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Review

Ecological Significance of Residual Exposures and Effects from the *Exxon Valdez* Oil Spill

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ABSTRACT

An ecological significance framework is used to assess the ecological condition of Prince William Sound (PWS), Alaska, USA, in order to address the current management question: 17 y following the *Exxon Valdez* oil spill (EVOS), are there any remaining and continuing ecologically significant exposures or effects on the PWS ecosystem caused by EVOS? We examined the extensive scientific literature funded by the Exxon Valdez Trustees or by ExxonMobil to assess exposures and effects from EVOS. Criteria to assess ecological significance include whether a change in a valued ecosystem component (VEC) is sufficient to affect the structure, function, and/or health of the system and whether such a change exceeds natural variability. The EVOS occurred on 24 March 1989, releasing over 250,000 barrels of crude oil into PWS. Because PWS is highly dynamic, the residual oil was largely eliminated in the first few years, and now only widely dispersed, highly weathered, or isolated small pockets of residual contamination remain. Many other sources of polycyclic aromatic hydrocarbons (PAHs) exist in PWS from past or present human activities or natural oil seeps. Multiple-lines-of-evidence analyses indicate that residual PAHs from EVOS no longer represent an ecologically significant exposure risk to PWS. To assess the ecological significance of any residual effects from EVOS, we examined the literature on more than 20 VECs, including primary producers, filter feeders, fish and bird primary consumers, fish and bird top predators, a bird scavenger, mammalian primary consumers and top predators, biotic communities, ecosystem-level properties of trophodynamics and biogeochemical processes, and landscape-level properties of habitat mosaic and wilderness quality. None of these has any ecologically significant effects that are detectable at present, with the exception of 1 pod of orcas and possibly 1 subpopulation of sea otters; however, in both those cases, PWS-wide populations appear to have fully recovered. Many other stressors continue to affect PWS adversely, including climate and oceanographic variability, increased tourism and shipping, invasive species, the 1964 earthquake, and overexploitation of marine resources, with associated cascading effects on populations of PWS fish and predators. We conclude that the PWS ecosystem has now effectively recovered from EVOS.

Keywords: Ecological significance   Ecological recovery   Ecological risks   *Exxon Valdez* oil spill   Prince William Sound

INTRODUCTION

Ecosystems are complex, diverse, dynamic, spatially and temporally variable, and continuously subject to a plethora of natural and anthropogenic stressors. A regional-scale ecological system, such as Prince William Sound, Alaska, USA, consists of many different habitat types and hundreds of species, organized hierarchically into a complex set of interactions between and among species, as well coupled physical, chemical, and biological processes. Any human activity, from 1 person walking along a stream bank to a massive spill of crude oil, causes some biological response somewhere in the system. That response could be as unimportant as crushing a few amphipods or as consequential as acute mortalities of significant portions of entire fish and bird populations. This spectrum of possible responses means that characterizing the ecological condition of a system after some stress event is not a trivial exercise, and there are no simple metrics to rely on. A critical issue is determining whether past or current biological responses rise to the level of being ecologically significant. This scientific issue needs to be resolved to answer an important current regulatory question: 17 y following the *Exxon Valdez* oil spill (EVOS), are there any remaining and continuing ecologically significant exposures or effects on the Prince William Sound (PWS) ecosystem caused by EVOS?

Gentile and Harwell (1998) proposed a systematic framework for assessing *ecological significance*. This was derived from the US Environmental Protection Agency (USEPA) framework for ecological risk assessment (USEPA 1992, 1998; Gentile et al. 1993), which was developed to assess the magnitude and likelihood of adverse ecological effects from environmental stressors. The ecological risk framework is organized into 2 essential risk components: 1) Characterization of the stressor (exposure) profile, where "stressor" is defined as any physical, chemical, or biological change that could affect an ecological system, and 2) characterization of ecological consequences from environmental stressors, evaluated as effects on a set of assessment endpoints, also termed valued ecosystem components (VECs). The ecological risk framework was designed to address exposures to multiple stressors (natural and/or anthropogenic), multiple ecological systems and attributes, natural variability and uncertainties, recovery potential, spatial and temporal heterogeneity, and scale issues in order to support informed decision making using the best available scientific knowledge. Gentile and Harwell (1998) expanded on the ecological risk framework to identify criteria for evaluating the ecological significance of effects and laid out a process for applying these criteria to real-world environmental problems.

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Here, we add a framework for assessing the ecological significance of exposures and use this expanded framework to assess the ecological significance of present (2006) residual risks to PWS, associated with the 1989 EVOS. To conduct this ecological significance assessment, we examined approximately 500 articles in the peer-reviewed literature or government-sponsored reports, including most of the research conducted by Trustees- and/or ExxonMobil-funded scientists. We assessed the ecological significance of the present sources and exposures of chemicals remaining from EVOS and the present ecological effects that were caused either by the initial, subsequent, or continuing exposures to EVOS chemicals or by the cleanup activities after the spill. We considered the nature and distribution of sources of remaining *Exxon Valdez* oil (EVO), other existing sources that may release polycyclic aromatic hydrocarbons (PAHs) or other toxic chemicals, and current exposures of those chemicals to receptor biota of PWS and potential interactions with other stressors, either chemical, physical, or biological and either natural or anthropogenic, that exist in PWS but are not associated with EVOS. We also considered each species identified by the Exxon Valdez Trustees as species of concern and included other VECs to evaluate the degree to which PWS ecosystem effects presently remain that 1) can be causally linked to EVOS or cleanup activities and 2) are ecologically significant, including being distinguishable from natural variability and the effects of other stressors. The present study is not a formal risk assessment but, rather, is an ecological significance assessment based on a multiple-lines-of-evidence approach and reasoned consideration of an extensive and often highly diverse literature.

The Exxon Valdez *Oil Spill*

The EVOS occurred on 24 March 1989, releasing over 250,000 barrels (4.2×10^7 L) of Alaska North Slope crude oil into northeastern PWS (Figure 1; cf. Galt et al. 1991; NOAA 1992; US Coast Guard 1993; Wolfe et al. 1994; Page et al. 1995). At the time of the spill, there were minimal wind and currents, but within 2 d a major windstorm, with wind speeds in excess of 70 mph (110 km/h), drove much of the surfaced oil to the southwest and facilitated the formation of mousse, a water-in-oil emulsion that affected the fate-and-transport and exposure to physical and chemical stressors from EVOS. During the initial period, an estimated 20% to 30% of EVO evaporated, including the volatile components (which were the most toxic constituents in EVO), or was diluted, dispersed, or photooxidized in the atmosphere (Wolfe et al. 1994). About one-half the spilled oil was distributed along the shoreline and inter- and subtidal areas of central and western PWS, eastern Kenai Peninsula, and the Kodiak Island Group and the Alaska Peninsula in the Shelikof Strait of the western Gulf of Alaska (GOA), as far southwest as Chignik Bay, 970 km (600 miles) from the spill site (Owens 1991; Wolfe et al. 1994; Neff et al. 1995). The vertical range of the beached oil was constrained by the extent of tidal excursions during the storm, which were between +0.2 and +5 m above mean lower low water. Within 3 weeks, less than 25% of the initial spill material remained on the sea surface and had moved into GOA (Galt et al. 1991). Owens (1991) and Neff et al. (1995) reported that 782 km (486 miles) of PWS (about 16% of the shoreline) and 1,315 km (818 miles) of GOA (about 14% of the shoreline) were oiled to some degree. Most of the beached oil dispersed back into the ocean during the 3 y postspill. It is estimated that about 14% of the spilled oil was recovered and disposed, 2% of the initial volume of spilled oil remained on PWS beaches by the end of 1992, and 13% was deposited in subtidal sediments, mostly as highly weathered residuals previously stranded on shore (Wolfe et al. 1994).

Biological effects immediately following the EVOS included toxicological responses from acute exposures to the chemical constituents of the spilled oil and physical oiling and associated smothering, hypothermia, and other physical responses. Especially visible was the significant loss of seabirds from oiling; for example, Ford et al. (1996) estimated 375,000 seabirds suffered direct mortality (see also Wiens et al. [1996] for overview of impacts on bird communities during the first 2 y after the spill). The chemical composition of the spilled oil showed a distinctive fingerprint of PAHs and other constituents as it moved through the coastal environment (Page et al. 1995; Bence et al. 1996). But as Neff and Stubblefield (1995) and Short and Harris (1996) noted, although EVOS-derived PAHs were readily detectable, all water concentrations were well below acute toxicity levels for marine animals, even right after the spill.

Because PWS is such a highly dynamic system, with extreme storms and wave/tidal action, particularly in winter, residual EVO on shore was significantly reduced in the 1st few years. By 1991, only widely dispersed, small pockets of residual sources remained, mostly under a boulder/cobble armor in the middle and upper intertidal zone. These residual EVOS sources are either highly weathered or are covered by sediments. Consequently, PAHs from these sources are released into the coastal environment only episodically; otherwise, the residues would long since have left the system, which was the case for EVO that was not protected from physical dynamics.

In addition to the chemical and oiling stressors, a massive cleanup operation was undertaken throughout the summers of 1989 through 1991 (Harrison 1991; Teal 1991; Stoker et al. 1992; Mearns 1996). During the first summer, more than 11,000 people, including 3,500 people on the beaches, and 1,400 boats were engaged in cleaning of over 1,600 km (1,000 miles) of shoreline and intertidal areas in PWS and the GOA (Harrison 1991). Cleanup activities included the use of high-pressure and/or high-temperature (70 °C) water removal, surfactants, and manual physical removal of oil. During 1990, cleanup operations focused on the intertidal areas where oil remained, including removal of tar mats, mousse, and subsurface deposits (Stoker et al. 1992), sediment relocation (Owens et al. 1991), and bioremediation to enhance microbial degradation (Prince et al. 1994). An assessment was undertaken in 1990 to determine if cleanup of subsurface oil was merited, and because most remaining deposits were isolated from the biological environment, it was concluded those deposits no longer represented a significant threat to recovery (Stoker et al. 1992). Thus, only a few, larger deposits of subsurface oil were relocated at that time to the middle and upper intertidal zones where it was subject to wave and tidal action and natural cleaning (Owens et al. 1991; Stoker et al. 1992). Less intrusive cleanup efforts continued through the summer of 1991, involving mostly manual removal and bioremediation (Prince et al. 1994; Sugai et al. 1997). Manual cleaning of birds and marine mammals continued through the summer of 1989; more than 1,600 birds representing 71 species were captured, cleaned, and returned to the environment, and more than 350 sea otters were captured and treated, with 223 surviving and subsequently released back

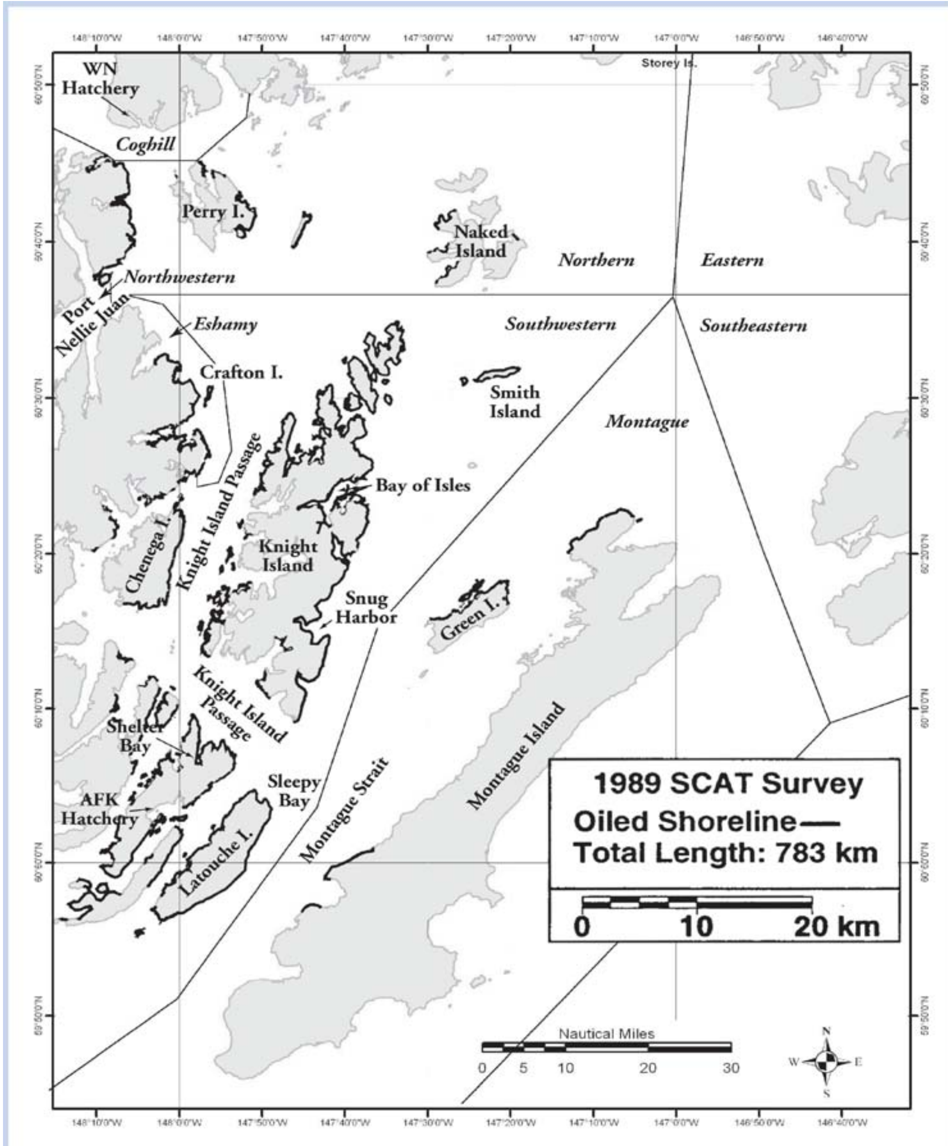

Figure 1. Map of oiled shoreline in Prince William Sound, AK, USA, attributed to the *Exxon Valdez* Oil Spill. Reprinted, with permission, from STP 1219-*Exxon Valdez Oil Spill: Fate and Effects in Alaskan Waters*, copyright ASTM International, 100 Barr Harbor Drive, West Conshohocken, PA 19428, USA.

into the environment (Monahan and Maki 1991; Stoker et al. 1992).

In many instances, there was clear evidence that the cleanup efforts constituted a more significant stressor to shoreline and intertidal biota and ecological communities than the physical and toxicological stressors associated most directly with EVO (Hoff and Shigenaka 1999). For example, some of the intertidal cleaning essentially removed all the macroalgal community from rocky surfaces, extending the time to recovery of intertidal communities by about 2 y (cf. Driskell et al. 1996; Highsmith et al. 1996, 2000; Lees et al. 1996; van Tamelen and Stekoll 1996; Skalski et al. 2001).

THE ECOLOGICAL SIGNIFICANCE FRAMEWORK

The ecological significance assessment framework (Gentile and Harwell 1998) begins by identifying the specific ecological attributes for assessing ecosystem condition. Since ecosystems have so many different biotic and abiotic constituents, the concept of ecological endpoints (Harwell and Harwell 1989; Kelly and Harwell 1990), assessment endpoints (Suter 1990; USEPA 1992, 1998), or VECs (CEAA 1999) was developed to identify those specific ecological attributes that are important ecologically and/or societally. Here we use the term VEC because it makes more explicit the idea that the attribute selected should have particular importance, but each term is essentially interchangeable. Each selected attribute should have either particular ecological importance to the structure, functioning, and/or health of the ecosystem or particular importance to society for economic, aesthetic, or values purposes. Ecologically important attributes include keystone species, important ecological processes, and higher-level attributes such as landscape mosaic or biodiversity. Societally important attributes include endangered species, economically important species, nuisance species, disease vectors, and aesthetic species.

The central objective of selecting VECs is to construct a suite of attributes that adequately encompass the range of plausible effects across ecological organizational hierarchies (ranging from population level to landscape levels of organization) such that if there is a significant change in 1 or more VEC, then that change would constitute a change in the condition (or health) of the ecological system. Conversely, if there is a change in the health of the ecosystem, then that change would be manifested in 1 or more VEC. One corollary of this criterion is a requirement for there to be sufficient information about the VEC to make this determination; that is, an ecological attribute that might otherwise be appropriate for selection as a VEC but for which there are too few data to characterize its numbers, distribution, variability, and so on should not be selected as a VEC simply because it cannot be used to characterize the health or recovery of an ecological system. This is particularly true if alternate VECs exist that reflect similar attributes of the ecosystem.

Similarly, a species that has a very small, isolated population may not be appropriate for selection as a VEC in the case of a single-event stressor such as EVOS. Consider the cutthroat trout (*Salmo clarki clarki*): At PWS it is at the extreme northern limit of its population distribution, which extends down through North America, mostly west of the Rocky Mountains, to California, Arizona, and New Mexico, and occurs in fresh, brackish, and saltwater. In PWS it occurs only in small, genetically distinct populations, located in small and isolated areas of the region. Thus, its numbers are quite small in PWS because it is at the upper limits of the conditions that it can survive in, making it particularly vulnerable to climatic variability. This species might be a good indicator for assessing the effects of global warming, as it plausibly could migrate further north as climate changes. But it would not seem to be a good indicator of an acute, single toxic event stressor because it would be very difficult to attribute a reduction in its very small population (even if that were detectable) to a single event. The point is not to suggest that a small population should never be used as a VEC, but simply that a small population is not a good indicator of a unique physical and toxic stressor like EVOS, especially if there are sufficient other species (other anadromous fish species, in this case) in PWS that can serve as VECs.

In assessing the ecological significance of an effect on a VEC, it is important to distinguish between a biological response and an ecological effect. An organism exposed to a stressor may show a biochemical, genetic, metabolic, behavioral, or other suborganismal- or individual-level response, but for there to be an ecologically significant effect, the biotic responses must collectively rise to the level of an effect at the population or higher level of ecological organization (Gentile and Harwell 1998). For example, a biochemical response in an individual exposed to a chemical stressor does not by itself constitute an ecological effect unless the biochemical response translates into a larger effect on survivability, reproductive ability, or other attribute that is emergent at the population level. Similarly, an individual-level response is ecologically important only if so many individuals in the population are affected that there emerges a population-level effect.

That does not mean there is no value in understanding biochemical or other suborganismal or individual responses. To the contrary, such responses may be appropriate indicators of exposure. Indeed in some cases biochemical responses (biomarkers) may be a much better indicator of exposure than measuring the chemical itself. But it does signify that detecting a biochemical or individual-level response does not necessarily mean there is an ecological effect. Most importantly, if the biochemical response or individual-level response is all there is and there are no changes at the population level or higher, then we assert that there is no significant ecological effect. The exception to this rule is where the VEC is a species for which individuals by themselves are societally important, for example, a marine mammal (e.g., individual orca) or an endangered species that has extremely low population numbers (e.g., individual California condor). Even in these cases, however, if the individual experiences only a biochemical response but otherwise is not affected with respect to behavior, longevity, hereditable genetics, reproductive potential, or other health metric, then there is only a response, not a significant effect.

Where multiple simultaneous stressors affect an ecological attribute, distinguishing ecological effects caused by 1 particular stressor from effects from other stressors or from natural variability even in the absence of stressors is a difficult task. To be diagnostic, causality between the observed stressor and the observed ecological effect needs to be established, requiring best professional judgment based on multiple lines of evidence. Another issue is multiple sources of the same stressor; in our case, source means the physical remnants of EVO somewhere in PWS that may continue to release PAHs into the environment, or it may mean another source of PAHs having nothing to do with EVOS, such as releases from natural seeps or abandoned industrial sites.

The criteria for determining ecological significance (Gentile and Harwell 1998) include 1) whether a change in a VEC is of sufficient type, intensity, extent, and/or duration to be important to the structure, function, and/or health of the system and 2) whether such a change exceeds natural variability or alters the natural variability regime. Both conditions are necessary and jointly sufficient for determining whether potential or measured ecological change can be regarded as ecologically significant.

A central tenet of determining ecological significance is that if a measured change in a VEC is considerably less than the normal range of natural variability for that VEC metric, then there can be no determination of ecological significance. This implies that the VEC is measurable either directly or indirectly and that there is a database that is either currently available or that can be obtained that will permit the assessment of the magnitude and direction of change in the VEC with an acceptable degree of confidence (USEPA 1998). If available data are insufficient to allow direct measurement of a change in the VEC, then those VECs are categorized as "unknowable" and should not be used to assess recovery or ecological significance.

Uncertainty is a fundamental component of all assessments that needs to be mentioned. While it is not our intent to characterize the uncertainty associated with the lines of evidence used in this study, it is important to recognize that the uncertainty will vary depending on the type of parameter being measured, differences among taxa, the sampling design and sample number, and extrapolations across species, from lab to field, and across spatial scales. Because it is not feasible to quantify all sources of potential uncertainty, we use a multiple-lines-of-evidence approach to reach our conclusions, following the USEPA guidelines (USEPA 1998). We recognize that the uncertainties in some lines of evidence are likely to be greater than others, but when viewed in the aggregate, we believe that the evidence presented here leads in almost all cases to a clear and defensible interpretation.

There are 2 further issues that need to be considered briefly: data sources and scale. In selecting information for each of the multiple lines of evidence, we examined scientific articles published in the peer-reviewed literature and a large body of information about EVOS developed by both Trustee- and/or Exxon-funded scientists. Obviously, we were limited by the availability of existing data, extensive as it is in this virtually unprecedented body of ecological research. Where there are scientific differences in opinion and interpretation of some of the data relevant to EVOS (and there are many), we have mostly left those debates to others and used available data as published. We have also noted differences in reported information in the literature and described the basis we used for reaching our conclusions.

The other important issue in addressing ecological significance, both of exposures and effects, is that of scale. This comes into consideration in several aspects, including spatial scale, temporal scale, organizational scale, and ecological scale. Spatial scale relates to the absolute and relative extent of exposure, as discussed in more detail in the following section, but also to the natural heterogeneity of the VEC of concern. For example, clams have a highly heterogeneous distribution at relatively small scales, determined largely by physical parameters such as sediment particle size, wave energy, exposure, and currents. Ideally, spatial heterogeneity needs to be well understood prior to designing a sampling regime to assess the status of a population like clams, but that was not always the case. This issue is particularly important in using a reference site approach for assessing the effects from EVOS (see Parker and Wiens 2005). If a VEC is spatially heterogeneous, then extra care needs to be taken in selecting reference sites to be used as surrogates of the oiled site in the absence of oiling. For example, as discussed later, this issue is a problem for assessing effects of EVOS on harlequin ducks in PWS: oiled and unoiled site comparisons have been used by some researchers to conclude continuing EVOS effects, when the data seem to indicate sufficient differences before EVOS in abundances between the oiled and reference sites to explain current patterns.

Similarly, temporal scale can be important in assessing ecological effects. Obviously, given enough time, PWS or any other ecosystem will change, but our time frame for assessing ecological recovery is the period from the aftermath of the EVOS event up to the present, some 17 y later. But longer-term processes prior to the spill must be taken into consideration, such as the multidecadal trends of decline in harbor seals throughout the region starting well before EVOS. Intrinsic time lags in the VEC recovery response may also be critical. An important case in point is the VEC killer whale, where an acute response of increased mortality in an exposed pod may take many years to diminish because of the long life spans, low fecundity, and social dynamics of the species.

By organizational scale, we refer to addressing effects on levels of organization higher than the individual (except for orcas) in order for there to be an ecologically significant effect; this issue is discussed in more detail elsewhere. However, a similar issue that also relates to spatial and temporal scales is referred to here as ecological scale. By that we mean issues like deciding on the particular population or subpopulation that is appropriate to assess.

For example, for the sea otter, as discussed later, 1 subpopulation on northern Knight Island may have numbers lower than 1 unpublished prespill estimate. Even if one accepts the numbers, the issue remains whether this is the appropriate scale to assess sea otter effects from EVOS. We argue, in this case, that since the lifetime home range of individual sea otters covers essentially all of PWS and since there is natural heterogeneity in otter numbers at small spatial scales and over time, then the appropriate ecological scale to assess EVOS impacts on sea otters is the total, PWS-wide population. By contrast, resident killer whales in PWS are organized into essentially permanent discrete pods, one of which was directly exposed and affected by EVOS. The PWS-wide population of resident killer whales has significantly increased since 1989, and we conclude that the PWS-wide population has recovered. But the AB pod has not yet recovered, so we characterize that as a continuing ecologically significant effect from EVOS.

Finally, Gentile and Harwell (1998) have proposed the use of reasonable expert judgment to assess ecological significance using criteria they developed. The intent is not to rely upon rigorous or quantitative metrics for determining ecological significance because, in most cases, there are insufficient data on one or more VEC for such an exercise. Prince William Sound is a case in point: hundreds of millions of dollars have been spent in research on ecological attributes selected by the Trustees for recovery assessment, but in very few cases are there sufficient data to make a quantitative statistical assessment of the dynamics of the attribute's response since EVOS. In the majority of cases, requiring precise quantitative metrics in order to make judgment is impractical. However, we assert that reasonable, objective, scientific judgments can be made based on Gentile and Harwell (1998) criteria and using a multiple-lines-of-evidence approach. Further, we believe that by making the criteria and the information used to make judgments transparent, others who might reach different conclusions would have a common basis for discussion.

One final point to keep in mind: there can be no absolute conclusions about ecologically significant effects. If one requires precise answers before making judgments about ecological significance in a complex environmental issue, then one may have to wait a very long time, particularly for large-scale issues such as climate change, successful restoration of the Everglades, or recovery of PWS from EVOS. Our approach, using multiple lines of evidence and a reasonable application of assessment criteria, we believe, allows defensible judgments to be made in the face of uncertainties, natural spatial and temporal variability, multiple stressors, and limited information.

EXPOSURE CHARACTERIZATION

Exposure characterization is the component of the ecological assessment framework (USEPA 1992, 1998) that is designed to predict or measure the temporal and spatial distribution of the stressor, its co-occurrence with the ecological components of concern, and its modification by biotic and abiotic factors. Issues related to the characterization of exposure include the following:

1. Identifying and quantifying the sources of stress, whether physical, chemical, biological, anthropogenic, or natural;
2. The spatial and temporal scales of exposure, including the frequency, duration, and persistence of the stressor at a given location;
3. Transformations of the stressor in the environment (e.g., chemical modifications from physicochemical or biological processes);
4. Bioavailability of the stressor (e.g., biouptake of chemicals and their transformation products); and

5. Interactions with the physical environment (e.g., wind and tidal conditions).

Quantifying exposure, whether to a population of a species or to an entire benthic community, requires the establishment of an explicit linkage between source(s) of the stressor, the spatial and temporal concentrations of the stressor, the stressor's co-occurrence in time and space and for sufficient duration with an appropriate ecological receptor, and some measure of the receptor's exposure (Gentile et al. 1993; USEPA 1998). From an exposure perspective, sources are considered "repositories" of stressors and are generally not in direct contact with the ecological receptor. Conversely, the stressor, which results from the mobilization and release of materials constituting the source, is in direct contact with a biological receptor and is responsible for eliciting a response in the receptor.

Measures of exposure can include concentrations in the surrounding medium or measures of biological responses (e.g., biomarkers, toxicity or body burdens). Biomarkers here refer to suborganismal-level responses (e.g., biochemical, physiological, and histological) that indicate that the receptor has been in sufficient contact with the stressor to elicit a response.

Assessing the significance of a chemical stressor's exposure profile in a regional ecosystem involves understanding the magnitude and distribution of sources, the spatial and temporal extent and intensity of the stressor in the environment, and its persistence and bioavailability. We have used a multiple-lines-of-evidence approach to evaluate the potential causal links between residual EVOS sources and observed biological/ecological responses. The objective of this approach is to address the issue of whether the residual EVO is a continuing and significant source of hydrocarbon exposure in PWS. Note that this approach follows the USEPA (1990) guidelines document, which explicitly addresses applicability to multiple stressors and recommends, given the intrinsic uncertainties associated with natural multistressed environments, using a multiple-lines-of-evidence approach (see EPA Guidelines Risk Description: Section 5.2.1).

Estimating the magnitude and extent of residual sources

Hydrocarbons derived from natural and anthropogenic sources are widespread in the marine coastal environments (Brassell and Eglinton 1980). Petrogenic hydrocarbon sources in PWS include eroding Tertiary sedimentary rocks and associated oil seeps in the eastern GOA, commercial and pleasure vessel operations (e.g., cruise liners, fishing fleet, Alaska marine highway ferries, oil tankers, and so on) (Page et al. 1995; Bence et al. 1996), and inputs to sediments from diesel and asphalt spills (Short et al. 1999, 2004; Page, Bence, et al. 2002). Pyrogenic inputs include combustion products from burning of coal, fuel oil, and wood and atmospheric inputs from forest fires and global industrial sources (Page et al. 1995; Page, Boehm, et al. 1999). In addition, there is a wide range of current and historic activities that release petrogenic and pyrogenic hydrocarbons into PWS, including recreation, fish processing plants, sawmills, mining, and the rupture of petroleum storage tanks during the 1964 earthquake (Kvenvolden et al. 1993; Wooley 2002).

Characterizing and quantifying the proportional contribution of the multiple sources and their stressors constitute a major challenge, particularly in PWS. While there has been considerable research on identifying sources, there appears to be no systematic, quantitative estimate or comparison of the total mass contributions from these sources that can be compared with residual EVO. Consequently, the wide variation and uncertainty in the mass contributions of these multiple sources to the coastal environment of PWS, particularly subtidal sediments, have made it difficult to determine their relative contributions with an acceptable degree of confidence. At issue is the relative proportion of the sum of the natural and anthropogenic sources to the recently estimated quantity of residual EVO remaining in the intertidal and nearshore subtidal sediments.

The seminal question is not whether sources of hydrocarbons from EVOS continue to exist, as clearly they do, but whether they pose a significant risk to the populations and communities comprising the PWS ecosystem. What is needed to address this question is a comparison of the current contributions of the various sources of hydrocarbon exposure to the PWS ecosystem. Assessing the importance and significance of the residual EVOS in a multi-source and -stressor environment, in the absence of quantitative information on the mass inputs from all sources, requires a multiple-lines-of-evidence approach that seeks to characterize the proportional contribution of the various stressors in PWS.

Subtidal sediments—Hydrocarbons sequestered in subtidal sediments have been hypothesized to provide a continuous chronic source of contamination to the intertidal and nearshore subtidal biological communities of PWS (Short et al. 2004). Three questions regarding the contaminant composition of these sediments need to be addressed: 1) Can different prespill sources of petrogenic PAHs be identified (i.e., fingerprinted) in PWS sediments? 2) Can the mass contribution of these multiple hydrocarbon sources (historic and current) be estimated? And 3) is the oil remaining from EVOS a significant source of PAHs when compared to other pre- and post-spill sources, both natural and anthropogenic? Distinguishing the individual contributions among these multiple sources is essential to evaluating current and future risk.

Studies conducted in PWS and GOA prior to EVOS indicated that petrogenic hydrocarbons exist in subtidal sediments (Kaplan and Venkatesan 1981). Four sources dominate the petrogenic hydrocarbon inputs to the PWS marine environment:

1. Eroding Tertiary sedimentary rocks (petroliferous shales) and associated natural oil seeps in the eastern GOA coast;
2. Oil from EVOS and other ships in PWS;
3. Diesel refined from Cook Inlet Crude and Alaska North Slope Crude; and
4. Heavy Monterey fuel oil and asphalt released to the Sound in the 1964 earthquake (Kvenvolden et al. 1995).

A further confounding factor is the hydrocarbons from the eroding sedimentary rocks and natural seeps contain the same suite of PAH compounds found in EVOS. Fortunately, however, the relative amounts of individual PAHs are different, and therefore state-of-the-science chemical analyses can be used to fingerprint the sources of the PAHs (Boehm et al. 1997; Burns et al. 1997).

In their comprehensive analysis of PWS subtidal sediments, including using PAH-fingerprinting tools, Page et al. (1995, 1996) concluded the following:

1. Eroding shales and natural oil seeps from the Katalla eastward to Yakutat Bay and along the northern coast of

the eastern GOA provide a region-wide PAH background;
2. Specific locations associated with past industrial activity in PWS have elevated levels of PAHs from both pyrogenic and petrogenic sources not associated with EVOS (Page, Boehm, et al. 1999; Page, Bence, et al. 2002);
3. The EVOS-derived PAHs found in 1989–1991 occurred in shallow, nearshore, subtidal sediments off some shorelines that had been heavily impacted by the EVOS; and
4. Most of the seafloor in PWS at present has no detectable PAH attributable to the EVOS.

Further, 32 of 84 nearshore sediment samples collected from sites that had been heavily oiled in 1989 had detectable residual EVOS residues in 1991 (Page et al. 1995). Of these 32 samples, 30 had total PAH (TPAH) levels less than 1.0 ppm, and 19 had levels less than 0.1 ppm (Boehm et al. 1995; Page et al. 1995). These results suggest that the vast majority of the approximately 4.5×10^3-km^2 seafloor area of PWS had no detectable traces of EVOS. This contrasts with the natural petrogenic PAH background concentrations in subtidal sediments, which are present at levels in excess of 1.0 ppm.

Where currently present, EVOS residues constitute a relatively small addition to the natural petrogenic background at specific nearshore (10–50 m) locations. This is evident from the distribution of PAH source components in PWS and eastern GOA subtidal sediments (Page et al. 1995, 1996). Clearly, the nearshore, embayment, and offshore sediments in the spill zone show only a marginal increase in EVOS-derived PAHs, with the highest percentage contribution being in the nearshore sediments (<50 m). It is also worth noting that the proportion of natural petrogenic-derived PAHs is highest (>2.0 ppm) in the eastern GOA nearest the streams along which there are natural seeps (Page et al. 1996; Boehm et al. 2001). The importance of oil seeps and eroding petroliferous shales as primary sources of hydrocarbons in subtidal sediments has been challenged by Short et al. (1999), who suggested that coal should be included as a plausible source of hydrocarbons in PWS. However, mass balance considerations and statistical analysis of hydrocarbon fingerprints independently indicate that coal contributes less than 1% of the PAHs and chemical biomarkers in this background (Boehm et al. 2001). While there are coal particles in the sediments of PWS, their PAH and chemical biomarker contributions are overwhelmed by those of seep oil residues and organic particles (e.g., kerogen) from the eroding shales. Huggett et al. (2003) have also shown that some of these PAHs are bioavailable. Therefore, the dominant sources of petrogenic hydrocarbon background in benthic sediments of PWS are eroding Tertiary shales and residues of natural oil seeps (Page et al. 1995; Bence et al. 1996).

Intertidal sediments—In 2001, a shoreline survey of 91 sites estimated the vertical distribution of oil remaining in western PWS areas that were heavily or moderately oiled in 1989 by EVOS (Short et al. 2002, 2004). Of the 4,249 quadrats sampled between +1.8 and +4.5 m above mean low water, 255 quadrats had very light to light subsurface oil residues, 72 quadrats had moderate subsurface oil residues, and 20 had heavy subsurface oil residues. The authors estimated that the beach surface area contaminated by subsurface oil in 2001 was 6.7 ha and, if surface tar projections were included, about 11.1 ha. This survey also determined that the mean mass of oil per unit area of beach ranged from 0.59 kg·m^{-2} for lightly oiled residues to 2.1 kg·m^{-2} for heavily oiled residues, which resulted in an estimate of 55,600 kg (approximately 65,000 L, or 16,400 gallons [US]) of total residual EVO. The residual volume of oil estimated from this 2001 study represents about 0.0014 to 0.0028 of the volume originally beached, and with an annual loss rate since 1992 of approximately 20% to 26% (Short et al. 2004), this suggests that the fraction remaining in 2006 is about 0.0003 to 0.0009.

Human activity sites—Recently, there has been an increasing awareness of the importance of sites associated with historical human activity (HA sites) as potential sources of hydrocarbons. Boehm et al. (2004) reported TPAH concentrations ranging from 2 to 12,056 ng·g^{-1} for mussel samples from 14 sites not oiled in 1989 but affected by past industrial activity, indicating that residual PAH from historical activities constitutes a continuing source of bioavailable PAHs in PWS. The TPAH values from HA sites were at least an order of magnitude greater than non–human activity sites, whether oiled in 1989 or not. A comprehensive survey in 2003 of 9 of these HA sites found ~3.6 ha (9 acres) of PAH-contaminated sediments with TPAH values in excess of 2,500 ppb (Page et al. 2004). More than 50 abandoned HA sites were identified, mostly within the spill zone in western PWS, and the total area of PAH-contaminated sediments was greater than the ~3.6 ha actually mapped and quantified (Page et al. 2004).

Assessing the relative importance of residual EVO as a hydrocarbon source—Spatial extent of contamination is one of the important lines of evidence that can be used to estimate the significance of residual EVO in PWS. Although absolute areal extent of the system at risk is 1 of the 1st spatial issues to be addressed, a small spatial impact should not be summarily disregarded until its role and/or use of the area (e.g., refugia or critical habitat) has been defined. Those issues do not seem to be relevant here. The lines of evidence chosen to assess the significance of residual EVO as a hydrocarbon source in PWS include 1) the linear extent of residual oiled shoreline (Wolf et al. 1994; Neff et al. 1995), 2) the areal extent of residual sediment contamination, and 3) the mass of residual EVO (Short et al. 2004).

Changes in the length of oiled shoreline in PWS from 1989 to 1992 are summarized in Table 1. The postspill shoreline oiling survey conducted in 1989 estimated that approximately 783 km (~16%) of the 5,000 km of PWS and 1,315 km of GOA (~14%) shorelines were oiled to varying degrees (Owens 1991; Neff et al. 1995). By 1992, only 10 km of PWS shoreline (0.21%) remained visibly oiled, 95% of which was classified as very light to light. Ultimately, only 2% of the volume of oil spilled initially remained on PWS beaches by the end of 1992 (Wolfe et al. 1994).

Using 783 km for the PWS oiled shoreline in 1989, of which 25% or about 200 km are beaches (Neff et al. 1995), estimates of intertidal area for the initially oiled beaches in PWS range from 2 to 10 km^2. Using the 2001 estimates for the residual areal extent of subsurface EVO of 11.3 ha (0.1 km^2; Short et al. 2004), the proportion of residual EVO to previously oiled shoreline is 1.25% to 0.25%, and the proportion of residual EVO to total PWS shore is 0.2% to 0.045%. Further, if we use the 10-km linear shoreline contaminated with residual EVO in 1992 estimated by Neff et al. (1995), the estimated upper limit of areal extent of residual EVO is between 0.10 and 0.50 km^2, which is consistent with the Short et al. (2004) estimate of approximately 11 ha and represents 1.4% of the originally oiled area in PWS.

Table 1. Linear distance (km) of oiled shoreline in Prince William Sound, Alaska, USA, 1989–1992 (from Neff et al. 1995)

Year	Oiling category[a]				
	Very light	Light	Moderate	Heavy	Total visibly oiled shoreline
1989	223	326	94	141	783
1990	323	80	46	21	420
1991	68	15	12	0.1	96
1992	8.7	0.8	0.6	0.2	10

[a] Shorelines were classified into 1 of 4 categories of visible oil status: Heavy = >6 m width and >50% oil coverage; moderate = >6 m width and 10–50% coverage, or 3–6 m width and >10% coverage; light = <3 m width and >10% coverage; and very light = any width <10% coverage.

On the basis of these estimates of the linear extent, spatial extent, and mass of residual EVO in PWS, we conclude that the magnitude of residual EVO as a continuing source of PAHs to PWS is a small fraction of 1% of what was available immediately after the spill. The areal extent and mass of residual EVO are a small fraction of the initial postspill amount, perhaps less than that contributed by HA sites, and decreasing by ~25% annually since 1992. However, while the total inputs of EVOS-derived PAH have been decreasing (Short et al. 2003), the PAH contribution from the HA sites remains chronic in nature because these sources are essentially terrestrial, occurring well above the beaches and thereby protected from wave action.

Estimating the ecological significance of residual EVO exposure

The extent of HA sites contaminated with PAHs (Page et al. 2004) is similar to the areal extent of residual EVOS estimated by NOAA (Short et al. 2004). The presence of a source of potential chemical stressors (e.g., EVO, oil seeps, coal) does not necessarily mean, however, that the source and material is either available for transport or bioavailable to induce direct or indirect toxic effects. Furthermore, even if there is demonstrated toxicity under the conditions of an experimental design in the lab, these observed effects must be extrapolated to the field and then placed within the larger ecological context that addresses the issues of scale and complexity. This is not meant to imply that "dilution is the solution to pollution"; rather, one has to place events in a population- or ecosystem-level context. Organisms often experience short-term adverse effects from disease or stress, yet their adaptive mechanisms and resiliency generally allow for a quick uneventful recovery. Unlike human health, where the focus is on the individual, ecological effects must be manifested at the population level in order to raise concern (except as noted previously for certain endangered or protected species, such as, in PWS, killer whales). Therefore, we have to look at the scale, duration, intensity, and persistence of exposure to ecological systems in order to interpret the significance of hydrocarbon stress in PWS.

Concentrations of EVO declined rapidly in both sediments and biota within a few years after the spill (Boehm et al. 1995; Murphy et al. 1999; Carls et al. 2001; Dean and Jewett 2001). In the previous sections, we identified the magnitude and spatial extent of residual sources of contamination that are critical to evaluating the importance of residual EVO. The purpose of this section is to examine multiple lines of evidence to infer the current magnitude and extent of persistent residual EVO exposure in PWS, drawing on sediment toxicity bioavailability and biomarker data. The intent is to view these data as lines of evidence to be used to accept or reject the hypothesis that residual EVO is a continuing source of PAH exposure that continues to pose a threat to PWS biota.

Sediment toxicity—There has been considerable study of the magnitude and extent of hydrocarbon contamination and its associated toxicity by both industry- and Trustee-sponsored scientists. An 11-y study (Page, Boehm, et al. 2002) sought to assess the initial levels of spatial impact, the effects of weathering on hydrocarbon chemistry and toxicity, the impacts on shoreline ecology, and the time frame for recovery. The results from this study show that by 1999 the potential for adverse toxic effects to the amphipod *Rhepoxinius abronius* at the 1989 worst-case sites from the remaining residues was negligible (Page, Boehm, et al. 2002, table 8). This toxicity test, along with sediment chemistry, has been widely used and accepted as an indicator of benthic community health (Long et al. 1988). The results from Page, Boehm, et al. (2002) are consistent with the generally rapid recovery of the shoreline biota observed in 1990–1991 (Gilfillan, Page, et al. 1995; Gilfillan, Suchanek, et al. 1995). The 1999 median sediment TPAH concentration at the 1990 worst-case sites was 117 $ng \cdot g^{-1}$, which is well below the threshold of 2,600 $ng \cdot g^{-1}$ for sediment toxicity threshold (Swartz 1999). These results indicate that widespread toxic effects in PWS from intertidal sediment hydrocarbon exposure are very unlikely and limited to a small number of low-energy marshy sites with high organic content. This means, as discussed more fully later, that present (2006) EVOS-derived PAH exposures do not constitute a significant risk to PWS VECs that function at larger spatial scales (e.g., marine birds, fish populations, orca).

Bioavailability, bioaccumulation, and removal of sediment hydrocarbons in mussels—An important key measure of residual risk is the potential bioaccumulation and trophic transfer of contaminants from sediments to important prey species and then to their predators. Concerns have been expressed that bird and wildlife populations foraging in the intertidal zone could be affected by ingesting contaminated prey (e.g., Trust et al. 2000; Bodkin et al. 2002). In the years following the spill, field studies were conducted to monitor the remaining shoreline oiled sites to assess both their residual toxicity and their bioavailability (Stubblefield et al. 1995; Babcock et al. 1996; Boehm et al. 1996, 2004; Hoff and Shigenaka 1999; Page, Gilfillan, et al. 1999; Carls et al. 2001; Carls, Harris, et al. 2004; Page et al. 2005). A particular focus was on PAH concentrations in the soft tissues of blue mussels (*Mytilus trossulus*), an organism that bioaccumulates PAHs.

This important intertidal prey species in PWS, which, along with other bivalves and crustaceans, forms a major component of the diet of intertidal foragers, is used as an indicator of the bioavailability of intertidal sediment hydrocarbons from oiled and unoiled sites in PWS.

Generally mussel beds were not cleaned after EVOS, resulting in these habitats potentially constituting a long-term continuing source of hydrocarbon contamination. Monitoring studies of mussel beds with significant contamination from 1992 to 1995 found TPAH concentrations as high as 8,100 $ng \cdot g^{-1}$ (dry wt; Babcock et al. 1996; Boehm et al. 1996; Carls et al. 2001). The implication of these results was that the oil-contaminated mussels and associated sediments posed a potential long-term risk to birds and wildlife foraging on intertidal mussels. In 1994, experimental restoration of 9 contaminated beds was conducted, involving the removal of contaminated surface sediment and the replacement of both sediment and mussels. The results of the 5-y postrestoration monitoring of mussel beds that were heavily oiled in 1989 indicated that by 1999 mussel TPAH concentrations were typically at baseline levels in both restored and nonrestored mussel beds (Carls, Harris, et al. 2004).

Boehm et al. (2004) documented that PAH levels in mussels from PWS had returned to the same as or near reference or baseline levels as defined by the US National Status and Trends Program (Daskalakes and O'Connor 1995). Sites were sampled during 1998–2002 at locations where remnants of subsurface EVO were known to persist based on postspill sampling. Seventy-two locations were sampled within the 1989 spill zone, as well as 28 sites not associated with human activity or oiling and 14 sites known to have human activity but not oiled. Mean TPAH for all non–human activity reference site mussels was 48 $ng \cdot g^{-1} \pm 62$ with a range of 3 to 355 $ng \cdot g^{-1}$, while that for human activity sites was 677 $ng \cdot g^{-1}$, with a range of 3 to 12,056 $ng \cdot g^{-1}$ (Table 2). Ninety-three percent (203 of 218) of the mussel samples from sites oiled in 1989 had TPAH concentrations less than 300 $ng \cdot g^{-1}$, with a mean of 100 $\pm$ 138 $ng \cdot g^{-1}$. Further, none of the mussel samples collected in 2002 from 17 sites where subsurface oil had been identified in the 2001 shoreline survey had TPAH concentrations exceeding 100 $ng \cdot g^{-1}$ dry wt, even though the sediment TPAH values ranged over 5 orders of magnitude (Boehm et al. 2004, figure 4). These results indicate that the likelihood of subsurface EVO hydrocarbon residues being bioavailable to mussels is low.

These results were confirmed by Page et al. (2005), who analyzed data for the concentrations of bioavailable PAH in mussels collected between 1990 and 2002 from 11 PWS sites that were heavily oiled in 1989. Page et al. (2005) found that concentrations of PAHs released from spill remnants decreased dramatically with time and by 2002 were at or near the background range of total TPAH of 3 to 355 $ng \cdot g^{-1}$ dry wt obtained for mussels from unoiled non–human activity sites in PWS (Boehm et al. 2004). These TPAH concentrations are well below those known to cause harm to intertidal foragers, using the minimum-effects criteria established by Carls et al. (2002). The time-series TPAH data indicate a mean TPAH half-life in mussel tissues of 2.4 (1.4–5.3) y, with an annual mean overall loss of bioavailable TPAH at the sites of 25%, which is consistent with loss rates derived by Carls, Harris, et al. (2004). The petroleum-derived TPAH fraction in mussel tissues has decreased with time, reflecting the decreasing bioavailability of EVO residues in shoreline sediments. These results show that PAHs from EVO residues that remain buried in shoreline sediments are in a form and at locations that have a low accessibility and bioavailability to marine organisms living on the shore or foraging in the intertidal zone and thus do not constitute a significant contaminant exposure pathway to foragers, such as mussel predators.

The PAH distributions in mussels from HA sites within western PWS not oiled in 1989 reflect chronic inputs of PAHs that are a legacy of the human activities at those sites prior to their abandonment before 1989. Mean TPAH values for these sites are 5 to 100 times greater than the minimum adverse effect threshold of 1,000 $ng \cdot g^{-1}$ in mussel tissue sufficient to predict abnormalities in herring larvae exposed to PAH in water as reported by Carls et al. (2002). Furthermore, the PAH profiles in mussel tissues at these sites were enriched in 4- to 6-ring pyrogenic PAHs compared to 2- and 3-ring primarily petrogenic PAHs. The higher–molecular-weight PAHs are known to be more potent inducers of cytochrome p450A (CYP1A) activity than the 2- and 3-ring PAHs that dominate the PAH distributions in residual EVO (Fent and Bätscher 2000). These PAH data indicate that the background condition for PWS is one of continuous exposure to low-level petrogenic and pyrogenic PAHs, and prior HA sites in western PWS remain local chronic contributors.

Bioavailability and bioaccumulation of sediment hydrocarbons to intertidal species—Spatial scale is an important consideration when evaluating the significance of exposure. Sites containing isolated EVO residues as well as combined sites with historical human activity represent a small percentage of the areal extent and volume of residual hydrocarbons of the western PWS shoreline. In addition, source reduction rates of 25% annually from EVOS sources suggest that it is unlikely that these sources constitute a significant bioavailable source of hydrocarbons causing risk to the ecology of PWS at this time.

Neff et al. (2006) surveyed 17 intertidal shoreline sites in PWS that included sites that were heavily oiled in 1989 by the EVOS and surveyed by Short et al. (2004), 3 unoiled reference sites, and 4 unoiled HA sites. The objective of this study was to determine if PAHs from buried oil residues in intertidal sediments were sufficiently bioavailable to intertidal prey organisms to pose a risk to foraging birds and wildlife populations. The results, summarized in Table 3, indicate that the overall bioavailability of residual EVO to a wide range of intertidal species is low and, though consistently higher than values for the reference sites (hermit crabs excepted), is not statistically different from the reference values. Values from HA sites are significantly higher than values from both previously oiled sites and reference sites, indicating that these sites are a potentially more important source of ongoing exposure than residual EVO sites. Mussels, clams, and worms collected from oiled sites contained both petrogenic and pyrogenic PAHs, while surface-dwelling intertidal fish, hermit crabs, whelks, and sea lettuce contained primarily pyrogenic PAHs. The low petrogenic PAH concentrations detected in intertidal biota at some oiled sites indicate that the PAHs from subsurface EVO residues persisting in intertidal sediments have a low accessibility and bioavailability to intertidal plants and animals. Consequently, the residual EVOS-derived PAH concentrations in intertidal organisms represent a low risk to populations of marine birds and mammals, such as oystercatchers, harlequin ducks, and sea otters, which forage in the intertidal zone (Neff et al., 2006).

Table 2. Summary of mussel tissue TPAH concentrations (ng·g^{-1} dry wt) in Prince William Sound, Alaska, USA, 1998–2002 (from Boehm et al. 2004)[a]

	Sites oiled in 1989	Unoiled HA sites	Unoiled NHA sites
Samples (*n*)	218	36	37
Arithmetic mean ± SD	100 ± 138	677 ± 2035	48 ± 62
Highest TPAH value	1,190	12,056	355
Lowest TPAH value	2.4	3.0	3.0

[a] HA = human activity; NHA = non–human activity; SD = standard deviation; TPAH = total polycyclic aromatic hydrocarbon.

Biomarkers of exposure—Biomarkers are becoming an increasingly useful tool for determining exposure of a variety of biota to specific classes of environmental stressors. Jewett et al. (2002) examined the spatial distribution and intensity of response of 3 biomarkers specific for hydrocarbon exposure in 2 species of intertidal/nearshore fish collected in PWS approximately 7 to 10 y after EVOS. The CYP1A in liver vascular endothelium, liver ethoxyresorufin-O-deethylase, and biliary fluorescent aromatic compounds were measured in masked greenling (*Hexagrammos octogrammus*) and crescent gunnel (*Pholis laeta*). These are both numerically dominant species nearshore and are important members of the food web. The 3 biomarkers were measured to determine whether these fish were exposed to hydrocarbons, and if so, to assess the spatial extent of exposure and to infer the possible sources of contamination. The authors indicated that all 3 biomarkers were elevated in fish collected from sites that were originally oiled by EVOS as compared to fish from unoiled sites.

Data collected in 1998 show that CYP1A in liver vascular epithelium measured by immunohistochemical scores, though low, differed significantly between sites for gunnel and greenling, but there clearly were differences between the responses of the species to known sources of EVO contamination. For example, Bay of Isles and Herring Bay samples were significantly different from all other oiled and non-oiled sites for greenling, yet exposure biomarkers for gunnel were not detected in Herring Bay. Fish were collected in 1999 from sites near intertidal mussel beds that were known to have high PAH concentrations in 1995 and from sites that were within the spill zone but unoiled. CYP1A immunohistochemical scores for greenling did not differ significantly between sites. The mean scores for gunnel were significantly different for oiled versus unoiled sites. However, there were no significant positive correlations between sediment hydrocarbon concentrations and any of the measured biomarkers in either species (Jewett et al. 2002).

It is clear from the results that while statistical differences were noted between oiled and unoiled sites, the levels of CYP1A and ethoxyresorufin-O-deethylase response were low and unrelated to the sediment PAH concentrations at either Bay of Isles or Herring Bay, and the patterns of response were inconsistent both across species and across biomarkers. Jewett et al. (2002) noted that the low levels of CYP1A and ethoxyresorufin-O-deethylase suggest that the exposure of the animals to residual hydrocarbon was low and that other potential sources of PAH exposure than EVOS probably existed.

A subsequent exchange of letters to the editor of the journal *Marine Environmental Research* between Jewett et al. (2003) and Boehm et al. (2003) identified 2 relevant points regarding the issue of exposure. First, while there have been several published studies documenting the potential sources of PAHs in PWS and GOA, there has been no systematic quantification of those sources so as to provide a context for evaluating the residual EVO sources. Second, when CYP1A levels of expression are as low as observed in this study, it makes assigning a specific causal source extremely difficult and uncertain. It is clear that while the biomarkers did show a positive response to the presence of PAHs, the low intensity of the responses, differences between species, and absence of correlation with levels of source contamination suggest that the exposures were quite low and difficult to assign to a particular source.

Huggett et al. (2003) tested the hypothesis that the biomarker levels of fish collected from PWS sites impacted by EVOS would be higher than those collected at unoiled

Table 3. Concentrations of polycyclic aromatic hydrocarbons (ng·g^{-1} dry wt) in intertidal plants and animals from Prince William Sound, Alaska, USA (from Neff et al. 2006)

	Sites oiled in 1989 (*N* = 17)	Unoiled reference sites (*N* = 3)	Unoiled human activity sites (*N* = 4)
Mussels	23.5 ± 18.3	14.4 ± 16.5	1,362 ± 2,369
Clams	26.9 ± 32.3	15.2 ± 7.0	1,651 ± 2,774
Worms	36 ± 38.6	10 ± 6.8	565 ± 879
Fish	22.4 ± 59.6	6.4 ± 4.9	108 ± 165
Hermit crabs	4.7 ± 2.6	7.7 ± 7.3	63.6 ± 82.7
Sea lettuce	11.3 ± 8.0	9.8 ± 5.1	2,209 ± 2,811
Whelks	5.3 ± 4.1	3.3 ± 1.8	5.7 ± 6.5

sites. The 1999–2000 study examined 5 species of fish (Pacific halibut, Pacific cod, rockfish, rock sole, and kelp greenling) and associated benthic sediments collected from 21 oiled and unoiled sites in PWS and reference sites in the eastern GOA. Fish were assayed for bile fluorescent aromatic compounds and CYP1A levels. The results indicated that for all species and all sites, the bile fluorescent aromatic compound and CYP1A levels were low and in the same range for fish collected from non–spill path sites, spill-path oiled sites, and spill-path unoiled sites in PWS and GOA. Further, the bile fluorescent aromatic compound results indicate a pervasive exposure of fish at all sites, including those in GOA that are well away from the effects of EVOS. The results for Pacific halibut indicated exposure to aromatic hydrocarbons is unrelated to EVOS or HA sites in PWS. Further, analysis of the benthic sediments indicated that the probable sources of this exposure were petrogenic hydrocarbons derived from natural oil seeps and kerogen particles from eroding sedimentary rocks in the eastern GOA (Page et al. 2004). The fact that biomarker levels for PWS fish were not zero but similar to exposure levels in fish from eastern GOA indicated that regional hydrocarbon sources may be involved.

It also is important to note that many HA sites have been shown to have elevated levels of 4- to 6-ring pyrogenic PAHs, which are much more potent inducers of cytochrome p450 than 2- and 3-ring PAHs. This is particularly important because western PWS has more HA sites from prespill industrial activity than eastern PWS. Further, the prevailing east-to-west current flow carries material inward to PWS through Hinchinbrook Entrance in the east and past Knight Island as the water flows west and south through Montague Strait (Royer et al. 1990). Therefore, one would expect baseline levels of CYP1A induction to be greater in western PWS than in eastern PWS, including sites in the spill zone. Claims that induction of CYP1A in harlequin ducks and sea otters are attributable to EVOS (e.g., Trust et al. 2000; Bodkin et al. 2002; Esler, Bowman, et al. 2002) have not included consideration of these other sources of bioavailable PAHs that are stronger CYP1A inducers than the PAH associated with EVO (Neff 2002; Barron et al. 2004; Boehm et al. 2004).

ECOLOGICAL EFFECTS OF *EXXON VALDEZ* OIL SPILL

Selection of valued ecosystem components for Prince William Sound

We begin the process of assessing the ecological significance of effects from EVOS by identifying the VECs appropriate for assessing ecosystem health, beginning with the set of ecological attributes that the Exxon Valdez Trustees use to assign recovery status. The list of species and other PWS attributes that are monitored by the Trustees (2002) for determining recovery status include bald eagles, black oystercatchers, clams, common loons, common murres, cormorants, cutthroat trout, designated wilderness areas, Dolly Varden, harbor seals, harlequin ducks, intertidal communities, Kittlitz's murrelets, marbled murrelets, mussels, orca, Pacific herring, pigeon guillemots, pink salmon, river otters, rockfish, sea otters, sediments, sockeye salmon, and subtidal communities.

The National Wildlife Federation (NWF 2003) also developed a list of indicators of the state of PWS, which is similar to the Trustees' list. The NWF added "water quality," defined to include various pollutants identified in NWF (2003) as routine and accidental releases from cruise ships and other vessel traffic. NWF (2003) also added "persistent organic pollutants," such as DDT derivatives and PCBs, and marine debris and invasive species, especially from ship ballast waters. Thus, the Trustees' and NWF lists of indicators are dominated by species as recovery assessment attributes, and although 2 communities were included (intertidal and subtidal), no higher levels of ecological organization (e.g., ecosystem- or landscape-level attributes) or ecological processes were included. We believe that it is important to address higher levels of ecological organization explicitly if the set of selected VECs is to be used to assess the health of the PWS ecosystem, particularly since any change to ecological processes or landscape characteristics would constitute a fundamental change in the ecosystem.

Some species (e.g., Kittlitz's murrelets and rockfish) were included on the Trustees' list, even though there is very little information about the prespill populations, the extent of direct injury from EVOS is unknown, and recovery objectives have yet to be established. Since our purpose of selecting VECs is to describe the ecosystem sufficiently to assess recovery, selected endpoints must have sufficient information about pre-spill conditions confidently to understand recovery, or else recovery status can never be determined. In effect, whereas the Trustees (2002) describe the recovery of some attributes as "unknown," the proper characterization is "unknowable," as without any reasonable baseline, recovery cannot be determined; consequently, some of the attributes selected by Trustees (2002) and NWF (2003) are not appropriate as VECs.

Our objective is to develop a suite of VECs that is sufficiently comprehensive to assess ecosystem recovery of PWS following the criteria discussed previously. No single set of ecological attributes meets the criteria; for example, 2 species may be sufficiently redundant with respect to function or vulnerability to a particular stressor that either could be selected as a VEC. But in principle, the suite of VECs selected should include ecological attributes at both species and higher levels of organization, the species selected should cover various trophic levels, the attributes should be potentially vulnerable to the stressors at hand, the attributes should be at risk from the stressors, and the attributes must have sufficient information to assess pre- and postevent condition or have appropriate unoiled reference sites (i.e., could plausibly reflect on recovery status).

Many but not all of the Trustees' list of attributes meet these criteria, but we believe that collectively they do not adequately cover the suite of attributes to constitute a complete set of VECs. Consequently, we propose to use Trustees' attributes that have sufficient information and to supplement this list with additional VECs to address higher-level conditions. We could argue for the sake of parsimony to reduce the set from the Trustees' list further to eliminate redundancies (e.g., there are 7 species of birds, many with overlapping risk profiles), but we choose not to debate the issue where sufficient information is available for determining ecological significance.

The following attributes in the Trustees' list are considered inappropriate as VECs and are not further considered here:

1. Sediments—This is not an ecological effect attribute, although the status as source of continuing exposure was considered in the exposure section.
2. Common loon—This species was excluded for the following reasons:

   a. The lack of prespill data to evaluate recovery;
   b. The small population size in PWS;
   c. Limited exposure to the stressors from EVOS; and
   d. Several other VEC bird species have sufficient data for use as indicators of effects and recovery.
3. Cutthroat trout—This species was excluded for the following reasons:
   a. The lack of virtually any prespill data;
   b. It exists as a small, isolated population in PWS at the extreme northern limit of its range; and
   c. Several other fish species VECs have sufficient data for use as indicators of effects and recovery.
4. Dolly Varden—This species was excluded for the following reasons:
   a. The lack of virtually any prespill data and
   b. Several other fish species VECs have sufficient data for use as indicators of effects and recovery.
5. Kittlitz's murrelet—This species was excluded for the following reasons:
   a. The lack of virtually any prespill data;
   b. Its habitat is exclusively in glaciated fjords that had very little exposure to EVOS;
   c. It presently is at great risk to population reductions driven by climate change and the associated rapid loss of PWS glaciers that has been under way for several decades (see Kuletz et al. 2003); and
   d. Several other bird species VECs have sufficient data for use as indicators of effects and recovery.
6. Rockfish—This species was excluded for the following reasons:
   a. The lack of sufficient prespill data and
   b. Several other fish species VECs have sufficient data for use as indicators of effects and recovery.

The remaining ecological attributes from the Trustees' list are supplemented here by the following additional VECs: 1) trophic structure of PWS and GOA, 2) water quality and biogeochemical processes, and 3) landscape mosaic of habitats. Since these additional VECs were not selected by the Trustees as recovery endpoints, the quantitative information about them is much more limited than other endpoints; nevertheless, we believe sufficient information is available to reach a reasonable conclusion about the ecological significance of effects attributable to EVOS on these VECs.

The complete set of VECs used for assessing ecological significance is shown in Table 4. This suite of VECs includes primary producers, filter feeders, fish and bird primary consumers, fish and bird top predators, a bird scavenger, mammalian primary consumers and top predators, biotic communities, ecosystem-level properties of trophodynamics and biogeochemical processes, and landscape-level properties of habitat mosaic and wilderness quality. It includes economically and aesthetically important attributes with high human value as well as ecologically important properties. Thus, we assert that this set both meets the criteria for VECs and is sufficient to assess ecological significance of any continuing ecological effects attributable to EVOS.

Table 4. List of valued ecosystem components (VECs) used to assess the ecological significance of effects of the *Exxon Valdez* oil spill on the Prince William Sound, Alaska, USA, ecosystem

Birds	
	Bald eagle
	Black oystercatcher
	Common murre
	Cormorants
	Harlequin duck
	Marbled murrelet
	Pigeon guillemot
Fish	
	Pacific herring
	Pink salmon
	Sockeye salmon
Mammals	
	Harbor seal
	Orca
	River otter
	Sea otter
Invertebrates	
	Clams
	Mussels
Higher levels of organization	
	Intertidal communities
	Subtidal communities
	Trophic structure of Prince William Sound and Gulf of Alaska
	Water quality and biogeochemical processes
	Designated wilderness areas
	Landscape mosaic of habitats

Summary of present status as determined by the Trustees for their endpoints

The Exxon Valdez Oil Spill Trustees periodically report on the status of the list of ecological attributes of concern, classifying them into categories as 1) recovered, 2) recovering, 3) not recovered (also listed occasionally as not recovering), and 4) unknown. The most recent update on status is presented in Trustees (2002) and reported on the Trustees' Web site (reported here as the information available in 2005 at http://www.evostc.state.ak.us/facts/status.html). This report shows that no consistent set of criteria was used by the Trustees for determining recovery status; for example, some endpoints are considered "recovered" when the population returns to prespill conditions, whereas for other species, "recovered" means the absence of further exposure. The knowledge of the endpoint's pre-spill status and trends varies widely, and many populations were considered to have been in significant decline prior to the EVOS event for reasons such as food limitation, habitat alteration, overexploitation, and

Table 5. Ecological significance of effects on valued ecosystem components (VECs) in Prince William Sound (PWS) from the *Exxon Valdez* oil spill (EVOS)[a,b]

	Valued ecosystem component (VEC)	Trustee status (Trustees 2002)	Potential EVOS-caused stressors	Intensity of response to EVO[b]	Spatial extent of response to EVO	Reversibility and potential for recovery of the VEC	Redundancy[c]	Duration of impact on VEC	Natural variability	Ecological significance[a]
1	Bald eagle	Recovered	Physical oiling; possible toxic effect on embryos and via food consumption	H 250 deaths out of population of 6,000–8,000 at risk	H	H	None	Few years expected and realized absence of effects in 2 y	Less than initial mortality	High significance in 1989 Not significant in 2006
2	Black oyster-catcher	Recovered	Physical oiling; disturbance from cleanup; possible toxic effect on embryos	M 9 carcasses, 50–280 deaths, or 4–20% of population	H	H	L Only VEC that feeds on intertidal invertebrates	Few years; 20-y life span; absence of effects by 1991	Less than initial mortality	Medium significance in 1989 Not significant in 2006
3	Common murre	Recovered	Physical oiling; toxic exposures; food limitations; social disruption	H Several 100,000 deaths; 40–60% of population in oiled areas	H	H	M Other birds feed on same species	Few years	Much less than initial mortality	High significance in 1989 Not significant in 2006
4	Cormorants	Not recovering	Physical oiling; toxicity	M–H? 800 carcasses	H	H	M Other birds feed on same species	Few years	Poor information on populations or variability	Medium–high significance in 1989 Not significant in 2006
5	Harlequin duck	Not recovering	Physical oiling; toxicity; exposures to polycyclic aromatic hydrocarbons via intertidal diet	M–H <1,000 total, and 400 from population of 9,000–18,000 in PWS	H	H	M Other birds feed on same species	Few years	Limited prespill data; less than initial mortality	Medium–high significance in 1989 Not significant in 2006
6	Marbled murrelet	Recovering	Physical oiling; toxicity	M–H 900 carcasses recovered approximately 7% total population killed (8,400)	H	H	M Other birds feed on same species	Few years	Limited prespill data but decline ongoing for 30 y; probably re fisheries resource competition	Medium–high significance in 1989 Not significant in 2006

Table 5. Continued

	Valued ecosystem component (VEC)	Trustee status (Trustees 2002)	Potential EVOS-caused stressors	Intensity of response to EVO[b]	Spatial extent of response to EVO	Reversibility and potential for recovery of the VEC	Redundancy[c]	Duration of impact on VEC	Natural variability	Ecological significance[a]
7	Pigeon guillemot	Not recovering	Physical oiling; toxicity	H 10–15% population loss from direct mortality	H	H	M Other birds feed on same species	Few years	Population declining for decades, but initial mortality exceeded variability	High significance in 1989 Not significant in 2006
8	Pacific herring	Not recovering	Toxicity; physical oiling of eggs	M–H? Population effects unknown from EVOS but critical species ecologically and societally	H	H	L This species is critical to the coastal ecosystem	Years?	Extremely high; order-of-magnitude variability controlled by physical processes in Gulf of Alaska	Medium–high significance in 1989 Not significant in 2006
9	Pink salmon	Recovered	Toxic exposures; indirect effects on predators/prey	Could not be measured but modeled as 11% reduction in adult returns	H	H	M	Few years	Order-of-magnitude variability	Not significant in 1989 Not significant in 2006
10	Sockeye salmon	Recovered	Toxic exposures; indirect effects on predators/prey	Unknown but overwhelmed by closure of fishery, causing overescapement	H	H	M	Few years	Order-of-magnitude variability	High significance in 1989 from fishery closure Not significant in 2006
11	Harbor seal	Not recovering	Physical oiling; toxicity; disruption from cleanup activities	M–H 13% population decline in oiled area in 1989	H	M	None	Many years	Direct effects exceed variability; population decline since 1970s	Medium–high significance in 1989 Not significant in 2006
12	Orca (resident pods)	Recovering	Toxicity; physical oiling?; disturbance from cleanup	M for AB pod, none for rest of population	H	L–M	None	Many years; social structure effects from loss of matriarchs	Very low; good data exist for resident pods; AB pod shot prespill	For AB pod, significant in 1989 and 2006 For all other PWS pods, not significant in 1989 or 2006

Table 5. Continued

	Valued ecosystem component (VEC)	Trustee status (Trustees 2002)	Potential EVOS-caused stressors	Intensity of response to EVO[b]	Spatial extent of response to EVO	Reversibility and potential for recovery of the VEC	Redundancy[c]	Duration of impact on VEC	Natural variability	Ecological significance[a]
13	River otter	Recovered	Toxic exposure via food	L 12 carcasses found	H	H	L	Few years	Population small but numbers highly uncertain	Probably significant in 1989 Not significant in 2006
14	Sea otter	Recovering	Physical oiling; toxicity; social organization disruption?	H Uncertain but significant population effects initially; Pacific population recovering from near extinction from hunting	H	H	None	Many years	PWS population increasing through mid-1990s, then declining in Gulf of Alaska	High significance in 1989 Not significant in PWS in 2006 One local subpopulation may still be reduced, but other factors present and uncertain prespill numbers
15	Clams	Recovering	Physical oiling; toxic exposures; high impacts from cleanup	Extensive from oil and cleanup though no quantification	H	H	L	Few years	Community composition highly variable; high spatial heterogeneity	Medium significance in 1989, especially from cleanup Not significant in 2006
16	Mussels	Recovering	Toxicity; physical oiling; cleanup	No population effect noted in oiled beds	H	H	L	Few years	Population ubiquitous; no noted changes from spill	Not significant in 1989 Not significant in 2006
17	Intertidal communities	Recovering	Physical oiling; toxic exposures; high impacts from cleanup	Extensive from oil and cleanup	H	H	None	Few years	Community composition highly variable; high spatial heterogeneity	High significance in 1989, especially from cleanup Not significant in 2006
18	Subtidal communities	Unknown	Physical oiling; toxic exposures; high impacts from cleanup	Extensive from cleanup	H	H	None	Few years	Community composition highly variable; high spatial heterogeneity	High significance in 1989, especially from cleanup Not significant in 2006

Table 5. Continued

	Valued ecosystem component (VEC)	Trustee status (Trustees 2002)	Potential EVOS-caused stressors	Intensity of response to EVO[b]	Spatial extent of response to EVO	Reversibility and potential for recovery of the VEC	Redundancy[c]	Duration of impact on VEC	Natural variability	Ecological significance[a]
19	Trophic structure	Not on Trustees' list	Overescapement	L Limited to lakes	L	H	None	Few years	Major changes under way from overfishing, climate variability	Limited initial period Not significant in 2006
20	Water quality/ biogeo-chemical processes	Not on Trustees' list	Physical oiling; toxic exposures; altered escape-ment/nutrients	L–M? Limited direct, indirect through fish populations	L	H	None	Months to few years	High turnover in PWS water	Limited initial period Not significant in 2006
21	Designated wilderness areas	Recovering	Physical oiling; toxic exposures; high impacts from cleanup; noise; odor; perception of loss of wilder-ness	H Extensive from oil and cleanup	H	M–H	None	End of clean-up; few years other effects	NA[d]	High significance in 1989 Not significant in 2006
22	Landscape mosaic of habitats	Not on Trustees' list	Cleanup activities; physical oiling; toxic exposures	L Limited to cleaned areas	L	M–H	None	Months to few years	Community composition highly variable but mosaic stable	Limited initial period Net positive effects in 2006 from habitat protection

[a] Based on factors for determining ecological significance; see text for criteria.
[b] See text for sources of effects estimates. H = high; M = medium; L = low.
[c] Redundancy refers to the degree to which other VECs could offset losses.
[d] NA = not applicable.

climate variability. Similarly, the estimates of the intensity of the initial EVOS-caused injury vary widely across the endpoints, from large numbers of carcasses (e.g., more than 20,000 dead common murres) to no quantification of effects on clams. The characterization of current status by the Trustees often does not distinguish between EVOS-related effects and effects caused by other stressors. Finally, for some of the ecological attributes, there is disagreement in the literature concerning the initial post-spill effects and current effects.

Ecological significance for each selected VEC

Here we consider each attribute in our assessment set of VECs with respect to the ecological significance criteria of Gentile and Harwell (1998); a summary is presented in Table 5, which also lists the current Trustees' categorization of the status of each of their endpoints. This section gives a brief background on the VEC, summarizes information from the literature on the initial effects from EVOS, summarizes information from the literature on recent condition and trends including effects from non-EVOS stressors, and summarizes the conclusion we reach about the significance of effects in the initial period and at present (2006) using the ecological significance criteria discussed previously.

1) Bald eagle (*Haliaeetus leucocephalus*) is an opportunistic forager species that eats a diversity of prey but generally prefers fish. It often scavenges prey when available, pirates food from other species when it can, and captures its own prey only as a last resort (Buehler 2000). The bald eagle populations in coastal Alaska are dense and believed to be increasing or stable in most areas (Hansen and Hodges 1985; Bowman et al. 1993). The breeding range extends throughout much of Alaska, and the overwintering habitat includes all of coastal Alaska (Buehler 2000). Historically, the bald eagle was actively hunted in Alaska because they were thought to impact salmon fishing adversely, with a bounty offered by the state for the 1st half of the 20th century (over 128,000 bounties were paid in Alaska from 1917 to 1952; Robards and King 1966; Bowman 1999; Buehler 2000). Other global population impacts from anthropogenic sources include effects of long-lived pesticides, such as DDT (Grier 1982); exposure to lead, mercury, and other toxic metals; habitat alteration; active poisoning; and collisions with vehicles and power lines (Buehler 2000). The species was listed as endangered in the contiguous U.S. states in 1967 but proposed for delisting in 1999 (Buehler 2000). Although it has never been listed in Alaska, it is protected against any takings throughout all of the United States, including Alaska, under the Bald Eagle Protection Act of 1940 (now the Bald and Golden Eagle Protection Act [16 USC §668 a–d]) and the Migratory Bird Treaty Act of 1918 (16 USC §703–712; 50 CFR §10.13). The pre-spill estimates of the total population in PWS, Kenai Peninsula, and Kodiak archipelago that was potentially at risk from EVOS were 8,000 (Bowman et al. 1993) and 6,000 (Trustees (2002).

An estimate of direct mortality from EVOS in all of the oiled region was 900, based on finding 150 carcasses and an assumption that 17% of dead birds were observed as carcasses (Bowman et al. 1993). That number was questioned by Paine et al. (1996) because they asserted that natural mortality had not been adequately considered. The Trustees estimated total mortality in PWS at 250 (Trustees 2002), or 4% to 11% of the Sound's population. Bowman et al. (1993) reported that while no information was available on the annual variability in productivity for PWS, the observed decrease in productivity in 1989 was outside the range of variability observed in a 10-y study of bald eagles nesting in southeast Alaska. They estimated that there were more than 220 fewer young the next year and calculated that the total production lost was 28% of expected. Mortality was judged to be primarily from physical oiling of birds or consumption of oiled food. Reproduction reduction in PWS was considered likely caused by mortality of 1 or more parent, abandonment of the nest, or toxic effects on embryos from oil transferred from adult feathers (Bowman et al. 1993). Those authors also reported that nest failure rates in 1989 were significantly higher near heavily oiled beaches than near unoiled beaches.

By 1990, however, nesting rates and productivity had returned to normal range, and no further effects were reported. In fact, Murphy et al. (1997) reported a significant increase in the bald eagle population in 1990 and 1991 compared with pre-spill conditions (1984–1985). White et al. (1995) reported that in 1990 all measures of nest productivity, nest occupancy, and nesting success were similar between oiled and unoiled areas and that other differences noted across areas, territories, and years could be attributable not to oil but to natural annual variability. Wiens et al. (2004) analyzed oiling and habitat variables, concluding that evidence of an oiling effect had disappeared by 1990. Bowman et al. (1993) and Bernatowicz et al. (1996) stated that direct mortality and reduced reproduction of eagles were considered unlikely to have a lasting effect on bald eagle populations in the spill area. Trustees (2002) reported that this was the case. Bowman et al. (1995) and Bowman and Schempf (1997) reported that the bald eagle population in PWS had returned to its estimated pre-spill size before 1995, survival rates had returned to normal, and the Alaskan population approached its carrying capacity. The Trustees (2002) list this ecological attribute as recovered.

Ecological significance—Based on this information, we assign the bald eagle effects to the high ecological significance category for the initial effects of the spill, but no ecologically significant effects remain in 2006.

2) Black oystercatcher (*Haematopus bachmani*) is a long-lived species that is completely dependent on marine shorelines for feeding and nesting. Black oystercatchers nest on supratidal boulders and gravel beaches and forage in the intertidal zone on invertebrates (Andres 1994, 1998b; Andres and Falxa 1995). Monogamous breeding pairs establish well-defined feeding and nesting territories, generally occupying the same territory across years on rocky shorelines where intertidal prey are abundant. Year-round populations exist from Baja Mexico along the Pacific coast up through the Aleutians (Andres and Falxa 1995). The world population is estimated to be just 11,000 individuals (Andres and Falxa 1995), of which half are in Alaska, with the PWS pre-spill population estimated to be 500 to 900 (Andres 1994). It is protected against takings under the Migratory Bird Treaty Act of 1918 (16 USC §703–712; 50 CFR §10.13). The population in general has declined, most likely because of habitat impacts from human activities (Andres 1998b). Predation on early life stages is the greatest source of mortality, and oystercatcher productivity in PWS is inversely correlated with the abundance of the common raven; additionally, the introduction of red and arctic foxes to the islands along the Alaskan coast caused local extirpation of breeding black oystercatchers (Andres and Falxa 1995). Diets of adults and chicks consist of benthic and epibenthic mollusks, mainly mussels (*Mytilus*) and limpets (*Tectura* and *Lottia*), and most feeding is done at low tide (Andres and Falxa 1995). Thus, this species was particularly vulnerable to EVOS effects (Murphy and Mabee 2000). However, because the life span is about 20 y (i.e., there is a relatively low annual population turnover rate), only multiple years of reproductive failure or high initial mortality would have significant long-term effects on the population.

Andres (1994) reported that whereas estimates of direct mortality of black oystercatchers from EVOS varied widely (from 3.6% of the total population in the spill zone to 10–57% of the population along the shoreline), mortality probably did not exceed 20% of the population in the EVOS zone. Andres (1994) also estimated that direct mortality of about 50 to 280 individuals occurred, or 4% to 20% of the total breeding population in the spill area (reported to be 1,388 individuals); however, Andres (1994) and Murphy and Mabee (2000) reported that only 9 carcasses were actually recovered. The low mortality likely occurred because few birds were occupying their territories at the time of the spill; less than 25% of the population remains in PWS over the winter (Murphy and Mabee 2000). The breeding activity of many pairs, however, was disrupted along oiled shorelines in PWS, and hatching success was significantly reduced in 1989 on Green Island. On the other hand, Andres (1994) reported that nesting sites on Green Island were recolonized rapidly in the early years after the spill and after cleanup activities ceased. Black oystercatcher exposure was most likely associated with direct oiling and consumption of contaminated food (primarily mussels). Sublethal toxic effects on chicks were noted (e.g., chick weight gain was slower near oiled sites), but it was also determined that the cleanup activities had significant effects on breeding success. Sharp et al. (1996) assessed effects on black oystercatcher reproduction and found decreased nesting effort, delayed reproduction, higher chick mortality, and lower chick growth rates in 1989.

Andres (1994) expected that no population effects from EVOS would be evident after a few years (e.g., he estimated 5 y for an 18% population loss to recover) and that recovery was expected to be particularly driven by the rate at which mussels were no longer to be contaminated. He also noted that reoccupation by oystercatchers into habitats disturbed by EVOS varied spatially across PWS and may be related to habitat quality independent of oiling severity, suggesting that the high quality habitats were reoccupied first.

Andres (1998b) reported no differences between oiled and unoiled sites by 1991, and Andres (1998a, 1999) reported that hatching success, fledging success, nesting success, and productivity were higher for pairs nesting in persistently oiled areas. Wiens et al. (2004) concluded that oystercatcher habitat occupancy had recovered by 1991. The Trustees (2002) supported a reassessment study in 1998 that reported that oystercatchers had fully reoccupied and were nesting at previously oiled sites in PWS (see also Murphy and Mabee 1999; Stephensen et al. 2001). The breeding phenology of nesting birds was found to be synchronous in oiled and unoiled areas, and no oil-related differences in clutch size, egg volume, or chick growth rates were detected. A high rate of nest failures on Green Island was attributed to predation rather than to lingering effects of EVOS. Murphy and Mabee (2000) concluded that EVOS oiling did not adversely affect breeding biology of the oystercatchers in 1998. The Trustees (2002) considered black oystercatchers recovered.

Ecological significance—Based on review of the available information, we assign the effects on the black oystercatcher to the medium ecological significance category for the initial spill, but no ecologically significant effects remain in 2006.

3) Common murre (*Uria aalge*), or thin-billed murre, is one of the most numerous marine birds in North America, with an estimated 4 million to 8 million murres in the west (Ainley et al. 2002). It is protected against takings under the Migratory Bird Treaty Act of 1918 (16 USC §703–712; 50 CFR §10.13). It is highly social, breeding in extremely high densities on island cliff ledges. It dives to depths of 100 m, feeding primarily on fish but also on pelagic invertebrates such as euphausiids (krill) and cephalopods (Ainley et al. 2002). Murres are particularly susceptible to injury from oil spills because they spend most of their time on the water, they are often concentrated in dense flocks (Nysewander et al. 1993), they occur near shipping channels, and they have low reproductive rates (Ainley et al. 2002). Oil spill–induced mortality is caused mainly by hypothermia and malnutrition as oiled feathers lose their insulating properties.

The EVO surrounded the colonies in April and early May 1989, the period in which murres were congregating prior to breeding. Over 30,000 dead birds were retrieved in GOA following EVOS, about 75% of which were murres (Piatt et al. 1990). These authors estimated the total direct mortality of all seabirds from EVOS to be 100,000 to 300,000, and Piatt and Ford (1996) estimated 250,000, whereas Ford et al. (1991, 1996) estimated 375,000 to 645,000. On the other hand, Wiens (1996) noted that the pre-spill data on murres were sparse and highly variable, so that translating direct mortality estimates to significant population reductions is difficult. In any case, these numbers were considered to be unprecedented for an oil spill (Piatt and Lensink 1989). Population reductions were associated with physical oiling, toxicity, food limitations, behavioral modifications, preferential mortality for experienced breeding pairs, and social disruptions (Nysewander et al. 1993). Nysewander et al. (1993) reported that common murre and thick-billed murre (*Uria lomvia*) colonies they surveyed contained the majority of the estimated 185,000 to 200,000 murres in the EVOS-affected area, with 80% of these as common murres (see also Piatt and Ford 1996). Nysewander et al. (1993) found reduced numbers at all study colonies following the spill, with population decreases estimated at 40% to 60% of pre-spill numbers. In addition, they stated that nesting was delayed and productivity rates were below normal following the spill, with almost total reproductive failure at many monitored sites in 1989 and effects on reproductive rates evident for 3 y. In contrast, numbers of murres did not decline and reproductive parameters were normal at 2 colonies surveyed outside the spill trajectory.

Dragoo et al. (1995) reported that productivity began to recover in 1992. Erikson (1995), Wiens (1995), Day et al. (1995), and Boersma et al. (1995) all concluded that any effects noted on common murres were relatively short term based on surveys in 1991, contrary to early predictions that recovery would take decades. Piatt and Anderson (1996) stated that longer-term effects (up to 1994) included depressed populations, reduced breeding success, and delayed breeding phenology, although Stephensen et al. (2001) reported greatly increased numbers in 1993. Piatt and Anderson (1996) noted there are other significant factors that could explain these effects, such as altered coastal circulation, variability in climate and sea-surface temperatures, altered food resources, and diet shifts, among other factors. They concluded that longer-term effects from these factors could not be distinguished from effects caused by EVOS. Trustees (2002) reported that monitoring at the breeding colonies in the Barren Islands indicated that reproductive success was within normal bounds by 1993 and afterward. Counts in 1997 and 1999 had increased to prespill numbers; Trustees (2002) considered this endpoint to be fully recovered.

Ecological significance—Based on review of the available information, we assign the effects on the common murre to the high ecological significance category for the initial spill, but no ecologically significant effects remain in 2006.

4) Cormorants species listed by the Trustees include the double-crested cormorant (*Phalacrocorax auritis*), the pelagic cormorant (*P. pelagicus*), and the red-faced cormorant (*P. urile*) (Trustees 2002). All 3 species are protected against takings under the Migratory Bird Treaty Act of 1918 (16 USC §703–712; 50 CFR §10.13). The double-crested cormorant feeds almost entirely on fish in shallow open waters over sandy bottoms, among rocks, or in seagrass or kelp beds, usually less than 8 m deep and within 5 km of the shoreline (Hatch and Weseloh 1999). The double-crested cormorant was listed as a species of concern in 1970, but recently there have been large increases in the population numbers, and the bird is now often considered to be a pest (Hatch and Weseloh 1999), with a 1990 global population estimated at 1 million to 2 million (Hatch and Weseloh 1999). The pelagic cormorant is the smallest, most widely distributed, and among the least gregarious of the North Pacific cormorants; it nests in loose colonies on steep cliffs along rocky and exposed shorelines and in spite of its name is rarely observed more than a few kilometers from land (Hobson 1997). It feeds mostly in swirling waters of riptides and storm tides on nonschooling fish in southern Alaska, particularly the Pacific sand lance (*Ammodytes hexapterus*), and on benthic invertebrates (Hobson 1997). The pelagic cormorant usually forages individually by diving from the water's surface and swimming in pursuit of its prey. The North American population totals about 130,000 birds, most of which are in Alaska (Hobson 1997). The red-faced cormorant is exclusively marine, using dry land only to breed or roost and never more than a few meters from the edge of the sea (Causey 2002). Similar to the pelagic cormorant, it is the least gregarious of all cormorants, nesting on steep, inaccessible cliff faces, and is shy of humans; because of this, it is 1 of the least known birds of the North Pacific (Causey 2002). It is occasionally observed well out at sea but more commonly is seen in inshore and coastal waters of islands. The red-faced cormorant prefers solitary fish and invertebrates near the bottom and feeds by underwater pursuit of its prey. Its range extends from the Aleutian Islands to a few sites in the GOA, with a worldwide breeding population roughly estimated to be about 155,000 individuals (Causey 2002).

Since all 3 PWS cormorant species are diving birds that feed on fish and spend much of their time on the water, they have high risk for exposure to oil spills. The overall population sizes for all these cormorants in the area are relatively small, with about 16,000 counted in colonies in the spill-affected area in 1996 (Trustees 2002). More than 800 carcasses of cormorants were found after EVOS, but it is unclear what the total injury was to the population. Day et al. (1997a) noted negative habitat impacts along the Kenai Peninsula in 1991 for the double-crested cormorant. They also reported that the pelagic cormorants dominate the cormorant populations in this area and that the large colony on nearby Middleton Island has been in decline since the mid-1970s. There were statistically significant declines in colonies in oiled areas between counts made in 1984 and after the spill, but the absence of sufficient population data led the Trustees (2002) to conclude that it is unknown what the populations would have been if EVOS had not occurred. Thus, while that report lists the cormorant species as "not recovering," the basis for that conclusion is unclear since there is such a poor baseline for comparison of population numbers prior to the spill.

By contrast, Stephensen et al. (2001) reported a stable population from 1990 to 2000. Day et al. (1997b) showed no significant effect of oiling on cormorant habitat use. Wiens et al. (2004) concluded EVOS had a direct effect on the pelagic cormorants, resulting in complete absence from the area in the summer of 1989, but that occupancy had recovered by 1991.

The double-crested and pelagic cormorants are widely perceived by commercial and recreational fishers as major competitors and thus have been subject to extensive destruction of nests, eggs, and young, and shooting of adults. These conflicts have probably led to the widespread decline of these species in the 20th century. While a definitive conclusion is uncertain because of insufficient pre-spill data, we believe that the effect of oiling from EVOS was transient, no continuing EVOS-caused stressor presently exists that could affect the cormorant populations, and other stressors not related to EVOS continue to exist and affect these birds.

Ecological significance—The initial effects of oiling on the cormorant species of PWS may have been ecologically significant, but the present ecological condition does not warrant a conclusion of any remaining ecologically significant effects attributable to EVOS.

5) Harlequin duck (*Histrionicus histrionicus*) breeds on fast-flowing rivers and streams and forages in intertidal and shallow subtidal habitats to a depth of 10 to 20 m. Their diet consists of a large diversity of benthic organisms, including crabs, amphipods, snails (*Littorina* and *Lacuna*), limpets (*Lottia*), and lesser amounts of 2 dozen other taxa (Robertson and Goudie 1999). They winter all along the coasts of British Columbia and southern Alaska and west to the Aleutian Islands. They have relatively high adult survival rates (75–85% in southeast Alaska) (Robertson and Goudie 1999), long reproductive life spans, and low annual productivity. They are protected against takings under the Migratory Bird Treaty Act of 1918 (16 USC §703–712; 50 CFR §10.13).

Because of their small body size, harlequin ducks may exist near an energetic threshold in winter, with little flexibility for increasing caloric intact or ability to rely on stored reserves (Esler, Bowman, et al. 2002), making them particularly susceptible to adverse effects of perturbations to their wintering environments. Crowley and Patten (1996) stated that the apparent low productivity of the harlequin duck population in PWS, even though there are rich and diverse food resources, indicates that other factors, such as habitat availability, predation, and climate, may be limiting; this is consistent with the fact that PWS is one of the northernmost wintering areas for the species (Robertson and Goudie 1999).

Patten et al. (2000) reported a study of 6 species of sea ducks, focusing after the 1st y only on harlequin duck. The population is present year-round in PWS, with the estimates of the breeding population of 2,600 to 3,300 from early May and late summer in 1972 and 5,500 during the summers of 1984–1985 (Patten et al. 2000). Klosiewski and Laing (1994) examined boat survey data, showing pre-spill estimates of 6,100 wintering harlequin ducks in March 1972 and 5,700 in March 1973 for portions of PWS that were oiled by EVOS. Esler, Bowman, et al. (2002) reported an overwintering population of 14,000 in PWS, and Wiens et al. (2006) reported the range of overwintering estimates of 12,000 to 18,000 and summer range of estimates of 5,000 to 10,000.

Piatt et al. (1990) estimated direct mortality of harlequin ducks from EVOS was over 400 in PWS and about 1,000 in the total spill-affected area, or an estimated 3% to 6% of the harlequin duck population present in PWS (Wiens et al. 2005). During March and April 1989 (i.e., during the period of maximal exposure to EVO), harlequin duck pairs would have populated offshore rocks and rocky points. Prior to the breeding seasons of March 1990 and 1991, harlequin ducks in oiled segments totaled about 2,800, indicating levels about half those of the early 1970s. Patten et al. (2000) noted that harlequin ducks have high site fidelity, contributing to the reduced breeding activity in years subsequent to EVOS and expectations of slower population recovery rates. It was also expected that harlequin ducks were significantly affected by the postspill cleanup activities (Patten et al. 2000; Wiens et al. 2006).

Patten et al. (2000) reported significant differences in population dynamics of oiled and unoiled sites but noted that other habitat differences between western and eastern PWS could have a significant effect as well. Wiens et al. (2004, 2006) suggested that differences in numbers in oiled and unoiled sites may be related to differing habitats as well as surveying methods; for example, they noted that some of the transects in the western PWS sites (oiled areas) included known pre-spill low-density areas, whereas the unoiled eastern PWS sites were all known as high-density areas. Patten et al. (2000) went on to state that recovery was expected to be slow in part because of small breeding population, interference with site fidelity, deferred breeding, and delayed maturity. (Note that whereas this article was published in 2000, it refers to data collected the 1st few years after the spill.) Patten et al. (2000) and Goatcher et al. (1999) predicted that the small number of breeding harlequin ducks that remain in the oil spill area and low recruitment rates of sea ducks in general would make recovery a long process, even in the absence of residual EVO.

Esler, Bowman, et al. (2002) concluded through comparisons of oiled versus unoiled areas that as of 1998, the harlequin duck population had not yet recovered, attributing effects to continuing oil exposure from EVOS. They reported elevated ethoxyresorufin-O-deethylase activity (an indicator of CYP1A induction, a biomarker of PAH exposure) in 1998 as the basis for concluding that harlequin ducks were still exposed to EVOS-derived toxic chemicals. However, they did not acknowledge that many other sources of PAHs exist in PWS, as we have documented previously; as a result, their conclusion is based on the presumption that any PAH exposure necessarily was attributable solely to EVOS. Other convincing evidence of EVOS-caused exposure was not presented, and there are many other factors that could adversely affect harlequin duck populations. Boehm et al. (2004) showed that even 1 y after the spill, risks to harlequin ducks from consumption of contaminated mussels was very low. Wiens et al. (2004, 2006) showed that the numbers of harlequin ducks were consistently much lower in the oiled than in the unoiled region of PWS both before and after the oil spill, suggesting that the 2 areas differ fundamentally in their environmental suitability or attractiveness to harlequin ducks, and thus those comparisons are of very limited value in assessing potential continuing effects. A habitat use study by Day et al. (1997b) indicated no effects from EVOS by 1991; they reported that harlequin ducks avoided heavily oiled areas immediately after the spill but not in subsequent years.

Data from Stephensen et al. (2001) show population numbers by 1993 equaled or exceeded the pre-spill counts for PWS. Rosenberg and Petrula (1998) concluded no significant differences between eastern and western PWS populations of harlequin ducks, with similar breeding propensity, recruitment, breeding success, and survival rates. They went on to conclude that breeding activity is limited by suitable breeding habitat in PWS. Finally, even if harlequin ducks continue to experience exposure to PAHs from remnant EVOS-contaminated mussel beds (as Patten et al. [2000] and Esler, Bowman, et al. [2002] suggested), the areal extent of these remnant mussel beds is so small compared to the other sources of mussels available to the harlequin ducks and the proportion of the harlequin duck diet attributable to mussels is so small that no plausible risk exists to the population (see also Wiens et al. 2006). Currently about 14,250 to 18,000 harlequins overwinter in PWS (Robertson and Goudie 1999; Trustees 2002) and 9,000 in summer (see also Esler, Bowman, et al. 2002; Esler, Schmutz, et al. 2002).

Wiens et al. (2006), in an exhaustive survey and reanalysis of available pre- and postspill data, concluded that EVOS had a limited immediate effect on harlequin duck distribution and abundance in PWS, followed by apparent recovery within a few years. They also concluded that 1) harlequin ducks were more abundant in the unoiled than in the oiled areas of PWS, even before the spill occurred; 2) some studies comparing pre- and postspill numbers found greater decreases in oiled areas, whereas other studies found greater decreases in unoiled areas; and 3) abundances increased during the mid- to late 1990s to levels equal to or greater than the prespill estimates in both oiled and unoiled areas. Trustees (2002) stated that conclusions of reproductive failure based on lack of broods in the oiled area no longer seem warranted. They classify this species as "recovering" (although the Trustees' Web site in 2005 continued to list this as "not recovering" [see http://www.oilspill.state.ak.us/facts/status.html]) but not yet having attained their recovery objectives.

Ecological significance—It is unclear how extensive the direct effects of EVOS were on the harlequin duck population. If, in fact, 1,000 birds died directly after the EVOS, then that would constitute a sufficient portion of the total population to be ecologically significant. Recent data indicate the populations have returned to prespill levels or higher and are well within the range of natural variability. The discussions in Patten et al. (2000) and Trustees (2002) concerning physiological responses of harlequin ducks do not indicate ecologically significant effects but rather suborganismal responses to exposures to PAHs that may derive from many sources in PWS, not just EVOS. Moreover, the present low levels of PAHs in mussels attributable to EVOS suggest (cf. Boehm et al. 2004; Neff et al. 2006) that the current exposure to harlequin ducks is not ecologically significant. Reported differences in oiled versus unoiled area populations apparently relate to factors other than residual EVOS, such as differences in habitats and differences in sampling methodologies between the oiled and unoiled sites. We conclude that there currently are no detectable ecologically significant effects from EVOS on harlequin ducks.

6) Marbled murrelet (*Brachyramphus marmoratus*) is a small seabird that nests in trees in coastal forests throughout most of its range in North America, from central California through the Aleutian Islands (Nelson 1997). In the breeding season, they feed primarily on small schooling fish, including Pacific herring, Pacific sand lance, and smelt, but prefer euphausiids, mysids, and amphipods in winter and spring (Nelson 1997). Marbled murrelets fly at high speeds, attend their breeding sites during periods of low light, and nest solitarily, making this species quite difficult to study. Consequently, information is very limited on its behavior, nesting habits, habitats, and population (Nelson 1997). The distribution and numbers of marbled murrelets have declined significantly since the 1970s, including in GOA and PWS, possibly because of decreased food resources, habitat loss through logging and coastal development, impacts from gill-net fishing, and exposures to oil spills (Wiens 1996; Nelson 1997; Trustees 2002). Rates of decline are estimated at 4% to 7% per y throughout its range (Nelson 1997). Being rare or uncommon in areas where it was common in the early 20th century, it was federally listed as a threatened species in California and the Pacific Northwest in 1992 and in British Columbia in 1990 (Nelson 1997). The species is protected against takings under the Migratory Bird Treaty Act of 1918 (16 USC §703–712; 50 CFR §10.13). Nevertheless, the marbled murrelet is still the most abundant seabird in PWS and breeds throughout the area affected by EVOS (Piatt and Ford 1993; Kuletz 1994).

More than 1,000 murrelet carcasses were found after the spill, 90% of which were marbled murrelets. Kuletz (1996) and Trustees (2002) stated that up to 7% of the total spill-area population of marbled murrelets suffered direct mortality from EVOS, totaling 8,400 directly killed (Kuletz 1994). Kuletz (1994) compared pre- and post-spill abundance and breeding activity of murrelets in central PWS and in Kachemak Bay, an unoiled bay in lower Cook Inlet. Murrelet numbers at Naked Island were lower in 1989 than in 1978–1980 but not in 1990–1992. At Kachemak Bay, murrelet densities did not change between 1988 and 1989. Fewer juvenile murrelets were observed in oiled sites, but not in unoiled sites, in post-spill years, but data were insufficient to determine if reproduction was disrupted. Murrelet numbers were negatively correlated with numbers of boats at both study sites, so cleanup activities likely contributed to the population disruption in 1989. Trustees (2002) reported that boat-based surveys were conducted in March and in July for 7 y during the period 1989–2000, and the counts of marbled murrelets in March in most years increased in oiled and unoiled areas, whereas the counts in July declined in both oiled and unoiled areas in these surveys.

Other studies indicate that some factor other than EVOS is responsible for the longer-term population responses. Kuletz (1996) stated that the marbled murrelet population in PWS has declined by 66% since 1973, and Nelson (1997) reported declines of 50% to 75% in the northern GOA in the past 20 y, so clearly this is not as a result of EVOS.

The Trustees (2002) report stated that the marbled murrelet productivity throughout PWS appears to be within normal bounds, supported by the data in Stephensen et al. (2001). Moreover, they reported that studies relating the marbled murrelet juvenile productivity to abundance of forage fish, such as Pacific herring, showed a direct correlation. Thus, although Trustees (2002) continued to list this species as "recovering" and "not fully recovered," it is unclear that the present status of the marbled murrelet population reflects any detectable residual effects from EVOS, given natural variability and the longer-scale population decreases for several decades.

Ecological significance—At an estimated 7% population loss from direct mortality, this threatened species did experience ecologically significant effects from EVOS; however, the absence of any population-level differences from oiled versus unoiled sites and the return to normal productivity throughout the PWS area are sufficient reasons to conclude that the marbled murrelet population at present has no detectable ecologically significant effects from EVOS.

7) Pigeon guillemot (*Cepphus columba*) nests in burrows or rock cavities in small colonies or scattered pairs along rocky coastlines, often on small islands that provide protection from predators (Ewins 1993). They forage by swimming underwater to feed on fish or by diving directly to the sea bed where they probe rocky recesses and vegetation, feeding on demersal or epibenthic prey primarily in nearshore waters shallower than 40 m (Ewins 1993; Sanger and Cody 1994; Kuletz 1998). Pigeon guillemot breeding numbers seem to be limited by the availability of food and of suitable nest sites. The estimated population is about 235,000 worldwide (Ewins 1993), with the pre-spill population in PWS estimated at 3,500 birds (Agler et al. 1994). The species is protected against takings under the Migratory Bird Treaty Act of 1918 (16 USC §703–712; 50 CFR §10.13). Like many other alcids, pigeon guillemots probably abandoned former breeding islands when predators were introduced, especially arctic fox and red fox (Ewins 1993). The Alaskan populations are thought to be stable but possibly depleted from losses caused by these introduced species. There is an absence of reliable population counts. However, widespread distribution along most coastlines reduces the vulnerability of pigeon guillemots at the population level to perturbations such as oil pollution, gill-netting, and mammalian predators (Ewins 1993).

It was estimated that 10% to 15% of the PWS population of pigeon guillemots suffered direct mortality from EVOS (Trustees 2002). Oakley and Kuletz (1994, 1996) studied a population breeding on Naked Island, Alaska, about 30 km from the *Exxon Valdez* grounding site. The post-spill population was about 40% less than the pre-spill population, but the authors could not attribute the entire decline to the spill because a decline in the PWS pigeon guillemot population predated the spill, perhaps associated with reduced food resources (Oakley and Kuletz 1994, 1996; Wiens 1996; Trustees 2002). Murphy et al. (1999) reported significant effects from oiling based on relative changes in densities at oiled versus unoiled sites and showed significant population effects through 1991. Day et al. (1997a, 1997b) showed a significant effect of EVOS on habitat use in 1989 but that the effect of habitat use was not significant by 1990. Hayes (1995) reported that the population of pigeon guillemots in PWS had decreased from about 15,000 (1970s) to about 5,000 and that while some local populations were affected by EVOS, there is evidence that the PWS-wide population was already declining prior to the spill. Wiens et al. (2004) concluded that pigeon guillemots demonstrated habitat use effects from EVOS, which had disappeared by 1991, overlaid onto a regional decline in overall abundance. They also noted that oiled areas had lower abundances in 1996 and 2001, which could reflect a delayed but inconsistent response, but could not identify an underlying ecological mechanism. Sanger and Cody (1994) noted that pigeon guillemot populations declined at both oiled and unoiled Naked Island sites, speculating reduced food availability and increased predation as causes.

Top-down trophic control may also be important. Warheit et al. (1997) suggested that the most efficacious restoration technique for pigeon guillemot is predator control, and populations have rebounded rapidly following removal of introduced fox populations (Byrd et al. 1994). Other predators include northwest crow, raven, magpie, Steller's jay, grey jay, mink, river otter, bald eagle, and peregrine falcon. Oakley and Kuletz (1996) reported that predation on pigeon guillemots on Naked Island has increased since the late 1970s and early 1980s and that more than 25% of nests there were depredated in 1994 (Hayes 1995). However, Oakley and Kuletz (1994, 1996) reported that relative declines in the population after the spill were greater along oiled shorelines, suggesting that some initial population decline was partly attributable to EVOS. Reproduction appeared largely unaffected, but the cryptic nature of pigeon guillemot nests limits the detectability of failed nests. Nesting success was reduced, but the apparent cause was greater losses of chicks to predators and thus was not related to the spill. Oakley and Kuletz (1994, 1996) stated that the most likely explanation for the few effects observed is that oil was present on the surface waters of the study area for a short period before the guillemots returned to begin their annual reproductive activities. Nevertheless, Trustees (2002) stated that because both oiled and unoiled areas in PWS had reduced numbers through 2000, they assigned pigeon guillemot as "not recovering."

Ecological significance—The initial effects on the pigeon guillemot may have been ecologically significant, but the present ecological condition does not indicate any remaining detectable ecologically significant effects attributable to EVOS. To the contrary, it appears that continuing population declines of this species may relate to other stressors, as seen in trends that have been under way for several decades.

8) Pacific herring (*Clupea pallasis*) is among the most important species ecologically and economically in PWS (Brown and Carls 1998), constituting a major part of the food base for marine mammals (e.g., humpback whales and harbor seals), many seabirds (e.g., murres), other fishes (e.g., haddock), and other species of concern (e.g., bald eagle; Brown and Carls 1998). This results from their high fat content, large abundance, and visibility and proximity to the surface. Commercially, the Pacific herring has historically been an important source of fish eggs and thus subject to intensive fishing pressure. In late March and early April, adult Pacific herring migrate to the spawning beaches in PWS, and spawning continues from mid-April to early June (Norcross et al. 2001). The species is abundant and spawns throughout the coastal Alaska region, including PWS (Norcross et al. 2001). The population attained a then-record size in 1989 in PWS and continued to increase annually thereafter through 1992. A major population crash occurred in 1993 (Pearson et al. 1999), resulting in the loss of 75% of the biomass, but the population has gradually increased since then (Brown and Carls 1998).

The EVOS event occurred a few weeks before the Pacific herring spawned in PWS. Brown et al. (1996) claimed that half the egg biomass in PWS was exposed to the oil spill trajectory. By contrast, Pearson et al. (1995) reported that most spawning in the spill path area was on the west coast of Naked Island and north shore of Montague Island, both sites of light and patchy oiling. Consequently, Pearson et al. (1995) estimated that about 96% of the total spawning area in PWS occurred along shorelines with no oiling from EVOS and that less than 1% of the spawning area occurred along shorelines with moderate to heavy oiling. In either case, clearly there were widespread sources of eggs and larvae well outside the spill zone and outside PWS (Norcross et al. 2001). Following the EVOS event, the Pacific herring fishery was closed to prevent contaminated food going to market, but the 1989 year class was 1 of the smallest returning to spawn. Field studies showed sublethal effects on newly hatched larvae, including premature hatch, low weights, reduced growth rates, and morphologic and genetic abnormalities in oiled versus unoiled areas (Brown et al. 1996), but these did not result in demonstrated population effects (Trustees 2002). Carls et al. (1999) argued that Pacific herring eggs in PWS were exposed to PAHs in concentrations sufficient to cause egg mortality or sublethal effects in the first 2 weeks following the spill. However, Pearson et al. (1995) noted that effects on Pacific herring eggs were minor even in oiled areas in 1989 and concluded that there was no significant injury to the Pacific herring population from EVOS.

Moreover, the lack of population-level effects could be attributed in part to the large natural interannual variability in larval production of Pacific herring. Even if all exposed areas had 99.9% loss of larval production caused by EVOS (as suggested by Brown et al. 1996), the unexposed areas still produced half the expected total PWS larval production in 1989, well within the range of normal variability. The species is subject to viral hemorrhagic septicemia disease, among other diseases (Marty et al. 1998; Kocan et al. 1999), which may be triggered by high densities and which apparently caused the collapse of the population in 1993. Evidence for this includes the observation of apparent viral hemorrhagic septicemia incidence in Pacific herring contained in the PWS spawn-on-kelp fishery in 1992 (Kocan et al. 1999). The authors also stated that no evidence could be found that exposure to oil increases the susceptibility of either laboratory-reared or wild Pacific herring to viral hemorrhagic septicemia. Thus, it is considered unlikely that EVOS led to the population crash (Brown and Carls 1998). No evidence exists of long-term pollution effects on Pacific herring caused by EVOS; for example, Carls et al. (1997) showed no evidence of oil-related reproductive impairment 6 y after the spill.

Norcross et al. (2001) reported on physical conditions that may control the population dynamics of adult Pacific herring, suggesting that the availability of food is the key to successful survival of juvenile herring and that food availability in their study year of 1995 was from GOA. They also noted that interannual variability in local climate and oceanographic conditions appear to control interannual variability in nursery success of Pacific herring. Consequently, these authors concluded that conditions in GOA directly affect the Pacific herring population within PWS. Marty et al. (2004) reported that because of disease-related declines in the population biomass, the Pacific herring fishery has been closed in PWS since 1999. The Trustees (2002) listed this species as "recovering," even though the population crash 4 y after EVOS and population dynamics since then are not likely attributable to EVOS; however, the Trustee Council downgraded the status to "not recovering," as reported in the *Anchorage Daily News* (8 August 2002), and the Trustees' Web site (http://www.oilspill.state.ak.us/facts/status.html) in 2005 continued to list this as "not recovering."

Ecological significance—It is unclear what population loss resulted from direct mortality attributable to EVOS, but the critical role in the ecosystem and for society of Pacific herring makes any effect rise to the level of ecological significance. However, the major collapse of the population several years after the spill was likely caused by factors other than EVOS, suggesting that there are no remaining ecologically significant effects that can be attributed to the oil spill. The continuing effect of disease on the PWS Pacific herring population and the linkage of the population to the much larger-scale physical properties of GOA suggests that the population dynamics and interannual variability of Pacific herring are dominated by natural processes, which are not caused by EVOS.

9) Pink salmon (*Oncorhynchus gorbuscha*) is one of the most important economic species in PWS. This species has a 2-y life cycle, with no interbreeding between odd- and even-year populations. Spawning occurs largely from July to mid-September. According to Trustees (2002), prior to the spill the wild pink salmon population numbers returning to spawn ranged from about 2 million (in 1988) to more than 23 million (in 1984) representing an order-of-magnitude interannual variability in the decade prior to the oil spill. Studies by Wertheimer et al. (1994) concluded that oil contamination from EVOS reduced the growth of juvenile pink salmon in western PWS in 1989. They speculated that populations were also reduced that year, but by 1990 they found that the size and growth of juvenile pink salmon were similar in oiled and nonoiled locations. Willette (1996) reported that EVOS-induced growth rate reduction likely caused about a 2% reduction in pink salmon survival to the adult stage among fish reared in oiled areas, an amount undetectable in the high natural variability.

Hatchery-reared pink salmon have increased since EVOS, with 84% of salmon caught in 1999 harvests consisting of hatchery-raised fish (Sharr et al. 1995; NWF 2002). Cooney et al. (2001) and Willette et al. (2001) estimated that about 75% of all pink salmon fry reared in PWS suffer predation mortality in their first 45 to 60 d. Cooney et al. (2001) estimated wild and hatchery population of fry of 650 million; thus, there typically would have been losses of 450 million juveniles in the PWS nearshore environment. Willette et al. (2001) reported losses of 546 million among a population of 726 million fry.

In the years after EVOS, returns of wild pink salmon ranged from about 2 million (1992) to almost 13 million (1990), leading Trustees (2002) to conclude that because of the tremendous natural variation in adult returns, the extent to which wild pink salmon returns were affected by the oil spill could not be measured directly. Cooney et al. (2001) and Willette et al. (2001) hypothesized that the high interannual variability in pink salmon is driven largely by trophodynamic interactions involving the macroplankton populations of PWS, which are controlled by hydrodynamics and upwelling processes. This is consistent with the findings of Morita and Fukuwaka (2006) that climate variability largely drives pink salmon population dynamics. Maki et al. (1995) and Brannon and Maki (1996) noted the high levels of pink salmon returns in PWS in 1990 and concluded that there were no population-level effects from EVOS. Brannon et al. (1995) examined water and sediment concentrations for PAHs in 1989 and through regression concluded that there would have been no substantial toxicological effects on the critical early life stages of pink salmon in PWS attributable to the spill. This was confirmed by chemical analysis of pink salmon embryo tissue sampled from oiled streams. In 1990 and 1991, embryos showed mean tissue-PAH concentrations that were at least 80 times lower than tissue concentrations reported toxic to pink salmon embryos (Brannon et al. 2001). These data and the sediment-PAH concentrations measured in oiled streams indicate that toxic levels of petroleum hydrocarbons were not present in the incubation environment (Brannon et al. 2001).

Geiger et al. (1996) derived modeled estimates that about 2 million fewer wild pink salmon returned to PWS in 1990 than would have in the absence of EVOS, even though they also reported that 1990 was a historical record catch year for pink salmon (with a catch of over 44 million fish). The 1991 catch was the 2nd largest recorded at over 37 million (Paine et al. 1996). Trustees (2002) estimated that fish returns in the Southwest District of PWS may have been reduced by 11%. Willette (1996) derived a model-based estimate of about 2% reduction in survival to the adult stage among pink salmon from hatcheries that were reared in oiled areas. The Geiger model treated the egg mortality reported by Bue et al. (1996) as being oil related to arrive at the estimated loss in returning adults. However, Brannon et al. (2001) concluded from reanalysis of the egg mortality data and field tests of the sampling protocol (Collins et al. 2000) that the observed egg mortality was not from oil but from the effect of sampling the eggs too soon after spawning during the period of extreme embryo sensitivity to shock. The Alaska Department of Game and Fish also reanalyzed its egg mortality data, concluding that because of sampling problems, oil effects on eggs could not be determined (Craig et al. 2002).

Both Maki et al. (1995) and Templin et al. (1996) noted that in 1990 and 1991, there was no obvious reduction in the numbers of returning pink salmon per spawner in the Southwestern District. While egg mortality may have been reduced in unoiled versus oiled streams through 1993, reduced growth rates of juveniles in oiled areas occurred only in 1989 (Wertheimer and Celewycz 1996), and PAH concentrations were below 1 $ng{\cdot}L^{-1}$ in initially oiled streams (i.e., well below any significant effects level). Moreover, Wertheimer and Celewycz (1996) attributed the higher abundance of juvenile pink salmon in 1989 and 1990 in unoiled areas compared to oiled areas to avoidance or habitat differences rather than toxicity from the oil. Rice et al. (2001) argued that differences between published estimates of initial population losses of pink salmon were driven by differing experimental methods or interpretations between research groups or by evaluation of differing sets of salmon streams, and in particular the mixing of data from oiled and unoiled streams.

However, the more relevant point here is that even if one accepts the largest estimate of potential population reduction (a modeled result of 2 million fewer returning adults than would have in the absence of EVOS), that must be placed in the context of an order-of-magnitude natural interannual variability and the fact that 1990 was a record-crop year. Clearly, those circumstances undermine any conclusion that ecologically significant population-level effects attributable to EVOS occurred in the immediate years after the oil spill. In any event, Trustees (2002) concluded that there are no continuing population effects from EVOS on pink salmon in PWS and declared the population to be completely recovered (see also Carls, Rice, et al. 2004).

Ecological significance—It is very difficult to make the case that there were detectable EVOS-caused population losses of pink salmon in the 1st year following the spill. Even if one accepts the model-based estimate of the initial wild pink salmon return reduction of 11% in the Southwestern District, that would not be ecologically significant for a population that typically has an order-of-magnitude variability in interannual numbers. The population numbers at present are within the normal range of variability, and there is no indication of significant EVOS-related population effects. Consequently, we conclude that there were no initial ecologically significant effects, and no ecologically significant effects exist today.

10) Sockeye salmon (*Oncorhynchus nerka*) is another economically important fish in PWS. This species is subject to large variability in numbers, in part driven by natural climate and circulation variability in the northeast Pacific over periods of years, decades, and centuries; fishing pressures and management; and other, more complex ecological relationships, such as nutrient dynamics (Finney et al. 2000). The salmon fishery was closed during 1989 following EVOS to avoid contaminated salmon reaching the market. This resulted in the next years having much higher than desired numbers of spawning sockeye salmon reaching the Kenai River and lakes on Kodiak Island (an event termed over-escapement). Adverse effects on the sockeye population were reported for the brood years 1989–1992 (Trustees 1999). However, sockeye freshwater growth returned to normal within 3 y, and the Trustees (2002) concluded that the population has now fully recovered.

Ecological significance—The effects of the EVOS event per se did not cause ecologically significant population reductions in the sockeye salmon, but the fishery closure, a management response to the EVOS, did lead to ecologically significant effects in the initial few years after the spill. However, no continuing, ecologically significant effects are evident today (2006).

11) Harbor seal (*Phoca vitulina*) is one of the most common marine mammals in PWS, feeding within 50 km of its haulouts in shallow, nearshore waters and diving for food at depths up to 100 m (Frost et al. 2001). It is protected under the Marine Mammal Protection Act of 1972 (16 USC §1361–1421). The primary food sources for harbor seals in PWS are pollock, octopus, capelin, Pacific cod, and herring. The species is prey for orca, Steller sea lions (*Eumetopias jubatus*), sharks, and humans (Frost 1997). The year-round populations in the PWS and GOA area were estimated at 125,000 in 1973, although more recent estimates showed significant declines in the population to about 21,000 to 34,000 by the 1990s (Frost 1997). Loughlin et al. (1996) reported a 40% decline in harbor seal counts from 1984 to 1988, and Frost et al. (2001) reported a 60% decline in harbor seal population since 1984, citing a hypothesized cause of the decline to changes in the trophic structure and availability of prey. Similarly, Mathews and Pendleton (2006) reported that the harbor seal population in Glacier Bay National Park (an unoiled area of coastal southeast Alaska) decreased from 6,200 in 1992 to 2,550 in 2002. While the reason for this major population decline is uncertain, since it has occurred throughout the region and largely pre-dated the EVOS event, it cannot be attributed primarily to the oil spill.

The most logical explanation is similar to the situation for the Steller sea lion, which has lost more than 80% of its population in GOA over the past 25 y. An NRC study (2003) suggested several plausible causes for its decline, which seem applicable to the harbor seal as well: Overfishing and climate variability reducing food resources, persistent organic pollutants such as PCBs and DDT derivatives, shifts in predator (transient orca) diets, and entanglement in fishing gear. A stable isotope study by Hobson et al. (2004) of Steller sea lions indicated a significant shift in the food base, again with plausible implications for the harbor seal populations.

The annual rate of harbor seal population decline in the 1980s prior to the spill was about 12%; however, in oiled areas, there was more than a 40% decline in local populations in 1988–1989 (compared to 11% in unoiled areas), and the total direct mortality was estimated at 300, or 13% of the estimated population of 2,200 harbor seals in PWS (Frost and Lowry 1994; Spies et al. 1996). By 1990–1994, the annual population decline rates were similar in oiled and unoiled sites (about 6% per y), and other indicators suggested no long-term population differences caused by EVOS (Frost and Lowry 1994). Nevertheless, Trustees (2002) listed harbor seals as "not recovering."

Ecological significance—Clearly, harbor seals are an appropriate VEC, and the population reductions in the immediate aftermath of EVOS were ecologically significant. The longer-term population issues, however, encompass a major decline in the population that may relate to altered food resources, climate change, predation, overharvesting of fisheries, human harvesting, or other non–EVOS related issues. This suggests that there no longer are any detectable ecologically significant effects on harbor seals that can be attributed to EVOS.

12) Orca (killer whale, *Orcinus orca*) is a highly charismatic, societally important species, the largest of the dolphin family, *Delphinidae*. It is protected under the Marine Mammal Protection Act of 1972 (16 USC §1361–1421). It is estimated that the resident population in GOA is about 800, increasing at a rate of about 2% per y, though the transient population may be in decline (Matkin and Saulitis 1997). Resident populations are primarily fish eating, whereas the transient populations are primarily mammal eating. Both populations maintain genetically and socially distinct populations, even when living in the same area (Matkin et al. 1999; Scheel et al. 2001). In PWS and the Kenai Fjords, 110 resident orcas lived in 6 pods in summer months prior to the oil spill. Three pods have been recorded in Kenai over the winter months, though the feeding range and behavior of the PWS pods over winter are unknown (Matkin and Saulitis 1994, 1997). Killer whales are the top predators in the ecosystem, and while the transient pods do not likely cause major population reductions in prey mammals in PWS, they may retard population recovery, such as for harbor seals (Matkin and Saulitis 1994; Matkin et al. 1999) and sea otters (Estes et al. 1998). The resident killer whales are estimated to consume only about 2% of returning salmon stocks, compared with up to 70% mortality by fishing pressure (Matkin and Saulitis 1994, 1997).

Dahlheim (1994) reported that photographic analysis from 1989–1991 revealed little impact from EVOS on PWS killer whale populations. However, Matkin et al. (1999) documented all the killer whales in PWS and coastal Alaskan waters, identifying the resident pods down to the individual level, in most cases with samples of skin and blubber for nuclear and mitochondrial (maternally inherited) DNA analyses. These authors also reported from their observations since the early 1980s that there has been a general and consistent increase in the resident orca population of PWS. The only exception was the PWS-resident AB pod, which contained 35 individuals in 1984 and was the most frequently sighted pod in PWS. Matkin and Saulitis (1997) and Matkin et al. (1999) reported that in April 1985, the AB pod (and only the AB pod) began to interfere with the commercial longline fishery in PWS, which was harvesting a record number of fish at that time, removing one-fourth of the total blackcod caught on the longlines and damaging fishing gear. During photo identification, bullet wounds were documented on 10 whales in the pod, 5 of which subsequently died. Between 1985 and 1986, 6 whales were lost from AB pod, a mortality rate 5 times higher than normal. The Marine Mammal Protection Act of 1972 (reauthorized in 1994) was modified in 1986 to outlaw the shooting of whales, which, along with major changes in the longline fishery regulations in PWS at the same time (Matkin et al. 1999), led to no further shootings of the AB pod. Five new calves were born in 1988, bringing the total number of whales in the pod to 36 a few months before EVOS occurred.

At the time of the oil spill, the AB pod was known to be directly in the area of the spill and was observed in oil-free waters and later within slicks (i.e., the killer whales did not avoid the oil). This indicates a very significant potential route of exposure to toxic volatile organic chemicals from EVOS through respiration (Matkin et al. 1999) and potentially an exposure route through consumption of oil-contaminated fish. Seven members of the AB pod were missing within a week of the spill, including 3 adult females and 4 juveniles who were later confirmed dead. Overall, the AB pod was known to lose 13 or 14 members over the next 2 y. Although no direct link to the EVOS event can definitively be established, the mortality rate (about 20% contrasted to a normal rate of about 2%) was unprecedented (Matkin et al. 1999), the exposure and risk were quite significant, and thus EVOS is the most plausible causal agent for the observed mortality in the early aftermath of EVOS (Dahlheim and Matkin 1993; Dahlheim 1994). Note that during the winter before and the months after the EVOS event, the longline fishery in PWS was closed, so there would have been no conflict with fishing activity (Matkin and Saulitis 1994).

Matkin et al. (1999) make a convincing story, from their extensive database and years of observations describing the close social structure of the matriarchal pods, that the loss of an important matriarch can continue to affect a pod for some years thereafter. Thus, the loss of key matriarchs from the 1986 shootings and from the 1989 EVOS event may have caused continuing further loss of AB pod members. This was particularly true for the AB6 and AB7 female lines, sister whales, both of which died prior to the spill and the offspring of which were completely eliminated by 1996. From 1990 to 2001, the AB pod hovered around 23 to 26 members (Matkin and Saulitis 1997; Matkin et al. 1999; Trustees 2002), potentially a continuing result of EVOS and/or the shooting deaths. However, the AB pod gradually increased from 22 in 1995 to 27 in 2003, and the total PWS population of resident killer whales increased from 117 in 1988 to 155 in 2003 (Matkin 2004). Matkin et al. (1999) also discussed how new pods may form by the gradual or abrupt separation of a matrilineal group, suggesting that the PWS AI pod separated from the AB pod in the 1970s and early 1980s, and other PWS pods have separated in the 1990s. The present status (Gay 2002) is listed as "recovering," downgraded from the draft Trustee report (2002), which labeled the population as "recovered."

Ecological significance—Clearly, killer whales are an appropriate VEC. Unique among the PWS VECs, there is a concern for killer whales about effects at the individual animal level. The population reduction of the AB pod in the immediate aftermath of EVOS was ecologically significant and most likely caused by EVOS, exacerbating an ongoing impact on that pod that resulted from recent human conflicts. The continued reduced population for the AB pod may still be partially associated with EVOS, as the numbers in that pod have not recovered to prespill levels. Thus, we conclude that local-scale ecologically significant effects from EVOS remain for that 1 pod. This continued ecologically significant effect, however, likely relates not to any continuing exposure to EVOS stressors but rather to long-term population dynamics and the intrinsic slow recovery rate for pods of this species. All the other resident pods in PWS have increased in numbers during the years since the 1989 spill, and the total resident population in PWS has increased in spite of the flat population level for the AB pod. Since other pods periodically intermingle with the AB pod, it is plausible to consider that the PWS orca population as a whole has recovered from EVOS and shooting stressors, while the AB pod specifically remains depressed in numbers. However, Matkin et al. (1999) has shown the demographics of the AB pod have stabilized (i.e., no more members seem vulnerable to future loss from those 2 events or their social consequences), so a continuing loss of AB pod members attributable to EVOS- or shootings-caused social disruptions seems unlikely. We believe there is no plausible risk from residual toxicity associated with EVOS because exposure rates are so low, as discussed previously. The larger PWS population of killer whale pods, both resident and transient, shows no signs of short- or long-term effects from EVOS; to the contrary, the resident populations continue the general trend of a gradual increase seen in the rest of GOA. Consequently, we conclude that no ecologically significant effects ever existed from the spill for any PWS or GOA resident killer whale pods other than the AB pod.

13) River otter (*Lutra* [also *Lontra*] *canadensis*) is an amphibious member of the family *Mustelidae* along with mink and sea otter. Its historical range extended over most of North America north of Mexico, although they have been virtually eliminated in the midwestern and eastern United States, especially around heavily populated areas. The river otter is still found throughout most of Alaska (Solf and Golden 1994), but the population in PWS is quite small. They are excellent swimmers that will also travel several miles over land between bodies of water on well-defined trails that are used year after year. River otters in Alaska breed in spring, usually in May. They hunt both on land and in freshwater and salt water, feeding on invertebrates (such as snails, mussels, clams, sea urchins, insects, crabs, shrimp, and octopi), as well as frogs, a variety of fish, and occasionally birds, mammals, and vegetation (Solf and Golden 1994; Ellis and Dewey 2003). Predators of North American river otters include birds of prey and other large predators, but humans are the primary predator; for example, over the past decade, 1,200 to 2,400 river otters have been harvested annually in Alaska for their pelts (Solf and Golden 1994; Ellis and Dewey 2003).

Twelve river otter carcasses were found after the oil spill (Bowyer et al. 2003), although no estimates were made of total direct mortality (Trustees 2002). Faro et al. (1994a) noted that during 1989–1991, there were some differences between river otters in oiled and unoiled areas in PWS, including biochemical changes, reduced prey diversity, reduced body size, and increased home-range area; however, they also noted that the lack of adequate prespill data make uncertain the assignment of these effects uniquely to EVOS. Faro et al. (1994a, 1994b) reported on studies to determine if physiological and morphological differences in river otter detected after EVOS remained in later years and concluded that no differences existed by 1992. Bowyer et al. (2003) found from a meta-analysis that detectable differences in metrics of river otter individuals (e.g., body mass, diet, home range, and liver enzyme levels) between oiled and unoiled areas in 1989–1992 were not detectable in 1996–1999, and they concluded recovery from the EVOS by the latter period. The Trustees (2002) concluded that the river otter had fully recovered.

Ecological significance—The initial effects of EVOS on the PWS river otter population were probably ecologically significant; the lack of prespill population data add uncertainty to this conclusion. However, no ecologically significant effects that can be attributed to EVOS exist at the present time (2006).

14) Sea otter (*Enhydra lutris*) is found in the coastal waters of Alaska, feeding exclusively on benthic invertebrates, including clams, mussels, sea urchins, snails, and crabs (Bodkin and Ballachey 1997). As a consequence, their habitat is limited to a relatively narrow zone along the shorelines in PWS that is sufficiently shallow for the sea otters to reach the benthic communities, primarily within the 40-m isopleth (Bodkin and Ballachey 1997). Rather than rely on fat tissues for thermal protection, sea otters rely on a dense fur with its trapped layer of air, making them prime targets of extensive human exploitation from the 18th into the 20th centuries. Sea otters were hunted almost to extinction by 1911, when protection through the International Fur Seal Treaty was implemented (Doroff et al. 2003). The population recovered from a few hundred at that time to more than 100,000 worldwide at present, about two-thirds of the prehunting population (Bodkin and Ballachey 1997). The sea otter population growth was accompanied by rapid expansion and return into about three-fourths of the original range. This return included a large and rapid areal expansion from the remnant Alaskan populations, such as those observed at Montague Island in 1936, at Latouche and Elrington Islands in 1949, at Hinchinbrook Island in 1951, and into western PWS during the 1960s (Johnson and Garshelis 1995). Estes (1990) reported the rate of increase of the sea otter population in southeast Alaska was 17.6% per year from 1975 to 1987. Expanding population trends have reversed in the past decade, however. Burn and Doroff (2005) reported broad-scale declines in sea otter populations in coastal Alaska in the 1990s, including a 70% decline in the Aleutians, 63% along the island coastlines of the south Alaska Peninsula area, and 56% to 68% for southwest Alaska as a whole; they noted the population decreases particularly involved reductions in the large rafts offshore. The US Fish and Wildlife Service (USFWS) estimated the total southwest Alaska population to be 94,050 to 128,650 in 1976, which was reduced to 41,865 in 2004. As a consequence, the USFWS listed the northern sea otter as a threatened species in southwest Alaska from Attu Island to western Cook Inlet effective 8 September 2005 (70 FR 46366).

Garshelis et al. (1984) showed that males lead the front of a population's expansion, abandoning an area as food resources are diminished to move into adjacent, unoccupied areas; females subsequently occupy the formerly male-dominated areas. Lensink (1962), cited in Johnson and Garshelis (1995), conducted an aerial survey in 1959, reporting 50 to 100 sea otters near Green Island, 450 to 575 near Montague Island, and 75 to 100 near Hinchinbrook Island. Sea otters had reached Knight Island by 1970 and continued expansion into eastern PWS during the 1970s and beyond to the GOA coast south of Cordova in the 1980s (Johnson and Garshelis 1995). This recent expansion of the species' range resulted in an estimated 5,000 to 10,000 individuals in PWS at the time of the oil spill. This population level was not considered to have reached carrying capacity for PWS (Burn 1994a, 1994b), except perhaps for localized areas such as Green Island that had been recolonized earlier (Johnson and Garshelis 1995). Tinker (2004) suggested that food resources limit carrying capacity, with density-dependent reduction of foraging success as prey are depleted and associated potential reduction of the health of the individual by increased susceptibility to disease through compromised immune systems. Adaptive strategies for optimal population fitness that appear to be used by sea otters include specialization of diets among individuals, passed on across generations (Tinker 2004), and complex strategies for establishing and maintaining territories (Garshelis et al. 1984).

The dense fur and unusually high metabolic rate also make sea otters particularly vulnerable to the physical effects of oiling, including decreased buoyancy and hypothermia from loss of insulating capacity (Lipscomb et al. 1994). The latter effect is especially important in PWS because it is at the northern limit of the population range (Johnson and Garshelis 1995). Sea otters are also vulnerable to inhalation of toxic volatile organics or ingestion of oil-contaminated food, exacerbated by increased rates of grooming following oiling (Johnson and Garshelis 1995). A total of 871 carcasses were found after EVOS, and 123 additional otters died in rehabilitation centers, totaling an observed mortality of about 1,000 (Estes 1991; Loughlin et al. 1996; Trustees 2002). More than 350 oiled sea otters were treated, and about 200 were rehabilitated and released (Monahan and Maki 1991), but there was a relatively low survival rate among the released sea otters (Estes 1991). Spies et al. (1996) estimated direct sea otter mortalities of 3,500 to 5,000, including 2,800 in PWS (Garrott et al. 1993). Estes (1991) commented on the lack of immediate prespill information on the sea otter populations in the exposed area and suggested a lack of power in the data to demonstrate population effects. Ballachey et al. (2003) reported about 500 carcasses in western PWS in the months following the EVOS, and they noted that chronic effects may have influenced survival through subsequent years, through sublethal exposures in 1989, continued exposure subsequently to residual oil, or alterations in prey populations.

Bodkin et al. (2002) stated that the sea otter populations were mostly missing the following year (1990) from their normal habitats in heavily oiled areas, although lack of sufficient prespill population data makes estimates of direct mortality highly uncertain. Burn (1994a, 1994b), on the other hand, conducted shoreline surveys of PWS by boat in 1989–1991, finding a 35% decline in shoreline density within oiled areas based on comparisons of pre- and postspill data. However, he found that further (albeit not statistically significant) declines in shoreline sea otter densities in 1989–1990 occurred in both oiled and unoiled areas. Burn (1994b) also reported speculation that unusually high mortalities of sea otters in unoiled areas of PWS the year after the oil spill may have been caused by disease introduced with the release of rehabilitated otters into those areas.

Johnson and Garshelis (1995) counted fewer otters than prespill counts only at 1 of 3 heavily oiled sites investigated in 1991. A 2nd of those heavily oiled sites showed a significant increase in numbers pre- to post-spill. They also showed no apparent spill-related effects on either sea otter distributions or pup production. Bodkin et al. (2002) reported that between 1993 and 2000, the number of sea otters in the spill area of PWS increased from 2,100 to nearly 2,700, indicating that the population in the potentially EVOS-affected area of PWS was recovering. Bodkin and Udevitz (1996) conducted a PWS-wide aircraft survey of the sea otter population, with an estimated total population of over 16,800 individuals in 1993. Bodkin and Dean (2000) reported that the PWS population was 12,289 in 1994. The 1995 PWS population was estimated to be 13,000 by Bodkin and Ballachey (1997), and the 1999 PWS population was estimated at 13,234 (Bodkin and Dean 2000; NOAA 2002).

These counts suggest to us that the PWS-wide population had recovered from the effects of EVOS by the mid-1990s. Bodkin et al. (2002), however, suggested that the population recovery had been delayed, based on a 1998 survey of northern Knight Island, which found about 75 otters, compared to a reported 235 in the same area surveyed in 1973. That 235 figure was derived from unpublished data reported in Dean et al. (2002), but we believe it would seem to exceed the carrying capacity for sea otters in so small an area (considering the Green Island numbers cited previously). Nevertheless, Bodkin et al. (2002) concluded from this single subpopulation comparison against an unpublished prespill number that at least local populations had not fully recovered after 10 y. This conclusion was contradicted by Johnson and Garshelis (1995), whose counts on northern Knight Island were larger in 1991 than the 1984 and 1985 counts they cited in an unpublished report by Irons et al. (1988). Johnson and Garshelis (1995) also noted that by 1991, the sea otter population in the oiled areas had equaled or increased over pre-spill levels (based on 1984 estimates) at all their survey sites and that western PWS continued to have increasing numbers of sea otters through 1996. Doroff and Bodkin (1994) found no differences in foraging success between oiled and unoiled areas 2 y after EVOS. However, Bodkin and Dean (2000) reported that subpopulation abundance remained unchanged from 1993 to 1999 at about half the pre-spill level.

Bodkin et al. (2002) concluded that continued toxic exposures to residual EVO may be a contributing cause of slower population recovery. However, we believe it is difficult to support that conclusion for several reasons. For example, Doroff and Bodkin (1994, 1997) found no differences in hydrocarbons in the sea otter's subtidal bivalve prey in oiled versus unoiled areas in sampling done in 1991. Rebar et al. (1996) found that hematologic and other chemical differences between sea otter pups and adults in eastern (unoiled) versus western (oiled) PWS were minimal and of equivocal biological significance in 1989–1990 (i.e., at a time shortly after the oil spill when the greatest differences would be expected). Boehm et al. (1996) concluded that only 1 y after the oil spill, the risks to wildlife, including sea otters, of toxic effects from consumption of EVOS-contaminated mussels were very low. Similarly, Boehm et al. (2004) showed very low levels of bioavailable hydrocarbons in mussels from the few sites identified by NOAA as having remnant EVO, having returned to background levels as determined from nonoiled and non–HA sites by 2002. They also found that PAHs from unoiled but previous HA sites exceed remnant EVOS sites, but total risk from all EVOS and non-EVOS sources is negligible. Neff et al. (2006) found that the small amounts of EVOS-derived PAH that persisted in intertidal sediments in 2002 have a low accessibility and bioavailability to intertidal plants and animals and consequently are not entering the food web of the Sound. Neff et al. (2006) also included a detailed discussion of potential pathways of exposure of sea otters to EVOS oil residues (food and pelage contamination) and concluded that sea otters are at low risk of injury from consuming clams and mussels from formerly oiled shores.

In contrast to their own conclusion about continued toxic effects affecting population recovery, Bodkin et al. (2002) found that sea otters foraged more successfully, and the condition of young females was seen to be superior at the oiled sites on Knight Island, suggesting that continued population differences may also be associated with different emigration and immigration rates. They also suggested that possible effects of social organization and behavioral responses may be contributory (Bodkin et al. 2002). However, Ballachey et al. (2003) concluded from their monitoring of pups in eastern and western PWS during 1992–1993 that area differences in survival rates, foraging success, and blood parameters suggested that the oiling history may have affected juvenile survival 3 to 4 y after the oil spill. Doroff et al. (2003) examined colonies of sea otters in the Aleutian archipelago, comparing data for 1965, 1992, and 2000; they showed a uniformly low population density throughout the area and concluded a common and geographically widespread cause for the recent population declines, attributing the cause to increased predation.

Estes et al. (1998) attributed the recent precipitous declines in the sea otter populations of large areas of western Alaska, discussed previously, to increased predation by killer whales. Such increased predation may be driven by shifts in the food resources for orcas caused by declines in the Steller sea lion and harbor seal populations (Hatfield et al. 1998), which in turn may have been caused by declines in fish populations (NRC 2003). Garshelis and Johnson (2001) further noted that one-third of the observed orca–sea otter predation events reported by Hatfield et al. (1998) occurred in PWS at Knight Island, suggesting that this could be the cause of the lack of recovery of the local sea otter population.

Peterson et al. (2003) suggested that a trophic cascade effect involving sea otters, kelp, and sea urchins (such as seen in coastal California and Aleutian Island ecosystems; see Estes and Palmisano 1974) could still be anticipated in PWS from EVOS, even though no such cascading effect had been seen as of that point in time (14 y after the oil spill) or since. Estes et al. (2004) and Reisewitz et al. (2006) reported that such a cascading effect did occur in the Aleutian archipelago from the loss of sea otters as a part of the post-1990 decline regionally, attributed to increased transient orca predation. They reported that a number of islands that previously had the typical sea otter–dominated trophic structure experienced a rapid ecosystem phase shift to a typical urchin-dominated state. This rapid response suggests to us that a time-lagged response hypothesized by Peterson et al. (2003) is not plausible.

We believe that an important difference between the Aleutian archipelago and PWS situations is the continuing, long-term reduction of sea otters in the Aleutians, contrasted with the transient reduction resulting from EVOS. Moreover, the sea otters of PWS have a quite varied diet, much less dominated by sea urchins, and the sea urchin–dominated kelp habitat is much less dominant in the habitat mosaic of the PWS ecosystem. We believe that the primary drivers for cascading effects in the PWS ecosystem relate to fundamental processes driven by climatic and oceanographic variability, especially involving nutrient upwelling and fish productivity.

Finally, Garshelis and Garshelis (1984), through tagging studies on sea otter movements in PWS, concluded that the lifetime home range for at least the males probably include the entire range of sea otters in PWS; this suggests that the appropriate scale for assessing sea otter population effects is PWS-wide rather than a local subpopulation. The Trustees (2002) list sea otter as "recovering."

Ecological significance—Sea otters are obviously important VECs, and the PWS population experienced uncertain but clearly ecologically significant effects from the initial stressors associated with EVOS. The current situation is more complicated, but we conclude overall there are no continuing ecologically significant effects on the PWS population. There may be a continued reduction in the local subpopulation in 1 heavily oiled area on northern Knight Island, although even this is uncertain because prespill numbers are not well-documented and are questionable, whereas other heavily oiled areas have the same or increased numbers compared to prespill. The possibly reduced numbers in this 1 subpopulation of sea otters may or may not be associated with EVOS, with speculations in the literature on EVOS-caused mechanisms including continued effects on population migration rates and continued exposures to toxic chemicals. However, the potential PAH exposures to sea otters via the food web, as documented in mussel body burdens near EVOS and HA sites, are well below levels to cause adverse effects and thus we believe do not constitute a current risk to the sea otters of PWS, including this subpopulation. Other plausible causal factors not associated with EVOS could relate to reduced numbers in this subpopulation, including observed predation by transient orcas in the area. Thus, local-scale ecologically significant effects on northern Knight Island sea otters may or may not remain and may or may not be attributable to EVOS. However, the PWS-wide population of sea otters has returned to prespill range of numbers and may even exceed the prespill population level. We conclude that at the PWS-wide scale no continuing ecologically significant effects exist. Moreover, recent larger-scale declines in sea otter populations throughout GOA suggest non-EVOS stressors have been at work since the 1990s, including altered food web structures that have caused diet shifts and increased predation on sea otters by transient orcas.

15) Clams of various species (including *Protothaeca staminea*, *Saxidomus giganteus*, *Clinocardium*, and *Macoma*) are important to PWS both ecologically and societally. With biomass estimated at 80 tons·km^{-2} in intertidal and subtidal areas of PWS, clams provide an important food resource for benthic invertebrates, marine mammals like sea otters, and shorebirds (NWF 2002). Because of the high site-to-site natural variability in clams, the absence of prespill characterization of the clam communities of PWS, and the absence of quantitative data characterizing the effects of EVOS or the cleanup, it could be argued that clams are not a good choice as a VEC. However, we include clams here because they were directly impacted by the extensive cleanup activities, and the reestablishment of clam populations in cleaned areas is an important indicator of recovery.

Shigenaka et al. (1999) and Trustees (2002) stated that littleneck and butter clams likely were affected by EVOS, with both mortality and lower growth rates following the oil spill, although no quantification of these effects was done. Because of differences between oiled sites on Knight Island and unoiled sites on Montague Island in the numbers of juvenile clams and numbers of clam species, Trustees (2002) listed clams as "recovering" but "not yet fully recovered." There is considerable variability in clam abundances across PWS, and many physical factors significantly affect local clam communities, including sediment particle size, tidal and current regime, and exposure to physical disturbances. As a result, we believe that it is quite plausible that site difference factors other than EVOS are responsible for present differences in clam communities. Several studies have shown that clams have not been significantly contaminated with EVOS PAH since the early 1990s (Doroff and Bodkin 1997; Roberts et al. 1998; Shigenaka et al. 1999; Neff et al. 2006) and thus are not an important vector of petroleum exposure to sea otters and sea ducks. Skalski et al. (2001) showed recovery of the intertidal community by 1994, including the clam populations, indicating no longer-term effects remain from the cleanup activities. Trustees (2002) assigned the subtidal community, which includes clams, as having recovered.

Ecological significance—Because a considerably large area of intertidal and subtidal habitats supporting clams was exposed to EVOS oil and cleanup stressors, it is likely that ecologically significant effects occurred following EVOS. However, we believe that the lack of continuing exposures and the recovery of the subtidal habitats indicate that effects on clam populations attributable to EVOS are no longer ecologically significant.

16) Mussels (*Mytilus trossulus*) constitute an important food resource for the intertidal ecosystems throughout the oil spill area and provide physical stability and habitat structure for other organisms in the intertidal zone. The PWS mussels were not noted as having significant population effects from EVOS, but they have been major sinks for residue of EVOS oil, continuing to hold residues for several years. This is particularly a concern in the underlying byssal mats and sediments, which provide physical protection of the oil residue from coastal wave action (Trustees 2002). A number of oiled mussel beds located on soft sediments were deliberately not a part of the initial cleanup activities. Since then, about 30 mussel beds have been located that contained remnant oil, mostly in relatively protected areas that are not as exposed to natural disturbances (Babcock et al. 1996, 1998). Twelve such beds were cleaned in 1993–1994;

however, some of these had become recontaminated in 2 y, presumably from oil deposits underneath the beds (Babcock et al. 1998).

Continued contamination of mussel beds offers a potential route of longer-term exposure of some vertebrate species feeding on mussels (e.g., black oystercatchers, harlequin ducks); the issue of significance thus becomes that of the ecological significance of exposure from the contaminated beds, discussed previously. The Trustees (2002) continue to list mussels as "recovering" because of purported continued contamination from oil residues. In contrast, Hoff and Shigenaka (1999) showed that the mean PAH levels in mussels in oiled versus unoiled sites were not significantly different after 1992, and Boehm et al. (1996) demonstrated that there was no EVOS-derived risk from consumption of mussels as early as 1993 and certainly not currently. Boehm et al. (2004) demonstrated that PAH levels in mussels in 2002 were at or near background levels, including sites known to have subsurface oil residues (see also Page et al. 2005).

Ecological significance—There apparently was no population-level effect on mussels from direct impacts of the EVOS and thus no ecologically significant effects on the mussels. However, the role of mussel beds as a potential source of release of EVOS chemicals into the environment after the oil spill made it a potentially ecologically significant exposure source. This issue is further discussed in the exposure significance section. Nonetheless, we conclude that the present residual contamination in mussels is insufficient to cause any continuing exposure of ecological significance to PWS.

17) Intertidal communities: The EVOS led to oiling of about 2,000 km of shoreline, much of it with an extensive intertidal zone in the Sound with typical tidal reach of 3 to 5 m. Like the subtidal zone, the intertidal communities of PWS are very important ecologically, providing a major food resource for sea otters, shorebirds, and other species and providing habitat for a diverse community of invertebrate species. The EVOS caused extensive damage to the intertidal habitats of PWS, including chemical toxicity and physical oiling from the oil spill as well as extensive physical and chemical stressors from the cleanup activities. However, Neff et al. (2006) found that the small amounts of EVOS-derived PAH that have persisted in intertidal sediments in 2002 have a low accessibility and bioavailability to intertidal plants and animals and consequently are not entering the food web of PWS.

The distribution and productivity of the alga *Fucus*, a species that was especially adversely affected by the cleanup activities, are particularly important to this habitat. Thus, *Fucus* is an important indicator of the health and recovery of the intertidal habitats of PWS. Trustees (2002) reported remaining differences between oiled/cleaned sites and unoiled sites with respect to *Fucus* populations as of 1997 and reported other intertidal habitats (e.g., estuaries and cobble beaches) not recovered as of 1991. The *Fucus* community recovered more rapidly in noncleaned areas; for example, Houghton et al. (1997) concluded that nontreated communities had nearly completely recovered by 1991, although oscillations in species abundances continued thereafter. In contrast, the cleaned areas showed slow recolonization through 1995, and areas apparently recovered in 1992 experienced subsequent severe declines in dominant taxa in 1995. As van Tamelen and Stekoll (1996) noted, observations of *Fucus* at heavily oiled sites suggest that *Fucus* may be able to withstand a fairly high degree of oiling. Thus, severe cleanup methods (e.g., high-pressure hot-water washing) resulted in the physical removal of the entire algal community in an attempt to prevent harm from the oil but actually may have caused more harm than the oil itself and delayed community recovery (Highsmith et al. 1996, 2000; Lees et al. 1996; Mearns 1996; van Tamelen and Stekoll 1996; van Tamelen et al. 1997).

In a study on the effects of the high-pressure hot-water washing, Houghton et al. (1996) concluded that although much of the rocky intertidal epibiota survived the oiling effects from EVOS, much of the community was displaced or killed by the cleanup. Driskell et al. (1996) also demonstrated that intertidal biota at oiled but untreated sites rebounded quickly, whereas recovery on cleaned sites lagged considerably. On the other hand, Gilfillan, Page, et al. (1995) and Gilfillan et al. (1999) showed that approximately 90% of the shoreline communities (including *Fucus*) had recovered by 1990, suggesting that the cleanup activities did not have as significant a long-lasting effect as others claimed. Moreover, statistical analyses by Skalski et al. (2001) on the extensive monitoring database collected between 1989 and 1997 (see Houghton et al. 1993) demonstrated that the intertidal community in oiled but uncleaned areas had recovered by 1992 and in oiled and cleaned areas by 1994. In any case, other than aesthetics and response to public perceptional pressures, ecologically a better strategy for recovery of the intertidal zone may have been to let the natural processes of waves and weather, especially associated with winter storms, clean up the coastline. Nevertheless, even if of questionable value, the physical removal of the intertidal communities is attributable to EVOS and thus is a proper subject of assessing its ecological significance.

Ecological significance—The adverse effects on intertidal habitats from EVOS and the cleanup activities were spatially extensive and ecologically significant. The recolonization of intertidal communities has proceeded as expected, with areas that were intensely cleaned essentially constituting primary succession on bare rock substrate. Whether cleaned or not, the intertidal communities had recovered within 5 y of EVOS. As a result, we conclude that the intertidal habitats a decade later no longer experience ecologically significant effects from EVOS.

18) Subtidal communities are important habitats in PWS, extending from the intertidal zone to a depth of about 20 m. These habitats are characterized by dense beds of kelp (including *Agarum*, *Nereocyctis*, and *Laminaria*) along exposed shorelines or eelgrass (*Zostera marina*) in more protected bays. These communities have a diversity of invertebrate species, including clams, sea urchins, amphipods, polychaetes, crabs, and other species that form a significant food resource for fish, seabirds, and marine mammals. Significant areas of subtidal habitat were exposed to EVOS, and ecologically significant effects on the plant and animal populations in the subtidal zones occurred as a result of chemical toxicity as well as turbidity, sedimentation, and other physical disturbances, especially associated with the extensive cleanup activities following the oil spill (Jewett and Dean 1997). Jewett and Dean (1997) concluded that most components of the seagrass community had recovered by 1995, with some exceptions that may relate to oiling or to inherent site differences not associated with EVOS. Jewett et al. (1995) reported that the nearshore subtidal community PAH concentrations in sediments had returned to background levels (<100 ng·g^{-1}) by 1993. Trustees (2002) concluded that the present differences between oiled and unoiled areas are attributable to natural factors, and the subtidal communities were assigned to the recovered category; however, the Trustees' Web site in 2005 (http://www.oilspill.state.ak.us/facts/status.html) still identify the recovery status as "unknown."

Ecological significance—Because extensive areas of subtidal habitats were directly affected by EVOS or associated cleanup activities, the initial effects were ecologically significant; however, we conclude that these habitats have recovered, and no detectable ecologically significant effects remain that are attributable to EVOS.

19) Trophic structure of PWS and GOA is one of the VECs that we have added to the list of predominantly species-level indicators identified by the Trustees. This additional VEC was included to enhance the capacity of the complete suite of VECs to capture ecologically important attributes and meet our criteria for assessing ecological significance. If EVOS (or any other stressor for that matter) changes the trophic structure of PWS, then that would constitute a very significant ecological effect because it relates to the fundamental structure and functioning of the ecosystem, with implications for all the other VECs as well as for the rate of ecosystem recovery.

A great deal of information indicates that these coastal ecosystems are, in fact, undergoing major shifts in trophic relationships, but only 1 transient linkage of this phenomenon applies to EVOS. That 1 instance involved the sockeye salmon fishery, which, as previously noted, was closed following the EVOS, resulting in excessive numbers of spawners in the Kenai River and Kodiak Island lakes in 1990. The resulting high numbers of sockeye juveniles overgrazed the zooplankton populations in their nursery lakes. This fundamentally affected the overall food webs of these lakes, causing reductions in sockeye juvenile growth rates during the freshwater part of the life cycle and reduction in returning sockeye adults in subsequent years (Trustees 2002). However, this situation returned to normal within 3 y (Trustees 2002), so no long-term trophic changes ensued from EVOS. Also, Celewycz and Wertheimer (1996) concluded from their study of zooplankton and epibenthic crustaceans in western PWS that EVOS did not reduce the availability of the prey resources of juvenile salmon, indicating the absence of an EVOS-caused food chain effect.

This localized, transient effect of EVOS on trophic structure of the PWS lakes should be considered in the larger context of other stressors. Many studies have shown that major changes in the trophic structure and dynamics of the northern Pacific, GOA, and PWS ecosystems have occurred over many decades, related to climate and ocean circulation variability, overfishing, and other non-EVOS factors. These trophic changes indicate the potential for significant cascading effects on PWS. For example, McGowan et al. (1998) documented the relationship between decadal-scale climate and ocean circulation variability and biological responses in the northeast Pacific, including zooplankton and fish populations, trophic structures, and other ecosystem attributes. These authors examined long-term daily coastal sea-surface temperatures since 1916 and higher-frequency data since 1947, indexing ecosystem changes in the northern Pacific to changes in sea-surface temperatures for 80 y, including ENSO events and interdecadal changes. They hypothesized climate-driven changes in upwelling, nutrients, plankton, fish populations, and many top predators. Finney et al. (2000) conducted a high-resolution study of diatoms, cladocerans, and nitrogen isotopes over the past 300 y from cores collected from lake sediments on Kodiak Island and Bristol Bay area. They determined that pronounced changes in salmon populations were related to climate change.

Reporting on the SEA study of the oceanography of PWS and northern GOA, Vaughan et al. (2001) stated that physical processes, including surface stratification, upper-layer circulation, and PWS/GOA exchange, are capable of regulating the spatial distribution and abundance of biological components in PWS, including Pacific herring (see also Kline 1999). The study by Eslinger et al. (2001) showed that phytoplankton and zooplankton variability in PWS is determined primarily by winds and associated convective currents and air temperatures during a relatively short period each spring. They also noted the importance of GOA-derived carbon to juvenile fish in PWS and that interannual differences in the PWS fish populations reflect variability in transport processes delivering plankton into nearshore waters.

Cooney et al. (2001) and Willette et al. (2001) hypothesized that the high interannual variability in salmon is in part driven by interactions between the variation in quantities and timing of macrozooplankton in PWS, the primary food source for the young salmon (itself driven by variation in hydrodynamic circulation and upwelling patterns), and the numbers and timing of fish predators (e.g., Pacific herring and walleye pollock) that consume early life stages of pink salmon (predators that may switch to pink salmon if the macrozooplankton densities are low). In addition, the vulnerability of salmon fry to fish predation is a function of fry and predator size (Willette 2001), which in turn is a response to food availability and water temperatures. Large mortality of pink salmon fry occurs by seabird predation (e.g., black-legged kittiwakes, Bonaparte's gulls, and Arctic terns; Cooney et al. 2001; Willette et al. 2001). Thus, the large interannual variability in pink salmon populations, measured by the numbers of fish returning to spawn, results from a complex set of interactions, fundamentally physically driven, with many biological interactions that have little to do with egg productivity/mortality (see also Finney et al. 2000). Further, Kaeriyama et al. (2004) hypothesized that high-seas salmon adapt to climate-induced changes in their prey resources by switching their diets either within or between trophic levels, further complicating the picture.

Another important example is the loss of over 80% of the Steller sea lion population in GOA over the past 25 y, which only recently appeared to be reversing (O'Harra 2002), and the spatially large-scale decline in the Steller sea lion populations since the 1960s and 1970s, including southern and central California, the GOA, the Aleutian Islands, the central Bering Sea, and Russia (Calkins et al. 1994). Miller et al. (2005) and Bograd et al. (2005) suggested that these effects related to interdecadal changes in GOA circulation and sea-surface temperatures following a 1976–1977 climate regime shift. This suggests that non–EVOS related drivers are affecting the upper trophic levels of the GOA and PWS ecosystems.

A recent report on Steller sea lions by the National Academy of Sciences (NRC 2003) offered 2 major hypotheses for the changes in the Steller sea lion population, a bottom-up and a top-down control, which have similar implications for harbor seals and other fish predators. These hypotheses include 1) bottom-up control (overfishing reducing food resources; climate variability in the 1970s changing abundance and distribution of fish food resources; nonlethal diseases affecting foraging efficiency; and pollutants, especially chronic exposures to PCBs and DDT derivatives, in food web affecting fecundity or mortality) and 2) top-down control (change in predator [e.g., transient orca] food preference, incidental take or entanglement in fishing gear, subsistence harvesting, poaching, and pollution/disease affecting mortality independent of nutritional effects; NRC 2003).

Studies by Hobson et al. (2004) using stable isotope ratios on tooth annuli of Steller sea lions covering the period of the 1960s through the 1980s indicated a significant shift in their food base, supporting the bottom-up control hypothesis. Wiens (1996) noted that changes in the flow patterns of the Alaska Coastal Current and sea-surface temperatures of the GOA during the 1970s to the 1990s and associated shifts in fish populations and seabird diets suggest that the seabird populations of the region may have been under significant food stress and already declining prior to EVOS. Similarly, Piatt and Anderson (1996) identified several potential factors that could explain adverse trends in common murre populations of the region, including depressed populations, reduced breeding success, and delayed breeding phenology. They suggested causal factors such as altered coastal circulation, variability in climate and sea-surface temperatures, and altered food resources and diet shifts, among others.

For the marbled murrelet population, Kuletz (1996) reported a 66% decline in PWS since 1973, and Nelson (1997) reported 50% to 75% declines in the northern GOA in the past 20 y. The Trustees (2002) reported on studies relating juvenile productivity of marbled murrelets to abundance of forage fish, such as Pacific herring, which Norcross et al. (2001) demonstrated is closely coupled to physical conditions in GOA. Similarly, the Trustees' Alaska Predator Ecosystem Experiment investigated the possible link between pigeon guillemot declines and the availability of high-quality forage fish, such as Pacific herring and sand lance, and found a strong connection between the availability of certain prey fishes, especially sand lance, and guillemot chick growth rates, fledging weights, and nesting population size. The PWS guillemot population had declined significantly before the oil spill, likely associated with reduced food resources and increased predation (cf. Oakley and Kuletz 1994, 1996; Sanger and Cody 1994; Wiens 1996; Trustees 2002).

As an important example of cascading indirect trophic effects of these changes, Estes et al. (1998) noted that declines in the Steller sea lion and harbor seal populations (see also Hatfield et al. 1998) may have been caused by declines in fish populations used as food resources (NRC 2003) that subsequently led to increased predation on sea otter by transient killer whales, perhaps causing precipitous declines in the sea otter populations of large areas of western Alaska.

The conclusion from these many lines of evidence is that the trophic structure of the PWS and GOA ecosystems is fundamentally affected by large-scale natural processes, especially climate and ocean circulation variability, as well as anthropogenic activities, especially overharvesting of fish populations. None of these stressors is related to EVOS. These drivers have caused major shifts in the trophic structure of these ecosystems, demonstrated by changes in fish biomass, changes in piscivorous bird populations, and changes in marine mammal abundances, occurring over decades and longer time frames. We believe that these fundamental relationships provide the basis for cascading effects on the PWS ecosystem from stressors, in contrast to an as-yet-unrealized hypothetical cascading effect on PWS from initial mortality of sea otters, as suggested by Peterson et al. (2003).

Ecological significance—The only indication of trophic changes associated with EVOS was transient and limited to lakes connected to PWS but not to PWS itself. Thus, the ecological significance of EVOS on PWS trophic structure in the initial period was modest and limited in scale. We conclude that there are no continuing ecologically significant effects of EVOS on the trophic structure of the PWS ecosystem. The fact that EVOS has not caused cascading effects on the PWS trophic structure, in contrast to major trophic changes that have cascaded throughout the PWS ecosystem over the past few decades from climate and oceanic variability, puts the EVOS event in the context of other environmental stressors that have had tremendous effects on the PWS ecosystem.

20) Water quality and biogeochemical processes: We added this VEC to the Trustee list of endpoints, again in order to enhance higher-level ecosystem attributes beyond the population level. This VEC also is one for which a significant change would have very significant ecological consequences because of the fundamental nature of the VEC. The NWF (2003) listed water quality as an attribute of concern for PWS, defining it to include pollutants, such as routine and accidental releases from cruise ships and other vessel traffic; "persistent organic pollutants," such as DDT derivatives and PCBs; marine debris; and invasive species, especially from ship ballast waters. Here we add issues of nutrients and biogeochemical processes.

The NWF (2003) report stated that while there is a paucity of information on water quality in the area, with the exception of some point-source pollution, the water quality in PWS is generally high. As the report notes, PWS is a rare fjord–estuary, with significant inputs of freshwater from streams and tidewater glaciers, mixing with marine waters entering through the Hinchinbrook Entrance, bringing the major source of nutrients into PWS. Turnover of the waters in PWS occurs several times each year, providing a significant flushing action that helps maintain water quality. The marine-derived nutrients lead to significant planktonic primary productivity in the spring when adequate sunlight reaches the Sound, and these plankton, supplemented by macroalgae and seagrass production, form the primary PWS food base, resulting in high diversity and large populations of invertebrates, fish, birds, and mammals.

Recreational tourism in Alaska has increased rapidly since the time of the oil spill. As an indicator, Colt (2001) reported that the number of summer visitors (May–September) arriving by air into Alaska increased from 150,000 to 450,000 from 1989 to 1998, and the number arriving by cruise ships increased from about 300,000 to almost 600,000. The highway tunnel to Whittier was completed in 2000, and the number of visitors to PWS has increased dramatically, with concomitant increases in anthropogenic stressors on the system. Moreover, NWF (2003) stated that 1.4 million people are expected to access PWS via Whittier by 2015. The Trustees (2005) reported 1.2 million visitors to PWS in 2001, double the amount in 1989; the number of sportfishers in PWS increased by 65% from 1989 to 1997. While the nutrient stressor implications of such an increase in human activity on and near PWS have not been quantified, they obviously greatly exceed any nutrient change implications from EVOS, either directly from the oil spill or indirectly, such as through transient changes in fish populations.

With respect to natural dynamics of nutrients, in this case involving PWS-affiliated lakes, Finney et al. (2000) found a positive relationship between sockeye abundance, estimated from marine nutrient loads in lake sediments near Kodiak Island, and climate variability, estimated as GOA sea-surface temperatures reconstructed from tree-ring analyses from 1700 to 1800. They noted that that relationship changed once commercial fishing began in the late 1800s, indicating the confounding influence of harvesting through effects on nutrient dynamics associated with reduced numbers of returning salmon. Large-scale coherence of variability in salmon populations both before and after commercial fishing began indicates that large-scale forcing of the ocean–atmospheric system of the northern Pacific and GOA contributed to interannual variations in salmon populations through impacts on nutrient dynamics (Finney et al. 2000). These authors noted that since >99% of the biomass of returning salmon entering lakes is marine derived, the salmon-contributed nutrients are significant sources of nutrients into these naturally oligotrophic systems. For example, they found that in Karluk Lake on Kodiak Island, marine salmon contributed more than half the annual water column nitrogen and phosphorus compared with insignificant atmospheric and anthropogenic inputs. These nutrients are subsequently transferred to higher trophic levels, thus fundamentally driving the biogeochemical processes of the lakes associated with PWS. Further, the lake diatoms and cladocerans in the sediment cores were closely correlated with sockeye escapement data (i.e., the numbers of fish returning to the lakes to spawn). The strong decline in escapement from 1939 to 1984 correlated to an increase in benthic diatoms, reflecting the relative decrease in planktonic production. Further, they noted that since commercial fishing began in the region in 1882, total production of sockeye salmon declined from 2 million fish around 1910 to 500,000 in the 1970s. They concluded that the prolonged decline in the sockeye fishery was driven in part by reduced nutrient inputs to the lakes because of fewer returning salmon. With respect to EVOS-derived impacts on lake nutrients, the greatly increased escapement of sockeye salmon because the fishery was shut down in 1989 plausibly led to nutrient impacts, but, as the Trustees (2002) noted, the sockeye freshwater populations returned to normal within 3 y, so no long-term impact occurred. Clearly, the large-scale dynamics of this example coupling lake nutrient status with GOA dynamics would vastly overwhelm any EVOS signal.

Ecological significance—As noted, there are few monitoring data on the water quality of PWS. However, we can reach some reasonable conclusions about the relative importance and ecological significance of EVOS compared to other nutrient and biogeochemical drivers in PWS:

1. The EVOS event itself released a modest amount of nutrients compared to the normal inputs of nutrients from marine waters; the rapid turnover time of PWS water (several times a year) would long since have eliminated any nutrient inputs directly from EVOS.
2. The natural nutrients/biogeochemical dynamics in PWS are driven by exchanges with marine waters and substantially affected by large-scale circulation dynamics of the northern Pacific and GOA systems and associated climate variability processes.
3. Order-of-magnitude variability of fish populations returning to PWS and its associated lakes and streams can have a tremendous effect on the freshwater components of the system, ultimately feeding back to affect fish productivity. This natural process potentially was affected by the overescapement in 1990 caused by the closure of the salmon fishery, but such an effect was transient, recovering within a few years. By contrast, the continuing overexploitation of Pacific fish populations has continuing impacts on the nutrient dynamics of these freshwater systems.
4. The tremendous increase in tourism and other human activities in and around PWS since EVOS has implications for the nutrient status of PWS vastly greater than any direct or indirect effect from EVOS.
5. Finally, we previously discussed at length the relative contributions of PAHs from residual sources from EVOS, from other HA sites, from continuing discharges from ship traffic and other pyrogenic sources, and from natural oil seeps in GOA. Our conclusion is that EVOS no longer is an ecologically significant source of such persistent organic pollutants, that its contribution at present is comparable to the other abandoned HA sites, and that natural and continuing anthropogenic sources are much more of a risk to the water quality VEC of PWS.

21) Designated wilderness areas: Among the areas oiled by the EVOS were regions that had been designated wilderness areas, including parts of the Katmai National Park, the wilderness study areas in the Chugach National Forest and Kenai Fjords National Park, and Kachemak Bay Wilderness State Park (Trustees 2002). The wilderness values that were impacted by EVOS include the physical and chemical contamination from the oil itself, the physical and chemical disturbances from the extensive cleanup activities in the first 3 y following the oil spill (Mearns 1996), the ecologically significant effects on VECs within the designated wilderness areas at the time of and following the oil spill, degradation of aesthetics associated with wilderness (e.g., noise pollution and noxious odors), and the perceived loss of the pristine character of the area. The Trustees (2002) considered this VEC to be "recovering" but "not yet recovered" because of the existence of residual contamination in the designated wilderness areas.

Ecological significance—The designated wilderness area is an appropriate VEC, following the criteria discussed previously for selecting VECs, because societally important endpoints are legitimate VECs and clearly PWS has important value to society, in part, because of its wilderness character. It is also clear that the nature and extent of the EVOS event and its cleanup activities constituted ecologically significant adverse impacts on this VEC in the months and years following the oil spill. The present (2006) extent of residual EVOS-derived contamination, however, is not ecologically significant. We believe that all other chemical and physical stressors caused by EVOS or the cleanup activities also are no longer ecologically significant. As a consequence of multiple and cumulative natural and non-EVOS anthropogenic stressors, the wilderness character of designated areas of PWS has significantly degraded over the past century or more and can be expected to continue to do so, particularly because of the increasing human activity in PWS. We believe that this continued degradation, however, is no longer significantly attributable to EVOS.

22) Landscape mosaic of habitats: This VEC was added to complete the higher-level ecological attributes for assessing ecological significance. In general, this VEC refers to the spatial extent, location, frequency distribution, and other landscape characteristics of the distribution of habitats across the landscape and seascape of PWS and associated ecosystems. Such a VEC is often used to indicate major changes associated with habitat alteration, which constitutes one of the most important anthropogenic stressors affecting ecosystems (cf. USEPA SAB 1990a, 1990b; Harwell et al. 1992). As with trophic structure and biogeochemical processes, changes to the landscape mosaic VEC would constitute fundamental changes to the overall ecosystem and thus are highly important for assessing major ecosystem changes and recovery.

The direct oiling effects of EVOS had no documented impact on the habitat mosaic in PWS, but there was ecologically significant habitat alteration for those areas of intertidal and shoreline communities experiencing intensive cleaning, as discussed previously (cf. Lees et al. 1996; Mearns 1996; van Tamelen et al. 1997; Highsmith et al. 2000). Similarly, significant areas of subtidal habitat were affected by EVOS or the cleanup, including effects on the subtidal biota from chemical toxicity, increased turbidity, sedimentation, and other physical disturbances (Jewett and Dean 1997); however, those authors concluded that most components of the seagrass community had recovered by 1995. The EVOS did not cause other documented habitat or landscape changes.

The PWS, in general, has had minimal habitat alteration as compared with most other coastal ecosystems of the United States. Potential sources of habitat alteration include natural resource extraction (e.g., oil and gas exploration, timber harvesting, mining, and agriculture), road building, and urbanization (NWF 2003). Probably the greatest impact on the landscape of PWS during the past century was the earthquake of 1964, which involved vertical displacement over an area of about 520,000 km^2 and the major area of uplift extending from southern Kodiak Island to PWS (State of Alaska 1964; Stanley 1968; USGS 2004). Vertical displacements ranged from about 11.5 m of uplift to 2.3 m of subsidence relative to sea level, and off the southwest end of Montague Island the absolute vertical displacement was 13 to 15 m. Such displacements significantly affected many habitats along the coastline, such as salmon streams, intertidal and subtidal areas, and other important habitats. Stanley (1968) reported that in PWS virtually all beaches were stranded out of reach of the sea, new beaches developed to fit the new sea levels, and there was substantial relocation of submarine sandbars by seismic waves. Stanley (1968) also reported extensive destruction or relocation of coastal habitats for seabirds, shellfish, and salmon in PWS. The spatial extent and intensity of habitat changes from the 1964 earthquake greatly exceeded any effects from EVOS, thus again providing an important context to evaluate long-term changes to PWS from EVOS.

The 1 other habitat-related consequence of EVOS was actually an important positive outcome resulting from the extensive protection of habitat by the Trustees using funds from the Exxon settlement. As of March 2002, the Trustees reported more than 260,000 ha (640,000 acres) of habitat had been protected through either acquisition of land or interests in land, including more than 2,250 km (1,400 miles) of coastline and over 300 streams for salmon spawning (Trustees 2002), using funds provided through the EVOS settlement agreement. Thus, because of this extensive habitat protection activity by the Trustees, the net effect of EVOS on the landscape mosaic of habitats has actually been beneficial for the long-term sustainability of the regional ecosystem. Note that we are certainly not suggesting that EVOS has had a net positive benefit; rather, we are pointing out that for this 1 attribute, the indirect consequences of EVOS may have guarded against adverse consequences of subsequent habitat alteration in PWS that was plausible to have occurred from other sources.

Ecological significance—The EVOS caused direct adverse impacts on inter- and subtidal habitats, but the overall mosaic of habitats in PWS was unaffected. No ecologically significant effects on the habitat mosaic are evident at present (2006).

SYNTHESIS

The present ecological significance of EVOS to the ecosystems of PWS relates to both 1) the ecological significance of any continuing chemical exposures from EVOS residue (as all the nonchemical stressors from the oil spill have long since disappeared) and 2) the ecological significance of any present residual effects on the biota, habitats, or ecosystems of PWS from EVOS and related cleanup activities. The evaluation of ecological significance used here is based on a multiple-lines-of-evidence approach to evaluating changes in exposure and ecological effects to populations and ecosystem structure and function.

The following lines-of-evidence approach was used to assess whether residual EVO is a significant contributor of hydrocarbon exposure in PWS:

1. The intensity of the source, that is, the concentration and the composition of known toxic constituents (especially PAHs);
2. The spatial (linear and/or areal) extent of remnant EVO and human activity sources;
3. The magnitude (volume or mass) of residual EVO and other hydrocarbon sources;
4. The residual and persistent toxicity to intertidal prey species and their primary foragers; and
5. The bioavailability of residual EVO and other sources of PAHs based on bioaccumulation and biomarker data.

The sources of residual EVO last surveyed in 2001 were very limited in linear (<10 km) and areal (11 ha) extent, so that <0.2% of the originally oiled shoreline had residual sources; by 2006, an estimated 0.03% to 0.09% of the original oil spill volume remained in PWS (primarily in subsurface sediments). Current sediment concentrations are well below any regulatory toxicity thresholds, and any residual sediment toxicity is limited to isolated areas. The bioavailability of PAHs in mussels and a wide range of other intertidal species from residual EVO is quite low and is not statistically different from reference sites except for a few sites that collectively represent <0.1% of the PWS shoreline. The concentration and composition of PAHs at these sites have decreased over the years following the oil spill and continue to decrease at approximately 25% annually, leaving only the higher-molecular–weight compounds that are lower in absolute concentration and less bioavailable than the initially spilled oil. Additionally, the spatial extent and volume of residual EVO have also decreased markedly since the oil spill. As a result, we conclude that these residual sources no longer represent a major route of hydrocarbon contamination to the food chain. However, residual EVO is not the only current source of hydrocarbon contamination to PWS. A survey of historical industrial activity sites within western PWS suggests that as much as 20 ha of contaminated sediments could be attributed to HA sites, comparable to that estimated for EVO, in addition to a suite of other anthropogenic sources (e.g., commercial shipping) and natural sources (e.g., oil seeps).

Biomarker studies indicate very low CYP1A activity throughout PWS and eastern GOA, differences across species, and absence of correlation with levels of source contamination. This supports a conclusion of the presence of multiple sources of PAHs throughout the region. The fact that biomarker levels for PWS fish are not zero but are similar to exposure levels in fish collected in the eastern GOA indicates that regional (i.e., non-EVOS) hydrocarbon sources are likely involved. As noted previously, it is not possible to quantify the attributable risk of exposure of PAHs to each source in PWS. Nevertheless, we believe that the current areal extent, mass, and bioavailability of residual EVO are so small that it is very unlikely that there remains significant exposure to ecological receptors in PWS from EVOS-derived PAHs.

Prince William Sound is a large and complex system that is subject to natural and anthropogenic sources of hydrocarbons historically and currently. Our multiple-lines-of-evidence approach seeks to use accepted scientific principles to produce a plausible assessment of the risks from residual sources of hydrocarbons within a context of scale. The results from such an analysis suggest that the scale of residual EVO has been sufficiently reduced over the 17 y since the oil spill so that the small localized remnant sources no longer constitute an ecologically significant risk of hydrocarbon exposure to the biota of PWS.

In evaluating the ecological significance of effects from EVOS today, there are several issues to address:

1. Are there any demonstrable current effects on the valued ecosystem components of PWS?
2. Do those effects occur at the population or higher level of organization?
3. Can those effects be attributed to stressors caused by EVOS?
4. Have the VECs individually and the PWS ecosystem holistically recovered from EVOS?

It is important to recognize that there are many historical and existing anthropogenic stressors that have adversely affected the relatively pristine character of PWS and continue to do so. These include harvesting for subsistence and the commercial fur trade (e.g., sea otter and harbor seals), commercial and subsistence whaling, commercial and recreational fishing within PWS and throughout the Pacific Ocean, hunting, logging, mining, tourism (especially the rapidly expanding cruise ship industry in PWS), commercial shipping of petroleum and other products, global climate change and associated glacial recession, air pollution from regional and longer-range sources, radioisotopes from nuclear testing, habitat alteration for human development, and introduced exotic species including foxes and hatchery-reared pink salmon (cf. Agler et al. 1995; Anderson and Piatt 1999; Arendt et al. 2002; Wooley 2002; NWF 2003). Many ecologically significant natural stressors also affect PWS, especially the 1964 earthquake (which caused major changes in shoreline elevations), an ongoing outbreak of spruce bark beetle (which may affect PWS nutrient dynamics), natural oil seeps, and natural variability in climate and ocean currents on annual, decadal, and longer time scales (cf. Eslinger et al. 2001; Vaughan et al. 2001). As a consequence of these multiple and cumulative natural and anthropogenic stressors, PWS has significantly degraded over the past century and can be expected to continue to do so, especially as the human presence in PWS is increasing. This provides a context in which to evaluate the significance of any residual effects from EVOS on the PWS ecosystem.

Another important attribute of the PWS ecosystem is that it is adapted to high levels of physical disturbances. The intertidal zone, as 1 example, is subject to tremendous stressors each tidal cycle, superimposed on which are longer-term episodic disturbances, such as very cold or hot temperatures concomitant with very low tides. This characteristic, along with the tremendous energy of winter storms and seas, means that this ecosystem has relatively high resilience to many types of stressors, in contrast to, for example, a desert system, which may show effects from physical disturbances for decades. It also means that some idealized ecological situation of steady-state or equilibrium condition simply never exists for PWS, with or without people and whether or not EVOS had occurred. Again, this provides an important context to evaluate recovery of the PWS ecosystem from EVOS.

The 3rd critical characteristic is the high natural variability of many VECs in PWS, most notably the anadromous fish populations and their predators. The driving forces underlying that variability are related to regional weather, climate, and oceanographic variability (Mundy 2005). As an important example, consider the few-week critical period in which the strength and direction of winds in GOA have a tremendous effect on upwelling processes, which in turn bring nutrients into the photic zone, resulting in major phytoplankton blooms that support the food resources for the juvenile fish. These changes in food resources lead to major differences in the return of adult fish populations into PWS, of importance not only to the fish VECs themselves but also to the broader food base in PWS supported by the fish biomass (e.g., many birds, sea otters, resident orca, and so on; Klein 1999; Eslinger et al. 2001).

As another example among many, consider the correlations between the phenology of breeding of black-legged kittiwake (*Rissa tridactyla*) and common guillemot (*Uria aalge*) populations and the North Pacific Oscillation, reported by Frederiksen et al. (2004). The example of a specific weather-driven sequence of events is nested within natural climate variability for the Pacific Ocean, GOA, and PWS that occurs on time scales of a few years (e.g., ENSO processes; cf. Diaz and Markgraf 2000) to several decades (e.g., the Pacific Oscillation; cf. McGowan et al. 1998; Chavez et al. 2003) and longer natural cycles (cf. Minobe 1997). These nested processes in turn may be further affected by global climate change (cf. IPCC 2001a, 2001b). Again, these documented large-scale cascading effects on the fundamental structure and processes of PWS provide a context to evaluate any long-term effects from EVOS.

We have noted that other anthropogenic stressors have had tremendous effects on the VECs of PWS, such as human overexploitation of sea otters. In many cases there are interactions among natural and anthropogenic stressors, such as the combined effects of the releases of hatchery-raised pink salmon, the pink salmon fishery itself, and climate variability–driven changes in food resources for the salmon (cf. McGowan et al. 1998; Finney et al. 2000).

The problem is how to distinguish the signal of an ecological effect from the noise of natural variability and how to distinguish reductions in a population caused by EVOS from reductions caused by other stressors. That problem was fairly easy to solve in the immediate aftermath of EVOS, when the physical and toxicity stressors were so pervasive and intense, but became much more difficult to address later, when the physical stressors had completely disappeared and the toxic chemical exposures were reduced to very low levels. Some would suggest that continuing exposures to PAHs from remnant EVOS sources pose a risk to the PWS ecosystem (cf. Bodkin et al. 2002; Carls, Harris, et al. 2004; Carls, Rice, et al. 2004). We conclude that PAH exposures are far below levels that could cause population-level effects. Moreover, the spatial extent of those few remnant sites is such that even if all the residual locations of EVOS oil were toxic to, for example, Pacific herring eggs, the sources for Pacific herring eggs that support the PWS population are so vastly larger than the 11 ha of remnant EVO sources that complete mortality there would literally be trivial to the PWS Pacific herring population. In essence, we conclude that today the EVOS-caused stressors are either completely gone from PWS or indistinguishable from background levels in the PWS system. Consequently, we assert that the issue is no longer one of ongoing stress-induced effects but of system resilience and recovery.

Wiens (1995) discussed some of the issues in determinations of a return to a normal prestressor condition (known as homeostasis) or a return to a trajectory of ongoing change (termed homeorhesis; see also Harwell et al. 1981; Levin et al. 1989; Kelly and Harwell 1990; Wiens et al. 2001). Skalski et al. (2001) argued that recovery occurs when an impacted population or community tracks or "parallels" the profiles of a control (nonimpacted) population or community and that erroneous conclusions can be reached by focusing only on absolute population numbers rather than the relative patterns of temporal trends. Similarly, Parker and Maki (2003) argued that expecting a poststress return to a steady-state or equilibrium condition is not ecologically realistic and can lead to erroneous conclusions about recovery. These theoretical and conceptual perspectives apply to the practical problem of assessing continuing effects from EVOS. Some would argue that so-far-unrealized indirect or cascading effects from EVOS could in the future plague PWS (such as suggested by Peterson et al. 2003). We quite disagree: the very real cascading effects on the PWS ecosystem over the past several decades from fundamental natural stressors, especially climate and oceanographic variability, and the pervasive anthropogenic stressor of overexploitation of marine resources are significant and would overwhelm any proposed EVOS signal.

In addition, there is often a confusion in the literature of responses of biomarkers (e.g., elevated CYP1A levels in sea otters or harlequin ducks) that actually are indicators of exposure to PAHs from whatever source, with ecologically significant effects on populations; Peterson et al. (2003) is only 1 example of this. Moreover, most individual VECs in PWS are affected by many different natural and anthropogenic stressors. For example, at the time of the oil spill, the Pacific population of sea otters was still recovering from the massive loss from 19th-century overexploitation. Additionally, as Wiens (1995) pointed out, spatial heterogeneity is an important factor. Thus, changes may be seen in 1 local subpopulation of sea otters but not seen elsewhere. Consider, for example, the Bodkin et al. (2002) finding of reduced population numbers only on northern Knight Island, whereas the PWS-wide population has significantly increased since the oil spill. In that case, we concluded that there are not ecologically significant effects at the total population level of PWS since we believe that is the appropriate scale for that particular VEC. Similarly, Wiens et al. (2001) suggested that canonical correspondence analyses conducted at the community level may be better indicators of ecological effects than a more common focus on individual species, especially where there are multiple stressors, but it is still extremely difficult to distinguish single-agent effects on communities, such as the EVOS event, from all the other stress effects seen in ecosystems.

Another important confounding factor in distinguishing EVOS-caused ecological effects from other effects is the complex interactions among species in the upper components of the food web. One example, noted previously, is the interaction among potentially controlling factors affecting the Steller sea lion populations (NRC 2003), including, among others, changes in fisheries activities, food resource availability, and prey selection by predators. Similarly, natural and anthropogenic variability in fish populations may have dramatic effects on seabirds. For example, Suryan et al. (2002) reported that short-term fluctuations in prey fish availability are responsible for dramatic, within-season changes in the breeding conditions of black-legged kittiwakes. Thus, complex foraging conditions limit the reproductive success of a place-foraging seabird species relying on an ephemeral food source, and effects on 1 species (the prey fish species) readily lead to cascading effects on other populations in the ecosystem.

The Pew Commission (2003) recently reported on the present status of the world's oceans, highlighting the many adverse effects from anthropogenic stressors, particularly including coastal development, causing loss of estuaries and wetlands; nutrient enrichment, primarily from surface runoff; overfishing, with an estimated 30% of assessed species being overfished and significant incidental catch effects on sea turtles, marine mammals, seabirds, and noncommercial fish species; invasive species, including excessive numbers of hatchery-raised salmon; and global climate change and climate variability, with associated changes in freshwater inputs, ocean circulation patterns, and sea-surface temperatures as well as changes in rates of sea-level rise. These issues have also been identified in several other studies as dominating the adverse effects of human activities on the global environment, often with particular consequences to the coastal and marine ecosystems. For example, the USEPA and its Science Advisory Board (SAB) have identified habitat alteration, climate change, overexploitation of living resources, and invasive species as the most important environmental stressors affecting the nation's environment (cf. USEPA 1987a, 1987b; USEPA SAB 1990a, 1990b; Harwell et al. 1992; see also Jackson et al. 2001 for discussion of overfishing as a major stressor on marine ecosystems). This is certainly not to say that global trends in coastal ecosystems excuse the very real and ecologically significant effects that occurred in PWS from EVOS, but it does show that since PWS is not an isolated, pristine ecosystem, it too is subject to many adverse stressors and that the residual effects from a transient EVOS can be considered only in the context of these continuing stressors and their continuing effects on the PWS ecosystem.

It is undeniable that over the past century there have been ecologically significant effects on coastal ecosystems from many different human activities, and many species and habitats have significantly degraded and become unsustainable. Unfortunately, this increasingly continues to be the case (Pew Commission 2003), and PWS is no exception. It is also undeniable that the ecological consequences on PWS from EVOS were extensive, intensive, and ecologically significant. The present issue is whether any remaining, current (2006) effects on PWS and associated ecosystems from EVOS are ecologically significant. Moreover, the latter question must also be answered in the context of the general degradation of the coastal environment of PWS and GOA from multiple stressors of natural and anthropogenic origin.

SUMMARY

To assess the ecological significance of any residual effects from EVOS, we examined the literature on more than 20 VECs, including primary producers, filter feeders, fish and bird primary consumers, fish and bird top predators, a bird scavenger, mammalian primary consumers and top predators, biotic communities, ecosystem-level properties of trophodynamics and biogeochemical processes, and landscape-level properties of habitat mosaic and wilderness quality. None of these has any ecologically significant effects that are detectable at present and attributable to EVOS, taking into consideration natural variability, limitations in prespill data, and effects from other stressors on each individual VEC. One exception exists for 1 pod of orcas; however, we conclude that the PWS-wide population of resident killer whales appears to have fully recovered.

We conclude that the 1989 EVOS and the associated cleanup activities clearly caused significant ecological effects in PWS at that time and for months to a few years thereafter, affecting many PWS VECs. However, we conclude that, at present, both natural variability and the occurrence of multiple anthropogenic stressors not associated with EVOS overwhelm any potential residual ecological effects of EVOS. The physical stressors from EVOS are completely gone. The chemical stressors from EVOS are essentially gone, with the few remnant EVOS source areas having comparable magnitude to other human activity sites and with the chemical releases from those remnant sites incapable of affecting the PWS ecosystem in an ecologically significant way.

We conclude that the only current ecologically significant residual effect from EVOS appears to be for 1 pod of orcas but not for the PWS population as a whole. We believe that this continuing effect does not derive from continuing direct or indirect toxic chemical exposures but rather appears to relate to the long-term population dynamics of orcas. In particular, the altered social structure caused by the loss of key matriarchs, which we believe was partly a result of EVOS effects and partly a result from preceding mortality from human conflicts over fish, continues to plague the AB pod. We also conclude that all the other attributes listed by the Trustees as not having recovered, as well as all the other VECs used in our assessment, either have attained recovered conditions based on ecological significance criteria or are responding to other natural and/or anthropogenic stressors that have nothing to do with EVOS or its cleanup activities.

Therefore, we conclude—on the basis of the extensive literature that we reviewed on the suite of ecosystem assessment VECs selected for our analyses and taking into consideration the limitations imposed both by the available data and by the often large natural spatial and temporal variability—that the Prince William Sound ecosystem has effectively recovered from EVOS.

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Disclaimer—The authors wish to acknowledge the financial support provided by ExxonMobil for the time needed to prepare this article; however, the opinions and conclusions expressed herein are strictly those of the authors and do not necessarily represent the opinions of ExxonMobil. We undertook this synthesis review without previous experience in PWS or with EVOS and with the understanding that regardless of the conclusions, this work would be submitted for publication. If anything, we began with a bias toward expecting to find evidence of continuing ecological effects on PWS based on our cursory reading of the popular science and public literature. We also felt, perhaps, an inherent bias, derived from our collective 60 y of experience working in or for the government, toward government-funded science rather than industry-funded science. However, after several years aboard research vessels on PWS, reading virtually all the available literature on EVOS, and applying the objective criteria we previously proposed for assessing ecological significance, we are confident that our conclusions reflect not our bias but rather our professional judgment based on the totality of the available information. Others may well disagree, but we encourage examination of the full literature, as we have done, before reaching conclusions.

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