Exxon Valdez Oil Spill
Restoration Project Final Report

2005 Assessment of Lingering Oil and Resource Injuries from the *Exxon Valdez* Oil Spill

Restoration Project 040776
Final Report

Integral Consulting Inc.
7900 SE 28th Street
Suite 300
Mercer Island, WA 98040

for:

Exxon Valdez Oil Spill Trustee Council
441 West 5th Avenue
Suite 500
Anchorage, AK 99501

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Study History: Restoration Project 040776 was initiated to provide the Trustee Council with an independent assessment of recent work on the nature and extent of lingering *Exxon Valdez* oil, an analysis of the ecological significance of lingering oil, a synthesis of relevant studies on resources classified in 2002 as "recovering" or "not recovering," and initial conclusions regarding the recovery status and general conditions of those resources. Most research and monitoring funded by the Trustee Council has focused on individual resources or selected ecosystems, and past assessments of the status of injured resources and services have been performed informally, with minimal information synthesis and technical citation. This project provides a more detailed synthesis of previous studies and uses a standardized framework for assessing the condition of resources. Evaluation efforts are guided by the Exxon Valdez *Oil Spill Restoration Plan* (EVOS Trustee Council 1994), which provides long-term guidance for restoring the resources and services injured by the oil spill.

This work effort is closely tied to more comprehensive evaluation of the status of injured resources and services that is in progress, Restoration Project 060783. In addition to the resources addressed in Restoration Project 040776, Restoration Project 060783 addresses injured services (subsistence uses, passive uses, commercial fishing, and recreation/tourism) and resources currently classified as having an unknown recovery status (cutthroat trout, Dolly Varden, Kittlitz's murrelet, rockfish, and subtidal communities). This more comprehensive evaluation is being conducted in consultation with agency and academic scientists who have been working in PWS over the past 17 years, and will culminate in recommendations to the Trustee Council on the recovery status of injured resource and services and recommendations for restoration actions, if needed.

Abstract: This report evaluates available scientific data to provide an independent evaluation of the nature and extent of lingering *Exxon Valdez* oil (EVO), the recovery status of key resources, and evidence linking residual effects from the original spill or lingering oil to the current condition of injured resources. Lingering oil remaining in Prince William Sound is found in the intertidal zone. Short et al. (2004a) predicted that approximately 10 acres of surface EVO and 19 acres of subsurface EVO were present in Prince William Sound in 2001. Additional related work performed in 2003 (Short et al. 2006) demonstrated that lingering oil extended further into the lower intertidal zone than originally anticipated, suggesting that the original estimate may be low by as much as 30 percent. Resources still experiencing the effects of the spill are generally those that are exposed to intertidal oil due to their feeding and foraging habits (e.g., sea otter, harlequin duck) or those that are long-lived and slow to reproduce (e.g., killer whale). Despite the fact that a number of studies have been directed at understanding the toxicity of oil to Pacific herring, which are of both ecological and commercial importance in Prince William Sound, the cause(s) of the highly depressed population are not well understood. Seventeen years after the spill, other stressors (predator–prey relationship, disease, climate or ocean current changes) made it more difficult to distinguish residual effects from the spill from the effects of other stressors.

Key Words: Lingering oil, *Exxon Valdez*, EVOS, Prince William Sound, sediment quality, resource injury, service injury, sea otter, killer whale, harbor seal, sea birds, harlequin duck, Pacific herring, sediments, wilderness areas, intertidal communities, clams, mussels

Project Data: No new data were collected for this project.

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ACRONYMS AND ABBREVIATIONS

ADEC	Alaska Department of Environmental Conservation
ADFG	Alaska Department of Fish and Game
AST	aspartate aminotransferase
CEM	conceptual exposure model
CYP1A	cytochrome P450 1A
BACI	before, after, control, impact (methodology)
EROD	7-ethoxyresorufin-*O*-deethylase
EVO	*Exxon Valdez* oil
EVOS	*Exxon Valdez* oil spill
GIS	geographical information system
GGT	gamma glutamyl transferase
ha	hectare
IHC	immunohistochemical
LDH	lactase dehydrogenase
LOEC	lowest-observed-effect concentration
LWO	less weathered oil
MLLW	mean lower low water
MWO	more weathered oil
NOAA	National Oceanic and Atmospheric Administration
PAH	polycyclic aromatic hydrocarbon
PCB	polychlorinated biphenyl
PDF	probability density function
PWS	Prince William Sound
SCAT	Shoreline Cleanup Assessment Team
SPMD	semipermeable membrane device
TPAH	total polycyclic aromatic hydrocarbon
USFWS	U.S. Fish and Wildlife Service

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This document represents our attempt to evaluate fairly and objectively a very broad body of work conducted in Prince William Sound and vicinity following the *Exxon Valdez* oil spill. The conclusions of this report represent those of Integral staff, and any errors and omissions are ours.

EXECUTIVE SUMMARY

The *Exxon Valdez* oil spill (EVOS) occurred as a result of the grounding of the *T/V Exxon Valdez* on Bligh Reef on March 24, 1989. Approximately 11 million gallons of the tanker's cargo of Alaskan North Slope crude oil was spilled into the open waters of Prince William Sound (PWS). The initial spreading of *Exxon Valdez* oil (EVO) in open water was exacerbated by a series of significant storm events resulting in the landing of EVO across shorelines of PWS over the course of a 2-month period. Seventeen years after the spill, EVO continues to persist to some degree in intertidal zones of the PWS shoreline and at patchy locations along the Gulf of Alaska shoreline southwest of PWS.

This report evaluates available scientific data to provide an independent evaluation of the nature and extent of lingering EVO, the recovery status of key resources, and evidence linking residual effects from the original spill or lingering oil to the current condition of injured resources. This work is intended to achieve the following objectives:

1. Provide the Trustee Council with an independent analysis of the ecological significance of lingering oil
2. Provide an initial independent evaluation of the recovery status of selected injured resources
3. Provide information to help focus and refine future work on oil-affected resources
4. Communicate the progress and results of these efforts to the Trustee Council and Trustee agencies
5. Communicate important technical conclusions to the public.

A more comprehensive evaluation of the status of injured resources and services is in progress. In addition to the resources addressed in this report, the evaluation will address injured services (subsistence, passive uses, commercial fishing, and recreation/tourism) and resources currently classified as having an unknown recovery status (cutthroat trout, Dolly Varden, Kittlitz's murrelet, rockfish, and subtidal communities). This more comprehensive evaluation is being conducted in consultation with agency and academic scientists who have been working in PWS over the past 17 years and will culminate in recommendations to the Trustee Council on the status of injured resource recovery and recommendations for restoration actions, if needed.

LINGERING OIL

Based on work performed in 2001, Short et al. (2004a) predicted that approximately 10 acres of surface EVO and 19 acres of subsurface EVO were present in PWS. Additional related work performed in 2003 (Short et al. 2006) demonstrated that lingering oil extended further into the lower intertidal zone than originally anticipated, suggesting that the original estimate may be low by as much as 30 percent. A recent Exxon-funded survey suggests that a smaller area than that estimated by Short et al. (2004a) may remain (Taylor and Reimer 2005). Consensus has been reached that whatever EVO does remain is a small fraction of the total area oiled in 1989.

In consultation with the National Oceanic and Atmospheric Administration (NOAA), a probabilistic technique was used by Integral to project the location of lingering oil in 2005, as well as the areal extent and volume. Based upon a Monte Carlo analysis, and assuming an annual loss rate of 20–26 percent, an estimated 4 acres of surface EVO and 7 acres of subsurface EVO were projected to be present in PWS in 2005, and less than 1 acre of EVO was project to be present in 2014 (i.e., 9 years). Because of uncertainties in the spatial extent and loss rate of EVO, additional modeling was conducted using more conservative assumptions. This additional modeling predicted a "worst case" estimate of approximately 35 total acres of EVO in 2005. Two locations were identified as areas of potential interest during the analysis. Northern Knight Island and Smith Island are two areas that were initially heavily impacted by the spill. Northern Knight Island has been the focus of additional investigation by NOAA during 2003 to evaluate further the presence of subsurface oil throughout the intertidal zone.

Sources

The vast majority of the area in PWS originally impacted by the spill is considered remote wilderness. Despite this remoteness, a variety of petroleum hydrocarbon and other polycyclic aromatic hydrocarbon (PAH) sources occur throughout PWS, including both natural and human sources. Prior to the spill, a major source of oil-derived hydrocarbons was the March 27, 1964, Alaska earthquake. The earthquake resulted in the rupturing of asphalt and fuel oil storage tanks in Valdez and other sites around the PWS and the Gulf of Alaska. These asphalt and fuel oil products originated from the California Monterey Formation oils, which are chemically distinct from EVO.

Hydrocarbons originating from other background sources are present throughout PWS in subtidal sediments. Sources of background hydrocarbons include eroding petroleum "source rocks" derived from Tertiary shales in areas east of PWS, coals originating in the outcrops of the Kulthieth Formation east of PWS (including the Bering River coalfields), and natural oil seeps (of doubtful significance) that occur throughout the northern Gulf of Alaska, but that are absent in PWS itself. Additional sources of hydrocarbons are associated with human activity, including more recent, smaller oil spills, and activities at villages, sawmills, canneries, and camps where coal, oil, or wood were used or burned (Page et al. 1998, 2006).

Ecological Significance

Intertidal sediments are the primary repository for lingering EVO and are the source of any ongoing exposure to resources that may live in proximity to or come in contact with oiled shoreline. A small amount of oil from the spill (conservatively estimated to be approximately 11 to upwards of 35 acres in 2005 for PWS) may remain in intertidal sediments and rocky shorelines located in and around PWS. Much of this lingering oil is of limited bioavailability and bioaccessibility. Direct testing of remaining oiled sediment patches using semipermeable membrane devices (SPMDs), SPMD extracts injected into fish, and toxicity tests have demonstrated that this oil can be toxic to some species of test organisms and that it can induce cytochrome P450 1A (CYP1A), an enzyme that is induced when an organism is exposed to PAHs. Mussels tested near the oil patches do not show elevated levels of PAHs, and the benthic communities living on the oil patches are not significantly different from benthic communities found in unoiled areas. Collectively, these data indicate that although some bioavailability and potential adverse impacts are possible for some organisms in direct proximity to areas with remaining EVO, the overall ecological significance of this for the PWS ecosystem is limited.

Given the estimated volume and areal extent, it is unlikely that lingering oil continues to be a significant source of exposure to more than a very few percent of the resources that come in contact with intertidal sediments in and around PWS.

Restoration Actions

Michel et al. (2005) identified two restoration approaches as most viable for lingering oil in PWS: 1) natural recovery, and 2) nutrient enrichment. Natural recovery consists of monitoring natural loss processes, to track the progress of natural recovery. The estimated cost for natural recovery based on biannual monitoring for 10 years was $874,000. Nutrient enrichment is a treatment technique used to hasten the biodegradation of oil and thus, its removal from the environment. The cost for treating an estimated 71,500 kg of subsurface EVO is expected to be $49,925,000.

Michel et al. (2005) acknowledged that the next step in the decision-making process is to conduct a detailed benefit-cost analysis, comparing the ecological benefits of removal of the lingering oil with the financial and ecological costs associated with remedial efforts.

INJURED RESOURCES

Sixteen resources are currently classified as not recovering or recovering.

Not Recovering	Recovering
Common loon	Clams
Cormorants (3 spp.)	Intertidal communities
Harbor seal	Killer whales (AB pod)
Harlequin duck	Marbled murrelet
Pacific herring	Mussels
Pigeon guillemot	Sea otter
	Sediments
	Wilderness areas

In addition to potential ongoing adverse effects from lingering oil, these resources can be adversely affected by lingering effects from the original spill or other natural or anthropogenic factors.

Sediments

"Sediments will have recovered when there are no longer significant residues of Exxon Valdez oil on shorelines (both intertidal and subtidal) in the oil spill area. Declining oil residues and diminishing toxicity are indications that recovery is well underway" (EVOS Trustee Council 2002b). Declining oil residues have been well established, and declining toxicity corresponds to declining oil residues. The fundamental issue is one of significance. The remaining oil has limited ecological significance; however, it likely results in localized exposure and (possibly) adverse impacts to some resources that live in direct proximity to the remaining oil patches.

Validation of the projected rate of recovery (i.e., loss rate) of lingering oil in PWS is recommended.

Designated Wilderness Areas

"Designated wilderness areas will have recovered when oil is no longer encountered in them and the public perceives them to be recovered from the spill" (EVOS Trustee Council 2002b). This recovery objective focuses entirely on the presence or absence of oil and on public perception. While oil residues have been dramatically reduced through natural recovery, some oil remains.

A quantitative assessment of residual pockets of lingering oil in wilderness areas would provide useful perspective on the nature and magnitude of lingering oil in designated wilderness areas. At present, the only recent assessment of oil in wilderness areas is largely qualitative in nature.

Intertidal Communities

Approximately 40–45 percent of the estimated 11 million gallons of EVO washed ashore in the intertidal zone of PWS. Aerial surveys by the Alaska Department of Natural Resources showed that light to heavy deposits of EVO washed ashore on approximately 24 percent (446 km) of the 1,891 km of the PWS shoreline. The spill area included all types of intertidal habitats, including exposed and sheltered rocky shores; boulder, cobble, and gravel beaches; mixed sand-gravel and fine sand beaches; exposed and sheltered tidal flats; and estuaries. Dominant species of algae and invertebrates that were directly affected by the spill were common rockweed, speckled limpet, several barnacle species, blue mussels, periwinkles, and oligochaete worms. Abundance of sediment infauna and clam densities at lower elevations on gravel and mixed sand/gravel beaches declined and intertidal fish were also affected.

"Intertidal communities will have recovered when such important species as *Fucus* have been reestablished at sheltered rocky sites, the differences in community composition and organism abundance on oiled and unoiled shorelines are no longer apparent after taking into account geographic differences, and the intertidal and nearshore habitats provide adequate, uncontaminated food supplies for top predators" (EVOS Trustee Council 2002b). There are important uncertainties that must be recognized in assessing recovery status, particularly the lack of monitoring of intertidal communities within the last decade. However, based on the weight of evidence for PWS, including the convergence of population trajectories of epibiota and infauna at untreated oiled sites and unoiled reference sites, the similarity in CYP1A expression between intertidal fish in most unoiled and oiled sites, and the localized, patchy distribution of lingering oil, most intertidal communities appear to have recovered.

The ability of intertidal communities to provide adequate, uncontaminated food supplies for top predators is another important consideration when assessing the recovery of the intertidal zone. Tissue data on PAHs in mussels indicates that bioaccumulation is no longer a concern for this resource. However, the most recent data on CYP1A in some bird species that forage in the intertidal zone (Ballachey et al. 2006) suggests that they may be exposed to PAHs during feeding. Harlequin duck, black oystercatcher, and Barrow's goldeneye, all of which forage on intertidal resources, continued to demonstrate exposure in 2004 and 2005 based on elevated levels of CYP1A that is distinguishable from background.

Clams

Clams in PWS are important as prey for sea otters, seabirds, and many other predators, as well as for subsistence use by native populations and recreational harvest. Within days of the EVOS in March 1989, clams on beaches along the shoreline of PWS were exposed to the spreading oil slick. Clams were killed by oiling and by the subsequent cleanup activities. Large numbers of dead and moribund clams documented on treated beaches suggest that they may have suffered toxic effects at the time of the spill (Lees et al. 1996), but those effects are difficult to separate from the effects of hydraulic washing. High-pressure, hot-water washing in particular caused thermal stress, oil dispersion, animal displacement and burial, and the transport of sediments from the upper intertidal into the lower intertidal zone.

Clams will have recovered when population and productivity measures (e.g., size and distribution) are comparable at oiled and unoiled sites, taking into account geographic differences (EVOS Trustee Council 2002b). It is likely that residual effects, if any, are limited to sites that experienced high-pressure, hot-water cleaning methods. At these sites, conditions have been improving over time. Residual effects of the original spill to intertidal populations of clams are small and unlikely to affect their value as a food resource to either human or non-human predators.

Mussels

Mussels in PWS are important prey items for harlequin ducks, black oystercatchers, juvenile sea otters, river otters, and many other species. They are also a key component of the intertidal habitat, providing physical stability and habitat for other organisms. Within days of the EVOS in March 1989, mussels on beaches along the shoreline of PWS were exposed to the spreading oil slick. Mussels at all tidal levels and substrates were directly exposed to EVO as oil coated mussels and penetrated the underlying mat of dense mussel beds.

In many areas, the initial high petroleum hydrocarbon concentrations in mussel tissue began declining soon after EVOS. Hydrocarbon concentrations in mussels on hard rock substrate have returned to background levels. Numerous investigations since the mid-1990s have documented elevated but declining PAH concentrations in mussels at some oiled beaches compared to mussels at unoiled locations or pre-spill levels. Existing data suggest that for nearly all of PWS, PAH concentrations in mussels have returned to background levels.

The recovery objective for mussels was defined as a return of petroleum hydrocarbon concentrations to background levels such that mussels do not contaminate their predators. It is highly unlikely that any remaining localized areas with elevated PAH would contaminate mussels at levels that would result in unacceptable risk to predators.

Harlequin Duck

Harlequin ducks are found throughout PWS, where they occupy shallow subtidal and intertidal zones, rich with benthic invertebrate food sources. They are at maximum density during the winter, and thus the full population was present in March 1989, when the spill occurred. An estimated 3–12 percent of the wintering population suffered acute mortality from oiling. Because they forage for crustaceans, molluscs, and other food in the intertidal zone, harlequin ducks have a high potential for exposure to lingering EVO, which is concentrated in the intertidal region of the shoreline.

Life history analysis of the harlequin duck suggests that recovery of a population after a perturbation like the EVOS could be extended (e.g., 10 years or more) because they are relatively long-lived birds (up to 12 years) with high survival and low reproduction rates. Dispersal and immigration rates are thought to be low. Thus, lowered survivorship over a several year period could have an extended effect on a population.

Long-term analysis (1989–2004) of harlequin duck populations in oiled areas of PWS indicates that the populations surveyed during March are stable, with no significant increasing or decreasing trends and no significant differences between population densities in oiled and unoiled areas (Sullivan et al. 2005; Rosenberg et al. 2005). Harlequin duck sex and age structure in oiled areas of PWS are similar to those elsewhere in their range, populations in oiled areas are stable, age ratios are similar in oiled and unoiled areas, and female survival has improved (Rosenberg et al. 2005). The slightly lower proportion of females in oiled areas remains a concern; however, the relevance of this condition to residual spill effects or biological recovery is not known.

Exposure of harlequin ducks to PAHs has been measured since 1998 using CYP1A, a liver enzyme that is induced in many vertebrate species following exposure to PAHs. Results from studies conducted in 2000–2002 initially indicated that levels in the oiled population had converged with levels in the unoiled population (Bodkin et al. 2003). However, the preliminary results for CYP1A data collected in March 2005 (Ballachey et al. 2006) indicate that levels were significantly higher (approximately doubled) in harlequin ducks collected from oiled areas relative to unoiled areas. Most researchers agree that as the spatial extent of lingering oil diminishes over time, the potential for or degree of exposure to harlequin duck also diminishes.

It is likely that harlequin duck populations in PWS are nearly recovered or recovering from the acute and chronic effects of the spill. However, the lack of an increasing population trend in western PWS and the apparent continuing PAH exposures as indicated by CYP1A elevations are still of concern to some researchers.

Additional population modeling is recommended to better understand the population dynamics of harlequin ducks, and to better assess the potential for residual impacts from the spill or impacts from lingering oil. Monitoring of populations and exposure (e.g., through CYP1A studies) should continue.

Pacific Herring

Pacific herring are of both ecological and commercial importance in PWS. Not only are they central to the marine food web, providing food to marine mammals, birds, and invertebrates, but herring are also fished commercially for food and bait, sac-roe, and spawn on kelp. Despite the fact that a number of studies have been directed at understanding the toxicity of oil to Pacific herring, the cause(s) of the highly depressed population, which has closed the fishery for all but 6 of the 17 years since the spill,[1] are not well understood.

Most life stages of Pacific herring were exposed to EVO to some degree following the initial spill. The spill occurred a few weeks before Pacific herring spawned in PWS—eggs and adults were exposed directly to residual EVO. Pacific herring did not spawn on the most heavily oiled beaches, but they did spawn on shorelines considered lightly to moderately oiled (Brown et al. 1996). Larval contact with EVO may also have occurred given that EVO was distributed in nearshore areas for several months following the spill. Early life stages are the most sensitive to the effects of oil. In addition, EVO was shown by researchers to be available in open water along shorelines where oiling was not directly evident. Carls et al. (2002) indicate that 25–32 percent of the Pacific herring embryos in PWS were damaged as a result of the EVO exposure.

There is little evidence linking the depressed herring population to residual effects from the spill, and it is likely that no single factor explains the population decline. In a synthesis of the literature, Carls et al. (1999, 2001b) surmise that the combination of high population density, poor nutrition, and epidemic infection by viral hemorrhagic septicemia virus caused the 1993 collapse in a boom-and-bust cycle typical of Pacific herring populations in the Gulf of Alaska and elsewhere. Pearson et al. (1999) also concluded that a combination of increasing biomass and decreasing food supply led to poor Pacific herring condition and the 1993 decline. Both studies acknowledged that natural factors and variability could explain the population decrease, and a clear link between EVOS and the 1993 population decline was neither established nor eliminated.

Present exposure of Pacific herring to and impacts from lingering EVO are likely minimal. The only route for Pacific herring exposure is during the egg stage when eggs are deposited on beaches. The extent of potential egg exposure to lingering oil is low because the proportion of Pacific herring spawning in areas that were moderately to heavily oiled, and therefore most likely to contain lingering oil, is a very small portion of known herring spawning areas.

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[1] The fishery was open in 1990, 1991, 1992, 1993, 1997, and 1998 (Johnson et al. 2002).

More recently, an expert group has been convened to address broader issues associated with the depressed herring population in PWS. Preliminary findings of the expert group indicate that the cause of the depressed Pacific herring population in PWS is apparently linked to continuing disease problems, a unique situation. While these diseases are ever present in all populations of herring (and other species), they are seldom suspected of limiting populations over an extended period of time. No other North American population has suffered acute population losses like those in 1993, nor has any other population suffered from chronic disease issues.

Additional research may be needed to determine the cause for the current condition of the Pacific herring population in PWS. It will likely be necessary to develop a more holistic approach to the problem, directly addressing issues such as the role of the fisheries management decisions, role of prey and predators, multiple disease factors affecting juvenile and adult fish, and climatic factors such as decadal oscillation or longer term changes.

Seabirds

Of the approximately 1 million seabirds inhabiting the oil spill area prior to the spill, it is estimated that between 100,000 and 300,000 were killed initially due to the acute effects of oiling (Piatt et al. 1990). Oil caused direct mortality either through coating of the feathers, leading to hypothermia and then death, or through lethal ingestion.

Pigeon guillemot, marbled murrelet, double-crested cormorant, pelagic cormorant, red-faced cormorant, and common loon[2] all spend a significant portion of their life histories in PWS and forage in the nearshore waters by diving for fish or benthic invertebrates. The common mechanism by which these birds may potentially be exposed to lingering EVO is indirectly through prey consumption or, in some cases, through direct contact with oiled intertidal sediments. The recovery objectives for these birds do not fully address the overall objective of a *return to conditions that would have existed if the spill had not occurred* because they do not take into account other factors or stressors that can affect population trends. For that reason, the overall objective was considered when evaluating seabirds.

It is unlikely that seabirds are continuing to experience any direct adverse effects from EVOS or from lingering EVO in the intertidal zone. There is currently minimal potential for exposure to lingering EVO in the intertidal zone, and it is likely that sufficient time has passed for the populations to recover from the initial acute mortalities caused by the spill. However, potential links between the oil spill and declining populations of sand lance and herring may be having a long-lasting, indirect effect on some seabirds in PWS.

Additional population monitoring and modeling are recommended to better understand the population dynamics of pigeon guillemots and marbled murrelets as indicators for the group of seabirds considered here. Collection of population survey data should continue for all injured species. Comparisons between PWS populations and those outside the sound should be made.

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[2] Although loons are not specifically seabirds, the similarity of their feeding mechanism and prey preferences allows them to be grouped with seabirds for this discussion.

Additional research to better understand the potential impacts of other stressors (e.g., climatic shifts or trends, predator-prey relationships) would also be beneficial.

Sea Otter

Sea otters are found throughout PWS, where they live in the nearshore environment and forage on subtidal and intertidal communities of clams, mussels, crabs, and sea urchins. Many sea otters were in the path of the oil as it spread south from Bligh Reef into bays and around islands of western PWS. Initial mortalities were due to acute injury with perhaps 2,000–3,000 mortalities throughout the spill area. Because sea otters forage in the intertidal and subtidal zones, digging up to several thousand intertidal pits per year, they have a high potential for exposure to lingering EVO, which is concentrated in the intertidal region of the shoreline.

The most thorough study of sea otters took place at northern Knight Island, which was heavily oiled in 1989. This study provides a "worst case" evaluation of ongoing impacts to sea otters. The sea otter subpopulation of northern Knight Island has generally been stable over the last several years, but remains well below the estimated pre-spill population levels. Exposure of sea otters on northern Knight Island to lingering oil has been measured since 1996 using CYP1A, an enzyme that is induced in vertebrate species following exposure to PAHs. The most recent biomarker data on exposure collected in summer of 2004 and 2005 (Ballachey and Bodkin 2006) indicate that levels of CYP1A in sea otters from northern Knight Island have declined to levels measured in sea otters from unoiled Montague Island, suggesting that exposure to PAHs is no longer a concern in this oiled area. Most researchers agree that as the spatial extent of lingering oil diminishes over time, the potential for sea otters to be exposed to lingering oil and their degree of exposure should also diminish.

It is likely that sea otter populations in some portions of western PWS are recovering from measurable acute and chronic effects of the spill, evidenced by a modest average annual rate of increase between 1993 and 2005. However, recovery of the subpopulation on northern Knight Island appears to be constrained by demographic lag, residual oil effects, continuing exposure, or by some other unknown stressors. These unknown stressors introduce some uncertainty into the interpretation of sea otter population trends on northern Knight Island, a relatively small subpopulation that is susceptible to small impacts on the total population. Predation by killer whales may be particularly important, and subsistence hunting could also be a factor. Emigration of sea otters must also be considered. Food resources apparently are not limiting on northern Knight Island and should not limit population growth.

Further investigation into the continued low numbers of the northern Knight Island subpopulation is recommended, including population modeling to address the sensitivity of this population to the impact of other stressors.

Harbor Seal

The harbor seal population in PWS began declining in the early 1980s, and perhaps earlier. The EVOS affected some of the largest harbor seal haul-out sites and waters adjacent to those sites in PWS. Immediately following the spill, dozens of seals became visibly oiled, including pups born to oiled mothers. Based on trend-count surveys conducted before and after the oil spill, the number of seals at oiled sites declined by 43 percent between 1988 and 1989 compared to an 11 percent decline at unoiled sites. In the last decade, the number of seals in the Kodiak Archipelago has been increasing (Small et al. 2003; Blundell et al. 2005), although the abundance level is much reduced from the 1970s. The Alaska Department of Fish and Game continued to conduct annual surveys in PWS following 1997, and recent counts indicate the population may be stabilizing after the sustained decline over the past two decades (Blundell et al. 2005).

It is unlikely that harbor seals are continuing to experience any residual adverse effects from the spill or from lingering EVO in the intertidal zone. Harbor seals could potentially be exposed to lingering oil as they move about on their haul-out sites. Exposure would be limited to dermal contact because seals do not forage in intertidal sediments. Lingering oil in surface sediments occurs primarily in the form of highly weathered, solid asphalt-like material, which would not be bioavailable to seals. Furthermore, the low likelihood of contamination in prey, combined with large feeding areas relative to the extent of lingering oil, makes continued exposure to lingering EVO through food resources for harbor seals unlikely.

Killer Whale

Within a week of EVOS in March 1989, the AB pod of killer whales was exposed to the spreading oil slick. By 1991, the resident AB pod had been reduced from 36 to 22 whales. Since 1991, the AB pod has fluctuated in number with additional births and mortalities, with a total number of 22 to 26 individuals. Despite the birth of seven new calves between 1990 and 1996, additional mortalities resulted in a total number of 27 whales in 2005.

The killer whale is a long-lived species (up to 90 years) with a relatively low rate of reproduction (on average, females have a calf every 5 years over a 30-year reproductive span). Low birth rates and long life spans make killer whales more likely to suffer long-lasting impacts and slow recovery from an acute event, particularly when females of reproductive age are lost. Modeling of the population dynamics (Matkin et al. 2003) indicates that the AB pod was impacted by EVOS primarily through the loss of young females, who offer the most potential contributions to future population growth. Based on this modeling effort, it was concluded that the AB pod has not yet recovered from the initial impacts of EVOS due to the loss of young females, but would have recovered had there been no loss of females at the time of the spill.

In addition, the AT1 transient population warrants additional study to evaluate the status and factors affecting population trends of this group.

1. INTRODUCTION

The natural resources and habitats of Prince William Sound (PWS) and other Alaskan waters have been studied extensively over the course of the 17 years since the occurrence of the *Exxon Valdez* oil spill (EVOS). The overall evaluation of the current condition of resources injured by EVOS begins with a comprehensive assessment of the existing literature related to the nature and extent of lingering oil in PWS and nearby areas. This literature assessment is followed by an evaluation and synthesis of research related to injured resources. The resources identified by the Trustee Council as “Recovering” or “Not Recovered” are the focus of this evaluation. Human use services that depend on natural resources are excluded from this evaluation.

In preparing this report, we have considered a broad range of scientific studies, regardless of their funding source. The conclusions and recommendations provided in this report reflect the best professional judgment of Integral scientists.

1.1 PURPOSE AND OBJECTIVES

This report evaluates available scientific data to provide an independent evaluation of the nature and extent of lingering *Exxon Valdez* oil (EVO), the recovery status of key resources, and evidence linking residual effects from the original spill or lingering oil to the current condition of injured resources. This work is intended to achieve the following objectives:

1. Provide the Trustee Council with an independent analysis of the ecological significance of lingering oil
2. Provide an initial independent evaluation of the recovery status of selected injured resources
3. Provide information to help focus and refine future work on oil-affected resources
4. Communicate the progress and results of these efforts to the Trustee Council and Trustee agencies
5. Communicate important technical conclusions to the public.

A more comprehensive evaluation of the status of injured resources is in progress. In addition to the resources addressed in this report, the evaluation will address injured services (subsistence, passive uses, commercial fishing, and recreation/tourism) and resources currently classified as having an unknown recovery status (cutthroat trout, Dolly Varden, Kittlitz’s murrelet, rockfish, and subtidal communities). This more comprehensive evaluation is being conducted in consultation with agency and academic scientists who have been working in PWS over the past 17 years and will culminate in recommendations to the Trustee Council on the status of injured resource recovery and recommendations for restoration actions, if needed.

1.2 BACKGROUND

EVOS occurred as a result of the grounding of the *T/V Exxon Valdez* on Bligh Reef on March 24, 1989. Approximately 11 million gallons of the tanker's cargo of Alaskan North Slope crude oil was spilled into the open waters of PWS. The transport and fate of spilled EVO in open water in the first several days after the spill has been widely researched and is well documented in the open literature. In general, a number of widely accepted chemical and physical processes led to the relatively rapid evaporation, dissolution, dispersion, and degradation of EVO in the open water environment.

The initial spreading of EVO in open water was exacerbated by a series of significant storm events resulting in the landing of EVO across shorelines of PWS over the course of a 2-month period. Several chemical and physical processes responsible for the transport and fate of EVO on these shorelines, particularly within the intertidal zone, have determined and continue to determine the nature and extent of remaining EVO. Seventeen years after the spill, EVO continues to persist to some degree in intertidal zones of the PWS shoreline and at patchy locations along the Gulf of Alaska shoreline southwest of PWS. This persisting EVO may act as a potential source of exposure to non-recovering and recovering resources.

1.3 RESTORATION PLAN AND RECOVERY GOALS AND OBJECTIVES

Restoration efforts in PWS are guided by the Exxon Valdez *Oil Spill Restoration Plan* (EVOS Trustee Council 1994). The purpose of the restoration plan is to provide long-term guidance for restoring the resources and services injured by the oil spill. It contains policies for making restoration decisions and describes how restoration actions will be implemented. The restoration plan includes five categories of restoration activities:

- General restoration
- Habitat protection and acquisition
- Monitoring and research
- Restoration reserve
- Public information, science management, and administration.

The restoration plan acknowledges that all injuries to natural resources cannot be studied or fully documented, because of the size of the area affected by the spill, multiple habitat types, and large number of species affected. The lack of baseline data is also a limiting factor. The list of injured resources and services provided in the restoration plan includes only those biological resources for which scientific research has demonstrated a population-level injury or continuing chronic effects. Periodic updates are provided when new information is available.

The goals, objectives, and restoration strategies developed for injured resources and services are intended to provide the blueprint for restoring the spill area. The overall goal of restoration is recovery of all injured resources and services, sustained by healthy, productive ecosystems that maintain naturally occurring diversity (EVOS Trustee Council 1994). The overall recovery objective of injured resources is a return to conditions that would have existed had the spill not occurred (EVOS Trustee Council 2002b). Resource- and service-specific recovery objectives are also identified and in some cases refined in subsequent updates of the recovery status of injured resource and services (September 1996, March 1999, August 2002, and June 2003).

Many resources and services currently classified as injured have as their recovery objective a "return to pre-spill levels" or "stable or increasing population trends". These recovery objectives do not fully address the overall objective of a *return to conditions that would have existed if the spill had not occurred,* because they do not take into account other factors or stressors that can affect the population. For that reason, both resource-specific objectives and the overall objective of a return to conditions that would have existed had the spill not occurred are considered when evaluating the status of injured resources in this report.

1.3.1 Restoration Actions Taken

Initial efforts to address the impacts of the spill focused on rehabilitation of oiled birds and mammals and removal of oil from oiled shorelines. Efforts to remove oil from the shoreline using various treatment and removal techniques met with varying degrees of success. Subsequent restoration actions have been funded by the $900 million civil settlement paid by Exxon. Restoration activities have included habitat protection, research, monitoring, and general restoration. Protection of habitat through land acquisition ($406 million) has involved the purchase of important habitat to limit logging and development projects that could inhibit restoration of injured resources or services. Large parcels (greater than 1,000 acres) are purchased to protect key habitat throughout the spill region. Small parcels (less than 1,000 acres) are typically strategically located habitat usually located by coves, by important stretches of river, or adjacent to valuable tidelands. A portion of the settlement ($145 million) has been set aside as a restoration reserve, to fund long-term restoration efforts through one or more of the categories identified above.

Natural recovery was selected as a preferred restoration method following the initial cleanup, which led to a comprehensive monitoring program to track the rate and effectiveness of natural recovery. Research was also directed toward restoration. Research and monitoring was primarily directed at tracking the recovery of resources following the spill and gaining a better understanding of the factors controlling the rate of recovery, the mechanism of impact, the nature of ongoing exposure, the location and bioavailability of residual pockets of residual EVO, and the interrelationships between injured and uninjured resources. Consideration of natural stressors and other sources of man-made stressors (e.g., fishing, localized industrial releases) was needed to distinguish lingering impacts from the spill from those associated with other stressors. Three major ecosystem research programs were conducted: 1) the Sound Ecosystem Assessment project, which addressed factors that influence the productivity of Pacific herring and pink salmon; 2) the Nearshore Vertebrate Predator program, which evaluated factors affecting the recovery of for indicator species, river otters, pigeon guillemots (a seabird), harlequin ducks, and sea otters; and 3) the Alaska Predator Ecosystem Experiment, which evaluated the productivity and recovery of seabirds based on the availability of forage fish.

1.3.2 Current Status of Lingering Oil

Based on the National Oceanic and Atmospheric Administration's (NOAA) most recent published findings for 2001, approximately 11.3 hectares (ha) (28 acres) of lingering oil is estimated to persist in intertidal surface and subsurface sediments of PWS beaches. A recent Exxon-funded survey suggests that even a smaller area remains (Taylor and Reimer 2005). Consensus has been reached that whatever area does remain is a small fraction of the total area oiled in 1989. However, lingering EVO nevertheless remains a potential source of ongoing exposure to some resources that have not recovered from injury caused by the initial spill.

Based on the work of Short et al. (2004b), O'Claire et al. (1996), Page et al. (2002b), and others (e.g., Michel and Hayes 1999; Hayes and Michel 1999; Gibeaut and Piper 1995), lingering EVO in surface sediments occurs primarily in the form of highly weathered, solid asphalt-like material sporadically present in the upper intertidal zone of sheltered areas. Weathered oil in the solid, asphalt form is insoluble, and therefore unlikely to be bioavailable. In contrast with remaining surface oil, EVO that penetrated the intertidal matrix of cobbles, gravel, and finer sediments to subsurface depths is less susceptible to weathering processes and is more persistent and soluble (Short et al. 2004a; Page et al. 2002b; Michel and Hayes 1999; Hayes and Michel 1999; Wolfe et al. 1994).

1.3.3 Current Status of Injured Resources

Sixteen resources are currently classified as not recovering or recovering.

Not Recovering	Recovering
Common loon	Clams
Cormorants (3 spp.)	Intertidal communities
Harbor seal	Killer whales (AB pod)
Harlequin duck	Marbled murrelet
Pacific herring	Mussels
Pigeon guillemot	Sea otter
	Sediments
	Wilderness areas

In addition to potential ongoing adverse effects from lingering oil, these resources can be adversely affected by lingering effects from the original spill or other natural or anthropogenic factors.

1.4 DOCUMENT OVERVIEW

The remainder of this document is organized as follows:

- Section 2, Lingering Oil—Describes the nature and extent of lingering EVO, other sources of oil, and predictions for future persistence and areal extent.
- Section 3, Intertidal Communities—Provides background on the ecology of intertidal communities, initial impacts of EVOS, and the basis for the current status of injury and recovery classification. Information on spatial and temporal changes in intertidal communities is then described, followed by an assessment of the linkage between EVOS (both lingering effects from the original spill and ongoing effects from lingering EVO), and the current condition of intertidal communities in PWS.
- Section 4, Clams—parallel to Section 3
- Section 5, Mussels—parallel to Section 3
- Section 6, Harlequin Duck—parallel to Section 3
- Section 7, Pacific Herring—parallel to Section 3
- Section 8, Seabirds—parallel to Section 3
- Section 9, Sea Otter—parallel to Section 3
- Section 10, Harbor Seal—parallel to Section 3
- Section 11, Killer Whale—parallel to Section 3
- Section 12, References.

2. LINGERING OIL

This section provides an overview of the major investigations conducted to determine the location and extent of EVO following the spill, the conceptual exposure model (CEM) used to describe the potential link between EVO and resources, the nature and extent of lingering EVO, other potential sources of oil, predictions for future persistence, and the areal extent of EVO. The section concludes with focused discussions on the current status of injury and recovery classification for sediments and possible restoration options.

2.1 OVERVIEW OF MAJOR INVESTIGATIONS

Three principal categories of investigations were performed after the spill to determine the location and extent of lingering oil. They are the historical Shoreline Cleanup Assessment Team (SCAT) surveys conducted in PWS between 1989 and 1993, the probability-based sampling work conducted by NOAA in 2001, and other recent studies conducted within and outside of PWS.

2.1.1 Historical SCAT Surveys

Following the initial EVOS spill in March 1989, a series of comprehensive ground surveys was performed to document the prevalence and magnitude of oiling present on beaches of PWS. The main surveys, referred to as SCAT surveys, were performed in 1989, 1990, 1991, 1992, and 1993. The goal of these surveys was to map the spill oil on shorelines and classify them according to extent and magnitude of oiling.

The most extensive survey was conducted in September 1989, during which approximately 1,200 km of the 5,000 km PWS shoreline was walked and surveyed by SCAT members (see Figure 2-1). SCAT surveys conducted in subsequent years covered a smaller extent of PWS (1,100 km in 1990, 390 km in 1991, 32 km in 1992, and approximately 50 km in 1993), owing to reductions in oil attributable to natural processes (e.g., dispersion, weathering) and direct cleanup efforts performed chiefly by Exxon.

Because the methods used in SCAT surveys are predominantly based on visual observations, the emphasis of SCAT surveys was on surface oiling. Subsurface oiling was examined to a lesser extent and usually in an opportunistic fashion to augment visual assessments of the surface. A chief objective of the SCAT surveys was to direct cleanup efforts. The SCAT surveys also provided a mechanism by which the persistence of EVO over time could be monitored.

According to the SCAT surveys, the extent of oiled shoreline had decreased from 583 km to 10 km from the period of 1989 to 1992 (ASGDC 2005; Neff et al. 1995). These trends suggested that oil remaining after 1992 would soon further disperse to negligible amounts (Koons and Jahns 1992; Neff et al. 1995; Boehm et al. 1995; Gibeaut and Piper 1998; Gilfillan et al. 2000). At some locations, however, oil appeared to be more persistent (Hayes and Michel 1999; Brodersen et al. 1999; Carls et al. 2001a; Short et al. 2004a). For example, the 1993 SCAT survey suggested that oil might persist in certain areas because it was no longer subjected to cleanup (Gibeaut and Piper 1998). A focused cleanup effort at Sleepy Bay in 1997 encountered unanticipated deposits of subsurface oil (Brodersen et al. 1999). Concerns regarding the persistence of lingering oil, particularly subsurface oiling, led to the 2001 probability-based shoreline sampling investigation of lingering EVO in PWS performed by researchers from NOAA (Short et al. 2004a).

2.1.2 Overview of 2001 NOAA Probability-Based Sampling Investigation in PWS

During the summer of 2001, researchers from NOAA performed a probability-based shoreline sampling study in PWS. The sampling design used for the study represented a significant departure from the ground survey methods used during the 1989–1993 SCAT surveys. As described by Short et al. (2004a), the study relied instead on a random sampling design to estimate the extent of remaining surface and subsurface oil.

For the sampling design, Short et al. (2004a) first identified the universe of sampled beach segments with oil present during the 1989–1993 SCAT surveys in PWS. These included beach segments previously described as heavily oiled (Category I), moderately oiled (Category II), and previously heavily oiled in 1989 but having little to no surface oil evident in subsequent years (Category III). Beach segments were then categorized into subsegments based on two length classifications, 100 m and < 100 m. The 100 m beach length was the basic sampling unit that the NOAA field crew could sample over the course of a low tide; however, some of the SCAT survey beach segments selected by NOAA were less than 100 m.[3] Collectively, a total of six sampling strata (three oiling categories factored with two length categories) were selected and used as the basis of a random sampling procedure.

No other characteristics, such as beach geomorphological conditions, were used in the sampling design. Also, beaches with medium and low oiling in 1989 and low oiling more recently were assumed to be clean and were excluded from sampling. As described by NOAA's statisticians, oiled beach segments (i.e., heavily, moderately, and originally heavily oiled) were selected for which NOAA's "expectations of finding oil were highest" (Pella and Maselko 2003).

Short et al. (2004a) focused their random sampling of surface and subsurface conditions on the middle and upper intertidal areas (ranging from +1.8 to +4.8 m above mean lower low water [MLLW]). The presence or absence of oil and the degree of oiling was recorded, and a limited number of oil samples were collected for chemical analysis. Using the results of the random samples, Short et al. (2004a) estimated the average mass and average area of persisting oil using probability-based estimation formulas.

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[3] The distinction between strata based on beach segment length was necessary for NOAA to extrapolate sampled areas to areal estimates representative of all of PWS. This extrapolation required separate calculation procedures for the two beach segment length categories.

The total beach area with surface oil was estimated to be 4.13 ha (10.2 acres). Surface oil was most commonly found in the upper half of the intertidal zone (i.e., within +3.3 to +3.8 m above the MLLW tidal elevation stratum) of previously described heavily and moderately oiled beaches, and was most often present as asphalt pavements and mousse. Based upon the chemical analysis, approximately 90 percent of surface oil corresponded to EVO. The remainder corresponded to Monterey Formation asphalt.

The total beach area with subsurface oil was estimated to be 7.80 ha (19.3 acres). Subsurface oil was also most commonly found in the previously described heavily and moderately oiled beaches. Unlike surface oil, subsurface oil was most often encountered near the middle (+1.8 to +2.8 m) of the intertidal zone. The majority of subsurface oil was classified as light oil residue. Based upon the chemical analysis, all of subsurface oil corresponded to EVO. The total mass of subsurface oil was estimated to be 55,600 kg, which represents less than 0.3 percent of the amount of oil originally beached in 1989. During the course of the 2001 work, NOAA also conducted limited opportunistic (i.e., nonrandom) sampling in the lower intertidal zone. This opportunistic sampling was performed below the +4.8 to +1.8 m MMLW tidal elevation stratum used during random sampling. Subsurface oil was identified below the +1.8 m tidal elevation stratum, particularly at Category I and II beaches. Based on this, Short et al. (2004a) speculated that if the lower intertidal zone were included in the random sampling, approximately 30 percent more subsurface oil would have been estimated.

The findings of Short et al. (2004a) suggest that more EVO had remained along shorelines of PWS, particularly in the middle and lower intertidal zone, than was projected based upon the 1989–1993 SCAT surveys (Koons and Jahns 1992; Boehm et al. 1995; Neff et al. 1995; Gibeaut and Piper 1998; Gilfillan et al. 2000). Based upon the combined use of the 2001 NOAA volumetric estimates coupled with data from the previous SCAT surveys, Short et al. (2004a) derived an intertidal decay rate for EVO in the range of 20–26 percent. Previously, Neff et al. (1995) estimated a higher rate of decay of 75 percent based on the decrease in linear kilometers of surface EVO recorded during the 1989–1992 SCAT surveys. The difference in estimates may be attributable to differences in both the depth and the location within the intertidal zone used to make the estimates. For example, the Short et al. (2004a) decay rate was based on an estimated subsurface intertidal mass in 2001[4] relative to an estimated mass remaining as of 1992. The 1992 mass estimated by Short et al. (2004a) was based on the work of Wolfe et al. (1994), who largely based their work on the linear shoreline estimates for surface oil during the 1989–1992 SCAT surveys. In addition, the SCAT surveys tended to not only focus on surface EVO but also focus specifically within the upper intertidal zone (Neff et al. 1995; Gibeaut and Piper 1998). The upper intertidal zone had likely been the historical focus given that 1) it is the area where EVO was generally accepted to have originally landed, 2) the upper intertidal zone is less frequently submerged and thus more conducive to weathering of surface EVO, and 3) surface

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[4] Short et al. (2004a) attempted to reconcile the additional mass potentially attributable to both surface oil and an underestimation of oil in the lower intertidal zone by presenting a decay rate as a range (i.e., 20–26 percent). This range is based upon the Short et al. (2004a) 2001 calculated mass of 55,600 kg and a presumed upper-end estimated mass of 111,200 kg (a factor of 2 higher than the calculated value of 55,600 kg). The range of 20–26 percent is based on a presumed mass of 860,000 kg in 1992, relative to the range of masses predicted by Short et al. (2004a) (i.e., 56,000–111,200 kg).

EVO is visually more apparent in the upper intertidal zone than it is in the middle and lower intertidal zones.

NOAA returned to PWS in 2003 to conduct follow-on sampling in some of the most heavily oiled beaches characterized in the 2001 study (Short et al. 2006). A total of 32 beaches were sampled in Herring Bay, Lower Pass, and Bay of Isles located along northern Knight Island. These beaches were some of the most heavily oiled areas following the initial spill and were protected from high-energy wave scouring. According to Short et al. (2006), the 2003 sampled beaches differed appreciably in intensity of initial oiling and the proportion of shoreline types from the broader area sampled throughout PWS in 2001. For subsurface oiling, NOAA identified a higher proportion of oiled beaches than were identified in 2001, and magnitude of oiling (categorized by the oiling sampling classification reported in Short et al. 2004a) was similarly greater among beaches sampled in 2003. The distribution of subsurface oiling with respect to tidal elevation was statistically similar for beaches sampled in both 2001 and 2003 at the +1.8 to +4.8 m range. Beaches sampled in the lower intertidal (0 to +1.8 m) accounted for approximately 36 percent of the oiling identified across all tidal elevations in 2003.

While this percentage approximates the 30 percent underestimation for lower intertidal subsurface oiling speculated by Short et al. (2004a) for the 2001 study, the 2003 results may not fully validate the 2001 underestimation. As noted by Short et al. (2006), comparison of the 2001 and 2003 study results requires some caution, based upon the above described differences in proportions of shoreline types and intensity of initial oiling among sampled beaches. To some extent, the 30 percent underestimation may be validated by the 2003 data in cases where similar beaches sampled in 2001 and 2003 are compared. However, the 30 percent underestimation applied for all of PWS remains somewhat uncertain, given the disparity in beach types sampled throughout PWS in 2001 versus those sampled in 2003 along northern Knight Island. Absent additional quantitative data, the 2003 study data may be most useful in providing qualitative perspective on the 30 percent underestimation and the need for additional quantitative confirmation data.

2.1.3 Other Recent Studies Within and Outside of PWS

In a 2002 study conducted in PWS, Taylor and Reimer (2005) performed a SCAT-based survey of sites based largely upon sites previously sampled during the 2001 NOAA study. At 15 of 39 sampled sites, EVO was found in weathered asphalt form at the surface and represented less than 0.2 percent of the surface area surveyed. Largely unweathered subsurface EVO was found at 32 of the 39 sites, with > 80 percent being found in the middle and upper intertidal zones and approximately 13 percent being found in the lower intertidal zone. Neither the exact tidal elevations for these zones nor the sampling design on selected beaches are described by Taylor and Reimer (2005), thus complicating direct comparison to the findings of Short et al. (2004a). Taylor and Reimer (2005) do not present an extrapolated quantification of the total area of remaining EVO in PWS, though they do acknowledge that slow weathering rates do exist at beaches where EVO is sequestered from natural weathering.

In a 2003 study, Neff et al. (2006) sampled sediments at historical human and industrial activity sites located in PWS. Nine of these sites, including former canneries, fish processing facilities, lumber mills and mines, were sampled for chemical analysis and estimation of areal extent of polycyclic aromatic hydrocarbon (PAH) contamination. Approximately 9 acres of sediment containing total PAH (TPAH) greater than 2,500 μg/kg dry weight was mapped. Based upon chemical analysis, TPAH in sediments was attributed to mixtures of petroleum products and combustion sources unrelated to EVO. A portion of these sediments was identified by immunoassay techniques as being potentially bioavailable. This study demonstrated that other sources of bioavailable PAHs unrelated to EVO may be present in PWS.

Outside of PWS along the Gulf of Alaska, a number of studies have been performed principally by the U.S. Geological Survey, the U.S. Fish and Wildlife Service (USFWS), and the Alaska Department of Environmental Conservation (ADEC) (e.g., Babcock et al. 1996, 1998; Irvine et al. 1999, 2002; Carls et al. 2001a) to evaluate lingering EVO on boulder armored beaches and mussel beds. These studies were performed along armored beaches at the Kenai Fjords and Katmai National Parks and the Kodiak Archipelago and for mussel beds along the Kenai and Alaska Peninsula coastlines. In 1995, a shoreline survey team from ADEC visited 30 sites along the Kodiak Archipelago that had measurable or reported oiling in 1990 and 1991 (ADEC 1996). By 1995, no oil or only trace amounts of oil were observed. Along the Kenai Fjords and Katmai National Parks, the majority of armored beaches visited[5] (typically five of six beaches visited) had largely unweathered EVO present as mousse in the subsurface, while areas of surface oiling were largely absent. These study sites were characterized as being relatively low in geomorphological diversity, consisting of boulder-armored, gravel beaches, most with an underlying bedrock abrasion platform at shallow depth (Irvine et al. 2002). This low diversity stands in stark contrast to the high diversity in geomorphology along shorelines throughout PWS (Irvine 2005, pers. comm.). In sediments at mussel bed sites, EVO was generally found, but by 2002, the areal extent and concentration of EVO was found to have declined at most sites (Irvine et al. 2002). As of 2005, the decreasing trend in EVO present in sediments at mussel bed sites appears to be continuing in these remote areas (Irvine 2005, pers. comm.).[6]

2.2 CONCEPTUAL EXPOSURE MODEL

A CEM was developed (Integral 2004) to support the evaluation of the nature and extent of lingering oil and present-day resource injuries that can be linked to lingering oil from the EVOS. The CEM developed for this evaluation follows the general framework described for the

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[5] Study sites included McArthur Pass, Cape Douglas, Kiukpalik Island, Ninagiak Island, Cape Gull and Kashvik (Irvine et al. 2002).

[6] Designated wilderness areas, which include the Katmai National Park, wilderness study areas in the Chugach National Forest and Kenai Fjords National Park, and Kachemak Bay Wilderness State Park, are injured resources categorized as "recovering" by the EVOS Trustee Council (EVOS Trustee Council 2002b). The recovery objective for this resource is stated as "[d]esignated wilderness areas will have recovered when oil is no longer encountered in them and the public perceives them to be recovered from the spill." Although lingering oil may be present in a relatively few, select locations within wilderness areas, it is unlikely to be encountered because it is predominant in subsurface conditions, largely in the form of mousse under boulder-armored shorelines. Because the collective recovery objective is largely based on "aesthetic" concerns of humans (inclusive of actual or perceived presence of lingering oil) rather than based on ecological resources, wilderness areas are not explicitly addressed in this report.

development and implementation of conceptual site models used in risk assessment as outlined by the U.S. Environmental Protection Agency (USEPA 1997, 1998) and ADEC (2000a,b; 2004).

The general CEM represents a series of working hypotheses developed to support the evaluation of whether, how, and to what degree lingering EVO can be linked to present-day resource injuries in PWS. Injuries cannot be ascribed in part or wholly to lingering EVO unless 1) resources come into contact with lingering EVO; 2) lingering EVO has the ability to cause adverse effects to those resources; and 3) an adverse effect to the resource has been measured or observed. The CEM seeks to establish to what extent, if any, these conditions co-exist. In its most general form, therefore, the CEM is a planning tool used to determine whether a complete environmental exposure pathway exists between lingering EVO and resources. In more complete terms, the CEM is a series of pictorial representations of past and lingering sources of EVO, chemical release and transport mechanisms, and environmental exposure pathways leading to recovering and non-recovering habitats and natural resources of concern.

The CEM serves as the foundation from which more detailed evaluations of the potential link between lingering EVO and present-day resources can ensue. These include evaluations of the weathering, toxic fraction, bioaccessibility, bioavailability, and nature and extent of lingering EVO, as well evaluations of the biological processes of exposure, uptake, and transfer of toxic EVO constituents among resources. The CEM is used as a starting point to assess which resources are clearly associated with lingering oil based upon their degree of spatial and temporal association with lingering EVO.

There are two major components of the CEM:

1. The first component consists of the chemical and physical processes in the environment that are responsible for determining the transport and fate, and ultimately the nature and extent, of EVO in PWS. These processes include physical forcing and dispersion of EVO (including sequestering mechanisms in intertidal sediments), chemical and microbial processes of degradation and transformation of EVO, and partitioning into abiotic and biotic environmental matrices.
2. The second component consists of an identification of the recovering and non-recovering ecological resources present in relation to lingering EVO. These include resources such as intertidal invertebrates coming into direct contact with lingering EVO, as well as the higher-trophic level resources, such as sea otters and harlequin ducks.

Each of these components is discussed and their corresponding depictions are presented in the following sections.

2.2.1 Chemical and Physical Processes

This section presents the chemical and physical processes that have dictated the nature and extent of EVO in PWS over time. These processes are depicted in Figure 2-2. Separate discussions are presented for these processes as they occurred immediately after the spill in open water (acting as a primary source of exposure) and from the time EVO landed on the beaches of PWS and its subsequent persistence in the intertidal zone of these beaches (potentially acting as a secondary and ongoing source of exposure). Because present-day resource injuries are being evaluated relative to lingering EVO present in the intertidal zone, the overall focus of this section is on those chemical and physical processes that dictate the nature and extent of lingering EVO in this area.

2.2.1.1 Open Water

Approximately 11 million gallons of the tanker's cargo of Alaskan North Slope crude oil was spilled into the open waters of PWS.[7] A number of widely accepted chemical and physical processes led to the fairly rapid evaporation, dissolution, dispersion, and degradation of EVO in the open water environment.

The left side of Figure 2-2 depicts the chemical and physical processes dictating the short-term nature and extent of EVO in open water. During the first two and a half days following the accident, EVO spread on the surface of the ocean as a slick (Galt et al. 1991; Kelso and Kendziorek 1991; Maki 1991; Wolfe et al. 1994; Neff and Stubblefield 1995; Patin 1999). Weathering of the oil started immediately as the more volatile and water-soluble fractions were rapidly lost to the atmosphere and water column through evaporation and dissolution, respectively (Galt et al. 1991; Neff et al. 1995; Neff and Stubblefield 1995; Wells et al. 1995; Wolfe et al. 1994). In response to wind and wave turbulence, emulsions (mixtures of tiny droplets of oil and water) formed both on the water surface (water-in-oil emulsions) and in the water column (oil-in-water emulsions) (Mackay et al. 1980; Payne et al. 1991, Wolfe et al. 1994). Emulsions on the water surface took the form of a mousse (a fairly stable form consisting of up to 70 percent water [Payne 1983; Patin 1999]). Tar balls also developed at the water surface as the oil slick and/or mousse fragmented, due to continued evaporation and spreading (Payne and Philips 1985). Oil on the water surface was additionally subjected to oxidation by exposure to ultraviolet light (typically photolysis [Patin 1999]). Within the water column, the tiny droplets of oil were also subjected to photooxidation to the depths to which light can penetrate. In addition, hydrocarbons may have been partially degraded or completely metabolized by microbes or other organisms (Patin 1999). Sorption of hydrocarbons to suspended solids and subsequent sedimentation also likely occurred (Patin 1999; Payne et al. 1991; Galt et al. 1991; ITOPF 2004). Overall, however, direct transport from the water column to the seafloor was considered to be insignificant (Bence et al. 1996).

Collectively, the persistence of EVO in open water following the spill was mediated by the relatively rapid degradation and dispersion processes in the sea (Neff et al. 1990; Wolfe et al. 1994; Short and Harris 1996; Spies et al. 1996; Neff and Stubblefield 1995). Today, 17 years after the original spill, EVO contamination in the open water is not present in measurable quantities.[8]

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[7] The actual volume of oil released during the spill is uncertain. Although the value of 11 million gallons does appear in the open literature, as do higher volume estimates, published reports documenting the quantification of the actual release volume do not exist.

[8] Similar conclusions regarding the general absence or very low concentrations of EVO in benthic sediments of PWS have also been reached (Page et al. 1995).

2.2.1.2 Intertidal Zone

The right side of Figure 2-2 depicts the chemical and physical processes that determine the long-term nature and extent of EVO in the intertidal zone of PWS. On the third day after the spill (March 26), a large storm generated and persisted for3 days producing winds ranging from 20 to 25 knots from the northeast, with gusts up to 70 knots. Winds rapidly moved the oil to the southwest, driving it to shorelines of several islands throughout PWS. Following 3 weeks of calm, clear weather, 2 weeks after the leading edge of the oil passed into the Gulf of Alaska, gale force winds (40–70 knots) occurred again, along the Kenai coast, churning and moving the oil toward Lower Cook Inlet (Wolfe et al. 1994). EVO and associated mousse washed ashore in PWS over the course of approximately 2 months (Wolfe et al. 1994; Gibeaut and Piper 1995; Neff et al. 1995), oiling an estimated 780 km (490 miles) of shoreline (Short et al. 2004a; Neff et al. 1995).

EVO coated beach surfaces resulting in surficial contamination predominantly within the intertidal zone. EVO additionally penetrated the surface through cobbles, gravel, and smaller-grained sediments resulting in subsurface contamination in the intertidal zone. Relatively little EVO was transferred in the first 2–3 weeks to shallow subtidal areas immediately adjacent to heavily oiled shorelines (Page et al. 1995). However, in the first years of cleanup activities and natural removal (e.g., intense winter storm events) of EVO from the shoreline following the spill, some EVO was transferred to subtidal sediments (Short et al. 1996; O'Claire et al. 1996). Oiled sediments from the intertidal zone were mobilized by storms and beach treatment activities to shallow subtidal sediments, though concentrations were generally low, and did not persist for longer than a year or two, except perhaps adjacent to the most heavily oiled beaches within protected embayments with entrance sills, such as Northwest Bay. By 2001, EVO was not identified in subtidal sediments along shorelines that had been heavily oiled (Short et al. 2003).

Within the intertidal zone itself, EVO present at the surface and subsurface may potentially result in exposures to resources. However, the chemical and physical processes acting on EVO vary substantially in type and degree with sediment depth, and result in distinctive characteristics and distribution profiles between the surface and subsurface settings. As depicted along the right-hand portion of Figure 2-2, within the intertidal zone, lingering EVO is predominantly found in highly weathered asphalt and tar ball forms at the surface, whereas it is usually less weathered when sequestered in the subsurface. A detailed discussion of the reasons for this distinct difference in form and composition between depth conditions is provided below in Section 2.3, "Nature and Extent of Lingering Oil."

2.2.2 Ecological Resources

The ecological resources component of the CEM entails a depiction of resources present in the food web of the nearshore coastal ecosystem of PWS. This depiction is presented in Figure 2-3.

The shallow nearshore coastal ecosystem in PWS is highly complex, yet conforms to well-understood and strong patterns of vertical zonation (Peterson 2001). The physical processes that have resulted in the distribution of lingering EVO in intertidal sediments also, in part, determine the marine communities that are present and which could be exposed to lingering EVO. The distributions of plants and animals of rocky shorelines are strongly influenced by both physical (e.g., wave action) and biotic (predation and competition) factors. Thus, nearshore marine communities are defined by the degree of exposure to waves (e.g., sheltered vs. exposed coast) and by their height on the shoreline. These nearshore communities are also subjected to foraging and predation by consumer organisms from other upland[9] and nearshore communities. Thus, the ecological resources of the CEM fall within the following ecological groupings:

- Intertidal and shallow, subtidal benthic communities
- Nearshore foragers.

The ecological resources present in the food web of the nearshore coastal ecosystem include those that are currently classified as not recovering or recovering, as indicated in Table 2-1.

Table 2-1. Resource Classification of Recovery Status.

Not Recovering	Recovering
Common loon	Clams
Cormorants (3 spp.)	Intertidal communities
Harbor seal	Killer whales (AB pod)
Harlequin duck	Marbled murrelet
Pacific herring	Mussels
Pigeon guillemot	Sea otter

This classification of recovery status, however, does not represent which resources have the greatest potential for exposure to EVO. Through emphasizing spatial position of each resource within the food web, the model is intended to communicate which resources are clearly associated with lingering EVO. Temporal aspects of the habits of each resource also can be used to communicate associations with lingering EVO. The collective degree of spatial and temporal association between lingering EVO and ecological resources is used to discriminate low-risk from high-risk resources.

Using the CEM as a starting point, detailed evaluation of the ecological resources in the context of their respective intertidal, nearshore subtidal, and nearshore communities, and the possible association of these resources with lingering EVO, is provided in Sections 3 through 11 of this report.

2.3 NATURE AND EXTENT OF LINGERING OIL

This section describes the nature and extent of lingering oil. Processes responsible for determining the nature, extent, and current conditions of lingering oil are presented. Because of

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[9] Exposure to upland foragers is not considered here because none are currently listed as an injured resource.

the distinctive processes that have occurred and continue to occur for oil at the surface versus the subsurface, separate discussions for each are provided.

2.3.1 Surface EVO

At the surface, EVO has been susceptible to a number of chemical and physical weathering processes that have acted to limit (relative to subsurface) its nature and extent in the intertidal zone. For example, surface EVO was directly subjected to Exxon's shoreline cleanup activities (e.g., bulk removal actions, hot-water washing) performed during the months after the spill. Some EVO remained on cobbles, gravel, and surficial sediments. This EVO was subsequently subjected to natural weathering and degradation processes, such as direct wave action and tidal flushing, evaporation, photolysis, and biodegradation. Over time, losses of EVO due to these processes have occurred at progressively slower rates, as the remaining hydrocarbon fractions gradually became less susceptible to weathering (Hayes and Michel 1999; Michel and Hayes 1999; Page et al. 2002b; Short et al. 2004a).

Under current conditions, surface EVO is present in a variety of weathered forms, including asphalt pavement/mousse, some tar balls and tar patties, and weathered surface oil residues (Gibeaut and Piper 1995; Short et al. 2002, 2004a). In addition, a small percentage of surface EVO consists of oil coats and oil films in the intertidal zone (Gibeaut and Piper 1995; Short et al. 2004a). NOAA researchers have recently estimated that approximately 4.13 ha (10.2 acres) of surface EVO may remain in PWS, with the majority being present in the upper half of the intertidal zone (Short et al. 2004a). The majority of surface oil present in the intertidal zone is believed to consist of EVO, with less than 10 percent considered attributable to other sources (e.g., Monterey Formation petroleum products release from tanks during the 1964 earthquake) (Short et al. 2004a).

2.3.2 Subsurface EVO

In contrast with remaining surface EVO, EVO that penetrated the intertidal matrix of cobbles, gravel, and finer sediments to subsurface depths is less susceptible to weathering processes. At depth, particularly below armored beaches, EVO is not subject to the same degree of natural weathering that occurs at the surface. Subsurface EVO was also not directly subjected to the cleanup efforts performed by Exxon. The collective research suggests that less weathered subsurface EVO has been sequestered in beaches armored by boulders and cobbles (Hayes and Michel 1999; Michel and Hayes 1999; Page et al. 2002b; Short et al. 2004a), in low-angle middle intertidal areas (Short et al. 2004a), in finer sediments beneath mussel beds (Babcock et al. 1998), and in thick sediment veneers over bedrock (Hayes and Michel 1999). Subsurface EVO appears particularly evident in moderately to highly sheltered shorelines that were heavily oiled soon after the initial spill (Hayes and Michel 1999; Wolfe et al. 1994; Short et al. 2004a). Researchers have recently estimated that approximately 7.8 ha (19.3 acres) of subsurface EVO, located predominantly in the middle intertidal zone, may remain in PWS (Short et al. 2004a).

Despite being largely sequestered, subsurface EVO is nevertheless subject to some degree of weathering and other transformation/partitioning processes. These processes include dissolution into pore waters and metabolic transformation by microbial and other benthic organisms. Such processes are most likely to result in relatively slow weathering. More rapid changes to sequestered EVO may occur as a result of reintroduction to the sediment surface, where the more rapid transformation and weathering processes occur. Reintroduction may occur following storm events, which may produce significant erosion and changes in beach morphology. For example, such changes were observed following the vigorous storms of the winter of 1992–1993 (Babcock et al. 1998). Re-introduction of subsurface EVO by bioturbation by benthic invertebrates and digging activity of sea otters has also been proposed (Peterson et al. 2003; Rice and Peterson 2004).

2.3.3 Bioavailability and Bioaccessibility

Based on the work of Short et al. (2004a), Page et al. (2002b) and others (e.g., Michel and Hayes 1999; Hayes and Michel 1999) and as described above, lingering oil in surface sediments occurs primarily in the form of highly weathered, solid asphalt-like material sporadically present in the upper intertidal zone of sheltered areas. Because it occurs at the surface, it is considered to be physically bioaccessible to resources. However, toxic components in weathered oil cannot readily dissolve into ambient seawater when the oil is in a solid, asphalt form, and therefore these components are not likely to be bioavailable.

In contrast with remaining surface oil, EVO that penetrated the intertidal matrix of cobbles, gravel, and finer sediments to subsurface depths is less susceptible to weathering processes and hence is less viscous, so that toxic components may dissolve more readily into seawater (Short et al. 2004a; Page et al. 2002b; Michel and Hayes 1999; Hayes and Michel 1999; Wolfe et al. 1994). Although it occurs at depth and may be less physically bioaccessible, this form of EVO is considered to be potentially more bioavailable than surface weathered oil.

Recent research conducted in 2004, using semipermeable membrane devices (SPMDs)[10] placed in the intertidal zone, indicates that where lingering subsurface oil is present, it is bioavailable to intertidal organisms, exists in a bioactive form that is capable of inducing cytochrome P450 1A (CYP1A)[11] in fish, and is distinguishable from stressed reference locations that have sediments contaminated by other non-EVO-related sources of petroleum hydrocarbons (Springman et al. 2005; Short et al. 2005). Although such evidence for bioavailability exists, mussels tested near oil patches do not show elevated levels of PAHs (Page et al. 2005), and the benthic communities living on the oil patches are not significantly different from benthic communities found in unoiled areas (Day 2005). With respect to higher trophic level organisms, in particular harlequin duck and sea otter, CYP1A results indicate that exposure to EVO is limited, as evidenced by recent biomarker convergence between oiled and unoiled areas. Additional details on the exposure and bioavailability of specific resources are discussed in subsequent sections of this report.

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[10] SPMDs are a biomimetic research tool that has been used to simulate water-mediated uptake and bioaccumulation potential of oil and other substances by aquatic organisms.

[11] CYP1A is an enzyme that is induced when an organism is exposed to PAHs.

2.4 OTHER SOURCES OF PETROLEUM HYDROCARBONS

The vast majority of the area in PWS originally impacted by the spill is considered remote wilderness. Despite this remoteness, a variety of petroleum hydrocarbon and other PAH sources occur throughout PWS, including both natural and human sources. An understanding of other potential sources is an important consideration when attempting to define the role lingering EVO may play in ongoing hydrocarbon exposures to animals of PWS. Other sources present in PWS may confound the understanding of EVO's precise role in animal exposures.

2.4.1 Types of Other Sources

There are two major types of hydrocarbon sources in PWS and the surrounding Gulf of Alaska:

- Petrogenic hydrocarbons
- Pyrogenic hydrocarbons.

A general overview is provided below for each of these hydrocarbon sources.

2.4.1.1 Petrogenic Hydrocarbons

Petrogenic hydrocarbons include crude oil and associated refined products as well as oil from natural seeps and eroding shales. Prior to the spill, a major source of petrogenic hydrocarbons was from storage tanks that ruptured during the March 27, 1964, Great Alaska Earthquake, the epicenter of which was just north of PWS. The earthquake was of sufficient magnitude (8.4 on the Richter scale) to uplift shorelines within the spill zone of PWS by up to 10 meters into the supratidal zone. On land, the earthquake resulted in the rupturing of asphalt and fuel oil storage tanks in Valdez and other sites around the Sound and the Gulf of Alaska (Kvenvolden et al. 1995). These asphalt and fuel oil products originated from the California Monterey Formation oils, which are chemically distinct from EVO. Monterey oil was also likely introduced into PWS and the Gulf of Alaska by various fishery activities (including canneries) and other industrial activities (e.g., mining) that previously relied upon California crudes or refined oils (Bence et al. 1996; Kvenvolden et al. 1993). Released asphalt was subsequently deposited as sporadic and small asphalt tar balls stranded by high tides and tightly adhered to cobbles and boulders on shorelines of PWS. As described above in Section 2.2, in its highly weathered form, EVO may also be present in similar asphalt tar balls. Kvenvolden et al. (1995) observed that the majority of small tar balls found in the spill zone were identified as Monterey in origin. Small tar balls observed by Short et al. (2004a) were similarly identified as Monterey, but the larger patches of weathered asphalt were identified as EVO.

Petrogenic hydrocarbon sources also include natural oil seeps that occur along the northern Gulf of Alaska east of PWS, but that are absent in PWS itself (Page et al. 1996). Page et al. (1995, 1996, 1997) suggested that active oil seeps in the Katalla, Yakataga, and other areas further east could contribute to natural background petroleum hydrocarbons in subtidal sediments of PWS. However, an analysis by Short et al. (2004b) suggests that such oil seeps are an insignificant source of PAHs to PWS. Other petrogenic sources contributing to the natural background of hydrocarbons in subtidal sediments include petroleum "source rocks" derived from Tertiary shales and, to a lesser extent, coal sources, both of which are present in the terrestrial margin of the Gulf of Alaska east of PWS. Outcrops of these hydrocarbon-rich rocks and coals are eroded by streams and glaciers and then transported into the westward-flowing Alaska Coastal Current (Bence et al. 1996; Hostettler et al. 1999; Boehm et al. 2001; Page et al. 1995, 1996, 1997, 2002a; Van Kooten et al. 2002; Short et al. 1999, 2004b). Eroded shelf rock originates from Katalla to Yakutat Bay. Short et al. (1999) and Hostettler et al. (1999) identified background hydrocarbon sources attributable to coal originating in the Bering River coalfields east of PWS, and Van Kooten et al. (2002) identified additional coal sources further east as far as the Malaspina glacier. Unlike the case with seep oils, PAHs in eroded coal or source rock would not be bioavailable because they are trapped within solid matrices (Short et al. 2004b).

2.4.1.2 Pyrogenic Hydrocarbons

Pyrogenic hydrocarbons include hydrocarbons generated by forest fires and fossil fuel combustion, including both coal and products refined from petroleum. Page et al. (1998; in press) described potential pyrogenic sources present in subtidal sediments of PWS. Potential sources included those associated with past and current human activities, including villages, sawmills, canneries, and camps where coal, oil, or wood were burned. Forest fires were also identified as a potentially significant source of PAHs in subtidal sediments (Page et al. 1998).

2.4.2 The Question of Other Sources and Lingering EVO

Investigations for other sources of petroleum hydrocarbons were undertaken largely over concerns that EVO initially stranded on intertidal shorelines could be transported to subtidal sediments following cleanup activities and storms in the first years after the spill. As described above in Section 2.2.1, by 2001, little EVO was found in subtidal sediments along shorelines that had been heavily oiled (Short et al. 1996; O'Claire et al. 1996; Neff et al. 1995). While the potential for some offshore contamination has been raised (e.g., Short et al. 1996), overall, a negligible amount of EVO is likely to exist under current conditions in subtidal sediments of PWS (Short et al. 2003).

In the case of the intertidal zone where lingering oil is known to persist, the issue of other sources is less widely debated. Prior to the original spill, little evidence existed to suggest other sources present in the intertidal zone. From 1977 to 1980, Karinen et al. (1993) studied petroleum hydrocarbons potentially present in sediments and mussels obtained from intertidal areas adjacent to the oil tanker vessel transportation corridor through PWS. The study was performed to determine baseline conditions prior to the start of oil tanker movement through PWS. At three of eight sampled stations, low-level hydrocarbon contamination was identified and attributed to localized vessel traffic (e.g., small fuel spills, ballast discharges and fuel-combustion exhaustion). Beyond these areas, intertidal sediments and mussels were found to be relatively free of petroleum hydrocarbon contamination. Karinen et al. (1993) concluded that contaminated sediments and mussels present along shorelines impacted by the EVOS would be readily discernible due to the general absence of other confounding sources of petroleum hydrocarbons. Because this study included relatively few samples collected over a relatively small area it cannot be considered a comprehensive assessment of pre-spill conditions in PWS. Further, the analytical methods employed for the 1977–1980 samples were not wholly sufficient for thoroughly characterizing petroleum hydrocarbons. However, analyses of samples by more modern methods from some of the 1977–1980 stations that were collected just prior to oil landfall in 1989 indicated no significant changes in PAH concentrations during the intervening period, and very low background concentrations of the wider PAH suite analyzed for the 1989 samples (Short and Babcock 1996).

Under current conditions, surface oil is present as weathered asphalts or tar balls originated from either Monterey oils or from stranded EVO (Kvenvolden et al. 1995; Short et al. 2004a). As described, these insoluble forms are not bioavailable. Slightly weathered subsurface oil, on the contrary, is identifiable as EVO alone as reported by Short et al. (2004a) and Taylor and Reimer (2005). Although the subsurface EVO is considered bioavailable, questions remain with respect to its potential access by resources.

2.5 PREDICTION OF CURRENT AREAL EXTENT AND FUTURE PERSISTENCE

This section presents the results of a modeling analysis conducted by Integral that was used to predict the current areal extent of lingering and future persistence of EVO in PWS. Based upon the 2001 NOAA study, the total area of lingering oil remaining in PWS was estimated at approximately 28 acres (Short et al. 2004a). However, because this estimate was based upon a probability-based random sampling approach and not a systematic observational survey, questions remain about the actual location of the lingering EVO within PWS. The goal of this analysis is to build upon the 2001 NOAA predictions by developing a mathematical mechanism through which both the location and probability of lingering EVO could be determined under current and future timeframes.

2.5.1 Objectives

The five main objectives of this modeling analysis are as follows:

1. Model probabilities of surface and subsurface EVO present in PWS in 2005
2. Relate these probabilities to historical SCAT data on oiling extent and location using geographical information systems (GIS) techniques
3. Map locations and probabilities of surface and subsurface EVO occurrence
4. Predict the future persistence of lingering EVO
5. Use results to assist in identification of possible geographic locations of interest.

An additional critical objective of this analysis is to combine the mapped locations and probabilities of lingering EVO with mapped distributions of unrecovered biological resources present in PWS. The results of this objective are described for selected resources in Sections 3 through 11 of this report.

The random sampling approach utilized in the 2001 NOAA study offered a number of distinct advantages over the comprehensive historical SCAT surveys (see Section 2.1). For example, the study provides quantitative, probability-based estimates of lingering EVO throughout western PWS. The statistical-based sampling approach also eliminated reliance on the timely and expensive comprehensive ground surveys traditionally used for SCAT surveys. However, a number of challenges are associated with the application of these estimates for the evaluation of the current extent of lingering EVO and its relation to unrecovered resources in PWS. Foremost, the 2001 NOAA estimates represent total average areas of lingering EVO based on extrapolations from a relatively few samples to all of western PWS. The exact geographic location of the estimated areas is unknown. In addition, information on the geographic distribution of unrecovered resources is available in a GIS format. To provide an understanding of the potential overlap between resources and current lingering EVO, the NOAA data require a translation to geographic location. Lastly, the prediction of both the current and future extent of lingering EVO necessitates the inclusion of an annual loss rate component in the modeling.

2.5.2 Modeling Approach

A mathematical model was used to map the current and future locations and probabilities of lingering EVO in PWS. The mathematical model was developed based upon Monte Carlo style probabilistic analysis techniques.[12] Monte Carlo analysis is a method of estimating the probability of a model result given variability and/or uncertainty in the underlying inputs of the model. Rather than relying upon single input values, Monte Carlo analyses utilize multiple values to characterize a given model input. Model inputs are expressed as distributions of values with known probabilities of occurrence and are referred to as probability density functions (PDFs). By including inputs characterized as PDFs, Monte Carlo analyses provide a more complete understanding of a model estimate by offering insight on both the magnitude and probability of the model output (e.g., the modeled number of remaining acres of EVO and the probability of occurrence).

The model relies upon data collected during the historical SCAT survey conducted between 1989 and 1993. These data are critical because 1) they provide information on the location and linear distances of oiled shorelines; and 2) they were used as the basis of the 2001 NOAA study to identify random locations to investigate. The 1989-1993 SCAT data were obtained in GIS data set layers primarily from the Alaska State Geospatial Data Clearinghouse (ASGDC 2005). The GIS data set layers were combined across years and then mapped. From this mapping, estimates of cumulative distances of oiled shoreline were calculated and segregated by oiling categories (i.e., Category I – heavily oiled, Category II – moderately oiled, Category III – previously heavily oiled and subsequently light to no oiling) as described by Short et al. (2004a).[13] The total lengths of shoreline were 21 km, 54 km, and 44 km for Categories I, II, and III respectively. These distances approximate the cumulative distances of shoreline locations used by NOAA to draw random sample locations.[14] To account for limited variability between the distances

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[12] The Monte Carlo analysis was conducted in accordance with established guidance available in the open literature, including that provided by Hoffman and Hammonds (1994), Burmaster and Anderson (1994), and Thompson et al. (1992).

[13] Shoreline segments initially classified as "lightly oiled" were not sampled.

calculated in this analysis and the reported estimates of Short et al. (2004a), distances were expressed as PDFs in the model.

The model additionally relies upon the probability-based estimates developed by NOAA of the acres of remaining EVO. Bootstrapped data sets of areas of EVO predicted by NOAA were obtained and expressed as PDFs for surface and subsurface oiling categories.[15] The area estimates were subsequently converted to shoreline lengths based upon NOAA's data on the average widths of beach segments sampled in the 2001 study. A PDF for the width of sampled beach segments was developed that is representative of all beach segments, and solving algebraically, the NOAA predicted areas were converted to linear shoreline distances.

To generate estimates of current EVO in 2005, an annual loss rate of 20–26 percent was used based upon Short et al. (2004a). Figure 2-4 depicts the exponential annual loss rate applied to various combinations of the linear shoreline distances of the NOAA surface and subsurface oiling categories.

After the NOAA data were converted to shoreline distances and the annual loss rate was incorporated for the estimation of current shoreline distances, a probability of a shoreline oiling occurrence could be calculated following Equation 1:

$$P_{EVO} = \frac{\sum_{EVO} 2005 shoreline}{\sum_{EVO} Historical\_shoreline} \qquad \text{Eq. 1}$$

Where:

P_{EVO} = probability of oil occurrence
Σ_{EVO}2005 shoreline = 2005 estimate of oiled shoreline distance
Σ_{EVO}Historical_shoreline = cumulative historical oiled shoreline distance

Equation 1 was run in a Monte Carlo computer simulation using commercially available software designed for such analyses.[16]

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[14] The NOAA beach lengths were 24.4, 49.1, and 43.1 km for Categories I, II, and III, respectively (Short et al. 2004a).

[15] Bootstrapping is a statistical method for characterizing the distribution of an estimated value (e.g., an estimated mean area containing EVO) by randomly selecting and propagating data values in the original sample data set. This technique is performed for the purpose of deriving more robust estimates of various statistics, such as standard errors and confidence intervals, from the original data set.

[16] For the Monte Carlo simulation, Crystal Ball software v.7.1 was used as a Microsoft® Excel add-in (Decisioneering 2005). The simulation was run over 10,000 iterations with Latin Hypercube sampling (500 subsamples). Stability in the tails of the predicted forecast (i.e., probability of oil occurrence) was reached under these conditions.

2.5.3 Results

Based upon the Monte Carlo analysis, there is an overall low probability of EVO being present along previously oiled intertidal shorelines in PWS in 2005. The probabilities of EVO occurrence range from 0.6 percent for Category III subsurface oil to 3.9 percent for Category I subsurface oil. Table 2-2 presents a summary of the probabilities of oiling occurrence.

Table 2-2. 2005 Probabilities of Oiling Occurrence in PWS.

Category of Oiling	Probability of Oil Present in PWS—2005[a]	
	Surface	Subsurface
Category I	0.021	0.039
Category II	0.016	0.017
Category III	0	0.006

[a] Probability estimates based on $1 – \alpha = 0.95$

Figure 2-5 depicts the location of oiled shorelines with their corresponding probabilities of oiling occurrence.

NOAA predicted on average that approximately 10 acres of surface EVO and 19 acres of subsurface EVO were present in PWS as of 2001 (Short et al. 2004a). Based upon the Monte Carlo analysis, an estimated 4 acres of surface EVO and 7 acres of subsurface EVO are present in PWS in 2005. Based upon the annual loss rate of 20–26 percent, less than 1 acre of surface and subsurface EVO will be present in 9 years (i.e., 2014). Two locations were identified as areas of potential interest during the analysis. Northern Knight Island and Smith Island are two areas that were initially heavily impacted by the spill. Northern Knight Island has been the focus of additional investigations by NOAA since the original 2001 study. Additional refinements to the Monte Carlo analysis could be integrated based upon the findings of these recent investigations (e.g., SCAT 3). Smith Island was originally identified in the 2001 NOAA study as having a large proportion of sampled locations with EVO present. No additional investigations have been published for Smith Island, though this island was originally identified as having the highest proportion of oiled locations in the 2001 NOAA study. The Monte Carlo analysis could further be used to perform a focused analysis to predict the current and future probability of oil occurrence specifically on Smith Island.

2.5.4 Additional Modeling of Current Extent of EVO

A series of additional model runs were performed to evaluate two key sources of uncertainty in the projection of the current extent of lingering EVO in PWS. Specifically, additional modeling was performed to individually and collectively evaluate 1) the effect of including a 30 percent increase in oiled area attributable to oil potentially present in the lower intertidal zone (undersampled in 2001 for PWS but fully sampled and corroborated for Northern Knight Island in 2003); and 2) the effect of considering a lower decay rate for EVO. Consideration of these uncertainties results in more conservative estimates of the remaining amount of EVO present in PWS as of 2005 as well as the future persistence of EVO.

For the additional model run to evaluate EVO attributable to the lower intertidal zone, the original model described in Section 2.5.2 was revised to include an upwards increase of 30 percent in the total area of oil. Based on the original model, a total of approximately 11 acres (2.2 linear kilometers) was predicted for 2005. Based on this additional model run, an increase to approximately 12 total acres (2.4 linear kilometers) was predicted for 2005.

For the additional model run to evaluate a potentially lower decay rate, the original model was revised to include a PDF incorporating a range of decay rates from 26 to 2.6 percent. The upper-end value of 26 percent is based on the high-end range cited by Short et al. (2004a) for PWS, and the lower-end value of 2.6 percent represents an order of magnitude decrease potentially representative of slower decay rates in locations were EVO may be more likely to persist (e.g., certain beach segments of northern Knight Island). Based on this additional model run, an increase to approximately 26 total acres (5 linear kilometers) was predicted for 2005.

Collectively, when both the uncertainty associated with the potential of 30 percent attributable to the lower intertidal and a potentially slower range of decay rates are incorporated into the model, an increase to approximately 35 total acres (7 linear kilometers) was predicted for 2005.

2.6 CURRENT STATUS OF INJURY AND RECOVERY CLASSIFICATION FOR SEDIMENTS

The comprehensive evaluation and synthesis of lingering EVO described throughout this section was primarily performed to determine and place into appropriate context the ecological significance of lingering EVO present in PWS and nearby areas. An additional objective of the lingering oil evaluation was to assess the current status and recovery classification for sediments.

The 2002 status assessment (EVOS Trustee Council 2002b) states as the recovery objective that "[s]ediments will have recovered when there are no longer significant residues of Exxon Valdez oil on shorelines (both intertidal and subtidal) in the oil spill area. Declining oil residues and diminishing toxicity are indications that recovery is underway."[17]

Though natural weathering processes are reducing the remaining amount of EVO in sediments, some patchily distributed EVO remains in PWS, particularly in subsurface sediments along sheltered and armored beaches that were initially heavily impacted by the spill (e.g., beaches along northern Knight Island). Some limited areas outside of PWS, notably certain beaches located along the Kenai and Alaska Peninsula coastlines, also contain patchily distributed subsurface EVO (present predominantly as mousse in subsurface sediments). Overall, sediments are considered to be recovering by the EVOS Trustee Council (EVOS Trustee Council 2002b). However, as described by the Trustee Council, the presence of surface and subsurface EVO continues to compromise wilderness and recreational values, expose and potentially harm living organisms, and offend visitors and residents, especially those who engage in subsistence activities along still-oiled shorelines.

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[17] Potential sediment toxicity is discussed in Section 3, "Intertidal Communities."

While EVO does undoubtedly remain in PWS and nearby areas, it is patchy in its distribution and is present along shorelines at only a small fraction of the shorelines initially impacted. Based upon Integral's modeled predictions, an estimated 4 acres of surface EVO and 7 acres of subsurface EVO (approximately 11 total acres) may be present in PWS in 2005. Based upon more conservative assumptions and additional modeling, a worst case estimate of approximately 35 total acres is predicted. The ecological significance of this oil is considered to be largely insignificant for biological resources present across PWS as a whole. However, for certain resources present in isolated locations with remaining EVO, exposures to EVO may occur. The ongoing natural decline of EVO will result in a continued decrease in the ecological relevance of remaining EVO across PWS.

2.7 RESTORATION OPTIONS

This section provides a focused review of potential sediment restoration options for locations where lingering EVO is known or suspected to persist. For the purposes of this overview, the subsurface EVO remediation options and technologies recently evaluated by Michel et al. (2005)[18] for PWS and surrounding areas are discussed. As described throughout Section 2, subsurface EVO is largely sequestered and thus has a greater likelihood to persist in a potentially bioavailable form compared to surface EVO. Subsurface EVO has also been predicted to persist over a broader extent and at a higher volume than surface EVO. The evaluation of restoration options therefore has been directed specifically at subsurface EVO.

2.7.1 Overview of Restoration Options for Lingering EVO

Michel et al. (2005) evaluated 11 potential technologies for the remediation of subsurface EVO. The technologies were screened on the basis of effectiveness, implementability, and operational considerations. Those treatment technologies that met a minimum score were further evaluated using a second scoring matrix to address environmental factors; namely, sediment recovery, intertidal community recovery, acute and chronic impacts during implementation, fish and wildlife disturbance during implementation, and the amount of bioavailable oil remaining after termination of cleanup activities.

Michel et al. (2005) identified two restoration approaches as most viable: 1) natural recovery, and 2) nutrient enrichment. Natural recovery precludes active remediation in favor of allowing sediments to recover under normal environmental conditions. Nutrient enrichment is a treatment technique used to hasten the biodegradation of oil and thus, its removal from the environment. Michel et al. (2005) next evaluated these options with respect to costs and environmental impacts. Michel et al. did not judge the explicit ecological benefits of the two options, but instead judged remedy benefits based largely on oil removal efficacy assuming that less oil would mean less of an ecological threat to intertidal and nearshore aquatic resources.

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[18] The Michel et al. (2005) report was prepared for NOAA under Restoration Project No. 050778.

2.7.2 Costs

The natural recovery option includes a monitoring program to document actual loss rates from natural recovery. A monitoring program performed every other year for a period of 10 years (and assuming a 3 percent annual increase in costs) would total $874,000.

Costs associated with the nutrient enrichment option were estimated in a step-wise fashion. A per kilogram mass treatment unit cost was derived by Michel et al. (2005) using cost information on 17 selected sites with known subsurface EVO. This unit cost was applied to the total estimated mass of subsurface EVO estimated by Short et al. (2004a) of 71,500 kg for all of PWS.[19] Based on this total mass, and including a complementary monitoring program similar to that envisioned under the natural recovery option, the total cost was estimated to be $49,925,000. This equates to a cost of approximately $700 per kilogram of subsurface EVO. Although the total mass PWS subsurface EVO has been estimated, the actual locations of subsurface EVO for all of PWS have not been determined. Therefore, as acknowledged by Michel et al. (2005), the total cost estimates do not include finding the subsurface EVO prior to remediation. The total costs also do not include project management, oversight by government agencies, and multi-year application.

Overall, Michel et al. found that:

- Natural recovery was the least expensive option. The predicted costs for natural recovery were less than a million dollars, whereas those associated with nutrient enrichment were predicted to reach nearly $50 million.
- Natural recovery has fewer environmental impacts associated with implementation than nutrient enrichment.[20]
- Nutrient enrichment has greater environmental benefit, with benefit defined as removal of bioavailable oil.

Michel et al. (2005) acknowledged that the next step in the decision-making process is to conduct a detailed benefit-cost analysis, comparing the ecological benefits of removal of the lingering oil with the financial and ecological costs associated with remedial efforts.

Some uncertainty remains with respect to the likely effectiveness of the nutrient enrichment option. According to technical experts, including Jacqui Michel and NOAA researchers Short and Rice 2006, pers. comm.), this technology has not been applied to remove subsurface oils in

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[19] This estimate is inclusive of the 30 percent additional amount of oil assumed to be present in the lower intertidal zone based upon Short et al. (2004a, 2006).

[20] No environmental impacts would occur under the natural recovery option because active remediation is not conducted. For nutrient enrichment, pits would have to be dug in order for nutrients to be placed within the appropriate proximity to lingering oil. This process would be expected to cause some physical disturbance of intertidal sediments and communities.

armored and rocky beaches prevalent in PWS. It has been used effectively to remove subsurface oil in sandy beach settings in Canada and was also effective in removing subsurface oil in PWS when applied to the surface in areas where surface and subsurface oil existed. These treatment scenarios are not directly applicable to the current subsurface oiling situation in PWS and therefore, additional research to identify factors limiting microbial degradation of subsurface oil and additional treatability studies would be needed before an effective treatment program could be developed.

2.8 SUMMARY AND CONCLUSIONS

On March 24, 1989, approximately 11 million gallons of Alaskan North Slope crude oil cargo from *T/V Exxon Valdez* was spilled into the open waters of PWS. Seventeen years after the spill, small quantities of EVO continue to persist in intertidal zones of the PWS shoreline and along the shoreline southwest of PWS. Lingering oil in surface sediments occurs primarily in the form of highly weathered, solid asphalt-like material sporadically present in the upper intertidal zone of sheltered areas. Because it occurs at the surface, it is physically bioaccessible to resources. However, weathered oil in the solid, asphalt form is insoluble, and therefore unlikely to be bioavailable. In contrast, EVO that penetrated the intertidal matrix of cobbles, gravel, and finer sediments to subsurface depths is less susceptible to weathering processes. Although it occurs at depth and may be less physically bioaccessible, this form of EVO has the potential to be more bioavailable than surface weathered oil.

2.8.1 Sources

The vast majority of the area in PWS originally impacted by the spill is considered remote wilderness. Despite this remoteness, a variety of petroleum hydrocarbon and other PAH sources occur throughout PWS, including both natural and human sources. There are two major types of hydrocarbon sources in PWS and the surrounding Gulf of Alaska: petrogenic (produced from unburned petroleum) and pyrogenic (produced from the combustion of fossil fuels or organic matter). Prior to the spill, a major source of petrogenic hydrocarbons was the March 27, 1964, Alaska earthquake. The earthquake resulted in the rupturing of asphalt and fuel oil storage tanks in Valdez and other sites around the PWS and the Gulf of Alaska. These asphalt and fuel oil products originated from the California Monterey Formation oils, which are chemically distinct from EVO.

Hydrocarbons originating from background sources are present throughout PWS in subtidal sediments. Sources of background hydrocarbons include eroding petroleum “source rocks” derived from Tertiary shales in areas east of PWS, coals originating in the outcrops of the Kulthieth Formation east of PWS (including the Bering River coalfields), and natural oil seeps (of doubtful significance) that occur throughout the northern Gulf of Alaska, but that are absent in PWS itself. Additional sources of hydrocarbons are associated with human activity, including more recent, smaller oil spills, and activities at villages, sawmills, canneries, and camps where coal, oil, or wood were used or burned (Page et al. 1998, 2006).

2.8.2 Remaining EVO

Based on work performed in 2001, Short et al. (2004a) predicted that approximately 10 acres of surface EVO and 19 acres of subsurface EVO were present in PWS. Additional related work performed in 2003 (Short et al. 2006) demonstrated that lingering oil extended further into the lower intertidal than originally anticipated, suggesting that the original estimate may be low by as much as 30 percent. A recent Exxon-funded survey suggests that a smaller area than that estimated by Short et al. (2004a) may remain (Taylor and Reimer 2005). Consensus has been reached that whatever EVO does remain is a small fraction of the total area oiled in 1989.

In consultation with NOAA, a probabilistic technique was used to project the location of lingering oil in 2005, as well as the areal extent and volume. Based upon a Monte Carlo analysis and an assumed annual loss rate of 20–26 percent, an estimated 4 acres of surface EVO and 7 acres of subsurface EVO were projected to be present in PWS in 2005 and less than 1 acre of EVO will be present in 2014 (i.e., 9 years). Based upon additional modeling using more conservative assumptions, a worst case estimate of approximately 35 total acres is predicted for 2005. Two locations were identified as areas of potential interest during the analysis. Northern Knight Island and Smith Island are two areas that were initially heavily impacted by the spill. Northern Knight Island has been the focus of additional investigation by NOAA during 2003 to evaluate further the presence of subsurface oil throughout the intertidal zone.

2.8.3 Ecological Significance of Lingering EVO

Intertidal sediments are the primary repository for lingering EVO and are the source of any ongoing exposure to resources that may live in proximity to or come in contact with sediments or oiled shoreline. The intertidal resources that live within or upon the sediment and rocky shorelines impacted by lingering EVO are discussed in Section 3. Other resources that may come in contact with sediments in early life stages or during foraging are discussed in subsequent sections.

A small amount of oil from the spill (conservatively estimated to be approximately 11 to upwards of 35 acres in 2005 for PWS) may remain in intertidal sediments and rocky shorelines located in and around PWS. Much of this lingering oil is of limited bioavailability and bioaccessibility. Direct testing of remaining oiled sediment patches using SPMDs, SPMD extracts injected into fish, and toxicity tests have demonstrated that this oil can be toxic to some species of test organisms and that it can induce CYP1A, an enzyme that is induced when an organism is exposed to PAHs. Mussels tested near the oil patches do not show elevated levels of PAHs and the benthic communities living on the oil patches are not significantly different from benthic communities found in unoiled areas.

Given the estimated volume and areal extent, it is unlikely that lingering oil continues to be a significant source of exposure to more than a very few percent at most of the resources that come in contact with intertidal sediments in and around PWS.

Recent research conducted using SPMDs in the intertidal zone indicates that where lingering subsurface oil is present, it is bioavailable to intertidal organisms and exists in a bioactive form that is capable of inducing CYP1A in fish (Springman et al. 2005; Short et al. 2005). Although such evidence for bioavailability exists, mussels tested near oil patches do not show elevated levels of PAHs (Page et al. 2005), and the benthic communities living on the oil patches are not significantly different from benthic communities found in unoiled areas (Day 2005). With respect to higher trophic level organisms, in particular sea otter, CYP1A results indicate that exposure to EVO is limited, as evidenced by recent biomarker convergence between oiled and unoiled areas. For some sea birds and sea ducks (Ballachey et al. 2006), CYP1A levels remain elevated in oiled areas relative to unoiled areas. Collectively, these data indicate that although some bioavailability and potential adverse impacts are possible for some organisms in direct proximity to areas with remaining EVO, the overall ecological significance of this for the PWS ecosystem is limited.

2.8.4 Sediment Recovery

"Sediments will have recovered when there are no longer significant residues of Exxon Valdez oil on shorelines (both intertidal and subtidal) in the oil spill area. Declining oil residues and diminishing toxicity are indications that recovery is well underway" (EVOS Trustee Council 2002b). Declining oil residues have been well established, and declining toxicity corresponds to declining oil residues. The fundamental issue is one of significance. The remaining oil has limited ecological significance; however, it likely results in localized exposure and (possibly) adverse impacts to some resources that live in direct proximity to the remaining oil patches. Validation of the projected rate of recovery (i.e., loss rate) of lingering oil in PWS is recommended.

2.8.5 Designated Wilderness Areas

"Designated wilderness areas will have recovered when oil is no longer encountered in them and the public perceives them to be recovered from the spill" (EVOS Trustee Council 2002b). This recovery objective focuses entirely on the presence or absence of oil and on public perception. While oil residues have been dramatically reduced through natural recovery, some oil remains.

A quantitative assessment of residual pockets of lingering oil would provide useful perspective on the nature and magnitude of lingering oil in designated wilderness areas. At present, the only recent assessment of oil in wilderness areas is largely qualitative in nature.

2.8.6 Restoration Actions

Michel et al. (2005) evaluated potential technologies for the remediation of subsurface EVO in PWS. The technologies were initially screened on the basis of effectiveness, implementability, and operational considerations. Selected technologies were subsequently evaluated to consider a variety of environmental factors. Based on this evaluation, Michel et al. (2005) identified two restoration approaches as most viable: 1) natural recovery, and 2) nutrient enrichment. Natural recovery precludes active remediation in favor of allowing sediments to recover under normal environmental conditions. Nutrient enrichment is a treatment technique used to hasten the biodegradation of oil and thus, its removal from the environment. Michel et al. (2005) next evaluated these options with respect to both costs and environmental impacts.

Overall, Michel et al. found that:

- Natural recovery was the least expensive option. The predicted costs for natural recovery based on biannual monitoring were less than a million dollars.
- Natural recovery has fewer environmental impacts associated with implementation than nutrient enrichment.
- The costs associated with nutrient enrichment were predicted on the basis of extrapolated costs associated with treating an estimated 71,500 kg of subsurface EVO predicted for PWS, for a total cost of $50 million.
- Nutrient enrichment is predicted to cost approximately $700 per kilogram of subsurface EVO.
- Total costs for nutrient enrichment of subsurface EVO do not include finding the subsurface EVO prior to remediation, nor do they include project management and oversight by government agencies.
- Nutrient enrichment has greater environmental benefit, with benefit defined as removal of bioavailable oil.

Michel et al. (2005) acknowledged that the next step in the decision-making process is to conduct a detailed benefit-cost analysis, comparing the ecological benefits of removal of the lingering oil with the financial and ecological costs associated with remedial efforts.

Some uncertainty remains with respect to the likely effectiveness of the nutrient enrichment option. For example, this technology has not been applied to remove subsurface oils in the types of armored and rocky beaches prevalent in PWS. Because treatment scenarios are not directly applicable to the current subsurface oiling situation in PWS, additional research to identify factors limiting microbial degradation of subsurface oil and additional treatability studies would be necessary.

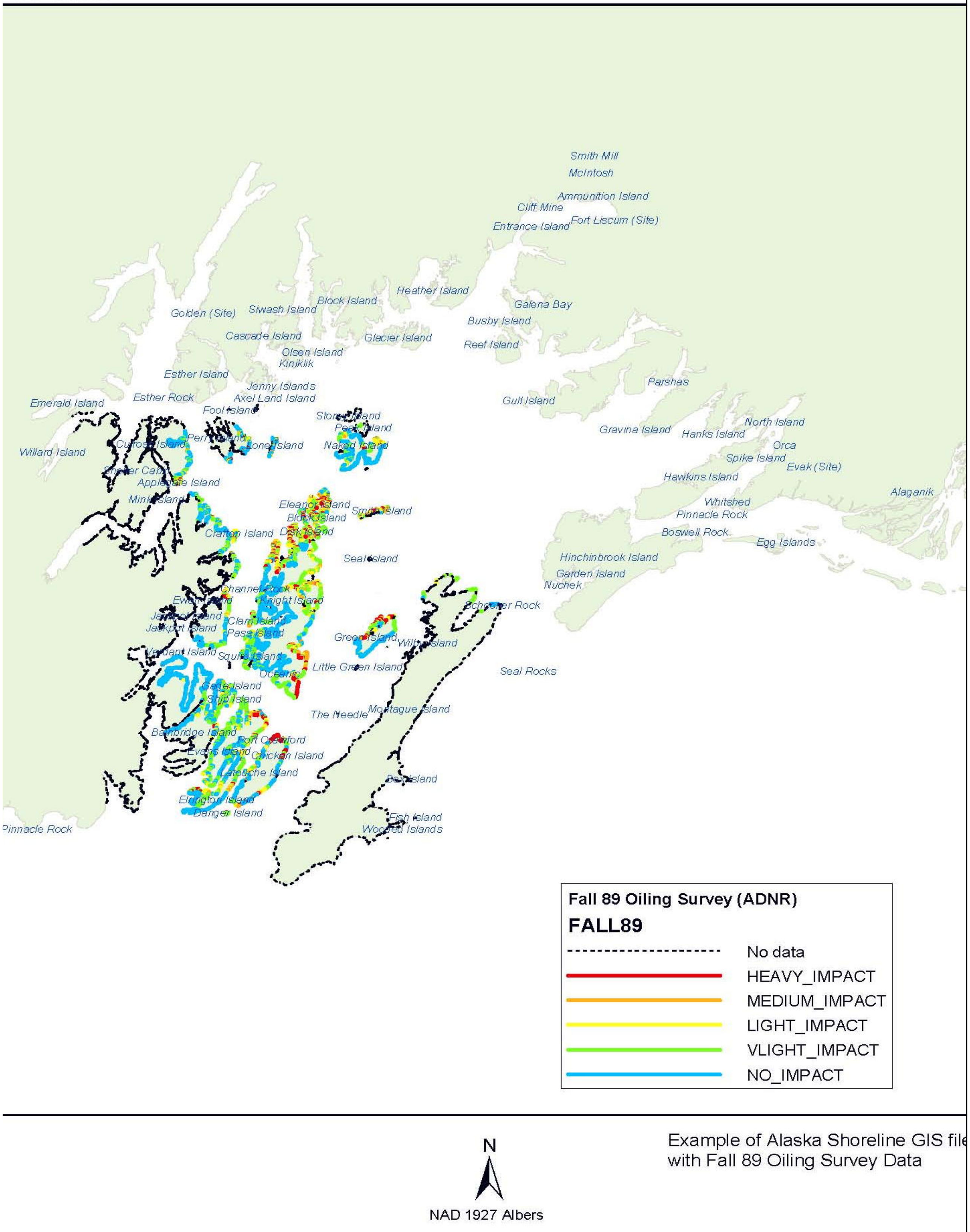
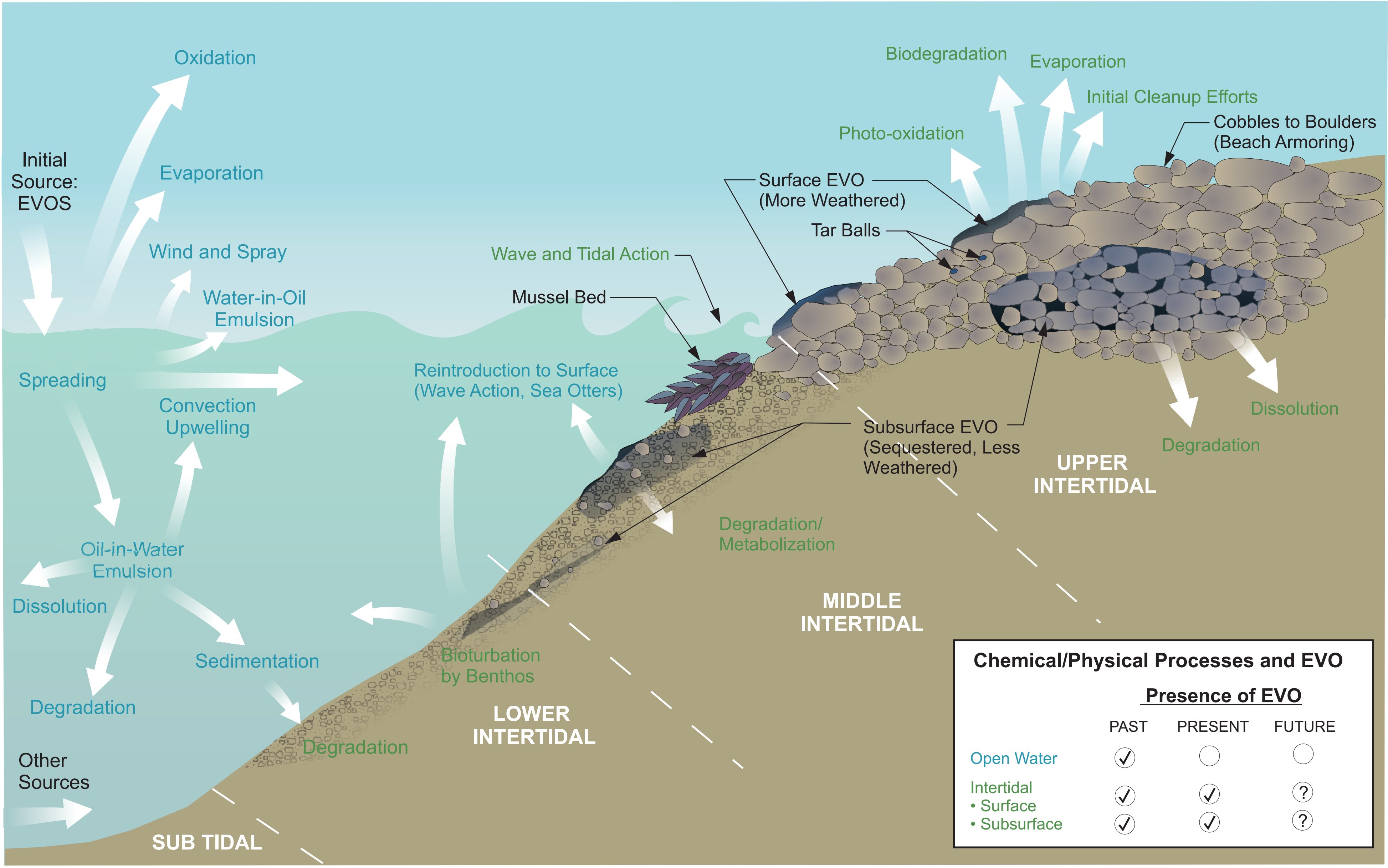


Figure 2-1. Depiction of Oiling in Prince William Sound According to Fall 1989 SCAT Survey.



Chemical/Physical Processes and EVO

	Presence of EVO		
	PAST	PRESENT	FUTURE
Open Water	✓	○	○
Intertidal			
• Surface	✓	✓	?
• Subsurface	✓	✓	?

Figure 2-2. Exxon Valdez Oil General Conceptual Exposure Model - Chemical/Physical Processes

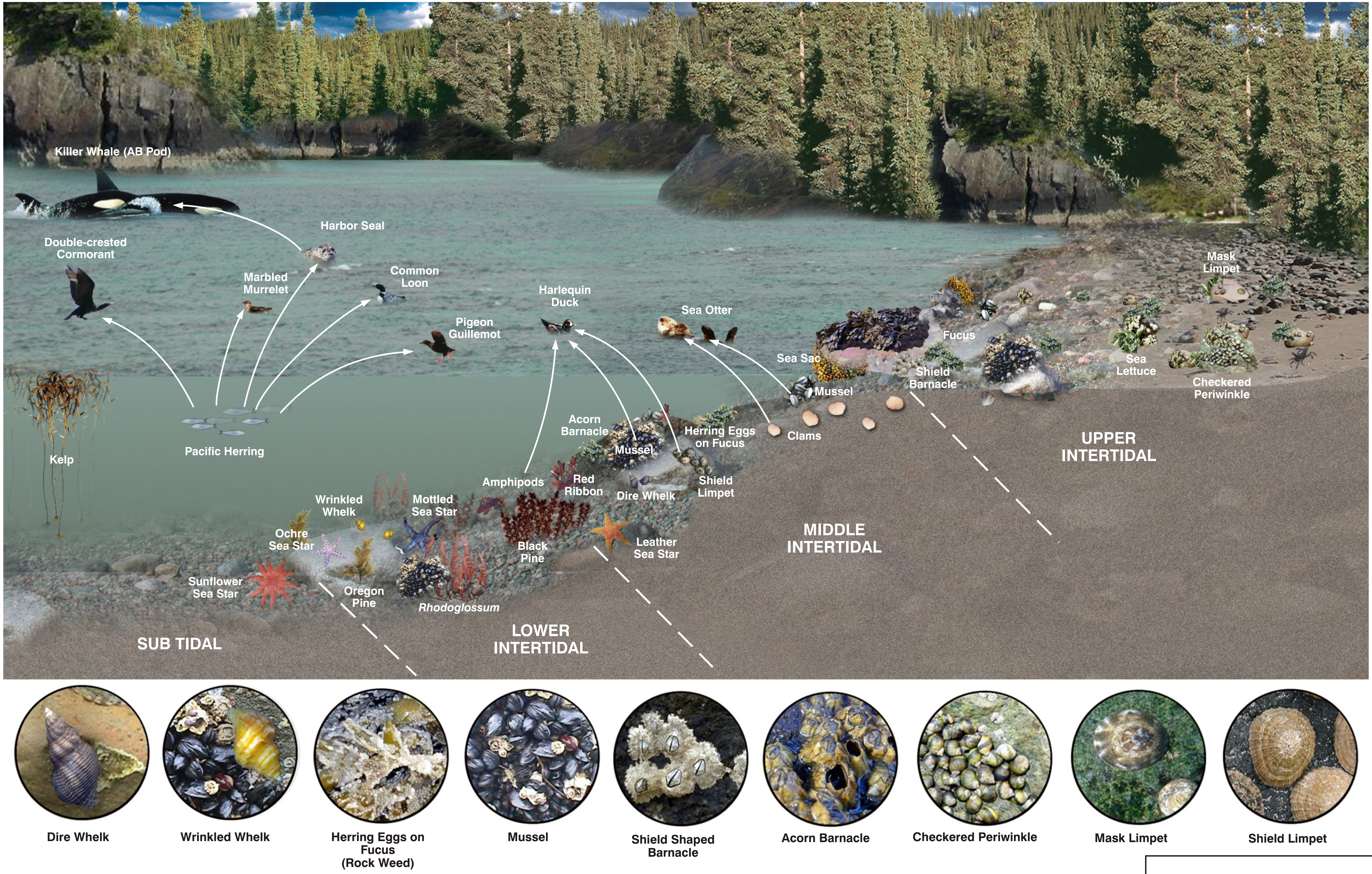


Figure 2-3 Exxon Valdez Oil General Conceptual Model - Food Web Relationships for Injured Resources

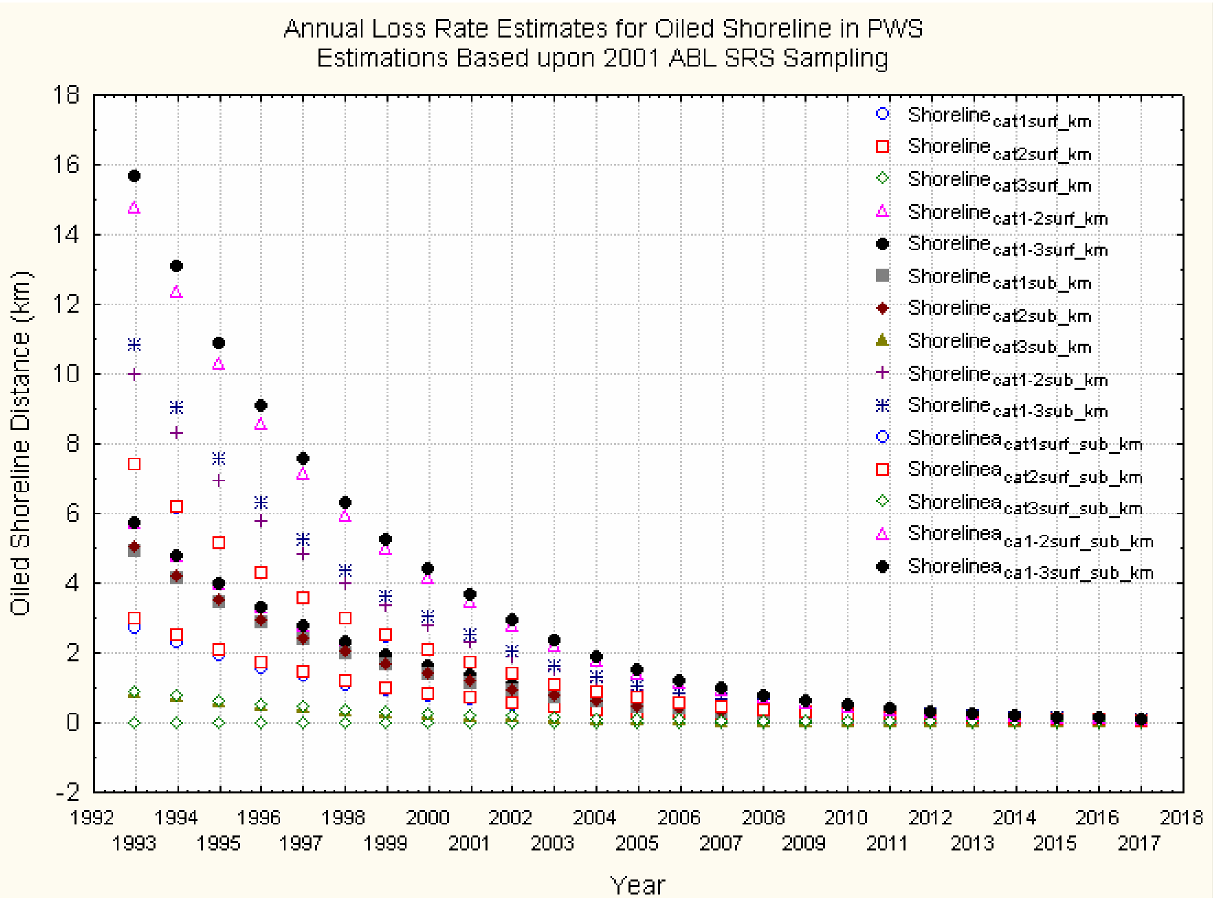
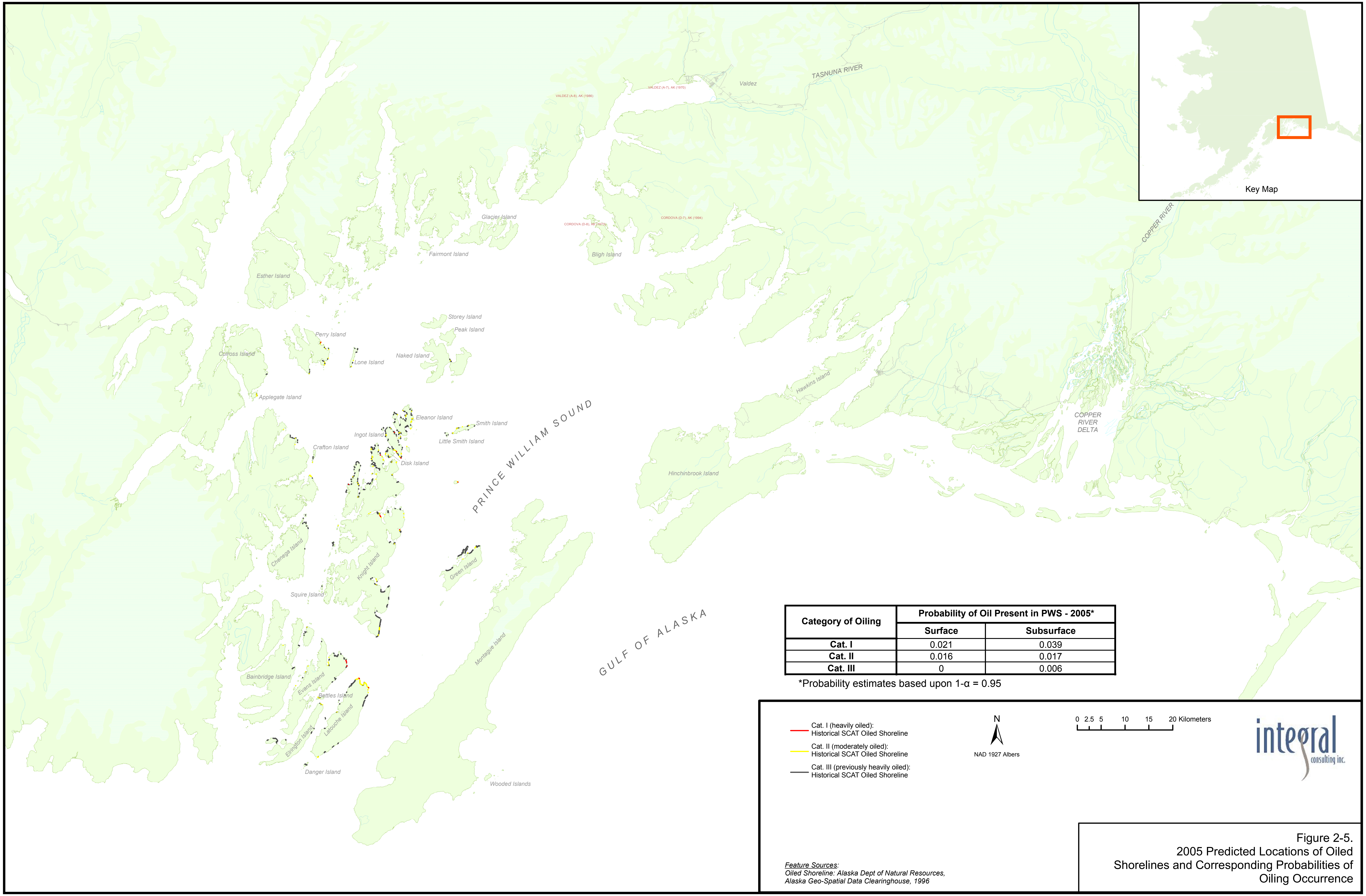


Figure 2-4. Exponential Annual Loss Rate of Lingering *Exxon Valdez* Oil.



Category of Oiling	Probability of Oil Present in PWS - 2005*	
	Surface	Subsurface
Cat. I	0.021	0.039
Cat. II	0.016	0.017
Cat. III	0	0.006

*Probability estimates based upon 1-α = 0.95

Feature Sources:
Oiled Shoreline: Alaska Dept of Natural Resources,
Alaska Geo-Spatial Data Clearinghouse, 1996

Figure 2-5.
2005 Predicted Locations of Oiled Shorelines and Corresponding Probabilities of Oiling Occurrence

\\Mi-gcocks\GIS_projects\C113_EVOS_NRTEVOS_Oiling-Probability-Resource-B-map_GJC-updated_11x17_12-02-05.mxd GJC @ 02-22-06

3. INTERTIDAL COMMUNITIES

The EVOS resulted in the oiling of approximately 1,500 km of south-central Alaska's coastline, with heavy oiling affecting approximately 350 km of this area. This event had major impacts on the intertidal communities, particularly to the upper intertidal zone (EVOS Trustee Council 2002a). The initial spreading of EVO in open water was exacerbated by a series of significant storm events resulting in EVO washing ashore across PWS over the course of a 2-month period. Beach cleaning occurring within the first months after the event, particularly high-pressure, high-temperature washing, also caused mortality of intertidal biota, with long-term repercussions on community composition in some cases (Houghton et al. 1996; Highsmith et al. 1996).

Following the spill, the EVOS Trustee Council identified recovery under way for the lower and mid-intertidal communities, but persistence of injury to communities in the upper intertidal zone, especially on rocky shores (EVOS Trustee Council 1994). The recovery objective was defined as a return to the community composition, intertidal abundance, population abundances, age-class distribution, and ecosystem function and services in each injured intertidal habitat, for each intertidal elevation (lower, middle, and upper), that would have prevailed in the absence of the oil spill. The Trustee Council further defined recovery objectives to include the reestablishment at sheltered rocky sites of important intertidal species such as *Fucus*, the lack of differences in community composition and organism abundance between oiled and unoiled shorelines, and the ability for intertidal and nearshore habitats to provide adequate, uncontaminated food supplies for top predators (EVOS Trustee Council 2002b). Strategies adopted by the Trustee Council to meet the recovery objectives included research into the causes of non-recovery, restoration efforts to assist or accelerate recovery, ongoing monitoring to detect recovery status, and protection of intertidal communities through maintenance of water quality and reduction of marine pollution.

In 2002, the Trustee Council identified the status of the intertidal community as recovering, based on substantial progress in recovery, but also acknowledged that recovery was not complete based on the lack of full recovery of some intertidal invertebrates (e.g., clams) occupying fine-grained sediment beaches (gravel/sand, sand, and silt substrates), the continued presence of residual oil in the intertidal zone, and the role of oil in initiating *Fucus* population instability (EVOS Trustee Council 2002b). Recent research conducted by NOAA shows the presence of approximately 28 acres of sporadically distributed lingering EVO on PWS beaches (Short et al. 2004a). This persisting EVO may act as a potential source of exposure to non-recovering and recovering resources. Consequently, this injury reevaluation independently reviews and assesses the technical basis for the recovery status of the intertidal communities of PWS. The available scientific information is critically evaluated and used to determine whether there is a linkage between lingering EVO and the incomplete recovery of the intertidal community in PWS.

3.1 BACKGROUND

This section describes the natural history of intertidal communities, initial impacts of EVOS, and the current status of injury and recovery.

3.1.1 Natural History of Intertidal Communities

Exceptionally productive biological communities are found in intertidal zones, as a result of the confluence of nutrient runoff from the land and planktonic foods from the photic zone of the ocean pushed into the coastline by winds and waves. Intertidal communities are of intrinsic importance in PWS, providing important ecosystem services such as food resources for a wide variety of marine and terrestrial consumers. Consumers of intertidal flora and fauna include human subsistence users, bear, sea and river otters, marine crabs and shrimp, rockfish, cod, juvenile fish that use the intertidal for foraging and refuge, and a variety of birds, including black oystercatchers, harlequin ducks, and pigeon guillemots (EVOS Trustee Council 1994; Peterson et al. 2001).

The intertidal communities of the Gulf of Alaska are characterized by a rich diversity of marine fauna and flora whose populations are governed by the physical and chemical gradients of their environment, as well as by biological forces including predation and competition. Important gradients that structure intertidal communities are exposure to air by tidal excursions, substrate type, and exposure to wave energy. These factors have led to a system of classification of intertidal habitat by its substrate type, degree of exposure to wave energy, and vertical position in relation to tidal elevations.

3.1.1.1 Vertical Gradients in Intertidal Habitat—Zonation

Tides in PWS are semidiurnal; two highs and two lows occur each lunar day, with successive high and low waters having different heights. Tidal heights vary depending on coastal geometry and are typically expressed relative to vertical distance above or below MLLW. In PWS, the intertidal zone ranges from extreme low water (–1.4 m) to extreme high water (+5.1 m). However, these tidal extremes occur only a few times a year. The intertidal zone is more typically characterized by the semidiurnal fluctuations between the elevations of MLLW (0 m)[21] and mean higher high water (+3.8 m).

Within these few meters of vertical space, intertidal communities are shaped by a series of gradients that govern organism abundance and distribution, and result in often visually striking changes, or zonation, in the intertidal community. Vertical zonation is a prominent feature of Gulf of Alaska intertidal communities, particularly on rocky shores. Plants and animals of temperate rocky intertidal zones tend to exhibit strong patterns of vertical zonation. Physical factors, including temperature and emersion, strongly influence species' distribution at the upper limit of the intertidal zone (though shorebird predation can play an important role in limiting upper limits for some intertidal species), whereas biological factors, particularly predation, often function to control the distribution of many plants and animals in the lower intertidal zone (Raffaelli and Hawkins 1996).

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[21] The actual elevation of the MLLW benchmark can vary slightly from the ideal of 0 m depending on month-to-month and year-to-year variability in local conditions. For example, in 1989, the average monthly elevation of MLLW in PWS was +0.1 m.

Although the exact terms used to describe vertical zonation patterns vary depending on researcher and geography, there is general agreement among descriptions of Pacific Coast intertidal communities that outlines distinct zones of communities along a vertical gradient. However, there is a variety of schemes for characterizing vertical gradients in the intertidal zone (Foster et al. 1990). Some are defined based on dominant taxa in the community. For example, Shigenaka (1997) defined lower, middle, and upper intertidal zones based on the distribution of rockweed (*Fucus*), which dominates the middle intertidal of rocky habitat. Other vertical classification schemes are based on the relative degree of exposure to air and position with respect to a fixed tidal benchmark. For example, Highsmith et al. (1996) defined zones based on vertical distance below the mean high water elevation. Consequently, low-, middle-, and high-intertidal zones may be defined differently depending on the objectives of a particular study. For this evaluation, we define a framework of three intertidal zones (high, middle, and low) divided evenly between extreme low water and extreme high water in PWS (Figure 3-1). Such a physically defined framework provides a common denominator for evaluating EVO, lingering oil, and resource injury for this highly complex and diverse community.

3.1.1.2 Horizontal Gradients in Intertidal Habitat—Substrate Type and Wave Energy

Substrate type is a major physical factor in shaping intertidal communities. Intertidal substrates range from bedrock or large boulder systems with little or no interstitial space, to cobble and mixed cobble/gravel beaches, to sandy beaches, and then mudflats. Compared to gravel or sandy beaches, the rocky intertidal area presents a stable substrate, whereas the animals of the sandy beach are well adapted to a mobile substrate. Whereas the rocky intertidal area of the northeast Pacific is one of the best-studied marine ecosystems in the world, research on sandy beach communities is less comprehensive.

Distinct changes in community composition are also seen along gradients of exposure to wave energy. High wave exposure results in a community dominated by algal species that can withstand abrasion and fast flow, and by animals such as mussels and barnacles that are resistant to detachment. In moderately wave-exposed areas, a broader array of more mobile fauna such as crabs and sea stars can be found using refuges such as rock overhangs to withstand wave exposure. A variety of more delicate plants and animals are found in more sheltered locations.

Based on beach geomorphology and degree of exposure, five principal intertidal habitats were identified for EVOS-related studies in PWS, the Kenai Peninsula, Lower Cook Inlet, and Kodiak Island (Sundberg et al. 1996; Highsmith et al. 1996) (Figure 3-2):

- Exposed rocky shores—These shores are predominantly composed of bedrock, very large boulders (>50 cm diameter), boulder-cobble, or wave-cut platform habitats that are exposed to high energy waves. Disturbance, both physical and biological, and larval recruitment rates are among the factors that can modify the community composition and abundance of species. Because of the intense biological interactions among species, changes in the abundance or distribution of one or more species can have strong direct and indirect effects on other members of the intertidal ecosystem. Bedrock shores consist of a supralittoral *Verrucaria* fringe and a mid-intertidal *Fucus*-barnacle-mussel zone; the low intertidal zone is dominated by fleshy red algae at five of seven sites, with average cover exceeding 50 percent, although *Fucus* and a number of green algae are also diagnostic of these sites (Lindstrom et al. 1999). Boulder-cobble sites are characterized by *Acrosiphonia arcta*, *Fucus gardneri*, and "*Ralfsia*" sp. in the low zone; the mid-zone is mostly devoid of vegetation (Lindstrom et al. 1999).

- Sheltered rocky shores—These shores are bedrock shores of variable slope (from vertical cliffs to wide, rocky ledges) that are sheltered from exposure to most wave and tidal energy. Wide shores may have some surface sediment, but bedrock is the dominant substrate type. Species density and diversity vary greatly, but biota are often very abundant (NOAA 2000).
- Coarse-textured beaches and exposed tidal flats—These beaches and tidal flats comprise intermediate-sized substrates, including gravel, sand, and mixed gravel/sand < 5 cm diameter. Gravel and sandy beaches may occur on shorelines that are directly exposed to oceanic waves, and in more protected coastal areas. Coarse-textured beaches are dynamic habitats where the primary substrate can be easily moved by wave-generated currents; therefore, the biological community of these beaches must be adapted to a mobile, abrasive substrate. Mixed gravel/sand/silt (mixed-soft) sites are characterized by rockweed (*Fucus gardneri*), mussels, and barnacles in the mid-intertidal zone and by the green alga *Cladophora sericea*, *Fucus*, and the filamentous brown alga *Pilayella littoralis* in the low intertidal zone; mussels and barnacles are common associates in the latter zone. The red alga *Polysiphonia aff. tongatensis* is found only at mixed-soft sites (Lindstrom et al. 1999).
- Fine-textured beaches—These beaches are generally flat and hard-packed. Though they are predominantly fine sand, there is often a small amount of shell hash. There can be heavy accumulations of wrack present. They are utilized by birds and turtles for nesting and feeding. Upper beach fauna are generally sparse, although amphipods can be abundant; lower beach fauna can be moderately abundant, but highly variable.
- Sheltered estuarine shores (tide flats, marshes)—Sheltered tidal flats are composed primarily of mud with minor amounts of sand and shell. They are present in calm-water habitats, are sheltered from major wave activity, and are frequently backed by marshes. The sediments are very soft and cannot support even light foot traffic in many areas (NOAA 2000). They can be sparsely to heavily covered with algae and/or seagrass. They can have very heavy wrack accumulations along the high-tide line. There can be large concentrations of shellfish, worms, and snails on and in the sediments. They are heavily utilized by birds and fish for feeding.

The majority of these shorelines (approximately 58 percent) are composed of sheltered rocky shores and coarse-textured (gravel and mixed sand-gravel) beaches. Exposed rocky shores account for the majority of the remaining types of shorelines (Ford et al. 1996). Fine-grained beaches and sheltered estuaries are a small, but biologically important, component of the shoreline habitats. All but fine-textured beaches were evaluated for EVOS-related impacts in PWS.

3.1.1.3 Temporal Variation

In addition to variation at spatial scales, intertidal communities also vary over time. Temporal patterns are not obvious or well described, and few long-term studies exist to describe such patterns. A general seasonal summer maximum and winter minimum of growth may be typical of intertidal communities, and may be most pronounced in Alaska, where seasonal variation in light exposure is highly variable. For example, coverage of the intertidal zone by the annual brown alga *Alaria* sp. varied from 20 percent in winter to 100 percent in summer of one year (Foster et al. 1991). However, finding temporal patterns is vastly complicated by variability of offshore conditions, localized disturbance, and patterns of recruitment of the larvae of intertidal organisms.

Intertidal communities in PWS have also been influenced by periodic disturbance associated with geophysical processes. The Great Alaska Earthquake of 1964 had profound, long-term effects on the intertidal biota of PWS. The entire spill zone is within the area that was uplifted up to 15 m, and in some areas subsided more than 1m, from its original pre-earthquake elevation (Hanna 1971). In areas where uplift exceeded 3 m, more than one-third of the clam population and virtually the entire mussel community were destroyed. The earthquake also decimated grazing animals and displaced algal species (e.g., *Fucus*) that are important structural elements of the community. Subtidal biota were uplifted to intertidal elevations and perished within a year. The earthquake also affected the intertidal and subtidal benthic habitat. In some areas, the violent shaking and subsequent washing by tsunami or strong currents scoured much of the nearshore sediment and washed it to deeper water. In summary, the earthquake devastated the intertidal community, transformed much of the benthic habitat, and triggered an ecological succession of recruitment and recolonization that continued for many of the 25 years prior to the spill.

3.1.2 Initial Impact of EVOS

Approximately 40–45 percent of the estimated 10.8 million gallons of crude oil spilled by the *Exxon Valdez* is estimated to have washed ashore in the intertidal zone of PWS (Wolfe et al. 1994). Aerial surveys by the Alaska Department of Natural Resources showed that light to heavy deposits of EVO washed ashore on approximately 24 percent (446 km) of the 1,891 km of the PWS shoreline observed (Peterson 2001), and Exxon surveys documented oil on 16 percent (783 km) of an estimated 5,000 km of PWS shoreline in 1989 (Neff et al. 1995). The spill area included all types of intertidal habitats, including exposed and sheltered rocky shores; boulder, cobble, and gravel beaches; mixed sand-gravel and fine sand beaches; exposed and sheltered tidal flats; and estuaries. Beginning in the summer of 1989, three kinds of studies were initiated to evaluate the initial impact of the spill, assess its rate of recovery, and document effects associated with various shoreline treatments to remove oil:

- The Shoreline Ecology Program (Page et al. 1995; Gilfillan et al. 1995)
- The Coastal Habitat Injury Assessment Program (Highsmith et al. 1996; Stekoll et al. 1996; van Tamelan and Stekoll 1996)
- NOAA Injury Assessment (Driskell et al. 1996; Houghton et al. 1996; Lees et al. 1996; Mearns 1996).

Although the methods and objectives varied among the studies (Peterson 2001), each was stratified to evaluate EVOS-related impacts along the horizontal and vertical gradients that define intertidal community composition (Figure 3-1). In addition to these studies, surveys of the distribution of oil in the outer tidal zone over time were conducted pursuant to the SCAT surveys (see Section 2.1.1).

3.1.2.1 Direct Effects of EVO

Initial impacts to intertidal organisms occurred at all tidal levels and in all types of habitats throughout the oil spill area. Direct assessment of effects included sediment toxicity testing, abundances of intertidal organisms, and ecological parameters of community structure.

Sediment toxicity tests were conducted to assess effects of EVO beginning in 1989 and continuing through 1991 (Page et al. 1995, 2002b; Boehm et al. 1995; Wolfe et al. 1996). Toxicity tests were performed with representative benthic organisms[22] (the amphipods *Rhepoxynius abronius* and *Ampelisca abdita*, and Pacific oyster larvae *Crassostrea gigas*) using standard bioassay protocols. Test sediments were collected from various intertidal elevations in boulder/cobble, coarse-textured (pebble/gravel), and soft-bottom habitats. Comparisons with reference locations or controls indicated significant sediment toxicity in the *R. abronius* test in primarily the upper intertidal zone of both cobble/boulder and coarse-textured habitats in 1990. *R. abronius* toxicity was also present but at diminished levels in coarse-textured habitat in 1991. This pattern of diminished toxicity over time was also evident in tests with *Ampelisca abdita* and Pacific oyster larvae. In comparisons with reference locations, significant toxicity was only evident in the *Ampelisca abdita* test in 1990 (Wolfe et al. 1996).

Dominant species of algae and invertebrates that were directly affected by the spill were common rockweed (*Fucus gardneri*), speckled limpet (*Tectura persona*), several barnacle species (*Chthamalus dalli*, *Balanus glandula*, *Semibalanus balanoides*), blue mussels (*Mytilus trossulus*), periwinkles (*Littorina sitkana*, *L. scutulata*), and oligochaete worms (Stekoll et al. 1996; Highsmith et al. 1996; van Tamelan and Stekoll 1996). Abundance of sediment infauna and clam densities at lower elevations on gravel and mixed sand/gravel beaches also declined (Peterson 2001). Large numbers of dead and moribund clams documented on treated beaches suggest that they may have suffered toxic effects (Lees et al. 1996), but those effects are difficult to separate from the effects of hydraulic washing (see below). Intertidal fish were also affected. In a study conducted across a gradient of exposure and both rocky and coarse-textured habitats, Barber et al. (1995) found declines in density and biomass of fish at oiled sites relative to reference in 1990. The Barber et al. (1995) study does not provide pre-spill data because the first year of sampling was 1990, after the spill.

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[22] Additional toxicity testing has been conducted on eggs and embryos of Pacific herring (*Clupea pallasii*), which are deposited in the lower intertidal and nearshore subtidal areas. These data are discussed below in Section 7.

Direct effects of EVO in the intertidal zone were also expressed in terms of important ecological parameters such as percent cover and biomass of algal canopy, the total number of species present, total abundances of organisms, and overall diversity of biota. Most of the ecological characterization of the intertidal zone was conducted one or two years following the spill. Consequently, successional processes of habitat recolonization by affected species were under way when most of the ecological studies were conducted. Nevertheless, the Shoreline Ecology Program identified alterations in community level indices at low to mid-intertidal elevations of exposed bedrock communities and at low or high intertidal elevations in boulder cobble or pebble/gravel habitats. The Coastal Habitat Injury Assessment Program identified significant alterations in canopy cover or in biomass of rockweed at all elevations in sheltered rocky and coarse-textured beach habitat, and at mid and upper intertidal elevations in exposed rocky and estuarine habitat (Highsmith et al. 1996; Stekoll et al. 1996; Stekoll and Deysher 2000). Diminished algal canopy cover created space for colonization by opportunistic barnacles and led to a cascade of short-term indirect effects (see below).

3.1.2.2 Effects of Cleaning Methods

Within the first few months following the spill, 89 percent (462) of the oiled beaches in PWS were cleaned using a variety of manual, hydraulic, mechanical, chemical, and bioremediation techniques—most often in combination with one another (Mearns 1996; Lees et al. 1996; Driskell et al. 1996). With the exception of high-pressure, hot-water washing, most of the treatments had minimal short-term impacts or were tolerated by the biota that survived the initial oiling. However, the hydraulic techniques involving some combination of hot-water and high-pressure washing were highly destructive, eliminating as much as 90 percent of the biota that survived the initial oiling. High-pressure washing was conducted at 12 percent (61) of the oiled beaches and hot-water washing was conducted at an additional 35 percent (180) of the oiled beach segments in PWS. One exception to this widespread cleaning was in mussel beds, which were generally not hydraulically cleaned because of fear of killing this high-value resource (Peterson 2001).

On fine-grained and mixed sand/gravel beaches, cleaning treatments caused massive disturbance of sediment and transportation of sediment down-slope (Driskell et al. 1996). Following hydraulic cleaning, an immediate reduction in abundance of clam species *Protothaca staminea* and *Saxidomus giganteus* was documented (Driskell et al. 1996). An evaluation of injury to clams is provided below in Section 4.

3.1.2.3 Indirect Effects of Oil and Cleaning

Effects of EVOS and cleanup efforts had short-term cascading effects in intertidal communities, particularly when species key to the organizational structure of this community were affected. For example, loss of the *Fucus* canopy meant a loss of protection provided by this alga from predation, desiccation, and abrasion to many other intertidal community members, including molluscs, crustaceans, and *Fucus* germlings, slowing recovery for both *Fucus* and its associated community (Highsmith et al. 1996).

EVOS, and the cleanup efforts following the spill, created a large-scale disturbance that reset ecological succession[23] in many areas of the intertidal zone. Mass mortality of established algal and invertebrate communities freed up space on the rocky substrate, a limiting resource in many intertidal communities. These clearings were quickly taken advantage of by early colonizers and species released from predation pressures (e.g., those with planktonic larvae). An initial bloom of ephemeral green algae was documented in the upper intertidal zone, probably facilitated by the loss of grazing molluscs that would normally limit the abundance of these algae (Stekoll et al. 1996). The upper-intertidal barnacle *Chthamalus dalli* was found much lower in the intertidal at oiled sites than unoiled. The increase in *C. dalli* in the mid-intertidal zone probably resulted from at least two spill-associated events: this species took advantage of lowered spatial competition from other barnacle species in denuded areas, and there was a reduction in mortality resulting from a decline in predatory molluscs (Highsmith et al. 1996). *F. gardneri* was found to take advantage of uncolonized substrates in the lower intertidal at oiled sites, outcompeting establishment of some of the annual algae that were removed as a result of oiling. *F. gardneri* establishment may have prevented the immediate recolonization of some annual algae, such as *Alaria*, that are normally common in the lower intertidal zone (Highsmith et al. 1996).

Increase in available space in the intertidal zone also favored species that could themselves quickly disperse or whose young were fast dispersers. The checkered periwinkle (*L. scutulata*), a strong disperser with planktonic larvae, was found in higher densities in EVOS-affected areas, while the Sitka periwinkle (*L. sitkana*), a poor disperser with crawl-away larvae, decreased at oiled sites (de Vogelaere and Foster 1994). Evidence that periwinkles with planktonic larval stages can rapidly recolonize was similarly seen after the *Torrey Canyon* spill, where the common periwinkle, *L. littorea,* which like *L.* scutulata has a planktonic larval stage, was one of the first littorines to recolonize oiled shorelines (Southward and Southward 1978).

3.1.3 Current Status of Injury and Recovery Classification

The 2002 status assessment (EVOS Trustee Council 2002b) states that "intertidal communities will have recovered when such important species as *Fucus* have been reestablished at sheltered rocky sites, the differences in community composition and organism abundance on oiled and unoiled shorelines are no longer apparent after taking into account geographic differences, and the intertidal and nearshore habitats provide adequate, uncontaminated food supplies for top predators."

The 2002 status assessment concludes that there has been substantial progress towards recovery, but that it is incomplete because of the continued presence of residual oil in sediments and on exposed surfaces (e.g., beaches) in the intertidal zone, the lack of full recovery of some soft-sediment (mixed gravel-sand, sand, and sand-silt) intertidal invertebrates (e.g., clams), and the role of oil in initiating *Fucus* population instability (EVOS Trustee Council 2002b).

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[23] Ecological succession is a natural process whereby animal and plant communities replace each other over time in response to changing environmental conditions.

These statements indicate that recovery of the intertidal community is partially contingent on the absence of lingering oil in sediments and on surfaces and the injury and recovery status of specific populations (clams) of intertidal organisms within the larger community. These data and other evidence for injury and recovery are reviewed and evaluated in the following sections. Clams and mussels are independently classified as injured resources. Consequently, injury status and recovery of clams and mussels are addressed in separate sections of this report (Sections 4 and 5, respectively).

3.2 SPATIAL AND TEMPORAL CHANGES IN INTERTIDAL POPULATIONS

The majority of research published in the mid-1990s suggests that, at a regional level, population trajectories of affected species and intertidal communities at oiled but untreated sites in PWS had converged or were close to convergence with patterns of abundance and distribution in unoiled reference sites (Barber et al. 1995; Driskell et al. 1996; Highsmith et al. 1996). However, recovery trends of populations of epibenthic biota and infauna at oiled and hydraulically treated beaches lagged that of oiled but untreated beaches.

The impacts of cleaning on intertidal epibenthos persisted in the first few years following treatment (Houghton et al. 1996; de Vogelaere and Foster 1994). Hydraulic shoreline treatment exacerbated reductions in abundance and biomass of intertidal communities caused by the initial oiling; delaying recovery relative to untreated shores (Houghton et al. 1996). Some organisms in the middle intertidal zone appeared to recover particularly slowly from cleaning; for example, the red algal turf, including coralline algae, was slower to re-grow than the *Fucus* canopy at some sites subjected to hot-water washing (Peterson 2001), although this recovery may have been impeded by increased competition from *Fucus* as well (Driskell et al. 2001).

3.3 LINKAGE BETWEEN EVOS AND THE CURRENT CONDITION OF INTERTIDAL POPULATIONS IN PWS

This section assesses the potential for ongoing injury to the intertidal community based on the current nature and extent of lingering EVO and the likelihood of exposure to a degree that could cause adverse effects and injury. The current condition of the intertidal community can potentially derive from one or more of the following:

- Residual effects from the original spill
- Ongoing exposure to lingering oil
- Other natural or anthropogenic factors that can influence the population.

The relative importance of these different factors is assessed based on the intertidal community ecology and the inherent ability of these communities to recover from the initial impacts of the spill, the likelihood that the intertidal biota could be exposed to lingering EVO to a degree that could cause adverse effects and injury, and the nature and magnitude of other factors that could affect the condition of the intertidal communities.

3.3.1 Residual Effects from the Original Spill

A variety of long-term indirect effects of trophic and interaction cascades were identified as a result of the initial spill and cleanup activities (Peterson et al. 2003). Indirect effects within the intertidal community were driven primarily by eradication of the *Fucus* canopy and most of the attached fauna, particularly in high-pressure and hot-water treated areas, which initiated an ecological succession.

EVOS initiated instability in populations of the abundant and widespread alga *F. gardneri*. Regrowth of this alga after the spill was followed 4–5 years later by a subsequent mass die-off of this species. Extensive denuding of the shoreline after the spill led to the establishment of large stands of *F. gardneri* that were all of a similar age class. The concurrent senescence of these single-aged stands is thought to be the reason for massive mortality of *Fucus* observed in 1994 and 1995 (Paine et al. 1996; Driskell et al. 2001). Such cycles could continue for several generations until a broader age-class distribution is achieved that would exhibit mixed-age, as opposed to evenly distributed age, patterns of cover.

Diversity, total numbers of species, and abundances of major infaunal groups (bivalves, polychaetes, gastropods) at hydraulically treated beaches significantly lagged those at untreated beaches and reference areas (Peterson 2001). This difference in recovery trajectory between untreated oiled sites and treated oiled sites suggests that the loss of fine sediments from hydraulic washing may play a key role in delaying recovery at treated sites (Peterson 2001). Nevertheless, with the exception of infaunal bivalve populations, intertidal community monitoring through 1997 at oiled-and-untreated, oiled-and-treated, and unoiled sites suggests that intertidal epibiota and infauna have recovered by 1994 (Skalski et al. 2001). Effects of the initial spill and beach treatment effects in clam populations have been monitored in recent years and are discussed below in the context of injury to clam resources.

3.3.2 Current Exposure to Lingering Oil

The distribution of lingering oil in the intertidal zone is described above in Section 2, "Lingering Oil." Information indicating ongoing potential exposure of intertidal communities to lingering oil draws upon horizontal and vertical patterns of their respective distributions.

Post-spill studies of lingering oil indicate that it persists at depth (25–50 cm)[24] primarily in cobble/boulder, coarse-textured beach, or mussel bed habitats with sufficient armoring to protect sequestered oil from weathering by processes of physical or biological turbation (Hayes and Michel 1999; Irvine et al. 2002). Two studies of the distribution of lingering oil in the intertidal zone of PWS were conducted in 2001 and 2003 (see Section 2 for detailed discussion). In 2001, Short et al. (2004a) surveyed a random selection of beaches in PWS that had been historically oiled for the presence of lingering oil in surface or subsurface sediments at intertidal elevations ranging from +1.8 to +4.8 m MLLW. Short et al. (2004a) indicate that the most heavily oiled

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[24] Depths to which subsurface oil has been sampled by other researchers (10 cm by Page et al. [2002a] to 0.5 m by Short et al. [2004a]) have not provided sufficient information about vertical profiling of remaining oil to assess its depth distribution.

beach segments are in sheltered embayments, but otherwise distinguish the distribution of lingering oil by elevation rather than by habitat type. The 2001 study results show that lingering oil is present in the middle to upper intertidal zone on approximately 60 percent of the beach segments surveyed. However, the results also indicate that lingering oil has a limited patchy distribution on beaches where it is found. They show that patches of lingering oil are present in the intertidal zone above +1.8 m MLLW, occurring on the surface most frequently at elevations of +2.8 to +3.8 m MLLW, and in the subsurface at elevations of +1.8 to +2.8 m MLLW (Short et al. 2004a). Opportunistic observations indicate that lingering oil also occurs below +1.8 m MLLW, but do not indicate the vertical extent of its distribution below this elevation (Short et al. 2004a).

NOAA returned to PWS in 2003 to conduct follow-on sampling in some of the most heavily oiled beaches characterized in the 2001 study (Short et al. 2006). The 2003 investigation covered the full range of tidal excursions and assessed the presence of lingering oil in surface or subsurface sediments at intertidal elevations ranging from –0.2 to +4.8 m MLLW. The results of the study at northern Knight Island show that lingering oil is present throughout the intertidal zone , but occurs at less than half (44 percent) of the beaches sampled and has a distribution limited to less than 0.5 percent of the beach surface. Beaches sampled in the lower intertidal (0 to +1.8 m) accounted for approximately 36 percent of the oiling identified across all tidal elevations in 2003.

In each of the major investigations of intertidal communities, sampling was stratified among low, middle, and high tidal elevations within the range of +0 to +3.5 m MLLW (Figure 3-1) (Page et al. 1995; Highsmith et al. 1996; Houghton et al. 1996). These studies were conducted several years before the Short et al. (2004a, 2006) surveys of lingering oil in 2001 and 2003. Nevertheless, the patterns of recolonization and vertical zonation of intertidal communities observed in these studies clearly overlap with the distribution of lingering oil where it occurs in the intertidal zone. Because the intertidal community is ubiquitous throughout the various shoreline habitats of PWS, only a small fraction of these communities coincides with lingering oil. For example, assuming that the beach habitat itself defines the intertidal community in the broadest sense, then the Short et al. (2006) results for northern Knight Island indicate that 1 percent or less[25] of the community would overlap with lingering oil depending on the embayment and elevation.

The Short et al. (2004a, 2006) studies of lingering oil focused exclusively on coarse beach habitat that could be excavated to assess lingering oil in the surface (0–5 cm) substrate as well as in the subsurface (5–50 cm) layer. However, almost half (49 percent) of the proposed sample locations in northern Knight Island were on impervious surfaces (e.g., bedrock cliffs or platforms) that could not be excavated. Lingering oil on the surface at these locations was not reported. Consequently, the degree to which surficial oil on these impervious surfaces overlaps with the distribution of the intertidal community adapted to hard-bottom habitat is not known.

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[25] The maximum probability of encountering either surface or subsurface oil is 0.0112 for the upper intertidal in Herring Bay and less than 0.006 for the remaining bays and locations (Short et al. 2006).

However, to the extent that lingering oil is present on hard-bottom habitat, it unlikely to exist in a form that is readily bioavailable (see Section 2).

Current exposure to the intertidal community in coarse, permeable beach habitat depends on the vertical distribution of lingering oil in beach sediments, and on its bioaccessibility and bioavailability. Subsurface lingering EVO is not directly accessible to surface-dwelling intertidal organisms, including snails, crabs, and algae, which dwell almost entirely on the surface of the intertidal substrate, unless these organisms enter the subsurface, for example, to feed, or if there is a mechanism by which it is remobilized to the surface. If the depths reported by Hayes and Michel (1999) described above are predictive of the depths of remaining oil across PWS, it is unlikely that the majority of infauna, most of which occupies the top 10–15 cm of the substrate, will directly contact remaining oil, and exposure will instead depend on other transport mechanisms. Possible transport mechanisms include physical disturbances, such as storms, which remove covering sediments to expose the oil; hydraulic gradients that move subsurface EVO to the surface; bioturbation by foraging vertebrate predators; and bioturbation by deep colonizing infauna (Rice et al. 2003; Shigenaka 1997; Shigenaka and Henry 1995). That such transport mechanisms exist seems evident from studies of bioaccumulation. Bioaccumulation in intertidal organisms or of chemical uptake using SPMDs deployed near oiled areas have demonstrated the presence of bioavailable lingering oil in surface or shallow subsurface intertidal sediments. Whether the bioavailable fraction is accumulated to levels sufficient to cause toxicity or impair intertidal community function is discussed below in Section 3.3.3.2.

3.3.3 Toxicity of Lingering Oil to Intertidal Communities

3.3.3.1 Acute and Chronic Toxicity to Individuals

Past studies of acute toxicity have focused on tests[26] with representative benthic organisms. Results of these tests indicated that toxicity associated with oiled habitat was initially present, but diminished within a few years following the spill.

Short et al.'s (2004a) characterization of lingering oil in previously moderate to highly oiled locations suggests that sediment exposure and toxicity may be prevalent in discrete patches at these locations. This possibility was evaluated by Day (2005), who sampled five pairs of intertidal stations in the vicinity of lingering oil patches, each composed of one oiled and one nearby non-oiled station. These locations were evaluated for potential ecological impacts due to the lingering oil based on benthic community structure (see below) and sediment toxicity. Sediments were not found to be toxic in the larval mussel bioassay using *Mytilus galloprovincialis*. Significant toxicity was noted in the 28-day survival and growth bioassay using the amphipod *Leptocheirus plumulosus*. None of the test amphipods survived exposure to sediments from the oiled sites, and amphipod survival ranged from 37 to 84 percent in sediments from the unoiled sites. Overall, amphipod survival from both oiled and unoiled locations was negatively correlated with sediment PAH concentrations. However, these results are

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[26] More sensitive chronic toxicity tests using herring eggs and embryos have been developed in recent years and are discussed below in Section 7.

complicated by additional stress that coarse-grained sediments may have imposed on the test organisms. USEPA (2001) recommends testing *L. plumulosus* in sediments that have greater than 5 percent silt-clay content. The substrate in PWS is exceptionally coarse and the silt-clay content expressed as percent fines was below this threshold at 8 of the 10 test sites. Consequently, low survival in some of the tests may be partially attributed to the coarse-grained nature of the sediments. Similar observations of grain-size effects in amphipod tests were made by Boehm et al. (1995). Nevertheless, the correlation with sediment PAH supports the idea that lingering oil may be a factor explaining the lack of amphipod survival at the oiled sites.

3.3.3.2 Bioavailability and Bioaccessibility of Lingering EVO

The majority of lingering EVO is subsurface, and subsurface EVO is generally less weathered and more bioavailable than lingering oil in surface sediments (see Section 2). The locations and depth of lingering oil indicate that it is unlikely that either infauna or epibiota will directly contact remaining oil, and exposure, if it occurs, will depend on other transport mechanisms. Such exposure has been investigated using sensitive biomarker techniques with intertidal fishes and SPMDs.

In 1990, studies of an intertidal fish, the high cockscomb prickleback (*Anoplarchus purpurescens),* in PWS showed elevated biomarkers (CYP1A) indicative of PAH exposure at EVOS-affected sites relative to that at unoiled sites (Woodin et al. 1997). In 1996, 1998, and 1999 similar follow-up studies of the biomarker were conducted with masked greenling (*Hexagrammos octagrammus*) collected subtidally and crescent gunnel (*Pholis laeta*) collected subtidally and intertidally (Jewett et al. 2002). In 1998, CYP1A induction in the intertidally collected crescent gunnel was elevated at Herring Bay, but not at Bay of Isles in comparison with two unoiled control sites at Mummy Bay and Port Chalmers. In 1999, CYP1A induction was studied in crescent gunnel from eight oiled sites and a single control site (Barnes Cove). This study showed a wide range in CYP1A induction in which the control was statistically similar to all of the oiled sites. The only significant differences were among two locations with the highest CYP1A levels (oiled sites at Sleepy Bay and Herring Bay) and the lowest CYP1A levels (oiled site at Disk Island). The 1999 study results were confirmed with an independent measure[27] of CYP1A induction in which there were no statistically detectable differences among oiled or unoiled sites. Collectively, these results indicate that the CYP1A biomarker in intertidal fish was detectable in many oiled locations throughout the spill area, but Herring Bay in 1998 was the only location where it was distinguishable from unoiled locations.

Elevated levels of CYP1A in birds that forage in the intertidal zone also suggest that lingering oil is bioavailable. The most recent data on CYP1A (Ballachey et al. 2006) in biological resources suggests that harlequin duck, black oystercatcher, and Barrow's goldeneye, all of which forage on intertidal resources, continued to demonstrate exposure in oiled areas in 2004 and 2005 based on CYP1A levels that that are elevated above background.

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[27] The 1996 and 1998 studies used immunohistochemical (IHC) analysis to assess CYP1A induction, and the 1999 study used IHC and another method based on the enzyme ethoxyresorufin-*O*-deethylase (EROD).

Bioaccessibility and bioavailability of lingering oil was also demonstrated in experiments conducted by Shigenaka and Henry (1995) three years following EVOS in which TPAH concentrations were measured in SPMDs, sediments, water and caged mussels placed on a previously heavily oiled beach on Smith Island and on an unoiled to lightly oiled beach also on Smith Island. Results of this study showed that TPAH concentrations were significantly correlated among the SPMDs, mussels, and sediments, further indicating the bioavailability of lingering oil to surface-dwelling organisms.

Recent research conducted in 2004 using SPMDs placed in the intertidal zone, indicates that where lingering oil is present, it is still bioavailable to intertidal organisms, exists in a bioactive form that is capable of inducing CYP1A in fish, and is distinguishable from stressed reference locations that have sediments contaminated by other non-EVO-related sources of petroleum hydrocarbons (Springman et al. 2005; Short et al. 2005).

3.3.3.3 Effects on Reproduction and Other Population Parameters

Day (2005) evaluated benthic community structure in a focused study of lingering oil patches. Classification analysis indicated that community composition over all eight stations was relatively similar. Study results indicate that residual oil sequestered in intertidal sediments is not causing substantial community-wide effects though populations of sensitive species such as amphipods may be impaired.

3.3.4 Indirect Effects of Lingering Oil

A variety of long-term indirect effects of trophic and interaction cascades were identified as a result of the initial spill and cleanup activities (Peterson et al. 2003). Indirect effects within the intertidal community were driven primarily by eradication of the *Fucus* canopy and most of the attached fauna, particularly in high-pressure and hot-water treated areas, which initiated an ecological succession. Intertidal community monitoring through 1997 at oiled-and-untreated, oiled-and-treated, and unoiled sites suggests that intertidal epibiota and infauna have recovered by 1994 (Skalski et al. 2001). Given the patchy and limited distribution of lingering oil and the similarity in intertidal community structure between beaches with lingering oil and reference beaches (Day 2005 and Section 3.3.2 above), there is little conclusive evidence to suggest that lingering oil is influencing community succession or mature reestablished benthic communities.

3.3.5 Other Factors Influencing Current Injury or Rate of Recovery or Population Variability

Other factors affecting intertidal communities that may confound an understanding of lingering oil effects include natural variability associated with predation, natural disturbance, and recruitment. Reduction in predator numbers can result in changes in predation rates and thereby affect community composition, abundance, and size distributions (Fukuyama et al. 2000). Natural disturbances such as storms may have important localized effects in community structure and function. Localized disturbances play important roles in establishing community structural heterogeneity at the patch scale when sessile communities are cleared from the substrate, restarting ecological succession. Long-term variation in rocky intertidal communities is not well understood (Foster et al. 1990). These processes increase variability in population structure, distribution, and abundance that are a part of a naturally functioning intertidal community, and can increase the ecological "noise," or variation in data, when examining trajectories of intertidal community recovery. Finally, there is significant interannual variation in recruitment of intertidal invertebrates based on natural conditions, including temperature, nutrient levels, predation rates on plankton, and currents that are difficult to measure and predict. Considerable year-to-year variation in recruitment of some intertidal community members, such as algae, can complicate analysis of population trajectories following a large-scale disturbance (Foster et al. 1990).

3.4 ACTIONS TAKEN OR PLANNED

Injury and restoration projects initiated by the EVOS Trustee Council (2003) for intertidal communities included multiple monitoring projects that focused on specific communities (e.g., eelgrass beds, populations of clams and Barrow's goldeneye), development of monitoring methods and options, and preparation of manuscripts based on past studies. Additional studies investigate ecological factors that influence intertidal and subtidal communities, monitor the persistence of oil along the Kenai and Katmai coasts, measure the extent of oil remaining in PWS, and map intertidal and subtidal shores in the coastal zone. Finally, habitat acquisition and restoration and reestablishment of clam populations have been initiated at selected locations.

3.5 SUMMARY AND CONCLUSIONS

3.5.1 Summary

Approximately 40–45 percent of the estimated 10.8 million gallons of EVO washed ashore in the intertidal zone of PWS. Aerial surveys by the Alaska Department of Natural Resources showed that light to heavy deposits of EVO washed ashore on approximately 24 percent (446 km) of the 1,891 km of the PWS shoreline. The spill area included all types of intertidal habitats, including exposed and sheltered rocky shores; boulder, cobble, and gravel beaches; mixed sand-gravel and fine sand beaches; exposed and sheltered tidal flats; and estuaries. Dominant species of algae and invertebrates that were directly affected by the spill were common rockweed, speckled limpet, several barnacle species, blue mussels, periwinkles, and oligochaete worms. Abundance of sediment infauna and clam densities at lower elevations on gravel and mixed sand/gravel beaches declined and intertidal fish were also affected.

Within the first few months following the spill, 89 percent (462) of the oiled beaches in PWS were cleaned using a variety of manual, hydraulic, mechanical, chemical, and bioremediation techniques—most often in combination with one another. With the exception of high-pressure, hot-water washing, most of the treatments had minimal short-term impacts or were tolerated by the biota that survived the initial oiling. However, the hydraulic techniques involving some combination of hot-water and high-pressure washing were highly destructive, eliminating as much as 90 percent of the biota that survived the initial oiling.

The current condition of the intertidal community can potentially derive from residual effects from the original spill and cleanup methods, ongoing exposure to lingering oil, or other natural or anthropogenic stressors that can influence the population. The spatial and temporal distribution of lingering EVO suggests that ongoing exposure to lingering oil and related impacts to intertidal communities are localized, limited, and patchy (see Section 2).

The majority of research published in the mid-1990s suggests that, at a regional level, population trajectories of affected species and intertidal communities at oiled but untreated sites in PWS had converged or were close to convergence with patterns of abundance and distribution in unoiled reference sites (Barber et al. 1995; Driskell et al. 1996; Highsmith et al. 1996). However, recovery trends of populations of epibenthic biota and infauna at oiled beaches treated with high-pressure, hot-water washing lagged that of oiled but untreated beaches. Diversity, total numbers of species, and abundances of major infaunal groups (bivalves, polychaetes, gastropods) at hydraulically treated beaches significantly lagged those at untreated beaches and reference areas even by 1998 (Driskell et al. 1996; Peterson 2001). This difference in recovery trajectory between untreated and treated oiled sites suggests that the loss of fine sediments from hydraulic washing may delay recovery at treated sites (Peterson 2001).

The ability of intertidal communities to provide adequate, uncontaminated food supplies for top predators is another important consideration when assessing the recovery of the intertidal zone. Tissue data on PAHs in mussels (Section 5) indicates that bioaccumulation is no longer a concern for this resource. However, the most recent data on CYP1A in some bird species that forage in the intertidal zone (Ballachey et al. 2006) suggests that they may be exposed to PAHs during feeding. Harlequin duck, black oystercatcher, and Barrow’s goldeneye, all of which forage on intertidal resources, continued to demonstrate exposure in 2004 and 2005 based on ongoing elevated CYP1A expression that is distinguishable from background rates.

3.5.2 Conclusions

Intertidal communities will have recovered when such important species as *Fucus* have been reestablished at sheltered rocky sites, the differences in community composition and organism abundance on oiled and unoiled shorelines are no longer apparent after taking into account geographic differences, and the intertidal and nearshore habitats provide adequate, uncontaminated food supplies for top predators (EVOS Trustee Council 2002b). There are important uncertainties that must be recognized in assessing recovery status, particularly the lack of monitoring of intertidal communities within the last decade. Additional sources of uncertainty include discrete patches of lingering oil that provide localized opportunities for intertidal faunal exposure; natural variability in recruitment and population dynamics of intertidal biota; and the continuation through at least the mid-1990s of differences in epibenthos and infauna between treated and untreated oiled sites. However, based on the weight of evidence for PWS, including the convergence of population trajectories of epibiota and infauna at untreated oiled sites and unoiled reference sites, the similarity in CYP1A expression between intertidal fish in most unoiled and oiled sites, and the localized, patchy distribution of lingering oil, intertidal communities appear to have recovered.

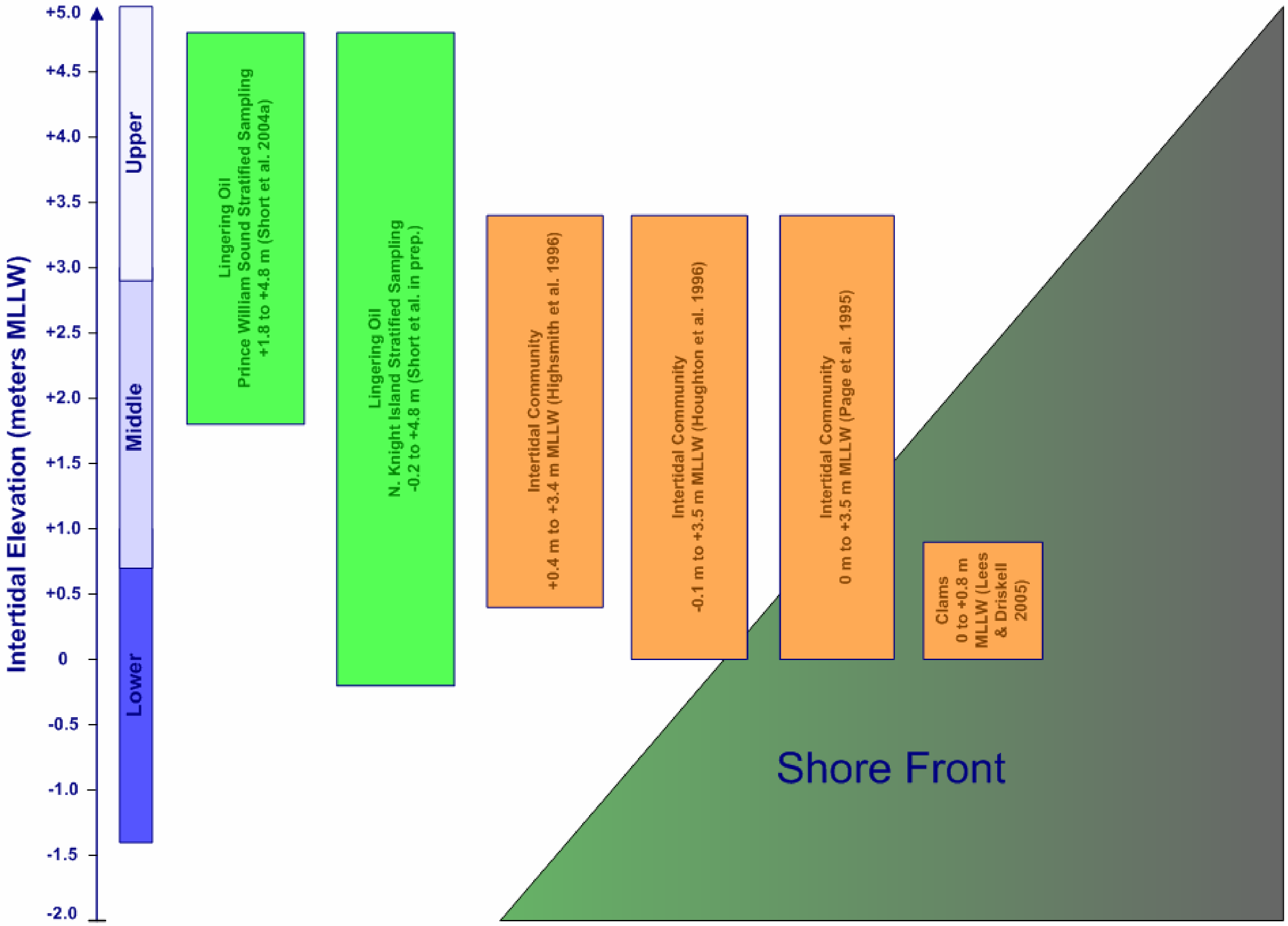
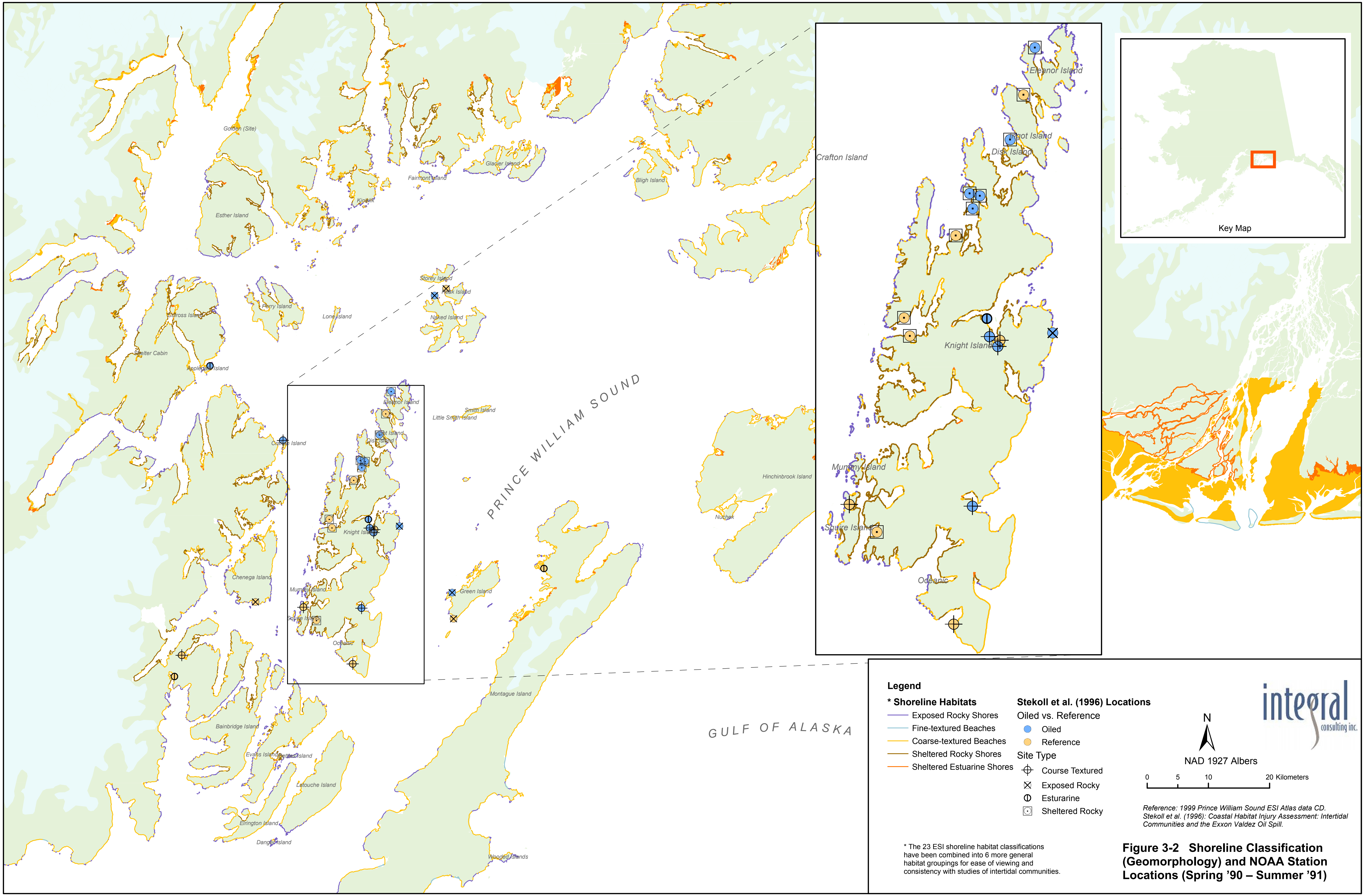


Figure 3-1. Sampling Elevations for Studies of Lingering Oil, Intertidal Communities, and Clams.



Legend

*** Shoreline Habitats**

- Exposed Rocky Shores
- Fine-textured Beaches
- Coarse-textured Beaches
- Sheltered Rocky Shores
- Sheltered Estuarine Shores

Stekoll et al. (1996) Locations

Oiled vs. Reference

- ● Oiled
- ● Reference

Site Type

- ⊕ Course Textured
- ☒ Exposed Rocky
- ⦶ Esturarine
- ⊡ Sheltered Rocky

Reference: 1999 Prince William Sound ESI Atlas data CD.
Stekoll et al. (1996): Coastal Habitat Injury Assessment: Intertidal Communities and the Exxon Valdez Oil Spill.

* The 23 ESI shoreline habitat classifications have been combined into 6 more general habitat groupings for ease of viewing and consistency with studies of intertidal communities.

Figure 3-2 Shoreline Classification (Geomorphology) and NOAA Station Locations (Spring ’90 – Summer ’91)

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4. CLAMS

Clams in PWS are important as prey for sea otters, seabirds, and many other predators, as well as subsistence use by native populations and recreational harvest. Within days of the EVOS in March 1989, clams on beaches along the shoreline of PWS were exposed to the spreading oil slick. Clams were killed by oiling and the subsequent cleanup activities.

The recovery objective for clams was defined as a return to populations and productivity levels that would have occurred in the absence of the spill. The main restoration strategy relied on natural recovery with monitoring and measures to protect injured clam beds (e.g., maintain water quality, reduce disturbance). In 2002, the EVOS Trustee Council (2002b) reviewed the status of clams in PWS and concluded that clams were continuing to recover, but were not yet fully recovered.

4.1 BACKGROUND

This section describes the natural history of clams, initial impacts of EVOS, and the current status of injury and recovery.

4.1.1 Natural History of Clams

This assessment of clam resources in PWS focuses on two species: the littleneck clam (*Protothaca staminea)* and the butter clam (*Saxidomus giganteus*). These species are emphasized for several reasons. First, they are the most abundant and widely distributed large clam species in PWS. Second, these clam species are both important for subsistence use and are popular species for recreational harvest in PWS. Third, littleneck and butter clams are major prey items for numerous invertebrate and vertebrate species, including the sea otter and sea ducks. Finally, for the above reasons, these species have been the focus of most research into the effects of oiling on clams in PWS, and in the case of the littleneck clam, studies in other areas. Key elements of the natural history of littleneck and butter clams are summarized in the following sections.

4.1.1.1 Life Cycle

Embryonic development and early life history stages are similar for littleneck clams and butter clams. For each species, sexes are separate, and eggs and sperm are released into the water where mass fertilization takes place. The timing of spawning depends on temperature. In PWS, the littleneck clam spawning period may last for up to 4 months and generally occurs between July and September (Chew and Ma 1987). Butter clam spawning also occurs in summer. After fertilization, the eggs divide rapidly, develop into trochophore larvae in about 12 hours, and then reach the veliger larval stage within the following 24 hours (Chew and Ma 1987). The eggs and larvae are planktonic, drifting with local currents and feeding on phytoplankton in the water column. After about 3 weeks, the littleneck clam larvae settle onto the substrate where they change from larvae into young clams known as “spat.” Development is similar for butter clams, with planktonic larvae that settle to the bottom 4–6 weeks after fertilization.

Young littleneck clam spat can move about; butter clam larvae do not have this ability and do not move once they settle (Chew and Ma 1987). Recruitment (i.e., addition of new individuals to a population, typically by reproduction or immigration) for all clam species is highly variable, and there is significant interannual variation in littleneck clam recruitment in PWS (Paul and Feder 1973). Environmental factors affecting recruitment include temperature, food supply, predation, and favorable conditions for settlement (Paul and Feder 1973).

Littleneck clams reach maturity at 2–3 years (Chew and Ma 1987). Some littleneck clams spawn in their second year but most spawn at the end of the third year, depending on size and maturity (Chew and Ma 1987; Ricketts and Calvin 1968). Butter clam spawning typically first occurs at the end of the third year (Ricketts and Calvin 1968). In PWS, the maximum life span is 15 to 16 years for littleneck clams, and may be up to 20 years for butter clams (Morris et al. 1980). Adult clams are sedentary, although they may re-burrow.

Littleneck clam growth rates in PWS are relatively slow (approximately 2–5 mm/year) when compared with other populations of this species in British Columbia (Paul and Feder 1973; Chew and Ma 1978), but vary both spatially and temporally. Factors influencing growth rates include food supply and location (including currents, protection from storms, substrate, and tidal level). Some researchers observed higher growth rates for littleneck clams on beaches with strong tidal currents compared to quiet bays and suggested that increased currents may increase food supply (Chew and Ma 1987). Others report slower clam growth rates in exposed areas compared to protected sites (McCrae 1995).

4.1.1.2 Abundance and Distribution

Littleneck and butter clams are widely distributed in PWS. They are most abundant in the lower intertidal to subtidal zones of sheltered beaches with gravel, sand, or mixed substrate. The preferred substrate for littleneck clams is coarse sand or fine gravel with mud (Paul and Feder 1973; Chew and Ma 1987). Butter clams are observed in a variety of substrates, preferring porous mixtures of gravel, broken shells, mud, and sand (Kozloff 1976; Morris et al. 1980).

Abundance and distribution of clams are strongly influenced by recruitment processes. Larval dispersion, which is dependent on environmental conditions including current patterns and strength, and larval settling patterns, establish initial patterns of sessile clam abundance and distribution that are subsequently shaped by factors including predation and abiotic stressors. While butter clam spat are not mobile, the young littleneck clam can to some extent migrate laterally by pedal movements and can therefore move around on the substrate to locate a suitable surface after settling (Chew and Ma 1987).

Adult littleneck and butter clams are not mobile and so cannot relocate in response to changes in environmental conditions. In Simpson Bay of PWS, butter clams were found in the lowest part of the middle intertidal zone and in the lower intertidal zone usually between –0.9 and +1.2 m MLLW, and littleneck clams were found between –0.6 and 1.5 m MLLW (Chew and Ma

1987).[28] Maximum butter clam density in the same study was observed at the 0-ft tidal level, and maximum littleneck clam was observed at the 1 ft (+0.3 m) MLLW tide level (Chew and Ma 1987). In Galena Bay of PWS, few littleneck clams were found above +0.43 m (+1.4 ft) MLLW (Paul and Feder 1973). Butter clams are also abundant in the lower intertidal zone, and while they may be co-located with littleneck clams, they are often found at somewhat lower elevations (Morris et al. 1980; Chu and Ma 1987).

The vertical distributions of littleneck and butter clams within the sediment can also overlap, although butter clams are generally found somewhat deeper. Adult littleneck clams are shallow burrowers and are commonly within 15 cm of the surface; they may burrow to a maximum depth of 20 cm (Chew and Ma 1987). In Galena Bay of PWS, most littleneck clams were found within 3.8 cm of the surface (Paul and Feder 1973). Butter clams can burrow as much as 25 to 30 cm deep, although they are more frequently found closer to the surface (Kozloff 1976). Adults of both species remain in the same burrow for life.

The Alaska earthquake of 1964 and the uplift of the shoreline in PWS had a major impact on the distribution of intertidal resources, including clams (Hanna 1971; Baxter 1971). At some locations, the magnitude of the uplift was great enough to elevate portions of the intertidal habitat to levels that were above the water depth needed to sustain the clam communities.

4.1.1.3 Feeding

Adult and larval littleneck clams are filter feeders, feeding on plankton and suspended organic matter in the water. There is no evidence for food selectivity.

4.1.1.4 Natural Mortality and Predation

Highest clam mortality is observed in larval stages and during the first year. Early survival depends on temperature, food supply, predation, currents, and substrate (McCrae 1995). In addition, clams are prey for many predators, including crab, octopus, and sea ducks. In PWS, clams are a preferred food item of sea otters (Fukuyama 2000; Van Blaricom et al. 2002; Dean et al. 2002) and may account for as much as 60–70 percent of the otter diet (Bodkin and Ballachey 1997).

Human use can contribute to clam mortality in some intertidal areas. Both butter and littleneck clams are important for subsistence use. In PWS, both species are harvested recreationally year-round, although most clamming occurs from April through September (ADFG 2006). There is no limit on the number of clams that can be collected, but they must be harvested by hand or using a rake, shovel, or other manually operated tool.

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[28] Based on elevations at Cordova, Alaska, approximate intertidal zone elevations are lower intertidal –1.45 to +0.75 m, middle intertidal +0.75 to +2.95 m, and upper intertidal +2.95 to +5 m.

4.1.2 Initial Impact of EVOS

In the first days to weeks following the initial spill, EVO was distributed both in open water and along the shoreline of PWS. The Trustee Council (2002b) reported that "the magnitude of immediate impacts on clam populations varied with the species of clam, degree of oiling, and location." Some clams were probably killed by oil and the cleanup activities (EVOS Trustee Council 2002b).

Cleanup operations immediately following the spill included pressure washing of intertidal areas using cold or hot water, as well as manual and mechanical removal of oil and tar, and manual removal of oiled sediments. Hot water pressure washing, in particular, caused thermal stress, oil dispersion, animal displacement and burial, and the transport of sediments from the upper intertidal into the lower intertidal zone. The results of this treatment included reductions in bivalve abundance observed on treated beaches 1–3 years after EVOS (Peterson 2001). Although recruitment and abundance were similar between unoiled and oiled-but-untreated sites, clam abundance was low on oiled-and-treated beaches (Driskell et al. 1996).

Trowbridge et al. (2002) sampled bivalves, including littlenecks and butter clams, at oiled and unoiled sites in PWS, Cook Inlet, the outer Kenai Peninsula, and around Kodiak Island, between 1989 and 1991 to examine hydrocarbon concentrations in sediment and bivalves and their effects, including mortality, growth and histopathological abnormalities in bivalves. In 1989, elevated levels of TPAH were documented in bivalve tissues. By 1990, weathered EVO was tentatively detected[29] in clam tissue at only 3 of 18 sites sampled, at levels ranging between 630 and 1,230 ng/g. Histopathological results were not significantly different between oiled and unoiled sites in either year of sampling. However, declines in growth rates of littleneck clams at oiled sites were observed in 1990 and 1991 (discussed below).

4.1.3 Current Status of Injury and Recovery Classification

The EVOS Trustee Council (EVOS Trustee Council 2002b) defines the recovery goal for clams as "a return to conditions that would have existed had the spill not occurred." The specific recovery objective described by the Trustee Council is to have comparable population and productivity measures (e.g., size and distribution) at oiled and unoiled sites, taking into account geographic differences (EVOS Trustee Council 2002b).

In 2002, the Trustee Council concluded that clam populations continue to recover, but are not yet fully recovered from the effects of the oil spill. This conclusion was based on observed differences in clam populations among unoiled, oiled-and-untreated, and oiled-and-treated-sites, as well as differences between study locations in clams available as prey to sea otters.

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[29] Detected oil was described by the authors as too weathered to confirm identity of the oil, but relative abundances of remaining PAH were consistent with extremely weathered EVO.

4.2 SPATIAL AND TEMPORAL DISTRIBUTION OF CLAM POPULATIONS

Spatial and temporal patterns of impact and recovery of clams within the intertidal zone were investigated as part of the NOAA injury assessment (Driskell et al. 1996; Houghton et al. 1996, 1997; Lees et al. 1996). Following the spill in 1989, much of the shoreline was cleaned by a variety of treatment technologies while other areas were not treated at all. Decisions to not treat some of the oiled shoreline areas were made with the intent of comparing initial impacts and rates of recovery among oiled-and-untreated, oiled-and-treated, and reference locations (see Section 3.1.1.2, "Effects of Cleaning Methods"). With the exception of high-pressure, hot-water washing, most of the treatments had minimal short-term impacts or were tolerated by the biota that survived the initial oiling (Driskell et al. 1996). Infaunal assemblages that were not treated by high-pressure hot-water washing were generally indistinguishable from those at unoiled reference sites by mid-summer 1991. However, bivalves and other infaunal assemblages on beaches treated by high-pressure, hot-water washing were still in the early stages of recovery in 1992. Further monitoring indicated that lower recruitment and abundances of little neck clams (*Protothaca staminea*) persisted at oiled-and-treated beaches in comparison to reference or oiled-and-untreated beaches through 1996 (Houghton et al. 1997). An additional study (Lees and Driskell in prep.) was conducted in 2002 to quantify residual effects of shoreline treatment on clam assemblages in the lower intertidal zone (see Section 4.3.1, "Residual Effects from the Original Spill").

4.3 LINKAGE BETWEEN EVOS AND THE CURRENT CONDITION OF CLAM POPULATIONS IN PWS

This section assesses the potential for ongoing injury to clams based on the current nature and extent of lingering EVO and the likelihood of exposure to a degree that could cause adverse effects and injury. The current condition of clam populations can potentially derive from one or more of the following:

- Residual effects from the original spill
- Ongoing exposure to lingering oil
- Other natural or anthropogenic stressors that can influence clam populations.

The relative importance of these different factors is assessed based on the clam life histories and inherent ability of clam populations to recover from the initial impacts of the spill, the likelihood that their behavior could result in ongoing exposure to lingering EVO to a degree that could cause adverse effects and injury, and the nature and magnitude of other factors that could affect the condition of these populations.

4.3.1 Residual Effects from the Original Spill

Recent monitoring data that compares trends in clam populations in oiled and unoiled areas provides some indication of possible residual effects attributable to the original spill. In 2002, Lees and Driskell (in prep.) studied bivalve assemblages at 23 oiled-and-treated locations and

17 oiled-and-untreated reference sites in PWS[30] to assess residual impacts of cleanup activities on mixed-soft beach habitat and its resident infaunal bivalve community in the lower intertidal zone. Two bivalve assemblages were studied at each location: large bivalves were excavated to 15 cm within a 0.0625 m^2 quadrat and small bivalves were sampled to the same depth with a 0.009 m^2 corer. For each sampling location, physical sediment characteristics, individual species abundances, total bivalve abundances, species richness, and two indices of diversity[31] were determined.

The Lees and Driskell (in prep.) investigation shows that median particle grain size at the oiled-and-treated areas (10.3 mm) was significantly higher than that at the reference locations (6.8 mm). Other sediment habitat characteristics (silt-clay, organic carbon, nitrogen, and carbon/nitrogen ratios) were highly intercorrelated, but nonetheless similar among oiled-and-treated and reference sites. Overall, Lees and Driskell (in prep.) believe that the difference among locations in particle grain size was small and not attributable to shoreline washing. Their analysis indicates that differences in particle grain size (percent fines) are largely determined by hydrodynamic processes as indicated by the degree of beach exposure to open water and wind (fetch).

Significant differences for each kind of assemblage were also identified. For the large bivalve assemblage, abundances of butter clams, abundances of littleneck clams, total abundances, and species richness were higher at reference locations than in oiled-and-treated locations. For the small bivalve assemblage, abundances of butter clams and overall bivalve diversity were higher in reference locations than in the oiled-and-treated locations.

Influence of sediment grain size on the bivalve community is also apparent from strong negative correlations between particle grain size and many of the biological parameters including individual abundances, total abundances, and species richness for both the large and small assemblages. The authors also note that large, long-lived clams in the infaunal assemblage at oiled-and-treated locations were about one-third as abundant as those in reference locations. Similarly, small, short-lived clams in the infaunal assemblage at oiled-and-treated locations were more abundant than those in reference locations. Differences in bivalve assemblages at oiled-and-treated sites in comparison with reference sites shown in their study follow the same pattern that was noted in NOAA’s post-spill investigation.

Lees and Driskell (in prep.) conclude that sediments have recovered from the disturbance associated with high-pressure, hot-water washing, but the bivalve assemblages lag and will not attain full recovery for many years. This conclusion is consistent with the long lifespan of some of the dominant clam species (15 years for littleneck clams and 20 years for butter clams). However, Lees and Driskell (in prep.) also note appropriately that “the results of this study do

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[30] Lees and Driskell also sampled three treated sites and one reference site known from previous NOAA studies, but excluded these locations from their quantitative evaluation of the data. Lees and Driskell (in prep.), represents the only recent data for clam populations in PWS. This report is discussed in greater detail in the report for Restoration Project 060783.

[31] The Shannon-Weiner diversity index (H’) and Simpson’s inverse diversity index.

not provide conclusive evidence that [oiled and] treated sites have not recovered from the effects of the HP-HW [high-pressure hot-water] washing.” This caveat is important because it acknowledges the alternative hypothesis that physical factors that were influential in the trajectory of the spill and the locations at which oil landed may also have defined differences in habitat characteristics (percent fines) that may have been extant prior to the spill.

4.3.2 Current Exposure to Lingering Oil

Exposure of clams to subsurface lingering oil depends on overlap in the distributions of clams and lingering oil. There are three components to clam and oil distribution: physical habitat characteristics (including substrate type and degree of physical disturbance), intertidal elevation, and depth below the surface. Available data on each of these three elements is discussed in the following paragraphs.

The recent study of lingering oil by Short et al. (2004b) was based on the likelihood that most lingering oil occurs in the middle and upper intertidal zone of previously heavily to moderately oiled beaches. Consequently, clam exposure to lingering EVO is limited to beaches that were heavily to moderately oiled in 1989. The locations of habitats conducive to clam beds and locations with potential lingering EVO are shown in Figure 3-2.

Habitat characteristics such as exposure, disturbance frequency, and geomorphology or substrate affect distribution of clams and lingering oil. Both littleneck and butter clams are most commonly found in protected beaches and bays. They are found in a variety of substrates, but prefer coarse sand and gravel beaches. Butter clams typically occur in beaches with a porous mixture of sand, broken shell, and small gravel; littleneck clams prefer coarse sand to firm gravel. Persistent EVO was found in sheltered bays and on beaches with boulder/cobble surface armoring or a thick sediment veneer over bedrock (Short et al. 2004b). Neither boulder/cobbles nor bedrock is the type of substrate preferred by clams.

Overlap in the tidal elevations at which lingering oil and clams are found is also required for exposure. Two studies of the distribution of lingering oil in the intertidal zone were conducted in 2001 (Short et al. 2004b) and 2003 (Short et al. 2006) (see Section 2 for detailed discussion). The 2001 study results show that lingering oil is present in the middle to upper intertidal zone on approximately 60 percent of the beach segments surveyed. However, the results also indicate that lingering oil has a limited patchy distribution on beaches where it is found. The 2003 investigation covered the full range of tidal excursions at three locations on northern Knight Island and assessed the presence of lingering oil in surface or subsurface sediments at intertidal elevations ranging from –0.2 to +4.8 m MLLW. The results of this study show that lingering oil is present throughout the intertidal zone, but occurs at less than half (44 percent) of the beaches sampled, has a distribution limited to less than 0.5 percent of the beach surface, and can be found at tidal elevations ranging from –0.2 to +4.8 m MLLW.

Lower intertidal elevations are preferred by littleneck and butter clams, and they are not found in the upper intertidal zone. Littleneck clams are distributed between –0.9 and +1.2 m MLLW. In PWS, the greatest littleneck clam abundance was found at 0 m MLLW (Chew and Ma 1987). Consequently, although lingering oil is limited and patchy in its distribution, it can overlap with the distribution of littleneck and butter clams in the lower intertidal zone in those areas in which it occurs.

Finally, the depth of lingering oil could also determine if there is potential clam exposure. Juvenile clams are found at the surface, within the upper inch of sediment, and so are unlikely to be exposed to subsurface lingering EVO. However, as the young clams develop and begin burrowing, there is exposure to subsurface sediments that could contain lingering oil. Adult littleneck and butter clams may burrow 8 to 12 in. deep, respectively, and so would encounter any subsurface oil at these depths. Subsurface oil may have a greater effect on clams than surface oil. Littleneck clams exposed to a surface layer of oiled sediments for 1 year did not contain detectable amounts of petroleum components, but clams in sediments mixed with oil did accumulate contaminants (Anderson et al. 1983).

4.3.3 Toxicity of Lingering Oil to Clams

4.3.3.1 Acute and Chronic Toxicity to Individuals

Several field studies examined the effect of EVO toxicity on clam mortality; no laboratory experiments evaluating the effects of EVO on clams have been reported. Trowbridge et al. (2002) reported that even though PAH concentrations were very high in bivalve tissues in PWS, there was no strong evidence that clam tissues were severely affected by EVO based on the presence of lesions or parasites. There was no significant difference in the occurrence of histopathological abnormalities in clams from oiled and control (unoiled) sites in PWS in 1990.

Fukuyama et al. (2000) used reciprocal transplants of clams between unoiled and oiled areas to examine effects on survival, growth rate, and depuration rates. The authors reported slower growth rates and higher mortality of clams transplanted to areas that had been heavily oiled. Mortality of clams from oiled areas was higher than that of clams from clean areas, even among animals moved from the oiled to the clean areas, suggesting residual effects on clams with longer-term exposure to the oil. Significant differences in clam mortality were observed 2 years, but not 1 year, after transplanting clams to PWS beaches with lingering oil, suggesting that perhaps longer term exposure may affect mortality (Fukuyama 2000).

Possible sublethal effects of hydrocarbon exposure include shallower and slower burrowing, reduced siphon activity, and decreased growth rates. Reduced clam growth rates due to oil exposure are the most frequently observed result of oiling (Fukuyama et al. 2000; Trowbridge et al. 2002). Experiments by Roesijadi et al. (1978) using large littleneck clams did not show significant differences in growth rate due to exposure to oiled sediment (887 µg/g), but other experiments with small littleneck clams and PWS field experiments have clearly shown decreased growth associated with hydrocarbon exposure. There were several observations of decreased growth rates in PWS in the years immediately after EVOS. In PWS in 1990 and 1991, growth of littleneck clams at oiled sites was less than at control sites. In transect sampling, clam growth rates in PWS decreased as PAH concentrations increased (Trowbridge et al. 2002). The growth of littleneck clams in PWS increased as the concentration of EVO decreased from 1990 to 1991 (Trowbridge et al. 2002).

While there is evidence that oil exposure in the first few years following EVOS affected clam growth, the potential effect of lingering oil on clam growth or other chronic effects have not been directly examined. PWS clams transplanted from unoiled to oiled areas 5–7 years following EVOS had significantly reduced growth rates, but these rates were still within the range seen in previous studies in PWS (Fukuyama 2000).

4.3.3.2 Bioavailability and Bioaccessibility of Lingering EVO

Clams both accumulate and depurate hydrocarbons. The primary routes for littleneck clams to accumulate hydrocarbons are direct uptake from water ventilated by the animals or ingestion of contaminated food or particulate matter. Filter or suspension feeders like littleneck clams appear to take up hydrocarbons from seawater (and interstitial water) at a slow rate until equilibrium is reached (Roesijadi et al. 1978). Clams transplanted from unoiled to oiled areas in PWS 5–7 years after EVOS showed increased PAH tissue concentrations (Fukuyama et al. 2000).

Although bivalves have limited ability to metabolize hydrocarbons, several studies indicate that clams are able to depurate hydrocarbons to some degree when the hydrocarbon source is removed (Fukuyama 2000). Trowbridge et al. (2002) observed EVO in bivalves collected from various oiled sites in 1989[32] following the spill, but reported a general absence of EVO or weathered EVO in bivalves collected in 1990, possibly indicating some level of depuration.

Based on the above observations, if clams and lingering oil co-occur, it is likely that elevated PAH concentrations may be present in clam tissue. Neff and Gilfillan (2004) investigated PAH concentrations in clams from beaches on Smith Island in PWS that contained subsurface lingering oil. In 2002, clams were collected in the mid-lower intertidal areas below the upper tidal elevations where lingering oil was documented. Clam TPAH ranged from <10 to approximately 400 ng/g dry weight; TPAH in clams from a reference area were <10 to approximately 200 ng/g dry weight (estimated from graph presented by Neff and Gilfillan 2004).

Bioaccessibility and bioavailability of lingering oil was also demonstrated in experiments conducted by Shigenaka and Henry (1995) 3 years following EVOS in which total TPAH concentrations were measured in SPMDs, sediments, water, and caged mussels placed on a previously heavily oiled beach on Smith Island and on an unoiled to lightly oiled beach also on Smith Island. Results of this study showed that TPAH concentrations were significantly correlated among the SPMDs, mussels, and sediments, further indicating the bioavailability of lingering oil to surface-dwelling organisms.

Recent research conducted in 2004 using SPMDs placed in the intertidal zone indicates that where lingering oil is present, it is still bioavailable to intertidal organisms, exists in a bioactive form that is capable of inducing CYP1A in fish, and is distinguishable from stressed reference locations that have sediments contaminated by other non-EVO-related sources of petroleum hydrocarbons (Springman et al. 2005; Short et al. 2005).

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[32] PAH concentrations in clam tissue ranged from less than 4,000 to 34,357 ng/g dry weight.

4.3.3.3 Effects on Reproduction

EVO effects on clam reproduction have not been assessed, but a possible change in clam recruitment has been investigated. Recruitment is discussed below in Section 4.3.5, "Other Factors Influencing Current Injury or Rate of Recovery or Population Variability."

4.3.4 Indirect Effects of Lingering Oil

The clam species of concern in this evaluation are filter feeders, ingesting plankton or other particles of suitable size in the water column. There is no evidence that lingering oil has affected plankton abundance or distribution or changed the availability or quality of food available to clams.

Studies of changes in sea otter predation pressure following the spill suggests a possible role of the oil spill in cascading effects on clam populations. Van Blaricom et al. (2002) compared clam abundance in oiled and unoiled intertidal and subtidal areas as part of their evaluation of sea otter populations in 1996–1998. The authors predicted that, consistent with general ecological theories of predator-prey relationships, a release in predation pressure following the decline in sea otter populations resulting from the spill might lead to an increase in size and abundance of preferred prey such as clams. Some findings, including an increase in mean size of littleneck and butter clams at Knight Island where sea otter abundance was significantly reduced after the spill, supported this theory (Van Blaricom et al. 2002; Dean et al. 2002). However, other results, including densities of both littlenecks and butter clams, were not different between areas, inconsistent with the theory of cascading effects of predation pressure release. The authors concluded that the relationship was not a simple causal relationship between predator and prey and suggested that other factors, including abiotic stressors and invertebrate predators, play an important role in moderating the relationship between sea otter and bivalve prey.

4.3.5 Other Factors Influencing Current Injury or Rate of Recovery or Population Variability

Other factors potentially affecting clam populations are predation, natural disturbance, and natural variability. Changes in predator (e.g., sea otter) numbers can result in changes in predation rates, affecting clam population attributes such as population abundance and size distributions (Kvitek et al. 1992; Van Blaricom et al. 2002). Natural disturbances such as storms may have localized effects but are unlikely to contribute to overall population variability. Finally, there is significant interannual variation in clam recruitment based on natural conditions that are difficult to measure and predict (Chew and Ma 1987).

4.4 ACTIONS TAKEN OR PLANNED

Immediately following the spill, there were extensive efforts to remove the oil deposited on the shoreline of PWS. Subsequent restoration effects focused on research to better understand the impacts of the spill on clams and to track recovery. Studies of clam populations following the spill indicated that clam populations in the lower intertidal habitat of oiled-and-treated beaches may have been altered in comparison with those in unoiled or oiled-and-untreated beaches. Subsequent restoration efforts focused on monitoring to assess recovery. Additional actions have included protection of clam beds through provisions in waste management plans and through continued studies in ecosystem projects, and replacement or enhancement of selected populations near several villages.

4.5 SUMMARY AND CONCLUSIONS

4.5.1 Summary

Clams in PWS are important as prey for sea otters, seabirds, and many other predators, as well as for subsistence use by native populations and recreational harvest. Within days of the EVOS in March 1989, clams on beaches along the shoreline of PWS were exposed to the spreading oil slick. Clams were killed by the initial oiling and by the subsequent cleanup activities that involved chemical dispersants and hydraulic washing. Large numbers of dead and moribund clams documented on treated beaches suggest that they may have suffered toxic effects (Lees et al. 1996), but those initial effects are difficult to separate from the effects of hydraulic washing. High-pressure, hot-water washing in particular caused thermal stress, oil dispersion, animal displacement and burial, and the transport of sediments from the upper intertidal into the lower intertidal zone.

4.5.2 Conclusions

The current condition of the clam population can potentially derive from residual effects from the original spill and cleanup methods, ongoing exposure to lingering oil, or other natural or anthropogenic stressors that can influence the population. The spatial and temporal distribution of lingering EVO suggests that ongoing exposure to lingering oil and related impacts to clams are localized, limited, and patchy (see Section 2).

Life history traits relevant to recovery from the original spill (and cleanup efforts) include high reproductive rates, rapid but variable recruitment, and long life spans (15 years for littleneck clams and 20 years for butter clams). The assessment of population-level effects of lingering oil is limited by the absence of baseline clam population data prior to the oil spill in 1989. However, comparison of populations in oiled and unoiled areas provides some indication of possible population effects due to lingering oil. Although recruitment and abundance were similar between unoiled and oiled-but-untreated sites, clam abundance was low on oiled-and-treated beaches (Driskell et al. 1996). Fukuyama et al. (2000) reported slower growth rates and higher mortality of clams transplanted to areas that had been heavily oiled. Lees and Driskell (in prep.) concluded that sediments have recovered from beach washing, but that clam assemblages lag in their recovery, but also noted that study results are not conclusive evidence that oiled, treated sites have not recovered. This caveat is important because it acknowledges the alternative hypothesis that physical factors that were influential in the trajectory of the spill and the locations at which it landed may also have defined important differences in habitat characteristics that may have been extant prior to the spill.

Clams will have recovered when population and productivity measures (such as size and distribution) at oiled sites are comparable to population and productivity measures at unoiled sites, taking into account geographic differences. It is likely that residual effects from EVO, if any, are limited to sites that experienced high-pressure, hot-water cleaning methods. At these sites, conditions have been improving over time. Residual effects of the original spill to intertidal populations are small and unlikely to affect their value as a food resource to either human or non-human predators. Although clam exposure to lingering oil is likely limited, there is insufficient recent and ongoing monitoring data to quantify the intersection of lingering oil, bioavailability to clams, and potential human and non-human consumers of this resource.

5. MUSSELS

Mussels (*Mytilus trossulus*) in PWS are important prey items for harlequin ducks, black oystercatchers, juvenile sea otters, river otters, and many other species. They are also a key component of the intertidal habitat, providing physical stability and habitat for other organisms. Within days of the EVOS in March 1989, mussels on beaches along the shoreline of PWS were exposed to the spreading oil slick. The recovery objective for mussels was defined as a return of petroleum hydrocarbon concentrations to background levels such that mussels do not contaminate their predators. The original restoration strategy relied on cleaning oiled mussel beds, monitoring, and measures to protect mussels and their habitat. In 2002, the EVOS Trustee Council (2002b) reviewed the status of mussels in PWS and concluded that mussels were continuing to recover, but were not yet fully recovered.

5.1 BACKGROUND

This section describes the natural history of mussels, initial impacts of EVOS, and the current status of injury and recovery.

5.1.1 Natural History of Mussels

5.1.1.1 Life Cycle

Mussels have separate sexes, and spawning in PWS runs from late February into August (Thomas et al. 1999). Eggs and sperm are released directly into the water column. Within 24 hours of fertilization, the embryo develops into a free-swimming larva. Within the next 24 hours, the larva grows into the more advanced veliger stage. The veliger has a ciliated velum that gives it more control in swimming and gathering food. In 2–3 weeks, veligers change from a swimming larva to a bottom-dwelling juvenile mussel (i.e., spat or seed). The newly settled mussels attach to substrates with byssal threads. Young mussels can detach and crawl or drift to a different location to seek a more favorable substrate.

Recruitment of mussels is highly variable from year to year (Gosling 1992). Growth rates are variable, depending on many factors, including temperature and the quantity and quality of food (Skidmore and Chew 1985; Gosling 1992). The life span of mussels ranges from 3 to 12 or more years, but mussels in PWS rarely live longer than 9 years (Seed 1976).

5.1.1.2 Feeding

Mussels are suspension feeders and filter a wide variety of food from the water column, including bacteria, phytoplankton, fine organic detritus, and inorganic material (Gosling 1992). Mussels feed only when submerged, so periodic inundation is a requirement for feeding and growth.

5.1.1.3 Abundance and Distribution

Mytilus trossulus is the dominant species of mussel found in PWS and the surrounding area. This species is found throughout the northern Pacific Ocean and is closely related to *M. edulis*, a species cosmopolitan throughout the northern hemisphere (Foster 1991). *M. trossulus* is abundant in the middle and lower tidal zones of rocky shorelines, can attain 100 percent coverage of the substrate in some areas, and in some cases can be found on compacted soft substrates. Like many sessile intertidal invertebrates, their upper range is often shaped by abiotic factors including temperature and risk of desiccation, while their lower limit is strongly influenced by predation (Gosling 1992). Sampling elevations reported for mussel collection in PWS are primarily in the mid-low to low intertidal zones, between 0 and +2 m MLLW (Highsmith et al. 1996), but have been reported as low as –0.8 m MLLW (Houghton et al. 1997).

Mytilus is known to form dense aggregations, or beds, which can provide structural habitat for a wide variety of other intertidal flora and fauna. Although mussels often occur in these dense beds, they can also occur as smaller patches or solitary organisms. The focus of most studies in PWS has been on mussel beds, primarily because these dense aggregations can sequester oil, which can act as a source of contamination to the community supported by the mussels.

5.1.1.4 Natural Mortality and Predation

Mussels are prey for a variety of vertebrate and invertebrate predators. They are a common food item for many seabirds (including harlequin ducks and black oystercatchers), river otters, and juvenile sea otters, as well as gastropods, crabs, and sea stars. Sea stars have been shown to be a major predator of *Mytilus,* strongly influencing the distribution and abundance of the mussel at the lower intertidal elevations (Gosling 1992).

5.1.2 Initial Impact of EVOS

In the first days to weeks following the initial spill, EVO was distributed both in open water and along the shoreline of PWS. Mussels at all tidal levels and substrates were directly exposed to EVO as oil coated mussels and penetrated the underlying mat of dense mussel beds. Cleanup operations immediately following the spill included pressure washing of intertidal areas using cold or hot water, as well as manual and mechanical removal of oil and tar, and manual removal of oiled sediments (Section 3 “Intertidal Communities”). However, dense mussel beds on unconsolidated substrates were generally not cleaned following the spill so that the stability and habitat they provide would be preserved.

Mussels were initially exposed to EVO through direct contact and ingestion of particulate oil. Petroleum hydrocarbons were initially accumulated as whole particulate oil rather than by absorption of hydrocarbons dissolved in seawater or associated with suspended sediment or particulates (Short and Babcock 1996). Petroleum hydrocarbon concentrations in mussels sampled in the spill area immediately following EVOS were much higher than in mussels sampled outside the spill area (Babcock et al. 1996). In addition, hydrocarbon concentrations in mussels immediately following the spill (1989) were higher than in mussels sampled prior to the spill (1977–1979). Mussels collected from 13 heavily oiled intertidal locations in PWS 6 weeks after EVOS had total petroleum hydrocarbon concentrations of up to 100,000 µg/kg dry weight compared to mean hydrocarbon concentrations of 143 to 544 µg/kg dry weight before EVOS (Short and Babcock 1996).

In many areas, the initial high petroleum hydrocarbon concentrations in mussel tissue began declining soon after EVOS. By 1991, there were no significant differences in mussel biomass and abundance between many oiled and non-oiled locations with similar habitat, and PAH concentrations in mussels from many areas had returned to background levels (Short and Babcock 1996). However, mussel populations from many areas with coarse-textured soft substrate still showed significant differences compared to reference locations and compared to mussels sampled prior to the spill (Highsmith et al. 1996; Babcock et al. 1996).

5.1.3 Current Status of Injury and Recovery Classification

The EVOS Trustee Council (EVOS Trustee Council 2002b) defines the recovery goal for mussels as “a return to conditions that would have existed had the spill not occurred.” The specific recovery objectives described by the Trustee Council are concentrations of oil in mussels at background concentrations and mussels that do not contaminate their predators (EVOS Trustee Council 2002b).

As of 1995, some mussel beds in PWS still contained lingering EVO, although more than half of the sites surveyed showed significant natural declines in oil concentrations (Carls et al. 2001a). While hydrocarbon concentrations in mussels from some sites were expected to reach background concentrations in a few years, contamination was expected to persist in mussels at other sites that were well protected or where oil penetrated into underlying sediments. In addition, there were concerns about continued hydrocarbon exposure to sea otters and harlequin ducks, although the pathway of exposure was not known. Based on this evidence, in 2002, the Trustee Council determined that “mussels continue to be recovering, but are not yet fully recovered from the effects of the oil spill” (EVOS Trustee Council 2002b).

5.2 SPATIAL AND TEMPORAL DISTRIBUTION OF MUSSEL POPULATIONS

M. trossulus is abundant in middle and lower tidal zones[33] of protected bays (Houghton et al. 1997; Lindstrom et al. 1999). The upper elevation is ultimately set by physical factors such as exposure to air and desiccation; predation is important in determining lower elevation limits (Seed 1976). Habitat in PWS includes exposed and sheltered bedrock and boulder shoreline, but mussels also form dense beds on soft (i.e., gravel) sediment. The beds are composed of interconnected living and dead mussel shells and accumulated sediments and debris. The beds can be up to 10 to 20 cm thick, as younger mussels settle and accumulate on established beds of older mussels. The mussels’ byssal threads can be so closely intertwined that it is possible to tear the bed away from the underlying sand or gravel in a solid mass (Rickets and Calvin 1968). Mussel bed sizes can range from 20 to 2,200 m^2 or more and beds may cover up to 60–100

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[33] Approximate intertidal zone elevations are shown in Figure 3-1.

percent of the sediment surface (Boehm et al. 1996; Lindstrom et al. 1999; Carls et al. 2001a). Reported mussel densities in PWS ranged from 20 to more than 10,000 individuals/m^2 (Babcock et al. 1996, 1998; Boehm et al. 1996; Carls et al. 2001a).

Mussel beds provide important intertidal habitat, both as a substrate for animals to attach to and as cover or protection for animals below. Additional information on these intertidal communities is provided in Section 3, "Intertidal Communities."

The variety of substrate that can be occupied by mussels results in a broad areal distribution in PWS. An overview of common intertidal habitats occupied by mussels (exposed and sheltered rocky shores, coarse-textured beaches) is provided in Section 3, "Intertidal Communities," and shown in Figure 3-2. There are no comprehensive data for mussel distribution or abundance throughout PWS.

The Alaska earthquake of 1964 and the uplift of the shoreline in PWS had a major impact on the distribution of intertidal resources, including mussels (Hanna 1971). The entire spill zone is within the area that was uplifted up to 15 m, and in some areas subsided more than 1 m, from its original pre-earthquake elevation (Hanna 1971). At some locations, the magnitude of the uplift was great enough to elevate portions of the intertidal habitat to levels that were above the water depth needed to sustain the mussel communities.

5.3 LINKAGE BETWEEN EVOS AND THE CURRENT CONDITION OF MUSSEL POPULATIONS IN PWS

This section assesses the potential for ongoing injury to mussels based on the current nature and extent of lingering EVO and the likelihood of exposure to a degree that could cause adverse effects and injury. The current condition of mussel populations can potentially derive from one or more of the following:

- Residual effects from the original spill
- Ongoing exposure to lingering oil
- Other natural or anthropogenic stressors that can influence mussel populations.

The relative importance of these different factors is assessed based on the mussel life history and inherent ability of mussel populations to recover from the initial impacts of the spill, the likelihood that their behavior could result in ongoing exposure to lingering EVO to a degree that could cause adverse effects and injury, and the nature and magnitude of other factors that could affect the condition of these populations.

5.3.1 Residual Effects from the Original Spill

The lack of baseline information on mussel population abundance in PWS prior to the spill prevents direct assessment of potential changes in mussel populations due to lingering oil. Initial decreases in mussel abundance were observed in spill areas (Houghton et al. 1991; Highsmith et al. 1996), particularly in areas that had been treated with high-pressure hot-water washing

(Houghton et al. 1996). However, post-spill comparison of localized mussel densities at oiled and unoiled sites indicated that, by 1991, significant differences in abundance and biomass were not apparent in sheltered rocky and PWS estuarine habitats; mussel populations from coarse-textured Cook Inlet area habitats still showed significant differences (Highsmith 1996). A study of intertidal epibiota in PWS found no statistically significant difference for mussels between oiled and unoiled sites by 1992 (Houghton et al. 1996).

As described in the section "Initial Impact of EVOS," mussels in many PWS locations showed recovery and natural declines in tissue PAH concentrations within a few years following the spill. Many mussel communities in PWS are not currently impacted by lingering oil. Hydrocarbon concentrations in mussels on hard rock substrate have generally returned to background levels. However, numerous investigations since the mid-1990s have documented elevated PAH concentrations in mussels at some oiled beaches compared to mussels at unoiled locations or pre-spill levels (Carls et al. 2001a). While PAH concentrations in mussels declined in PWS and the Gulf of Alaska, some beaches remained contaminated 10 years after the spill (Carls et al. 2001a, 2004; EVOS Trustee Council 2002b).

Carls et al. (2001a) studied mussel beds at 80 beaches in PWS and 18 beaches in the Gulf of Alaska over four years from 1992 to 1995. By the end of the study in 1995, TPAH in mussels declined at many locations, but remained significantly greater than background concentrations (90 µg/kg dry weight) at six locations. In a subsequent study, Carls et al. (2004) noted that by 1999 TPAH concentrations in mussels were below background levels at all but one of seven restored mussel beds and that TPAH concentrations in mussels had generally declined to background levels throughout PWS. Carls et al. (2004) found that TPAH concentrations in mussels were initially reduced by beach restoration, but rebounded in subsequent years, suggesting that transport mechanisms invaded the restored sediments and recontaminated the mussels. In a separate 1999 study of residual oiling of armored beaches and mussel beds in the Gulf of Alaska, Irvine et al. (2002) measured TPAH in mussels from six locations. Tissue concentrations were less than background levels established by Carls et al. (2001a, 2004) at five of these locations and were less than twice background at the remaining site. Twice background is a benchmark established by Carls et al. (2001a, 2004) below which concentrations were not considered to be significantly elevated.

In 2002, Boehm et al. (2004) measured TPAH concentrations in composite samples of mussels from beaches identified by Short et al. (2001) as having lingering oil. Mean tissue TPAH in mussel samples from lingering oil beaches were 54 µg/kg dry weight compared to mean concentrations of 28 and 106 µg/kg dry weight in beds not exposed to EVO.[34] These results were from composite samples based on a sampling grid and, in contrast to Carls et al. (2001a, 2004), did not target areas where lingering oil was visibly present or elevated.

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[34] Boehm et al. (2004) report TPAH concentrations minus parent naphthalene. The two reported PAH levels are the mean value for sites not associated with human and industrial use, and the mean for sites associated with human and industrial use, respectively.

5.3.2 Current Exposure to Lingering Oil

Despite the direct measurements described above, the extent and magnitude of the current exposure of the mussel populations in PWS to lingering subsurface oil cannot be directly quantified. Investigations assessing PAH concentrations in mussel tissue have been based on non-random sampling and many targeted known or visibly oiled locations, and often the "most oiled" portions of mussel beds (Babcock et al. 1996). Therefore, they do not allow estimates of the current extent of mussel exposure to lingering oil relative to the overall mussel population in PWS.

Two studies of the distribution of lingering oil in the intertidal zone were conducted in 2001 (Short et al. 2004b) and 2003 (Short et al. 2006) (see Section 2 for detailed discussion). The 2001 study was a stratified random survey of lingering oil that did not include all mussel habitat. The 2001 study shows that lingering oil is present in the middle to upper intertidal zone (+1.8 to +4.8 m MLLW) on approximately 60 percent of the beach segments surveyed. However, the results also indicate that lingering oil has a limited patchy distribution on beaches where it is found. In comparison, mussels are most abundant at lower intertidal and lower-mid intertidal elevations (Section 5.1.1.3). The 2003 investigation covered the full range of tidal excursions (from –0.2 to +4.8 m MLLW) at three areas on northern Knight Island. The results of this study showed that lingering oil was present at less than half (44 percent) of the beaches sampled, had a distribution limited to less than 0.5 percent of the beach surface, and was found at tidal elevations ranging from –0.2 to +4.8 m MLLW.

Lingering oil is clearly present in surface and subsurface sediments in the lower intertidal zone and was historically observed below mussel beds (Short et al. 2004b; Short and Babcock 1996; Carls et al. 2001a). Researchers agree that the total area of oil-contaminated mussel beds as a percentage of all mussel beds in PWS was small in the 1990s (Boehm et al. 1996; Carls et al. 2001a; Babcock et al. 1996). Mussels are widely distributed on both hard and soft coarse substrates throughout PWS, including areas such as Smith Island and northern Knight Island that were moderately to heavily oiled by EVOS. Using estimates of habitat types and mussel densities in the different habitats, Boehm et al. (1996) estimated that fewer than 3 percent of the mussels in PWS were exposed to lingering subsurface oil, and more recently reduced the estimate to ~0.1–0.2 percent (Boehm et al. 2004). Other mussel and subsurface oil sampling investigations (e.g., Carls et al. 2001a, 2004) have not used random sampling and so cannot be used to estimate the proportion of the mussel population in PWS potentially affected by lingering oil. Finally, in a recent review, Page et al. (2005) summarized mussel data for PWS for the years 1990, 1991, 1993, 1998, 1999, 2000, 2001, and 2002. These results show that PAHs from EVOS residues that remain buried in shoreline sediments after the early 1990s are in a form and at locations that provide a low accessibility to mussels living in the intertidal zone.

Outside of PWS along the Gulf of Alaska, a number of studies have been performed to evaluate lingering EVO on boulder armored beaches or mussel beds at the Kenai Fjords and Katmai National Parks and the Kodiak Archipelago (see Section 2). Although measurable oiling was reported along the Kodiak Archipelago in 1990 and 1991, no oil or only trace amounts of oil were observed in 1995. Along the Kenai Fjords and Katmai National Parks, the majority of armored beaches visited had largely unweathered EVO present as mousse in the subsurface, while areas of surface oiling were largely absent. EVO was generally found in sediments at mussel bed sites, but by 2002, the areal extent and concentration of EVO declined at most sites (Irvine et al. 2002). As of 2005, the decreasing trend in EVO present in sediments at mussel bed sites appears to be continuing in these remote areas (Irvine 2005, pers. comm.).

5.3.3 Toxicity of Lingering Oil to Mussels

The primary route for mussel exposure to subsurface lingering EVO was ingestion of petroleum hydrocarbons in water (Carls et al. 2001a; Boehm et al. 1996). Mussels are found on the sediment surface, and there is little direct contact with subsurface lingering oil. Mussels are suspension feeders and filter large volumes of seawater. Oil from contaminated subsurface sediment may enter interstitial or overlying water with suspended sediment or as dissolved hydrocarbons (Carls et al. 2001a). Both particulate and dissolved hydrocarbons are ingested by mussels as they filter seawater. Specific investigations of EVO or lingering EVO toxicity to mussels are presented in the following sections.

5.3.3.1 Acute and Chronic Toxicity to Individuals

Mussels both accumulate and depurate hydrocarbons. The primary routes for mussels to accumulate hydrocarbons are direct uptake from water filtered by the animals or ingestion of contaminated particulate matter. Mussels will depurate PAH rapidly if not chronically exposed to hydrocarbons in water or sediment.

Many physiological responses in mussels were not correlated with chronic exposure to lingering oil 3–4 years after EVOS. In a 1992 study by Thomas et al. (1999), total PAH tissue concentrations[35] in mussels ranged from 0 to 6,000 µg/kg dry weight and were significantly different between mussels from oiled and non-oiled beaches. However, there was no significant difference in byssal thread production, condition index, feeding rate, or glycogen content between mussels from oiled beaches and those from reference locations, or between mussels from beds overlying oiled sediment and from nearby hard substrate.

Mussels collected at oiled locations in PWS in 1993 (Morado et al. 1998; Shigenaka 1997) showed changes in histopathology compared to mussels from an unoiled control site. Although sampling was limited, observations indicated that mussels from the oiled locations had increased digestive gland metaphasia, increased brown cells, decreased abundance of storage cells, and increased hemocytic infiltrates in gonads compared to controls.

5.3.3.2 Bioavailability and Bioaccessibility of Lingering EVO

At present, the very small quantities of lingering oil have a patchy distribution and limited bioaccessibility to mussels in PWS (see “Residual Effects from the Original Spill” and “Current Exposure to Lingering Oil” sections above). Where it was present in the past, there is evidence that subsurface oil is unevenly distributed within mussel beds (Harris et al. 1996). Factors such

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[35] Nondetected values in this study were reported as a zero.

as the original degree of oiling, differences in exposure and water flow over the bed, and grain size all affect the distribution of subsurface oil within a mussel bed.

Although bioaccessibility is low, subsurface lingering oil remains in a bioavailable state as indicated by the continued presence of elevated hydrocarbons in some mussel beds with lingering oil (Carls et al. 2001a). Bioaccessibility and bioavailability of lingering oil was also demonstrated in experiments conducted by Shigenaka and Henry (1995) 3 years following EVOS in which TPAH concentrations were measured in SPMDs, sediments, water, and caged mussels placed on a previously heavily oiled beach on Smith Island, and on an unoiled to lightly oiled beach also on Smith Island. Results of this study showed that TPAH concentrations were significantly correlated among the SPMDs, mussels, and sediments, further indicating the bioavailability of lingering oil to surface-dwelling organisms.

The PWS Regional Citizens' Advisory Committee long-term environmental monitoring program documented a return of PAH and AHC (aliphatic hydrocarbons) levels in mussel tissue in PWS to background levels by 2003 (Payne et al. 2005). With the exception of two sites, the Alyeska Marine Terminal site and the Gold Creek site in Port Valdez, 10 regional sites did not show elevated concentrations of hydrocarbons. At the two elevated sites, where PAH and AHC contaminants from the Alyeska Marine Terminal Ballast Water Treatment Facility were detected, concentrations were small and the authors suggested that PWS is not heavily contaminated from ongoing anthropogenic activities (Payne et al. 2005).

The 2001 (Short et al. 2004b) and 2003 (Short et al. 2006) studies of lingering oil in the intertidal zone (described above) confirm its presence in patchy localized areas in the middle to lower intertidal zones inhabited by mussels. Recent research conducted in 2004 using SPMDs placed in the intertidal zone indicates that where lingering oil is present, it is still bioavailable to intertidal organisms, exists in a bioactive form that is capable of inducing CYP1A in fish, and is distinguishable from stressed reference locations that have sediments contaminated by other non-EVO-related sources of petroleum hydrocarbons (Springman et al. 2005; Short et al. 2005).

5.3.3.3 Effects on Reproduction

Shigenaka (1997) reported that there was some indication of offset reproductive timing in oil-exposed mussels collected at Smith Island in 1993, but no further information was located. More than 10 years after the initial spill, mussel abundance does not differ significantly between sampling locations with subsurface oil and locations without subsurface oil. However, mussel recruitment patterns can exhibit large temporal and spatial variability (Gosling 1992), making it difficult to assess the extent to which local reproduction contributes to local recruitment and abundance. Therefore, effects on reproduction may not be discernible from local patterns of adult mussel abundance.

5.3.4 Indirect Effects of Lingering Oil

Mussels are filter feeders, ingesting plankton or other particles of suitable size in the water column. There is no evidence that lingering oil has affected plankton abundance or distribution or changed the availability or quality of food available to mussels.

5.3.5 Other Factors Influencing Current Injury or Rate of Recovery or Population Variability

There are a large number of natural factors contributing to mussel population variability, including predation, food supply, competition for food and space, currents and larval dispersion, storm disruption, and climatic factors. Estimates of the time for mussel recovery (e.g., PAH tissue concentrations declining to background or pre-spill levels) have varied widely (Babcock et al. 1996; Carls et al. 2001a; Page et al. 2005). Natural factors will affect recovery rates. For example, Irvine et al. (2002) suggested that disturbance by storms or mussel mortality could be important factors in determining the recovery of mussel beds. Between 1995 and 1999, they observed greater decreases in subsurface oil concentrations in mussel beds compared to armored beaches. This observation could be due to more frequent or severe disturbance in the mussel beds (i.e., death of mussels in beds) although there are also differences in the tidal elevations. In addition, based on unpublished data, Carls et al. (2004) reported that there was evidence of unexplained regional declines in mussel populations in the last decade, but no supporting information or evidence was provided.

5.4 ACTIONS TAKEN OR PLANNED

Immediately following the spill, there were extensive efforts to remove the oil deposited on shorelines of PWS. Subsequent restoration efforts focused on research to better understand the impacts of the spill on mussels and to track recovery. Studies of mussel beds following the spill indicated that mussels at some locations may accumulate lingering oil at concentrations greater than background levels. Subsequent restoration efforts have focused on monitoring to assess recovery, active cleaning and restoration of some beds, and acquisition of habitat important to mussels.

5.5 SUMMARY AND CONCLUSIONS

5.5.1 Summary

Mussels in PWS are important prey items for harlequin ducks, black oystercatchers, juvenile sea otters, river otters, and many other species. They are also a key component of the intertidal habitat, providing physical stability and habitat for other organisms. Within days of the EVOS in March 1989, mussels on beaches along the shoreline of PWS were exposed to the spreading oil slick. Mussels at all tidal levels and substrates were directly exposed to EVO as oil coated mussels and penetrated the underlying mat of dense mussel beds. Cleanup operations immediately following the spill included pressure washing of intertidal areas using cold or hot water, as well as manual and mechanical removal of oil and tar, and manual removal of oiled sediments (Section 3, “Intertidal Communities”). However, dense mussel beds on unconsolidated substrates were generally not cleaned following the spill so that the stability and habitat they provide would be preserved.

In many areas, the initial high petroleum hydrocarbon concentrations in mussel tissue began declining soon after EVOS. Hydrocarbon concentrations in mussels on hard rock substrate have returned to background levels. However, numerous investigations since the mid-1990s have documented elevated but declining PAH concentrations in mussels at some oiled beaches compared to mussels at unoiled locations or pre-spill levels. Carls et al. (2001a) studied mussel beds at 80 beaches in PWS and 18 beaches in the Gulf of Alaska over four years from 1992 to 1995. By 1995, TPAH in mussels declined at many locations, but remained significantly greater than background concentrations at six locations. In a subsequent study, Carls et al. (2004) noted that by 1999, TPAH concentrations in mussels were below background levels at all but one of seven restored mussel beds and that TPAH concentrations in mussels had generally declined to background levels throughout PWS.

In 2002, Boehm et al. (2004) measured TPAH concentrations in composite samples of mussels from beaches identified by Short et al. (2001) as having lingering oil. Mean tissue TPAH in mussel samples from lingering oil beaches were comparable to levels in beds not exposed to EVO. These results were from composite samples based on a sampling grid and, in contrast to Carls et al. (2001a, 2004), did not target areas where lingering oil was visibly present or elevated.

In a separate 1999 study, Irvine et al. (2002) measured TPAH in mussel beds from six locations in the Gulf of Alaska, all of which were either below or not significantly elevated in comparison to background levels.

5.5.2 Conclusions

The recovery objective for mussels was defined as a return of petroleum hydrocarbon concentrations to background levels such that mussels do not contaminate their predators. Existing data suggest that for nearly all of PWS, PAH concentrations in mussels have returned to background levels. It is highly unlikely that any remaining localized areas with elevated PAH would contaminate mussels at levels that would result in unacceptable risk to predators.

6. HARLEQUIN DUCK (*Histrionicus histrionicus*)

Harlequin ducks are present year-round in the nearshore environment of PWS. They are at maximum density during the winter, and thus the full population was present in March 1989, when the EVOS occurred. Ducks in the path of the oil slick became coated with oil, and an estimated 7 percent (range = 3–12 percent) of the wintering population of PWS and a higher percentage of those wintering in oiled areas suffered acute mortality from oiling (Piatt 2005, pers. comm.). In the years following the oil spill, population counts were variable by season and year; no clear trend of increasing population was established. The EVOS Trustee Council identified demographic parameters during breeding and nonbreeding seasons and biochemical measures as key indicators of recovery status. This section evaluates these and other population parameters to assess the likelihood of ongoing injury to harlequin ducks from EVO.

6.1 BACKGROUND

This section describes the natural history of harlequin ducks, initial impacts of EVOS, and the current status of injury and recovery.

6.1.1 Natural History of Harlequin Duck

Harlequin ducks are found throughout PWS, where they occupy shallow subtidal and intertidal zones, rich with benthic invertebrate food sources. They are seasonal migrants, with the wintertime population of roughly 15,000 in PWS (average abundance during 1990–2004 winters from Sullivan et al. 2005) reduced by more than half in June, when breeding birds have flown to their freshwater breeding grounds on inland rivers and streams.

6.1.1.1 Life History

Harlequin ducks are a relatively long-lived waterfowl with a diverse, primarily invertebrate diet that uses both fresh and saltwater habitats over its lifecycle. The coastal breeding range of harlequin ducks extends from northern California to Alaska including the Aleutians. Females become reproductively mature at 2 years of age, but have low reproductive success until 5 years of age (Robertson and Goudie 1999). A harlequin duck of at least 10 years of age was identified in British Columbia, suggesting the life span is at least this long or longer (Robertson and Goudie 1999). Low annual productivity is compensated for by relatively high adult survival (Esler et al. 2002). Population change is most sensitive to changes in adult female survival (Robertson and Goudie 1999). Because of limited dispersal, immigration to an area after a population decreases due to some natural or anthropogenic factor may be limited. Recovery time from a perturbation like an oil spill may be extended by demographic lags associated with low immigration rates, high site fidelity, and low reproductive rates.

6.1.1.2 Seasonal Behavior and Habitat

Harlequin ducks nest inland, but they overwinter in coastal, marine waters. They are known for their high site fidelity (philopatry[36]) on both wintering and breeding areas (Robertson and Goudie 1999; Patten et al. 2000). PWS is prime wintering habitat and is one of the northernmost wintering areas in the species' range (Esler et al. 2002). Harlequin ducks in PWS exhibit a high degree of fidelity to specific wintering areas (Iverson et al. 2004) (e.g., within approximately the same few kilometers of shoreline). Depending on spatial scale, within-winter emigration rates of wintering harlequin ducks within PWS range from 4 to 10 percent (Iverson and Esler in press).

Wintertime activity is focused on foraging (Robertson and Goudie 1999). During the day, they remain close to shore, spending most of their time foraging in shallow subtidal and intertidal waters (Robertson and Goudie 1999). At night, they often "raft-up" offshore in small groups to rest (Robertson and Goudie 1999; Rizzolo 2004).

Pairs form on winter range as early as October, and pair formation continues through spring (approximately May) until breeding birds depart for their nesting grounds farther inland (Robertson and Goudie 1999). Some nonbreeding individuals remain on marine waters over summer. Breeding males remain on the nesting grounds for only a short period, departing sometime during egg incubation (which is carried out by the female), and arriving back on their wintering grounds around the end of June (Robertson and Goudie 1999). Females with broods appear some time in late August or September. Wing molting, during which adult and juvenile birds (but not fledglings) are flightless, occurs in the fall on marine waters.

Most of the breeding locations for the PWS wintering population are unknown. Rosenberg and Petrula (1998) conducted post-spill brood surveys and concluded that breeding habitat in the PWS watershed is limited and that most harlequin ducks leave the area to breed. A few harlequin ducks breed locally on streams draining into PWS and rear their broods in estuaries where they feed on salmon roe and intertidal invertebrates.

6.1.1.3 Diet

During the winter, harlequin ducks feed on small (≤ 25 mm) intertidal and subtidal invertebrates (Esler et al. 2002). North American winter diets were found to consist of 57 percent crustaceans (decapods, amphipods, isopods, and barnacles), 25 percent molluscs (snails, limpets, chitons, and bivalves [especially blue mussel]), 10 percent insects and 5 percent echinoderms and fish (Robertson and Goudie 1999). Local diets of harlequin ducks in PWS were quantified after the EVOS in 1989–1990. They consisted of 20 percent snails of the genus *Littorina*, 18 percent snails of the genus *Lacuna*, 12 percent blue mussels (*Mytilus* sp.), and 10 percent limpets (*Lottia*); the remaining 40 percent consisted of 24 different taxa, all in small amounts (Patten et al. 2000). Harlequins were observed grazing, dabbling, and diving for invertebrates, at or near the surface of the water in the intertidal zone (Patten et al. 2000).

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[36] The tendency of an individual to return to, or stay in, its home area or another adopted locality.

In summer, PWS harlequin duck diets are supplemented with seasonally available salmon eggs and in spring, with herring roe (Patten et al. 2000). Breeding birds forage on freshwater aquatic insects, including midge larvae, caddis flies, stone flies, and mayflies (Robertson and Goudie 1999).

6.1.1.4 Predators and Disease

Predation on harlequin ducks occurs on both wintering and breeding grounds. Bald eagles occur in PWS and are known to prey on harlequin ducks. They attack aggressively by swooping repeatedly and causing the ducks to dive until exhausted (Robertson and Goudie 1999). On breeding grounds, predators include mink, martens, hawks, great horned owls, arctic foxes, and common ravens (Robertson and Goudie 1999). There is no evidence to evaluate the degree to which predation limits the size of wintering populations of harlequin duck (Rosenberg and Esler 2005, pers. comm.).

Studies of disease suggest that harlequin ducks do not have parasite loads as heavy as those of other sea ducks (Robertson and Goudie 1999). Feather lice, acanthocephalan worms, and trematodes have been identified as parasitic organisms on harlequin ducks, and one trematode, *Paramonostomum histrionici*, appears to be unique to harlequin ducks, not having been identified in any other host species (Robertson and Goudie 1999).

6.1.2 Initial Impact of EVOS

The EVOS occurred in March, when the full wintering population of harlequin ducks was present in PWS. Harlequins and other sea ducks are among the most vulnerable to oil spills because they spend most of their time in nearshore waters where they feed on benthic invertebrates (Piatt et al. 1990). Within days of the spill, hundreds of harlequin ducks became coated with oil and suffered acute mortality due to loss of insulation (caused by oiled feathers) and subsequent hypothermia. Most carcasses recovered were completely coated with oil. Others may have suffered from ingestion of oil and subsequent toxicological effects (Esler et al. 2002; Patten et al. 2000). Patten et al. (2000) found hydrocarbon metabolites in 4 percent of harlequin ducks collected (shot) from oiled areas in 1989 and 1990, consistent with exposure to oil. Using an estimated carcass recovery rate of 15 percent, harlequin duck mortality for the years 1989 and 1990 was estimated at 1,413 for the entire spill area and 980 in PWS, which is about 7 percent of the wintering population (Piatt and Ford 1996; Esler et al. 2002). Rosenberg (2006) estimated 3,199 ducks in the spill area of PWS based on an average of 6 years of counts in west PWS and 5 years of counts in southwest PWS, plus a 20 percent estimate to cover areas not surveyed, yielding an estimate of 31 percent of harlequin ducks in the spill area of PWS killed by oiling (980 out of 3,199).

6.1.3 Current Status of Injury and Recovery Classification

Pre-spill data on harlequin duck population trends and demographics in PWS are limited, making it difficult, if not impossible, to compare post-spill trends with pre-spill data. Most of the information available for evaluating the status of injury comes from post-spill comparisons of oiled and unoiled parts of PWS. Esler et al. (2002) concluded that harlequin duck populations had not recovered by 1998 and that adverse effects from oil continued for at least 9 years after the spill. Their conclusion is based on the following:

- Elevated CYP1A in ducks wintering in previously oiled areas compared with those wintering in unoiled areas
- Lower winter survival of adult females in previously oiled areas compared with those in unoiled areas
- A declining trend in oiled areas from 1995 to 1997 based on fall population surveys by the Alaska Department of Fish and Game (ADFG)
- Lower population densities in oiled areas in 1996 and 1997 than would be expected based on habitat.

The 2002 recovery plan states as the recovery objective "[h]arlequin ducks will have recovered when breeding- and nonbreeding-season demographics return to prespill levels and when biochemical indicators of hydrocarbon exposure in Harlequins in oiled areas of PWS are similar to those in Harlequins in unoiled areas" (EVOS Trustee Council 2002b). The EVOS Trustee Council (2002b, p. 12) concludes that "[a]lthough some of the indicators show signs of recovery, the majority of the indicators do not indicate recovery. Taken together, the population census trends, survival measures, and indicators of exposure suggest that the Harlequin Duck has not recovered from the effects of the oil spill."

These data and other evidence for injury and recovery are reviewed and evaluated in the following sections.

6.2 SPATIAL AND TEMPORAL CHANGES IN HARLEQUIN DUCK POPULATION

Population trend data are complicated by two main factors aside from sampling design issues: 1) a lack of pre-spill data, and 2) temporal variation (seasonal and annual) in populations. Population trends of harlequin ducks in PWS prior to EVOS are unknown. The only pre-spill population data available were collected in the summers of 1984 and 1985. These data have been of little use for comparison to post-spill populations since the number of harlequin ducks that leave the area to breed (and timing of return by post- and non-breeding males) may fluctuate from summer to summer. Also, there are no data on normal year-to-year variation in the absence of a spill. There is relatively little breeding activity that occurs within the PWS watershed; thus, pairs begin leaving for breeding grounds in May, and individuals return to PWS from June through August or September. Breeding propensity, phenology, and success may vary from year to year, making annual comparisons of spring and summer data difficult. By September, most birds have returned to PWS although some movements (redistribution) may occur following molt. Winter populations are the most stable (Rosenberg and Petrula 1998; Esler et al. 2002), and therefore winter population data should provide the best indicator of overall population trends in PWS.

The overall winter population trend of harlequin ducks in PWS has been stable since the spill through 2004 (Sullivan et al. 2005). Analysis of the 15 years of population data collected by USFWS (Sullivan et al. 2005) indicates that harlequin ducks showed no increase or decrease in densities in the oiled area during summer (July) and winter (March) and no absolute difference in trends in oiled and unoiled areas over the 15-year study period. Focusing on winter data, researchers for ADFG (Rosenberg et al. 2005) found no difference in the change in density of harlequin duck between oiled and unoiled areas and no overall trend in PWS (increasing or decreasing) from 1997 to 2005. Using summer pre- and post-spill data for 10 bays (nine of them moderately to heavily oiled during the spill and located in the Knight Island area) in PWS, Murphy et al. (1997) did not identify a significant change in harlequin duck numbers pre- and post-spill.

6.3 LINKAGE BETWEEN EVOS AND THE CONDITION OF HARLEQUIN DUCK POPULATIONS IN PWS

The current condition of the harlequin duck population in PWS can potentially derive from one or more of the following:

- Residual effects from the original spill, including reduced abundance due to demographic lags
- Ongoing exposure to lingering oil
- Other natural or anthropogenic factors that can influence the population.

The relative importance of these different factors is assessed based on the life history of the harlequin duck and the inherent ability of the population to recover from the initial impacts of the spill, the nature and degree of ongoing exposure to lingering EVO that could cause adverse effects, and the nature and magnitude of other factors that could affect the condition of the population. A great deal of research has focused on the relationship between lingering EVO and harlequin duck, and this work is the central focus of the discussion below.

6.3.1 Residual Effects from the Original Spill

Residual effects of the EVOS on harlequin ducks in PWS have been studied by comparing duck populations in oiled and unoiled areas, including analyses of long-term trends in the population abundances. Insights into potential recovery times from initial perturbations may also be gained by analysis of species life history characteristics. Each of these approaches is discussed below.

Comparisons of population trends in oiled vs. unoiled portions of the spill area have been made by Murphy et al. (1997), Rosenberg and Petrula (1998), Agler and Kendall (1997), Lance et al. (2001), Irons et al. (2000), Wiens et al. (2004, 2005), Sullivan et al. (2005), and Rosenberg et al. (in review). Our summary of analyses of temporal trends and comparisons between areas will generally focus on winter data because several investigators (e.g., Esler et al. 2002 and Rosenberg and Petrula 1998) believe winter is the best time to survey for population comparisons. Because of the statistical limitations of field surveys of duck populations in their ability to detect relatively small changes (e.g., < 10 percent) in abundance over time periods of several years, the best information available is provided by the most recent analysis of population data by USFWS (Sullivan et al. 2005).

Rosenberg and Petrula (1998) found a decline in harlequin duck populations in western PWS during the period 1995–1997, whereas none was detected in eastern PWS. They believe the decline is due to lower survivorship in western PWS, not due to lower recruitment. Based on their data on population structure and size, Rosenberg and Petrula (1998) concluded "[the harlequin duck] population in oiled areas of WPWS [western PWS] has the potential to recover from the effects of the EVOS. However, our trend analysis indicates that the population in oiled areas is still declining".

Several population studies using overlapping data sets have found some time periods within which the harlequin duck population in oiled areas of PWS increased (e.g., Agler and Kendall [1997] for the March population in 1989, 1990, 1991, 1993, 1994, and 1996; Lance et al. [2001] for the March population in 1989–1998). However, the analysis by Sullivan et al. (2005) of USFWS long-term data on the harlequin duck population of PWS showed that the overall winter population trend of harlequin ducks in oiled areas of PWS from 1989 through 2004 has been stable since the EVOS despite some evidence of an increase in the population from March 1990 to March 1996. According to the analysis by Sullivan et al. (2005), there is no evidence of increasing populations in oiled areas or PWS as a whole over the full time period (1989–2004).

Sullivan et al. (2005) found no population trends for harlequin duck in oiled portions of PWS surveyed in March and July and no absolute difference in trends relative to unoiled areas (Sullivan et al. 2005). Given the overall lower abundance of harlequin duck in oiled areas compared with unoiled areas after the EVOS, Sullivan et al. (2005) interpret[37] the absence of a difference in population trends between oiled and unoiled areas of PWS to be an absence of evidence of a recovering population. Rosenberg et al. (2005) conducted a more comprehensive analysis of sex and age composition and population trends in PWS, and concluded that 1) harlequin duck demographics in oiled areas of PWS are similar to those elsewhere in their range, 2) populations in oiled areas are stable, 3) age ratios are similar in oiled and unoiled areas, and 4) female survival has improved. They noted that the lower proportion of females in oiled areas is a concern, but concluded that based on population data the outlook for full recovery is good.

Wiens et al. (2004) evaluated habitat use by harlequin ducks and determined that harlequin duck populations in oiled areas were not adversely impacted relative to unoiled areas. However, Wiens et al. (2004) surveyed harlequin duck populations during summer only, which may not be the best time of year to detect impacts of EVOS.

Despite extensive evaluations of population survey data and winter survivorship of harlequin ducks in PWS, as discussed above, there has been no detailed demographic modeling to assess

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[37] To test whether the populations in east and west PWS were changing at different rates, Sullivan et al. (2005) examined the homogeneity of the slopes of the logarithms of the densities over time between the oiled and the unoiled areas using linear models. They concluded that significantly different slopes indicated that densities of a species or species group in the oiled area were changing at a different rate than in the unoiled area.

the significance of the initial mortality due to the spill. Moreover, modeling to estimate recovery time after the initial impact and the effect of demographic lags has not been conducted. Life history analysis of the harlequin duck suggests that recovery of a population after a perturbation like the EVOS could be extended (e.g., 10 years or more) because naturally they are relatively long-lived birds (up to 12 years) with high survival and low reproduction rates. Their low productivity is associated with a late age at first breeding, a small average clutch size, and a high proportion of non-breeding birds in some years (Canadian Wildlife Service 2006). Dispersal and immigration rates are thought to be low (Iverson et al. 2004). Thus, lowered survivorship over a several year period (as observed by Esler et al. 2002 after the EVOS) could have an extended effect on a population.

6.3.2 Current Exposure of Harlequin Duck to Lingering EVO

Harlequin ducks are well distributed throughout PWS. Based upon the mapping analysis of current lingering EVO presented in Section 2, there is a small geographical overlap between shorelines used by harlequin ducks and shorelines with EVO (Figure 6-1). Bird surveys published in the PWS Environmental Site Investigation Data Atlas, supplemented by additional data provided by D. Rosenberg, indicate that concentrations of harlequin duck within the oiled area (1989–2001 oil data) of PWS are distributed over approximately 1,800 km of shoreline. Approximately 15 km (0.8 percent) of this distribution coincides with previous heavily to moderately oiled areas where lingering oil has been identified. Harlequin ducks could be exposed to lingering oil in this area through ingestion of contaminated prey. They could also be exposed through contact with sediments while foraging beneath rocks in the intertidal zone (Esler et al. 2002).

Trust et al. (2000) measured CYP1A levels in harlequin ducks (and Barrow's goldeneyes) in 1998 as an indicator of exposure to oil constituents 9 years after the spill. CYP1A is a liver enzyme that is induced in many vertebrate species following exposure to PAHs (Golet et al. 2002) and polychlorinated biphenyls (PCBs) (Trust et al. 2000).

Trust et al. (2000) sampled harlequin ducks in March-April of 1998 at oiled sites (Crafton Island and Main Bay) and unoiled Montague Island. They collected liver samples from 37 harlequin ducks (19 from oiled sites, 18 from unoiled sites) and assessed them for CYP1A induction by measuring 7-ethoxyresorufin-*O*-deethylase (EROD) activity. EROD activity was significantly ($p<0.001$) higher in the ducks from the oiled sites. Levels of PCB congeners suspected of inducing CYP1A in birds did not differ between oiled and unoiled populations. One particular congener (PCB congener 138) was measured above detection limits in all samples and was positively related to EROD activity, although it did not differ between the populations at oiled and unoiled sites. PCB congener 138 was present at elevated levels in only four blood plasma samples. After accounting for variation due to this congener, birds from oiled areas still had significantly ($p<0.001$) higher EROD activity than those from unoiled areas. In other words, CYP1A was higher in the oiled population even after accounting for the effects of PCB congeners.

CYP1A testing in harlequin ducks has continued since the 1998 studies reported above (Bodkin et al. 2003). Results from studies conducted in 2000–2002 initially indicated that levels in the oiled population have converged with levels in the unoiled population (Bodkin et al. 2003). However, the preliminary results for CYP1A data collected in March 2005 (Ballachey et al. 2006) indicate that levels were significantly higher in harlequin ducks collected from oiled areas relative to unoiled areas. The difference in the CYP1A results between the 2000–2002 time period and the 2005 results is partly because the variance of the mean for the 2005 data sets (oiled and unoiled treatment means) was much lower than that for the 2000–2002 data sets. Nevertheless, a trend of decreasing CYP1A induction over time is evident in the oiled areas, with the level of CYP1A induction in oiled areas being about double that of unoiled areas in 2005.

6.3.3 Toxicity of Lingering EVO to Harlequin Duck

6.3.3.1 Acute and Chronic Toxicity to Individuals

In 2000–2002, Bodkin et al. (2003) conducted experimental feeding of hydrocarbon-contaminated food to captive harlequin ducks. The researchers measured CYP1A levels as an indicator of exposure in their experimental and control populations. They also observed behavior and measured food consumption, body mass, and metabolism at two levels of contamination in their experimental population and in a control group. Results of this unpublished study showed that CYP1A was induced at levels greater than in the control population. However, there were no significant differences in any of the parameters that would indicate effects of ingestion of contaminants on either behavior or energetics in the experimental population. These results are difficult to compare with wild populations of harlequin ducks potentially exposed to lingering oil because the degree of exposure in the wild population is unknown.

Another potential cause of mortality is oiling of plumage, which can lead to ingestion of oil and possibly effects on behavior and physiology, as well as hypothermia. Although contact of harlequin duck with lingering oil is possible (see below), the probability of plumage oiling has not been assessed.

6.3.3.2 Bioavailability and Bioaccessibility of Lingering EVO

Lingering oil buried in the sediment may be bioaccessible to the extent that it can migrate to the sediment–water interface through physical processes or be exposed through the burrowing or foraging activities of organisms. Physical transport of lingering oil to the sediment–water interface was demonstrated in experiments conducted by Shigenaka and Henry (1995) 3 years following the EVOS. Shigenaka and Henry (1995) assessed bioavailability of lingering oil using SPMDs. SPMDs are a research tool that has been used to simulate water-mediated uptake and bioaccumulation potential of oil and other substances by aquatic organisms. Shigenaka and Henry (1995) measured TPAH concentrations in SPMDs, sediments, water, and caged mussels placed on a previously heavily oiled beach on Smith Island and on an unoiled to lightly oiled beach also on Smith Island. Results of this study showed that TPAH concentrations were significantly correlated among the SPMDs, mussels, and sediments, further indicating the bioavailability of lingering oil to surface-dwelling organisms. Preliminary results demonstrating the bioavailability of lingering EVO and its ability to induce CYP1A has recently been demonstrated by Short et al. (2005), who compared the potential for CYP1A induction in PAHs collected by SPMDs deployed in different part of PWS where different types of petroleum contamination was known to occur. Extracts were injected into trout fry (Springman et al. 2005) and EVOS was demonstrated to be a potent inducer of CYP1A, unlike samples collected from human use sites or other area-wide sources.

Harlequin ducks forage on small invertebrates that are either attached to or sequestered beneath coarse gravel, cobble, or larger sediments. Consequently, bioturbation of surficial sediments during foraging and ingestion of contaminated prey are two mechanisms that have been hypothesized for exposure to lingering oil (Esler et al. 2002; Bodkin et al. 2003). Plumage oiling through contact with oiled sediments may also occur, but the importance of this mechanism is unknown.

6.3.3.3 Effects on Reproduction and Other Population Parameters

Rosenberg and Petrula (1998) compared demographics of harlequin ducks in oiled western PWS with unoiled eastern PWS during surveys conducted in both spring and fall from 1995 to 1997. They found more males than females in both areas. They found no major differences in recruitment between the two areas as evidenced by similar proportions of subadults in the population. Few broods were observed in eastern PWS and no broods were observed in western PWS, leading them to conclude that breeding habitat is limited in PWS, and most pairs go elsewhere to breed and fledge their young. Overall, they did not detect any substantial differences in population, age, and sex structure between eastern PWS and western PWS that would indicate effects of oil on demographics (but see discussion of results of Esler et al. [2000a] below for information on winter survivorship). However, despite similarity in population structure, they did find that harlequin duck populations were declining in western PWS and when interpreted in context with other EVOS studies (Holland-Bartels et al. 1998), attributed this decline to EVO exposure.

In 1995–1997, Esler et al. (2000a) compared harlequin duck densities on oiled Knight Island and unoiled Montague Island. They examined the relationship between habitat variables, food availability, and oiling history on densities of wintering harlequin ducks. After accounting for differences in habitat, densities were lower in oiled than unoiled areas.

Esler et al. (2000b) also found winter survival of females to be lower in previously oiled areas in western PWS than on unoiled Montague Island. Kaplan-Meier estimates of survival were 78 and 84 percent, respectively, during the winters of 1995–1996 and 1997–1998 (data from all three years were pooled). The study projects annual population growth of 0.5 percent for the unoiled sites on Montague Island, and a population decrease of 5.4 percent annually for the oiled areas. In contrast, Lance et al. (2001) measured overall winter population increases in the oiled area during an overlapping time period (1990–1998). The winter survival study was repeated in 2000–2003 (Bodkin et al. 2003). Preliminary findings of this unpublished study indicate that there are no significant differences in cumulative winter survival of female harlequin ducks between previously oiled (81 percent survival) and unoiled (84 percent survival) areas.

6.3.4 Indirect Effects of Lingering Oil

Studies have been done to determine whether EVO has affected the availability of prey for harlequin duck. Both Esler et al. (2000a) and Holland-Bartels (2000) reported that the availability of prey was the same between oiled and unoiled sites during 1996 and 1997. Prey availability does not appear to be a factor in female winter survival (Esler et al. 2000b). Similar body mass between females from oiled and unoiled sites also indicates that food is not limited for the population in the oiled area (Esler et al. 2000b). There is no evidence indicating harlequin ducks have altered their use of habitats or their migratory behavior subsequent to the spill.

Bald eagles are natural predators of harlequin ducks, and some subsistence harvest by Native Americans may contribute to mortality. However, predation on harlequin ducks in PWS has not been quantified, and there are no data to assess whether lingering oil has affected predation levels on harlequin duck. Mortality and survival studies (Esler et al. 2000a) do not provide data on the amount of mortality that is due to predation or the relative amounts of predation between oiled areas in PWS and unoiled areas on Montague Island.

6.3.5 Other Factors Influencing Current Injury or Rate of Recovery or Population Variability

There are numerous environmental stressors that can affect harlequin duck populations and complicate interpretation of their recovery status. Rosenberg and Petrula (1998) identified seasonal, interannual, and spatial factors that affect reproductive fitness and success. Influencing factors include seasonal cues that affect the onset and timing of breeding chronology, availability, and competition for limited, quality breeding habitat, timing and success of migration to winter habitat, variability in the quality of winter habitat and over-winter survival, immigration/emigration particularly for unmated bachelor males, and predatory–prey cycles. There is no evidence to suggest that any of these factors cause any differences between harlequin duck populations in oiled and unoiled areas of PWS or account for the observed population trends.

6.4 ACTIONS TAKEN OR PLANNED

Immediately after the spill, there were extensive efforts to rescue and rehabilitate oiled birds. Subsequent restoration efforts focused on oil cleanup to minimize exposure to oil and on research to better evaluate harlequin duck recovery and mechanisms of injury. Research has included assessment of biomarkers indicating exposure (CYP1A), assessment of the impact of the spill on prey availability, population modeling, survival analysis, habitat assessment, population distribution and movement, and genetic evaluations.

6.5 SUMMARY, CONCLUSIONS, AND RECOMMENDATIONS

6.5.1 Summary

Harlequin ducks are found throughout PWS, where they occupy shallow subtidal and intertidal zones, rich with benthic invertebrate food sources. They are at maximum density during the winter, and thus the full population was present in March 1989, when the spill occurred. An estimated 7 percent (range of 3–12 percent) of the wintering population suffered acute mortality from oiling. Because they forage for crustaceans, molluscs, and other food in the intertidal zone, harlequin ducks have a high potential for exposure to lingering EVO, which is concentrated in the intertidal region of the shoreline. Short et al. (2005) recently presented preliminary results demonstrating the bioavailability of lingering EVO and its ability to induce CYP1A.

Life history analysis of the harlequin duck suggests that recovery of a population after a perturbation like the EVOS could be extended (e.g., 10 years or more) because naturally they are relatively long-lived birds (up to 12 years) with high survival and low reproduction rates. Dispersal and immigration rates are thought to be low. Thus, lowered survivorship over a several year period could have an extended effect on a population.

Long-term analysis (1989–2004) of harlequin duck populations in oiled areas of PWS indicates that the populations surveyed during March are stable, with no significant increasing or decreasing trends and no significant differences between population densities in oiled and unoiled areas (Sullivan et al. 2005; Rosenberg et al. 2005). Harlequin duck sex and age structure in oiled areas of PWS are similar to those elsewhere in their range, populations in oiled areas are stable, age ratios are similar in oiled and unoiled areas, and female survival has improved (Rosenberg et al. 2005). The slightly lower proportion of females in oiled areas remains a concern; however, the relevance of this condition to residual spill effects or biological recovery is not known.

Exposure of harlequin ducks to PAHs has been measured since 1998 using CYP1A, a liver enzyme that is induced in many vertebrate species following exposure to PAHs. Results from studies conducted in 2000–2002 initially indicated that levels in the oiled population have converged with levels in the unoiled population (Bodkin et al. 2003). However, the preliminary results for CYP1A data collected in March 2005 (Ballachey et al. 2006) indicate that levels were significantly higher (approximately doubled) in harlequin ducks collected from oiled areas relative to unoiled areas. Most researchers agree that as the spatial extent of lingering oil diminished over time, the potential for or degree of exposure to harlequin duck also diminished.

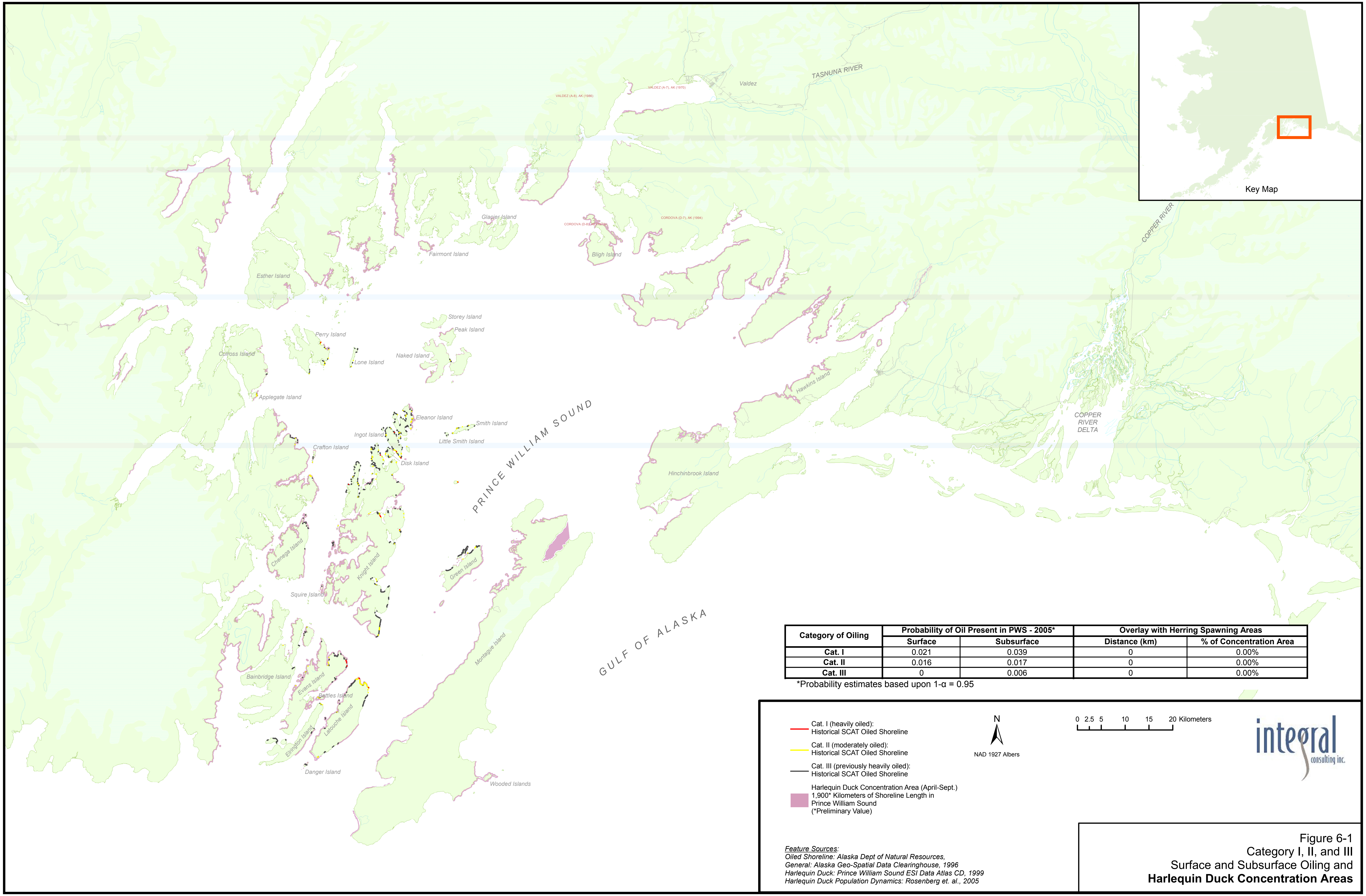
6.5.2 Conclusions

The recovery objective for harlequin ducks states that they "will have recovered when breeding- and nonbreeding-season demographics return to prespill levels and when biochemical indicators of hydrocarbon exposure in Harlequins in oiled areas of PWS are similar to those in Harlequins in unoiled areas" (EVOS Trustee Council 2002b). It is likely that harlequin duck populations in PWS are nearly recovered or recovering from the acute and chronic effects of the spill.

However, the lack of an increasing population trend in western PWS and the apparent continuing PAH exposures as indicated by CYP1A elevations are still of concern to some researchers.

6.5.3 Recommendations

Additional population modeling is recommended to better understand the population dynamics of harlequin ducks, and to better assess the potential for residual impacts from the spill or impacts from lingering oil. Modeling should include a component to estimate oil encounter and ingestion rates for harlequin duck and the potential for CYP1A induction. Additional research to better understand the potential impacts of other stressors (e.g., climatic shifts or trends, predator–prey relationships) would also be beneficial. Monitoring of populations and exposure (e.g., through CYP1A studies) should continue.



Category of Oiling	Probability of Oil Present in PWS - 2005*		Overlay with Herring Spawning Areas	
	Surface	Subsurface	Distance (km)	% of Concentration Area
Cat. I	0.021	0.039	0	0.00%
Cat. II	0.016	0.017	0	0.00%
Cat. III	0	0.006	0	0.00%

*Probability estimates based upon 1-α = 0.95

🟥	Cat. I (heavily oiled): Historical SCAT Oiled Shoreline
🟨	Cat. II (moderately oiled): Historical SCAT Oiled Shoreline
⬛	Cat. III (previously heavily oiled): Historical SCAT Oiled Shoreline
🟪	Harlequin Duck Concentration Area (April-Sept.) 1,900* Kilometers of Shoreline Length in Prince William Sound (*Preliminary Value)

<u>Feature Sources</u>:
Oiled Shoreline: Alaska Dept of Natural Resources,
General: Alaska Geo-Spatial Data Clearinghouse, 1996
Harlequin Duck: Prince William Sound ESI Data Atlas CD, 1999
Harlequin Duck Population Dynamics: Rosenberg et. al., 2005

Figure 6-1
Category I, II, and III
Surface and Subsurface Oiling and
Harlequin Duck Concentration Areas

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7. PACIFIC HERRING (*Clupea pallasii)*

Within a week of the EVOS in March 1989, Pacific herring (*Clupea pallasii*) and eggs deposited on beaches were exposed to the spreading oil slick in open water and along the shoreline. Although egg mortality and larval deformities were documented, the population level effects of these injuries were not clearly established. However, in 1993, the Pacific herring population in PWS declined dramatically. Based on this decline, and the speculative relationship between this decline and EVO, the EVOS Trustee Council (1994) established recovery objectives and defined a restoration strategy for Pacific herring in PWS. The recovery objective was defined as a return to healthy and productive pre-spill herring population abundances. Research into the cause(s) of the Pacific herring decline, monitoring, and habitat protection were adopted as the restoration strategies that would be implemented to meet the recovery objective. In 2002, the EVOS Trustee Council (2002b) reviewed the status of the Pacific herring in PWS and concluded that it could not be considered to be recovering. The recovery objectives were modified to specify that Pacific herring will have recovered when the next highly successful year class is recruited into the population and when other indicators of population health are within normal bounds in PWS. The restoration strategy was not modified.

7.1 BACKGROUND

This section describes the natural history of Pacific herring, initial impacts of EVOS, and the current status of injury and recovery.

7.1.1 Natural History

Pacific herring are of both ecological and commercial importance in PWS. Not only are they central to the marine food web, providing food to marine mammals, birds, and invertebrates, but they are also fished commercially for food and bait, sac-roe, and spawn on kelp (Thomas and Thorne 2003). Key elements of Pacific herring natural history in PWS are summarized in the following sections.

7.1.1.1 Life Cycle and Distribution in PWS

There are four Pacific herring life stages—eggs, larvae, juveniles, and adults—and all are found in PWS at various seasons and locations (Brown and Carls 1998). Spawning in PWS typically takes place in April and the spawning season varies from 5 days to 3 weeks. Pacific herring typically spawn along the same beaches each year, although the volume of eggs and shoreline distance varies (Brown and Carls 1998; Carls et al. 2002). For example, from 1994 to 1997, the annual spawning beach length ranged from 23.3 to 68.5 km (Willette et al. 1998). Figure 7-1 shows Pacific herring spawning beds located throughout PWS based upon 1989–1998 data provided by ADFG (Moffitt 2004, pers. comm.). During spawning, the eggs attach to eelgrass, rockweed (i.e., *Fucus*), and kelp in shallow subtidal and intertidal areas. The eggs hatch in May, about 24 days after spawning depending on temperature (Hart 1973; Brown and Carls 1998).

After hatching, the larval herring migrate to the surface and continue to grow and congregate nearshore. The larvae become juveniles in July, about 10 weeks after hatching. In the fall, the juveniles move into deeper water but nearshore habitat remains important for at least the first year, and they may spend up to 2 years in nearshore areas or bays before joining the adult population residing in deeper waters (Brown and Carls 1998).

In PWS, adult Pacific herring rarely spawn before their third year and may live up to 12 years. After spawning in the spring, adult Pacific herring disperse from the spawning aggregations to multiple schools in deeper waters, presumably close to the entrances of PWS (Brown and Carls 1998). In the fall, adult and 2-year old Pacific herring return from summer feeding areas and overwinter in central and eastern PWS.

7.1.1.2 Feeding

Newly hatched larvae carry a yolk sac that is typically depleted in the first week. The earliest larval stages begin feeding on invertebrate eggs and small zooplankton such as copepods. While the larval Pacific herring grow and congregate nearshore through their first summer, they continue to live mainly on copepods but may also eat other crustaceans, barnacle larvae, mollusc larvae, or young fishes (Brown and Carls 1998). As they move into deeper water, copepods remain an important food for both juvenile and adult Pacific herring, but adults also feed on larger crustaceans and small fish. Adult Pacific herring stop feeding in the fall and live off storage lipids throughout the winter, and their reproductive tissues begin to mature through the winter without feeding prior to spawning in the following spring (Carls et al. 2001b).

7.1.1.3 Natural Mortality

All Pacific herring life stages are important prey items in the marine food web (Hart 1973). Egg mortality caused by foraging birds or wave scouring can be as high as 90 percent. In addition, up to 50 percent of eggs that do hatch may exhibit morphological abnormalities due to natural factors (Carls et al. 2002). Newly hatched larvae are preyed on by invertebrates and fish and can be swept away by currents. Juvenile and adult Pacific herring are a critical food resource for salmon, seabirds, and marine mammals (Brown and Carls 1998). In addition, Pacific herring and their eggs are an important commercial fishery in PWS with most harvest occurring in the spring (Pearson et al. 1995).

7.1.2 Initial Impact of EVOS

In the first days to weeks following the initial spill, EVO was distributed both in open water and along the shoreline of PWS. As such, it is reasonable to presume that most life stages of Pacific herring were exposed to EVO to some degree following the initial spill. The spill occurred a few weeks before Pacific herring spawned in PWS, so eggs and adults were exposed directly to EVO, ranging from oiled habitat to dissolved concentrations of the oil in the water column. Larval contact with EVO may also have occurred given that EVO was distributed in nearshore areas for several months following the spill and was often resuspended in the water column as a result of the large cleaning effort transpiring in multiple areas and bays throughout PWS. Because there are similarities in the mechanisms of uptake and toxicological modes of action are comparable during these life stages, it is difficult to separate impacts of exposure as larvae from earlier exposure as eggs. Observations of juvenile Pacific herring exposure to EVO are anecdotal, based on observations of juveniles in contaminated intertidal habitat (Carls et al. 2001b). Although larvae may also have been exposed to EVO through ingestion of oil-contaminated food,[38] this was not a significant route of exposure (Kline 1999). Prespawning adults in nearshore areas generally do not feed and so did not ingest EVO or oil-contaminated food.

The early life stages are the most sensitive to the effects of oil. Based on the extent of visible oil on beaches observed during initial surveys conducted in 1989 by Pearson et al. (1995) estimated that 4–10 percent of the total Pacific herring spawn length in PWS occurred along shorelines with oil. In addition, EVO contaminants were shown by Trustee-sponsored researchers to be available in open water along shorelines where oiling was not directly evident. Using PAH accumulation in mussels from spawn beaches as an index of oil exposure, Brown et al. (1996) estimated that 40–50 percent of the eggs were exposed to EVO in 1989. By combining mussel accumulation data with the timing of the egg deposition at various spawn sites, Carls et al. (2002) refined this estimate down to 25–32 percent of the 1989 eggs that were exposed to EVO. While Pacific herring did not spawn on the most heavily oiled beaches, they did spawn on shorelines considered lightly to moderately oiled (Brown et al. 1996).

Pacific herring larvae from eggs from oiled beaches showed increased sublethal impacts and mortality compared to eggs from non-oiled beaches (Hose et al. 1996; McGurk and Brown 1996). Given the fragile nature and vulnerability of herring larvae, sublethal laboratory effects are very likely to be lethal in the wild. Both field and laboratory studies demonstrated sublethal impacts that included premature hatching, low larval weights, reduced growth, and increased incidence and severity of morphological deformities and genetic abnormalities (McGurk and Brown 1996; Marty et al. 1997; Norcross et al. 1996; Brown and Carls 1998; Carls et al. 1999). Of the embryos present in PWS in 1989, 25–32 percent may have been damaged (Carls et al. 2002). By 1991, sublethal larval measurements had returned to probable baseline levels (Hose et al. 1996).

There is no direct measurement of the percentage of the adult population that encountered EVO. Severe lesions and elevated PAH levels were observed in some adult Pacific herring from oiled areas (Brown and Carls 1998; Marty et al. 1999). Herring typically rise to the surface after darkness and would have had increased exposure probability at this time. Laboratory studies showed abnormalities and possible depressed immune functions in Pacific herring exposed to oil (Carls et al. 1999, 2001b). Significant adult mortality was not observed in 1989 (Carls et al. 2002), but this would not be unexpected given the heavy predation or scavenging by different groups of predators.

EVO exposure was suspected as a potential cause of the herring population decline in 1993. However, numerous other reasons could also explain this decline: disease aggravated by high population density, food scarcity, and poor ocean conditions (EVOS Trustee Council 1999; Brown et al. 1996).

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[38] Copepods, one of the major herring prey items, have been shown to accumulate and concentrate petroleum hydrocarbons (EVOS Trustee Council 1998; Brown et al. 1996).

It is likely that no single factor explains the 1993 population decline. In a synthesis of the literature, Carls et al. (1999, 2001b) surmise that the combination of high population density, poor nutrition, and epidemic infection by viral hemorrhagic septicemia virus caused the collapse in a boom-and-bust cycle typical of Pacific herring populations in the Gulf of Alaska and elsewhere. Pearson et al. (1999) also concluded that a combination of increasing biomass and decreasing food supply led to poor Pacific herring condition and the 1993 decline. Both studies acknowledged that natural factors and variability could explain the population decrease, and a clear link between EVOS and the 1993 population decline was not established, nor eliminated. Disease in the Pacific herring population in PWS continues and it appears to be a limiting factor in the recovery today. The role of disease and other factors that may be important in controlling Pacific herring populations are discussed below in Section 7.3.1 ("Residual Effects from the Original Spill").

7.1.3 Current Status of Injury and Recovery Classification

The EVOS Trustee Council (2002b) defines the recovery goal for Pacific herring as "a return to conditions that would have existed had the spill not occurred". The specific recovery objectives described by the Trustee Council are 1) Pacific herring will have recovered when the next highly successful year class is recruited into the population, and 2) when other indicators of population health (such as biomass, size-at-age, and disease expression) are within normal bounds in PWS (EVOS Trustee Council 2002b).

As described above, the Pacific herring population began increasing in 1997 following the 1993 decline. However, the population increase stalled in 1999. In 2002, because the population had yet to exhibit a highly successful year class, the Trustee Council stated that "…Pacific herring can only be considered to be not recovering from the effects of the oil spill" (EVOS Trustee Council 2002b). The EVOS Trustee Council also reported that 70 percent of the 2002 population was composed of young, 3-year old fish and that this could be an indication that the next large year class is present in the population and that recovery was under way. However, as of 2004, the PWS Pacific herring biomass had still not reached the minimum threshold and the commercial fishery remained closed.

7.2 SPATIAL AND TEMPORAL CHANGES IN PACIFIC HERRING POPULATION

Pacific herring populations in PWS were increasing in the late-1980s before EVOS, and record numbers for the modern fishery were reported in 1989. However, in 1993 the Pacific herring fishery collapsed: the overall 1993 harvest was only 14 percent of the 1992 harvest (Pearson et al. 1999) and the 1989 year class was one of the smallest cohorts ever to return to spawn (EVOS Trustee Council 1998; Brown et al. 1996).[39] The Pacific herring fishery in PWS was closed from 1994 to 1996.

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[39] Recruitment of the 1989 year class was also low in other Alaskan herring stocks (Funk 1995).

Pacific herring populations in PWS began increasing in 1997 and 1998 but numbers leveled off beginning in 1999 and the fishery was again closed, which has continued through 2005. In 2002, the Pacific herring population had still not met the recovery objective of recruitment of the next highly successful year class into the population, although there were indications that the population was increasing (EVOS Trustee Council 2002b). However, as of spring 2004, ADFG reported that Pacific herring biomass in PWS remained slightly below the minimum spawning biomass threshold (i.e., 22,000 tons) for establishing a Pacific herring fishery in PWS (ADFG 2004) and the fishery was again closed and remains closed in 2005. Although Pacific herring are renowned for their high interannual variability in so-called boom-or-bust cycles of productivity, the persistence of the 1993 crash appears unprecedented and is of concern because upward swings in the population are characteristic of this species in other locations in Alaska.

7.3 LINKAGE BETWEEN EVOS AND THE CONDITION OF HERRING POPULATIONS IN PWS

The current condition of Pacific herring population can potentially derive from one or more of the following:

- Residual effects from the original spill
- Ongoing exposure to lingering oil
- Other natural or anthropogenic stressors that can influence the population.

The relative importance of these different factors is assessed based on the Pacific herring life history and inherent ability of the population to recover from the initial impacts of the spill, the likelihood that the behavior of Pacific herring could result in ongoing exposure to lingering EVO to a degree that could cause adverse effects and injury, and the nature and magnitude of other factors that could affect the condition of the population.

7.3.1 Residual Effects from the Original Spill

Whether the initial spill continues to exert residual effects on Pacific herring populations is unknown and is the subject of an ongoing review and research (Rice et al. 2005). This research, which is scheduled for completion in May 2006, will focus on uniqueness of the PWS herring stocks relative to oil, disease, recruitment success, and resiliency through genetic diversity. The task group involved with this project met in October 2005 to discuss their preliminary findings. Preliminary (unpublished) findings into residual effects of EVO focused on four areas of investigation:

- Persistent Toxicological Mechanisms—Two mechanisms that could promote long-term multigenerational toxicity were discussed: long-term immunosuppression to disease from initial exposures in 1989, and effects from lingering oil exposures that cause continuing or cascading effects. Neither mechanism was considered plausible. The overlap between herring use and lingering oil does not exist.

- Fish Pathogens—Disease associated with two fish pathogens (viral hemorrhagic septicemia virus and *Icthyophonus hoferi*[40]) continue to be the leading suspected cause for limiting recovery of Pacific herring. The time period over which these two diseases have been limiting recovery appears to be unique and is on the order of a decade. The cause of the continuing disease problem is unknown.
- Population Dynamics—In a recent retrospective study, it was hypothesized that the decline in Pacific herring populations was initiated by the spill in 1989, but was not detectable until 1993 (Thomas and Thorne 2003; Thorne 2005). Two independent measures of spawning success were used in a population model to test this idea but yielded contradictory results. Resolution of these competing hypotheses does not appear possible.
- Genetic Diversity—It was hypothesized that historical fishing practices coupled with the population crash in 1993 resulted in a population with low genetic diversity that limits it ability to tolerate disease or other environmental stressors. Preliminary review indicates that genetic diversity in Pacific herring from PWS is relatively high and no different from that in other populations along the Pacific Coast of North America.

Based on these preliminary findings, the cause of the continued depressed Pacific herring population in PWS is linked to continuing disease problems. Although the cause of the disease is not clear, it does not appear to be linked to lingering EVO, and any other mechanisms (immunosuppression, cascade effects) linking back to residual effects are possible but not evident. The continuing disease problem for Pacific herring in PWS is unusual; the cause of the vulnerability to disease remains unknown.

7.3.2 Current Exposure of Pacific Herring to Lingering EVO

Pacific herring are most likely to encounter lingering oil as eggs and developing embryos affixed to aquatic vegetation near the benthic-water interface in the intertidal and shallow subtidal zones of the shoreline. The eggs incubate for 24 days, during which time they could be continually exposed to lingering oil if it is present. Assuming that lingering oil is present, this would be considered a chronic exposure because it occurs over the entire duration of a sensitive life history stage for Pacific herring. In 1989, toxicity of EVO to Pacific herring eggs and larvae occurred via water-borne exposure (Carls et al. 2001b, 2002), and this is also the likely pathway for exposure and toxicity to lingering oil today.

Short et al.'s (2004b) recent study of lingering oil was based on the likelihood that most lingering oil occurs in the intertidal zone of previously heavily to moderately oiled beaches. Consequently, Pacific herring exposure to lingering oil is limited to spawning beaches that were heavily to moderately oiled in 1989. While the actual geographical distribution of lingering oil within Pacific herring spawning areas has not been determined, a relatively small portion of all

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[40] A fungus-like organism associated with massive mortalities in herring in the Atlantic Ocean, and has recently been reported to cause disease in wild Pacific herring from Washington through Alaska.

Pacific herring spawning areas is in heavily to moderately oiled areas that may still contain lingering oil (Figure 7-1).

Willette et al. (1998) located Pacific herring spawning areas using aerial surveys followed by direct diver observation. While the total length of shoreline spawned each year between 1994 and 1997 varied, their maps confirmed that the same general areas were spawned each year. General spawning areas are similar to those documented in the 1970s and 1980s (Moffitt 2004, pers. comm.). There is little overlap in herring spawning areas and lingering oil areas, and no indication that herring spawning areas have changed substantially in the last decade in response to lingering oil.

Subsequent life stages of Pacific herring are even less likely to be exposed and are less susceptible to the toxic effects of lingering EVO. Because Pacific herring migrate away from the intertidal and nearshore subtidal benthic habitats after hatching, it is unlikely that they would encounter lingering oil either by direct contact or by foraging activities in their later life history stages. Although zooplankton can accumulate petroleum hydrocarbons, ingestion of zooplankton is not a significant route of petroleum hydrocarbon exposure to larval or juvenile Pacific herring (Carls 1987; Kline 1999). Adult Pacific herring were also exposed to EVO in 1989 via contact with water as indicated by liver lesions, elevated concentrations of PAH in their tissues, and possibly increased sensitivity to viral infection[41] (Carls et al. 2001b). However, these symptoms were not evident in adult fish in years subsequent to the spill when aqueous concentrations of EVO in the nearshore environment returned to background levels.

In summary, while lingering oil exists in some parts of the sound, particularly in the intertidal zone of northern Knight Island, there is little to no overlap with this habitat and the spawning areas. Lingering oil is not likely to be a cause for poor recovery.

7.3.3 Toxicity of Lingering EVO to Pacific Herring

Mechanisms of toxicity of petroleum hydrocarbons to Pacific herring are largely known from research conducted on EVO immediately following the spill and for the ensuing decade following the sustained collapse of Pacific herring in PWS. Although there is very little overlap of lingering oil with Pacific herring spawning areas, as indicated in Section 7.3.2, these mechanisms are reviewed here to summarize potential effects of lingering oil should it occur.

7.3.3.1 Acute and Chronic Toxicity to Individuals

Developing eggs are the only Pacific herring life stage with possible significant exposure to lingering oil (see “Current Exposure” section above). Developing eggs of Pacific herring are also the most vulnerable and sensitive life stage for exposure to petroleum hydrocarbons (Carls et al. 1999, 2001b). In laboratory experiments, Carls et al. (1999) simulated the weathering

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[41] Viral hemorrhagic septicemia virus (Kocan et al. 1999).

process for EVO[42] to establish quantitative relationships between EVO exposure and toxicity to Pacific herring eggs and embryos and the larvae that hatched from the oil-exposed eggs. Oil exposure was expressed in two ways—as the initial TPAH concentration in water, and as the TPAH concentration in the developing egg tissue after 16-days of continuous aqueous exposure. Toxic endpoints that were measured were mortality of eggs and newly hatched larval fish, and a variety of sublethal effects, including time to hatching, abnormal development, chromosomal aberrations, physical deformities, and swimming behavior of neonates. The lowest-observed-effect concentration (LOEC) for sublethal effects in developing eggs is 0.4 µg/L based on TPAH in water, and is 22 µg/kg wet tissue weight based on TPAH in eggs. The LOEC for mortality to nascent larvae is 0.7 µg/L based on TPAH in water, and is 108 µg/kg wet tissue weight based on TPAH in eggs.

7.3.3.2 Bioavailability and Bioaccessibility of Lingering EVO

Pacific herring eggs are deposited on the surface of marine plants (brown seaweeds and eelgrass) that live at the sediment-water boundary in the intertidal zone. Consequently, lingering oil is bioaccessible to developing Pacific herring eggs to the extent that it can be mobilized to the sediment-water boundary. This was demonstrated in experiments conducted by Shigenaka and Henry (1995) 3 years following EVOS. Shigenaka and Henry (1995) assessed bioavailability of lingering oil using SPMDs. SPMDs are a research tool that has been used to simulate water-mediated uptake and bioaccumulation potential of oil and other substances by aquatic organisms. Shigenaka et al. (1995) measured TPAH concentrations in SPMDs, sediments, water, and caged mussels placed on a previously heavily oiled beach on Smith Island, and on an unoiled to lightly oiled beach also on Smith Island. Results of this study showed that TPAH concentrations were significantly correlated among the SPMDs, mussels, and sediments, further indicating the bioavailability of lingering oil to surface-dwelling organisms. However, the study was not designed to develop predictive relationships between water and sediment exposures and tissue concentrations or adverse effects in sediment dwelling biota and cannot be used to assess injury to Pacific herring eggs. Recent research conducted in 2004 using SPMDs placed in the intertidal zone indicates that where lingering oil is present, it is still bioavailable to intertidal organisms, exists in a bioactive form that is capable of inducing CYP1A in fish, and is distinguishable from stressed reference locations that have sediments contaminated by other non-EVO-related sources of petroleum hydrocarbons (Springman et al. 2005; Short et al. 2005).

7.3.3.3 Effects on Reproduction

There is no indication of Pacific herring reproductive impairment attributable specifically to lingering oil based upon laboratory and field studies. Although laboratory experiments showed that Pacific herring eggs were negatively affected (i.e., increased genetic and morphological defects) by exposure to weathered oil, field studies show no evidence of oil-related effects on reproductive success. In 1992, three years after EVOS, reproductive success of adult Pacific herring collected from two previously oiled sites and one unoiled site were compared. Although

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[42] Two fractions of Alaska North Slope Crude Oil were tested—less weathered oil (LWO) and more weathered oil (MWO). Carls et al. (1999, 2001b) emphasize the results of the MWO testing to represent exposure to EVO stranded on PWS beaches in 1989.

significant differences in some reproductive measures were observed, they could not be directly related to previous EVO exposure and could have been due to natural variability and environmental conditions unrelated to oil (Kocan et al. 1996). Another field comparison of oiled and unoiled sites was conducted in 1995. Results indicated that Pacific herring in PWS appeared to be reproductively fit and similar to Pacific herring from other areas (Johnson et al. 1997; Carls et al. 1997). Carls et al. (1997) concluded that factors other than reproductive impairment are affecting Pacific herring recovery in PWS.

7.3.4 Indirect Effects of Lingering Oil

As described above, Pacific herring encounter lingering oil as eggs attached to algae in the intertidal and shallow subtidal zones. Egg exposure to lingering oil is via water-borne oil contaminants. In addition to direct contact, there may be potential impacts resulting from habitat changes due to lingering oil. These potential indirect impacts are discussed in the following sections.

7.3.4.1 Food

Zooplankton in the nearshore water column, particularly copepods, are the primary food of Pacific herring during the juvenile and larval life stages (Carls et al. 2001b; Hart 1973), and zooplankton can accumulate petroleum hydrocarbons (Brown et al. 1996). However, Pacific herring are not likely to be exposed to lingering oil via ingestion of zooplankton. Even immediately following EVOS, ingestion of contaminated zooplankton was not a significant route of petroleum exposure to early Pacific herring life stages (Carls 1987; Kline 1999). In addition, there are no data in the literature indicating that zooplankton in the nearshore water column are currently affected by exposure to lingering oil.

Changes in fish population structure can affect feeding competition between species. For example, pollock populations in PWS have increased at the same time Pacific herring populations have decreased, and juveniles of both fish species may compete for zooplankton prey. However, there is no evidence linking these changes in species abundance to the presence of lingering oil, and other causes have been proposed (Sturdevant et al. 2001).

7.3.4.2 Shelter/Habitat

Pacific herring eggs are deposited on algae (primarily *Fucus* and kelp) in intertidal and shallow subtidal areas. All other Pacific herring life stages are found in the water column. If lingering oil exposure results in changes to algal species distribution in spawning areas (e.g., *Fucus* replaced by green, ephemeral algae [Peterson et al. 2003]), it is possible that this could affect spawning and/or egg survival. However, there are no reports in the literature to indicate that this has occurred, and no change in spawning locations outside normal variability has been observed (Willette et al. 1998). In addition, as stated previously, a relatively small proportion of total Pacific herring spawning area is potentially impacted by lingering oil.

7.3.4.3 Migratory Behavior

Pacific herring migrate away from the shoreline after hatching, so except for during the egg stage, Pacific herring do not encounter lingering oil. No change in Pacific herring distribution and migration patterns has been reported, and Pacific herring movement is not affected by lingering oil.

7.3.5 Other Factors Influencing Current Injury or Rate of Recovery or Population Variability

Worldwide, herring populations show large fluctuations, with crashes followed by periods of recovery that may take a decade or longer (Hay et al. 2001). Only a single stock of Pacific herring has shown an extended collapse. Pacific herring from the Hokkaido-Sakhalin region in the western Pacific was once one of the world's largest fisheries, but collapsed for unknown reasons and has remained depressed most likely due to uncontrolled fishing pressure for more than 40 years (Hay et al. 2001).

Natural environmental factors are also associated with population variability of Pacific herring. However, the reasons and mechanisms for this natural variability are poorly understood. Predation, disease, food availability, intertidal exposure of eggs to air and larval drift are all examples of natural factors contributing to the large natural variability observed in Pacific herring populations (Pearson et al. 1999; Carls et al. 2001b; Marty et al. 2003; Rooper et al. 1999; Sturdevant et al. 2001). Climate, as it affects food availability, may also affect Pacific herring populations (Brown 2002; Schweigert et al. 2002). Brown (2002) noted that trends in abundance of Pacific herring in the northern Gulf of Alaska appear to be in phase with decadal-scale climate indices, which may also affect growth and spawn timing. Schweigert et al. (2002) reported that herring stocks throughout British Columbia and Alaska have shown a decline since the late 1970s that may result from climatic conditions and declining food availability.

In addition, there is evidence that declines in Pacific herring biomass in PWS are attributable to disease (viral hemorrhagic septicemia virus, *Ichthyophonus hoferii)*, and that disease is a significant variable in population fluctuations (Johnson 2002; Marty et al. 2003, 2004; Quinn et al. 2001). Marty et al. (2004) concluded that disease is the most important variable limiting recovery of the Pacific herring populations in PWS and predicted that Pacific herring in PWS will not recover until both viral hemorrhagic septicemia virus and *I. hoferii* are at background levels for several years. These disease factors are common to Pacific herring throughout its range, but appear to have a unique controlling influence on Pacific herring populations in PWS.

7.4 ACTIONS TAKEN OR PLANNED

Because of its importance near the base of the nearshore marine food web and its value as a commercial fishery, numerous projects have been initiated to evaluate and restore Pacific herring populations. Injury and restoration projects include research on the mechanisms limiting recovery, surveys to collect information for fisheries management, investigation of ecological factors governing herring populations, and acquisition of habitat protective of herring.

7.5 SUMMARY, CONCLUSIONS, AND RECOMMENDATIONS

7.5.1 Summary

Pacific herring are of both ecological and commercial importance in PWS. Not only are they central to the marine food web, providing food to marine mammals, birds, and invertebrates, but herring are also fished commercially for food and bait, sac-roe, and spawn on kelp. Despite the fact that a number of studies have been directed at understanding the toxicity of oil to Pacific herring, the cause(s) of the highly depressed population, which has closed the fishery for all but 6 of the 17 years since the spill,[43] are not well understood.

Most life stages of Pacific herring were exposed to EVO to some degree following the initial spill. The spill occurred a few weeks before Pacific herring spawned in PWS—eggs and adults were exposed directly to residual EVO. Pacific herring did not spawn on the most heavily oiled beaches, but they did spawn on shorelines considered lightly to moderately oiled (Brown et al. 1996). Larval contact with EVO may also have occurred given that EVO was distributed in nearshore areas for several months following the spill. Early life stages are the most sensitive to the effects of oil. In addition, EVO was shown by researchers to be available in open water along shorelines where oiling was not directly evident. Carls et al. (2002) indicate that 25–32 percent of the Pacific herring embryos in PWS were damaged as a result of the EVO exposure.

Pacific herring populations in PWS were increasing in the late-1980s before EVOS, and record numbers were reported immediately after EVOS primarily due to the strong recruitment of the 1988 (pre-spill) year class. However, in 1993, the Pacific herring fishery was only 14 percent of the 1992 harvest and the 1989 year class was one of the smallest cohorts ever to return to spawn. The Pacific herring fishery in PWS was closed from 1994 to 1996. The Pacific herring population began increasing in 1997 following the 1993 decline, and the fishery was opened in 1997 and 1998. However, the population increase stalled in 1999, and recent research by Thomas and Thorne (2003) have suggested that the opening of the fishery in 1997 and 1998 stressed the already weakened population and contributed to the 1999 decline, when the fishery was again closed. As of spring 2005, ADFG reported that Pacific herring biomass in PWS remained slightly below the minimum threshold for establishing a Pacific herring fishery in PWS and the fishery remains closed.

7.5.2 Conclusions

It is likely that no single factor explains the population decline. In a synthesis of the literature, Carls et al. (1999, 2001b) surmise that the combination of high population density, poor nutrition, and epidemic infection by viral hemorrhagic septicemia virus caused the 1993 collapse in a boom-and-bust cycle typical of Pacific herring populations in the Gulf of Alaska and elsewhere. Pearson et al. (1999) also concluded that a combination of increasing biomass and decreasing food supply led to poor Pacific herring condition and the 1993 decline. Both studies acknowledged that natural factors and variability could explain the population decrease, and a

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[43] The fishery was open in 1990, 1991, 1992, 1993, 1997, and 1998 (Johnson et al. 2002).

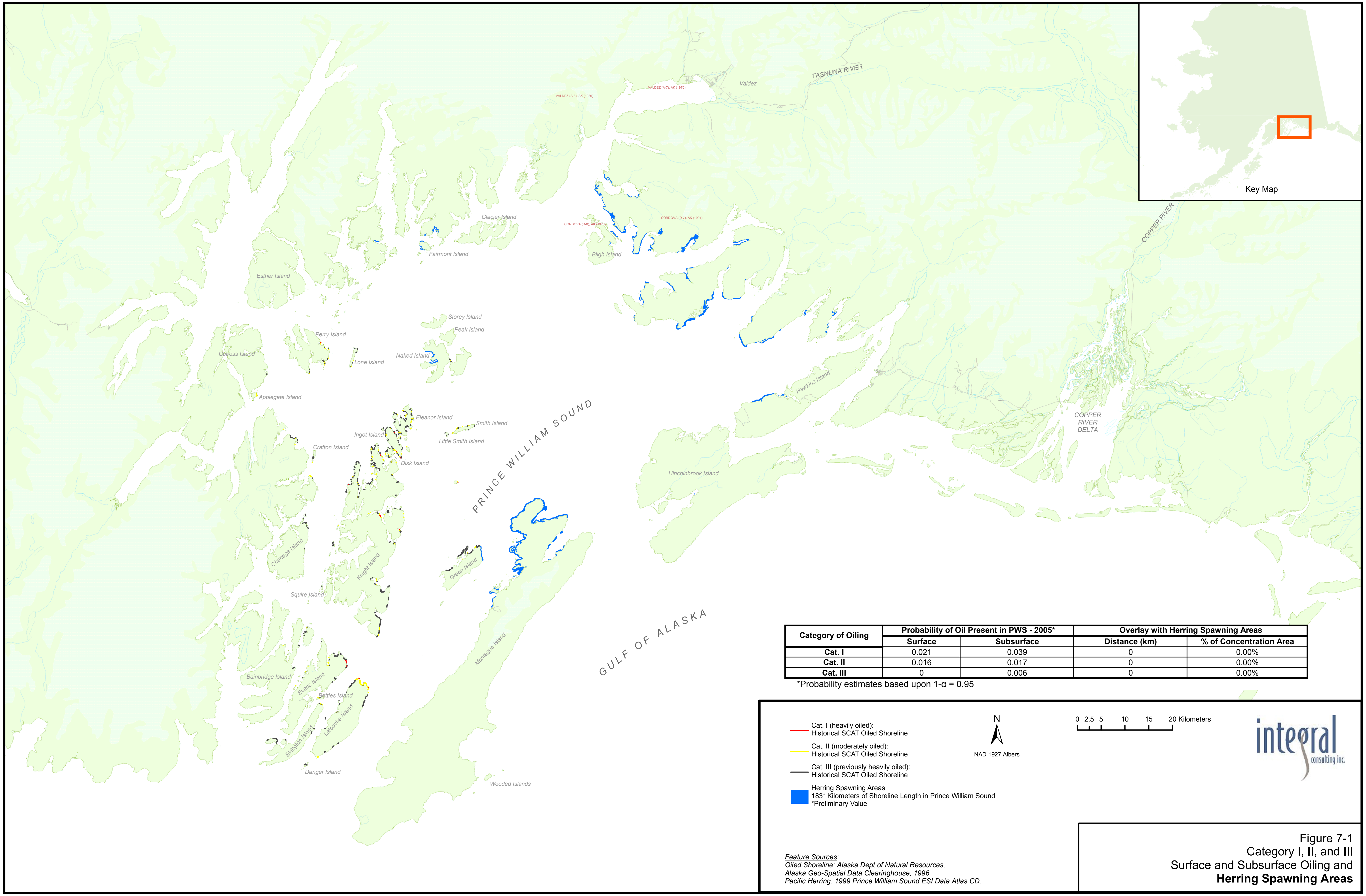
clear link between EVOS and the 1993 population decline was neither established nor eliminated.

Present exposure of Pacific herring to and impacts from lingering EVO are likely minimal. The only route for Pacific herring exposure is during the egg stage when eggs are deposited on beaches. The extent of potential egg exposure to lingering oil is low because the proportion of Pacific herring spawning in areas that were moderately to heavily oiled, and therefore most likely to contain lingering oil, is a very small portion of known herring spawning areas. Although toxicity thresholds and concentration-response curves have been developed for sublethal effects and mortality to eggs and larvae, monitoring data are not available for concentrations of lingering oil in water or in eggs to estimate the likelihood of these effects.

There is little evidence linking the depressed herring population to residual effects from the spill. More recently, an expert group has been convened to address broader issues associated with the depressed herring population in PWS. Preliminary findings of the expert group indicate that the cause of the continued depressed Pacific herring population in PWS is apparently linked to continuing disease problems. The continuing limiting of the population by disease is a unique situation, and while these diseases are ever present in all populations of herring (and other species), they are seldom suspected of limiting populations over an extended period of time. No other North American population has suffered acute population losses like those in 1993, nor has any other population suffered from chronic disease issues.

7.5.3 Recommendations

Additional research may be needed to determine the cause for the current condition of the Pacific herring population in PWS. It will likely be necessary to develop a more holistic approach to the problem, directly addressing issues such as the role of the fisheries management decisions, role of prey and predators, multiple disease factors affecting juvenile and adult fish, and climatic factors such as decadal oscillation or longer term changes.



Category of Oiling	Probability of Oil Present in PWS - 2005*		Overlay with Herring Spawning Areas	
	Surface	Subsurface	Distance (km)	% of Concentration Area
Cat. I	0.021	0.039	0	0.00%
Cat. II	0.016	0.017	0	0.00%
Cat. III	0	0.006	0	0.00%

*Probability estimates based upon 1-α = 0.95

- Cat. I (heavily oiled): Historical SCAT Oiled Shoreline
- Cat. II (moderately oiled): Historical SCAT Oiled Shoreline
- Cat. III (previously heavily oiled): Historical SCAT Oiled Shoreline
- 🟦 Herring Spawning Areas 183* Kilometers of Shoreline Length in Prince William Sound *Preliminary Value

Feature Sources:
Oiled Shoreline: Alaska Dept of Natural Resources,
Alaska Geo-Spatial Data Clearinghouse, 1996
Pacific Herring: 1999 Prince William Sound ESI Data Atlas CD.

Figure 7-1
Category I, II, and III
Surface and Subsurface Oiling and
Herring Spawning Areas

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8. SEABIRDS

The EVOS Trustee Council (2002b) concluded that six seabird species have not yet met the recovery objectives of the EVOS restoration plan:

- Pigeon guillemot (*Cepphus columba*)
- Marbled murrelet (*Brachyramphus marmoratus*)
- Double-crested cormorant (*Phalacrocorax auritus*)
- Pelagic cormorant (*Phalacrocorax pelagicus*)
- Red-faced cormorant (*Phalacrocorax urile*)[44]
- Common loon (*Gavia immer*).[45]

Of the approximately 1 million seabirds inhabiting the oil spill area prior to the spill, it is estimated that between 100,000 and 300,000 were killed initially due to acute oiling effects (Piatt et al. 1990). The common mechanism by which these birds may potentially be exposed to lingering EVO is through consumption of contaminated prey. Some species (e.g., pigeon guillemot, cormorants) may also experience oiled plumage through direct contact with benthic patches of oil. This section evaluates population trends and other parameters to assess the likelihood of ongoing injury to seabirds from the initial impact of EVOS and from lingering oil.

8.1 BACKGROUND

This section describes the natural history of the six seabird species of interest, the initial impacts of EVOS, and the current status of injury and recovery.

8.1.1 Natural History of Seabirds

Pigeon guillemots, marbled murrelets, cormorants, and common loons all spend a significant portion of their life histories in the nearshore waters of PWS. All except loons are seabirds, spending 80 percent of their lives at sea and coming ashore for a few weeks during the breeding season to nest. All six species forage in the nearshore waters of PWS by diving for fish or benthic invertebrates.

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[44] The vast majority (approximately 99 percent) of cormorants observed in PWS are pelagic cormorant (David Irons, pers. comm. to R. Pastorok on January 27, 2006). Any text below that refers in general to cormorants is based on knowledge of pelagic cormorant biology.

[45] Although loons are not specifically seabirds, the similarity of their feeding mechanism and prey preferences allows them to be grouped with seabirds for this discussion.

8.1.1.1 Feeding and Foraging Behavior

Pigeon Guillemot—Pigeon guillemots forage close to shore on mid-water schooling fish and on bottom-dwelling organisms, by diving directly to the seabed and probing with their bills. Optimal foraging habitat is thought to be in water 10–20 m deep, usually over rocky substrate (Ewins 1993). Breeding birds forage within 0.2 to 7 km of the colony (Ewins 1993). They select prey from a wide variety of fish and benthic invertebrates, presumably based on availability (Ewins 1993). The adult diet is primarily Pacific herring (*Clupea pallasi)*, Pacific sandfish (*Trichodon trichodon*), capelin (*Mallotus villosus*), cods (*Gadus* sp.), sculpins (family Cottidae), gunnels (family Pholidae), pricklebacks (family Stichaeidae), and flounders (families Bothidae and Pleuronectidae). Primary invertebrate prey include red rock crab (*Cancer productus*), shrimp (family Pandilidae), and occasionally polychaetes, gastropods, and bivalve molluscs (Ewins 1993). In Alaska, chicks are fed primarily Pacific sand lance (*Ammodytes hexapterus*), Pacific herring, capelin, gadids (family Gadidae), gunnels, pricklebacks, and other fish 6–15 cm in size (Ewins 1993; Golet et al. 2000).

Marbled Murrelet—Marbled murrelets are generalist feeders, consuming a variety of small fish species in addition to invertebrates depending on season and prey availability (Nelson 1997; Kuletz 2005). Murrelets generally forage by diving in shallow (less than 50 m deep) water near shore and take prey from throughout the water column, including the bottom (Nelson 1997). They are usually found feeding within 1 to 2 km of shore; in PWS, the greatest densities of marbled murrelets are found within 1 km of shore (Kuletz 1997, 2005). A large variety of fish is consumed throughout the summer, including but not limited to Pacific sand lance, Pacific herring, capelin, northern anchovy (*Engraulis mordax)*, walleye pollack (*Theragra chalcogramma*), and sea perch (*Sebastes alutus)*, as well as various other smelt and gadid species (Nelson 1997; Kuletz 2005). Primary winter prey include capelin, surf smelt (*Hypomesus pretiosus*), and herring (Nelson 1997), while various invertebrates (e.g., krill and amphipods) make up a smaller proportion of the diet (Nelson 1997; Kuletz 2005). Overall, both winter and summer foraging behavior and associated diet are varied based upon prey availability (Kuletz 2005). Prey requirements are more specific for raising chicks, which require prey of high energy density, such as Pacific sand lance and Pacific herring (Kuletz 2005).

Pelagic, Double-Crested, and Red-Faced Cormorants—Foraging behavior and diets of cormorants are similar to those of the other seabirds discussed here. They are all nearshore feeders, but whereas red-faced and pelagic cormorants tend to forage over rocky bottoms (like pigeon guillemots), double-crested cormorants are more likely to forage over sandy bottoms. Consequently, pelagic cormorant diets in some locations have been found to overlap nearly 100 percent with pigeon guillemots and only 20 percent with double-crested cormorants (Hobson 1997). Adult pelagic cormorants eat a variety of small fish and invertebrates. Chicks are fed mainly fish (Pacific sand lance, gunnels, pricklebacks, and sculpins) and shrimp (Hobson 1997). Red-faced cormorants eat predominantly demersal fish (Causey 2002); their diet includes smelt, Pacific sand lance, flounder, and sculpin.

Common Loon—In the marine environment, common loon foraging activity occurs primarily nearshore, in the upper 5 m of the water column. Activity is often centered over shoals, around islands and outcrops, and at the entrances to tributaries (McIntyre and Barr 1997). Wintering individuals often feed in the same locations each day (McIntyre and Barr 1997). The diet consists primarily of small fish (up to 25 cm/10 in. long), including eels, herring, sand lance, pipefish, goby, and sculpin (McIntyre and Barr 1997). As opportunistic feeders, they will also take aquatic invertebrates such as crustaceans and molluscs.

8.1.1.2 Winter and Breeding Habitats

The northern Gulf of Alaska, including PWS, serves as important breeding range for pigeon guillemots and marbled murrelets, all of which nest on land, close to their coastal feeding areas. Cormorants nest along the seaward edge of PWS and on other coastal islands in the Gulf of Alaska. Common loons breed inland, on freshwater lakes. Breeding season for all species is, roughly, May through August. Cormorants and common loons winter in the Gulf of Alaska, whereas marbled murrelets and pigeon guillemots disperse more widely.

Pigeon Guillemot—In PWS, pigeon guillemots gather in small colonies in May to establish pair bonds and nest. They nest in cavities or burrows on rocky coastlines, often on islands for protection from predators (Ewins 1997). Occasionally, large nesting colonies of 1,000 or more pairs can be found (Ewins 1997). Only 25–50 percent of the summer pigeon guillemot population remains in PWS to overwinter (Ewins 1997); the remainder disperse to unknown wintering areas. In general, they move from exposed coastline to sheltered inshore waters to winter (Ewins 1997), although in PWS they nest in sheltered fjords and bays and migrate in winter. Little else is known about their winter range.

Marbled Murrelet—While the breeding range of the marbled murrelet overlaps with pigeon guillemots and cormorants, their breeding habitat is distinctly different. Throughout most of their range (northern California to Alaska) marbled murrelets nest in coastal forests, laying their single egg on moss-covered branches of large trees. In Alaska, they nest both on the ground and in trees, but tree nesting is believed to be predominant (Piatt and Ford 1993). The northern Gulf of Alaska is a key area of concentration for this species (Agler et al. 1998). It is likely that murrelets nest near good foraging areas (Piatt and Ford 1993). In PWS, radio-tagged murrelets foraged 5–40 km from suspected nest sites, with considerable variability among individuals (Kuletz 2005). Only about 25 percent of the summer population of marbled murrelets in PWS remains in the sound to overwinter (Kuletz 1997).

Pelagic, Double-Crested, and Red-Faced Cormorants—Cormorants typically nest in colonies at sites relatively safe from predators, including small, rocky or sandy islands and rocky cliffs over deep water. Double-crested cormorants breed and winter from the Aleutian Islands to Mexico. They nest in trees or on the ground, usually within 10 km of feeding areas and, preferably on small islands (Hatch and Weseloh 1999). Pelagic cormorants are resident from the Bering Strait to Baja California, including the Gulf of Alaska, often locating their colonies on suitable cliffs of forested, grassy, and rocky headlands and islands (Hobson 1997). The red-faced cormorant has the narrowest range of all six species discussed here. In North America, they are found only in Alaska, extending from a few scattered sites in the Gulf of Alaska, west through the Aleutian Islands. Winter range of red-faced cormorants probably overlaps with breeding range, but is not well-documented (Causey 2002).

Common Loon—The common loon is widely distributed across North America. Although they are freshwater breeders, most loons winter in coastal waters, generally inshore, over shoals, and in sheltered bays, inlets, and channels (McIntyre and Barr 1997). Specific breeding locations of loons wintering in PWS are unknown (EVOS Trustee Council 2002b).

8.1.2 Initial Impact of EVOS

The immediate effect of EVOS was to reduce the size of the local breeding populations of seabirds due to direct mortality of breeding adults for 1989 and possibly future years (Piatt et al. 1990). Of the approximately 1 million seabirds inhabiting the oil spill area prior to the spill, it is estimated that between 100,000 and 300,000 were killed initially due to the acute effects of oiling (Piatt et al. 1990). The most vulnerable species included loons, grebes, sea ducks, and alcids (murres, auklets, puffins, murrelets, and guillemots) because they spend most of their time on the surface of the water, often in dense flocks (Piatt et al. 1990). Oil caused direct mortality either by coating feathers and causing fatal hypothermia or through toxicity of ingested oil. Murres accounted for 74 percent of the injured seabirds recovered after the spill, followed by other alcids (7 percent) and sea ducks (5.3 percent) (Piatt et al. 1990).

Soon after EVOS, the availability of food resources for seabirds in PWS may have decreased with the population declines of herring and Pacific sand lance. Golet et al. (2002) speculated that, given the characteristics of Pacific sand lance, their populations in PWS may have declined in response to EVOS, and subsequently impacted seabird productivity through decreases in high quality forage fish availability. Studies have not been done to confirm this speculation, and pre-spill data on Pacific sand lance populations are lacking.

Pigeon Guillemot—Estimates of pigeon guillemot mortality directly attributable to EVOS vary widely. Piatt et al. (1990) recorded 2.2 percent of recovered oiled bird carcasses as unspecified guillemots (from which a proxy for oil-related mortality of pigeon guillemots could be derived), and Oakley and Kuletz (1996) noted a 43 percent population decline in pigeon guillemots due to the spill and other factors. Pigeon guillemots may have exhibited long-term negative effects of the oil spill through at least 1998 (Irons et al. 2000). Based on pre-spill and post-spill population statistics, impacts of the spill were more pronounced for pigeon guillemots than for any other seabird (Golet et al. 2002; Murphy et al. 1997).

Marbled Murrelet—PWS is a key area of concentration in the distribution of marbled murrelets, the most abundant seabird in PWS (EVOS Trustee Council 2002b). As much as 7–12 percent of the population of marbled murrelets in the spill area died (estimated mortality was between 12,800 and 14,800 individuals) as a direct result of the spill (Kuletz 1996). In addition to direct mortality, murrelets showed evidence of ingestion of oil as shown through analysis of aliphatic liver compounds (Oakley et al. 1996); they may also have been deterred from foraging areas during the cleanup effort as indicated by the reduction of murrelets in areas where cleanup-related boat and plane traffic was high (Kuletz 1996).

Pelagic, Double-Crested, and Red-Faced Cormorants—In 1996, USFWS Alaska Seabird Colony Catalog reported counts of 7,161 pelagic cormorants, 8,967 red-faced cormorants, and 1,558 double-crested cormorants in the oil spill area. These numbers indicate that cormorant populations remain small and EVOS-related injury (exceeding 1,000 acute mortalities) to all three cormorant species was significant.

Common Loon—Loons in the spill area may have suffered high losses relative to their population sizes (Piatt et al. 1990). For example, hundreds of loons were lost from a population that may number only a few thousand (EVOS Trustee Council 2002b).

8.1.3 Current Status of Injury and Recovery Classification

Of the six species of seabirds addressed in this review, none was considered to have met the recovery objectives of the restoration plan (EVOS Trustee Council 2002b). For pigeon guillemots and marbled murrelets, the recovery objectives stated in the 2002 recovery plan are "populations that are stable or increasing". Pigeon guillemots are considered to be "not recovering," due to continued population declines (EVOS Trustee Council 2002b). In the 2002 assessment, the marbled murrelet is considered to be "recovering," due to some indication of stable or increasing March populations, but not fully recovered due to continued summertime population declines (EVOS Trustee Council 2002b). However, March surveys may not be relevant for gauging recovery, because it appears to be a period of early migration for marbled murrelets and the timing of migration might be affected by climate change (Kuletz 2006, pers. comm.). The 2002 recovery plan states that the recovery objective for cormorants and common loons is a return to pre-spill population levels within the oil spill area. The EVOS Trustee Council (2002b) considered common loon to be "not recovered" and cormorant species "not recovering," due to lack of significant increasing population trends.

For each of the six species, the 2002 recovery plan states the following:

Pigeon Guillemot—"Boat surveys have indicated that numbers of guillemots in the summertime continue to decline along both oiled and unoiled shorelines in the PWS through 2000. March surveys reveal no significant trends in abundance although the data appear to suggest a decline at this time of year as well. For these reasons the pigeon guillemot is still considered to be not recovering from the effects of the oil spill." (EVOS Trustee Council 2002b, p. 17)

Marbled Murrelet—"The summertime marbled murrelet population is not stable or increasing, but the March population is stable over time. Marbled murrelet productivity, as measured by surveys of adults and juveniles on the water in PWS, appears to be within normal bounds. Based on these results, it appears that the marbled murrelet is at least recovering from the effects of the oil spill, but clearly has not yet recovered." (EVOS Trustee Council 2002b, p. 15)

Pelagic, Double-Crested, and Red-Faced Cormorants—"More recent surveys (through 2000) have not shown a significant increasing population trend since the oil spill, and, for that reason, these species are considered to be not recovering." (EVOS Trustee Council 2002b, p. 9)

Common Loon—"One year of high counts in the unoiled areas is insufficient to indicate that recovery has started. Thus the common loon is considered still not to have recovered from the effects of the spill" (EVOS Trustee Council 2002b, p. 8)

8.2 SPATIAL AND TEMPORAL CHANGES IN SEABIRD POPULATIONS

Seabird populations in PWS were surveyed before and after the spill (1972, 1984, 1985 and 1989, 1990, 1991, 1993, 1996, 1998, 2000, 2004, 2005, respectively) by USFWS and others (Klosiewski and Laing 1994; Agler et al. 1994; Agler and Kendall 1997; Lance et al. 1999; Murphy et al. 1997; Wiens et al. 2004; Sullivan et al. 2005). Habitat features were characterized and included as a category of variables in some surveys for some years (1991, 1998, 2001; Wiens et al. 2004). Interpretation of the cited surveys has focused on pre- and post-spill patterns in population density and geographic population distinctions in oiled and unoiled parts of PWS. The relationship between seabird populations in oiled and unoiled areas of PWS is discussed in Section 8.3.1. Other factors affecting seabird populations are discussed in Section 8.3.5.

Pigeon Guillemot—Data from Naked Island collected in 1985 suggest that the pigeon guillemot population of PWS was already in decline prior to the spill (Oakley and Kuletz 1996). Most recently, Sullivan et al. (2005) reported on long-term pigeon guillemot population data collected from 1989 through 2004. Based on a combination of statistical tests, the authors report that the pigeon guillemot populations in oiled areas have not demonstrated a trend towards recovery.

Marbled Murrelet—Marbled murrelet populations in PWS and elsewhere have undergone dramatic decreases over the last 30 years; the marbled murrelet is formally listed as a threatened species in California, Oregon, Washington (McShane et al. 2004), and British Columbia (Burger 2002). The summer murrelet (marbled and Kittlitz's murrelets [*B. brevirostris*]) population in the northern Gulf of Alaska declined from an estimated 304,000 in 1972 to 97,000 shortly after the spill (Klosiewski and Lang 1994). Based on a comparison of population surveys in 1972, 1973, and 1985, it appears that 67 percent of the marbled murrelet population decline occurred prior to EVOS (Klosiewski and Laing 1994). On the south coast of Alaska, climatic shifts have caused changes in forage fish abundance, with a reduction in certain high-quality (i.e., high lipid) forage fish, such as capelin, upon which murrelets and other seabirds depend (Kuletz et al. 1997; Agler et al. 1999; Anderson and Piatt 1999).

Sullivan et al. (2005) also reported on long-term population data collected from 1989 through 2005 for murrelets. With the exception of an unusually high population estimate in 1993, the overall murrelet population decreased between 1989 and 2005 (Sullivan et al. 2005). Based on a combination of statistical tests, the authors report that the murrelet populations in oiled areas have not demonstrated a trend towards recovery. However, the nesting distribution and foraging ranges of murrelets makes it impossible to separate murrelets into oiled or unoiled areas, so such designations may be irrelevant for this species. Most significantly, marbled murrelet breeding population declines have been documented in both oiled and unoiled areas. The similar trends throughout PWS suggest that either EVO or some other factor, such as prey availability, is influencing their population (EVOS Trustee Council 2002b). Declines in the herring population of PWS could account for the observed decline in the marbled murrelet population of the sound because juvenile herring are very important for murrelets raising chicks (see Kuletz 2005) and the similarity in rate and amount (percentage) of population decline in herring and murrelet suggests a link (Kuletz 2006, pers. comm.).

Pelagic, Double-Crested, and Red-Faced Cormorants—There were statistically significant declines in cormorant populations in the oiled area between pre-spill (1984–1985) and post-spill (1989–1991) surveys (Irons et al. 2000). Studies on the three species of cormorants (Irons et al. 2000; Klosiewski and Laing 1994) indicated continued population declines through at least 1998. Murphy et al. (1997) found an overall decrease in pelagic cormorant (the only cormorant species sampled) abundance from 1989 through 1991. In addition, Sullivan et al. (2005) reported a statistically significant increase in summer populations of cormorants but a statistically significant decrease of cormorant winter populations.

Common Loon—In 2000, there was statistical evidence of an increase in PWS winter population density of common loons consistent with recovery, but no evidence of summer population recovery (Sullivan et al. 2005 based on data from Irons et al. 2000*)*. Sullivan et al. (2005) tentatively concluded that winter populations of loons may be recovering based on significantly increased winter population density in 2000.

8.3 LINKAGE BETWEEN EVOS AND THE CURRENT CONDITION OF SEABIRD POPULATIONS IN PWS

The current condition of the seabird and common loon populations in PWS can potentially derive from one or more of the following:

- Residual effects from the original spill
- Ongoing exposure to lingering oil
- Other natural or anthropogenic factors that can influence the populations.

The relative importance of these different factors is assessed based on the species life history and inherent ability of the population to recover from the initial impacts of the spill, the nature and degree of ongoing exposure to lingering EVO that could cause adverse effects and injury, and the nature and magnitude of other factors that could affect the condition of the population. The relationship between lingering EVO and seabirds is the central focus of the discussion below.

8.3.1 Residual Effects from the Original Spill

Comparisons of population trends in oiled vs. unoiled portions of the spill area have been made by Irons et al. (2000), Murphy et al. (1997), Klosiewski and Laing (1994), and Wiens et al. (2004). The most comprehensive evaluations of the relationship between EVOS and seabird populations in the years following the spill have been performed by Irons et al. (2000) and Wiens et al. (2004).

Irons et al. (2000) used a methodology known as BACI (before, after, control, impact) on population data collected from 1985 to 1998 from 187 to 212 shoreline transects[46] in PWS. The BACI design assumes that 1) birds in the reference locations were not affected by the spill, 2) bird populations in the spill area and reference area are closed populations, and 3) changes in

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[46] In the 1984/1985 survey, 772 transects were surveyed.

bird density in the reference area reflect changes that would have occurred in the oiled area had the spill not taken place.[47]

Wiens et al. (2004) conducted surveys in 10 bays in 1989–1991, 1996, 1998, and 2001 and assessed potential ongoing impacts based on abundance and habitat change. The authors evaluated bird abundance, as estimated by nearshore and offshore surveys and habitat data: 1) using oil as a quantitative variable with and without habitat measures as covariates to assess spill effects and their changes over time, and 2) using oiling as a categorical variable (oiled vs. unoiled) to conduct between-year, repeated-measures analysis with 1984 data as baseline. Results were combined into a weight-of-evidence interpretation for each bird species.

In addition to the potential effect of EVOS on seabird population trends discussed below, an indirect link between EVOS and seabirds exists through potential effects of the oil spill on the food resources of seabirds. The most persistent effect of concern is the potential relationship between EVOS and the decline in the herring population of PWS. All of the seabird species discussed here depend to some extent on herring as a food resource. For example, juvenile herring are very important for murrelets and other seabirds when raising chicks (see Kuletz 2005). Thus, if a link exists between EVOS and the persistent depression of PWS herring populations, any observed reductions in current seabird populations could be indirectly linked to the spill. However, there is a large amount of scientific uncertainty in the linkage between the continuing reduction of the PWS herring population and the EVOS (see Section 7, “Pacific Herring”). Thus, any indirect linage between seabird declines and EVOS via reductions in the herring food resource is also highly uncertain.

Pigeon Guillemot—From 1989 to 1991, pigeon guillemots decreased in overall abundance, and the decrease was greater in oiled areas than in unoiled areas (Murphy et al. 1997). Pigeon guillemots continued to show statistically significant negative effects in oiled areas in most years through 1998 according to the BACI method (Irons et al. 2000). Wiens et al. (2004) concluded that pigeon guillemot populations in oiled areas were adversely affected relative to unoiled areas, but concluded recovery was complete by 1991.

Marbled Murrelet—Irons et al. (2000) compared July murrelet (*Brachyramphus* sp.) populations pre- and post-spill in oiled and unoiled areas and found that populations increased in oiled areas relative to unoiled areas in 1993 to 1998. Agler and Kendall (1997) compared July population trends for Kittlitz’s murrelets in oiled and unoiled areas from 1989 to 1996. The overall population decreased during these years, but the number in oiled areas increased. Agler and Kendall (1997) speculated that increases in oiled areas relative to unoiled areas may represent a geographical shift in the population, possibly in response to prey distribution. Wiens et al. (2004) concluded that marbled murrelet populations in oiled areas were not adversely impacted relative to unoiled areas.

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[47] The authors noted that assumptions 1 and 2 were probably not met, but pointed out that that would bias the interpretation to be more conservative (i.e., underpredict impacts).

Pelagic, Double-Crested, and Red-Faced Cormorants—Cormorants as a group showed negative responses to oiling in most years from 1989 to 1998 (Irons et al. 2000). Murphy et al. (1997) found an overall decrease in pelagic cormorant (the only cormorant species they sampled) abundance from 1989 through 1991, as well as a greater decrease in oiled vs. unoiled areas. Wiens et al. (2004) concluded that pelagic cormorant populations in oiled areas were adversely affected relative to unoiled areas, but concluded recovery was complete by 1991.

Common Loon—Common loons exhibited declines in population numbers and use of habitat in oiled areas in 1989 but not in 1990 (Day et al. 1997; Irons et al. 2000). Irons et al. (2000) found a weak negative effect of oiling on population numbers again in 1993, but not in 1996 or 1998.

8.3.2 Current Exposure to Lingering Oil

While initial seabird mortalities caused by oil were through direct contact (i.e., coating of feathers with oil and eventual death from hypothermia), or through lethal ingestion, the pathway for exposure to oil that remains in the environment would most likely be through food chain transfer from invertebrate prey (Golet et al. 2002). This pathway has been suggested following the finding of elevated CYP1A in adult pigeon guillemots from oiled sites (Golet et al. 2002). The authors assumed the transfer would be from invertebrate rather than fish prey because 1) fish-eating chicks did not exhibit elevated CYP1A, whereas fish-and-invertebrate-eating adults did, and 2) invertebrates are more likely to sequester PAH compounds and pass them on through the food chain while fish more readily metabolize them (Golet et al. 2002).

Pigeon Guillemot—No studies have been conducted to provide direct evidence of the invertebrate pathway of PAH-transfer to pigeon guillemots or other seabirds. The diversity of fish and invertebrate prey consumed by seabirds, combined with the size of foraging territories (extending to several kilometers offshore) suggests that lingering oil in prey would impact only a fraction of the diet of birds residing in oiled areas. Some local populations may focus on invertebrate prey seasonally; spring and winter preferences for invertebrate prey have been shown in marbled murrelets (Nelson 1997) and pigeon guillemots (Ewins 1993). Some local populations may also have very localized feeding areas; for example, loons often feed in the same location each day (McIntyre and Barr 1997), and the highest densities of murrelets in PWS occur within 200 m of shore (Kuletz 1997).

Studies designed to examine such a link could be done. For example, Andres (1999) looked at the effects of persistent shoreline oil on black oystercatchers in PWS in 1992 and 1993. He examined the diets of oystercatchers nesting on oiled and unoiled beaches, quantified their diets, foraging behavior, and foraging habitat use (including quantifying the amount of foraging done in oiled vs. unoiled mussel beds). He found some differences (such as slower mass gain in chicks and elevated hydrocarbon indices in chick feces from chicks raised on oiled shorelines). Overall, however, he found that the patchiness of lingering oil led to exposure for only a relatively few individuals and concluded that lingering oil presented little risk to the PWS oystercatcher population as a whole.

As a follow-on to the work of Seiser et al. (2002) on activity of liver enzymes in pigeon guillemot, Golet et al. (2002) found significantly higher hepatic CYP1A activity in the livers of pigeon guillemots from oiled parts of PWS compared to pigeon guillemots from unoiled parts, indicating higher exposure to CYP1A inducers (potentially PAHs associated with EVO) in the oiled areas. Even in the oiled populations, however, absolute CYP1A activity was low, indicating a low-level of exposure. The authors attribute this finding to the patchy distribution of lingering oil. Even within the oiled area, there would be differences in the amount of oil ingested based on microsite-specific foraging patterns of individuals. CYP1A was also measured in chicks, and no difference was found between oiled and unoiled sites. Adults would be more likely to be exposed than chicks because their diet includes invertebrates whereas chicks eat primarily fish (Golet et al. 2002). Invertebrates are more likely to sequester toxins whereas fish metabolize them (Golet et al. 2002). More recent CYP1A data collected in 2004 indicate that there is no difference in the CYP1A levels in pigeon guillemots collected in oiled and unoiled parts of PWS (Ballachey et al. 2006).

Golet et al. (2002) suggested that elevated levels of CYP1A in pigeon guillemots inhabiting oiled sites could be due to food chain transfer of PAHs through the invertebrate portion of their diet. There is no evidence to indicate any exclusive or extensive overlap between local pigeon guillemot foraging areas and lingering oil, but by the nature of their feeding habits (as nearshore opportunists) such an overlap could possibly exist. More specific studies examining pigeon guillemot diets and levels of PAHs in invertebrate prey could help answer some of these questions.

Other Seabird Species—Evidence of ongoing exposure of other seabird species to lingering oil related to the EVOS is lacking.

Collectively, the existing information suggests that negligible overlap in both time and space exists between lingering EVO and seabirds. Therefore, Integral has not performed a direct mapping comparison for seabirds like those performed for other selected resources discussed in this report.

8.3.3 Toxicity of Lingering Oil to Seabirds

Pigeon Guillemot—Seiser et al. (2000) measured blood parameters in pigeon guillemot populations of oiled vs. unoiled portions of PWS in 1997. An objective of this study was to identify evidence of toxic responses to oil contamination in both chicks and adults. Based on data for chicks, Seiser et al. (2000) concluded there was little evidence of chicks being impacted by lingering oil. Preliminary data from adults, however, indicated elevated aspartate aminotransferase (AST) activity in oiled populations. This Seiser et al. (2002) finding is an indication of hepatocellular injury, which would be consistent with continued exposure to lingering oil. Seiser et al. (2002) additionally suggest that follow-up research be conducted to evaluate fully the health of adult pigeon guillemots residing in oiled areas.

Golet et al. (2002) also tested blood parameters in guillemots. There were significant differences between birds from oiled vs. unoiled sites, but not consistently across years. For example, guillemots at the oiled site had higher concentrations of the enzyme lactate dehydrogenase (LDH) in both 1998 and 1999 than guillemots at unoiled sites. In 1999, guillemots at the oiled site also had elevated AST activity. The combined effect of elevated LDH and AST is indicative of toxicological response (Golet et al. 2002), but this effect was present only in 1999 and not 1998, when there was no difference between AST levels at oiled vs. unoiled sites (Golet et al. 2002).

Other Seabird Species—Although the toxicity studies were done solely on pigeon guillemots, similar mechanisms of toxicity, if it occurs at all, could exist for any of the other five species considered in this review.

8.3.4 Indirect Effects of Lingering Oil

Pacific sand lance is a high-lipid fish important in the diets of pigeon guillemot and other seabird chicks (Golet et al. 2002). Sand lance burrow in intertidal sediment to avoid predators; they also spawn in fine gravels or sand near shore and exhibit high site fidelity (Golet et al. 2002; Hobson 1986). They are sensitive to oil contamination and avoid contaminated substrate, preferring the water column when sediments are contaminated and thus having greater exposure to predators (Pearson et al. 1984; Golet et al. 2002). Thus, it is possible that lingering oil affects sand lance, which in turn leads to less productivity of seabird populations through limitation of a key food resource. However, no direct evidence of this mechanism is available.

Although declines in the herring population of PWS occurred following the EVOS, these declines are probably not related to lingering oil effects (see Section 7, “Pacific Herring”). Overlap in the distribution of lingering oil and herring habitat is minimal. Indirect effects of the decline in PWS herring populations on seabirds are discussed in Section 8.3.1.

8.3.5 Other Factors Influencing Current Injury or Rate of Recovery or Population Variability

Ocean climate in the Gulf of Alaska cycles between warm and cold regimes approximately every 20–30 years (Anderson and Piatt 1999). In 1977, the regime shifted from warm to cold. This shift was followed by marked changes in fish and shellfish community structure (Anderson and Piatt 1999). For seabirds, this meant a decline in capelin, a high-energy, fatty forage species. By 1988, many seabirds had shown a shift in diet from primarily capelin to primarily sand lance and juvenile pollock (Piatt and Anderson 1995). It also signaled an increase in predatory groundfish (such as cod, flounder, and halibut), which compete with marine birds for smaller forage fish (Anderson and Piatt 1995). Population declines in many marine bird species have occurred coincident with these climatic and ecological shifts in the Gulf of Alaska (Piatt and Anderson 1995; Agler et al. 1999).

The decline in seabird populations of PWS could also be related to decreases in their key food resources (see Section 8.3.1 and 8.3.4 above). Herring are important to murrelets and other seabirds, both for raising chicks (they need 1+ juveniles) and for adults (0+ to 1+ classes) (Kuletz 2005). Golet et al. (2002) discussed the importance of herring and sand lance together as “high-lipid” food resources for guillemots and other seabirds. Whether the decline in herring in PWS is related to EVOS is questionable (see Section 7, “Pacific Herring”). Thus, the link between the declines in seabird populations and EVOS through an indirect effect on their key food resources is highly questionable. Golet et al. (2002) speculated that sand lance populations may have declined in response to EVOS, and subsequently impacted seabird productivity through decreases in high quality forage fish availability. Studies have not been done to confirm this speculation, and pre-spill data on Pacific sand lance populations are lacking.

Pigeon Guillemot—An apparent increase in nest predation of pigeon guillemot chicks and incubating adult birds occurred after the spill (Hayes and Kuletz 1997, Golet et al. 2002). A proposed hypothesis was that, after the spill, predators such as river otters and minks preyed more heavily on nesting guillemots due to heavy oiling of their customary shellfish prey (Hayes and Spencer 1997). Pigeon guillemots are also caught and drowned incidentally in gill nets (Wynne et al. 1991, 1992; Manley et al. 2004).

Marbled Murrelet—Marbled murrelet populations have declined throughout the species' range, and the species is listed as threatened under the U.S. Endangered Species Act. The primary cause is considered to be loss of old growth nesting habitat (McShane et al. 2004), although prey availability also appears to be important to reproductive success (Peery et al. 2004; Kuletz 2005). During the early 1990s, Carter et al. (1995) estimated from limited data that mortality through incidental capture in gill nets killed about 3,300 individuals in Alaska annually. Murrelets were one of the most commonly caught seabirds in gill nets in PWS (Wynne et al. 1991, 1992) and Kodiak (Manly et al. 2003; NOAA, unpubl. data). Population trends projected from demographic analyses suggest that populations are declining throughout their range in North America as much as 4–7 percent per year (Beissinger 1995).

Given the declining trend for marbled murrelet throughout its range and the potential for cumulative impacts from multiple stressors in PWS, it is doubtful that removal of oil spill impacts alone will accomplish the recovery objective listed in the 2002 status assessment.

Other Seabird Species—The factors discussed earlier for all seabirds, including the change in the ocean climate in the Gulf of Alaska and declines in the herring population of PWS potentially linked to the EVOS, may contribute to population trends in cormorants and loons. Specific evidence of the action of other factors on these species is lacking.

8.4 ACTIONS TAKEN OR PLANNED

Immediately after the spill, there were extensive efforts to rescue and rehabilitate oiled seabirds. Subsequent restoration efforts focused on oil cleanup to minimize seabird exposure to oil and research to better evaluate seabird recovery and mechanisms of injury. Some oil spill studies have evaluated links between forage-fish abundance and seabird reproductive success. Other studies have evaluated the toxic effects of oil and the nutritional attributes of different prey in relation to guillemot productivity. Predators that are believed to be responsible for the low densities of pigeon guillemot (i.e., foxes) on two of the Shumagin Islands have been removed to facilitate population growth.

Seabird habitat has also been acquired and protected. For marbled murrelets, restoration activities have addressed the protection of nesting habitat by identifying and acquiring tracts of land and preserving old growth forests and areas where murrelets are known to occur. For pigeon guillemot, nest sites located in forested regions near cliff edges are protected from logging or other coastal development. Predator-proof nest boxes have also been installed for pigeon guillemots, but use has been limited.

8.5 SUMMARY, CONCLUSIONS, AND RECOMMENDATIONS

8.5.1 Summary

Six seabird species have not yet achieved the recovery objectives of the EVOS restoration plan (EVOS Trustee Council 2002b): pigeon guillemot, marbled murrelet, double-crested cormorant, pelagic cormorant, red-faced cormorant, and common loon.[48] Of the approximately 1 million seabirds inhabiting the oil spill area prior to the spill, it is estimated that between 100,000 and 300,000 were killed initially due to the acute effects of oiling (Piatt et al. 1990). Oil can cause direct mortality either through coating of the feathers, leading to hypothermia and then death, or through lethal ingestion.

These birds all spend a significant portion of their life histories in PWS and forage in the nearshore waters by diving for fish or benthic invertebrates. The common mechanism by which these birds may potentially be exposed to lingering EVO is indirectly through prey consumption or, in some cases, through direct contact with oiled intertidal sediments.

Evaluation of seabird populations in terms of their recovery objectives stated by the EVOS Trustee Council (2002b) is challenging. Marbled murrelets and pigeon guillemots will have recovered "when their populations are stable or increasing." Loons and cormorants will have recovered "when their populations return to pre-spill levels in the oil spill area." These recovery objectives do not fully address the overall objective of a *return to conditions that would have existed if the spill had not occurred* because they do not take into account other factors or stressors that can affect population trends. For that reason, the overall objective of a return to conditions that would have existed had the spill not occurred is considered when evaluating seabirds.

Pigeon Guillemot—The life history of the pigeon guillemot suggests that there was a potential for exposure to intertidal EVO post-spill that diminished as lingering oil in the intertidal diminished. Preliminary data from a 2004 study of CYP1A in pigeon guillemots from oiled and unoiled areas in PWS (Ballachey et al. 2006) indicate that there is no longer any elevated exposure to PAH in oiled areas. However, based on the conservative criterion of "increasing population since the spill," Sullivan et al. (2005) suggest that the pigeon guillemot population in PWS from 1989 through 2004 has not demonstrated a trend towards recovery in oiled areas.

Marbled Murrelet—PWS is a key area of concentration in the distribution of marbled murrelets, the most abundant seabird in PWS. The life history of the marbled murrelet suggests

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[48] Although loons are not specifically seabirds, the similarity of their feeding mechanism and prey preferences allows them to be grouped with seabirds for this discussion.

that there was a potential, albeit a relatively low potential, for exposure to intertidal EVO post-spill that diminished as lingering oil in the intertidal zone diminished.

The population of the marbled murrelet in PWS has changed dramatically in the 17 years since the spill. It is likely that stressors other than the direct effects of the spill are currently affecting the population. Indirect effects, due to changes in the murrelet's primary prey (sand lance and juvenile herring) are possible. Irons et al. (2000) and Agler and Kendall (2005) found increasing populations of murrelets in oiled areas. Most recently, Sullivan et al. (2005) reported on long-term murrelet population data collected through 2005. Based on a combination of statistical tests, the authors reported that the murrelet populations in oiled areas have not demonstrated a trend towards recovery. Most significantly, marbled murrelet breeding population declines have been documented in both oiled and unoiled areas, and it is likely some other factor, such as prey availability, which may or may not have been influenced by the oil spill is influencing their population.

Pelagic, Double-Crested, and Red-Faced Cormorants—The life history and foraging behavior of cormorants in PWS suggest that there was a potential for exposure to intertidal EVO post-spill that diminished as lingering oil in the intertidal zone diminished.

Cormorants as a group showed negative responses to oiling in most years from 1989 to 1998 (Irons et al. 2000). Based on the most recent population data collected through 2005 (Sullivan et al. 2005), the populations of cormorants in both oiled and unoiled areas have an overall positive trend since the spill. Murphy et al. (1997) found an overall decrease in pelagic cormorant (the only cormorant species they surveyed) abundance from 1989 through 1991, as well as a greater decrease in oiled vs. unoiled areas.

Common Loon—The life history of the common loon suggests that there is low potential for exposure to lingering EVO in the intertidal zone. Common loons exhibited declines in population numbers and use of habitat in oiled areas in 1989 but not in 1990 (Day et al. 1997; Irons et al. 2000). Irons et al. (2000) found a weak negative effect of oiling on population numbers again in 1993, but not in 1996 or 1998. Based on the most recent population data collected through 2005 (Sullivan et al. 2005), the populations of loons in oiled and unoiled areas in winter have increased steadily since the spill.

8.5.2 Conclusions

It is unlikely that seabirds are continuing to experience any direct adverse effects from EVOS or from lingering EVO in the intertidal zone. There is currently minimal potential for exposure to lingering EVO in the intertidal zone, and it is likely that sufficient time has passed for the populations to recover from the initial acute mortalities caused by the spill. However, potential links between the oil spill and declining populations of sand lance and herring may be having a long-lasting, indirect effect on some seabirds in PWS. Moreover, population modeling to evaluate the effects of EVOS and the recovery potential of seabirds has not been performed.

8.5.3 Recommendations

Additional population monitoring and modeling are recommended to better understand the population dynamics of pigeon guillemots and marbled murrelets as indicators for the group of seabirds considered here. Collection of population survey data should continue for all injured species. Population modeling should focus on the use of age- or stage-structured models of pigeon guillemot to project population densities with and without EVOS. Comparisons between PWS populations and those outside the sound should be made. Residual effects of EVOS should be evaluated by projecting the recovery of the population after the initial acute impact of the spill, assuming no chronic effects of the initial spill or lingering oil. The importance of food availability could also be evaluated by implicitly modeling the effects of decreases in the abundance of high-quality prey (herring and sand lance) by modifying age- or stage-specific fecundity and survivorship appropriately based on monitoring data. One specific issue other than residual effects of EVOS that might be modeled is the potential population-level impact on seabirds from incidental take in gill net fisheries. Additional research to better understand the potential impacts of other stressors (e.g., climatic shifts or trends, predator-prey relationships) would also be beneficial.

9. SEA OTTER

Prior to the EVOS, sea otters populated the nearshore marine environment of PWS. This species had made a dramatic comeback from the brink of extinction in the early 1900s, when fur hunting had reduced their numbers to a few hundred worldwide. By 1985, the estimated population in PWS was 5,800 and at the time of the spill, Garrott et al. (1993) estimated the western PWS sea otter population size at 6,546 and still growing. Hundreds of sea otters became coated with oil in the days following the spill. A total of 871 carcasses were recovered, although many more were estimated to have died (Ballachey et al. 1994; Bodkin et al. 2002), and mortality rates up to 90 percent were observed in heavily oiled areas of the northern Knight Island Archipelago (Bodkin and Udevitz 1994). Despite the high level of initial mortality, reproduction in sea otters after the spill was not impaired with reproductive rates in some of the heavily oiled areas as high or higher than they were before the spill, or as high as in unoiled areas (Bodkin et al. 2002; Garshelis and Johnson 2001). However, not all vital parameters were within the normal ranges that would be indicative of recovery in the years following the spill. Higher than normal mortality in sea otters after the spill, particularly in older adults, was evident through at least 1998 (Monson et al. 2000b). By the mid 1990s, the sea otter population of western PWS showed population growth, but elevated mortality and emigration continued to constrain population growth of otters in heavily oiled areas for at least a decade after the spill (Bodkin et al. 2002). Specifically, there was no evidence of population growth through 2005 in the heavily impacted subpopulation at northern Knight Island (Bodkin et al. 2002; Ballachey and Bodkin 2006). Exposure to lingering oil in nearshore areas, as indicated by expression of the biomarker CYP1A in this subpopulation, provides evidence that oil remained a possible factor affecting recovery through at least 2002 (Bodkin et al. 2003).

9.1 BACKGROUND

This section describes the natural history of sea otter, initial impacts of EVOS, and the current status of injury and recovery.

9.1.1 Natural History of Sea Otters

Sea otters were originally widespread throughout the northern Pacific, inhabiting nearshore marine environments in northern Japan, Russia, Alaska, Canada, the continental U.S., and northern Mexico. Their thick pelts were highly sought after by fur traders, who began hunting them in Alaska in the 1740s (Riedman and Estes 1990). By the end of the 19th century, the worldwide population of sea otters was nearly extinct. In 1911, sea otters came under protection of the International Fur Seal Treaty. Since then, populations have increased and through translocations and dispersal, much of their former range has been recolonized.

A remnant population of sea otters survived the fur harvest in southwestern PWS. The population increased through the 1950s, and began to expand north and eastward through the 1970s. By the time of the EVOS, the sea otter population in western PWS had been established for at least 25 years. Before the spill occurred, this population was likely near equilibrium density, and limited by prey availability, although some habitat in northwest PWS was below carrying capacity (Estes et al. 1981; Garshelis et al. 1986).

9.1.1.1 Food and Foraging

Sea otters forage in the benthos of rocky and soft-sediment communities, as well as within the algal understory and canopy (Riedman and Estes 1988). They forage most often in subtidal zones, although the intertidal is used as well (Riedman and Estes 1988). In PWS, the percent of foraging dives varies among individuals from less than 5 to more than 30 percent (Ballachey and Bodkin 2006).

Sea otter prey preferences are largely determined by the availability of prey species, which varies with geographic location and habitat and the length of time that the area has been occupied by sea otters (Riedman and Estes 1988). Sea otters prey on calorie-rich sea urchins where they are available, particularly in rocky habitats, but studies have shown that they can deplete an area of their preferred prey and then switch to other, less energetically profitable prey (Kvitek et al. 1993). Studies in PWS have shown clams (including *Saxidomus giganteus* and *Mya truncata*) to be their primary prey, as well as the most abundant food resource (Riedman and Estes 1988; Dean et al. 2002). It has been reported that 60–70 percent of the diet is clams, while mussels constitute 10–20 percent (Garshelis et al. 1986; Doroff and Bodkin 1994; Dean et al. 2002). In specific areas, however, mussels may account for a larger proportion of the diet. Estes et al. (1981) found mussels and clams equally common in diets at Green Island, an area long occupied by sea otters and possibly depleted in its clam population. In 1991, juvenile sea otters in PWS foraged more frequently on mussels in the intertidal zones, whereas adults foraged more on subtidal clams (Doroff and Bodkin 1994). During 1996–1997, clams accounted for 72–80 percent and mussels 9–14 percent of the sea otter's diet in PWS.

9.1.1.2 Reproduction

Female sea otters begin breeding as early as age 2, and by age 3 most females are reproductively mature (Bodkin et al. 1993). Pups are usually born singly and can be born in any month; however, a peak in pupping occurs in PWS in April and May (Ballachey et al. 2003). Within about 6 months, pups are independent of their mothers. Dominant adult males establish and defend territories that provide access to females (Riedman and Estes 1990). Male territories vary in size, averaging about 75 acres (Riedman and Estes 1990). Some males have been observed to move between their breeding territory and locations of all-male aggregations (Garshelis et al. 1986). Non-territorial males live in those same all-male aggregations, which are often established adjacent to areas where reproductive females and territorial males occur (Riedman and Estes 1990). Home range or use estimates for otters from the northern Knight Island subpopulation average 23 km^2 for males and 21 km^2 for females (Ballachey and Bodkin 2006).

9.1.2 Initial Impact of EVOS

Sea otters are considered particularly susceptible to the effects of oil spills (Davis et al. 1988, Bodkin and Ballachey 1997). Unlike seals that have blubber for insulation, sea otters rely on air trapped within their thick pelage to maintain a constant body temperature. Sea otters also have a high metabolism and spend nearly all of their time in the water foraging and resting. Many sea otters were in the path of the oil as it spread southwest from Bligh Reef into bays and around islands of western PWS. Of the 871 recovered carcasses, 493 were collected from PWS, 181 from the Kenai Peninsula, and 197 from the Kodiak Island/Alaska peninsula area (Ballachey et al. 1994).

Hundreds of sea otters became coated with oil in the days following the spill. The 871 carcasses that were recovered included some that died prior to the spill but total mortality was underestimated to the extent that not all carcasses were recovered (Ballachey et al. 1994; Bodkin et al. 2002; Garshelis 1997; DeGange et al. 1994). Initial mortalities were due to acute injury (primarily oil coating leading to hypothermia, stress, and specific pathologies as described in Section 9.3.3; Lipscomb et al. 1993, 1994). In PWS, Garrott et al. (1993) estimated 40 percent (2,650 sea otters) of the population was lost to acute mortality, and that the pre-spill sea otter population size in oiled areas of western PWS was 6,546. However, using different assumptions, Garshelis (1997) estimated that fewer than 1,000 otters died. It is likely that the true number will never be known. Due to the convoluted nature of the shorelines, some otters survived by finding refuge in bays and inlets that escaped oiling or were only lightly oiled (Garrott et al. 1993). A massive effort to rehabilitate injured sea otters took place in the weeks following the spill, although survival of rehabilitated and released otters was relatively low (Monnett et al. 1990; Bodkin and Ballachey 1997).

9.1.3 Current Status of Injury and Recovery Classification

The 2002 recovery plan states as the recovery objective "[s]ea otters will have recovered when the population in oiled areas returns to its prespill levels and distribution, and when biochemical indicators of hydrocarbon exposure in otters in the oiled areas are similar to those of otters in unoiled areas" (EVOS Trustee Council 2002b). The Trustee Council concludes "[s]ea otter recovery is underway for much of the spill-affected area, with the exception of subpopulations at the most heavily oiled bays in western PWS. For this reason, sea otters continue to be in the recovering category."

9.2 SPATIAL AND TEMPORAL CHANGES IN SEA OTTER POPULATIONS

The sea otter population of western PWS was thought to be near equilibrium density prior to the spill (Estes et al. 1981; Garshelis et al. 1986). Garrott et al. (1993) estimated that the population of western PWS grew only 2 percent per year between 1985 and 1989. After an estimated loss of about 40 percent of the western PWS population following the spill in 1989, the population was slow to begin recovery, with no observed growth in 1990 and 1991 (Garrott et al. 1993).[49]

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[49] Raw numbers of estimates of the pre-spill population size and the number of mortalities in western PWS spill vary greatly between researchers (e.g., Garrott et al. [1993] and Garshelis [1997]). However, it is widely agreed that the decrease in sea otter population following the spill was extensive and widespread.

But after 1993, the sea otter population in western PWS began to increase at a rate about one-half the estimated long-term pre-spill rate of increase (Bodkin et al. 2002). From 1993 to 2000, the population increased by about 600 otters, representing an annual growth rate of approximately 4 percent (Bodkin et al. 2002). However, within some of the most heavily oiled areas on northern Knight Island, numbers of sea otters were well below pre-spill levels, with no population growth observed (Bodkin et al. 2002), and since 2002 the population there had declined significantly, in contrast to the larger western PWS area (Ballachey and Bodkin 2006).

Given the discrepancy in estimates of population loss, it is useful to look at growth rates. Garshelis and Johnson (2001) estimated an annual growth rate from 1990 to 1996 of 2.5 percent in western PWS. Bodkin et al. (2002) report an annual growth rate from 1996 to 2000 in western PWS of 5 percent. They conclude that the western PWS population growth rate is lower than expected, based both on the observed growth rate of 10 percent throughout most of the 20th century (Bodkin et al. 1999) and population modeling predictions of 9 percent growth by Garrott et al. (1993) and 10–14 percent growth by Udevitz et al. (1996). The 9 percent growth rate suggested by Garrott et al. (1993), however, is for the entire PWS and is an average of higher growth rates in the more recently colonized northeast with an estimated 1985–1989 growth rate of 2 percent in the spill area. Thus, the observed annual growth rate of 5 percent in the spill area between 1996 and 2000 is higher than the estimated pre-spill growth for that area from 1985 to 1989 and appears in keeping with expected growth rates, given that they are lower in the longer-occupied western PWS than the more recently colonized eastern PWS. The longer term rate of change in western PWS between 1993 and 2005 is about 2 percent (Ballachey and Bodkin 2006).

The subpopulation of sea otters on northern Knight Island has shown no evidence of population growth from 1993 through 2000 (Bodkin et al. 2002) and has demonstrated a significant negative population trajectory during the period 1993–2005 (Ballachey and Bodkin 2006). This area received heavy oiling and sea otter mortality after the EVOS approached 90 percent (Bodkin and Udevitz 1994). From 1993 to 2000, the mean population size was 77, and from 2003–2005 the mean population size declined significantly to 39 individuals. A mark-resighting study from 1996 to 1999 showed retention rates of otters at Montague Island three times as high as on northern Knight Island (Bodkin et al. 2002). The Montague Island subpopulation had been stable for many years, while the Knight Island subpopulation appears to never have regained stable population structure (age and sex distribution) after being greatly reduced by the oil spill. Food limitation has been ruled out as the probable cause of lack of population growth, because food resources are at least as plentiful for the northern Knight Island subpopulation as for the Montague Island subpopulation (Dean et al. 2002; Bodkin et al. 2002). Nevertheless, Bodkin et al. (2002) suggest that it is possible that interactions between food availability and chronic exposure to oil contamination are influencing sea otter mortality and may contribute to the lack of recovery at Knight Island.

A closer look at the known population dynamics on northern Knight Island provides some insight into differences between oil-impacted and unimpacted otter populations. Data reported on population structure show that in 1996–1998, there was a lower than usual proportion of females in the population, and that 48 percent of the females were in the 0–3 age class (Bodkin et al. 2002). Also, in 1996, a bachelor group of 26 males immigrated into the area, but were no longer there in 1997. These data indicate that the northern Knight Island subpopulation has not achieved a stable structure. Many of the females were below reproductive age and in 1996, one-third of the population consisted of young non-territorial males. Given the unstable dynamics, the population would not be expected to grow at the same rate as the stable Montague Island subpopulation.

Monson et al. (2000a) looked for evidence of long-term effects of EVOS by examining the age distribution of ages at death from sea otter carcasses collected in western PWS between 1989 and 1998. They compared the age distribution of post-spill mortalities to that of pre-spill mortality. Pre-spill carcasses were collected from Green Island in 1977–1985. They also compared pre-spill age-distribution to three time periods of post-spill mortality, 1989 post-spill, 1990–1991, and 1992–1998, using the Kolmogorov-Smirnov two-sample test. The post-spill mortality age distribution in 1989 differed significantly from pre-spill; the post-spill mortality age distribution in 1990–1991 also differed significantly from the pre-spill distribution. The 1992–1998 data did not differ significantly from the pre-spill data, nor did it differ significantly from the 1990–1991 data, but it was significantly different from the 1989 post-spill distribution, suggesting a gradual return to the pre-spill pattern (Monson et al. 2000a).

Because of the small number of animals at northern Knight Island, a small amount of predation (or an additional source of mortality) on this population (i.e., loss of three animals annually) would also have a greater impact on the Knight Island subpopulation than on larger populations that could better absorb the loss (Bodkin et al. 2002). Given all of the unanswered questions surrounding social dynamics and other influences on the Knight Island subpopulation, the mechanism underlying its lack of growth is far from clear, but is most likely due to elevated mortality and emigration rates potentially linked to chronic exposure to lingering oil (Bodkin et al. 2002).

9.3 LINKAGE BETWEEN EVOS AND THE CURRENT CONDITION OF SEA OTTER POPULATIONS IN PWS

The current condition of the sea otter population in PWS can potentially derive from one or more of the following:

- Residual effects from the original spill
- Ongoing exposure to lingering oil
- Other natural or anthropogenic factors that can influence the population.

The relative importance of these different factors is assessed based on the sea otter life history and inherent ability of the population to recover from the initial impacts of the spill, the nature and degree of ongoing exposure to lingering EVO that could cause adverse effects and injury, and the nature and magnitude of other factors that could affect the condition of the population. A great deal of research has focused on the relationship between lingering EVO and sea otters, and this work is the central focus of the discussion below.

9.3.1 Residual Effects from the Original Spill

Studies to assess damage to sea otters from long-term exposure to oil have been focused on examination of population growth rates, mortality rates and mortality age-distribution, reproductive success, and juvenile survival (see Section 9.3.3). Due to limited pre-spill data, many studies relied heavily on comparisons between sea otters from oiled portions of PWS and sea otters from areas not affected by the oil spill. Some pre-spill data were available on sea otter abundance and mortality, but data specific to western PWS are particularly lacking for the years immediately prior to the oil spill (1986–1989).

Two modeling efforts projected recovery rates for the sea otter population in PWS following the spill. Garrott et al. (1993) used the very simple method of applying the pre-spill annual growth rate of 9 percent, and estimated at least 3 years but more likely 5 years would be needed for the population to recover to pre-spill abundance. This simple model applied to all of PWS, and assumed that all sea otters contribute equally to growth, did not consider potential ongoing exposure, and did not address chronic effects that may influence other important population parameters. Garrott et al. (1993) acknowledged that population growth following the spill was not evident based on post-spill surveys through 1991.

Udevitz et al. (1996) applied an age-specific population model to western PWS and predicted recovery times ranging from 10 to 23 years, depending on assumptions regarding survival rates. Like the Garrott et al. (1993) projection, the Udevitz et al. (1996) model did not incorporate potential chronic effects related to the spill into projected recovery times.

9.3.2 Current Exposure to Lingering Oil

Exposure of sea otters to lingering oil has been evaluated in terms of 1) home ranges and feeding behavior and their relations to lingering oil, and 2) biomarker measures of PAH exposure.

9.3.2.1 Feeding Behavior

Exposure to lingering oil for sea otters would occur primarily through ingestion of contaminated prey such as clams and mussels, or through exposure to oil resuspended in the water following some source of disturbance (Bodkin et al. 2002). The overlap between nearshore sea otter foraging habitats and likely locations of lingering oil in the intertidal zone supports this assumption.

Because of small home range size and strong fidelity to home ranges, sea otters inhabiting areas with lingering oil would be expected to encounter oil that remained in foraging habitats, including the intertidal zone. In the decade following the spill, many studies were conducted in western PWS to measure sea otter survival, reproduction, and mortality, but this information was never linked to any measured exposures to lingering oil at the level of the individual. In 2002, researchers began collecting data that would help quantify the frequency and duration of sea otters' use of habitats that likely still contained lingering oil (i.e., the intertidal of previously heavily oiled sites). Using Knight Island as a study area, USGS researchers looked at foraging habits of 16 individual sea otters, including measurement of the proportion of foraging time spent in the intertidal zone, where lingering oil is known to occur (Ballachey and Bodkin 2006). They found that sea otters conducted most of their foraging in the subtidal zone, with an average of 13 percent of their foraging occurring in the intertidal zone. Foraging primarily for clams, this translates to digging about 7 intertidal pits per otter per day, or an average of 2,500 (range <200 to >10,000) intertidal pits per otter per year. Preliminary analyses failed to detect significant correlations with the amount of intertidal foraging and CYP1A levels in individuals. However, the study demonstrates the propensity for individual otters to contact intertidal sediment frequently. Ballachey and Bodkin (2006) also discovered that the proportion of foraging in the intertidal zone was significantly less in the summer, when biomarkers of oil exposure were obtained, than in the spring.

9.3.2.2 Biomarker Measurements

A 3-year study was conducted from 1996 to 1998 to examine the biomarker levels in blood of PWS sea otters (Ballachey et al. 2002). Levels were compared between otters from areas that were oiled by EVOS and otters from nearby unoiled locations. The study also compared the general health of the two populations through blood serum chemistries and body condition analysis. The oiled population was sampled from northern Knight Island and Naked Island; the unoiled population was sampled from Montague Island.

Measurement of activity of CYP1A has proven to be a sensitive and specific biochemical indicator of exposure to PAHs. Selected PAH compounds induce CYP1A activity, which can then be used to assess the degree of exposure to PAH.[50] RT-PCR[51] testing was conducted on sea otter blood samples in 1996, 1997, and 1998. In all three years, RT-PCR results demonstrated greater induction of CYP1A for the oiled population than the unoiled population. Analysis of variance on ranks of values showed the area effect to be significant ($p < 0.001$) while age, sex, year, and capture method were not. CYP1A levels generally declined from 1996 to 1998, suggesting that CYP1A in sea otters corresponded to declining residual oil in the environment (Ballachey et al. 2002). Preliminary results demonstrating the bioavailability of lingering EVO and its ability to induce CYP1A have been recently demonstrated by Short et al. (2005), who compared the potential for CYP1A induction by PAHs collected by SPMDs deployed in different areas of PWS where different types of petroleum contamination were known to occur. Extracts were injected into trout fry (Springman et al. 2005) and EVO was demonstrated to be a potent inducer of CYP1A, unlike samples collected from human use sites or other area-wide sources.

Additional studies were conducted in 2001–2005 (Ballachey and Bodkin 2006). Data from these studies show an apparent trend of convergence of CYP1A levels in the two populations, with no difference by 2005, which would be consistent with diminishing persistence of EVO in the environment.

The 1996–1998 study also compared blood serum enzymes for indication of general health differences between the oiled and unoiled populations. Significantly higher levels of gamma

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[50] PCBs can also induce CYP1A.

[51] "RT-PCR" is a reverse-transcriptase-polymerase chain reaction that quantifies mRNA for CYP1A.

glutamyl transferase (GGT), a blood serum enzyme used as an indicator of liver function, were found in sea otters from oiled areas ($p < 0.001$) in all three study years, 1996, 1997, and 1998 (Ballachey et al. 2002). None of the other serum enzymes (AST, ALT, AP, and LDH) were elevated. In follow-up studies conducted in 2001–2005, the pattern continued. Few to none of the sea otters from the unoiled population had high GGT, whereas about 20 percent of those from the oiled population had relatively high (defined as >20 U/L) GGT levels (Ballachey and Bodkin 2006).

9.3.3 Toxicity of Lingering Oil to Sea Otter

9.3.3.1 Acute and Chronic Toxicity to Individuals

Necropsies were performed on many of the sea otter carcasses recovered after the spill to examine the pathological response to oil. Many different agencies were involved in the rehabilitation and necropsy of sea otters, but a lack of coordination led in some cases to incomplete collection and documentation of data. Nevertheless extensive pathological examinations were conducted on a large sample of carcasses (Lipscomb et al. 1993, 1994).

A general picture of acute toxicity from the initial oil spill has been generated based on 214 carcasses of sea otters that died in the spill (carcasses recovered from the wild rather than those that died in captivity). Of these carcasses, 152 were oil-coated, and 62 were not oil-coated. Cause of death for many of the sea otters was shock. Based on necropsy data collected, Lipscomb et al. (1994) proposed that a series of events took place. Heavily oiled otters initially became hypothermic. Some otters died of hypothermia, while many did not. Those that survived the immediate effects of hypothermia by grooming themselves of the oil undoubtedly ingested large amounts of it. These otters stopped feeding to groom, and body fat stores became depleted. Oil exposure (likely through inhalation or ingestion) led to interstitial pulmonary emphysema. Of 152 oiled carcasses recovered from the wild, 100 (66 percent) had interstitial pulmonary emphysema while only 13 (21 percent) of unoiled carcasses recovered had pulmonary emphysema. The extreme stress on the otters from all of the preceding events led to gastric erosions which resulted in gut hemorrhage. Of the oiled carcasses, 83 (55 percent) had gastric erosions and hemorrhage, while only 4 (6.5 percent) of the unoiled carcasses had gastric hemorrhaging. Finally, the oiled otters went into shock, followed by death.

Chronic effects of either the initial oiling or lingering oil have not been clearly established. Some indicators of chronic effects have been measured. In 1992, wild sea otters from the spill area, compared to counterparts from an unoiled area, were found to have differences in blood serum chemistry that were consistent with liver damage (Ballachey et al. 2003). This finding indicates there were some otters that survived the initial oiling, but suffered chronic effects and, possibly, premature mortality. Studies of river otters have shown that low levels of petroleum hydrocarbons in the diet were associated with anemia and increased metabolic costs (Duffy et al. 1994). Continuing studies (1996–2005) have found that blood values have largely returned to normal levels, but a low proportion of otters still show elevated serum enzymes indicative of liver dysfunction (Ballachey and Bodkin 2006). From 2001 to 2005, livers of sea otters in western PWS were examined grossly and biopsied using endoscopy (Ballachey and Bodkin 2006). Generally, livers of sea otters from the oiled areas of northern Knight Island were observed to be slightly swollen and pale, and these changes were seen to a lesser extent in sea otters captured at Montague Island. However, when biopsies were examined for histopathological changes, few differences were noted between the two regions.

Monson et al. (2000a) used data from sea otter carcasses collected between 1989 and 1998 to develop demographic models with varying survival rates from year to year. Each model was run for 9 years, simulating the post-spill years 1990–1998. For each simulation, they compared the predicted age-distribution of otters dying in each year with those actually recovered from the field, and used maximum likelihood methods to determine the most likely patterns of change. The pattern predicted by the models and supported by the data, shows that the youngest animals suffered the greatest mortality after the spill, but their survivorship increased in the years following the spill (Monson et al. 2000a). The older animals, on the other hand, showed greater and greater rates of mortality as the years went by. These long-term effects were strongest on animals that were 4–5 years old at the time of the spill. The models also showed that through the mid 1990s, there were likely reduced survival spill effects on otters born after 1989, suggesting either maternal influences or exposure to lingering oil in the environment. Analysis of age-at-death data collected in western PWS since 1998, using the models developed by Monson et al. (2000a), further indicate that the anomalous mortality patterns continue at least through 2005 (Ballachey and Bodkin 2006). Advances to the earlier Monson et al. (2000a) models describe a pattern of higher survival among a peripheral or “less spill affected” area in PWS that is contributing individuals to a “core” area, such as Knight Island, where mortality is higher (Ballachey and Bodkin 2006).

Ballachey et al. (2003) compared first-year survival of sea otter pups between oiled and unoiled parts of PWS in 1992–1993. Survival was significantly higher (0.74 vs. 0.52, p=0.05) in pups from eastern PWS than those from western PWS. An earlier study by Rotterman and Monnett (1991) also found lower post-weaning survival of sea otter pups in oiled western PWS, compared to those from unoiled areas in eastern PWS. Ballachey et al. (2003) noted that sea otters (both adults and pups) from eastern PWS were in significantly better condition (when comparing the condition variable of mass/length) than western PWS sea otters. A few aspects of the study make it difficult to draw a conclusive link between juvenile survival and oiling.

First, cause of death could not be established for most of the pups that died. The few that were subject to necropsy were found to have suffered from either malnutrition or trauma, and evidence of fighting (presumably among conspecifics) was apparent, but no link was made to pathologies that would indicate oil contamination. Second, the western PWS sample may not have been representative of the oil-spill-affected population (Ballachey et al. 2003). Because of low sea otter densities, Ballachey et al. were not able to obtain sufficient numbers of animals from the most heavily oiled portions of PWS, and thus sampled otters from areas that were not as strongly affected by oil according to the Alaska Shoreline Fall 1989 Oiling Survey Map (NOAA). Third, the two populations varied in duration of occupation. The eastern PWS was more recently colonized (1970s) than the western PWS (1950s). Sea otters are known to reduce the density and mean sizes of their prey over time (Kvitek 1993), and greater foraging efficiency has been noted in areas recently occupied by sea otters than areas with longer established populations (Garshelis et al. 1986). More recently occupied areas would be expected to have greater food resources, possibly leading to better survival.

A possible link to oil contamination from this study is the finding of elevated levels of certain blood serum enzymes and other blood chemistry parameters in the western PWS population similar to those that have been linked to oil contamination in laboratory tests (Ben-David et al. 2000; Mazet et al. 2000). There was no relationship, however, between blood variables and subsequent survival of juvenile otters (Ballachey et al. 2003). Thus, the study provides evidence of lower juvenile survival in the western PWS population, but the effects of oil cannot be isolated from other potential area effects.

9.3.3.2 Reproductive Rates within Populations Relative to Reference

Reproduction is not a likely limiting factor and does not vary among populations demonstrating different growth rates (Monson et al. 2000b). Garshelis and Johnson (2001) compared pre- and post-spill pup production in western PWS, using data that had been collected in 1977–1985, 1990–1991, and 1993–1996. They chose three sites in western PWS: Applegate Rock, Green Island, and Montague Island. Only Montague Island was not directly affected by EVO. At Applegate Rock, the most heavily oiled of the sites sampled, the pup ratio (number of pups to number of adults) was significantly higher in 1990 than pre-spill. Pup production was generally greater than or equal to the pre-spill values in all of the post-spill samples. There was no significant difference in pup ratios between pre-spill and 1990 at unoiled Montague Island or heavily oiled Green Island.

There was no difference in reproductive rates between otters on oiled northern Knight Island and otters on unoiled Montague Island in 1995 to 1997 (Bodkin et al. 2002). In fact, reproductive rates (as measured by pup to adult ratios) exceeded those reported from other parts of Alaska and Russia.

9.3.4 Indirect Effects of Lingering Oil

Lingering oil impacts to sea otter food supplies could occur through either contamination of prey or a decrease in prey populations. Clams (particularly butter clams *Saxidomus giganteus*) are of particular interest, because clams make up about 70–80 percent of western PWS sea otter diets and butter clams are a preferred prey species. Mussels are also of interest, because they make up 10–20 percent of adult sea otter diets, and possibly much more for individual adults and most juveniles (Doroff and Bodkin 1994; Dean et al. 2002).

Several studies have examined hydrocarbon levels in clams and mussels from the oiled area (Carls et al. 2004; Trowbridge et al. 2002). Overall, clams sampled from the intertidal of oiled beaches had elevated levels of hydrocarbons after the spill, while clams sampled from subtidal locations did not. Blue mussels found in the intertidal zone of oiled beaches were also shown to have elevated levels of hydrocarbons in their tissues for at least 2 years after the spill; by 2002, these levels were no longer elevated over background.

Doroff and Bodkin (1994) found no significant difference in mean hydrocarbon concentrations in clam tissue from subtidal areas of two oiled sites compared to an unoiled site in 1991. A previous study had shown elevated levels of hydrocarbons in mussels located at the intertidal portion of one of the oiled sites (Green Island) in 1989 (Andres and Cody 1993).

Summaries of recent studies of PAH bioavailability and accumulation in bivalves are presented in Section 4 for clams and Section 5 for mussels. In general, PAH concentrations in mussels have declined to background levels in recent years and no longer represent a substantive pathway to EVO exposure to sea otters. Recent data for clams are limited and suggest that clam populations located near patches of lingering EVO may have slightly elevated TPAH concentrations in comparison with those in reference locations.

Based on these studies, there is more evidence of lingering oil in bivalves from the intertidal than the subtidal zone. Sea otters forage primarily in the subtidal zone (Bodkin et al. 2004; Ballachey and Bodkin 2006). The risk of contamination from lingering oil, then, would be highest for juveniles and certain individual adults that focus their predation in intertidal habitats (Doroff and Bodkin 1994; Bodkin et al. 2004; Ballachey and Bodkin 2006).

While some studies have shown slower growth rates and higher mortality of clams in heavily oiled areas (Fukuyama et al. 2000), a comparison of food resources between heavily oiled, northern Knight Island and unoiled Montague Island measured more (but not significantly more) prey energy available to sea otters in the oiled area (Dean et al. 2002). However, sea otters required significantly less foraging time, and juvenile female sea otters were in significantly better condition (weight/total length) at northern Knight Island than at Montague Island (Dean et al. 2002). They concluded that the greater foraging efficiency and female condition could be attributed to the effects of reduced sea otter density on their prey populations, resulting in greater prey densities and mean prey sizes (Dean et al. 2002).

9.3.5 Other Factors Influencing Current Injury or Rate of Recovery or Population Variability

Hatfield et al. (1998) report that killer whales were preying on sea otters in PWS in the early 1990s. There are no data quantifying the impact of this predation on sea otters in PWS. The amount of predation, as well as removals by subsistence hunting, would be useful information to help evaluate the current circumstances of the sea otter population in western PWS. Killer whales have been implicated in sea otter population declines in the Aleutian Islands (Estes et al. 1998). A small number (as few as three or four) of otters lost to predation or harvest on Knight Island could have a significant impact because of the small size of the population (Bodkin et al. 2002). However, evidence contrary to predation or harvest as the sole cause of sea otter decline comes from beach-cast carcasses, an indication of mortality from other sources than predation (Monson et al. 2000a; Bodkin et al. 2002). In the juvenile sea otter survival study (Ballachey et al. 2003), most of the carcasses were not recovered, and some could have been lost to predation.

It is possible that cascading social or cultural effects resulting from the loss of specific age or sex components to the population, or disruption to the social structure, may continue to affect recovery of sea otter populations in ways similar to those proposed for the PWS killer whale groups and that may occur with other social species of mammals (e.g., wolves). However, these avenues of injury would be difficult to assess and they have not been addressed as part of post-spill studies.

9.4 ACTIONS TAKEN OR PLANNED

A massive effort to rehabilitate injured sea otters took place in the weeks following the spill, although survival of rehabilitated and released otters was relatively low. The restoration strategy for sea otters has included development and application of aerial survey methods and a spectrum of research to examining feeding behavior and potential causes of delayed recovery.

In 2004, Bodkin (2004, pers. comm.) recommended the following actions to further evaluate recovery of the sea otter population of western PWS:

- Monitor decline of oil in sediments
- Continue surveys in western PWS and Knight Island through recovery
- Model recent survival patterns through analysis of ages at death
- Assay biochemical markers in sea otters and historical samples
- Synthesize information on chronic losses compared to acute losses.

9.5 SUMMARY, CONCLUSIONS, AND RECOMMENDATIONS

9.5.1 Summary

Sea otters are found throughout PWS, where they live in the nearshore environment and forage on subtidal and intertidal communities of clams, mussels, crabs, and sea urchins. Sea otters are considered one of the most vulnerable of marine animals to the effects of oil spills. Many sea otters were in the path of the oil as it spread south from Bligh Reef into bays and around islands of western PWS. Initial mortalities were due to acute injury with perhaps 2,000–3,000 mortalities throughout the spill area. Because sea otters forage in the intertidal and subtidal zones, digging up to several thousand intertidal pits per year, they have a high potential for exposure to lingering EVO, which is concentrated in the intertidal region of the shoreline. Preliminary results demonstrating the bioavailability of lingering EVO and its ability to induce CYP1A has recently been presented by Short et al. (2005).

Sea otters in PWS overall in 2004 number approximately 10,000 and appear stable. However, while the numbers in western PWS have increased by more than 700 animals between 1993 and 2003, at northern Knight Island, the numbers remain at less than half the 1989 abundance.

The most thorough study of sea otters took place at northern Knight Island, which was heavily oiled in 1989. This study provides a “worst case” evaluation of ongoing impacts to sea otters. The sea otter subpopulation of northern Knight Island has generally been stable over the last several years, but remains well below the estimated pre-spill population levels. Exposure of sea otters on northern Knight Island to lingering oil has been measured since 1996 using CYP1A, an enzyme that is induced in vertebrate species following exposure to PAHs. The most recent biomarker data on exposure collected in summer of 2004 and 2005 (Ballachey and Bodkin 2006) indicate that levels of CYP1A in sea otters from northern Knight Island have declined to levels measured in sea otters from unoiled Montague Island, suggesting that exposure to PAHs is no longer a concern in this oiled area. Most researchers agree that as the spatial extent of lingering oil diminishes over time, the potential for sea otters to be exposed to lingering oil and their degree of exposure should also diminish.

9.5.2 Conclusions

It is likely that sea otter populations in some portions of western PWS are recovering from measurable acute and chronic effects of the spill, evidenced by a modest average annual rate of increase between 1993 and 2005. However, recovery of the subpopulation on northern Knight Island appears to be constrained by demographic lag, residual oil effects, continuing exposure, or by some other unknown stressors. These unknown stressors introduce some uncertainty into the interpretation of sea otter population trends on northern Knight Island, a relatively small subpopulation that is susceptible to small impacts on the total population. Predation by killer whales may be particularly important, and subsistence hunting could also be a factor. Emigration of sea otters must also be considered. Food resources apparently are not limiting on northern Knight Island and should not limit population growth.

9.5.3 Recommendations

Further investigation into the continued low numbers of the northern Knight Island subpopulation is recommended, including population modeling to address the sensitivity of this population to the impact of other stressors (e.g., hunting and predation).

10. HARBOR SEAL

Harbor seals (*Phoca vitulina richardsi*) are year-round residents of and one of the most common marine mammals in PWS and the Gulf of Alaska. The EVOS affected some of the largest harbor seal haul-out sites and waters adjacent to those sites in PWS. Immediately following the spill, dozens of seals became visibly oiled, including pups born to oiled mothers. At least 300 seals were unaccounted for at oiled sites after the spill, although the number of carcasses recovered was much lower. Based on trend-count surveys conducted before and after the oil spill, the population at oiled sites declined by 43 percent between 1988 and 1989 compared to an 11 percent decline at unoiled sites. In response to the injury, the EVOS Trustee Council established a recovery objective of stable or increasing harbor seal populations. By 1990, there was no longer any difference in the rate of population decline at oiled and unoiled sites. But populations continued to decline approximately 4.6 percent per year from 1990 to 1997. The harbor seal continues to be listed as "not recovering from the effects of the spill" (EVOS Trustee Council 2002b). This section evaluates these and other population parameters to assess the likelihood of ongoing injury to harbor seals from EVO.

10.1 BACKGROUND

This section describes the natural history of harbor seals, initial impacts of EVOS, and the current status of injury and recovery classifications.

10.1.1 Natural History of Harbor Seals

Harbor seals are one of the most common marine mammals in PWS and the Gulf of Alaska. They are year-round residents, their habitat is near-coastal, and haul-out sites in PWS are often located in estuaries and protected coves. Harbor seals spend much of their time hauled out on land during pupping and molting season; haul-out sites are selected for protection from land predators and access to deep water and food. Haul-out sites include intertidal reefs, rocky shores, mud and sand bars, floating glacial ice, and gravel and sand beaches (Frost 1997). Harbor seals show high fidelity to their haul-out sites, with individual seals averaging only two to four haul-out sites per season (Frost 1997).

Harbor seals prey primarily upon fish and invertebrates. Major food items of PWS harbor seals include pollock, capelin, Pacific cod, herring, and octopus (Frost 1997). Harbor seals are also known to eat other schooling fish, flatfish, squid, and crustaceans (Wynne 1993). Detailed studies on fatty acid composition of seal blubber show that harbor seals from different haul-outs within PWS may have very different diets (Iverson et al. 1997). This may, in part, reflect their tendency to eat whatever is locally and seasonally available. Harbor seals tend to remain within 50 km of shore, and typically feed in water less than 200 m deep (Frost et al. 2001; Lowry et al. 2001; Hastings et al. 2004; Small et al. 2005).

10.1.2 Initial Impact of EVOS

In the early weeks after the spill, harbor seals swam in oiled water, surfacing in oil slicks to breathe at the air–water interface where volatile hydrocarbon vapors were present (Frost et al. 1994a). Through spring and summer, seals crawled over and rested on oiled rocks and algae at haul-outs. Harbor seals inhabiting central PWS, including Eleanor Island, the north part of Knight Island, and the west side of Knight Island Passage became heavily coated with oil. More than 80 percent of the seals observed in these areas in May 1989 had oil on them (Frost 1997). Many seals remained oiled until their annual molt in August (Frost et al. 1994a). Some of the haul-out sites remained oiled through the May/June pupping season. Many pups became oiled shortly after birth (Frost 1997). In the Bay of Isles and Herring Bay on the north end of Knight Island, 89–100 percent of all seal pups seen were oiled (Lowry et al. 1994).

Nineteen seal carcasses were recovered from PWS and the Gulf of Alaska following the spill, 15 of which were oiled (Loughlin et al. 1996). Spraker et al. (1994) examined the seals, but because of carcass decomposition, were unable to determine exact cause of death. To assess injuries, researchers then harvested 27 additional seals, both oiled and unoiled. They documented brain and liver lesions, as well as skin irritation and conjunctivitis in oiled seals. Brain lesions may have been responsible for the abnormal behavior exhibited by many seals immediately after the spill in April and May 1989 (Loughlin et al. 1996). They were lethargic, disoriented, and unusually tame. These atypical behaviors had ceased by September 1989 and were not observed again (Frost 1997).

Using data from trend counts at major haul-out sites in PWS, Frost et al. (1994a) estimated the total mortality of harbor seals caused by the EVOS to be 302. This number was based on the number of seals missing from oiled haul-out sites relative to previous counts. This estimate was called into question by Hoover-Miller et al. (2001), who suggested that invalid assumptions were used in Frost's analysis. They agree that seals declined at oiled sites in 1989, but suggested mortality was lower than the 302 estimate, and that many seals simply relocated (i.e., moved to unoiled haul-out sites). They suggested that Frost et al. (1994a) did not sample a large enough geographic region to determine whether the missing seals had died, and that a greater portion of PWS (beyond the 25 sample sites in central and eastern PWS) should be sampled.

Frost et al. (1994b) examined tissues (brain, liver, blubber, muscle, etc.) from seals collected in heavily oiled parts of PWS and compared them with seals from moderately oiled and unoiled areas in 1989 and 1990. In 1989, concentrations of PAHs were significantly higher in blubber of seals from the oiled population. Blubber was the only tissue sampled that exhibited elevated PAH concentrations. For seals from both populations, PAH concentrations in other tissues (brain, liver, muscle, etc.) were near or below detection limits. By 1990, PAH concentrations in oiled seals were significantly lower than they had been in 1989, indicating oil impacts had decreased 1 year later.

They also examined metabolites of petroleum-related aromatic compounds in bile samples from the same two populations. Seals from the 1989 and 1990 oiled populations had significantly higher concentrations of fluorescent aromatic compounds than the control population. They concluded that the level of exposure to petroleum-related hydrocarbons for harbor seals in PWS had declined by 1990, but was still higher than exposure for seals outside the spill path.

10.1.3 Current Status of Injury and Recovery Classification

Harbor seal populations were in decline for at least 5 years prior to the EVOS, and perhaps longer. Since the oil spill, populations continued to decline, yet recent counts indicate the population may have started to stabilize. In the year of the oil spill, populations at oiled sites declined at a greater rate than unoiled sites. Since then, there has been no observable difference between oiled and unoiled sites. The recovery objective for the harbor seal is a stable or increasing population (EVOS Trustee Council 2002b). In 2002, the EVOS Trustee Council (2002b, p. 11) concluded that harbor seals “are considered not recovering from the effects of the oil spill.”

10.2 SPATIAL AND TEMPORAL CHANGES IN HARBOR SEAL POPULATIONS

Some evidence exists to suggest that the population abundance of harbor seals in the Gulf of Alaska began to decrease in the early 1970s, yet reliable estimates of the number of harbor seals prior to the mid-1970s are not available because of a lack of standardized surveys. The declining trend for harbor seals in the Gulf of Alaska is based on the 63 percent decline in PWS during 1984–1997 reported by Frost et al. (1999), and the 85 percent decline on Tugidak Island during 1976–1988 observed by Pitcher (1990). In other parts of the Kodiak Archipelago, counts likely declined substantially between the mid 1970s and the early 1990s (Small et al. 2003). The declining trend in these regions was evident prior to EVOS. Population trends after the spill are discussed in Section 10.3.1.

10.3 LINKAGE BETWEEN EVOS AND THE CURRENT CONDITION OF HARBOR SEAL POPULATIONS IN PWS

The current condition of the harbor seal population in PWS can potentially derive from one or more of the following:

- Residual effects from the original spill
- Ongoing exposure to lingering oil
- Other natural or anthropogenic factors that can influence the population.

The relative importance of these different factors is assessed based on the harbor seal life history and inherent ability of the population to recover from the initial impacts of the spill, the likelihood that the behavior of harbor seals could result in ongoing exposure to lingering EVO to a degree that could cause adverse effects and injury, and the nature and magnitude of other factors that could affect the condition of the population.

10.3.1 Residual Effects from the Original Spill

Surveys were conducted after the spill to compare counts of harbor seals at oiled and unoiled sites (Frost et al. 1994a). The study used a survey route that had been established to document harbor seal population trends in PWS in 1984. The number of seals at each of the 25 haul-out sites within the survey route was counted in 1983, 1984, and 1988. Seven of the 25 sites were oiled by the EVOS, whereas the other 18 sites remained unoiled. Surveys continued annually after 1989. The number of harbor seals at oiled sites declined from an estimated 675 harbor seals in 1984 to 418 seals in 1988, and at unoiled sites declined from 1,121 in 1984 to 637 in 1988. The estimated annual rate of decrease for both unoiled and oiled sites was approximately 12 percent per year between 1984 and 1988. In 1989, the average number of seals at oiled sites declined by 43 percent compared with an 11 percent decline at unoiled sites (Frost et al. 1994a). By 1990, there was no longer any difference in the rate of decline between oiled and unoiled sites, and the authors concluded that the effects of the oil spill were evident only in population declines of 1989 (Frost et al. 1994a).

Frost et al. (1999) continued post-spill surveys at the same 25 sites from 1990 to 1997. Surveys were conducted during the annual molt (August-September) each year. The objective was to determine the overall population trend of harbor seals in PWS and included both oiled and unoiled areas. Counts were found to be substantially influenced by time of day, date, and time relative to low tide each day. Because of this, the authors chose to use a model with a standardized set of covariates to adjust the annual counts to the expected count under optimal conditions. Bootstrap methods[52] were used to address uncertainty. Counts were then analyzed with both linear and loglinear regression. Adjusted counts showed significant annual population declines of 4.6 percent per year. There was no significant trend in unadjusted counts, which fluctuated up and down annually.

In the last decade, the number of seals in the Kodiak Archipelago has been increasing (Small et al. 2003; Blundell et al. 2005), although the abundance level is much reduced from the 1970s. ADFG continued to conduct annual surveys in PWS following 1997, and recent counts indicate the population may be stabilizing after the sustained decline over the past two decades (Blundell et al. 2005).

Residual effects of the original spill are unlikely. In the initial oiling, harbor seals were exposed via direct contact exposure with EVO and possibly via indirect exposure through consumption of contaminated food. These exposures were short-lived, however, and were probably unimportant in the years subsequent to the oil spill (Hartung 1995). Although harbor seal populations declined at a greater rate at oiled sites in comparison with unoiled sites in 1989, these differences were transient and no longer discernible by 1990.

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[52] In statistics, bootstrapping is a method for estimating the sampling distribution of an estimator by resampling with replacement from the original sample.

10.3.2 Current Exposure to Lingering Oil

Harbor seals could potentially be exposed to lingering oil as they move about on their haul-out sites. Exposure would be limited to dermal contact because seals do not forage in intertidal sediments. Lingering oil in surface sediments occurs primarily in the form of highly weathered, solid asphalt-like material (Short et al. 2004a). Weathered oil in this form is insoluble and, therefore, unlikely to be bioavailable to seals. Although some seals may come into contact with this oil, impacts from this type of contact would be negligible. In contrast to surface oil, subsurface lingering oil is less weathered and more soluble, and therefore potentially bioavailable. However, it is highly unlikely that seals would ever come in contact with subsurface EVO at isolated subsurface locations in the intertidal environment.

Indirect effects of lingering oil on harbor seals via exposure to nearshore food resources are also expected to be negligible. The low likelihood of contamination in prey, combined with large feeding areas relative to the extent of lingering oil, makes continued exposure to lingering EVO through food resources for harbor seals unlikely. To the extent that such exposure may occur, its influence on harbor seal populations would probably be small relative to the conditions that affected population declines prior to the spill, which may continue to exist today.

In summary, there is no evidence that harbor seals are affected by lingering EVO. Although lingering oil may remain in small amounts at some haul-out sites, the low potential for exposure suggests that its overall direct impact on harbor seals is likely to be negligible.

10.3.3 Toxicity of Lingering Oil to Harbor Seal

There is no evidence that harbor seals are affected by lingering EVO. While lingering oil may remain in small amounts at some haul-out sites, the low potential for exposure suggests that its overall direct impact on harbor seals is likely to be negligible.

10.3.4 Indirect Effects of Lingering Oil

Indirect effects of lingering oil on harbor seal via exposure to nearshore food resources are also expected to be negligible. To the extent that such exposure may occur, its influence on harbor seal populations would probably be small relative to the conditions that affected population declines prior to the spill, which continue to exist today. These conditions could include climate-driven ecosystem shifts of the late 1970s, predation, interactions with fisheries, subsistence hunting, contaminants, and emigration. Any or all of these factors could play a role in the continuing decline of the harbor seal in the Gulf of Alaska.

10.3.5 Other Factors Influencing Current Injury or Rate of Recovery or Population Variability

The number of harbor seals in PWS and the Kodiak Archipelago began to decline in the 1970s and 1980s, before EVOS. Researchers reported a 63 percent decline in PWS during 1984–1997 (Frost et al. 1999) and an 85 percent decline on Tugidak Island during 1976–1988 (Pitcher 1990). Although those declines may have begun earlier, comparable counts are not available. In other parts of the Kodiak Archipelago, counts also likely declined substantially between the mid 1970s and the early 1990s (Small et al. 2003).

Causes of harbor seal population declines in PWS and the Gulf of Alaska prior to EVOS are not known with certainty. Many possible factors may have contributed to the declines and could continue to affect harbor seal populations. These include predation by killer whales, disease and parasitism, reduced prey biomass and quality due to either commercial fisheries or climate-driven ecosystem shifts, and human-caused mortality from illegal shooting and subsistence harvests, contaminants, and incidental takes from fishery operations. The extent to which these factors influenced population declines, either individually or cumulatively, is unknown (Frost 1997; Jemison and Kelly 2001; Small et al. 2003).

Fadely et al. (1998) did a broad study of blood chemistry, blubber parameters, and body condition of seals in PWS, Southeast Alaska, and Kodiak Island to determine if there were differences between declining (PWS) and non-declining (Southeast Alaska) populations. They found no differences among the populations sampled (Kodiak Island, PWS, and Southeast Alaska) during 1993–1996. Based on the measured blood values, the majority of seals sampled showed no indication of poor health. Some researchers have suggested that the population declines may be due, in part, to a decrease in carrying capacity created during the oceanic regime shift of 1976. Average climate in the Gulf of Alaska shifted in 1977 from cool to warm, and over subsequent years, a reorganization of fish and shellfish communities occurred (Anderson and Piatt 1999). The impact of such reorganization of the fish community on apex predators, such as the harbor seal, is unknown.

10.4 ACTIONS TAKEN OR PLANNED

Studies of harbor seal populations following the spill indicated an increase in the rate of decline at oiled sites in comparison with historical trends for PWS. Subsequent restoration efforts have focused on monitoring to assess population behavior in the context of long-term trends, research to assess causative factors associated with declining populations, and acquisitions of important habitat.

10.5 SUMMARY AND CONCLUSIONS

10.5.1 Summary

The harbor seal population in PWS began declining in the early 1980s, and perhaps earlier. The EVOS affected some of the largest harbor seal haul-out sites and waters adjacent to those sites in PWS. Immediately following the spill, dozens of seals became visibly oiled, including pups born to oiled mothers. Based on trend-count surveys conducted before and after the oil spill, the number of seals at oiled sites declined by 43 percent between 1988 and 1989 compared to an 11 percent decline at unoiled sites. In the last decade, the number of seals in the Kodiak Archipelago has been increasing (Small et al. 2003; Blundell et al. 2005), although the abundance level is much reduced from the 1970s. ADFG continued to conduct annual surveys in

PWS following 1997, and recent counts indicate the population may be stabilizing after the sustained decline over the past two decades (Blundell et al. 2005).

Residual effects of the original spill are unlikely. In the initial oiling, harbor seals were exposed via direct contact exposure with EVO and possibly via indirect exposure through consumption of contaminated food. These exposures were short-lived, however, and were probably unimportant in the years subsequent to the oil spill (Hartung 1995). Although harbor seal populations declined at a greater rate at oiled sites in comparison with unoiled sites in 1989, these differences were transient and no longer discernible by 1990.

Harbor seals could potentially be exposed to lingering oil as they move about on their haul-out sites. Exposure would be limited to dermal contact because seals do not forage in intertidal sediments. Lingering oil in surface sediments occurs primarily in the form of highly weathered, solid asphalt-like material, which is insoluble and, unlikely to be bioavailable to seals. Although some seals may come into contact with this oil, impacts from this type of contact would be negligible. Although subsurface oil is potentially bioavailable, it is highly unlikely that seals would ever come in contact with subsurface EVO at isolated subsurface locations in the intertidal environment. Furthermore, the low likelihood of contamination in prey, combined with large feeding areas relative to the extent of lingering oil, makes continued exposure to lingering EVO through food resources for harbor seals unlikely.

10.5.2 Conclusions

The recovery objective for harbor seals is "stable or increasing harbor seal populations" (EVOS Trustee Council 2002b). This recovery objective does not fully address the overall objective of a *return to conditions that would have existed if the spill had not occurred* because it does not take into account other factors or stressors that can affect population trends. It is unlikely that harbor seals are continuing to experience any residual adverse effects from the spill or from lingering EVO in the intertidal zone.

11. KILLER WHALE (*Orcinus orca*)

Within a week of EVOS in March 1989, the AB pod of killer whales was exposed to the spreading oil slick. Within the next few months to years, approximately 36 percent of the individuals in the AB pod were no longer seen in their maternal groups and were assumed to have died. The actual connection between individual deaths within the AB pod and exposure to the oil spill had not been formally established; however, the pod was observed surfacing in an EVO slick in the weeks following the spill and nearly all these deaths occurred at the time of the spill or the following winter. Based on this observed exposure and other possible exposures and the potential relatedness to killer whale mortalities, the EVOS Trustee Council (1994) established recovery objectives and defined a restoration strategy for the AB pod in PWS. The recovery objectives and restoration strategy were not modified in 2002.

The AT1 transient population has not been identified by the EVOS Trustee Council as an injured resource, but it is worthwhile to note that this population also suffered losses subsequent to the oil spill.

11.1 BACKGROUND

This section describes the natural history of killer whales, initial impacts of EVOS, and the current status of injury and recovery classifications.

11.1.1 Natural History of Killer Whales

The killer whale, or orca (*Orcinus orca*), is the largest member of the dolphin family (Delphinidae), attaining sizes of 8 m in length for males and 7 m for females, and weighing as much as 4,000 kg in males and 3,000 kg in females (Wynne 1993; Matkin and Saulitis 1997). They are found worldwide, but have been studied most heavily in the northern Pacific, including waters of the western U.S., Canada, and Alaska, where researchers have identified and tracked pods and individual orcas for more than three decades (Reeves et al. 2002; Gordon and Flaherty 1990; Matkin et al. 1994).

The killer whale is a long-lived species with a relatively low rate of reproduction (Matkin and Leatherwood 1986). Males live 50 to 60 years and females up to 80 or 90 years (Reeves et al. 2002). The maximum reproductive life span of a female is about 30 years, with a calf born approximately every 5 years; thus, each female produces only 4 to 6 calves throughout her life (Gordon and Flaherty 1990).

Unique markings on individual killer whales make it possible to identify them as individual and thus conduct population studies on them using photographic techniques (Matkin and Saulitis 1997). Males and females can be distinguished by the shape and size of the dorsal fin. Males have a large (1–1.8 m), triangular dorsal fin. The dorsal fin of females is much smaller (less than 0.9 m) and is distinctly curved, or sickle-shaped. In addition, individual killer whales can be identified by the unique saddle-patch markings immediately behind the dorsal fin (Gordon and Flaherty 1990; Reeves et al. 2002).

Three genetically distinct lineages occur throughout the northern Pacific. They are known as residents, transients, and offshores. Residents and transients are found in PWS and are discussed in this report. Residents forage on fish and maintain extremely stable social groups over time. Transients feed on marine mammals and there may be some emigration/immigration from groups.

Resident pods may number up to 40–50 individuals. Pods comprise smaller, maternal groups, which include a female and her offspring of both sexes. Resident whales remain in these maternal groups for life (Matkin and Saulitis 1997). One or more maternal groups may occasionally split off and form a new pod, but there is no exchange of individuals between established resident pods (Matkin et al. 1994). Transient groups are much smaller than resident pods, averaging fewer than six individuals per pod (Gordon and Flaherty 1990), and often range over a wider area than residents (Matkin et al. 1994 (although the AT1 transient group is an exception). The social dynamics of these groups are not well understood, but are more complex than that of resident pods (Dahlheim and Matkin 1994).

Resident pods of killer whales enter PWS during any time of year but primary use is in the summer (July, August, and September) when large numbers of coho salmon pass through the sound on their way to spawning grounds. Southwestern PWS, Resurrection Bay, and Aialik Bay all appear to be important feeding areas for resident killer whales during salmon migration (Matkin and Saulitis 1997). During this time, AB pod is one of the more commonly sighted resident pods in PWS (Matkin and Saulitis 1997).

One group of transient killer whales, the AT1 population, appears to center its range in PWS/Kenai Fjords and may be found there year-round. These whales hunt harbor seals along the shorelines of Knight and Montague islands and in the glacial fjords of Kenai Fjords. They also hunt Dall's porpoises in Knight Island Passage, Montague Strait, and Kenai Fjords (Matkin and Saulitis 1997). Unlike other groups of transients, this population has shown up repeatedly and consistently in the same area year after year, appearing to center their range on PWS (Matkin and Saulitis 1997).

Of the few behaviors that bring killer whales into direct contact with the shoreline, one that has been documented for both residents and transients of PWS is beach rubbing. Groups of whales enter shallow water for brief periods of time to rub their bodies on smooth, rounded stones. Some pods appear to do this quite regularly while others only sporadically (Matkin 2004, pers. comm.). Specific beaches in PWS and Kenai Fjords appear to be selected for this behavior, including Point Nowell, Sleepy Bay on Latouche Island, southern Perry Island, and Sunny Cove in Resurrection Bay (Matkin and Saulitis 1997; Matkin 2004, pers. comm.).

11.1.2 Initial Impact of EVOS

As many as eight resident pods and members of the AT1 transient population are sighted regularly in the PWS/Kenai Fjords region. Initial impacts of EVOS were assessed by observing and photographing the individual whales of these pods during the spring and summer for three years (1989, 1990 and 1991) following the spill. These observations were compared with known demographics of the pods to determine if there were changes in overall population, seasonal abundance, birth rates, mortality rates, and/or habitat usage (Matkin et al. 1994; Dahlheim and Matkin 1993).

The resident AB pod suffered an unusually high rate of mortality after the spill. Prior to the spill, the pod numbered 36 individuals. One week following the grounding of the *Exxon Valdez*, seven members of this pod were unaccounted for. The following spring, six more members of the pod were missing. By the third year of the study, a total of 14 whales were missing from the AB pod. No carcass of any of these whales was ever discovered, but that is not unusual because whales apparently sink when they die (Matkin and Saulitis 1997). In addition, in studies from Puget Sound to Kenai Fjords, resident whales missing from their matrilineal groups have never appeared in other pods or matrilineal groups. In studies all along the west coast, it is accepted that if an animal is missing from a resident pod for more than a year, it is considered dead (Dahlheim and Matkin 1994). While matrilineal groups may occasionally split off from the main pod and form a new pod, the demographics of the whales that went missing from the AB pod indicate that was not the case. (The missing whales included individuals from four different subpods, and two of the missing females left behind 2- and 3-year old juveniles [Dahlheim and Matkin 1994].) Therefore, although there was a lack of material evidence, researchers concluded that all 14 missing whales had died (Matkin and Saulitis 1997).

The mortality rate for the AB pod was 19 percent in 1989 and 21 percent in 1990 (Matkin et al. 1994). This rate compares to a natural mortality rate for killer whales of 2.2 percent or less (Dahlheim and Matkin 1994). Although the exact cause of mortality of these whales has not been proven, killer whales were observed swimming through and surfacing in oil slicks in the days and weeks following the spill. It is possible that petroleum or petroleum vapors were inhaled by these whales (Matkin and Saulitis 1997). Killer whales may have been exposed to petroleum hydrocarbons at the time of the spill via consumption of contaminated fish (Dahlheim and Matkin 1993). The AB pod is a resident group of killer whales that forages primarily on coho salmon during its seasonal foraging excursions into PWS and may also prey on other fishes (herring, halibut, and sablefish). However, although nearshore rockfish were affected by the spill, concentrations of petroleum hydrocarbons in rockfish or other species of fish were not identified as a causative factor in the decline of the AB pod. Similar mechanisms of exposure have been proposed for the AT1 population of transient killer whales, including nearshore predation on harbor seals, but have not resulted in determination of injury.

The AT1 transient population has not been identified by the EVOS Trustee Council as an injured resource, but it should be noted that this group also suffered losses subsequent to the oil spill. At the time of the spill, members of the AT1 population were observed in the area of the spill and adjacent to the tanker when it was leaking oil. Two stranded AT1 whales were found in 1990, but the cause of death was not determined. Nine whales from the AT1 transient population disappeared shortly after the spill (Matkin and Saulitis 1997). Because transient whales do not always remain in their natal groups, it could not be determined immediately that the missing AT1 whales were dead. However, after 5 years, these whales had still not reappeared in their original groups or in any other observations of killer whales and therefore were considered dead. It was suspected the missing whales died from the protracted effects of either inhaling oil or oil vapors or as a result of extensive feeding on heavily oiled harbor seals following the spill (Matkin and Saulitis 1997). Because of the initial difficulty in confirming deaths of the missing AT1 whales, a listing of "injured" for this population was not pursued in the years immediately following the spill. Consequently, the EVOS Trustee Council (1999, 2002b) has not recognized the sharp decline in the AT1 group following the spill and has not identified this population as an injured resource. The EVOS Trustee Council (1999, 2002b) has noted the presence of high levels of non-EVO contaminants in tissues of AT1 whales and has recommended continued monitoring of these substances as possible contributing factors in recent populations trends in the AT1 group.

11.1.3 Current Status of Injury and Recovery Classification

In 2002, the EVOS Trustee Council (2002b, p. 14) concluded that "since AB pod has not regained its pre-spill size of 36 individuals, killer whales are considered to be recovering, but not fully recovered from the effects of the oil spill." The recovery objectives and restoration strategy were not modified in 2002, although monitoring was expanded to include the AT1 population of killer whales. The recovery objective for killer whales is a return to pre-spill numbers of 36 individuals for the AB pod. Monitored natural recovery was adopted as the restoration strategy that would be implemented to meet the recovery objective.

11.2 SPATIAL AND TEMPORAL CHANGES IN KILLER WHALE POPULATION

Spatial and temporal patterns in killer whale populations provide an important framework for judging recovery in the context of past and present exposure to EVO as well as natural variability. By 1991, the resident AB pod had been reduced from 36 to 22 whales. Since then, the population has fluctuated with additional births and mortalities, to between 22 and 26 individuals. Despite the birth of seven new calves between 1990 and 1996, the pod still only numbered 26 whales in 2001. As of 2005, the pod numbered 27 (Matkin 2005, pers. comm.). The remaining six resident pods that frequent PWS and Kenai Fjords have seen an overall increase in population from 86 in 1984 to 105 in 1992 (Matkin et al. 1994) and 134 in 2004 (Matkin 2005, pers. comm.).

The post-spill distribution of killer whales in PWS is shown in Figure 11-1. There is nothing reported in the literature to suggest that killer whales have shifted their geographic distribution following EVOS.

11.3 LINKAGE BETWEEN EVOS AND THE CURRENT CONDITION OF KILLER WHALE POPULATIONS IN PWS

The current condition of the killer whale population in PWS can potentially derive from one or more of the following:

- Residual effects from the original spill

- Ongoing exposure to lingering oil
- Other natural or anthropogenic factors that can influence the population.

The relative importance of these different factors is assessed based on the killer whale life history and inherent ability of the population to recover from the initial impacts of the spill, the likelihood that the behavior of killer whales could result in ongoing exposure to lingering EVO to a degree that could cause adverse effects and injury, and the nature and magnitude of other factors that could affect the condition of the population.

11.3.1 Residual Effects from the Original Spill

The killer whale is a long-lived species with a relatively low rate of reproduction (Matkin and Leatherwood 1986). Males live 50 to 60 years and females up to 80 or 90 years (Reeves et al. 2002). The maximum reproductive life span of a female is about 30 years, with a calf born approximately every 5 years; thus, each female produces only 4 to 6 calves throughout her life (Gordon and Flaherty 1990). These characteristics suggest that recovery rate of a population following a significant loss of individuals will be relatively slow.

Modeling of killer whale population dynamics in PWS was performed by Matkin et al. (2003). They also discuss the characteristics of the AB pod relative to those of other resident pods in PWS and in northern British Columbia. The modeling analysis indicates that the AB pod was impacted by EVOS primarily through the loss of young and reproductive females. These are the age/sex classes that usually demonstrate very low mortality rates. The reproductive value of the AB pod decreased due to the EVOS because the young females offer the most potential contributions to future population growth. Based on this modeling effort, it is concluded that the AB pod has not yet recovered from the initial impacts of EVOS due to the loss of young females, but would have recovered had there been no loss of females at the time of the spill.

The analysis of birth and death rates shows that although birth rates were not affected by the spill, the death rate for the AB pod was much higher than expected during and immediately following the spill (Matkin et al. 2003). During 1989–1990, there were 14 deaths in this pod, when only 1 would have been expected. By 1996, the death rate returned to within the range of other resident pods in PWS. The resident population of killer whales in PWS excluding the AB pod grew 3.3 percent per year. This rate is somewhat higher than the 2.6 percent growth rate for the resident killer whale population in northern British Columbia in the 1970s and 1980s. Matkin et al. (2003) hypothesize that the PWS population is recovering from a previous perturbation (prior to EVOS) or that the carrying capacity of the habitat for killer whale in PWS has increased.

Matkin et al. (2003) looked at the effect that loss of females had on the potential recovery of the AB pod. Their analysis of reproductive value and birth and death rates does suggest that the AB pod would not yet have recovered to a pre-spill abundance given its age/sex structure and reproductive condition after EVOS. In addition to loss of females, AB pod recovery has suffered from higher mortality rates since the spill compared to other pods. It might be useful to analyze the potential population growth rate and projected abundance of AB pod based on age/sex structure after EVOS and fecundity/mortality rates estimated from PWS pods as a whole.

11.3.2 Ongoing Exposure to Lingering Oil

At the time of the spill in 1989, contact with liquid oil on the sea surface and inhalation of oil vapor at the air–sea interface were the environmental pathways of most concern for resident killer whales, although transient AT1 killer whales may have contacted oil through ingestion of oiled harbor seals. These pathways and routes of exposure are not significant today because except for occasional sheens from beaches, lingering oil does not occur at the sea surface, but rather is predominant in the intertidal zone as highly-weathered asphalt-like material in surface sediments or is largely sequestered in subsurface sediments.

Dermal contact with intertidal sediments during beach rubbing is a possible but unlikely route of exposure to lingering oil for killer whales. Three of the rubbing locations used by killer whales in PWS (Point Nowell, Perry Island, and Sleepy Bay) were within the trajectory of the oil spill, and Sleepy Bay was heavily oiled. However, there are three major factors, which when considered collectively, indicate that significant dermal exposure to lingering oil is unlikely. These factors are as follows:

1. The AB pod has been observed visiting rubbing beaches only on an infrequent basis (Matkin and Saulitis 1997) and most rubbing occurs for only 10 to 20 minutes at a time (Matkin 2004, pers. comm.). Hence, the frequency and duration of exposure via dermal contact with rubbing beaches is likely to be low, if it occurs at all.
2. Rubbing locations would need to coincide with the elevation and patchy distribution of lingering oil, and the depth of disturbance caused by rubbing must be sufficient to mobilize subsurface lingering oil and bring it into contact with the killer whale. Because subsurface lingering oil is patchily distributed over a relatively small fraction of the beach area of PWS, the likelihood of whales encountering lingering oil is low.
3. The depth of disturbance that could be caused by rubbing is unknown, but must be sufficient to bring the oil into contact with the skin of the whale. The bioavailability of lingering oil via dermal contact with killer whales is also unknown,[53] and is likely to be mitigated by continuous washing by seawater during routine swimming following beach rubbing.

In summary, none of the exposure pathways and routes considered at the time of the spill is present today. Because lingering EVO is sequestered in subsurface sediments or occurs predominantly as a weathered solid in surface sediments of the intertidal zone, it is not bioaccessible via the inhalation route of exposure, and exposure is also unlikely via dermal contact during beach rubbing. Significant food-chain exposure is unlikely because fish metabolize PAHs rapidly (Lawrence and Weber 1984; Eisler 2000), and significant

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[53] The dermal route of exposure would be difficult to quantify even for well-known mammalian wildlife species that could be studied in controlled laboratory settings and is almost never conducted in ecological risk assessments where the frequency and duration of contact is much higher than indicated here.

bioaccumulation is unlikely. Moreover, the prey fish favored by resident killer whale do not occur in the vicinity of lingering oil or, in the case of Pacific herring, are present in the intertidal zone only transiently during spawning (Matkin and Saulitis 1997; Hart 1973; Brown and Carls 1998).

Based on the low likelihood of exposure and the low bioaccessibility of oil to killer whales, chances for lingering oil impacts to killer whales are very low.

11.3.3 Other Factors Influencing Current Injury or Rate of Recovery or Population Variability

Conflicts between killer whales and fisheries may have contributed to the AB pod mortalities. In 1985, the AB pod began removing fish from the longlines of commercial fishing boats. Fishermen attempted many methods of deterring the whales, including attempts to frighten them away by shooting at them (Dahlheim and Matkin 1994). At least 10 whales were documented with bullet wounds in 1985 and 1986. Five of these whales subsequently died (Matkin and Saulitis 1997). The shooting of killer whales apparently ceased after 1986 when it was outlawed by the Marine Mammal Protection Act, and there is no indication that it has been a problem since that time. It is possible that the wounding of some whales may have weakened them and, over time, contributed to their premature mortality (EVOS Trustee Council 2002b). However, AB pod increased in numbers in 1987 and 1988 and it is unlikely that bullet wounds suffered 3 years previously caused the unprecedented mortalities of 1989–1990.

Transient whales prey on harbor seals, whose population in PWS has been in decline since before the oil spill (Matkin and Saulitis 1997). In addition, AT1 and other transients contain high concentrations of non-EVOS related contaminants in their blubber (EVOS Trustee Council 2002b; Ylitalo et al. 2001). However, levels of contaminants are 10–20 times lower in resident whales and it is unlikely that exposure to non-EVOS related chemicals is adversely affecting recovery of the AB pod.

11.4 ACTIONS TAKEN OR PLANNED

Restoration actions related to resident killer whales have focused on research to better evaluate killer whale recovery and mechanisms of injury. This has included development of photo identification techniques, biopsies of skin and blubber, genetic and contaminant analysis, and population modeling.

11.5 SUMMARY, CONCLUSIONS, AND RECOMMENDATIONS

11.5.1 Summary

Within a week of EVOS in March 1989, the AB pod of killer whales was exposed to the spreading oil slick. The mortality rate for the AB pod was 19 percent in 1989 and 21 percent in 1990 (Matkin et al. 1994). By 1991, the resident AB pod had been reduced from 36 to 22 whales. The actual connection between individual deaths within the AB pod and exposure to the oil spill had not been formally established; however, the pod was observed surfacing in an EVO slick in the weeks following the spill. None of the carcasses of the missing AB pod whales has been discovered, which is not unusual because whales generally sink when they die. Since 1991, AB pod has fluctuated in number with additional births and mortalities, with a total number of 22 to 26 individuals. Despite the birth of seven new calves between 1990 and 1996, additional mortalities resulted in a total number of 27 whales in 2005.

The AB pod is a "resident" pod," and as such forages on fish and maintains stable social groups over time. Maternal groups may occasionally break off from a resident pod and form a new pod, but there is no exchange of individuals between established resident pods (Matkin et al. 1994). The killer whale is a long-lived species (up to 90 years) with a relatively low rate of reproduction (on average, females have a calf every 5 years over a 30-year reproductive span). Low birth rates and long life spans make killer whales more likely to suffer long-lasting impacts and slow recovery from an acute event, particularly when females of reproductive age are lost.

The AT1 transient population, unlike other groups of transients, is found repeatedly and consistently in the same area year after year, appearing to center its range on PWS (Matkin and Saulitis 1997). These whales forage primarily on harbor seals. The AT1 population has not been identified by the EVOS Trustee Council as an injured resource, but this population also suffered losses subsequent to the oil spill.

11.5.2 Conclusions

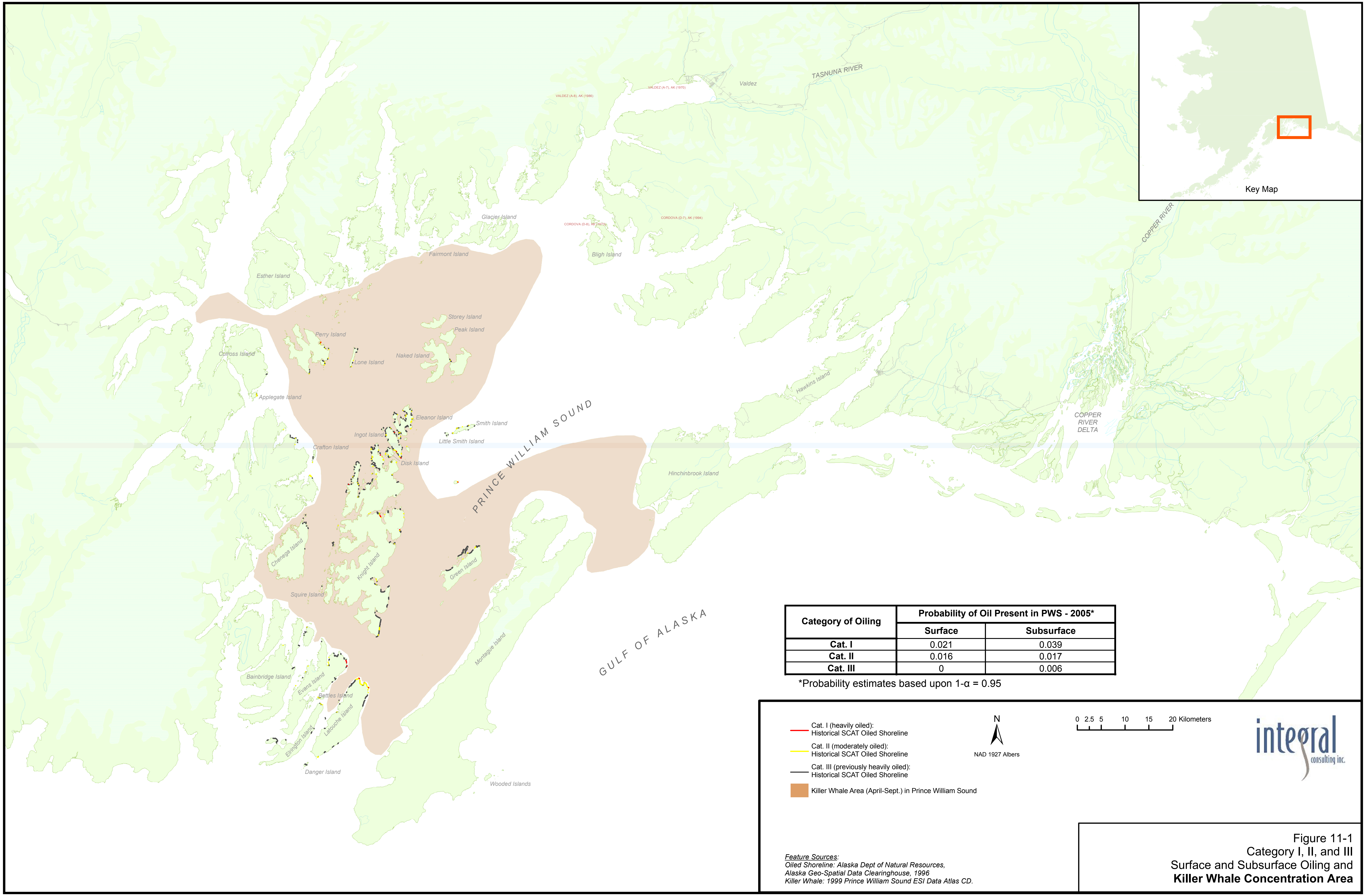
Population modeling can be used to determine if the initial population losses suffered by the AB pod after the spill would be expected to persist over years and decades following the spill. Modeling of the population dynamics (Matkin et al. 2003) indicates that the AB pod was impacted by EVOS primarily through the loss of young females, who offer the most potential contributions to future population growth. Based on this modeling effort, it was concluded that the AB pod has not yet recovered from the initial impacts of EVOS due to the loss of young females, but would have recovered had there been no loss of females at the time of the spill.

It is unlikely that killer whales could be exposed to lingering oil in intertidal sediments. Because lingering EVO is sequestered in subsurface sediments or occurs predominantly as a weathered solid in surface sediments of the intertidal zone, it is not bioaccessible via the inhalation route of exposure, and exposure is also unlikely via dermal contact during beach rubbing. Significant food-chain exposure is unlikely because fish metabolize PAHs rapidly.

11.5.3 Recommendations

It would be useful to extend the modeling work of Matkin et al. (2003) and analyze the potential population growth rate and projected abundance of the AB pod based on age/sex structure after EVOS and fecundity/mortality rates estimated from PWS pods as a whole, as well as those estimated for the AB pod. Refinement of the recovery objective should also be considered. The recovery objective for killer whales is a return to pre-spill numbers of 36 individuals for the AB pod. This recovery objective does not fully address the overall objective of a *return to* *conditions that would have existed if the spill had not occurred* because they do not take into account other ecological factors or stressors that can affect population trends.

In addition, the AT1 transient population warrants additional study to evaluate the status and factors affecting population trends of this group.



Category of Oiling	Probability of Oil Present in PWS - 2005*	
	Surface	Subsurface
Cat. I	0.021	0.039
Cat. II	0.016	0.017
Cat. III	0	0.006

*Probability estimates based upon 1-α = 0.95

Cat. I (heavily oiled): Historical SCAT Oiled Shoreline

Cat. II (moderately oiled): Historical SCAT Oiled Shoreline

Cat. III (previously heavily oiled): Historical SCAT Oiled Shoreline

🟧 Killer Whale Area (April-Sept.) in Prince William Sound

Feature Sources:
Oiled Shoreline: Alaska Dept of Natural Resources,
Alaska Geo-Spatial Data Clearinghouse, 1996
Killer Whale: 1999 Prince William Sound ESI Data Atlas CD.

Figure 11-1
Category I, II, and III
Surface and Subsurface Oiling and
Killer Whale Concentration Area

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