U.S. DEPARTMENT OF
HOMELAND SECURITY
United States Coast Guard

Biological Assessment for the Northwest Area Contingency Plan for the Response to Spills of Oil and Hazardous Substances

Appendices A through E

Prepared for:

United States Environmental Protection Agency
Region 10

and

United States Coast Guard
Thirteenth Coast Guard District

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Appendix A. Available Supplemental Response Information

This appendix is intended to provide a non-exhaustive list (Table A-1) of documents and tools that may be used during a spill response in the northwest area. The documents and tools in Table A-1 are provided here as a resource and reference for those interested in obtaining additional technical information about spill response planning and implementation. These documents and tools are external to the Northwest Area Contingency Plan (NWACP) and are, therefore, outside the scope of the Biological Assessment of that plan. Not all information available in these documents will be accessed during a spill response, nor will all guidance in these documents be relevant or appropriate within the legal or environmental context of a spill response in the northwest and an associated implementation of the NWACP.

Table A-1 Supplemental Documents and Tools That May be Utilized During a Response Action

Document Name	Agency/ Author	Brief Synopsis
Geographic Response Plans	Ecology, EPA, ODEQ	Contains local-scale descriptions of spill response strategies, natural resources, and logistics information.
Alaska Clean Seas Technical Manual Vol. 1 and 2	Alaska Clean Seas	The manual contains descriptions of spill response tactics and equipment.
Characteristics of Response Strategies: a Guide for Spill Response Planning in Marine Environments	NOAA	This document summarizes the technical rational for selecting response methods. The guide discusses developing incident specific strategies and describes the characteristics of individual response methods.
Environmental Considerations for Marine Oil Spill Response	API, NOAA, USCG, EPA	A tool for contingency planners and field responders to identify response techniques that have minimal ecological impacts and also minimize the impact of the oil. Guidance is provided through matrix tables indicating the relative environmental consequences of the different response options used for various categories of oil in open water and shoreline habitats.
Understanding Oil Spills and Oil Spill Response: Understanding Oil Spills in Freshwater Environments	EPA	This publication provides information about oil spills, potential effects to the environment, clean up methods, and how various agencies deal with oil spills.
Oil Spill Response Field Manual	ExxonMobil	This field manual outlines and describes in-depth proper procedures, clean up measures, and best management practices for handling oil spills.
FWS National Contingency Plan: procedures for removal and response [online]	USFWS	Creates a national consistency for responding to oil spills. Sets procedures so responses are implemented to best protect fish and wildlife resources and their habitats in the event of an oil and hazardous substance release.
Pacific Northwest Environmental Response Management Application (ERMA)	NOAA	Spatial data and mapping tool to inform a response action in the Pacific Northwest. Includes information on weather, bathymetry, natural resources, water quality, and more.
Oil Spill Response in Fast Currents: a Field Guide	USCG	This field manual describes proper procedures, clean up measures, and best management practices for handling oil spills in flowing waters.
Evaluation of New Approaches to the Containment and Recovery of Oil in Fast Water	USDOT and USCG	Report on the abilities and limitations of different spill response equipment and methods when applied in flowing waters.

Table A-1 Supplemental Documents and Tools That May be Utilized During a Response Action

Document Name	Agency/ Author	Brief Synopsis
National Oil and Hazardous Substances Pollution Contingency Plan (NCP) and NCP Product Schedule	EPA	National plan outlining general spill response framework. Product schedule describes specifically which chemical response tools (e.g., Corexit EC9500A dispersant) are permitted at the federal level during a spill response action.
Western Response Resources List	Region 10 RRT and NW Area Committee	Inventory of available spill response equipment for use in the northwest, including locations and amounts.
Canada-United States Joint Inland Pollution Contingency Plan, CANUSWEST – South Annex	EPA, Environment Canada	Plan outlining cooperative spill response activities between the US and Canada in the event of an international spill to inland waters.
Canadian Coast Guard-United States Coast Guard Joint Marine Pollution Contingency Plan, CANUSPAC Annex	USCG and Canadian Coast Guard	Plan outlining cooperative spill response activities between the US and Canada in the event of an international spill to coastal waters.
Technical Information Papers (Nos. 1 through 17)	ITOPF	Reports providing guidance on a range of response topics including but not limited to aerial reconnaissance; oil fate and effects; use of booms, dispersants, skimmers, and sorbent materials; shoreline monitoring and cleanup; and oil sampling protocol.
Technical Reports and Good Practice Guidance documents (multiple documents)	IPIECA	The IPIECA has generated several technical reports and guidance documents on topics including but not limited to spill monitoring, equipment selection and use, recovery of oil and decanting, shoreline cleanup, inland responses, waste management and minimization, and wildlife response preparedness.
General NOAA Operational Modeling Environment (GNOME) and Automated Data Inquiry for Oil Spills (ADIOS)	NOAA	Computer modeling tools used to predict oil spill trajectory and fate.
In-situ Burning: A Decision-maker's Guide to In-situ Burning	API	Regulatory Analysis and Scientific Affairs Publication No. 4740, April 2005; focuses on operational *in situ* burn considerations to help decision making both on land and on water
Field Operations Guide for In-situ Burning of Inland Oil Spills	API	API Technical Report 1251, July 2015; contains a set of operational checklists, tools, and references to assist in the conduct of inland in-situ burning of spilled oil.

ADIOS – Automated Data Inquiry for Oil Spills
API – American Petroleum Institute
Ecology – Washington State Department of Ecology
EPA – US Environmental Protection Agency
ERMA – Environmental Response Management Application
FWS – Fish and Wildlife Service
GNOME – General NOAA Operational Modeling Environment
IPIECA – International Petroleum Industry Environmental Conservation Association (
ITOPF – International Tanker Owners Pollution Federation
NCP – National Contingency Plan
NOAA – National Oceanic and Atmospheric Administration
NW – Northwest
ODEQ – Oregon Department of Environmental Quality
RRT – regional response team
USCG – US Coast Guard

USDOT – US Department of Transportation
USFWS – US Fish and Wildlife Service

APPENDIX B. DISPERSANT AND DISPERSED OIL AQUATIC EXPOSURE AND TOXICITY EVALUATION

This appendix (Appendix B) to the *Biological Assessment for the Northwest Area Contingency Plan for the Response to Spills of Oil and Hazardous Substances* (hereafter referred to as the NWACP BA) provides a detailed review of the fate, transport, and toxicity of chemical dispersants. This review was developed previously on behalf of the US Environmental Protection Agency (EPA) and US Coast Guard (USCG) for the *Biological Assessment of the Alaska Federal/State Preparedness Plan for Response to Oil & Hazardous Substance Discharges/Releases (Unified Plan)* (hereafter referred to as the Alaska Unified Plan BA), which was finalized in 2014 (Windward and ERM 2014). The analysis presented in the Alaska Unified Plan BA was heavily utilized and expanded upon for the *Biological Assessment of Dispersant Preauthorization Under Section I of the California Dispersant Plan, Appendix XII of the Regional Response Team IX Regional Contingency Plan* (hereafter referred to as the California Dispersant Plan BA and incorporated by reference) (EPA and USCG 2015). Both of the documents include region-specific analyses of the potential effects of chemical dispersant use on Endangered Species Act (ESA)-listed species and their critical habitats. The analysis of effects from exposure to chemical dispersants and dispersed oil that is presented in the main text of the NWACP BA relies on the analyses presented in both of these documents.

Though the effects analyses contained in this appendix from the Alaska Unified Plan BA (Windward and ERM 2014) and California Dispersant Plan BA (EPA and USCG 2015) are focused on the regions for which they were developed, there is substantial information contained within those BAs that pertains to the Pacific Northwest (NW) and the NWACP. For example, chemical fate, transport, and toxicity are all expected to be similar between Alaska, California, and the NW, with differences depending on local conditions (e.g., weather and sea state, water temperature and salinity, and presence of sea ice [specific to Alaska]) and local species (NRC 2005).

This appendix from the Alaska Unified Plan BA (Windward and ERM 2014) contains uncertainty analyses related to responses in Arctic conditions that are not directly comparable to conditions in the NW (where sea ice cannot influence a dispersant application). However, analyses in this appendix that focus on “cold-water” species are expected to be relevant to the NW (excepting data specific to Arctic species), given that water temperatures in the NW can be similar to those in Alaska. This appendix also contains substantial text and analysis describing the chemical dispersant Corexit® 9527, which was used during the Deepwater Horizon event but is no longer stockpiled for response actions; analyses of that specific dispersant chemical are now virtually irrelevant, whereas analyses of Corexit® EC9500A (referred to in this appendix as Corexit® 9500) remain relevant. Corexit® EC9500A is the only chemical dispersant that is stockpiled in the NW (as of 2017). Several of the ESA-listed species analyzed in this appendix are also present in the NW (e.g., several whales and short-tailed albatross).

The California Dispersant Use Plan BA (EPA and USCG 2015) provides analyses for several of the ESA-listed species that are included in the NWACP BA (e.g., marbled murrelet and western snowy plover) as well as additional information on the fate, transport and toxicity of dispersants. The California Dispersant Use Plan BA also contains some information that is not relevant to the NWACP BA (e.g., information specific to California or ESA-listed species that not present in the NW).

References

EPA, USCG. 2015. Biological assessment of dispersant preauthorization under Section I of the California Dispersant Plan, Appendix XII of the Regional Response Team IX Regional Contingency Plan. US Environmental Protection Agency and US Coast Guard.

NRC. 2005. Oil spill dispersants: efficacy and effects. Committee on Understanding Oil Spill Dispersants, Efficacy, and Effects, National Research Council. National Research Council of the National Academies. National Academies Press, Washington, DC.

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U.S. DEPARTMENT OF
HOMELAND SECURITY

United States Coast Guard

DISPERSANT AND DISPERSED OIL AQUATIC EXPOSURE AND TOXICITY EVALUATION

FINAL

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Table of Contents

Attachment B-1. Toxicity Data

Tables

Figures

Acronyms

ANS	Alaska North Slope
ARRT	Alaska Regional Response Team
BA	biological assessment
BO	biological opinion
BMP	best management practice
CAS	Chemical Abstracts Service
CDC	Centers for Disease Control and Prevention
DHOS	Deepwater Horizon oil spill
DOSS	dioctyl sulfosuccinate sodium
DPnB	1-(2-butoxy-1-methylethoxy)-2-propanol
DPS	distinct population segment
EC50	concentration that has an effect on 50% of an exposed sample
EPA	United States Environmental Protection Agency
EROD	ethoxyresorufin-O-deethylase
ESA	Endangered Species Act
ESU	evolutionarily significant unit
EVOS	*Exxon-Valdez* oil spill
GNOME	General NOAA Operational Modeling Environment
GOA	Gulf of Alaska
HC	hazardous concentration (for a given proportion or percentile of a species sensitivity distribution)
HPAH	high-molecular-weight polycyclic aromatic hydrocarbon
IQR	interquartile range
LC50	concentration that is lethal to 50% of an exposed sample
LPAH	low- molecular-weight polycyclic aromatic hydrocarbon
NOAA	National Oceanic and Atmospheric Administration
NOEC	no-observed-effect concentration
NPRW	North Pacific right whale
OECD	Organisation for Economic Cooperation and Development
PAH	polycyclic aromatic hydrocarbon
ppb	parts per billion

ppm	parts per million
PWS	Prince William Sound
SSD	species sensitivity distribution
TPH	total petroleum hydrocarbons
USCG	United States Coast Guard
Y-K Delta	Yukon-Kuskokwim Delta

1 Introduction

1.1 PURPOSE AND THE BASELINE CONDITION

This document is a Appendix B to the *Biological Assessment of the Alaska Federal/State Preparedness Plan for Response to Oil & Hazardous Substance Discharges/Releases (Unified Plan)*, hereafter referred to as the BA. The purpose of this appendix is to describe the known or potential adverse impacts of chemical dispersants, alone or in a mixture with oil, both directly on species listed under the Endangered Species Act (ESA) (or similar surrogates) and indirectly on their prey. These impacts must be weighed against the baseline condition: that petroleum has been spilled, and that a response can be taken in accordance with the Unified Plan. Such a response may involve the application of chemical dispersants under certain circumstances, which are elaborated upon in the BA.

In order for adverse impacts related to chemical dispersants to be considered relevant to this BA, dispersants must be shown to meet one or more of the following qualifications:

- Be inherently more toxic than oil (i.e., causing toxicity when alone in solution).
- Increase the exposure concentration and/or duration of exposure to oil of ESA-listed or candidate species or their prey to oil or its component chemicals (e.g., polycyclic aromatic hydrocarbons [PAHs]).
- Increase the toxicity of petroleum or its component chemicals to ESA-listed or candidate species or their prey (Milinkovitch et al., 2011a; Ramachandran et al., 2004; Wolfe et al., 1998; Wolfe et al., 2001; Yamada et al., 2003).

If the application of dispersants to an oil spill can be shown to mitigate the known impacts of a non-dispersed oil spill (i.e., the baseline condition), then the impacts of dispersants as a potential response tool can be considered negligible (or even beneficial by comparison) (Fingas, 2008; NRC, 2005).

The synthesis of available data regarding the known impacts on ESA-listed or candidate species and their prey, toxicity in laboratory testing, and fate and transport testing is weighed with species-specific information (i.e., life history, seasonal use of Alaska waters, feeding strategies, and habitat associations) in the final determination of direct and/or indirect adverse effects on individual ESA-listed or candidate species. This synthesis is presented in Section 5 and summarized in Section 7.

1.2 SPECIES CONSIDERED

1.2.1 ESA-listed or candidate species

Table 1. Protected species status, habitats, and distribution

Protected Species	Status	Habitat Type in Potentially Affected Area	Critical Habitat?	Geographic Location
Marine Mammals				
Beluga whale (*Delphinapterus leucas)* – Cook Inlet DPS	E	nearshore, open water (including polynyas)	yes	Cook Inlet
Blue whale (*Balaenoptera musculus)*	E	open water	no	Aleutian Islands, Bering Sea, GOA
Bowhead whale (*Balaena mysticetus)*	E	open water, ice edge	no	Bering Sea, Beaufort Sea, Chukchi Sea
Fin whale (*Balaenoptera physalus)*	E	open water	no	Bering Sea, Beaufort Sea, Chukchi Sea, GOA, Aleutian Islands
Gray whale (*Eschrichtius robustus*) – Western North Pacific stock	E	nearshore, open water	no	Okhotsk Sea, Sakhalin Island, Russia, South China Sea (Potentially: Bering and Chukchi Seas, Aleutian Islands, GOA)
Humpback whale (*Megaptera novaeangliae)*	E	open water, nearshore	no	Bering Sea, Aleutian Islands, Kodiak Island, PWS, GOA including Inside Passage, Chukchi Sea, western Beaufort Sea
North Pacific right whale (*Eubalaena japonica)*	E	open water	yes	Bering Sea, Aleutian Islands, GOA
Sei whale (*Balaenoptera borealis)*	E	open water	no	Bering Sea, Aleutian Islands, GOA
Sperm whale (*Physeter macrocephalus)*	E	open water, ice edge	no	Bering Sea, Aleutian Islands, GOA
Steller sea lion (*Eumetopias jubatus)* – *western population*	E	shoreline, nearshore, open water	yes	Bering Sea, PWS, Kodiak Island, Aleutian Islands, GOA
Steller sea lion (*E. jubatus*) – *eastern population*[a]	T	shoreline, nearshore, open water	yes	GOA, southeast Alaska
Polar bear (*Ursus maritimus*)	T	terrestrial, shoreline, nearshore, ice	no[b]	Bering Sea, Beaufort Sea, Chukchi Sea, North Slope, western Alaska
Northern sea otter (*Enhydra lutris kenyoni*) – southwest Alaska DPS	T	shoreline, nearshore	yes	Aleutian Islands, Bristol Bay, Alaska Peninsula, Kodiak Island, Pribilof Islands
Pacific walrus (*Odobenus rosmarus,* ssp. *divergens*)	C[d]	shoreline, nearshore, open water, ice	no	Chukchi Sea, Bering Sea, Bristol Bay

Protected Species	Status	Habitat Type in Potentially Affected Area	Critical Habitat?	Geographic Location
Ringed seal (*Phoca hispida)*	T	nearshore, open water, ice	no	Chukchi Sea, Beaufort Sea
Bearded seal (*Erignathus barbatus)*	T	nearshore, open water, ice	no	Chukchi Sea, Beaufort Sea, Bering Sea
Birds				
Eskimo curlew (*Numenius borealis*)	E	terrestrial (tundra)	no	Arctic, although likely extinct
Short-tailed albatross (*Phoebastria albatrus*)	E	open water	no	Aleutian Islands, Bering Sea, GOA
Spectacled eider (*Somateria fischeri*)	T	shoreline, tidal marsh/delta, nearshore, open water, ice	yes	Beaufort Sea, Bering Sea, Arctic coastal plain, Y-K Delta
Steller's eider (*Polysticta stelleri*) – Alaska breeding population	T	tidal marsh/delta, nearshore, open water	yes	Bering Sea, Alaska Peninsula, Aleutian Islands, Kodiak Island, Cook Inlet, Arctic coastal plain, Y-K Delta
Kittlitz's murrelet (*Brachyramphus brevirostris*)	NL[c]	shoreline, nearshore, open water	no	Alaska Peninsula, Aleutian Island, Glacier Bay, Kenai Peninsula, Kodiak Island, Point Lay, PWS, Seward Peninsula, Yakutat Bay
Yellow-billed loon (*Gavia adamsii*)	C[d]	riverine/riparian, lake/wetland/bog, nearshore, open water	no	Aleutian Islands, Kodiak Island, Seward Peninsula, southeast Alaska, St. Lawrence Island, Arctic coastal plain
Fish				
Chinook salmon (*Oncorhynchus tshawytscha*) – Lower Columbia River ESU	T	open water, nearshore	no	GOA
Chinook salmon (*O. tshawytscha*) – Upper Columbia River, spring run ESU	E	open water, nearshore	no	GOA
Chinook salmon (*O. tshawytscha*) – Puget Sound ESU	T	open water, nearshore	no	GOA
Chinook salmon (*O. tshawytscha*) – Snake River, fall run ESU	T	open water, nearshore	no	GOA
Chinook salmon (*O. tshawytscha*) – Snake River, spring/summer run ESU	T	open water, nearshore	no	GOA, Bering Sea
Chinook salmon (*O. tshawytscha*) – Upper Willamette River ESU	T	open water, nearshore	no	GOA, Bering Sea
Coho salmon (*Oncorhynchus kisutch*) – Lower Columbia River ESU	T	open water, nearshore	no	GOA, Aleutian Islands, Bering Sea (north to Point Hope), Southeast Alaska
Steelhead trout (*Oncorhynchus mykiss*) – Lower Columbia River DPS	T	open water, nearshore	no	GOA, Aleutian Islands

Protected Species	Status	Habitat Type in Potentially Affected Area	Critical Habitat?	Geographic Location
Steelhead trout (*O. mykiss*) – Middle Columbia River DPS	T	open water, nearshore	no	GOA, Aleutian Islands
Steelhead trout (*O. mykiss*) – Snake River basin DPS	T	open water, nearshore	no	GOA, Aleutian Islands
Steelhead trout (*O. mykiss*) – Upper Columbia River DPS	T	open water, nearshore	no	GOA, Aleutian Islands
Pacific herring (*Clupea pallasi*) -- Southeast Alaska DPS	C	open water, nearshore	no	GOA, Aleutian Islands, Bering Sea, Southeast Alaska
Reptiles				
Leatherback sea turtle (*Dermochelys coriacea*)	E	open water	no[e]	GOA
Loggerhead turtle (*Caretta caretta*)	E	open water	no[e]	GOA
Green turtle (*Chelonia mydas*)	T	open water	no	GOA
Olive Ridley turtle (*Lepidochelys olivacea*)	T	open water	no	GOA
Plants				
Aleutian shield fern (*Polystichum aleuticum*)	E	terrestrial	no	Adak Island

a The eastern population of Steller sea lion is currently proposed for delisting (NMFS, 2012).

b On 10 January 2013, the US District Court for the District of Alaska issued an order vacating the rule designating critical habitat for the polar bear (US District Court District of Alaska, 2013). Therefore, at this time, there is no critical habitat designated for the polar bear.

c The Kittlitz's murrelet was designated as a candidate species during the preparation of the BA. On 3 October 2013, USFWS issued a determination finding that listing the Kittlitz's murrelet is not currently warranted (78 FR 61764, 2013). This listing determination was published during finalization of the BA. Therefore, the Kittlitz's murrelet has been included in the BA but an effects determination has not been made because listing under ESA is not imminent.

d The Pacific walrus and yellow-billed loon have been designated as candidate species. A 12 July 2011 court settlement agreement established that USFWS would either submit a proposed rule to list the species, or issue a not-warranted finding. The dates of submittal established in the settlement agreement are October 2014 for the yellow-billed loon and October 2017 for the Pacific walrus (US District Court for the District of Columbia, 2011).

e Critical habitat has been designated for leatherback sea turtles (77 FR 4170, 2012) and proposed for loggerhead turtles (78 FR 43006, 2013) outside of Alaska.

BA – biological assessment
C – candidate
DPS – distinct population segment
E – endangered
ESA – Endangered Species Act
ESU – evolutionarily significant unit
GOA – Gulf of Alaska
NL – not listed
T – threatened
USFWS – US Fish and Wildlife Service

Chemical dispersants are not intended for terrestrial application. Therefore, terrestrial species protected by the ESA (i.e., Aleutian shield fern [*Polystichum aleuticum*] and Eskimo curlew [*Numenius borealis*]) are not described in this appendix. It is assumed that the probability of exposure of these species to dispersants or dispersed oil is very small. This is particularly true of Aleutian shield fern, which is found in only one area, removed from the marine environment. Eskimo curlew, if still in existence,[1] could conceivably come into contact with oil spill responders in the terrestrial environment. This scenario is outside the scope of this discussion, because upland oil spill responses will not consider the use of chemical dispersants as a response tool (Section 1.3).

ESA-listed or candidate species for which multiple distinct population segments (DPS) or evolutionarily significant units (ESUs) are recognized by ESA will be considered as a single species in this appendix. It is not expected that impacts will differ greatly between either, nor is sufficient information available to determine whether one DPS or ESU is more susceptible to exposure than another. DPS and ESU information is important for identifying stock information (e.g., population size) and information about spawning locations and timing, none of which directly relate to chemical exposures that occur in Alaska. For example, ESA-listed species of salmon that are found in Alaska do not spawn in Alaska waters.

1.2.2 Non-ESA-listed or candidate species

Those ecological receptors at greatest risk of exposure to dispersants and dispersed oil include plankton, embryonic or larval forms of fish, and embryonic, larval, and adult forms of invertebrates that reside in the upper water column (Rico-Martinez et al., 2013; Ortmann et al., 2012). This risk is due to the relative immobility of these species relative to ocean currents; they are carried with currents and are not expected to be able to move away from the area of a spill response. Many larger species of fish and invertebrates (e.g., squid, octopus, herring) gain mobility as they mature, and others (e.g., crab, bivalves, echinoderms, worms) settle to the ocean floor. These species generally represent the prey of the ESA-listed or candidate mammals, birds, fish, and some reptiles evaluated in this BA. Data specific to protected species are assessed in Section 3.2. Impacts on non-ESA-listed or candidate species can be considered indirect impacts on ESA-listed species, if the non-listed or candidate species are prey items of listed species.

1.3 Description of Dispersants and Conceptual Model

Chemical dispersants are mixtures of surfactants and hydrocarbon-based solvents that alter the spatial distribution, physical transport, and chemical and biological fate of spilled oil in aquatic environments. The intended purpose of dispersant application is to reduce the concentration of oil at the surface of the ocean by breaking the oil slick into emulsified droplets that can be suspended and distributed (and subsequently diluted and biologically degraded) throughout the water column. The process of the chemical dispersion of oil is portrayed in Figure 1. Dispersant application is also a useful tool for reducing oil in shoreline habitats, when applied appropriately and in a timely manner (i.e., prior to migration of the slick into shallow waters, where oil

[1] Eskimo curlew have not been sighted for decades (since 1969) and are suspected to be extinct in the wild (USFWS, 2011a).

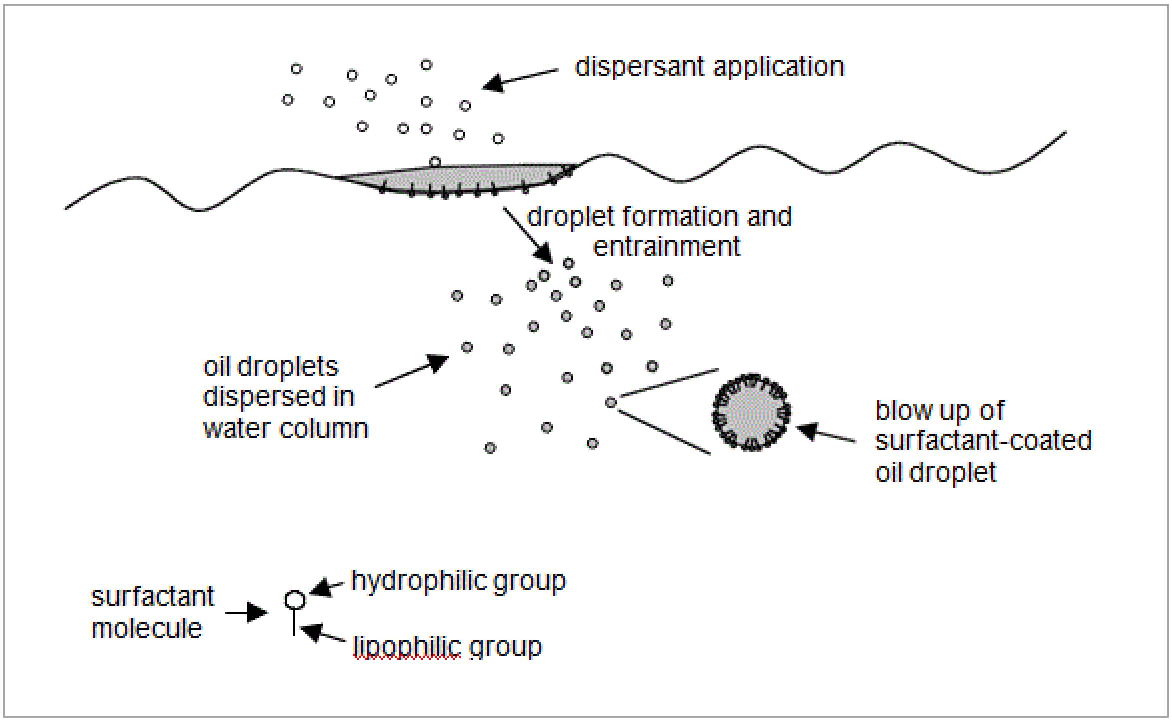
cannot be greatly diluted, and prior to significant weathering of the oil), and is expected to substantially reduce the known long-term impacts of shoreline oiling (Peterson et al., 2003; Cross and Thomson, 1987).

When released into the aquatic environment, crude oil tends to form a thin layer, < 1 mm thick on average (Lee et al., 2011a) and typically ~0.1 mm (NRC, 2005), that spreads over the surface of the water; after oil is spilled, a number of physical, chemical, and biological factors affect its dispersion and ultimate fate (NRC, 2005). Physical factors such as surface tension (a measure of attraction between the molecules of a liquid), density, and viscosity (a measure of resistance to flow) cause the oil molecules to generally stay together, if there are no other forces at work (NRC, 2005). A chemical dispersant can cause an oil slick to either spread rapidly and then disperse, or to spread slowly through "herding" (NRC, 2005), after which additional dispersant applications may be required to remove the oil slick from the ocean's surface.

In the event of a subsurface release, spreading is different; the presence of natural gas in crude oil makes it buoyant, driving it quickly to the surface as a uniform plume (NRC, 2005). The resulting surface slick may be similar to a surface release, particularly when the subsurface release is shallow (NRC, 2005). In the event of deep releases, such as the Deepwater Horizon oil spill (DHOS), density stratification and ambient currents can cause denser oil components to split from gaseous components (i.e., natural gas and methane), resulting in a much slower and less uniform ascent to the surface (NRC, 2005). The resultant surface slick is expected to be thinner and spread over a larger area (NRC, 2005). Thinner slicks are less affected by chemical dispersion (NRC, 2005), making the spill less likely to be contained and mechanically recovered. The application of chemical dispersant at the wellhead during DHOS may have been in response to such expectations. The application of chemical dispersants at the wellhead during DHOS represented an unprecedented use of this chemical countermeasure; such a response has never been conducted in Alaska, nor is it approved for use in Alaska. For that reason, deepwater response actions are not being assessed as part of this consultation.

Wind, waves, and other physical forces (such as the movement of sea ice) can either enhance dispersion or mix the oil and water, forming an emulsion that remains relatively cohesive and does not disperse easily (NRC, 2005; MMS, 2010; Brandvik et al., 2010). Over time, chemical processes (e.g., volatilization and oxidation) can change the makeup and density of oil, which affects, in turn, its fate in the environment (Mackay and McAuliffe, 1988). Biodegradation occurs over time, as fractions of the oil become bioavailable (i.e., dissolve in the water column) (Prince et al., 2013); however, oil thickness, cohesiveness, viscosity, and other factors affect bacterial access to oil molecules (Prince et al., 2003).

The concepts laid out in this section are further expanded in Section 2, and are incorporated in the conclusions regarding the likelihood of impacts on certain species in Sections 4 and 5.



Source: NRC (2005)

Figure 1. Mechanism of chemical dispersion

2 Fate and Transport of Dispersants and Dispersed Oil

This section expands upon the conceptual model (Section 1.3) of how dispersed oil behaves in an aquatic environment, and discusses the factors that affect the toxicity of dispersed oil under field conditions. Oil is assumed to be fresh or slightly weathered crude petroleum, the most likely material for which dispersants would be used (Alaska Clean Seas, 2010; Nuka Research, 2006; NOAA, 2012b; ARRT, 2013). Diesel fuel is the most common type of petroleum spilled in Alaska waters (See Appendix D to the BA), but it is very rarely, if ever, treated with chemical dispersants (Appendix D). The rapid rate at which refined fuels (such as diesel) naturally attenuate (i.e., volatilize, disperse, and degrade) makes dispersant application impractical for such spills.

Factors affecting oil dispersion and dilution are discussed in Section 2.1, dispersants and dispersed oil degradation is discussed in Section 2.2, and transport is discussed Section 2.3.

2.1 Dispersion and Dilution

Dispersion is a natural process that distributes petroleum at the ocean's surface into the water column over time, resulting in many small droplets that may or may not resurface and coalesce with the oil slick (NRC, 2005). This process can be very slow under natural conditions, but the addition of chemical dispersants greatly increases the rate of dispersion (NRC, 2005).

The application of dispersants in a typical spill response involves the release of a large tank of undiluted dispersant chemical (commonly referred to as a sortie) from deployed vehicles (e.g., airplanes, boats, or helicopters) onto the surface of a spill on open water (Nuka Research, 2006). The volume released depends largely on the vehicles' carrying capacities for liquid dispersants (Nuka Research, 2006); however, the rate of application (i.e., volume per unit area) is expected to be as consistent as possible over a large area (Nuka Research, 2006), resulting in a more or less uniform input of dispersant chemicals. Ideally, the dispersant droplets come into contact with the oil and mix rapidly, resulting in nearly instantaneous dispersion into the water column. Although dispersant is applied as evenly as possible, because oil slicks tend to be unevenly distributed across the ocean's surface (NRC, 2005), the true dispersant-to-oil ratio (DOR) is expected to vary spatially. The required volume of chemical dispersant is assumed to be that which is needed to coat the surface of an oil slick with minimal volume allowed for overspray (Scelfo and Tjeerdema, 1991) and to achieve a recommended DOR, typically between 1:10 and 1:50 (Rico-Martinez et al., 2013), and more specifically, 1:20 in Alaska (Alaska Clean Seas, 2010).

The goal of dispersant application is to break the surface tension of the water-oil interface such that droplets of oil form that are small enough to remain suspended in the water column (Brandvik et al., 2010). Dispersant chemical formulations are designed to bind to non-polar substrates and crude oil specifically, so the individual chemicals in dispersants tend to move through the water column with plumes of dispersed oil (Kujawinski et al., 2011).[2] Once broken into droplets, the oil mixes into the water column, effectively lowering the surface concentration of oil and thus the exposure of aquatic organisms at the ocean's surface. Note that pelagic species (e.g., fish) may be more exposed to oil after chemical dispersion, because typical concentrations of oil in the water column are very low prior to dispersion, even just below the slick (Mackay and McAuliffe, 1988). Also, the exposure of species to toxic components of oil (i.e., PAHs) is likely to increase immediately after dispersant application (Yamada et al., 2003; Ramachandran et al., 2004; Milinkovitch et al., 2011a), and may result in increased toxicity (Barron, 2003; Barron et al., 2008). PAHs are likely to decrease rapidly in concentration as a result of natural processes (e.g., wave action, wind-driven currents and advection, photo-oxidation, and biodegradation), though toxicity may still occur (French-McCay, 2010). These possible impacts are discussed at length in Section 3.

The rate of oil and chemical dispersant mixing is primarily determined by the energy of the environment into which the dispersant is applied, although some additional factors contribute to effective dispersion (e.g., spill size, dispersant droplet size, penetration of spill upon impact, thickness of spill, extent of weathering, and the formation of less dispersible emulsions) (NRC, 2005). A calm sea will mix more slowly than churning waters, where waves stir the oil and dispersant together. Wind also produces turbulent mixing, facilitating dispersion (NRC, 2005). Both wave action and wind energy act on any oil, regardless of the presence of dispersants, and cause the natural dispersion of oil droplets. In the Arctic, sea ice can dampen the effect of wind and waves, requiring the deliberate addition of turbulence (e.g., propeller wash from a response vessel) (Sørstrøm et al., 2010). However, the movement of the ice itself has been shown to sufficiently mix oil and dispersant, such that chemical dispersion is highly effective even in the presence of broken ice (Sørstrøm et al., 2010; Potter et al., 2012). It is also important to note that the effectiveness of dispersion at Arctic temperatures is not dissimilar to its effectiveness in warmer waters (Potter et al., 2012; Sørstrøm et al., 2010; Brandvik et al., 2010; MMS, 2010). Still, under certain circumstances, it is possible that dispersion will be less effective in areas covered by sea ice due to decreases in surface water salinity (Brandvik et al., 2010; Chandrasekar et al., 2006) or sheltering from sea energy (Sørstrøm et al., 2010).

The environment in which dispersants are applied is often much different than the system in which a controlled toxicology study is conducted. In an artificial test system with well-defined boundaries, oil is constrained even when dispersed, limiting dilution. In a large water body, such as an ocean or embayment, dispersed oil is less

[2] Therefore, free dispersant in the water column is unlikely in the presence of oil; overspray into unoiled water is an exception and would result in partitioning to water.

constrained. Typically, field applications are more effective in reducing surface oiling than are applications in laboratory tests, as shown by Nedwed and Coolbaugh (2008).

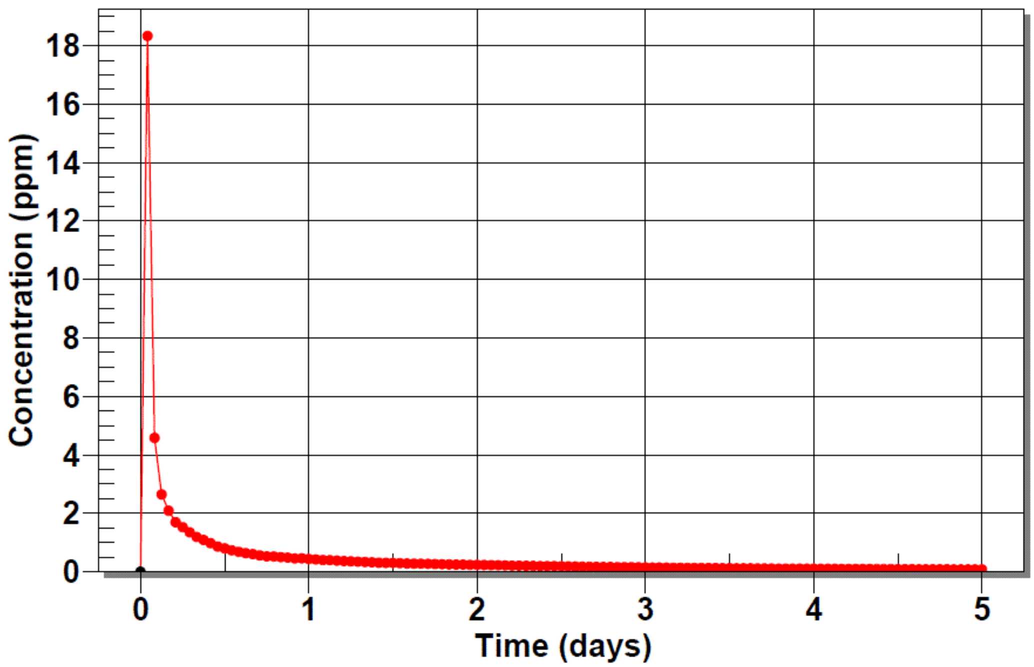
Gallaway et al. (2012) modeled the expected concentration of dispersant released to the environment assuming an application rate of 5 gal. of Corexit® 9500 per acre, a 10-km^2 area, and a total volume of 5,000 gal. of dispersant. The receiving waters were modeled as having a local initial value of approximately 18 parts per million (ppm) of Corexit® 9500, which was diluted rapidly over time (Figure 2). Within approximately one hour, the concentration of dispersant was diluted to below the 5th percentile of the species sensitivity distribution (SSD), the Hazardous Concentration-5 (HC5), calculated for this BA (i.e., 5.53 ppm Corexit® 9500) (Section 3.2; Table 3). The implication of this model is that the concentration of a dispersant is diluted rapidly after application to below protective concentrations (specific to dispersants alone); overspray is unlikely to result in significant acute toxicity to planktonic, embryonic, or larval species of fish or invertebrates, because the duration of exposure to toxic concentrations is very short, much shorter than in controlled toxicity experiments. The rate of dispersant dilution indicated by the Gallaway et al. (2012) model is similar to that reported by Nedwed (2012), who indicated that concentrations of dispersant decreased to < 1 ppm within a matter of hours (and to the parts per billion [ppb] range within 24 hours). Similar modeling conducted by the National Oceanic and Atmospheric Administration (NOAA) using the General NOAA Operational Modeling Environment (GNOME) provides similar results (NOAA, 2012b): dispersion is rapid, and dilution drives concentrations of dispersants to < 1 ppm within 24 hours.[3]

McAuliffe et al. (1980, 1981) and Mackay and McAuliffe (1988) showed that dispersed oil, although highly concentrated in the water column below an oil slick immediately after dispersion, decreased to below what the authors considered to be protective levels[4] within a matter of hours. Furthermore, the time-averaged concentration of dispersed oil was low (i.e., 0.46 ppm C_1-C_{10} hydrocarbons), even over short time periods immediately following the application of dispersant (i.e., between 10 and 30 minutes after application) (Mackay and McAuliffe, 1988). Although Mackay and McAuliffe (1988) measured only the light fraction of oil as it dispersed, it can be assumed that heavier fractions of oil (i.e., C_{11} and larger molecules) will disperse and dilute at the same rate (i.e., be transported within the same droplets of oil). That is not to say that dissolution and biodegradation of hydrocarbons into the water column

[3] GNOME model inputs used to derive dispersant concentration dilution models assumed idealized conditions for dispersion, such as 100% effectiveness (NOAA, 2012b).

[4] A direct comparison to the protective concentrations presented in Table 5 is not appropriate, because Mackay and McAuliffe (1988) reported the concentration of hydrocarbons as a light fraction, C_1-C_{10} hydrocarbons, rather than total petroleum hydrocarbons (TPH), a broader fraction of the possible hydrocarbons found in dispersed oil. The concentrations presented in Tables 4 and 5 are based on TPH, the broader fraction.

from oil droplets will be equivalent, as heavier organic molecules tend to be inherently less soluble and less biodegradable than lighter fractions even in the presence of chemical dispersants (Yamada et al., 2003).



Source: Gallaway et al. (2012)

Note: Concentration (ppm) refers to Corexit® 9500. The rapid decrease in Corexit® 9500 concentration is driven by dilution. Degradation occurs concurrently, but at a much slower rate.

Figure 2. Model of Corexit® 9500 concentration as a function of time after 5,000-gal. application over 10 km^2

In all cases, concentrations of dispersant or dispersed oil are shown to be diluted below their respective HC5s in less than the 48- to 96-hour exposure durations used in toxicity tests (Section 3). For this reason, it is expected that the chemical dispersion of oil will result in mitigated acute toxicity, even in relatively sensitive species, due to the reduction in exposure duration and concentration driven primarily by dilution. Mackay and McAuliffe (1988) stated the same conclusion. Furthermore, it is expected, based on previously published models of oil and dispersant dilution and the HC5s calculated in Section 3, that limited acute toxicity will occur in pelagic species, such as ESA-listed or candidate fish or prey species of ESA-listed wildlife. These findings are restated in Sections 4 and 5.

2.2 Degradation of Dispersants and Dispersed Oil

The purpose of this section is to describe the effect on the concentration of oil resulting from the biological and abiotic degradation of oil components or chemical dispersant components. Unlike dilution (Section 2.1), degradation results in the complete destruction of oil or chemical dispersants. Dilution is a rapid process that occurs immediately after chemical dispersion, but the rate and extent to which components of chemical dispersants and oil will degrade are dependent on various environmental factors, as well as the chemical itself.

Biological degradation, as discussed in Section 2.2.1, is strictly limited to microbial degradation, so the section does not relate to metabolism in larger organisms. Metabolism of oil components (e.g., PAHs) is discussed in Section 3.1.2; such metabolism has been linked to various toxic impacts (Shemer and Linden, 2007; Albers and Loughlin, 2003; Payne et al., 2003).

2.2.1 Biodegradation

Dispersants, once released into the environment, undergo physical and chemical processes much like spilled oil or other degradable substances. Neff (1988) indicated that as the volatile components of dispersants evaporate, physical processes initially control the rate of elimination of dispersants from a marine system.[5] After initial evaporation, biological processes determine the rate of removal from the environment.[6]

In a spiked laboratory exposure, Corexit® mixtures were reported to have a 107-minute half-life (i.e., time required for 50% degradation of chemical) in solution (George-Ares and Clark, 2000), indicating rapid removal from water under certain conditions. Mulkins-Phillips and Stewart (1974) also noted that dispersants are biodegradable, but that degradation occurred only after a microbial lag period in growth; this lag period is likely due to observed shifts in natural microbial communities in response to oil spills (Hazen et al., 2010; Lu et al., 2011; Baelum et al., 2012). A study by Okpokwasili and Odokuma (1990) observed that Corexit® 9527 biodegraded 90% or more within 16 days, and the half-life of the chemical mixture was approximately 2 to 3 days. Baelum et al. (2012) measured total Corexit® 9500 and the glycol and dioctyl sulfosuccinate sodium (DOSS) components individually in the presence of oil; the authors report rapid biodegradation of Corexit and DOSS within 5 to 20 days, but glycol components that were largely unaffected after 20 days. Mudge et al. (2011) specifically observed 1-(2-butoxy-1-methylethoxy)-2-propanol (DPnB), for which a half-life of approximately 30 days was determined.

Studies by Staples and Davis (2002), Kim and Weber (2005), the US Environmental Protection Agency (EPA) (2005, 2009, 2010), the Organisation for Economic Cooperation and Development (OECD) (1997), and West et al. (2007) indicate that the component chemicals of Corexit® 9500 and Corexit® 9527 are marginally or readily biodegradable (as well as abiotically degradable; see Section 2.2.2). Table 2 provides a

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[5] Refer to Table 2, which indicates that current Corexit® formulations contain only one potentially volatile component, petroleum distillates.

[6] Dilution is also a major factor in determining the concentration of dispersed oil in the water column, although such redistribution of oil does not, in itself, result in removal from the environment.

summary of biodegradation information for Corexit® component chemicals. The rates are given as either the half-life or percent degradation. Percent degradation is accompanied by the duration of the microbial exposure. The percent loss over time is used in determining biodegradability, such that a > 60% loss of a chemical within 28 days characterizes that chemical as readily biodegradable.

Table 2. Biodegradation information for Corexit® component chemicals

CAS No.	Chemical Name (Common Name)	Biodegradability	Half-Life (Days)	Concentration Loss (%, Duration)	Source(s)
57-55-6	1,2-propanediol (propylene glycol)	readily biodegradable	13.6	81%, 28 days	West et al. (2007); Dow AgroSciences (2012)
111-76-2	2-butoxyethanol[a]	readily biodegradable	nr	> 60%, 28 days	OECD (1997)
577-11-7	butanedioic acid, 2-sulfo-, 1,4-bis(2-ethylhexyl) ester, sodium salt (1:1) (DOSS)	readily biodegradable[b]	nr	66.4%, 28 days	EPA (2009)
		readily biodegradable	nr	91 to 97.7%, 3 to 17 days	TOXNET (2011)
1338-43-8	sorbitan, mono-(9Z)-9-octadecenoate (Span™ 80)	readily biodegradable	nr	58 to 62%, 14 to 28 days	EPA (2005, 2010)
9005-65-6	sorbitan, mono-(9Z)-9-octadecenoate, poly(oxy-1,2-ethanediyl) derivs. (Polysorbate 80)	not readily biodegradable	nr	52%, 28 days	Fisher Scientific (2010)
9005-70-3	sorbitan, tri-(9Z)-9-octadecenoate, poly(oxy-1,2-ethanediyl) derivs (Polysorbate 85)	readily biodegradable	nr	60 to 83%, 28 days[c]	EPA (2005)
29911-28-2	1-(2-butoxy-1-methylethoxy)-2-propanol (glycol ether DPnB)	readily biodegradable	10.3 – 28	> 60%, 28 days	Howard et al. (1991); Dow (1993, 1987); Staples and Davis (2002)
64742-47-8	petroleum distillates, hydro-treated, light[a]	readily biodegradable	nr	> 97%, 4.7 days	Rozkov et al. (1998)

[a] Potentially volatile component

[b] EPA states that DOSS did not biodegrade readily; however, the rate at which biodegradation occurred was greater than 60%, above the typical criterion for ready biodegradability. Therefore, it has been changed in the table to reflect the more widely accepted criterion.

[c] Value is expected based on the degradation of chemicals with similar chemical structures.

CAS – Chemical Abstracts Service
DOSS – dioctyl sulfosuccinate sodium
DPnB – dipropylene glycol n-butyl ether
EPA – US Environmental Protection Agency
nr – not reported
OECD – Organisation for Economic Cooperation and Development

Kujawinski et al. (2011) reported only minimal evident biodegradation of DOSS, a component of Corexit® formulations, in samples collected up to 64 days after dispersant application had ceased at the Deepwater Horizon wellhead.[7] It is important to note that dilution of the chemical over time resulted in barely detectable concentrations of DOSS (0.07 ppb); initial concentrations were assumed to be ~7 ppb, 3 orders of magnitude greater than was measured after 64 days. Baelum et al. (2012) reported that that DOSS, in particular, was substantially degraded during a 20-day experiment, but found that glycol components were less biodegradable during that time period.

The biodegradation of dispersed oil is well studied, although results vary among studies (NRC, 2005; Fingas, 2008; Bruheim et al., 1999). In general, biodegradation testing results indicate that oil dispersion increases the rate of oil elimination from the water column under a variety of conditions (Hua, 2006; Lindstrom et al., 1999; Lindstrom and Braddock, 2002; Hazen et al., 2010, as cited in Lee et al., 2011a; McFarlin et al., 2012b; Otitoloju, 2010; MacNaughton et al., 2003; Prince et al., 2003; Zahed et al., 2010; Zahed et al., 2011; Prince et al., 2013; Baelum et al., 2012). Zahed et al. (2011) reported Corexit® 9500-dispersed oil half-lives of 28, 32, 38, and 58 days at oil concentrations of 100, 500, 1,000, and 2,000 ppm, respectively; concentrations of dispersed oil have rarely exceeded 100 ppm during testing, and have not been shown to exceed 500 ppm (McAuliffe et al., 1980, 1981; Mackay and McAuliffe, 1988). These half-lives were all less than those of untreated oil: 31, 40, 50, and 75 days at the same respective oil concentrations. Baelum et al. (2012) reported that non-dispersed oil degraded only 20% within 20 days, whereas dispersed oil degraded by 60%, an increase of 40% caused by the addition of Corexit® 9500. Prince et al. (2013) reported half-lives for oil and Corexit® 9500-dispersed oil of 13.8 days and 11 days, respectively, corroborating previous results (2011; Baelum et al., 2012). It is important to note that the test conditions applied by Prince et al. (2013) and Baelum et al. (2012) (i.e., water temperatures of 8 and 5°C, respectively) were more relevant to Alaskan waters than those applied by 2011) (i.e., water temperature of 27.5°C). McFarlin et al. (2012b) reported that biodegradation increased in response to dispersant application when observing an Arctic microbial community exposed at -1 and 2°C (in two tests). Biodegradation in the Arctic has been shown to progress rapidly (Lee et al., 2011a), but there have been concerns over temperature limitations on microbial activity (Venosa and Holder, 2007). Rapid degradation under Arctic conditions may occur due to the presence of cold-adapted communities of symbiotic bacteria (Lee et al., 2011a; McFarlin et al., 2012a), and such adaptations are not adequately addressed when using one community at various temperatures, as was done by Venosa and Holder (2007).

[7] Kujawinski et al. (2011) did not observe degradation directly, but assumed that minimal degradation had occurred based on the small discrepancies from modeled concentrations (which assumed minimal degradation). In addition, the study was conducted on an atypical spill and response action; impacts related to deepwater applications of chemical dispersants are not being assessed under this consultation.

Increased biodegradation in the presence of dispersant chemicals is significant, but often incomplete. Biodegradation processes are limited largely to the lighter components of oil, and the addition of dispersants appears to facilitate the mineralization of oil only somewhat (McFarlin et al., 2012b). Studies investigating individual components of oil over time found that heavy components within degraded oil made up a larger proportion of the whole volume (Lindstrom and Braddock, 2002; Lindstrom et al., 1999). This has been shown to be true in field observations as well (Hazen et al., 2010; Atlas and Hazen, 2011). Heavier organic components of oil become enriched over time for both oil and dispersed oil (Lindstrom et al., 1999), so this phenomenon does not constitute a negative long-term impact on the degradation of oil relative to baseline conditions. Reductions in the biodegradation of some hydrocarbons due to the addition of chemical dispersant may be linked to selective inhibition of hydrocarbon-degrading bacteria in the marine environment (Hamdan and Fulmer, 2011). The results of such tests are not relevant to field conditions, considering the rapid community-level shifts that occur under natural conditions when oil and dispersant are introduced to a diverse microbial community (Hazen et al., 2010; Lu et al., 2011).

2.2.2 Abiotic degradation

Lyman et al. (1990) indicate that components of Corexit® 9500 are not expected to be susceptible to photolysis, although hydrolytic degradation may occur in the absence of microbial action. The half-lives indicated for individual components range from 77 days for Tween 85® to 7.7 years for Span® 80 (TOXNET, 2011). Rates of hydrolytic degradation vary greatly based on pH. For example, DOSS has a half-life of 240 days at pH 8, but a half-life of 6.7 years at pH 7, in the absence of microbial degradation (TOXNET, 2011). Because these chemicals have much shorter half-lives for biodegradation than under abiotic conditions, (George-Ares and Clark, 2000; Baelum et al., 2012), it is not expected that abiotic degradation pathways play a major role in initial degradation of Corexit® dispersants in the field.

Similarly, it is expected that abiotic degradation is limited relative to biodegradation (and physical effects) in decreasing the dispersed oil in an aquatic system over an extended period of time. However, physical weathering is known to have a marked impact on the initial concentration of oil, primarily since evaporation from the ocean's surface can result in the loss of approximately 20–50% of an oil spill within 24 hours (Mackay and McAuliffe, 1988; Suchanek, 1993). Similarly, many components of oil (e.g., PAHs) are susceptible to photolysis (Shemer and Linden, 2007).

2.3 TRANSPORT OF DISPERSANTS AND DISPERSED OIL

Horizontal transport of dispersants and dispersed oil is largely driven by ocean currents. Both oil and dispersed oil will assumedly be carried in the direction of major currents. It has been noted that the spread of oil across the ocean's surface can rapidly increase after dispersant application (preceding dispersion into the water column)

(NRC, 2005), and that dispersants sprayed at the edge of a slick can cause oil to be herded, whereby the slick area decreases somewhat (Fingas, 2008).

The long-distance transport of dispersants was studied by Kujawinski et al. (2011), who observed a component of Corexit® dispersant formulations, DOSS, after application in deep water (900 to 1,400 m) during the DHOS event. The compound was found within plumes of dispersed oil and gas from the point of application up to 315 km away at a detectable concentration (0.07 ppb) up to 64 days later.[8] The transport of dispersant components within oil plumes is expected due to the known partitioning characteristics of the surfactant components of Corexit® formulations, as well as the creation of surfactant micelles (Figure 1) (TOXNET, 2011; Nalco, 2005, 2010). It has been noted that, at very dilute concentrations of dispersant, surfactants may slowly partition to the water column and be lost from the dispersion process (Fingas, 2008).[9] Although such transport was observed after DHOS, that instance may not be an entirely relevant case study, because the application of chemical dispersants at the wellhead in deepwater represented an atypical response action, one that is not being assessed as part of this consultation.

Vertical transport of dispersants and dispersed oil is limited by density gradients within the water column that are controlled by temperature and salinity. Temperature gradients are referred to as thermoclines, and the salinity gradient is referred to as the pycnocline; each represents a density barrier against sea water mixing. Typically, the pycnocline is between 5 and 10 m below the ocean's surface (NOAA, 2012b), and thermoclines exist even deeper (i.e., 100 m or more). The presence of density barriers does not hinder the rapid dilution of dispersants and dispersed oil, because in addition to being transported vertically to approximately 10 m, they also are transported horizontally through advection caused by ocean currents (NRC, 2005; NOAA, 2012b).

The buoyancy of dispersed oil droplets is driven by their size (i.e., diameter), such that smaller droplets disperse deeper and rise to the surface more slowly (NRC, 2005). Also, the presence of suspended sediment can regulate droplet buoyancy through the creation of oil-mineral aggregates that tend to sink (Fingas, 2008). In the event that stable emulsions do not form, which can be common (Fingas, 2008), dispersed oil tends to remain in the water column for between 4 and 24 hours before resurfacing.

[8] The application of dispersants at depth will not occur in Alaskan waters because oil exploration and drilling occurs in waters less than 300 m deep. Some components of Corexit® were not detected after DHOS in any samples collected by EPA (data available through Socrata, 2012). Similar monitoring by the United States Coast Guard (USCG) (2010) resulted in no exceedances of established dispersant chemical component benchmarks. However, USCG did observe detectable concentrations of dispersant constituent chemicals in 60 of 4,850 samples (2010). Discrepancies among the results of Kujawinski et al. (2011), EPA (data available through Socrata, 2012), and USCG (2010) may be due to differences in sampling depth, location, and target analytes.

[9] Note that this occurs specifically under conditions of dilute concentrations (Fingas, 2008); this process is unlikely to contribute sufficient chemicals to illicit toxic effects in marine biota.

Based on the dilution modeling conducted by Nedwed (2012), Gallaway et al. (2012), and Mackay and McAuliffe (1988) (Section 2.1, Figure 2), 4 to 24 hours is sufficient to greatly dilute the concentrations of dispersant and dispersed oil. Lewis et al. (1995) also showed that subsequent sprayings can increase the effectiveness of dispersion when oil resurfaces quickly, resulting in a rapid removal of oil from the ocean's surface.

3 Effects

3.1 SUMMARY OF KNOWN EFFECTS OF OIL, DISPERSANTS, AND DISPERSED OIL

3.1.1 Effects of chemical dispersants

The purpose of this section is to discuss the mechanisms of toxicity or physical impacts of dispersants alone (i.e., without oil). The toxicity of dispersants is typically less than that of oil (Fingas, 2008; NRC, 2005), so impacts of dispersants alone on aquatic species are not expected to be greater than those of oil on its own; however, the combination of oil and dispersants can be either more toxic (NRC, 2005; Fingas, 2008) or less toxic than oil alone.[10]

Dispersants are not intended to be applied to wildlife at all, neither directly nor indirectly; therefore, concentrated exposure to dispersants alone is not expected as a result of their application. Exposures to very diluted concentrations may occur as a result of leaching to the water column from micelles over time (Fingas, 2008) or, to a limited extent, as a result of overspray during application (Butler et al., 1988; Scelfo and Tjeerdema, 1991). The effects caused by dispersed oil are discussed in Sections 3.1.2 and 3.2.5. Although dispersants are shown to have inherently toxic characteristics in this section, later discussions (Sections 3.1.2 and 3.2.5) provide evidence that dispersants may mitigate the acute (i.e., lethal) toxicity of oil alone to certain species (e.g., larval fish and invertebrates), or have little to no effect on species that pass through the upper 10 m of the ocean, but generally reside much deeper (e.g., cetaceans, pinnipeds, fish, and marine reptiles).

3.1.1.1 Fish

The toxicity of dispersants to sensitive species and life stages of fish are discussed at length in Section 3.2, and so will only be noted briefly here. Abnormal development and narcosis are the most often cited modes of toxicity (NRC, 2005). At very low doses, dispersants have been shown to be embryotoxic to fish exposed at early life stages (Lonning and Falk-Petersen, 1978; Falk-Petersen et al., 1983). This is only relevant to Pacific herring (*Clupea pallasii*), which spawn in Alaska nearshore waters. While the direct application of dispersants is not intended for nearshore waters, dispersion in open water that, over time, results in diluted dispersant concentrations in nearshore waters could have a marked impact on Pacific herring, a species highly sensitive to dispersed oil. However, given the toxicity of oil alone and the potential impacts caused by oiling of nearshore areas and intertidal shorelines, it may still be

[10] The analysis presented in Sections 3.3 and 3.4 of this appendix show that the lethality of chemically dispersed oil is less than that of oil. Figures 8 and 9 clearly show the differences between oil and chemically dispersed oil, particularly oil dispersed by Corexit® 9500.

beneficial (relative to baseline oiling) to apply dispersants, if done at a distance from known spawning habitat. This is further explained in Section 3.2 and Section 4.

ESA-listed Chinook (*Oncorhynchus tshawytscha*) and coho salmon (*Oncorhynchus kisutch*) and steelhead trout (*Oncorhynchus mykiss*) are present only as juveniles and adults in Alaska waters, and therefore are not as susceptible as Pacific herring to the toxic effects of dispersants. This is further discussed in Section 4.

3.1.1.2 Birds

Chemical dispersants are known to impact bird species in various ways. Dispersants have been shown to substantially alter the structure and function of common murre (*Uria aalge*) feathers; the impact of dispersants alone on feather structure has been shown to be greater than that of dispersed oil or oil alone (Duerr et al., 2009; 2011). Such alterations in feather structure have been observed in lesser scaup (*Aythya affinis*) that were exposed to oils and/or dispersants (Stephenson, 1997), and these alterations are known to lead to a loss of thermoregulatory ability (Jenssen and Ekker, 1991a, b). Lost thermoregulation in experiments has been largely associated with oil, rather than dispersants (Lambert et al., 1982). Lambert et al. (1982) observed that birds became wetted and lost buoyancy when exposed to dispersants, although this did not immediately impact their metabolic rate. This suggests that, although oil drives the loss of thermoregulation, dispersants may contribute to lost thermoregulation by allowing greater wetting of feathers, facilitated in part by the alteration in function (Duerr et al., 2011). Diminished thermoregulation is particularly important to birds in Alaska, where temperatures are often low enough to induce hypothermia, and where birds have adapted specialized feathers for trapping heat. For example, Jenssen and Ekker (1991a, b) showed that common eider (*Somateria mollissima*) were more affected by alterations to their feathers (made incrementally worse by the addition of dispersants to oil) than were mallards. Furthermore, molting birds, which already have functionally compromised plumage, are more susceptible to the impacts of oil or dispersants (Stephenson, 1997), and are less able to avoid oil. This is an important consideration for any dispersant application, particularly near critical molting habitat for Steller's (*Polysticta stelleri*) and spectacled eiders (*Somateria fischeri*) (Petersen et al., 1999). The ecology of ESA-listed species is discussed at length in Section 3 of the BA. Other ESA-listed bird species, including short-tailed albatross (*Phoebastria albatrus*), yellow-billed loon (*Gavia adamsii*), and Kittlitz's murrelet (*Brachyramphus brevirostris*), could be similarly impacted at the individual level (e.g., reduced survival) if directly coated with chemical dispersants. Since dispersants are not intended for direct application to birds, the probability of such an undesirable incident occurring is remote (Butler et al., 1988). If dispersants were applied to a slick that later came into contact with birds, negative impacts on bird plumage could increase relative to the baseline condition (Duerr et al., 2009, 2011; Jenssen and Ekker, 1991a, b). However, the volume of oil at the ocean's surface is expected to diminish once dispersant has been applied (Lewis et al., 1995; Section 2), thereby reducing the area in which birds could be impacted by dispersed oil. Furthermore, it has been claimed (CDC and ATSDR, 2010; Lessard and Demarco, 2000) that the application of dispersants to oil (and the subsequent formation of oil droplets) may reduce the likelihood of birds becoming oiled, at least by dispersed oil droplets.

In one study, ingestion of concentrated Corexit® 9527 was shown to have acute but non-lasting neurological impacts on birds that persisted for a few hours (Rocke et al., 1984). All birds returned to normal within 24 hours, and none died from such exposure. This effect was not observed in either crude oil only or dispersed oil treatments (Rocke et al., 1984). Behavioral impacts resulting from temporary intoxication may result in decreased fitness or the death of some individuals (e.g., if birds could not escape predation). It is not likely that highly concentrated doses of dispersants will be directly ingested by birds immediately following application, given the rapid rate of dilution expected to occur (Section 2). Birds are also expected to disperse due to noise caused by response workers, equipment, and airplanes, or be dispersed (i.e., hazed using noise), such that they would not be present in an area at a time when dispersants were most concentrated in the water column.

The inhalation of fumes from dispersants poses little risk to birds and other animals, unless they are directly exposed to undiluted dispersants. Such exposure is unlikely considering the best management practices (BMPs) or response actions (e.g., avoidance of wildlife, monitoring for bird presence, and hazing in an area to intentionally disperse wildlife) that could be implemented prior to chemical dispersion.

Of the chemicals in Corexit® 9527 and Corexit® 9500, both petroleum distillates and 2-butoxyethanol are volatile, although the manufacturer notes inhalation as a potential route of exposure. Inhalation (or aspiration) of sprayed droplets during application is perhaps the more likely pathway of exposure for the non-volatile components of chemical dispersants than volatilization from the ocean surface. Nalco (2005, 2010) and the Centers for Disease Control and Prevention (CDC) (CDC and ATSDR, 2010) report that prolonged inhalation of Corexit® chemicals may cause chemical pneumonia, respiratory irritation, and eye irritation. Corexit® 9527 specifically contains 2-butoxyethanol, which, after prolonged exposure, can cause damage to the blood (i.e., hemolysis), liver, and kidneys, central nervous system depression, nausea, vomiting, anesthesia, and narcotic effects (Nalco, 2010; CDC and ATSDR, 2010). Oil alone is also known to contain approximately 20 to 50% volatile chemicals (by volume) (Mackay and McAuliffe, 1988; Suchanek, 1993), which may cause similar impacts in birds through inhalation. The inhalation or aspiration of chemical dispersants is a possible outcome of a worst-case scenario in which the chemical is sprayed in the immediate vicinity of ESA-listed or candidate species; in the main text of the BA, this is noted as a possible impact on all air-breathing ESA-listed or candidate species (i.e., excluding fish species).

Various studies have observed the embryotoxicity of Corexit® 9500 to birds by directly applying the chemical to mallard (*Anas platyrhyncos*) eggs (Wooten et al., 2012). Direct exposure of mallard eggs to Corexit® 9500 resulted in significantly reduced hatch success at an application of 20 µl (of pure dispersant), and significantly reduced the developmental stage (mortality occurred at 40 µl of pure dispersant). As mentioned above, the direct application of dispersants to adult birds (i.e., nesting parents) is neither intended nor likely (Butler et al., 1988), nor is application of dispersants to terrestrial habitats where birds nest (Wooten et al., 2012). There are currently no studies available that investigate the embryotoxicity of Corexit® 9527 alone.

3.1.1.3 Mammals

Dispersants have no visible impact on sea otter fur structure (Duerr et al., 2009; 2011), but the effects of oil on thermoregulation have been shown (Geraci and St. Aubin, 1980; St. Aubin, 1988; Geraci, 1990). This is particularly significant to marine mammals that do not have subcutaneous blubber to regulate their body temperature (Geraci and St. Aubin, 1980). The sea otter is the most relevant marine mammal in this BA that utilizes dense fur to trap air against the skin (Williams et al., 1988). It is not clear if dispersants will physically affect mammals.

Data on the toxicity of dispersants to mammals are very limited. The inhalation of fumes from dispersants poses a possible route of exposure, and could lead to various localized or systemic impacts including chemical pneumonia; inflammation of organ tissues (e.g., eyes and respiratory tract); increased difficulty breathing (not directly related to inflammation) (Roberts et al., 2011); injury to kidneys, liver, and blood cells (i.e., hemolysis); nausea; vomiting; narcosis; defatting and drying of skin; dermatitis (Nalco, 2005, 2010; CDC and ATSDR, 2010); and acute neurological impacts (e.g., altered neurotransmitter signaling) potentially leading to chronic depression, lack of motor coordination, and short-term memory loss (Sriram et al., 2011). It is unclear how neurological impacts could affect ESA-listed mammals at the individual level (e.g., reduced survival), but behavioral impacts could assumedly result in a diminished ability to forage or avoid predation. It is not clear whether ecologically relevant concentrations of chemical dispersants will result in such impacts on marine mammals, particularly after dispersants mix into the water column. Direct application to mammals is not the intended or suggested use of chemical dispersants, and BMPs or response actions (e.g., avoidance of wildlife, monitoring for mammal presence, and hazing in an area to intentionally disperse wildlife) should mitigate animal exposures to concentrated dispersant chemicals.

3.1.1.4 Invertebrates

The toxicity of dispersants to invertebrates (which may compose part of the diet of ESA-listed species) is discussed at length in Section 3.2. Abnormal development and narcosis are the most often cited modes of toxicity (NRC, 2005), although numerous sublethal impacts on invertebrates may also occur. Dispersants have been shown to be toxic to invertebrates at early life stages at very low doses (Lonning and Falk-Petersen, 1978; Falk-Petersen et al., 1983), but dispersants have also been shown to be less toxic than oil alone (Attachment B-1; Fingas, 2008; NRC, 2005). Therefore, dispersants alone do not pose a greater threat than that of the baseline condition for a spill cleanup.

3.1.1.5 Marine reptiles

At present, there are no known studies investigating the impacts of dispersants alone on marine reptiles, such as sea turtles. There is extensive research on the effects of oil alone, and at least one study investigating dispersed oil. Dispersants are not intended for direct application to sea turtles, so direct toxicity due to dispersants alone is unlikely. Various other factors limiting the likelihood of exposure of marine reptiles to oil response actions in Alaska are discussed in Sections 2 and 3 of the BA. Nesting does not occur in Alaska (Section 3 of the BA), so ESA-listed marine reptiles in sensitive life stages would not be exposed to dispersants (or dispersed oil) as a result of an oil spill response in Alaska. Furthermore, the presence of marine reptiles in Alaska is “accidental or uncommon” (Section 3.4.4 of the BA), which limits the likelihood of an individual coming into contact with dispersants, spilled oil, or dispersed oil in Alaska waters.

3.1.2 Known effects of oil and dispersed oil

Dispersants are known to have a variety of effects on aquatic species (Sections 3.1.1.1 to 3.1.1.5). However, the toxicities of various dispersants (e.g. Corexit® 9500 and Corexit® 9527) are known to be less than that of crude oil alone (Fingas, 2008; NRC, 2005); conversely, some have shown dispersed oil to be more toxic than either oil or dispersants alone (Attachment B-1; Fingas, 2008; NRC, 2005). Therefore, the impacts of dispersed oil are caused primarily by the toxicity of oil, and may be enhanced by its interaction with dispersants. The enhanced toxicity of dispersed oil (over oil alone) is frequently attributed to the increased bioavailability of the toxic components of oil, principally PAHs (Wolfe et al., 1998; Wolfe et al., 2001; Yamada et al., 2003; Ramachandran et al., 2004; Milinkovitch et al., 2011a). Dispersants have been shown to increase the acute toxicity (e.g., lethality) of oil in only about half of the comparable studies (Attachment B-1, Section 3.4.1); the other half of these studies showed that chemical dispersants actually decrease the lethality of oil in a mixture. These studies are discussed in Sections 3.2 through 3.4, which present , SSDs developed to show how oil and dispersed oil compare across the available studies of acute toxicity (Figures 8 and 9). When considering the available, relevant, and comparable acute toxicity data in Attachment B-1 (including studies in which oil toxicity was enhanced by chemical dispersants), it appears that the acute lethality of oil is generally decreased by chemical dispersants.

The sublethal impacts of dispersed oil are generally enhanced relative to those of oil alone (Attachment B-1), suggesting that an immediate response to dispersed oil exposure is generally less likely than a delayed response (e.g., decreased fitness leading to death). Due to diminishing concentrations of dissolved and dispersed components of oil in the water column over time (Section 2), long-term impacts are unlikely within an area. Observed impacts (i.e., toxicity endpoints) of chronic exposure to PAHs include genotoxicity, immunotoxicity, histopathological impacts (e.g., hepatic lesions), behavioral impacts, and reproductive impacts (Payne et al., 2003; Albers and Loughlin, 2003; Malcolm and Shore, 2003; Besten et al., 2003; Meador, 2003; Barron, 2012; Godschalk et al., 2000; Lemiere et al., 2005; Carls et al., 1999; Jonsson et al., 2010). The likelihood of such impacts affecting listed species as a result of short-term exposure is a point of uncertainty, although the rapid reduction in exposure concentrations and biodegradation of dispersed oil within a relatively short time period (Section 2) may limit the likelihood.[11] Changes in enzyme activity, blood plasma chemistry, and increased PAH metabolites in bile have been observed in various species after exposure to dispersed oil, suggesting that exposure increases, but not necessarily that impacts at the individual level (i.e., reduced growth, reproduction, or survival) occurs (Lee and Anderson, 2005; Cohen et al., 2001; Ramachandran et al., 2004; Baklien et al., 1986).

3.1.2.1 Fish

The exposure of fish to oil (and its component chemicals) appears to occur predominately across the gill surface or through ingestion of contaminated food (Baussant et al., 2001; Cohen et al., 2001; Milinkovitch et al., 2011b). If exposed continuously to PAHs dissolved in the water column, oil may require as many as seven days to reach a maximum concentration in fish (Logan, 2007). The more soluble components of oil (e.g., low-molecular-weight PAHs [LPAHs]) are internalized across the gills more efficiently than the larger molecules, resulting in a greater exposure to LPAHs than to high-molecular-weight PAHs (HPAHs) over short time periods (Baussant et al., 2001; Cohen et al., 2001; Wolfe et al., 2001). HPAHs may be quickly and efficiently metabolized and depurated from some fish (e.g., turbot) (Baussant et al., 2001), whereas they are concentrated in invertebrates (e.g., *Mytilus edulis*) (Baussant et al., 2001). Due to the rapid depuration of the LPAHs, Wolfe et al. (2001) did not find a significant increase in the accumulation of an LPAH (i.e., naphthalene) or its metabolites after 12 hours of depuration in larval topsmelt.

HPAHs, which fish can also internalize across the gills, are metabolized and excreted from the fish body at a slower rate than LPAHs (Logan, 2007; Payne et al., 2003); their solubility also increases after dispersant application, resulting in greater exposure for fish to HPAHs than after exposure to untreated crude oil (Couillard et al., 2005; Cohen et al., 2001). HPAH accumulation is more strongly correlated with enzymatic

[11] Impacts of chronic PAH exposure have historically been reported for species found in areas impacted by spilled but untreated oil (e.g., sea otters in PWS after EVOS) or in areas with significant anthropogenic inputs of contaminants (e.g., beluga St. Lawrence waterway), including but not limited to PAHs. Therefore, such impacts cannot be directly related to dispersants or PAHs alone.

responses indicative of metabolism in fish (and subsequent exposure to toxic PAH metabolites) (Couillard et al., 2005). The correlation between HPAH exposure and metabolic activity further indicates that these chemicals are efficiently metabolized to forms that can be removed from the body, limiting trophic transfer.[12]

Similarly, the accumulation of oil and its components in invertebrates, which is enhanced by the addition of chemical dispersants (Wolfe et al., 1998; Jensen et al., 2011), can influence uptake in fish species through ingestion. Ingestion of contaminated food appears to be more important in the exposure of fish to HPAHs, because lipids in prey items, specifically invertebrates, accumulate organic, lipophilic compounds such as HPAHs (Logan, 2007). However, the apparent exposure of fish to HPAHs when fed dispersed oil-contaminated prey was not significantly different from the exposure of fish fed crude oil-contaminated prey (Cohen et al., 2001). Wolfe et al. (2001) reported a similar result for the accumulation of naphthalene and its metabolites in larval topsmelt exposed to both contaminated food and exposure solution.

Reported individual-level impacts (i.e., impacted growth, survival, or reproduction) on fish include abnormal growth, reduced growth (Claireaux et al., 2013; Couillard et al., 2005), reduced hatch (Greer et al., 2012; Anderson et al., 2009), and mortality (Van Scoy et al., 2012). An additional impact of note is the onset of blue sac disease, which was observed in Atlantic herring (*Clupea harengus*) by Greer et al. (2012). Reduced hatch and diseases in early-life-stage individuals pose a significant threat at the individual and population levels for fish species known to spawn in Alaska (e.g., Pacific herring). However, Greer et al. (2012) showed that dispersion reduced the acute toxicity of oil to Atlantic herring embryos 5, 30, and 60 minutes post-dispersion, even though blue sac disease had been induced.[13] This disease has been observed in fish exposed to either oil alone or dispersed oil (Greer et al., 2012; Colavecchia et al., 2006). Reduced acute toxicity in Chinook salmon was observed by both Lin et al. (2009) and Van Scoy et al. (2010). Therefore, the impact of chemical dispersion on oil toxicity to fish is uncertain, although likely to be enhanced in embryonic and larval life stages in planktonic fish species (e.g., Pacific herring).

In addition to causing internal impacts, dispersed oil affects transfer across the gills of fish (Singer et al., 1996), particularly by affecting Na^+/K^+-ATPase pumps (Duarte et al., 2010), which are necessary for regulating ionic and osmotic gradients in fish tissues. Duarte et al. (2010) showed that the flux of ions across fish gills significantly

[12] HPAHs are known to be broken down into much more toxic metabolites prior to egestion, and metabolites have been linked to various sublethal impacts on fish (Logan, 2007; Payne et al., 2003). Although PAHs are actively metabolized and excreted, it is not implied here that sublethal impacts will not result.

[13] Solution collected 15 minutes post-dispersion from the wave tanks where dispersion was conducted was more toxic than oil alone (Greer et al., 2012); it is unclear why this duration resulted in a conflicting result.

increased (both influx and efflux), and that the net flux significantly decreased, such that more sodium was lost from the gill surface, when fish were exposed to dispersed oil, relative to the control, dispersant-only, or oil-only treatments. Such a disruption could lead to increased stress in fish. However, the effect does not directly relate to an impact at the individual level (i.e., reduced survival, growth, or reproduction).

Although bioaccumulation of PAHs has been shown to occur in fish over short time periods, efficient metabolic processes limit the bioconcentration of PAHs in fish tissues over time (Logan, 2007; Payne et al., 2003) and the transfer of parent PAHs from fish to higher trophic levels (i.e., birds and mammals) (Payne et al., 2003; Albers and Loughlin, 2003). The transfer or bioconcentration of PAH metabolites in higher trophic levels has not been extensively studied; it is possible that metabolites stored in fish lipids could be transferred to higher trophic levels, resulting in PAH-related toxicity in those species.

3.1.2.2 Birds

The impacts of oil on birds are well documented. For example, Holmes et al. (1979) showed that mallards that ingested large quantities of oiled food succumbed to stress-related exhaustion more frequently than those that did not ingest oiled food. Eastin and Rattner (1982) observed that oil ingestion resulted in altered blood chemistry and lost osmoregulation (i.e., retaining of salt after seawater ingestion), and cited reduced growth as also possible after oil exposure through ingestion. The same authors noted that such impacts appeared to be mitigated when exposed to Corexit® 9527-dispersed oil. Rocke et al. (1984) observed immunological impacts on waterfowl exposed to ingested crude and dispersed oil.

Oiling causes hypothermia in birds by altering the function of feathers that regulate body heat (O'Hara and Morandin, 2010; Jenssen, 1994; Stephenson, 1997; Jenssen and Ekker, 1991a, b). Duerr et al. (2009; 2011) showed that dispersed oil had a greater impact on common murre feathers than did oil alone, likely leading to a loss of thermoregulatory ability, hypothermia, and death. This result has been corroborated in mallard and common eider (Jenssen and Ekker, 1991a, b); conversely, Lambert et al. (1982) showed that mallards exposed to dispersed oil experienced changes in basal metabolic rate not significantly different from those caused by oil, and that dispersants alone did not increase their metabolic rate relative to the control; the key difference between Lambert et al. (1982) and Jenssen and Ekker (1991a, b) is that the latter exposed birds on water, whereas the former exposed birds on water briefly, then moved them to dry land. Lambert et al. (1982) speculated that prolonged exposure to cold water and dispersed oil would have different results than exposure to only dispersed oil, which Jenssen and Ekker (1991a, b) later definitively showed. The CDC (CDC and ATSDR, 2010) and Lessard and Demarco (2000) noted that dispersants could make oil droplets "less likely to stick to birds and other animals," so oiling may be mitigated somewhat by chemical dispersion. However, it is likely that dispersed oil has greater physically impacts than oil alone at equivalent concentrations (Jenssen and

Ekker, 1991a, b). Section 2 discusses how the dilution of dispersed oil and its subsequent removal results in a marked decrease in the concentration of oil at the ocean's surface.

The toxicity of oil to birds has been reported in the literature, and various impacts have been observed. For example, Esler et al. (2010) reported that harlequin ducks (*Histrionicus histrionicus*) in areas oiled by the *Exxon Valdez* oil spill (EVOS) had elevated levels of ethoxyresorufin-O-deethylase (EROD) compared to birds that frequented nearby, un-oiled areas, indicating exposure to oil-related hydrocarbons some time after shoreline oiling had occurred. Exposure to oil during the EVOS event resulted in mass bird mortalities related to the ingestion of hydrocarbons (in addition to the loss of thermoregulatory ability) (Peterson et al., 2003). Stubblefield et al. (1995a, b) indicated that impacts on adult mallards related to oil ingestion were not significantly different from impacts on control birds, but that significant impacts on egg production, shell thickness, and hatch success resulted from exposure to oil; hatch success was reduced when oil was directly applied to the mallard egg. Eastin and Rattner (1982) observed that ingestion of oil was related to alterations in blood chemistry, potentially leading to immunological impacts and reduced osmoregulation; the authors suggested that mallards could probably ingest low levels of oil for months without exhibiting effects. Barron (2012) cites additional sublethal impacts on birds exposed to petroleum products, which include hemolytic anemia, the presence of Heinz bodies in red blood cells,[14] cachexia,[15] and diminished resistance to bacterial infection.[16] Reduced immune response was also noted in oiled, rehabilitated, and released American coots (*Fulica americana*) (Newman et al., 2000). It is not clear if the chemical dispersion of oil would increase such impacts on birds, but it is expected that any measure reducing the direct oiling of birds would diminish the likelihood of such impacts; therefore, chemical dispersion, which is expected to reduce such oiling (CDC and ATSDR, 2010; Section 2), is expected to reduce the likelihood of sublethal impacts related to oiling.

Modeling conducted by French-McCay (2004) estimated that waterfowl and other surface-dwelling birds that came into contact with oil spills in open ocean environments (i.e., where dispersants would be applied) had a 99% probability of

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[14] Heinz bodies are inclusions within red blood cells that have been linked to various blood disorders, including hemolytic anemia. Heinz bodies are caused by heritable mutations or oxidative stress; oxidative stress is generally caused by reactive oxygen species or "oxygen radicals." PAHs are known to react in the body to create oxygen radicals (Altenburger et al., 2003).

[15] Cachexia is also referred to as "wasting syndrome," and is characterized by weight loss, fatigue, muscle atrophy, and weakness that cannot be corrected nutritionally. Cachexia has been observed in cases of advanced cancers, infectious diseases such as AIDS or tuberculosis, and exposure to contaminants such as mercury.

[16] Barron (2012) also notes that studies with mallards exposed to Bunker C and dispersed Bunker C (through ingestion) did not show significantly reduced antibody production or resistance to viral infection.

mortality. French-McCay (2004) also noted that species of loon (i.e., yellow-billed loon), which do not behaviorally avoid oil, are more susceptible to oiling than those species of birds that do avoid oiled areas. It is clear that oiling alone poses a significant threat to ESA-protected birds.

Dispersed oil may be more toxic to and have greater physical impacts on bird species than oil alone. Butler et al. (1988), Finch et al. (2012), and Peakall et al. (1987) showed that dispersed oil is more toxic to developing birds exposed *in ovo*[17] than oil alone. However, the application of chemical dispersants is expected to reduce the exposure of birds to oil; this assumption is discussed further in Section 4, and is corroborated by modeling reported by French-McCay (2010). Also, it has been observed that the application of dispersants can, under certain circumstances, reduce embryotoxicity from oil in birds (Albers and Gay, 1982; Albers, 1979; both as cited in Wooten et al., 2012). In these ecologically relevant tests, which observed the toxicity of dispersed oil applied to eggs via contact with an oiled nesting parent, it was shown that dispersants more often increased the toxicity of oil to the developing embryo (Albers and Gay, 1982, as cited in Wooten et al., 2012; Peakall et al., 1985, as cited in Peakall et al., 1987).

Corexit® formulations may contribute volatile petroleum distillates or 2-butoxyethanol (Table 2; TOXNET, 2011) to the environment, possibly resulting in increased inhalation exposure relative to oil alone. However, approximately 20 to 50% of crude oil is composed of volatile chemicals that are lost on the first day after an oil spill (Mackay and McAuliffe, 1988; Suchanek, 1993), a greater volume of volatile chemicals than is added by the application of dispersants. More importantly, dispersants decrease the amount of chemical that is released through evaporation (NRC, 2013), so chemical dispersant application may mitigate impacts on ESA-protected species of birds (as well as other animals, including human responders) caused by inhalation of multiple chemicals, relative to the baseline condition. The dispersion of volatile chemicals into the water column represents a trade-off in toxicity between protecting species that breathe air (e.g., birds) and protecting those that do not surface to breathe (e.g., fish). This is also an important consideration for human safety during a response action (NRC, 2013).

Chemical dispersants have been shown to decrease the amount of oiling of shorelines, thereby reducing the chronic input of hydrocarbons to filter-feeders such as bivalves, and reducing the long-term (i.e., > 2 years) uptake of hydrocarbons in those species from oiled sediment (Humphrey et al., 1987). Since both shoreline and bird oiling are known to have severe impacts, chemical dispersant application may, under certain circumstances, have an immediate benefit to ESA-listed species. It is not clear whether short-term benefits (e.g., reduced oiling of birds or forage habitat) outweigh potential

[17] Butler et al. (1988) and (Peakall et al., 1987) exposed eggs indirectly, applying the oil to the parent's breast. Finch et al. (2012) exposed eggs directly, brushing the oil onto the egg.

long-term impacts (e.g., altered prey base, increased PAH contamination in prey, and sublethal effects of PAH toxicity).

3.1.2.3 Mammals

Geraci and St. Aubin (1988) and Williams et al. (1988) showed that sea otter are susceptible to lost thermoregulation after contact with crude oil. This impact can result in either hypothermia and death (Geraci and St. Aubin, 1988), or sublethal effects on behavior (Davis et al., 1988). The effect is likely to depend on the season in which the exposure occurs, as colder ambient temperatures result in more severe effects once thermoregulation is compromised. Geraci and St. Aubin (1988) also note that oil alone can impact buoyancy, which can result in drowning.

Results from Duerr et al. (2011) suggest that dispersants do not increase the impacts of oil on thermoregulation, since ecologically relevant concentrations of dispersed oil (12 to 320 ppm) do not alter the functional structure[18] of otter fur. This was corroborated by Williams et al. (1988), who found the increase in metabolic activity in oiled otters to be similar to that of otters exposed to dispersed oil. The application of dispersants is expected to decrease the exposure of mammals to oil that are sensitive to its physical impacts (e.g., sea otter); this is discussed further in Section 4. Note that the CDC (CDC and ATSDR, 2010), as well as Lessard and Demarco (2000), claim that dispersants may reduce the likelihood of oil droplets sticking to animals, so the physical impacts on sea otter of oiling may be reduced by the application of dispersants.

It is important to note that most of the marine mammals assessed in this BA, particularly those that develop subcutaneous blubber, are not expected to be impacted by physical effects of oiling. Primary examples include cetaceans and pinnipeds, which regulate their body heat with blubber. According to modeling conducted by French-McCay (2004), the probability of surface oiling in the open ocean leading to death is 0.1% for cetaceans, 1% for pinnipeds, and 75% for furbearing marine mammals (e.g., sea otter). Clearly, sea otter is the ESA-listed species assessed in this BA most susceptible to the physical impacts of oiling.

Toxicity and altered behaviors in mammals relating to oil has been documented extensively. Geraci and St. Aubin (1988) provided a review of the known impacts of oil alone on marine mammals, including sea otter, polar bear, pinniped, and cetacean species. Examples of known impacts of oil alone on pinnipeds and otters include irritation of the eyes, skin, and other sensitive tissues or mucous membranes; reduced body weights in pups; altered maternal care for pups (potentially due to olfactory disturbance); altered swimming behaviors; loss of thermoregulatory ability; gastrointestinal distress after direct ingestion; organ lesions when vapors are inhaled; and reduced resilience to stress (Geraci and St. Aubin, 1988). Duffy et al. (1994)

[18] Weisel et al. (2005) provides a discussion of the functionality of otter fur in relation to maintaining body heat.

observed that otters abandoned latrine sites that had been oiled, even after two years had elapsed since the oiling.

Cetaceans are likely to be affected in similar ways, such that oiling may lead to localized irritation of tissues, and gastrointestinal problems relating to the ingestion of oil. Fouled baleen is another possible effect, assumedly resulting in decreased feeding efficiency. Feeding at the surface is uncommon among whales, although some species may skim feed or surface in oil, resulting in some ingestion of oil alone. Skim feeding has been observed in North Pacific right whales and sei whales, which are assessed more specifically in Sections 5.1.7 and 5.1.8, respectively.

Taylor et al. (2001) and Duffy et al. (1994) observed altered blood chemistry in otters exposed to oil alone, but it is unclear the extent to which such impacts relate to effects at the individual level (i.e., reduced survival, growth, or reproduction). Toxic impacts relating to ingestion are fairly minimal, unless very large volumes of oil are ingested; Geraci and St. Aubin (1988) indicated that given the small volumes of oil found in pinniped stomachs after oiling events and the infrequency of grooming, this is unlikely for pinnipeds. Cetaceans do not groom either, but sea otters groom frequently; among the marine mammals, sea otters are the most likely to ingest large quantities of oil from their coats. The low toxicity of ingested oil is corroborated by other studies (Rogers et al., 2002; Stubblefield et al., 1995a), although tissue damage was noted at relatively high rates of ingestion (in mouse and ferret tests). Sea otters have been shown to suffer from immunological impacts resulting from modifications to gene expression after exposure to PAHs from crude oil (Bowen et al., 2007).

Dispersed oil sometimes has greater toxicity than oil alone, assumedly due to the higher bioavailability of toxic components such as PAHs (Wolfe et al., 2001; Wolfe et al., 1998; Ramachandran et al., 2004; Yamada et al., 2003; Milinkovitch et al., 2011a). PAHs are known carcinogens that cause oxidative stress and DNA damage (Lemiere et al., 2005), as well as narcosis (DiToro et al., 2000), topical lesions, developmental deformities, decreased growth, and ultimately mortality (Albers and Loughlin, 2003; Logan, 2007). They are also known to become more toxic when released into the environment than when studied under controlled laboratory conditions (due to photo-enhanced toxicity) (Barron, 2006; Barron et al., 2008; Barron and Ka'aihue, 2001) particularly after dispersant application (Barron, 2003; Ramachandran et al., 2004; Milinkovitch et al., 2011a). PAHs are bioaccumulated in the tissues of many species that may then be ingested by mammals; for example, bivalves and other invertebrates accumulate PAHs (Wolfe et al., 1998; Logan, 2007; Meador, 2003).

It is unclear whether mammals exposed to increased PAHs in a dispersed oil plume will develop any symptoms or be directly impacted at the individual level (i.e., reduced survival, growth, or reproduction).

Trophic transfer of parent PAHs (i.e., non-metabolized PAHs) from invertebrates to marine mammals is not thought to be significant (Albers and Loughlin, 2003), because metabolisms at higher trophic levels (i.e., above invertebrates) limit such accumulation (or biomagnification) (Wolfe et al., 2001; Albers and Loughlin, 2003). Fish may accumulate PAHs in their tissues, but they also are able to readily metabolize these chemicals (Logan, 2007), somewhat limiting the trophic transfer of parent PAHs to predominantly piscivorous mammals (Albers and Loughlin, 2003). Wolfe et al. (2001) found that Corexit® 9527 significantly increased the uptake of naphthalene from the water column by larval topsmelt (*Atherinops affinis*), but dispersants also resulted in significantly increased depuration; the result after 12 hours was a slightly decreased final tissue concentration of naphthalene. Using a simplified food chain, Wolfe et al. (2001) found that the dietary uptake of naphthalene was different between oil and dispersed oil. For this reason, piscivorous mammals are less likely to accumulate (or biomagnify) high concentrations of parent LPAHs as a direct result of dispersant application.

HPAHs are also metabolized by fish, though the rate of excretion is slower than for LPAHs (Payne et al., 2003; Wolfe et al., 2001). Therefore, HPAHs are more likely to be transferred from fish tissue to mammals through the latter's diet than are LPAHs (Payne et al., 2003; Wolfe et al., 2001). Toxicity caused by PAHs is generally associated with highly toxic metabolites (Albers and Loughlin, 2003), so the transfer of metabolites rather than parent PAHs may result in some toxicity.

Although historical data of PAH toxicity in marine mammals is available (Albers and Loughlin, 2003), it is not clear whether deceased marine mammals found with high concentrations of PAHs in tissues were chronically exposed to PAHs, nor is it clear to what concentrations they were exposed, what the source of the PAHs was, or whether they were exposed to various other chemicals at the same time (as a mixture) (Albers and Loughlin, 2003).

One component in each of the Corexit® dispersants is potentially volatile (i.e., petroleum distillates in Corexit® 9500 and 2-butoxyethanol in Corexit® 9527) (Table 2) and may become volatile soon after application. Exposure of mammals to toxic volatile chemicals through inhalation of dispersed oil is expected to be less than exposure through inhalation of oil alone, because volatile components in oil are effectively dispersed into the water column (Section 1.2.2; NRC, 2013). Volatilization may be reduced through increased dispersion and dilution of volatile chemicals into the water column (NRC, 2013); this represents another trade-off in toxicity between protecting species that breathe air (e.g., mammals and birds) and protecting those that do not surface to breathe (e.g., fish).

3.1.2.4 Invertebrates

Invertebrates are known to bioaccumulate hydrocarbons and PAHs (Boehm et al., 2004; Meador, 2003), which can lead to narcosis (Logan, 2007). Early-life-stage exposures to oil (including PAHs) can lead to developmental impacts, reduced growth, and death (Lee, 2013; Lonning and Falk-Petersen, 1978; Falk-Petersen et al., 1983; Albers and Loughlin, 2003). Exposure to oil can also lead to localized lesions on organ tissues (Brown, 1992), although it is unclear whether lesions in invertebrate species would have an impact at the population level that would, in turn, indirectly impact ESA-listed species by significantly reducing their prey base (i.e., invertebrates). Various other effects have been noted, including reduced respiration and movement (related to physical smothering), cytotoxicity and cytogenotoxicity, and altered feeding and excretion (Suchanek, 1993). These sublethal impacts may lead to mortality, but it is unclear whether, in an oil dispersion situation, PAH concentrations would be high enough, or exposures to PAHs sufficiently long, to cause such impacts on a broad scale (i.e., in a large enough area to reduce the prey base of ESA-listed or candidate species).

Measured toxicities of dispersed oil and dispersants alone to invertebrates are discussed at length in Section 3.2; sensitivities are modeled in Section 3.3. It has been commonly noted that dispersants are less toxic than oil alone, but that dispersed oil is more toxic than oil alone (Fingas, 2008; NRC, 2005);[19] therefore, the addition of dispersants is typically considered a direct threat to pelagic invertebrates and fish, and an indirect threat to mammals, birds, and reptiles. An example of such impacts on a planktonic community is presented by Jung et al. (2012), who observed greater impacts in a mesocosm study after dispersants had been applied to oil (relative to oil alone). Similarly, Scholten and Kuiper (1987) observed impacts on planktonic communities relating to the bioavailable fraction of oil; they warned against the use of dispersants, which enhance the dissolved (and therefore bioavailable) fraction of hydrocarbons in the water column. Many invertebrates, particularly during larval life stages, are found in shallow water, where they are exposed to high concentrations of oil and dispersed oil during a spill event. Acute mortality in the vicinity of the dispersed spill may occur in many sensitive species (French-McCay, 2010; Scholten and Kuiper, 1987; Stige et al., 2011), but widespread mortality will result from the uncontrolled spread of an oil spill (i.e., associated with baseline condition) (Abbriano et al., 2011).

Historical applications of dispersants have shown that planktonic species are increasingly exposed to oil after dispersant application (Lee, 2013), that such exposures may result in decreased growth and reproductive capabilities (Lee, 2013), and that these species may be at greater risk under natural conditions due to photo-enhanced toxicity (Barron et al., 2008). These are points of uncertainty that have not been incorporated into the analysis provided in Section 3.3. Uncertainties are described in further detail in Sections 6.2 and 6.3.1.

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[19] This position is brought into question in Sections 3.3 and 3.4 when considering the available, relevant, and comparable acute toxicity data (Attachment B-1). See Figures 8 and 9 for a clear comparison of the SSDs for dispersants, oil, and dispersed oil. The analysis presented in Sections 3.3 and 3.4 does not incorporate potential adverse impacts due to sublethal effects or photo-enhanced toxicity.

Ultimately, indirect impacts on prey species must be weighed against direct benefits to ESA-listed birds, marine reptiles, and mammals (i.e., reduced oiling of feathers and fur or other dermal contact and reduced ingestion, inhalation, and aspiration of crude oil). In the context of the survival of an ESA-listed or candidate species, the localized (i.e., in the area directly under a dispersed oil spill) mortality of quickly reproducing planktonic prey may be relatively unimportant compared to the possible mortality or impaired reproduction in a relatively slowly reproducing, geographically limited, and/or sparsely populated species of bird, marine reptile, or marine mammal.

It is possible that the addition of oil and dispersant to a natural system may cause a planktonic or benthic community to become dominated by species that are already present (i.e., to tolerant species) (Ortmann et al., 2012; Atlas and Hazen, 2011; Parsons et al., 1984), but such a shift may not result in an overall reduction in biomass (Varela et al., 2006) or a sustained impact (Abbriano et al., 2011), even in low-productivity environments (Cross and Martin, 1987). For that reason, it is not necessarily true that acutely lethal responses in sensitive species will result in significant reductions in the prey bases of listed or candidate species. This is particularly relevant for non-specific planktivores like baleen whales. It is less relevant for species that consume specific invertebrates that only exist as plankton during embryonic or larval life stages; examples of such species include bivalves, crab, some finfish, and many others.

Infaunal invertebrates in subtidal habitats exposed to a dispersed oil slick were found to be adversely affected relative to those in a similar shoreline that was exposed to a non-dispersed slick; but conditions returned to baseline within 2 years, and little difference was noted between the two shorelines thereafter (Cross and Thomson, 1987; Mageau et al., 1987; Humphrey et al., 1987). Behavioral responses (e.g., migrating out of sediment burrows to the sediment surface) and limited mortality were observed, but mass mortality of infaunal invertebrates did not occur during either the oil-only scenario or the dispersed oil scenario (Cross and Thomson, 1987; Mageau et al., 1987). Although hydrocarbon uptake did increase notably, particularly in filter-feeding species (e.g., bivalves), bivalve species metabolized or depurated the hydrocarbons within 1 year (Humphrey et al., 1987). It was noted that the immediate effects on infauna were not likely to have a long-term impact on populations (except in sensitive species) (Mageau et al., 1987), whereas untreated crude oil that reached the shoreline posed a long-term, chronic source of contamination for these species (Humphrey et al., 1987). Long-term (i.e., > 2 years) impacts were obvious in an echinoderm and a bivalve on the dispersed shoreline (Cross and Thomson, 1987). Peterson et al. (2003) observed long-term impacts on benthic invertebrates along oiled shorelines after EVOS, suggesting that removing oil from the ocean surface before it heavily oils shorelines may serve to protect these productive communities (Fingas, 2008).

Sublethal responses (e.g., reduced superoxide generation and phagocytic activity, as well as impairment of "several aspects of immune competence,"[20] indicating reduced immunosuppression) measured in invertebrate communities resulting from chronic exposures to oil (and PAHs in particular) are often temporary within a population, such that a community may return to pre-spill conditions within a matter of months or years (Edwards and White, 1999; Dyrynda et al., 2000). It is unclear whether temporary fluctuations in invertebrate populations will have a marked adverse impact on predator individuals (Section 6.4).

3.1.2.5 Marine reptiles

The impacts of oil on marine reptiles have been studied to a lesser extent than the impacts on other groups. Oil is known to cause mortality in sea turtles, as evidenced by strandings of dead individuals after DHOS (Barron, 2012) and other major oil spills. As with other species, this is likely related to PAHs in oil, which have been shown to significantly impact developing turtles (Albers and Loughlin, 2003; Van Meter et al., 2006). Other noted impacts include effects on respiration, skin, blood chemistry, and salt gland functioning (Albers and Loughlin, 2003). Turtles are especially susceptible to oil spills that foul nesting areas (ITOPF, 2011), which suggests that the baseline condition under consideration by this BA would pose a great risk to sea turtles if it were to occur in nesting areas. However, nesting does not occur in Alaska; rather the presence of marine reptiles in Alaska is considered "accidental or uncommon" (Section 3.4.4 of the BA).

Since PAHs are the primary cause of toxicity in marine reptiles, it may seem logical that an increase in PAHs resulting from the application of dispersants would result in greater toxicity. However, as discussed in Section 2, many factors in a field application of dispersants to an oil slick may mitigate such impacts, namely rapid dilution of an oil slick into the water column and removal of oil from the ocean's surface.

Another aspect of dispersion that is not described at length in the BA, but that is important to the assessment of sea turtles, is that dispersants are known to reduce the formation of buoyant tarballs (Shigenaka, 2003). It is speculated that the major route of oil exposure for adult sea turtles ingestion, particularly the ingestion of tarballs (Shigenaka, 2003); this is based on the facts that oil has been found in turtle stomachs following field exposure, turtles apparently do not avoid oiled waters (Shigenaka, 2003), and tarballs are known hazards for turtles (Shigenaka, 2003). It is therefore suggested that dispersant use would reduce the concentration of oil at the surface, and sea turtles' contact with it, or reduce the prevalence of tarballs that might be ingested incidentally by sea turtles. This conclusion was also reached by Shigenaka (2003), who noted that, prior to dispersant application, on-scene coordinators must take into account area contingencies (e.g., presence of eelgrass beds, depth of water column, presence of nesting habitat, etc.) in order to ensure the protectiveness of dispersion. It

[20] Quote taken from Edwards and White (1999)

is not suggested that oil dispersion will entirely mitigate the mortality of sea turtles, since observations during the DHOS event suggest the opposite (Barron, 2012).

It is also important to note that the only available study observing the impacts of dispersed oil on sea turtle embryos resulted in no adverse impacts (Van Meter et al., 2006); it was found that the percolation of oil through sediment in simulated nests resulted in a very low transfer of PAHs to the interior of the nest and eggs. It is still possible that the emergence of juveniles would result in exposure to those PAHs, but the bioavailability of PAHs in sediment would be significantly less than the bioavailability of dissolved PAHs initially in the water column (Albers and Loughlin, 2003). Exposure of adults to increased PAHs is not likely to result in acute toxicity, due to the rapid dilution and degradation of oil and its components after a dispersant application (Section 2). Also, reptiles are able to efficiently metabolize and excrete ingested hydrocarbons (Albers and Loughlin, 2003), which should limit the bioaccumulation of PAHs after a dispersant application.

Exposure of reptiles to toxic volatile chemicals through inhalation of dispersed oil is expected to be less than through inhalation of oil alone (NRC, 2013), even though at least one component of dispersants is volatile (i.e., petroleum distillates, 2-butoxyethanol) (Table 2). This is achieved through the dispersion of volatile chemicals into the water column, another trade-off in toxicity between protecting species that breathe air (e.g., reptiles) and protecting those that do not surface to breathe (e.g., fish).

The relatively low abundance of sea turtles in Alaska (Section 3.4.4 of the BA) and the potential reduction in the routes of exposure (i.e., ingestion of tarballs while foraging; inhalation or aspiration, ingestion, and oiling when surfacing to breathe) suggest that the application of dispersants may have a negligible or beneficial effect on marine turtles relative to the baseline condition.

3.2 Analysis of Oil, Dispersants, and Dispersed Oil Toxicities

The purpose of this section is to describe in detail the method for developing SSDs and HC5s for dispersants, crude oil, and dispersed oil as they relate to prey species of ESA-listed or candidate species. In some cases, data that are directly (i.e., species-level data) or closely (i.e., genus-level data) related to ESA-listed or candidate species are available. For example, Chinook salmon, coho salmon, steelhead (or rainbow trout [*Oncorhynchus mykiss*]), and Pacific herring toxicity data are all available, as are data from possible surrogates such as sockeye salmon (*Oncorhynchus nerka*) and Atlantic herring. Regardless, the majority of the data represent species that can be considered planktonic prey or early life stages of prey species (i.e., fish and invertebrate embryo, larvae, or juveniles).

3.2.1 Overview of toxicity data

The majority of the toxicological studies were conducted with established test species (e.g., mysids, daphnids, and inland silverside [*Menidia beryllina*]), which are sensitive to chemical perturbation, and are relatively short-lived (compared to cetaceans, for example). The majority of individuals were exposed at an early life stage, the goal being to observe the response in each species at its most sensitive stage of development. Such studies are conducted to determine the relative toxicity of a chemical (or a mixture) compared to other chemicals, or to address the relative sensitivity of many species or groups of species (i.e., genera) to a single chemical. Of the species tested, rainbow trout (which is not evolutionarily distinct from steelhead trout), Chinook salmon, coho salmon, and Pacific herring were the only protected or candidate species included in the calculations of HC5s; among these, only Chinook salmon had directly comparable oil and dispersed oil toxicity data.[21] All other test species are considered surrogates for the prey of endangered species, and are important when considering food web interactions that result from the chemical dispersion of oil. Potential food web interactions are discussed for endangered species identified in this BA, as applicable.

The criteria used for the development of SSDs are discussed below. The SSDs were created using reported acute aquatic toxicity data from the literature (Attachment B-1) to assess the relative toxicity of Corexit® 9500 and Corexit® 9527 to a number of model species. The HC5s reported are the concentrations of dispersants or dispersed oil below which no expected acutely toxic effects will occur in 95% of aquatic species. There are exceptions to this method of threshold derivation, which are discussed below. Emphasis was placed on Arctic, Alaska, or cold-water species, although these species were not disproportionately weighted in the determination of the HC5s. All species were treated equally in the calculations. Limiting the dataset to only the most relevant species would have resulted in too few tests to create meaningful SSDs for Corexit® 9500 and dispersed oils.

3.2.2 Toxicity data acceptability criteria for developing SSDs

Acute aquatic toxicity values were compiled from the literature available for dispersants and dispersed oil, as summarized in Attachment B-1. SSDs for each mixture were developed using the median lethal concentrations (i.e., concentration that is lethal to 50% of an exposed population) (LC50) for exposure durations of between 48 and 96 hours for all species, with continuous (i.e., static, static renewal, or

[21] Median lethal concentrations were directly comparable, in that the endpoints and exposure durations were the same, the species was the same, and the exposure scenario was the same. Furthermore, the oil types were the same: Prudhoe Bay Crude Oil (PBCO). Dispersed oil is less toxic than oil alone to Chinook salmon (Van Scoy et al., 2010; Lin et al., 2009; Moles et al., 1979 as cited in Barron et al., 2013).

flow-through) and spiked exposures.[22] Only 96-hour exposures were included for larval or juvenile fish, but 48-hour exposures were included for embryonic or embryolarval fish; only 4 data were included for 3 species (i.e., Atlantic menhaden [*Brevoortia tyrannus*], spot croaker [*Leiostomus xanthurus*], and red drum [*Sciaenops ocellatus*]).

Continuous exposures are the most common in the dataset (Attachment B-1), but spiked exposures are typically considered the most applicable to the use of a chemical dispersant in the field (Clark et al., 2001), assuming the dispersant is applied to a surface slick rather than a subsurface release (e.g., wellhead blowout). Spiked exposures result in non-specific durations of exposure, but are perhaps the most relevant to a real-world spill. Spiked exposures should result in realistic LC50 values for surface applications. Dispersant application to subsurface releases, such as occurred during the DHOS, are atypical, but not impossible. This type of application may be mimicked during toxicity testing by a continuous exposure scenario. For this reason, toxicity data using either exposure type is considered valid for the calculation of HC5s. The inclusion of such data does not greatly affect the calculation of protective HC5 values, because the lower SSDs (i.e., the most sensitive tests) are generally composed of constant exposures; spiked exposures often result in much higher LC50 values. The HC5s calculated in this appendix are similar to those reported elsewhere for oil or dispersants (Barron et al., 2013). Dispersed oil SSDs have not been previously developed, so no such comparison can be made for dispersed oil.

Aquatic plant and algae bioassays were included if they satisfied the other criteria for inclusion (i.e., mortality endpoint reported as LC50, 48- to 96-hour exposure). Plants were not obviously more or less sensitive to dispersants, so their inclusion in the HC5 calculations did not bias the distribution.[23] Lastly, both freshwater and saltwater species were used, particularly because of the availability of rainbow trout data. The inclusion of both types of species did not ultimately affect the HC5 values.[24]

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[22] Continuous exposures imply that the toxicant is cycled through the test chamber at a constant concentration, or added at appropriate intervals to ensure that significant degradation does not occur during the toxicity test. Spiked exposures imply that the toxicant is added once during the test and allowed to diminish over time (e.g., to degrade or evaporate).

[23] Exclusion of the plant species would not have resulted in the selection of a different best-fit model. Neither plant species was at the lower end of the distribution, and therefore did not affect the selection of the HC5.

[24] HC5s were calculated using both freshwater and saltwater species, and then omitting freshwater species. The calculated HC5 did not change, because the freshwater species tended to be less sensitive to dispersants or dispersed oil. The lower end of the SSD was composed of sensitive saltwater species.

3.2.3 Summary of acute lethality data for dispersants

3.2.3.1 Corexit® 9527

Acute toxicity data for 48- and 96-hour exposures to Corexit® 9527 were compiled from 48 tests on 34 species within 31 different genera. Specifically, for invertebrates and aquatic plants, toxicity tests that lasted only 48 hours were included, because these species tend to have shorter periods of development than fish. Only 96-hour toxicity test data were included for fish species, with the exception of embryo-larval tests using Atlantic menhaden, red drum, and spot croaker (Fucik et al., 1995; Slade, 1982). Spiked tests had non-specific exposure durations, but they are expected to be ecologically relevant (Clark et al., 2001). Of the tests conducted, 2 used plants, 28 used invertebrates, and 18 used fish species. The observed LC50s for all species were between 2.4 and 840 ppm or mg dispersant/L water. Only bounded data were included in the calculation of HC5s; unbounded values (e.g., LC50 > 1,000 ppm) were omitted. Tests were carried out under various temperatures, each assumedly appropriate to the test species; therefore, not all tests are entirely applicable to waters in Alaska. As applicable, Arctic and sub-Arctic Alaska species are identified and discussed below.

Invertebrate species had more varied LC50s than did fish or plants, likely due to the greater number of tests and test conditions conducted for invertebrates. Green hydra (*Hydra viridissima*) and grass shrimp (*Palaemonetes pugio*) were the least sensitive invertebrate species and least sensitive species, overall. Various crustaceans (*Allorchestes compressa, Pseudocalanus minutes, Penaeus setiferus*) and Pacific oyster (*Crassostrea gigas*) were the most sensitive invertebrates and most sensitive species, overall.

The majority of fish were less sensitive than invertebrates, and as sensitive as plant species. The range of LC50s for rainbow trout, the only tested species that can be considered endangered (i.e., Steelhead trout), was between 96 and 260 ppm Corexit® 9527 (Doe and Wells, 1978; Wells and Doe, 1976).

Two aquatic plant species were tested: a brown alga (*Phyllospora comosa)* and turtle grass (*Thalassia testudinum*). The 48-hour LC50 for the brown alga was 30 ppm (Burridge and Shir, 1995), and the 96-hour LC50 for turtle grass was 200 ppm (Baca and Getter, 1984).

3.2.3.2 Corexit® 9500

Acute toxicity data for spiked and 48- to 96-hour exposures to Corexit® 9500 were compiled from 48 tests with 26 species and 24 genera. Of the tests conducted, 26 used invertebrates and 22 used fish. The observed range of 48- to 96-hour LC50s was between 3.5 and 1,038 ppm, the highest values being for spiked exposures.

Invertebrates that were less sensitive to Corexit® 9527 included the green hydra and Eastern oyster (*Crassostrea virginica*). Sensitive species included the amphipod

(*A. compressa*), copepods (*Eurytemora affinis* and *Tigriopus japonicus*), and red abalone (*Haliotis rufescens*).

Fish were generally less sensitive to Corexit® 9500 than to Corexit® 9527. Of the fish tested, rainbow trout and red drum were the least sensitive; rainbow trout had a 96-hour LC50 of 354 ppm, and red drum had a spiked LC50 of 744 ppm. Other relatively insensitive species included the sheepshead minnow (*Cyprinodon variegatus*) and gulf killifish (*Fundulus grandis*). In addition some tests, but not all, indicated inland silverside to be relatively insensitive.

3.2.3.3 Corexit® toxicity to cold-water species

Most laboratory toxicity tests use temperate or warm-water species, warm exposure conditions (e.g., 20–25°C), and variable exposure scenarios or test types. There is a paucity of data representing those conditions more likely to be encountered by species of concern in Alaska waters. Recent tests by McFarlin et al. (2011) were conducted under conditions that would be observed during an oil spill response in Alaska. These tests incorporated cold-water temperatures, spiked exposures, and Arctic test species.

A second study was conducted by Ordzie and Garofalo (1981) with Corexit® 9527. Reported 6-hour LC50s were between 200 ppm at 20°C and 2,500 ppm at 2°C. This toxicity test was conducted using temperatures similar to those of Alaska waters and an appropriate exposure duration, but using a test species (a scallop [*Argopecten irradians*]), not present in Alaska. These values were excluded from the SSD due to the short exposure duration. However, it is important to note that this exposure duration (in addition to the exposure temperature) is ecologically relevant (Gallaway et al., 2012).

The following studies used species that may be present in Alaska, or tested species under conditions approximating the application of dispersant under Arctic field conditions:

- Clark et al. (2001) reported an LC50 of 13.9 ppm Corexit® 9527 for larval Pacific oyster using a spiked exposure system. The Pacific oyster is found in Alaska, although it is a non-native species primarily valued for aquaculture.
- Clark et al. (2001) determined a spiked LC50 of > 1,055 ppm Corexit® 9500 for turbot (*Scophthalmus maximus*), a fish present in the North Atlantic. This value is unbounded, and was therefore not included in SSD.
- Nalco (2005, 2010) determined 96-hour LC50s of 75 ppm Corexit® 9500 and 50 ppm Corexit® 9527 for turbot.
- Rhoton et al. (2001) reported an LC50 of 355 ppm Corexit® 9500 for larval tanner crab (*Chionoecetes bairdi*), an Alaska species, in a spiked exposure system.
- Duval et al. (1982; cited in NRC, 2005) reported a 96-hour continuous exposure LC50 of > 1,000 ppm Corexit® 9527 for the isopod *Gnorimosphaeroma oregonensis*,

which can be found in intertidal areas of Alaska. This value is unbounded, and therefore was not included in SSD.

- Hartwick et al. (1982; cited in NRC, 2005) reported a 96-hour LC50 of 100 ppm Corexit® 9527 for littleneck clam (*Protothaca stamiea*), an important aquaculture species that is present throughout nearshore and intertidal areas of the Gulf of Alaska (including the Aleutian Islands).
- Foy (1982; cited in NRC, 2005) reported 96-hour LC50s for four Arctic amphipod species—*Anonyx laticoxae, Anonyx nugax*, *Boeckosimus edwardsi*, and *Onisimus litoralis*—as well as an unidentified species within the genus *Boeckosimus*; all were exposed continuously to Corexit® 9527. The LC50s were as follows: > 140 ppm for *A. laticoxae*; 97 to 111 ppm for *A. nugax*; > 80 ppm for *B. edwardsi*; > 175 ppm for *Boeckosimus* sp.; and 80 to 160 ppm for *O. litoralis.* The same study reported 96-hour LC50s of < 40 and > 80 ppm Corexit® 9527 for fourhorn sculpin (*Myoxocephalus quadricornis*) and a copepod (*Gammarus oceanicus)*, respectively. Unbounded values were not included in the SSD.
- Rainbow trout 96-hour LC50 toxicity values were reported by Wells and Doe (1976; cited in NRC, 2005) and by Doe and Wells (1978; cited in NRC, 2005) as being between 96 and 293 ppm Corexit® 9527.
- George-Ares and Clark (2000) reported a 96-hour LC50 of 354 ppm Corexit® 9500 for rainbow trout.

Not all studies listed herein report the temperatures at which exposures were conducted. It can be assumed that all studies were conducted under conditions appropriate to the test species, such that temperatures were not outside the species' tolerable limits.[25] Exposures of Alaska species using temperatures higher than those typically observed in Alaska would likely result in an overestimate of toxicity, based on the findings of Ordsie and Garofalo (1981; cited in NRC, 2005), rather than an underestimate.

3.2.3.4 Sublethal or chronic toxicity of dispersants

Although sublethal and chronic toxicity data were not included in the calculation of HC5s, some data have been compiled; it is presented here for comparison to acutely toxic concentrations, as well as to identify known sublethal impacts. In a small number of studies, exposure to chemical dispersants has been shown to cause sublethal or chronic[26] toxic responses. Singer et al. (1991) reported a concentration at which 50% of

[25] This assumption is based on the use of a negative control treatment in each study that indicated the health or condition of the test species under the given test conditions.

[26] Chronic responses are those following exposure of a duration that includes a notable portion of a species' entire life cycle or early life stages. The duration is characteristically longer than acute exposures, and endpoints often include sublethal effects that are slow to manifest and continual (e.g., abnormal growth).

the number of exposed organisms were affected (EC50) of 13.6 ppm Corexit® 9527, based on abnormal growth in red abalone after a 48-hour exposure to spiked concentrations. Nalco (2010) reported a 72-hour reduced biomass EC50 of 9.4 ppm Corexit® 9527 for the diatom *Skeletonema costatum* when it was continuously exposed. The bioluminescent marine bacterium *Vibrio fischeri* was observed to have a reduced bioluminescence EC50 of 104 ppm Corexit® 9500 (NRC, 2005) after a 15-minute exposure; reduced bioluminescence is an indication of lowered metabolic activity. The 15-minute *V. fischeri* bioassay is considered a chronic test because of the bacterium's very short life span. Mitchell and Holdway (2000) reported chronic, 7-day no-observed-effect concentration (NOEC) values of 13 and < 15 ppm for green hydra exposed (static, daily renewal) to Corexit® 9527 and Corexit® 9500, respectively. Other studies found that dispersants inhibited reproduction (Singer et al., 1991), growth, development (Singer et al., 1991; Wells et al., 1982), and other endpoints (Gulec et al., 1997; Norwegian Institute for Water Research, 1994; Burridge and Shir, 1995; all cited in NRC, 2005) in various species (e.g., giant kelp [*Macrocsytis pyrifera*], amphipods, diatoms, mysids, and red abalone) when these species were exposed over a relatively long period of time.

Very short-lived species are also briefly discussed in this appendix. The 48-hour time-to-molt EC50 for *Artemia* sp. (42 ppm) and the 72-hour biomass production EC50 for *S. costatum* (9.4 ppm) are within the range of LC50s for Corexit® 9527 (i.e., from 2.4 to 840 ppm). Similarly, the *V. fischeri* chronic 15-minute bioluminescence EC50 (104 ppm) and the 72-hour biomass production EC50 for *S. costatum* are within the range of acute LC50s for Corexit® 9500 (i.e., from 3.5 to 744 ppm).

3.2.4 Summary of acute lethality data for crude oil

A number of studies were compiled to characterize the toxicity of oil alone in an aquatic system. Oil toxicity data represent exposure durations between 48 and 96 hours with established test species. The same assumptions and limitations that applied to the dispersant toxicity data (Section 3.2.3) apply to this dataset. However, the interpretation of this dataset is less straightforward, because additional variables exist when dealing with oil, which is a complex mixture. In order for a definitive statement to be made regarding the change in toxicity due to the application of dispersants, it is important to establish the toxicity of crude oil relative to that of dispersants and dispersed oil.

Lacking a singular source or composition, oil is expected to elicit variable acute responses in ecological receptors. More specifically, different types of oil have different fractions of toxic components, such as PAHs (Ramachandran et al., 2004). In addition, degrees of weathering are included in the dataset; a single oil type can be either fresh or weathered, depending on the time the oil has spent exposed to natural conditions (e.g., ultraviolet radiation, wind and water, biodegradation, and evaporation). Weathered oil tends to have fewer bioavailable components due to the volatilization and biodegradation of its lighter (and typically more acutely toxic) constituents (NRC, 2005; 2003b as cited in NRC, 2005; 2003a). This was a particular point of study by Barron et al. (2013), who developed SSDs and reported HC5 values for different oil types; HC5 values ranged from 0.285 to 3.53 ppm TPH, depending on the type of oil.

Unlike the toxicity datasets for dispersants or dispersed oil, the majority (56%) of species tested with oil alone were cold-water species. A total of 134 tests were conducted; 73 tests were conducted on invertebrates, and 61 tests were conducted on fish. A total of 59 species were tested, of which 34 were invertebrates and 25 were fish. A total of 45 genera were tested, of which 27 were invertebrates and 18 were fish. Approximately half of all the species tested (as well as within the groups of species or genera) are found in cold-water environments. Not all tests with cold-water species were conducted under cold-water conditions, but it is assumed that the exposure conditions were appropriate (i.e., tolerable range of temperatures) for the species.[27]

Two warm-water invertebrates (*Palaemon serenus* and *A. compressa*) and one warm-water fish (Australian bass [*Macquaria novemaculeata*]) were found to have 96-hour LC50 values between 258,000 and 465,000 ppm TPH; these three LC50 values are more than three orders of magnitude greater than the fourth-least sensitive species (*T. japonicus*), and more than four orders of magnitude greater than the fifth-least sensitive genera (*Platichthys*). The four highest LC50 values (i.e., *P. serenus, A. compressa, M. novemaculeata,* and *T. japonicus*) were confirmed as outliers using the Interquartile Range (IQR) method.[28] When developing the SSD, two distributions were fit using the entire dataset, excluding the upper three data points.[29] The removal of the three highest data points resulted in the selection of a distribution that fit the entire dataset better, both visually and statistically (based on the Anderson-Darling statistic). Therefore, the statistical distribution was fit to the empirical SSD with the three highest LC50 values omitted to minimize (i.e., improve) the best-fit statistic and more realistically predict values at the lower end. It is un clear, based on the studies available (Gulec and Holdway, 2000; Gulec et al., 1997), why the LC50 values are so much higher than those of other similar exposures.

After removing the three highest LC50 values, the least sensitive invertebrates were the copepod *T. japonicus* and a polychaete worm, *Platynereis dumerilli*. Insensitive fish included flounder (*Platichthys* sp.) and topsmelt. Sensitive invertebrates included pale octopus (*Octopus pallidus*), black chiton (*Katharina tunicate*), Alaska shrimp (*Crangon*

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[27] This assumption is validated by the use of a negative control during toxicity testing. The control indicated the condition of the test species under the given exposure conditions.

[28] Outliers are defined according to the range between the 25th and 75th percentiles of the dataset (or the IQR), such that values that are greater than 1.5 or 3 times the IQR plus the 75th percentile value are considered outliers. The method also applies to low outliers that are less than 1.5 or 3 times the IQR below the 25th percentile.

[29] Removal of the 4th highest data point resulted in no change in the best-fit distribution selected or the calculated HC5.

alaskensis), and green hydra. The range of LC50 values at the genus level was between 0.39 and 124.3 ppm (excluding the values between 258,000 and 465,000 ppm). These values (e.g., 0.39 to 124.3 ppm) are somewhat similar to those reported for dispersed oils (Section 3.3), although the SSDs and HC5s calculated in this appendix (Sections 3.3 and 3.4, Tables 3 through 5, and Figures 8 and 9) suggest that oil is more acutely toxic than dispersed oil. This finding is consistent with much of the literature, although contrary to what has been suggested in past literature reviews (Fingas, 2008; NRC, 2005) and many toxicity studies (Attachment B-1).

3.2.4.1 Sublethal or chronic toxicity of crude oil

Smit et al. (2009) synthesized chronic exposure data and developed an SSD of chronic or sublethal endpoints (i.e., DNA damage; oxidative stress; and reduced survival, growth, and reproduction, or "whole-organism" responses). The data compiled by Smit et al. (2009) will be briefly discussed here.

The most sensitive species to DNA damage were blue mussel (*M. edulis*) and green sea urchin (*Strongylocentrotus droebachiensis*), with chronic 210-day LOECs of 2.8 and 4 ppb TPH, respectively. Iceland scallop (*Chlamys islandicus*) was the most sensitive to oxidative stress, with a chronic 30-day LOEC of 2.3 ppb TPH. Blue mussel was the most sensitive to whole-organism responses, with a 33-day chronic reproductive NOEC of 30 ppb TPH.

Sheepshead minnow was the least sensitive to DNA damage, with a 21-day chronic LOEC of 100 ppb TPH; blue mussel and Atlantic cod (*Gadus morhua*) were the least sensitive to oxidative stress, with a chronic 30-day LOEC of 63.4 ppb TPH and sublethal 3-day LOEC of 69.4 ppb TPH. Longnose killifish (*Fundulus similis*) was the least sensitive to whole-body responses, with a chronic 8-day NOEC of 9,900 ppb TPH.

HC5 values for different groups of endpoints were between 1.4 and 70.5 ppb TPH; 70.5 ppb TPH, the HC5 for whole-body responses, was identified as the maximum allowable threshold for chronic exposures of aquatic life (based on various fish and invertebrates). This chronic threshold is approximately 15% of the HC5 calculated for oil alone based on acute toxicity (Section 3.3).

3.2.5 Summary of acute lethality data for dispersed oil

A number of studies were compiled to characterize the toxicity of dispersed oil in an aquatic system. Dispersed oil data represent exposure durations between 48 and 96 hours with established test species. The same assumptions and limitations applied to dispersant toxicity data (Section 3.2.3) apply to this dataset. However, the interpretation of this dataset is less straightforward due to the complex nature of oil (Section 3.2.4), as well as the varied interaction of dispersant chemicals with different types of oil (Fingas, 2008).

3.2.5.1 Corexit® 9527-dispersed oil

Acute values used in the calculation of SSDs for dispersed oil were based on the minimum calculated spiked or 48- to 96-hour LC50 of exposure. This dataset is the smallest of those presented in this appendix, particularly as regards the number of species represented (n = 12), those that can be considered cold-water species (n = 2), and those that are ESA listed (n = 0). Corexit® 9527-dispersed oil data were available for 29 tests with 13 different species, each from a different genus. Of the tests performed, 8 were conducted with fish (5 different species), and 21 were conducted with invertebrates (8 different species). LC50s ranged from 0.74 to 75 ppm Corexit® 9527-dispersed oil, analyzed as TPH.

LC50s from tests spiked with Corexit® 9527-dispersed oil (n = 11) ranged from 1.8 to 111 ppm. Pacific oyster, a cold-water species, had a spiked LC50 between 1.92 and 2.28 ppm dispersed oil (depending on the oil type). Data from 7 static renewal tests were available, with LC50s ranging from 0.74 to 28.5 ppm.[30] Constant exposure 48- to 96-hour LC50s ranged from 0.11 to 75 ppm; excluding the maximum value for this exposure type (75 ppm), all other values were ≤ 1.09 ppm.

3.2.5.2 Corexit® 9500-dispersed oil

Corexit® 9500-dispersed oil data were available for 51 tests with 18 different species, each from a different genus. Of these, 28 tests were conducted with fish (9 different species) and 23 with invertebrates (9 different species). The range of LC50s was from 0.186 to 155.9 ppm as TPH. The species geometric mean LC50s used to develop the SSD were between 1.37 and 76.0 ppm.

LC50s from 27 spiked tests conducted with Corexit® 9500-dispersed oil ranged from 2.84 to 72.6 ppm. Clark et al. (2001) reported LC50s between 0.81 and 3.99 ppm dispersed oil for spiked exposures of Pacific oyster; a single LC50 of 48.6 ppm dispersed oil was reported for turbot under the same exposure conditions.

LC50s from 24 tests using constant exposure (i.e., continuous, static, and static renewal) to Corexit® 9500-dispersed oil were in the range of 0.19 to 155.9 ppm, the highest value being for Chinook salmon, an ESA-listed species.

Five cold-water species or genera are represented in the dataset, three fish (sculpin [*Myoxocephalus* sp.], Arctic cod [*Boreogadus saida*], and Chinook salmon) and two invertebrates (Pacific oyster and *Calanus glacialis*). Cold-water species were the most insensitive to Corexit® 9500-dispersed oil, with the exception of Pacific oyster, which was relatively sensitive. McFarlin et al. (2011) reported LC50 values for three of the

[30] Static renewal is similar to a static exposure, in that the chemical is premixed with the exposure solution prior to testing. In a renewal test, the solution is periodically replaced with fresh solution; the result is an exposure scenario similar to a continuous exposure, such that the chemical remains relatively constant over the exposure period. It is not held constant throughout (i.e., continuous), nor is it allowed to degrade or partition without replacement (i.e., static, without renewal).

four relatively insensitive cold-water species (sculpin, *C. glacialis,* and Arctic cod), indicating that different methodologies may result in decreased toxicity. All three species were exposed to a spiked dispersed oil scenario in very cold water (2°C), whereas others (e.g., Pacific oyster) were exposed in warmer water (Clark et al., 2001; as cited in NRC, 2005).

The geometric mean 96-hour LC50 value for Chinook salmon exposed to Corexit® 9500-dispersed oil under constant conditions was approximately 76.0 ppm TPH. This is the only ESA-listed species for which toxicity data is available.

3.2.5.3 Sublethal or chronic toxicity of dispersed oil

The chronic and sublethal effects of dispersed oil have not been studied extensively. A study by Lee et al. (2011b) reported hatchability of Atlantic herring embryos exposed to Corexit® 9500-dispersed oil over a period of 2.4 to 336 hours. The chronic LC50s were time dependent and ranged from < 0.25 to 18 ppm for 336- to 2.4-hour exposures, respectively. In the same study, chronic 336-hour LC50s for Corexit® 9500-dispersed oil were between 1.75 and 1.94 ppm for Pacific herring, and between 2.03 and 4.33 ppm for Atlantic herring. Although these values are not represented in the SSDs for Corexit® 9500-dispersed oil, they have important implications for Pacific herring, which is a candidate for listing under ESA. Even under the short, ecologically-relevant exposure durations associated with the dispersion of surface spills, the concentration of dispersed oil caused embryotoxicity to Pacific herring. Pacific herring typically spawn in kelp beds in shallow areas, where severe oiling may occur under baseline conditions; concentrations of crude oil as low as 1.22 ppm TPH are sufficient to cause mortality in Pacific herring (Rice et al., 1979; cited in Barron et al., 2013), so this species may be adversely impacted under any condition that allows oil (dispersed or not) to enter spawning habitat. The application of dispersants is not intended for nearshore areas, but dilute dispersed oil may wash into such areas; thus, longer-term exposures within this range of LC50 values are possible, and Pacific herring could be adversely impacted by dispersants.

Ramachandran et al. (2004) reported 48-hour EC50s between 1.00E-7 and 6.60E-6 ppm (volume/volume) of Corexit® 9500-dispersed oil for rainbow trout. The endpoint was measured by the EROD enzyme activity bioassay, which can indicate general toxicant exposure at very low concentrations; EROD activity does not result from any sort of effect at the individual level (e.g., reduced growth, reproduction, or survival), although it implies that sublethal impacts caused by PAH metabolites may occur (Lee and Anderson, 2005). Concentrations required to cause acute, individual-level effects (i.e., reduced survival, growth, or reproduction) in salmon (using Chinook salmon as a representative) (Van Scoy et al., 2010; Lin et al., 2009) are more than eight orders of magnitude greater than those reported by Ramachandran et al. (2004).

3.3 SSDs and Calculation of HC5s for Dispersants, Oil, and Dispersed Oil

In order to assess the potential risk to plankton, invertebrates, and fish associated with dispersant application, SSDs were developed for simplified scenarios of exposure to Corexit® 9500 and Corexit® 9527, crude oil (including all oil types, weathered or fresh), and oil dispersed by the Corexit® products. This approach has been recently applied to similar datasets for crude oil, dispersants alone (Barron et al., 2013; Smit et al., 2009; de Hoop et al., 2011), and dispersed oil (Gardiner et al., 2012). The SSDs were developed using toxicological data from the literature, and HC5s were calculated from the lower (i.e., more sensitive) ends of the distributions for each mixture. The HC5 was chosen to represent a concentration that was protective of 95% of aquatic species (Barron et al., 2013).

LC50s for each species[31] were ranked according to increasing acute 48- to 96-hour LC50s (Table 2) for dispersants, and plotted on a logarithmic scale (Figure 3). Additional criteria for data acceptability were applied (Section 3.2.1.1). Similar data for dispersed oil are provided in Table 3 and Figure 4. The geometric mean of each species was used when multiple valid tests were available for a single species, and the geometric mean of a genus was used when data existed for multiple species within the same genus. If a single test was replicated for a single species in a single study, only the lowest LC50 (i.e., the most protective value) was included.

The distribution of empirical data was described using @Risk® software (Palisade Decision Tools, Version 6.1.1) as a Microsoft Excel® add-in. Distributions can take a number of theoretical forms (e.g., normal, logarithmic, etc.), so the best-fitting distribution (i.e., the distribution most like the empirical data from the literature) was used based on the Anderson-Darling statistic. This statistic is specifically useful for describing the ends of a distribution. It was also assumed that predicted LC50 values could not be less than 0 ppm. For crude oil, Corexit® 9500, and Corexit® 9527, a Pearson 6 distribution best described the empirical data. A log-logistic distribution best fit to Corexit® 9500-dispersed oil toxicity data, and a lognormal distribution best fit to Corexit® 9527-dispersed oil toxicity data.

The Latin Hypercube method was used to simulate 5,000 iterations of hypothetical data points from the selected distributions, which were then plotted and compared to the empirical datasets (Figures 3 through 9). The data simulated by @Risk® for each distribution was ranked from low to high, and the 250th value of 5,000 (i.e., the 5th percentile) was selected as the HC5.

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[31] The dataset of LC50 values was limited to exposure durations between 48 and 96 hours for invertebrates and 96 hours for fish; only juvenile or other early life stages of fish were acceptable, although adult life stages of small, short-lived invertebrates (e.g., kelp forest mysid [*Holmesimysis costata*]) were also deemed acceptable. All exposure types (e.g., static, flow-through, etc.) were included in the calculation of HC5.

Table 3. Summary of LC50 geometric mean values, best-fit distributions, and calculated HC5s for Corexit® 9500 and Corexit® 9527

Dispersant	Genus	Cold Water?	Genus Geomean LC50 (ppm)	Rank	Distribution selected in @Risk®	HC5 (ppm)
Corexit 9500	*Allorchestes*	no	3.5	1	Pearson 6	5.53
	Eurytemora	no	5.2	2		
	Tigriopus	no	10	3		
	Haliotis	no	12.8	4		
	Macquaria	no	19.8	5		
	Artemia	no	20.8	6		
	Litopenaeus	no	31.1	7		
	Acartia	yes	34	8		
	Chionoecetes	yes	44.6	9		
	Penaeus	no	48	10		
	Atherinosoma	no	50	11		
	Americamysis	no	50.4	12		
	Menidia	no	51.1	13		
	Scophthalamus	yes	74.7	14		
	Palaemon	no	83.1	15		
	Lates	no	143	16		
	Sarotherodon	no	150	17		
	Fundulus	no	155.4	18		
	Holmesimysis	no	158	19		
	Hydra	no	160	20		
	Crassostrea	yes	167	21		
	Cyprinodon	no	262.8	22		
	Oncorhynchus	yes	354	23		
	Sciaenops	no	744	24		

Dispersant	Genus	Cold Water?	Genus Geomean LC50 (ppm)	Rank	Distribution selected in @Risk®	HC5 (ppm)
Corexit 9527	*Allorchestes*	no	3	1	Pearson 6	7.18
	Pseudocalanus	yes	5	2		
	Crassostrea	yes	6.6	3		
	Macquaria	no	14.3	4		
	Holmesimysis	no	20.6	5		
	Acartia	yes	23	6		
	Americamysis	no	23.7	7		
	Litopenaeus	no	24.1	8		
	Phyllospora	no	30	9		
	Menidia	no	35.4	10		
	Atherinops	no	38.9	11		
	Leiostomus	no	40.9	12		
	Brevoortia	no	42.4	13		
	Artemia	no	46.0	14		
	Palaemon	no	49.4	15		
	Scophthalamus	yes	50	16		
	Sciaenops	no	52.6	17		
	Cyprinodon	no	74	18		
	Daphnia	yes	75	19		
	Callinectes	no	77.9	20		
	Onisimus	yes	80	21		
	Fundulus	no	89.5	22		
	Anonyx	yes	97	23		
	Platichthys	yes	100	24		
	Protothaca	yes	100	25		
	Oncorhynchus	yes	158.0	26		
	Corophium	no	159	27		
	Thalassia	no	200	28		
	Pimephales	no	201	29		
	Hydra	no	230	30		
	Palaemonetes	no	840	31		

HC5 – hazardous concentration, 5th percentile
LC50 – concentration that is lethal to 50% of an exposed population
ppm – parts per million

Table 4. Summary of LC50 geometric mean values, best-fit distribution, and calculated HC5s for crude oil alone

Genus	Cold Water?	Genus Geomean LC50 (ppm TPH)	Rank	Distribution selected in @Risk®	HC5 (ppm TPH)
Octopus	no	0.39	1	Pearson 6	0.46
Katharina	yes	0.44	2		
Crangon	yes	0.56	3		
Hydra	no	0.7	4		
Sciaenops	no	0.85	5		
Holmesimysis	no	1.11	6		
Pagurus	yes	1.14	7		
Boreogadus	yes	1.2	8		
Clupea	yes	1.22	9		
Cryptochiton	yes	1.24	10		
Melanotaenia	no	1.28	11		
Pandalus	yes	1.29	12		
Eualus	yes	1.32	13		
Capitella	yes	1.44	14		
Salvelinus	yes	1.49	15		
Oncorhynchus	yes	1.68	16		
Theragra	yes	1.73	17		
Aulorhynchus	yes	1.85	18		
Myoxocephalus	yes	1.89	19		
Farfantepenaeus	no	1.9	20		
Chlamys	yes	1.90	21		
Americamysis	no	1.91	22		
Thymallus	yes	2.04	23		
Paralithodes	yes	2.22	24		
Eleginus	yes	2.28	25		
Xenacanthomysis	yes	2.31	26		
Calanus	yes	2.4	27		
Cottus	yes	3	28		
Menidia	no	4.02	29		
Palaemonetes	no	4.60	30		
Neanthes	yes	4.82	31		
Spiochaetopterus	no	4.92	32		
Notoacmea	yes	5.32	33		
Leander	no	6	34		

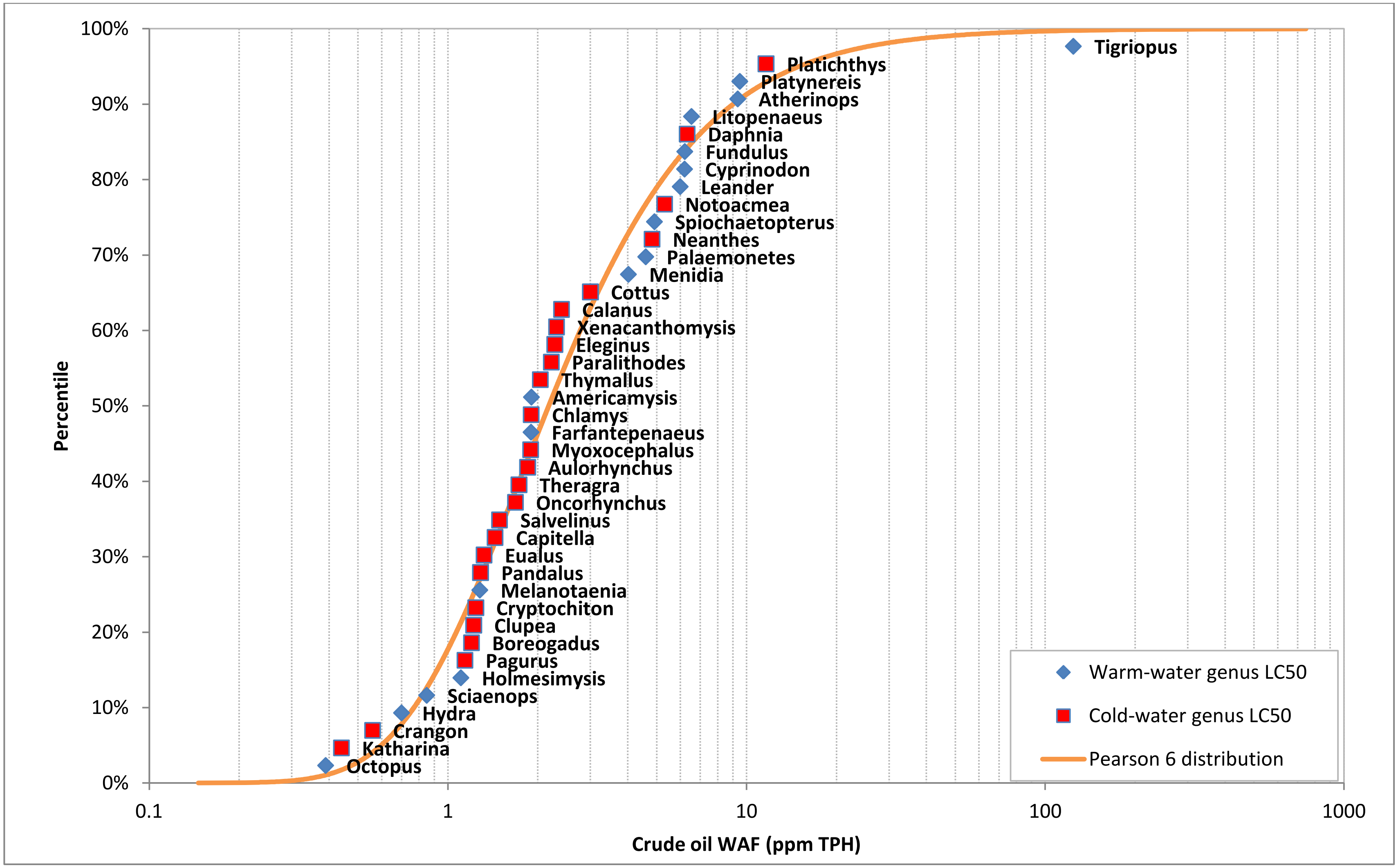
Genus	Cold Water?	Genus Geomean LC50 (ppm TPH)	Rank	Distribution selected in @Risk®	HC5 (ppm TPH)
Cyprinodon	no	6.21	35		
Fundulus	no	6.22	36		
Daphnia	yes	6.32	37		
Litopenaeus	no	6.54	38		
Atherinops	no	9.35	39		
Platynereis	no	9.5	40		
Platichthys	yes	11.62	41		
Tigriopus	no	124.3	42		
Palaemon	no	258,000	43		
Allorchestes	no	311,000	44		
Macquaria	no	465,000	45		

HC5 – hazardous concentration, 5th percentile
LC50 – concentration that is lethal to 50% of an exposed population
ppm – parts per million
TPH – total petroleum hydrocarbons

Table 5. Summary of LC50 geometric mean values, best-fit distributions, and calculated HC5s for Corexit® 9500- and Corexit® 9527-dispersed oil

Dispersant	Species	Cold Water?	Species Geomean LC50 (ppm TPH)	Rank	Distribution selected in @Risk®	HC5 (ppm TPH)
Corexit 9500	*Melanotaenia fluviatilis*	no	1.37	1	log-logistic	1.71
	Crassostrea gigas	yes	1.8	2		
	Palaemon serenus	no	3.6	3		
	Americamysis bahia	no	3.7	4		
	Sciaenops ocellatus	no	4.23	5		
	Menidia beryllina	no	6.2	6		
	Hydra viridissima	no	7.2	7		
	Holmesimysis costata	no	7.4	8		
	Litopenaeus setiferus	no	7.5	9		
	Tigriopus japonicus	no	10.7	10		
	Atherinops affinis	no	11.1	11		
	Macquaria novemaculeata	no	14.1	12		
	Allorchestes compressa	no	14.8	13		
	Myoxocephalus sp.	yes	17	14		
	Cyprinodon variegatus	no	18.6	15		
	Calanus glacialis	yes	20.5	16		
	Boreogadus saida	yes	45	17		
	Oncorhynchus tshawytscha	yes	76.0	18		
Corexit 9527	*Melanotaenia fluviatilis*	no	0.74	1	lognormal	0.69
	Crassostrea gigas	yes	1.03	2		
	Octopus pallidus	no	1.8	3		
	Holmesimysis costata	no	2.35	4		
	Menidia beryllina	no	2.55	5		
	Americamysis bahia	no	3.65	6		
	Palaemon serenus	no	8.1	7		
	Hydra viridissima	no	9	8		
	Daphnia magna	yes	15.28	9		
	Allorchestes compressa	no	16.2	10		
	Macquaria novemaculeata	no	28.5	11		
	Atherinops affinis	no	28.6	12		
	Platichthys flesus	no	75	13		

HC5 – hazardous concentration, 5th percentile
LC50 – concentration that is lethal to 50% of an exposed population
ppm – parts per million
TPH – total petroleum hydrocarbons



Note: The three highest LC50 values were removed, and the distribution was fit to the remaining points.

Figure 3. SSDs for crude oil water-accommodated fraction with the selected distribution fit to empirical toxicity data

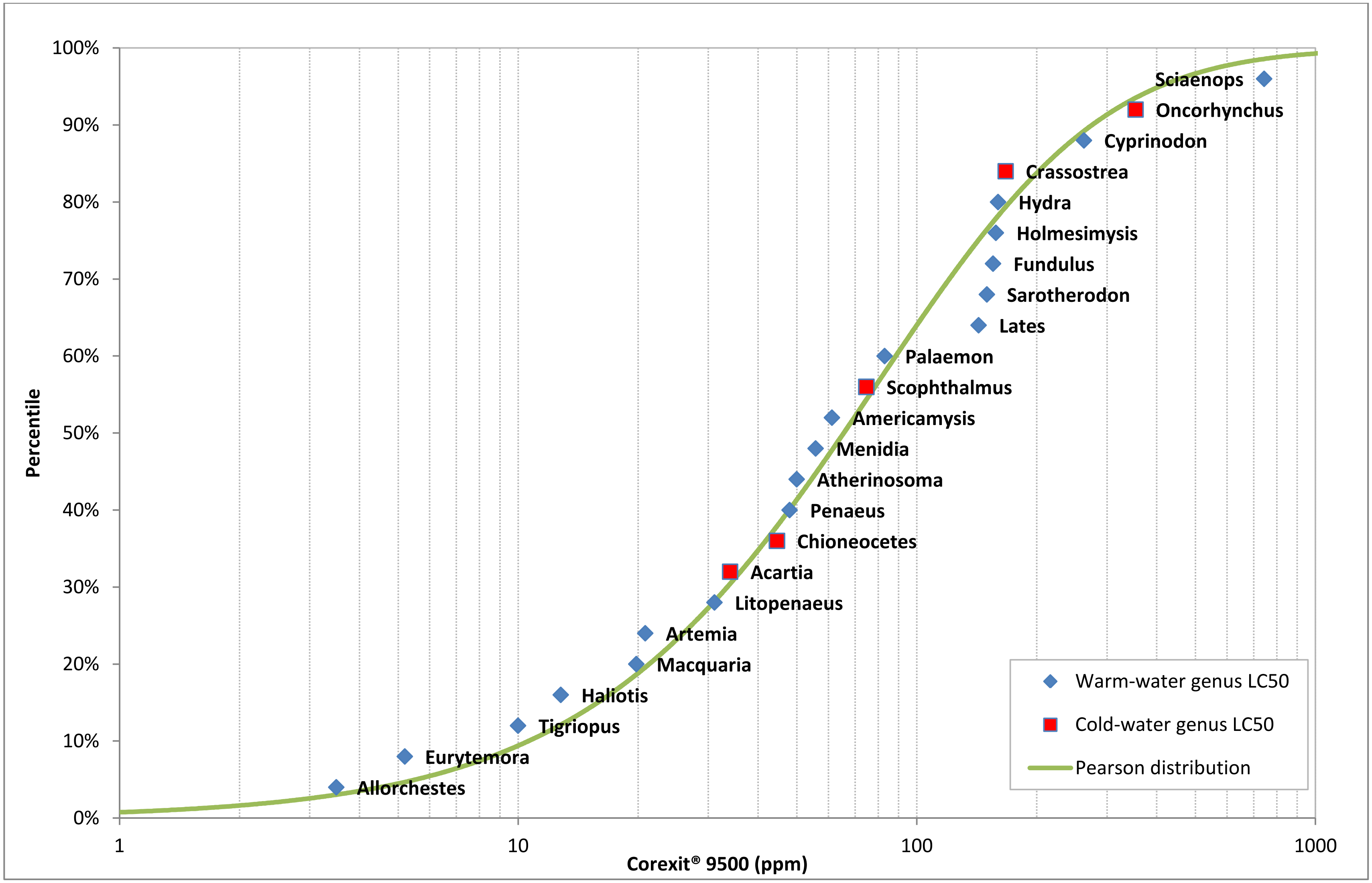


Figure 4. SSDs for Corexit® 9500 with the selected distribution fit to empirical toxicity data

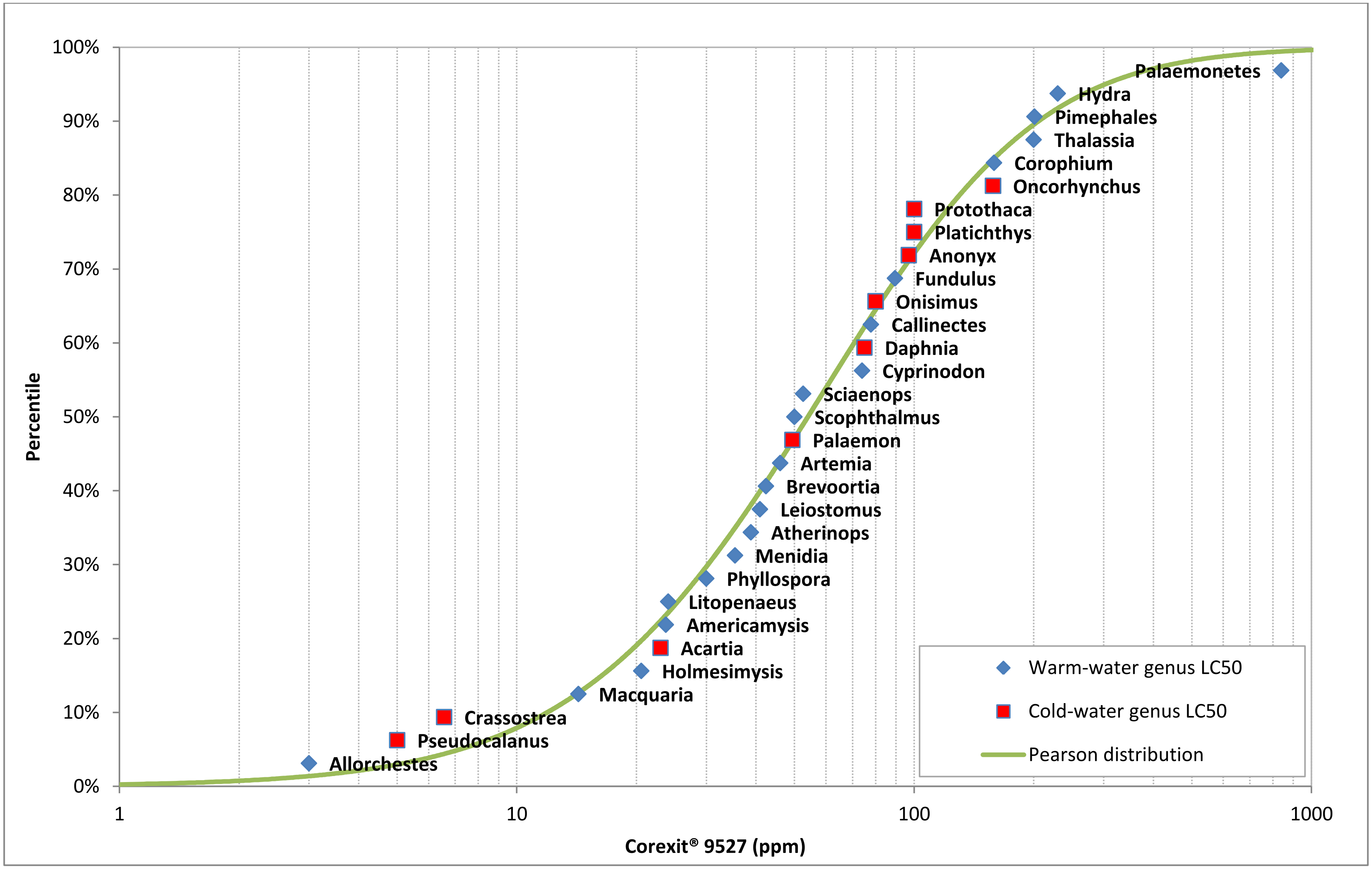


Figure 5. SSDs for Corexit® 9527 with the selected distribution fit to empirical toxicity data

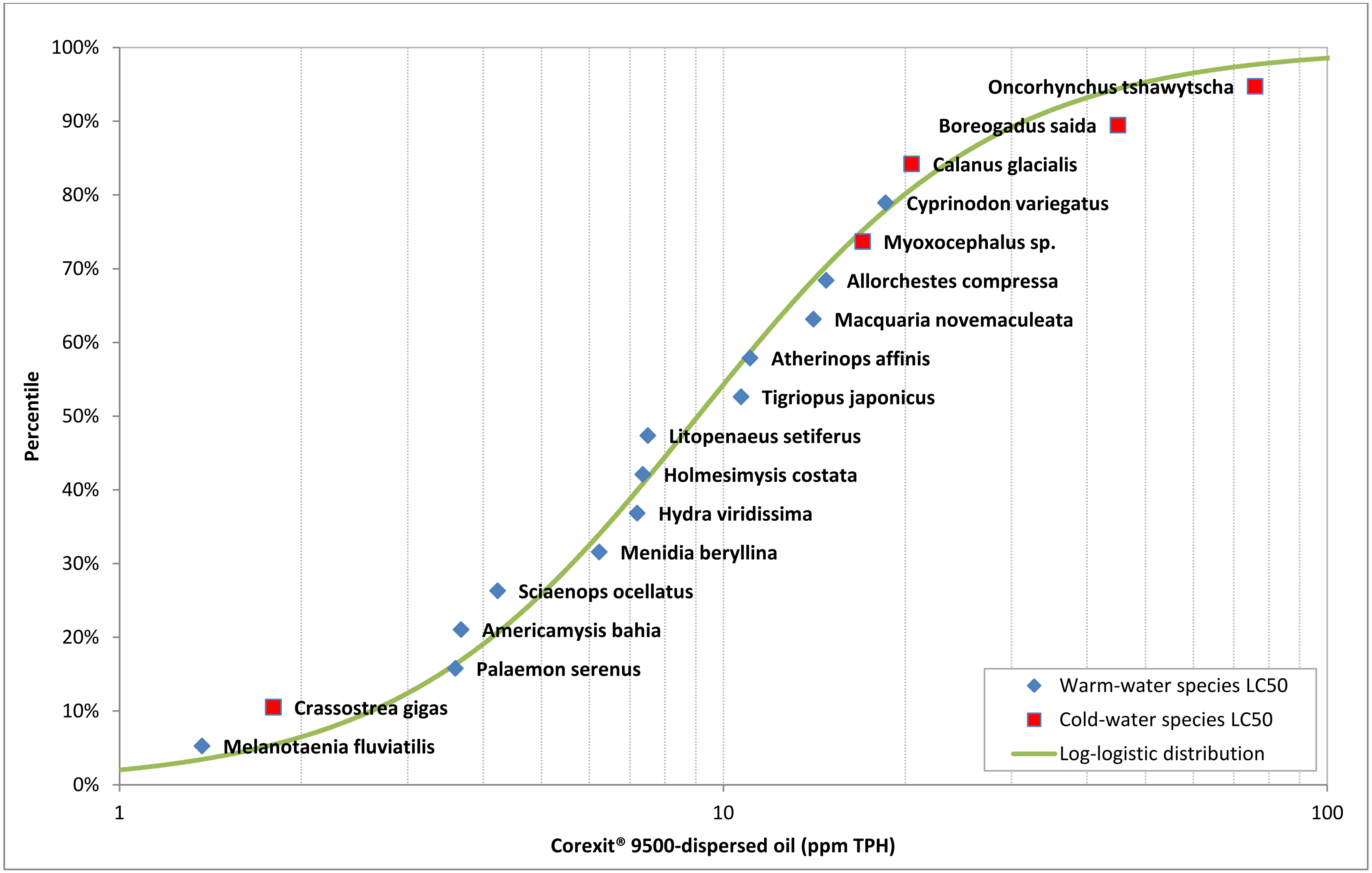


Figure 6. SSDs for Corexit® 9500-dispersed oil with the selected distribution fit to empirical toxicity data

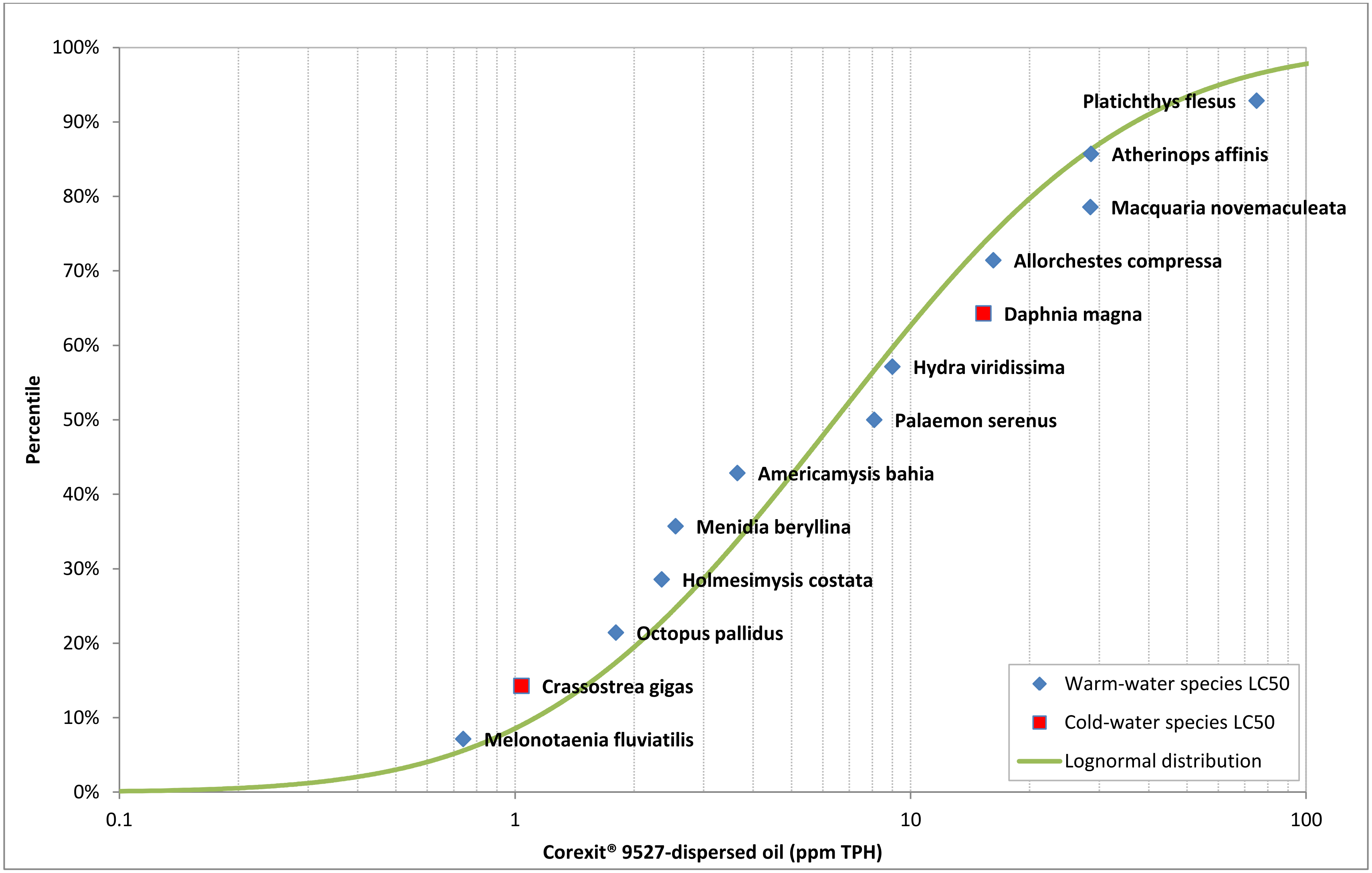


Figure 7. SSDs for Corexit® 9527-dispersed oil with the selected distribution fit to empirical toxicity data

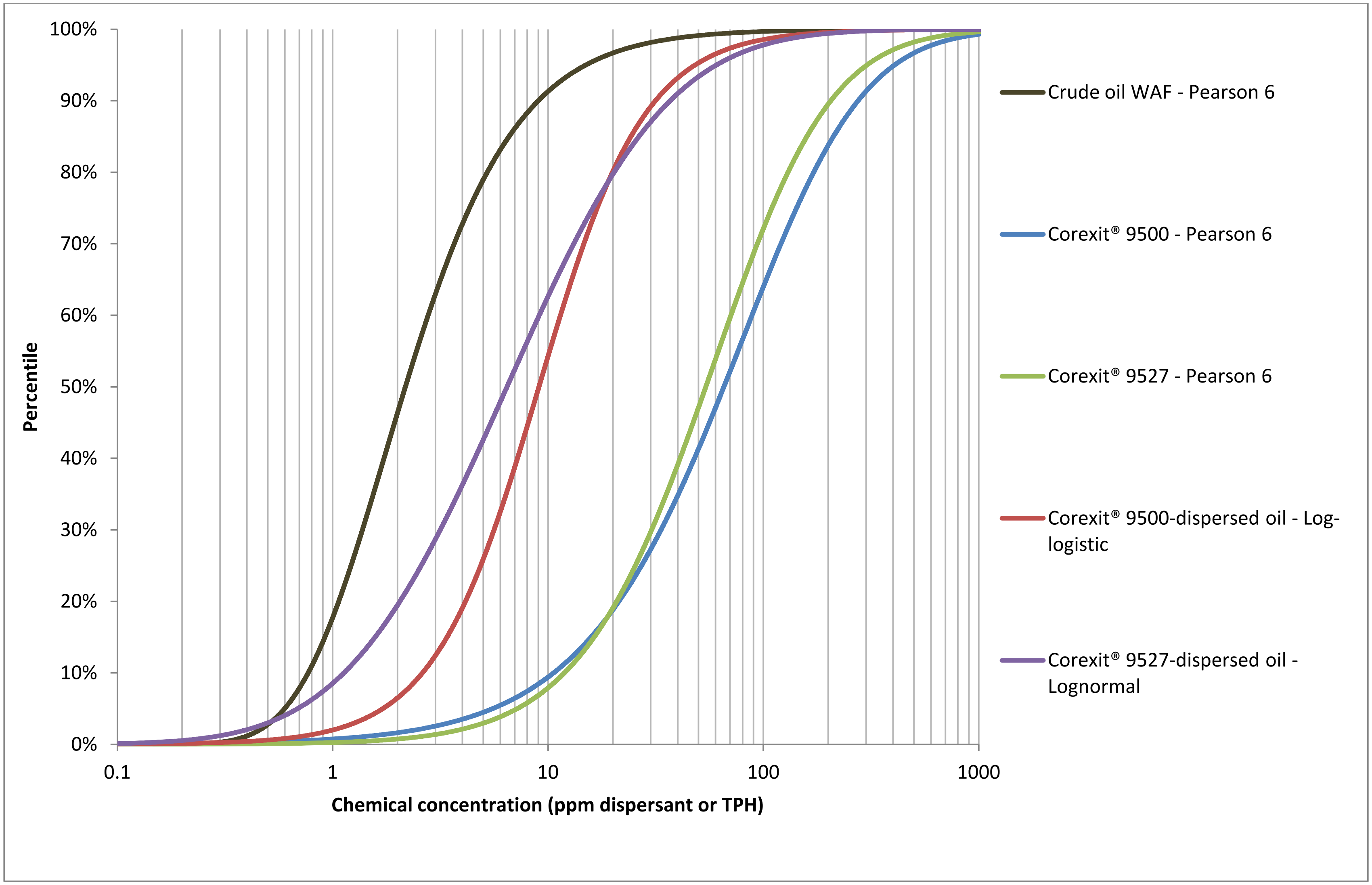


Figure 8. Comparison of selected distributions for multiple toxicity datasets

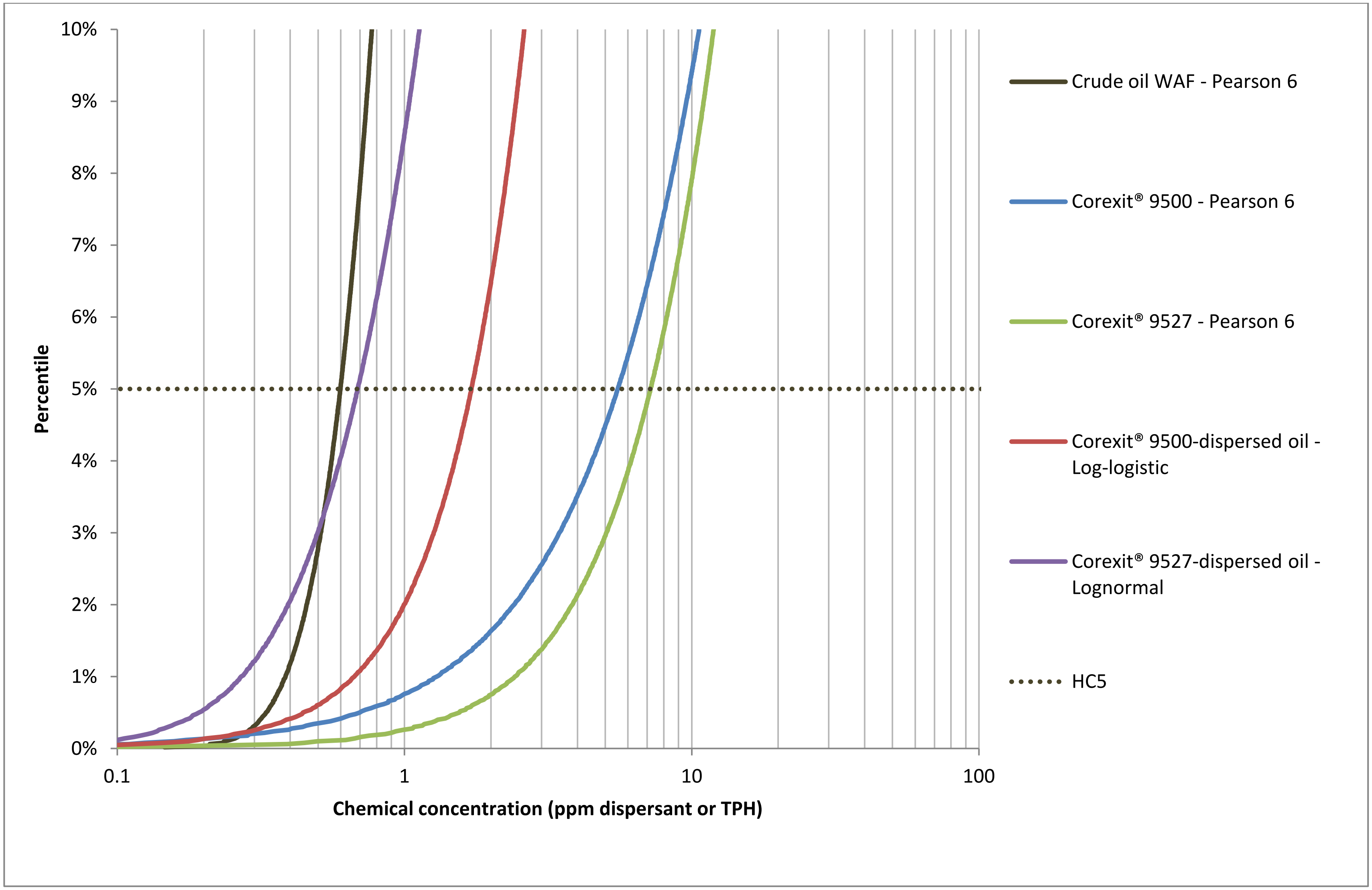


Figure 9. Comparison of selected distributions for multiple toxicity datasets, lower end with HC5 shown

The resulting HC5s for Corexit® 9500 and Corexit® 9527 were 5.53 and 7.18 ppm, respectively, indicating that the Corexit® 9527 appears to be less acutely toxic at the lower end (related to the HC5) of the SSDs than Corexit® 9500 (Figure 9). This finding runs contrary to what has been reported previously (NRC, 2005). However, Figure 8 shows that Corexit® 9527 is more acutely toxic at higher concentrations than Corexit® 9500, in accordance with the accepted view of the two dispersant formulations (NRC, 2005). The fact that the two SSDs appear to overlap can be explained by the similarities in the chemical composition of each formulation.

The crude oil HC5 was calculated as 0.46 ppm TPH. This value is similar to (i.e., within 20%) HC5 values reported by de Hoop et al. (2011), but low compared to those reported by Barron et al. (2013), except for No. 2 fuel oil (0.285 ppm TPH), which Barron et al. (2013) reported as lower. The HC5 calculated by Barron et al. (2013) for Bunker C was similar that calculated here for crude oil (i.e., 0.561 ppm TPH), but about 22% more. The HC5 reported by Gardiner et al. (2012) for Alaska North Slope (ANS) crude oil was similar to that reported here (within 5% for non-Arctic species), but lower than the HC5 for Arctic species (i.e., 0.80 ppm TPH). Variability in calculated HC5 values for crude oil can be explained by variability in oil types used (Barron et al., 2013) and species included Gardiner et al. (2012). Although de Hoop et al. (2011) report lower HC5 values for polar species than for temperate species, the differences were slight; Gardiner et al. (2012) reported a larger difference between cold- and warm-water species, but used fewer species to develop the SSDs than did de Hoop et al. (2011). It is not clear whether cold- or warm-water species are more sensitive to oil.

The Corexit® 9500-dispersed oil HC5 was 1.71 ppm TPH, and the Corexit® 9527-dispersed oil HC5 was 0.69 ppm TPH. The HC5s for Corexit® 9500-dispersed oil reported by Gardiner et al. (2012) were higher than that calculated here by factors of 1.52 and 4.91 for non-Arctic and Arctic species, respectively.

3.4 RELATIVE ACUTE TOXICITY OF OIL VERSUS DISPERSED OIL

The purpose of this section is to place the discussion of dispersed oil toxicity in the context appropriate for this BA. The toxicity of dispersed oil relative to the toxicity of oil alone is the primary concern that must be considered in order to provide a determination of effect for ESA-listed species. This is due to the fact that the exposure to and toxicity of oil, alone, represents the baseline condition against which dispersed oil toxicity and exposure must be compared. Neither the toxicity of dispersants compared to natural seawater nor the toxicity of oil alone compared to natural seawater are considered appropriate discussions for the BA.

Although many laboratory studies have shown that oil is more acutely toxic than or similarly toxic to dispersed oil (Section 3.3; Attachment B-1), dispersed oil is generally thought to be more toxic than oil alone (Singer et al., 1998; McFarlin et al., 2011; Ramachandran et al., 2004), because dispersants increase the solubility of the toxic components of oil (e.g., PAHs) (Wolfe et al, 1998, 2001; Ramachandran et al., 2004). Bioavailability is assumed to increase via the spatial redistribution of oil into the water column, the spread of the oil-water interface on the ocean's surface as droplets form, and the increased solubility of hydrophobic constituent components drawn into solution by surface active components and solvents in dispersants. The formation of oil droplets is facilitated by the surface active chemicals (i.e., surfactants) in dispersants (e.g., DOSS, Tween®80, Tween®85, and Span® 80) (Figure 1).

Although some studies have shown PAH concentrations in tissue and water to increase in the presence of dispersants (Yamada et al., 2003; Milinkovitch et al., 2011a; Ramachandran et al., 2004; Couillard et al., 2005; Faksness et al., 2011), others have shown that retention or net uptake of oil (as TPH) in tissue decreases (relative to oil alone) when the oil is dispersed (Wolfe et al., 2001; Mageau et al., 1987; Lin et al., 2009; Chase et al., 2013). Wolfe et al. (1998) showed a non-significant increase in uptake of an LPAH, and Milinkovitch et al. (2012) showed a lack of effects related to the increased uptake.

Other possible mitigating factors of acute toxicity include temperature (i.e., lower exposure at lower temperatures) (Lyons et al., 2011) and salinity (i.e., exposure decreases as salinity increases) (Ramachandran et al., 2006). Lin et al. (2009) note that dispersed oil droplets may be unavailable due to the creation of bulky, stable micelles (see "surfactant-coated oil droplet" in Figure 1) that encapsulate oil and render PAHs and other oil components non-bioavailable. This effect has been verified by others in biodegradation experiments with surfactants and PAHs (Volkering et al., 1995; Liu et al., 1995; Kim and Weber, 2003; Guha et al., 1998); PAHs have also been shown to partition to non-aqueous phases upon microbial degradation of non-ionic surfactants, again resulting in non-bioavailable forms of PAHs (Kim and Weber, 2003).

3.4.1 Relative acute lethal toxicity

The purpose of this section is to discuss all available acute toxicity data (Attachment B-1), without the limitations placed on data for inclusion in the SSDs. The available literature shows that chemical dispersants either increase or decrease the acute toxicity (i.e., lethality) of oil under laboratory conditions (Attachment B-1). Increased toxicity is generally associated with increased solubility of toxic PAHs or other hydrocarbons; decreased toxicity is often explained by variable oil chemical compositions, variable rates of oil and dispersant degradation, and the relatively low toxicity of dispersants alone (Pollino and Holdway, 2002). Fucik et al. (1995) speculated that the creation of oil droplets increased the rate of volatilization of the lighter toxic components of oil (NRC, 2005), but it has since been shown that volatilization is reduced after chemical dispersion due to the increased solubility of lighter volatile components (NRC, 2013).

A number of studies reported reduced toxicity associated with the application of chemical dispersants to oil; several studies that reported unbounded LC50 values for oil or dispersed oil are discussed here (though they were not included in the calculations of HC5 values).[32] Based on the entire dataset for comparable 46- to 96-hour acutely lethal LC50 values, approximately 54% of comparable studies had decreased toxicity when oil was dispersed, and approximately 46% had increased toxicity. Thus, contrary to popular opinion, it is slightly more likely that toxicity will decrease once dispersants have been applied.

The addition of Corexit® 9527 in spiked exposures increased toxicity in 75% of tests (n = 4), and the addition of Corexit® 9500 in spiked exposures decreased toxicity in 80% of tests (n=21)[33]. In static renewal exposures with Corexit® 9500, 64% of tests (n = 11) showed increased toxicity, and in static renewal exposures with Corexit® 9527, 75% of tests (n = 8) showed increased toxicity. Static tests without renewal have not been conducted extensively. Only one test for Corexit® 9500 and two for Corexit® 9527 have occurred with comparable LC50s for dispersed oil and oil, alone; all three tests resulted in decreased toxicity in dispersed oil treatments. In continuous exposures (i.e., flow through), 80% of tests with Corexit® 9500-dispersed oil showed increased toxicity, but 60% of tests with Corexit® 9527-dispersed oil showed decreased toxicity.

Based on the most applicable laboratory test results (using spiked or static exposure scenarios) for Corexit® 9500-dispersed oil and oil-only exposures, the use of chemical dispersants may decrease the acute lethality of oil. This is evidenced by the relative toxicity observed in 18 of 21 studies (Attachment B-1). Among the studies that reported comparable LC50 values for dispersed oil and oil alone, 60% of the tests conducted with Corexit® 9500-dispsersed oil (n = 38) showed reduced toxicity (Attachment B-1), indicating that, regardless of exposure conditions, toxicity may decrease more often than it increases with the use of dispersants.

The reported LC50s for ESA-listed fish (e.g., Chinook salmon) and larger invertebrate species (e.g., tanner crab, scallop) indicate that these species are less sensitive to dispersed oil than smaller species at early life stages (Figures 3 through 7, Tables 3 through 5, Attachment B-1). Ordzie and Garofalo (1981) showed that exposures under

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[32] Only the lowest LC50 values reported in studies for each endpoint were used for this discussion. Note that some unbounded values are included in this section as well. If an unbounded LC50 indicates a range that excludes the other LC50 to which the first is compared, then it can be said to be more or less toxic, depending on the circumstance. For example, Singer et al. (1998)Singer et al. (1998)Singer et al. (1998) reported a 96-hour LC50 for a spiked exposure of kelp forest mysid as > 25.45 ppm oil and equal to 10.54 ppm for Corexit 9527-dispersed oil; because the range of possible LC50 values greater than 25.45 ppm excludes the value 10.54 ppm, the latter value can be said to be more toxic. Note that SSDs and calculated HC5s exclude unbounded values that are not appropriate for that specific type of analysis.

[33] The majority of these studies were conducted by McFarlin et al. (2011). Where unbounded LC50s were reported for "water-accommodated fractions" of oil, "breaking water-water-accommodated fractions" were used. These tests used oil that had been vigorously mixed into exposure water prior to exposures. Excluding this study (which was methodologically different than the others), the percentage of tests indicating decreased toxicity after Corexit® 9500 application is 66.67% (n = 9).

Arctic conditions (i.e., 2°C) may result in lower toxicity (in scallop) at relevant dispersed oil concentrations in the water column (i.e., up to 28 ppm dispersed oil immediately after application), particularly during short exposures (i.e., 6 hours) within the initial period of dilution (Mackay and McAuliffe, 1988; Nedwed, 2012; Gallaway et al., 2012).[34]

3.4.2 Relative sublethal toxicity

The data available for sublethal toxicity are very limited. Three tests with Corexit® 9500-dispersed oils (i.e., Terra Nova, Mesa, and Scotian light crude oils) were available for a single species (rainbow trout) (Ramachandran et al., 2004). Dispersants increased the exposure in all three of these tests, as indicated by the induction of cytochrome P4501A and measured using the EROD enzyme activity bioassay (Ramachandran et al., 2004). After the oil was treated with Corexit® 9500, EC50s decreased by factors of 5.91 to 1,116. It should be noted that these tests were conducted under laboratory conditions with closed systems and a static-renewal exposure scenario, both of which may overestimate the exposure of test species to dispersed oil under expected field conditions.[35] Also, EROD activity is a biomarker of exposure and does not necessarily indicate an adverse effect.

Four tests comparing Corexit® 9527-dispersed oil and oil alone were available. A study by Singer et al. (1998) tested Corexit® 9527 and red abalone larval shell abnormalities, as well as initial narcosis in topsmelt and kelp forest mysid. In the abnormal growth assay, EC50s for dispersed oil (17.81 to 32.70 ppm) were less (i.e., more toxic) than concentrations for oil alone (33.58 to 46.99 ppm, measured as total [C_7-C_{30}] hydrocarbons); however, toxicity decreased in the initial narcosis bioassays. A second study (Mitchell and Holdway, 2000) showed changes in the modeled population growth rate of green hydra. Over a period of 168 hours, the toxicity of the oil increased after dispersant had been added. The mortality endpoint for green hydra measured during the same study indicated that oil alone was more acutely toxic than dispersed oil.

3.5 Uncertainties Associated with the Application of HC5s

The data presented in Sections 3.2 and 3.3 and Attachment B-1 often do not consider ecologically-relevant exposure durations. This is a major shortcoming of the current analysis and those presented elsewhere (Barron et al., 2013; Smit et al., 2009; de Hoop

[34] This statement is based on the reported 6-hour LC50 values for *Argopecten irradians* (a scallop) of 1,800 and 2,500 ppm Corexit 9527-dispersed oil at 10°C and 2°C, respectively. The species was not impacted by oil alone, but was impacted by dispersants alone, suggesting that in this case, dispersants were driving toxicity.

[35] This statement assumes that exposed species are mobile rather than held within a plume. The former assumption is relevant for the test species, rainbow trout in question, but the latter condition is relevant for many planktonic species. In that case, exposures can be expected to increase, as observed by Ramachandran et al. (2004).

et al., 2011); however, the inclusion of less relevant data was necessary to develop meaningful SSDs from the available data. The use of spiked exposures is perhaps most relevant (for surface application), as discussed in Section 3.2.1.1; these tests were specifically investigated by Gardiner et al. (2012), who noted that dispersed oil was approximately 5 to 10 times less toxic than oil alone, and that Arctic species were less sensitive than non-Arctic species. Although analysis was limited by the number of available studies with Arctic species (n = 5), the results generally corroborated the findings presented in Section 3.3, specifically the comparison of crude oil and Corexit® 9500-dispersed oil.

Exposure durations in a real spill event are expected to vary by individual, species, and population or community. The dilution of oil and dispersant over time was discussed by Nedwed (2012) and Gallaway et al. (2012) and modeled by Mackay and McAuliffe (1988). Nedwed (2012) indicated that the rate of dilution of dispersed oil results in a concentration of dispersed oil < 10 ppm within minutes of application, approximately 1 ppm within hours, and in the parts per billion range (i.e., < 1 ppm) within one day. Previous measurements of immediate dispersed oil concentrations after dispersant application have been as high as 50 to 150 ppm (Belore et al., 2009), although usually lower (between 10 and 30 ppm) (Mackay and McAuliffe, 1988; McAuliffe et al., 1980; McAuliffe et al., 1981). However on average, over short time periods (i.e., 10 to 30 minutes after dispersant application), concentrations have been shown to be in only the parts per billion range (Mackay and McAuliffe, 1988),[36] suggesting that while instantaneous spikes in concentration may occur, dilution is rapid. Mackay and McAuliffe (1988) state, "the measured field exposures to C_1-C_{10} dissolved hydrocarbons from untreated and chemically dispersed crude oils are thus much lower (by a factor of 150 to 1 million) than those observed to kill a wide range of organisms in laboratory bioassays." When considering whether the increased concentration of dissolved hydrocarbons in the water column could cause "irreversible damage" to species that would otherwise not be exposed at depth to dispersed oil, Mackay and McAuliffe (1988) state that, "it appears that in many cases there is an adequate safety margin."

Other important uncertainties regarding the HC5s include the variety of exposure scenarios used in their development. Exposure temperatures, salinities, oil conditions (i.e., weathered or fresh), oil types, and species life stages all potentially contribute to variability in observed toxicity. For example, tests using different species exposed at different temperatures or salinities could result in different rates of ingestion, respiration, and depuration; an indirect example is provided by Venosa and Holder (2007), who observed that microbial activity in a single consortium slowed at colder temperatures. Fresh oils characteristically contain higher concentrations of small,

[36] MacKay and McAuliffe (1988) report these time-averaged concentrations as TPH (C_1-C_{10}), the lightest fraction of hydrocarbons and the most volatile. Other, less volatile fractions of hydrocarbons (e.g., C_7-C_{30}) may be expected to be concentrated under a dispersed oil plume also.

volatile, and more bioavailable hydrocarbons than weathered oil (Bobra et al., 1989; Rhoton, 1999; Singer et al., 2001; Rhoton et al., 2001); in this analysis, HC5s were calculated using results from either fresh or weathered oils. Similarly, different oil types or sources (e.g., ANS, Cook Inlet, and Prudhoe Bay) have different chemical compositions, and may illicit varying toxicity (Barron et al., 2013). Species life stage is known to affect toxicity testing, such that earlier life stages (particularly embryonic or larval life stages) tend to be much more susceptible to chemical intoxication. Attachment B-1 includes data from various literature reviews that did not explicitly state the life stage of the tested species, so the HC5 calculations may have inadvertently included mature life stage LC50s.

4 Synthesis of Fate and Transport, Exposure, and Toxicity Data

The purpose of this section is to synthesize the information provided in Sections 2 and 3, as well as information in Section 3 of the BA. The likely exposures of groups of species and their relative sensitivities to dispersants and dispersed oil are discussed to assess the likelihood of physical or toxicological impacts. Oil toxicity is discussed only in relation to the baseline condition. Species-specific discussions are provided in Section 5.

4.1 Likelihood of Physical Effects

Based on the available dispersant application guidelines for response actions in Alaska (Alaska Clean Seas, 2010; Nuka Research, 2006 [STAR]) and the life histories and behaviors of the wildlife addressed by this BA (Section 3 of the BA), it is unlikely that the bird and mammal species protected under the ESA would be directly exposed to undiluted dispersants as a result of a spill response action. This will limit potential physical impacts on birds and furbearing mammals (e.g., sea otter and polar bear), such as reduced thermoregulation of feathers or fur (Section 3.1) caused by dispersants alone.

Pinnipeds will not likely be impacted due to their use of nearshore and intertidal habitat (i.e., near haulouts, where dispersant application is unlikely to be permitted), and the subcutaneous blubber that maintains their body heat (Section 3 of the BA). If exposure to dispersants alone were to occur for any species, it is likely that the concentration would be very dilute, based on the rate of dilution after application (Gallaway et al., 2012). Species will more likely be exposed to dispersed oil. Cetaceans are likely to be exposed to dilute dispersed oil, but physical impacts are unlikely based on the function of subcutaneous blubber in these species.

If birds are exposed to dispersed oil, the physical impacts may be greater than those of oil alone (Duerr et al., 2011). However, at least three factors may reduce the overall impact of oil on these species under field conditions: reduced spill area (NRC, 2005), reduced spill volume and concentration (NRC, 2005), and reduced extent of oiling (CDC and ATSDR, 2010; Lessard and Demarco, 2000). Birds and furbearing mammals that use feathers or fur for thermoregulation or buoyancy on water tend to spend much of their time resting (among other activities) at the ocean's surface (Section 3 of the BA). If the area of the oil slick is reduced at the surface, then the likelihood of a slick coming into contact with such ESA-protected species should be reduced relative to the baseline condition. Modeling by French-McCay (2004) highlighted the importance, particularly for birds and furbearing mammals, of reducing oil at the ocean's surface. The same study indicated that cetaceans and pinnipeds are not at risk of such physical effects. Additionally, reducing the volume and concentration of oil at the surface should mitigate the extent of oiling of these species (NRC, 2005). Although it is not clear whether this will entirely protect these species from becoming oiled, complete dispersion and removal of an oil slick from the surface should reduce oiling to negligible levels. The CDC and ATSDR (2010) and Lessard and Demarco (2000) found that dispersed oil is less likely to "stick to birds and other animals," so it is possible that reduced oiling of birds and mammals (in combination with a reduction in surface slick area and oil concentration) will ultimately reduce the likelihood of lost thermoregulatory or swimming ability. This is a potential diminishment of physical impacts relative to the baseline condition.

Physical impacts caused by dispersants or dispersed oil are not expected in other ESA-listed groups, such as fish or reptiles; French-McCay (2004) modeled the likelihood of mortality in marine reptiles within an oiled area, and found the likelihood of such mortality to be very low (i.e., 1% probability). Fish and reptile species do not regulate their body heat as do birds and mammals, and assumedly do not suffer physically from oiling in a similar way.

4.2 Likelihood of Acute Toxicity

As stated, dispersants are intended exclusively for use on an oil slick at the ocean's surface, and would not be applied directly to water where oil was not present. It has been noted that dispersants will slowly leach from dispersed oil droplets over time (Fingas, 2008), but at a concentration expected to be low relative to acute LC50 values observed in the lab (Attachment B-1, Table 3, Figures 3 and 4). Some overspray is expected during application, but spraying of areas with wildlife is not expected or suggested; certain BMPs or wildlife deterrence measures (if permitted) are intended to preclude wildlife from areas where dispersants are being sprayed. Furthermore, spotter aircraft are used during aerial applications to ensure that overspray is minimized (Brown et al., 2011).

HC5s are provided for Corexit® 9500 and Corexit® 9527 (Table 3) in order to show the relative acute toxicity of dispersants, crude oil, and dispersed oil (i.e., dispersed oil is more acutely toxic than dispersants alone, but less acutely toxic than oil alone) (Tables 3 through 5, Figures 8 and 9). Approximately half of the comparable data suggest that oil is more toxic than dispersed oil, particularly according to the most relevant laboratory testing scenarios (Section 3..2).

The rapid dilution of dispersant after application is expected to result in a very short duration of exposure to concentrated dispersant, even for the most sensitive and vulnerable of aquatic species (e.g., sea surface microlayer, larval fish and invertebrates, and plankton).[37] Dispersant chemicals, when applied during a response action, mix rapidly into an oil spill (ExxonMobil, 2008), are transported and diluted with the motion of waves and currents (NRC, 2005; Nedwed, 2012; Gallaway et al., 2012), and

[37] Shallow-dwelling pelagic and neustonic species are most often represented in the SSDs (Section 3; Attachment B-1). They are also the most likely to be impacted by dispersants applied at the surface of the ocean (as well as by any oil that would be dispersed).

are biodegraded over time (Section 2). Dilution alone is expected to greatly reduce the concentration of dispersants within a matter of hours (Gallaway et al., 2012). Durations of dispersant exposure above the dispersed oil HC5 (Table 5) at a given location may be a matter of minutes or hours (Mackay and McAuliffe, 1988), although repeated dispersant applications may occur over the course of days (Fingas, 2008), potentially resulting in multiple short pulses of dispersed oil into the water column. As the HC5s for dispersed oil (and dispersants alone) are based on constant 48- to 96-hour toxicity tests, a typical response action is not expected to cause acute toxicity to sensitive aquatic life, let alone larger ESA-listed or candidate species. Repeated dispersions may result in mortality of sensitive species, but are unlikely to result in concentrations high enough to cause acute mortality at higher trophic levels (i.e., ESA-listed or candidate species).

Many of the ESA-listed birds and mammals are wide ranging, occur in specific areas only seasonally, forage throughout the water column (some to great depths), and avoid areas of human activity. These activities are discussed at length in Section 3 of the BA. The observance of BMPs is required during any spill response, and these practices are intended to ensure that wildlife are not impacted by the response action. Together, these limiting factors are expected to reduce the likelihood of exposure to dispersed oil and any possibility of acute toxicity resulting from the application of chemical dispersants.

Indirect oil embryotoxicity in birds (i.e., transfer from oiled parent to nest), which can increase after exposure to dispersants (Wooten et al., 2012), is not likely, because the direct exposure of nesting birds or birds on the water to chemical dispersants is unlikely (Butler et al., 1988). This conclusion has also been reached by previous studies (Peakall et al., 1987; French-McCay, 2004; NRC, 2005). BMPs or other response actions (e.g., hazing) could be used (if permitted by a regulating agency) to disperse birds from an area where dispersants were to be applied.

Exposure of marine reptiles to dispersed oil has been specifically studied at least once (Rowe, 2009), and findings suggest that dispersed oil is unlikely to be toxic to turtles *in ovo*. Previously reported toxicity to marine reptiles (Yender and Mearns, 2003; cited in Rowe, 2009) is likely overestimated, as the percolation of oil and dispersed oil through sediment (i.e., where sea turtle eggs are deposited) results in a very low transfer of toxic oil components to eggs under realistic conditions. Species-specific considerations are stated in Section 5.

4.3 Likelihood of Chronic or Sublethal Toxicity

Chronic, large-scale exposures of ESA-listed or candidate species to chemical dispersants alone are not expected to occur in the natural environment, largely due to the rapid rate of dilution and biodegradation after a dispersant application. This is specifically true of larger, less sensitive individuals. However, Pacific herring, a candidate species for listing under ESA, is known to spawn in Alaska and is present during all life stages (Section 3 of the BA). Although dispersants alone are not likely to be sufficiently concentrated in the water column to cause acute toxicity (due to partitioning to oil and sediment, Section 2), over time, the increased surface area of droplets containing dispersants and oil may allow dispersants to leach into the water column in dilute concentrations (Fingas, 2008); also, overspray is possible, but is not expected to be substantial (Butler et al., 1988), and the use of spotter aircraft to guide aerial dispersant applications minimizes overspray (Brown et al., 2011). Leaching and oversprayed dispersants may result in sublethal toxicity in sensitive species (e.g., early-life-stage Pacific herring, Section 5.3.4). It is not clear what concentration of dispersants is likely to leach from dispersed oil droplets into the water column, but it is likely to be dilute (Section 2.1). Chronic, sublethal toxicity in fish is likely to manifest as abnormal development (Lonning and Falk-Petersen, 1978; Falk-Petersen et al., 1983), possibly leading to altered fitness and death.[38] Delayed development has also been observed at high concentrations of Corexit® 9527 (100 ppm) (Lonning and Falk-Petersen, 1978; Falk-Petersen et al., 1983), but this is not an ecologically-relevant concentration, nor is it clearly linked to adverse impacts on survival, growth, or reproduction.

Short-term, sublethal effects on sensitive species and life stages are possible from exposure to dispersed oil at ecologically-relevant concentrations. One study with Atlantic herring embryos (Lee et al., 2011b) reported that concentrations of Corexit® 9500-dispersed oil of 11.08 and 18.00 ppm (ANS and Arabian light crude oils, respectively) were sufficient to cause reduced hatching in half of the exposed embryos after only 2.4 hours. A similar effect was noted for concentrations of 2.21 and 3.07 ppm (using the same dispersant and oil types) after an 8-hour exposure (Lee et al., 2011b); a range of 0.49 to 1.94 ppm was reported as the 24-hour EC50, and a range of < 0.25 to < 0.37 ppm was reported as the 14-day EC50 (Lee et al., 2011b). Even if the concentration of dispersed oil in the water column decreases below the calculated HC5 within a matter of minutes to hours (Mackay and McAuliffe, 1988), it may still be possible for a significant adverse effect to occur in planktonic species at sensitive embryonic life stages. This may have implications for the decision to use dispersants in areas where fish are spawning, particularly for ESA candidate species and concentrations of prey of protected species.

No SSDs were created for sublethal or chronic effects due to the variety of measured endpoints and exposure durations reported in the literature, as well as the paucity of data and species assessed in chronic or sublethal tests (that reported meaningful toxicity values). Without SSDs, HC5s were not calculable for chronic or sublethal endpoints.

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[38] Death in this case is distinct from mortality resulting from exposure to chemicals; the former is indirectly caused by chemical exposure but directly results from reduced fitness (e.g., reduced growth and survival in response to normal environmental factors, such as temperature or dissolved oxygen changes).

5 Summary of Species-Specific Impacts

The purpose of this section is to make a definitive statement about the likelihood of adverse impacts on each species at the individual level (i.e., reduced survival, growth, or reproduction) caused by the use of chemical dispersants. These conclusions are applied to the larger discussion in the BA, and represent just one of many potential adverse impacts on ESA-listed or candidate species that could be caused by an implementation of the Unified Plan.

As noted in Section 1, terrestrial species are not included in this assessment, so Eskimo curlew and the Aleutian shield fern are omitted. Dispersants are intended for use in open water, marine environments; neither of these species utilizes such habitats, so exposures to dispersants or dispersed oil is considered highly unlikely (i.e., discountable) under expected circumstances.

5.1 MAMMALS

5.1.1 Beluga whale, Cook Inlet DPS

Beluga whale are unlikely to be impacted by the physical effects of dispersants (Section 3.1). They rely on subcutaneous blubber to regulate their body heat, instead of fur, which could be compromised by oiling, dispersants, or dispersed oil.

This species is very large and will not likely be exposed to concentrations of dispersants or dispersed oil in quantities large enough to cause acutely toxic effects (e.g., mortality); such effects are unlikely even in lower trophic levels (Section 4). Dispersed oil rapidly dilutes and degrades over time, so chronic exposure to dispersants or dispersed oil in the water column is unlikely (Section 2). Accumulation of PAHs in tissue over time as a result of chemical dispersion is unlikely due to the ability of mammals to metabolize and excrete PAHs, as well as the expected acute nature of a PAH exposure after a chemical dispersion event (Sections 2.1 and 2.2).

Acute exposures to PAHs, which may become more bioavailable in the shallow water column after chemical dispersion, have been linked to various effects on wildlife in PWS after EVOS, although toxicity is noted as secondary to the physical impacts of fouling (e.g., hypothermia) (Albers and Loughlin 2003). Larger marine mammals with subcutaneous blubber (e.g., cetaceans), which would not suffer from hypothermia caused by fouling, were observed to experience sublethal impacts (e.g., lesions) after EVOS (Albers and Loughlin 2003). It is unclear whether the application of chemical dispersants would increase the exposure of whales to PAHs, resulting in a greater prevalence of lesions. It is also unclear whether lesions caused by increased exposure to PAHs would lead to significant effects resulting in the impairment of essential behavioral patterns (e.g., breeding, feeding, and sheltering). The impact of PAHs on whale species as a result of acute exposure after chemical dispersion is a point of uncertainty (Section 6.3.4).

Beluga spend much of their time in fairly shallow water, so they may be more exposed to dispersed oil than other cetaceans. However, they also may be more exposed to oil alone, in the event that dispersant is not applied, because they remain at the surface, where oil becomes concentrated. Dispersion would assumedly remove much of the oil from the ocean's surface, effectively reducing the exposure of beluga. And, as noted in Section 3.1, exposure to oil alone when surfacing to breathe is more likely to cause severe impacts on cetacean species than exposure to dispersed oil in the water column.

The prey base of beluga whale is largely composed of juvenile or adult fish species, often anadromous fish. Anadromous fish are unlikely to be impacted by dispersants or dispersed oil during spawning or rearing (i.e., not present in marine waters during those activities), but they may be exposed to sufficient levels of dispersed oil as juveniles to elicit sublethal effects (Section 3.2.3.4). Beluga also prey upon marine fish, which may be impacted to a greater extent if spawning occurs in shallow waters (i.e., less than 10 m deep) (Section 1.3). As stated in Sections 3.1.1 and 3.1.2, embryonic fish are much more likely to suffer from the acutely toxic impacts of dispersant application. Such impacts may be greater than those caused by oil alone if spawning occurs between 1 and 10 m deep, since embryos at such depths would not be exposed to oil, but would be exposed to dispersed oil.

Based on the rationale provided above, Cook Inlet beluga whale are anticipated to be exposed to dispersed oil in the event of a chemical dispersant application, potentially resulting in adverse impacts. Exposures to dispersed oil and increased uptake of PAHs from the water column may result in sublethal responses (e.g., lesions and irritation of sensitive tissues). The likelihood and duration of exposure of beluga whale to dispersed oil may be facilitated by their localized, year-round distribution within Cook Inlet, and the importance of their critical habitat (e.g., shallow waters used for feeding, calving, and predator evasion), which may be degraded by dispersed oil (NMFS, 2008a). Furthermore, the likelihood of exposure is greater due to the frequency of oil or other petroleum products spills in Cook Inlet (Appendix D).

5.1.2 Blue whale

Blue whale are unlikely to be impacted by the physical effects of dispersants (Section 3.1). They rely on subcutaneous blubber to regulate their body heat, instead of fur, which could be compromised by oiling, dispersants, or dispersed oil.

This species is extremely large and will not likely be exposed to dispersants or dispersed oil in quantities significant enough to cause acute toxic effects (e.g., mortality); such effects are unlikely even at lower trophic levels (Section 4). Dispersed oil rapidly dilutes and degrades over time, so chronic exposure to dispersants or dispersed oil is unlikely (Section 2).

It is possible (although unlikely) that dispersed oil will be ingested by blue whale, which feed through their baleen on planktonic species. However, the ingestion of even large quantities of crude oil by much smaller species has been found to cause minimal effects (Section 3.1), and cetaceans are likely able to efficiently metabolize hydrocarbons (Albers and Loughlin, 2003). It is highly unlikely that blue whale will ingest large quantities of dispersed oil due to the depth at which they are found (Wade and Friedrichsen, 1979; as cited in Reeves et al., 1998). Given that embryonic and larval life stages of blue whale prey may be found in shallow water during a chemical dispersant application, it is possible that these prey species may be impacted (Section 3.2).

The trophic transfer of PAHs to invertebrates in dispersed-oil exposures does occur, but fish metabolize PAHs fairly efficiently (Wolfe et al., 2001; Logan, 2007). The magnification of PAHs in blue whale through their diet is unlikely (Albers and Loughlin, 2003), because the higher trophic levels, including cetaceans, metabolize PAHs efficiently. Accumulation of PAHs in tissue over time as a result of chemical dispersion is unlikely due to the ability of mammals to metabolize and excrete PAHs, as well as the expected acute nature of a PAH exposure after a chemical dispersion event (i.e., rapid dilution and increased rate of degradation) (Sections 2.1 and 2.2).

Acute exposures to PAHs, which may become more bioavailable in the shallow water column after chemical dispersion, have been linked to various effects on wildlife in PWS after EVOS, although toxicity is noted as secondary to the physical impacts of fouling (e.g., hypothermia) (Albers and Loughlin 2003). Larger marine mammals with subcutaneous blubber (e.g., cetaceans), which would not suffer from hypothermia caused by fouling, were observed to experience sublethal impacts (e.g., lesions) after EVOS (Albers and Loughlin 2003). It is unclear whether the application of chemical dispersants would increase the exposure of whales to PAHs, resulting in a greater prevalence of lesions. It is also unclear whether lesions caused by increased exposure to PAHs would lead to significant effects resulting in the impairment of essential behavioral patterns (e.g., breeding, feeding, and sheltering). The impact of PAHs on whale species as a result of acute exposure after chemical dispersion is a point of uncertainty (Section 6.3.4).

Blue whales periodically surface to breathe, which requires that they potentially come into contact with oil at the ocean's surface. Because dispersants remove oil from the ocean's surface and, through dilution, reduce the concentration of oil, it can be expected that the exposure of blue whale to oil will be mitigated by dispersants. Exposure will increase as the species moves from deep waters through the upper 10 m (before reaching the surface), but this is expected to result in minimal impacts (Section 3.1). It is not expected that exposures will last, as blue whales surface briefly and then return to deeper water to feed.

For these reasons, blue whale are not anticipated to be negatively impacted by the application of dispersants if BMPs are implemented during the response action. For example, dispersant applications should not occur in areas where blue whales are known to be present.

In the unlikely event that BMPs fail, and the implementation of the Unified Plan fails to protect marine mammals, blue whales may be adversely impacted by the application of dispersants. Potential impacts on blue whales in a worst-case scenario are provided in the main text of the BA.

5.1.3 Bowhead whale

Bowhead whale are unlikely to be impacted by the physical effects of dispersants (Section 3.1). They rely on subcutaneous blubber to regulate their body heat, instead of fur, which could be compromised by oiling, dispersants, or dispersed oil.
It is possible that dispersed oil will be ingested by bowhead whale, which feed through their baleen on planktonic species, particularly in shallow waters. The amount of hydrocarbons accumulated will be limited by the use of dispersants to break up oil and facilitate metabolic breakdown and the ability of cetaceans to efficiently metabolize ingested hydrocarbons (Albers and Loughlin, 2003). Therefore, substantial bioaccumulation or magnification of oil components from direct ingestion of dispersed oil are not likely to occur over time (Sections 2.1 and 2.2). Oiling of bowhead whale habitat, such as broken sea ice, breathing holes, or polynyas, could result in pools and concentrations of oil, severely impacting bowhead whale. Dispersion in these areas, particularly where bowhead whale surface to breathe, could mitigate such impacts by reducing the amount of surface oil (Section 3.1). However, ingestion of dispersed oil during feeding may increase, leading to fouled baleen and sublethal impacts (e.g., vomiting and tissue irritation). Such effects may reduce the feeding efficiency of bowhead whale (BOEMRE, 2011). Bowhead whale will likely be most susceptible to such impacts during summer, when feeding increases (BOEMRE, 2011).

During migration from April to June, calves are born (Koski et al., 1993; cited in NMFS, 2002). Calves tend to reside in the upper 20 m of the water column (Koski and Miller, 2009), which puts them at particular risk of exposure to both dispersed oil and oil alone. As noted in Section 3.1, the acute impacts of dispersed oil on cetaceans are less than those of oil alone, due to the altered route of exposure (i.e., ingestion of dispersed oil as opposed to inhalation or aspiration of surface oil).

The trophic transfer to invertebrates of PAHs in dispersed-oil exposures has been shown, but fish metabolize PAHs fairly efficiently (Wolfe et al., 2001). The magnification of PAHs in bowhead whale through their diet is unlikely (Albers and Loughlin, 2003), because higher trophic levels, including cetaceans, metabolize PAHs efficiently. Accumulation of PAHs in tissue over time as a result of chemical dispersion is unlikely due to the ability of mammals to metabolize and excrete PAHs, as well as the acute nature of a PAH exposure after a chemical dispersion event (i.e., rapid dilution and increased rate of degradation) (Sections 2.1 and 2.2).

Acute exposures to PAHs, which may become more bioavailable in the shallow water column after chemical dispersion, have been linked to various effects on wildlife in PWS after EVOS, although toxicity is noted as secondary to the physical impacts of fouling (e.g., hypothermia) (Albers and Loughlin 2003). Larger marine mammals with subcutaneous blubber (e.g., cetaceans), which would not suffer from hypothermia caused by fouling, were observed to experience sublethal impacts (e.g., lesions) after EVOS (Albers and Loughlin 2003). It is unclear whether the application of chemical dispersants would increase the exposure of whales to PAHs, resulting in a greater prevalence of lesions. It is also unclear whether lesions caused by increased exposure to PAHs would lead to significant effects resulting in the impairment of essential behavioral patterns (e.g., breeding, feeding, and sheltering). The impact of PAHs on whale species as a result of acute exposure after chemical dispersion is a point of uncertainty (Section 6.3.4).

For the reasons noted, chemical dispersion may affect bowhead whales by causing increased baleen fouling and reduced feeding efficiency. However, the incremental benefit of removing oil from the surface (i.e., reducing inhalation or aspiration) outweighs the potential for exposure in the water column (i.e., increasing ingestion and potentially fouled baleen). This conclusion assumes that dispersants are not directly applied to areas where bowhead whale are known to be congregated.

In the unlikely event that BMPs fail, and the implementation of the Unified Plan fails to protect marine mammals, bowhead whales may be adversely impacted by the application of dispersants. Potential impacts on bowhead whales in a worst-case scenario are provided in the main text of the BA.

5.1.4 Fin whale

Fin whale are unlikely to be impacted by the physical effects of dispersants (Section 3.1). They rely on subcutaneous blubber to regulate their body heat, instead of fur, which could be compromised by oiling, dispersants, or dispersed oil.

This species is extremely large and will not likely be exposed to dispersants or dispersed oil in quantities large enough to cause acute toxic effects (e.g., mortality); such effects are unlikely even at lower trophic levels (Section 4). Dispersed oil rapidly dilutes and degrades over time, so chronic exposure to dispersants or dispersed oil is unlikely (Section 2).

It is possible (although unlikely) that dispersed oil will be ingested by fin whale, which feed through their baleen on planktonic species. The ingestion of crude oil, even in large quantities, in much smaller species has been found to cause minimal impacts (Section 3.1), and cetaceans are likely able to efficiently metabolize hydrocarbons (Albers and Loughlin, 2003). It is highly unlikely that fin whale will ingest large quantities of dispersed oil due to the depths at which they are often found (i.e., between 50 and 600 m) (US Navy, 2011; Croll et al., 2001; Goldbogen et al., 2006; Panigada et al., 2003). Assuming that fin whale feed at depths > 10 m, it is likely that their prey are also found primarily at depths > 10 m; therefore, the prey population of fin whale is unlikely to be exposed to high concentrations of dispersed oil, if any at all (Section 2). However, the larval life stages of these species may be found in shallower waters, so impacts may occur in very sensitive species (Section 3.2). Within the overall community, acute toxicity is expected to decrease as a result of chemical dispersion relative to oil alone (Section 3.3).

The trophic transfer to invertebrates of PAHs in dispersed-oil exposures has been shown, but fish metabolize PAHs fairly efficiently (Wolfe et al., 2001). The accumulation and/or magnification of PAHs in fin whale through their diet is unlikely (Albers and Loughlin, 2003), because higher trophic levels, including cetaceans, metabolize PAHs efficiently. Also, rapid dilution, biodegradation, and transportation of an oil plume are expected to result in acute, temporary exposures in the water column (Section 2).

Acute exposures to PAHs, which may become more bioavailable in the shallow water column after chemical dispersion, have been linked to various effects on wildlife in PWS after EVOS, although toxicity is noted as secondary to the physical impacts of fouling (e.g., hypothermia) (Albers and Loughlin 2003). Larger marine mammals with subcutaneous blubber (e.g., cetaceans), which would not suffer from hypothermia caused by fouling, were observed to experience sublethal impacts (e.g., lesions) after EVOS (Albers and Loughlin 2003). It is unclear whether the application of chemical dispersants would increase the exposure of whales to PAHs, resulting in a greater prevalence of lesions. It is also unclear whether lesions caused by increased exposure to PAHs would lead to significant effects resulting in the impairment of essential behavioral patterns (e.g., breeding, feeding, and sheltering). The impact of PAHs on whale species as a result of acute exposure after chemical dispersion is a point of uncertainty (Section 6.3.4).

Fin whale surface periodically to breathe, requiring that they potentially come into contact with oil at the ocean's surface. Because dispersants remove oil from the ocean's surface and, through dilution, reduce the concentration of oil, the exposure of fin whale to oil will be mitigated through dispersion. Exposure will increase as they move from deep water through the upper 10 m (before reaching the surface), but this is expected to result in minimal or minimized impacts (Section 3.1); fin whale surface briefly, then return to deeper water to feed. Fin whale spend approximately 44% of their time in water less than 50 m deep (Goldbogen et al., 2006), a depth that will be mostly unaffected by dispersed oil.

For these reasons, fin whale are not anticipated to be negatively impacted by the application of dispersants if all BMPs are implemented during the response action. For example, dispersant applications should not occur in areas where fin whale are known to be present.

In the unlikely event that BMPs fail, and the implementation of the Unified Plan fails to protect marine mammals, fin whale may be adversely impacted by the application of dispersants. Potential impacts on fin whales in a worst-case scenario are provided in the main text of the BA.

5.1.5 Gray whale, Western North Pacific DPS

Gray whale are unlikely to be impacted by the physical effects of dispersants (Section 3.1). They rely on subcutaneous blubber to regulate their body heat, instead of fur, which could be compromised by oiling, dispersants, or dispersed oil.

This species is very large and will not likely be exposed to concentrations of dispersants or dispersed oil in quantities large enough to cause acute toxic effects (e.g., mortality); such effects are unlikely even at lower trophic levels (Section 4). Dispersed oil rapidly dilutes and degrades over time, so chronic exposure to dispersants or dispersed oil is unlikely (Section 2).

It is possible that dispersed oil will be ingested by gray whale, which feed through their baleen on benthic species suctioned from sediment (Nerini, 1984). The ingestion of crude oil, even in large quantities, in much smaller species has been found to cause minimal impacts (Section 3.1), and cetaceans are likely able to efficiently metabolize hydrocarbons (Albers and Loughlin, 2003). It is highly unlikely that gray whale will ingest large quantities of dispersed oil due to where they feed, typically 50 to 60 m deep along the continental shelf (Nerini, 1984; ADF&G, 2008). Benthic prey species that live at these depths will not be exposed to dispersed oil in large concentrations, so indirect effects on gray whale are unlikely.

Acute exposures to PAHs, which may become more bioavailable in the shallow water column after chemical dispersion, have been linked to various effects on wildlife in PWS after EVOS, although toxicity is noted as secondary to the physical impacts of fouling (e.g., hypothermia) (Albers and Loughlin 2003). Larger marine mammals with subcutaneous blubber (e.g., cetaceans), which would not suffer from hypothermia caused by fouling, were observed to experience sublethal impacts (e.g., lesions) after EVOS (Albers and Loughlin 2003). It is unclear whether the application of chemical dispersants would increase the exposure of whales to PAHs, resulting in a greater prevalence of lesions. It is also unclear whether lesions caused by increased exposure to PAHs would lead to significant effects resulting in the impairment of essential behavioral patterns (e.g., breeding, feeding, and sheltering). The impact of PAHs on whale species as a result of acute exposure after chemical dispersion is a point of uncertainty (Section 6.3.4).

Gray whale surface periodically to breathe, requiring that they potentially come into contact with oil at the ocean's surface. Because dispersants remove oil from the ocean's surface and, through dilution, reduce the concentration of oil, the exposure of gray whale to oil will be mitigated through dispersion.

For these reasons, gray whale are not anticipated to be negatively impacted by the application of dispersants if all BMPs are implemented during the response action. For example, dispersant applications should not occur in areas where gray whales are known to be present.

In the unlikely event that BMPs fail, and the implementation of the Unified Plan fails to protect marine mammals, gray whale may be adversely impacted by the application of dispersants. Potential impacts on gray whales in a worst-case scenario are provided in the main text of the BA.

5.1.6 Humpback whale

Humpback whale are unlikely to be impacted by the physical effects of dispersants (Section 3.1). They rely on subcutaneous blubber to regulate their body heat, instead of fur, which could be compromised by oiling, dispersants, or dispersed oil.

This species is very large and will not likely be exposed to concentrations of dispersants or dispersed oil in quantities large enough to cause acute toxic effects (e.g., mortality); such effects are unlikely even at lower trophic levels (Section 4). Dispersed oil rapidly dilutes and degrades over time, so chronic exposure to dispersants or dispersed oil is unlikely (Section 2).

It is possible that dispersed oil will be ingested by humpback whale, which feed through their baleen on various species, small fish in particular, which are captured by various methods (Ingebrigtsen, 1929; Jurasz and Jurasz, 1979; Watkins and Schevill, 1979; Hain et al., 1982; Weinrich, 1983; Baker, 1985; Baker and Herman, 1985; Hays et al., 1985; Winn and Reichley, 1985; D'Vincent et al., 1985; as cited in NMFS, 1991). The ingestion of crude oil, even in large quantities, has been found to cause minimal impacts in much smaller species than humpback whales (Section 3.1), and cetaceans are likely able to efficiently metabolize hydrocarbons (Albers and Loughlin, 2003). It is unlikely that humpback whale will ingest large quantities of dispersed oil due to the depths at which they feed, typically between 92 and 120 m deep (NMFS, 2011a), and as deep as 500 m (US Navy, 2011).

Humpback whales can also be found in the nearshore environment, where exposures to chemical dispersants should not be substantially different. Dispersant applications are not intended for nearshore habitats, although tides and currents may move a dispersed spill into the nearshore environment. If an oil spill has been appropriately dispersed (i.e., all BMPs have been implemented by the On-Scene Coordinator and dispersion has been effective), dilution and biodegradation are likely to occur to some extent prior to a plume reaching the nearshore environment.

Acute exposures to PAHs, which may become more bioavailable in the shallow water column after chemical dispersion, have been linked to various effects on wildlife in PWS after EVOS, although toxicity is noted as secondary to the physical impacts of fouling (e.g., hypothermia) (Albers and Loughlin 2003). Larger marine mammals with subcutaneous blubber (e.g., cetaceans), which would not suffer from hypothermia caused by fouling, were observed to experience sublethal impacts (e.g., lesions) after EVOS (Albers and Loughlin 2003). It is unclear whether the application of chemical dispersants would increase the exposure of whales to PAHs, resulting in a greater prevalence of lesions. It is also unclear whether lesions caused by increased exposure to PAHs would lead to significant effects resulting in the impairment of essential behavioral patterns (e.g., breeding, feeding, and sheltering). The impact of PAHs on whale species as a result of acute exposure after chemical dispersion is a point of uncertainty (Section 6.3.4).

For these reasons, humpback whale are not anticipated to be negatively impacted by the application of dispersants if all BMPs are implemented during the response action. For example, dispersant applications should not occur in areas where humpback whale are known to be present.

In the unlikely event that BMPs fail, and the implementation of the Unified Plan fails to protect marine mammals, humpback whale may be adversely impacted by the application of dispersants. Potential impacts on humpback whales in a worst-case scenario are provided in the main text of the BA.

5.1.7 North Pacific right whale, eastern stock

North Pacific right whale (NPRW) are unlikely to be impacted by the physical effects of dispersants (Section 3.1). They rely on subcutaneous blubber to regulate their body heat, instead of fur, which could be compromised by oiling, dispersants, or dispersed oil.

This species is very large and will not likely be exposed to concentrations of dispersants or dispersed oil in quantities large enough to cause acute toxic effects (e.g., mortality); such effects are unlikely even at lower trophic levels (Section 4). Dispersed oil rapidly dilutes and degrades over time, so chronic exposure to dispersants or dispersed oil is unlikely (Section 2).

Acute exposures to PAHs, which may become more bioavailable in the shallow water column after chemical dispersion, have been linked to various effects on wildlife in PWS after EVOS, although toxicity is noted as secondary to the physical impacts of fouling (e.g., hypothermia) (Albers and Loughlin 2003). Larger marine mammals with subcutaneous blubber (e.g., cetaceans), which would not suffer from hypothermia caused by fouling, were observed to experience sublethal impacts (e.g., lesions) after EVOS (Albers and Loughlin 2003). It is unclear whether the application of chemical dispersants would increase the exposure of whales to PAHs, resulting in a greater prevalence of lesions. It is also unclear whether lesions caused by increased exposure to PAHs would lead to significant effects resulting in the impairment of essential behavioral patterns (e.g., breeding, feeding, and sheltering). The impact of PAHs on whale species as a result of acute exposure after chemical dispersion is a point of uncertainty (Section 6.3.4).

It is possible that dispersed oil will be ingested by NPRW, which feed through their baleen on various species, particularly copepods. The ingestion of crude oil, even at large quantities, has been found to cause minimal impacts in much smaller species than NPRW (Section 3.1), and cetaceans are likely able to efficiently metabolize hydrocarbons (Albers and Loughlin, 2003). It is unlikely that NPRW will ingest large quantities of dispersed oil due to the depths at which they feed, between 80 and 175 m (as assumed from NPRW behavior) (US Navy, 2011).

In NPRW critical habitat (Section 3.4.1.6.1 of the BA), NPRW prey species are known to be very dense, and dense aggregations of copepods are directly related to NPRW movements (Shelden et al., 2005). Although NPRW are thought to feed deeper (i.e., > 10 m) in the water column (US Navy, 2011), dispersant application could impact the sensitive prey species of NPRW. However, based on the information presented in Section 3.3, dispersion will reduce toxicity in aquatic species, particularly at the ocean's surface. Those prey species that NPRW feed upon at depth should be unaffected by oil or dispersed oil due to environmental restrains on vertical mixing (Section 2). Furthermore, toxicity data indicates that Arctic copepod species (e.g., *C. glacialis*) are less sensitive to dispersed oil toxicity than other species (Figure 6), and approximately 20 times more sensitive to oil alone than dispersed oil (McFarlin et al., 2011). Based on these two indications of toxicity, a significant portion of the planktonic community (as well as specific dietary components for NPRW [i.e., copepods]) will not be significantly affected by dispersant application, making indirect impacts on NPRW unlikely.

NPRW surface periodically, approximately every 5 to 15 minutes, to breathe (US Navy, 2011), requiring that they potentially come into contact with oil at the ocean's surface. Because dispersants remove oil from the ocean's surface and, through dilution, reduce the concentration of oil, the exposure of NPRW to oil will be mitigated through dispersion. As noted in Section 3.1.2.3, oil at the ocean's surface is likely to cause more severe impacts than dispersed oil due to the altered route of exposure (i.e., inhalation and aspiration at the surface when breathing, as opposed to ingestion and dermal contact in the water column).[39]

For these reasons, NPRW are not anticipated to be negatively impacted by the application of dispersants if all BMPs are implemented during the response action. For example, dispersant applications should not occur in areas where NPRW are known to be present, particularly not in critical habitat for this species, where a larger portion of the population could be exposed.

[39] This statement is based on the assumption that acute lung, kidney, and liver tissue damage are more likely to result in observable impacts than exterior irritation, inflammation, or lesions or gastrointestinal irritation. Lung functionality in particular has been deemed important for cetaceans, which rely on their ability to dive and remain underwater for long periods of time.

In the unlikely event that BMPs fail, and the implementation of the Unified Plan fails to protect marine mammals, NPRW may be adversely impacted by the application of dispersants. Potential impacts on NPRW in a worst-case scenario are provided in the main text of the BA.

5.1.8 Sei whale

Sei whale are unlikely to be impacted by the physical effects of dispersants (Section 3.1). They rely on subcutaneous blubber to regulate their body heat, instead of fur, which could be compromised by oiling, dispersants, or dispersed oil.

This species is very large and will not likely be exposed to concentrations of dispersants or dispersed oil in quantities large enough to cause acute toxic effects (e.g., mortality); such effects are unlikely even at lower trophic levels (Section 4). Dispersed oil rapidly dilutes and degrades over time, so chronic exposure to dispersants or dispersed oil is unlikely (Section 2).

It is possible that dispersed oil will be ingested by sei whale, which feed through their baleen on planktonic species, fish, and large invertebrates (e.g., squid) (Nemoto and Kawamura, 1977; Kawamura, 1982; both cited in NMFS, 2011b). Sei whale feed throughout the water column, periodically skimming the surface (NOAA Fisheries, 2013). Surface skimming and feeding in the shallow water column put sei whale at particular risk of ingesting oil at the ocean's surface. Although oil ingestion is not likely to be the most toxic route of exposure for mammals (Section 3.1), excessive feeding at the ocean's surface could result in the ingestion of very large quantities of oil. Diving among sei whale is limited, with dives typically lasting 5 to 10 minutes and rarely being deeper than 300 m (MarineBio, 2012). It is possible that sei whale surface more frequently to breathe than do other deeply diving whales (e.g., blue whale), so inhalation and aspiration of oil fumes is also a potential route of exposure, more so than for other ESA-listed cetaceans, particularly when oil is left at the surface (i.e., not dispersed). The application of dispersants greatly reduces the concentration of oil at the surface, as well as the volatilization of the oil spill (Section 2), so chemical dispersion should reduce the exposure of sei whale to oil, specifically limiting the more harmful routes of exposure (Section 3.1).

Acute exposures to PAHs, which may become more bioavailable in the shallow water column after chemical dispersion, have been linked to various effects on wildlife in PWS after EVOS, although toxicity is noted as secondary to the physical impacts of fouling (e.g., hypothermia) (Albers and Loughlin 2003). Larger marine mammals with subcutaneous blubber (e.g., cetaceans), which would not suffer from hypothermia caused by fouling, were observed to experience sublethal impacts (e.g., lesions) after EVOS (Albers and Loughlin 2003). It is unclear whether the application of chemical dispersants would increase the exposure of whales to PAHs, resulting in a greater prevalence of lesions. It is also unclear whether lesions caused by increased exposure to PAHs would lead to significant effects resulting in the impairment of essential behavioral patterns (e.g., breeding, feeding, and sheltering). The impact of PAHs on whale species as a result of acute exposure after chemical dispersion is a point of uncertainty (Section 6.3.4).

Indirect impacts on sei whale due to dispersant application are not expected, because dispersants decrease toxicity in the overall planktonic community relative to oil alone (Sections 3.3 and 3.4, Figures 8 and 9). Sei whale are known to be opportunistic feeders (Flinn et al., 2002; Tamura et al., 2009; as cited in NMFS, 2011b) and often feed on large species (e.g., adult squid and mackerel) (Nemoto and Kawamura, 1977; Kawamura, 1982; both cited in NMFS, 2011b), so the prey species of sei whale are likely to be insensitive, large-bodied fish and invertebrates in later life stages, which are known to be less sensitive than small species in early life stages (Attachment B-1).

For these reasons, sei whale are not anticipated to be negatively impacted by the application of dispersants if all BMPs are implemented during the response action. Rather, dispersion would likely result in a net benefit for sei whale relative to the baseline condition.

In the unlikely event that BMPs fail, and the implementation of the Unified Plan fails to protect marine mammals, sei whale may be adversely impacted by the application of dispersants. Potential impacts on sei whales in a worst-case scenario are provided in the main text of the BA.

5.1.9 Sperm whale

Sperm whale are unlikely to be impacted by the physical effects of dispersants (Section 3.1). They rely on subcutaneous blubber to regulate their body heat, instead of fur, which could be compromised by oiling, dispersants, or dispersed oil.

This species is extremely large and will not likely be exposed to concentrations of dispersants or dispersed oil in quantities large enough to cause acute toxic effects (e.g., mortality); such effects are unlikely even at lower trophic levels (Section 4). Dispersed oil rapidly dilutes and degrades over time, so chronic exposure to dispersants or dispersed oil is unlikely (Section 2).

Acute exposures to PAHs, which may become more bioavailable in the shallow water column after chemical dispersion, have been linked to various effects on wildlife in PWS after EVOS, although toxicity is noted as secondary to the physical impacts of fouling (e.g., hypothermia) (Albers and Loughlin 2003). Larger marine mammals with subcutaneous blubber (e.g., cetaceans), which would not suffer from hypothermia caused by fouling, were observed to experience sublethal impacts (e.g., lesions) after EVOS (Albers and Loughlin 2003). It is unclear whether the application of chemical dispersants would increase the exposure of whales to PAHs, resulting in a greater prevalence of lesions. It is also unclear whether lesions caused by increased exposure to PAHs would lead to significant effects resulting in the impairment of essential behavioral patterns (e.g., breeding, feeding, and sheltering). The impact of PAHs on whale species as a result of acute exposure after chemical dispersion is a point of uncertainty (Section 6.3.4).

Sperm whale generally prey on large and deep-dwelling species of cephalopod and fish (NMFS, 2010), species highly unlikely to be impacted by dispersed oil or baseline oiling. As larvae, these species may be found in the shallow ocean as plankton. As shown in Sections 3.3 and 3.4 and Figures 8 and 9, the toxicity of dispersed oil is expected to be less than that of oil alone. This is particularly true for large fish species (e.g., *Oncorhynchus* sp. and Arctic cod) and cephalopods (e.g., pale octopus) (Attachment B-1). For these reasons, it is unlikely that the application of dispersants will have a significant adverse impact on sperm whale prey; rather, dispersants may have a positive net impact due to decreased toxicity. Thus, an indirect impact on sperm whale is unlikely.

Because sperm whale tend to dive very deeply to seek prey, as much as 30 minutes at a time and often > 400 m (and up to 2,000 m) (Watkins et al., 2002; cited in US Navy, 2008), it is not expected that sperm whale will be exposed to oil or dispersed oil for extended periods of time. However, surfacing to breathe poses a potential point of exposure. Oiling where sperm whale surface could result in severe impacts (Section 3.1), so the application of dispersants to reduce the volume, concentration, and areal extent of surface oiling would reduce impacts on surfacing sperm whale. The resulting increase in dispersed oil in the shallow water column should not cause as severe of impacts (Section 3.1), and dispersed oil is expected to be less toxic than oil alone (Section 3.1).

For these reasons, sperm whale are not anticipated to be negatively impacted by the application of dispersants if all BMPs are implemented during the response action.

In the unlikely event that BMPs fail, and the implementation of the Unified Plan fails to protect marine mammals, sperm whales may be adversely impacted by the application of dispersants. Potential impacts on sperm whales in a worst-case scenario are provided in the main text of the BA.

5.1.10 Steller sea lion, eastern and western populations

Steller sea lion are unlikely to be impacted by the physical effects of dispersants (Section 3.1). They rely on subcutaneous blubber to regulate their body heat, instead of fur, which could be compromised by oiling, dispersants, or dispersed oil.

This species is large and will not likely be exposed to concentrations of dispersants or dispersed oil in quantities large enough to cause acute toxic effects (e.g., mortality); such effects are unlikely even at lower trophic levels (Section 4). Dispersed oil rapidly dilutes and degrades over time, so chronic exposure to dispersants or dispersed oil is unlikely (Section 2). Sublethal impacts related to dispersed oil are certainly possible, but it is unlikely that dispersed oil will have greater impacts than oil alone, particularly on Steller sea lion, which frequently dive through the ocean's surface and use shoreline haulouts. The application of dispersants is expected to result in diminished oiling of shorelines (Fingas, 2008) and haulouts, as well as a reduced volume, concentration, and areal extent of oil at the ocean surface (NRC, 2005), where Steller sea lions could be exposed. Allowing haulouts or rookeries to be oiled (i.e., No Action alternative) may result in the chronic exposure of this species, as the oil degrades slowly on the shoreline over many years (Peterson et al., 2003).

Dispersants are expected to reduce the volatilization of oil by dissolving its lighter components (Section 2). Thus, the risk of inhalation or aspiration exposure for Steller sea lions at the ocean's surface or on haulouts may be diminished by dispersant application. Inhalation and aspiration of oil may have severe impacts in mammals (Section 3.1).

Ingestion of oil in the shallow water column (as deep as 10 m) may be increased by dispersion, but ingestion results in less severe impacts on mammals than does inhalation (Section 3.1). Mammals are known to effectively metabolize and excrete PAHs when ingested (Albers and Loughlin, 2003), so ingested hydrocarbons are unlikely to accumulate or magnify in Steller sea lions over time as a result of chemical dispersion; exposures to chemical dispersants are expected to be acute and temporary (Section 2). Acute exposures to PAHs have been linked to various effects on wildlife in PWS after EVOS, although toxicity is noted as secondary to the physical impacts caused by fouling (e.g., hypothermia) (Albers and Loughlin, 2003). Larger marine mammals with subcutaneous blubber (i.e., those that would not suffer from hypothermia) experienced sublethal impacts (e.g., lesions) after EVOS, although it was not determined whether observed impacts corresponded to impacts on survival, growth, or reproduction (Albers and Loughlin, 2003).

Steller sea lions generally feed on schooling fish (62 FR 24345, 1997), which could, as larvae, be exposed to dispersants and dispersed oil. The application of dispersants has a severe impact on sensitive species, particularly herring (Lee et al., 2011b), but dispersed oil is less toxic to these species than oil alone (Lee et al., 2011b; Sections 3.3 and 3.4; Figures 8 and 9). Impacts on herring are discussed in Section 5.3.4. Allowing important spawning habitat for sea lion prey species (e.g., walleye [*Sander vitreus*], pollock species, Atka mackerel [*Pleurogrammus monopterygius*], herring species, and capelin [*Mallotus villosus*]) to be oiled will likely result in greater toxicity than if dispersants are applied (Sections 3.3 and 3.4, Figures 8 and 9), and long-term impacts on kelp beds or other intertidal shorelines will be reduced (Peterson et al., 2003). Appropriately planned and executed dispersant applications (i.e., all BMPs properly implemented) will have a net positive benefit on Steller sea lion prey species relative to baseline conditions.

For these reasons, the application of dispersants is not expected to have significant adverse effects on Steller sea lion relative to the baseline condition. All BMPs should be implemented to avoid applying dispersants directly where sea lion are present, or where sensitive prey species are spawning.

In the unlikely event that BMPs fail, and the implementation of the Unified Plan fails to protect marine mammals, Steller sea lions may be adversely impacted by the application of dispersants. Potential impacts on Steller sea lions in a worst-case scenario are provided in the main text of the BA.

5.1.11 Polar bear

Polar bears selectively avoid oil on water when given the choice (St. Aubin, 1988), so it is unlikely that polar bears will approach and dive through oiled waters. It is not clear whether the dispersion of oil into the water column will result in behavioral changes in polar bears, or whether polar bears will dive into waters where oil has been dispersed. It is possible that slight oiling will occur on polar bears that dive into waters where dispersed oil exists. This may result in increased physical impacts.

Polar bears are furbearing mammals that may be significantly impacted by the physical effects of oiling or dispersant exposure (Section 3.1). Polar bears that dive through heavily oiled surface waters will themselves become heavily oiled, resulting in a decreased ability to maintain their body temperature. Hypothermia and death could result (St. Aubin, 1988). Thermal regulation is also important to keep polar bears cool during the summer (St. Aubin, 1988), so oiling could result in heat exhaustion or other heat-related maladies. The application of chemical dispersants in areas with heavily oiled surface water would result in a decreased concentration, volume, and areal extent of surface oil, likely reducing the potential for polar bears to be oiled. Although severe oiling is unlikely (and behaviorally avoided) (St. Aubin, 1988), slight oiling may have less extensive sublethal impacts on polar bears. Impacts would be less extensive due to the lower concentration or volume of oil, as well as the decrease in the stickiness of the oil (CDC and ATSDR, 2010; Lessard and Demarco, 2000).

Polar bears groom their fur, so oiling results in the ingestion of large volumes of oil (St. Aubin, 1988). Ingestion of oil in bears caused vomiting, gastrointestinal distress, serious liver and kidney damage, blood cell damage, and death (St. Aubin, 1988). It is not clear whether such effects would occur in polar bears if oil were chemically dispersed, but it is expected that the lower concentrations ingested would result in less exposure and reduced toxic effects (Section 3.1). It can be assumed that polar bears would avoid oil associated with the baseline condition.

Ringed and bearded seals are the primary prey of polar bears in Alaska; neither species is expected to be more adversely impacted by dispersed of oil than by the baseline condition. Rather, oiling of these species is more likely under the baseline condition, as they frequently dive through small holes in sea ice where oil could accumulate. Dispersing any oil under the ice would likely decrease the oiling of ice seals, and thereby reduce the potential transfer of oil from seal pelts to polar bears.

It is unlikely that hydrocarbons would bioaccumulate in seal tissues as a result of acute exposure, because seals are able to metabolize PAHs (Albers and Loughlin, 2003). Similarly, polar bears have efficient mechanisms for metabolizing and excreting hydrocarbons, so the transfer of parent PAHs from seals to polar bears as a result of chemical dispersant application in the arctic is unlikely, as is the accumulation of PAHs in polar bears resulting from the consumption of seal tissue. The impacts of PAH exposures on polar bears, and whether such exposures would result in reduced survival, growth, or reproduction, are points of uncertainty, discussed in Section 6.3.4. Acute exposures to PAHs have been linked to various effects on wildlife in PWS after EVOS, although toxicity is noted as secondary to the physical impacts caused by fouling (e.g., hypothermia) (Albers and Loughlin, 2003), which may be reduced by chemical dispersion (Lessard and Demarco, 2000; CDC and ATSDR, 2010).

Based on the improbability of polar bears becoming significantly oiled by dispersed oil or under baseline conditions, it is not expected that polar bears will be adversely impacted due to the dispersion of oil. It is possible that minimal oiling will occur as a result of eliminating concentrated oil at the ocean's surface and the associated sensory cues for avoidance (i.e., smell and clearly visible sheen), but it is not expected that exposures to dilute, dispersed oil or dispersants will significantly impact polar bears at the individual level (i.e., reduced survival, growth, or reproduction). Similarly, indirect effects on polar bear prey are unlikely, as discussed in Sections 5.1.14 and 5.1.15, for ringed and bearded seal, respectively.

In the unlikely event that BMPs fail, and the implementation of the Unified Plan fails to protect marine mammals, polar bears may be adversely impacted by the application of dispersants. Potential impacts on polar bears in a worst-case scenario are provided in the main text of the BA.

5.1.12 Northern sea otter, southwest Alaska DPS

Sea otters are furbearing mammals that may be significantly impacted by the physical effects of oiling or dispersant exposure (Section 3.1). Otters that dive through heavily oiled surface waters will themselves become heavily oiled, resulting in a decreased ability to maintain their body temperature. Hypothermia and death could result (Geraci and St. Aubin, 1988). The application of chemical dispersants in areas that are heavily oiled at the ocean's surface would result in a decreased concentration, volume, and areal extent of surface oil, which would likely reduce the potential for oiling of sea otters.

Sea otters rely on critical nearshore habitat and shallow areas, where oiling could cause significant ecological damage and long-term effects (Peterson et al., 2003). The application of chemical dispersants is intended to reduce the oiling of shorelines (Fingas, 2008), thereby protecting sea otter habitat. The application of dispersants is not intended for nearshore application, so direct and concentrated exposures of sea otters to dispersants and dispersed oil are fairly unlikely (Section 2).

Sea otters groom their fur, which, if oiled, may result in ingestion of significant quantities of oil. The elimination of oil from the ocean's surface is expected to reduce oiling of sea otters, and therefore the ingestion of oil through grooming.

Inhalation and aspiration of oil is a potential route of exposure for sea otters, particularly because they spend much of their time at the water's surface (Kenyon, 1969; as cited in USFWS, 2010a; Riedman and Estes, 1990) or hauled out on the shoreline (Kenyon, 1969; as cited in USFWS, 2010a; Riedman and Estes, 1990). Chemical dispersion has been shown to reduce the evaporation of volatile oil components (NRC, 2013), which should in turn reduce the inhalation or aspiration of vapors by sea otters.

Clams, sea urchins, and finfish are the primary dietary components of sea otter (USFWS, 2010a), but they will shift their diet when certain species become scarce (USFWS, 2010a). Because sea otter are generalist feeders, it is unlikely that small changes in their prey base will cause significant impacts at the individual or population levels. The toxicity of oil alone is greater than that of dispersed oil (Sections 3.3 and 3.4, Figures 8 and 9), so chemical dispersion may reduce toxicity to the overall community, and indirect impacts on the food web are therefore not expected.[40] Chronic exposure of benthic species should be less under dispersed oil conditions than under baseline conditions (Humphrey et al., 1987).

Although PAHs and other hydrocarbons are known to accumulate in benthic invertebrates (Wolfe et al., 1998), such chemicals are unlikely to be biomagnified at higher trophic levels (Wolfe et al., 2001; Logan, 2007) due to more efficient PAH metabolisms in mammals (Albers and Loughlin, 2003). The impact of dietary PAHs in mammals is a point of uncertainty, discussed in Section 6.3.4. Acute exposures to PAHs have been linked to various effects on wildlife in PWS after EVOS, although toxicity is noted as secondary to the physical impacts caused by fouling (e.g., hypothermia) (Albers and Loughlin, 2003), which may be reduced by chemical dispersion (Lessard and Demarco, 2000; CDC and ATSDR, 2010). It is unclear whether such exposures would result in reduced survival, growth, or reproduction.

For these reasons, it is expected that sea otters will not be adversely impacted, either directly or indirectly, by the application of chemical dispersants relative to baseline oiling, particularly in the event that oil slick reaches nearshore, critical habitat.

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[40] The relative sensitivities of species that might be consumed by Northern sea otter (i.e., large epibenthic invertebrates, bivalves, and finfish) vary substantially, essentially bracketing the SSDs presented in Section 3.3 (Figure 7). Sensitive larval bivalves (e.g., *Crassostrea* sp.) may be more impacted by chemical dispersion of oil than larval or juvenile finfish. Adult bivalves may be less impacted over the long term in areas where oil is dispersed than in areas where oil is not treated. For example, increased rates of depuration of hydrocarbons in impacted benthos communities have been previously observed (Humphrey et al., 1987; Wolfe et al., 1998).

In the unlikely event that BMPs fail, and the implementation of the Unified Plan fails to protect marine mammals, Northern sea otters may be adversely impacted by the application of dispersants. Potential impacts on Northern sea otters in a worst-case scenario are provided in the main text of the BA.

5.1.13 Pacific walrus

Walruses are unlikely to be impacted by the physical effects of dispersants (Section 3.1). They rely on subcutaneous blubber to regulate their body heat, instead of fur, which could be compromised by oiling, dispersants, or dispersed oil.

This species is large and will not likely be exposed to concentrations of dispersants or dispersed oil in quantities great enough to cause acute toxic effects (e.g., mortality); such effects are unlikely even at lower trophic levels (Section 4). Dispersed oil rapidly dilutes and degrades over time, so chronic exposure to dispersants or dispersed oil is unlikely as well (Section 2). Sublethal impacts related to dispersed oil are certainly possible, but it is unlikely that dispersed oil will have a greater impact than oil alone, particularly on walruses, which frequently dive through the surface of water and use shoreline haulouts. Rather, oil alone is expected to cause greater toxicity (Section 3.1) due to its build up at the ocean's surface under baseline conditions (NRC, 2005).

The application of dispersants is expected to result in diminished oiling of shorelines (Fingas, 2008) and haulouts, as well as a reduced volume, concentration, and areal extent of oil at the ocean surface (NRC, 2005), where walruses could be exposed. Allowing haulouts or rookeries to be oiled (i.e., No Action alternative) may result in the chronic exposure of this species, as the oil degrades slowly on the shoreline over many years (Peterson et al., 2003).

Haulouts on sea ice are expected to be impacted differently by oil than shorelines, since ice does not trap and slowly release oil over time to the same extent as sediment. Still, baseline conditions in areas covered by sea ice are expected to cause substantial oiling of walruses that dive into water to forage, and the increased concentration of volatile oil at the surface (associated with baseline conditions) is expected to result in increased inhalation and aspiration of oil. This is particularly true at points where oil may concentrate, such as spatially constrained polynyas or breathing holes in the ice. Dispersants are expected to reduce the volatilization of oil by dissolving its lighter components (Section 2). Thus, the risk of inhalation or aspiration for hauled-out or surfacing walruses may diminish after dispersant application (NRC, 2013). Inhalation and aspiration of oil may have severe impacts on mammals (Section 3.1).

Ingestion of oil in the shallow water column (as deep as 10 m) may increase due to dispersion, but it has been shown that ingestion has less severe impacts on mammals than does inhalation (Section 3.1). Ingestion of PAHs is not expected to be a major source of parent PAH body burdens in marine mammals, because mammals are known to effectively metabolize and excrete PAHs (Albers and Loughlin, 2003). Ingested hydrocarbons are unlikely to accumulate or magnify in walruses over time as a result of chemical dispersion; exposures to PAHs after dispersion is expected to be acute rather than chronic (Section 2).

Acute exposures to PAHs have been linked to various effects on wildlife in PWS after EVOS, although toxicity is noted as secondary to the physical impacts caused by fouling (e.g., hypothermia) (Albers and Loughlin, 2003). Larger marine mammals with subcutaneous blubber (i.e., those that would not suffer from hypothermia) experienced sublethal impacts (e.g., lesions) after EVOS, although it was not determined whether those impacts corresponded to reductions in survival, growth, or reproduction (Albers and Loughlin, 2003).

Walruses are unique among the ESA-listed pinnipeds, in that they forage on benthic invertebrates (e.g., bivalves) exclusively (Richard, 1990; as cited in USFWS, 1994). These species are known to accumulate hydrocarbons and PAHs (Wolfe et al., 1998), although they do not readily transfer PAHs to higher trophic levels, which can efficiently metabolize those chemicals (Albers and Loughlin, 2003; Wolfe et al., 2001). The application of dispersants increases PAHs in the water column, which may increase the uptake of such chemicals in walrus prey species. It is not likely that this will provide a major route of exposure to toxic chemicals, but it could contribute to toxicity in sensitive prey species (e.g., Pacific oyster). Invertebrate larvae have been shown to be particularly sensitive to dispersants and dispersed oil (Attachment B-1). However, impacts on benthic communities are anticipated to be short-term and of a low magnitude (Mageau et al., 1987; Cross and Martin, 1987; Cross and Thomson, 1987); mass mortality has not occurred in field observations with dispersed oil. Still, long-term reproduction in bivalves may be inhibited by oil dispersion (Cross and Thomson, 1987), which may impact foraging by walruses. The potential for reduced populations of sensitive bivalves suggests that indirect impacts at the local scale are possible, as are indirect impacts at the individual walrus level.

The impact of dietary PAHs in mammals is a point of uncertainty, discussed in Section 6.3.4. Walrus are perhaps at a higher risk than other species, but it is not clear if sublethal impacts caused by PAHs will manifest as an effect on growth, survival, or reproduction, given that exposures to PAHs through the diet as a result of chemical dispersant application will likely attenuate within a year (Humphrey et al., 1987).

Based on the rationale provided in this section, it is not expected that Pacific walruses will be directly affected by dispersed oil or dispersants, however, indirect effects are possible, due to the selective diet of walrus on species that are particularly sensitive to dispersants and dispersed oil.

In the unlikely event that BMPs fail, and the implementation of the Unified Plan fails to protect marine mammals, Pacific walruses may be directly impacted by the application of dispersants. Potential direct impacts on Pacific walruses in a worst-case scenario are provided in the main text of the BA.

5.1.14 Ringed seal

Ringed seals are unlikely to be impacted by the physical effects of dispersants or dispersed oil (Section 3.1), because they use subcutaneous blubber to regulate body heat. Although slight surface oiling of seal fur may occur after oil is dispersed into the water column, the oil is expected to be dilute (Section 2) and less likely to stick to fur (CDC and ATSDR, 2010; Lessard and Demarco, 2000) than oil alone.

Ringed seals live near sea ice and maintain holes through which they can breathe or haul out to rest, pup, or molt (Kelly et al., 2010). Oil under ice could pool in breathing holes and affect seals that surface to breathe, or coat seals as they move in and out of the holes. Heavy coating of seal fur may result in localized irritation (Section 3.1). Surfacing in untreated oil poses a greater threat to ringed seal, as oil could be inhaled (volatile components) or aspirated (vapors and liquid oils) (Section 3.1), leading to various systemic impacts or death. The removal of oil from the ocean's surface by chemical dispersion should reduce the likelihood of such impacts.

Ringed seals primarily feed on fish and large epibenthic invertebrates under sea ice. These species are unlikely to be exposed to oil under baseline conditions as adults, but may be exposed to toxic levels at early life stages. As shown in Sections 3.3 and 3.4 and Figures 8 and 9, dispersants reduce the toxicity of crude oil to early life stages of aquatic species in general, although some sensitive species are more sensitive to dispersed oil. It is not expected that the application of dispersants will significantly impact adult benthic invertebrates or finfish (Section 4), nor will dispersants increase toxicity to sensitive life stages of benthic invertebrates or finfish relative to baseline conditions. Therefore, indirect impacts on ringed seals are unlikely.

Ingestion of dispersed oil is possible among ringed seals as they feed in the shallow water column, but they are not expected to ingest large volumes of oil in this way, since oil concentrations decrease rapidly over time and throughout the water column after chemical dispersion (Section 2). Ingestion of oil in the shallow water column (as deep as 10 m) may increase due to dispersion, but ingestion results in less severe impacts on mammals than does inhalation (Section 3.1). Ingestion of PAHs is not expected to be a major source of PAH body burdens in marine mammals, because mammals are known to effectively metabolize and excrete PAHs (Albers and Loughlin, 2003); ingested hydrocarbons are unlikely to magnify in ringed seals as a result of chemical dispersant applications. Body burdens are expected to return to background levels after depuration, metabolism, and excretion, particularly after a short-term exposure (Albers and Loughlin, 2003).

Based on the rationale presented in this section, ringed seals are not anticipated to be significantly impacted, either directly or indirectly, by chemical dispersion. Rather, under most circumstances, the removal of oil from the ocean's surface will benefit ringed seals, eliminating the most impactful routes of exposure and reducing toxicity to the planktonic base of the food web (i.e., early life stages of prey species).

In the unlikely event that BMPs fail, and the implementation of the Unified Plan fails to protect marine mammals, ringed seals may be adversely impacted by the application of dispersants. Potential impacts on ringed seals in a worst-case scenario are provided in the main text of the BA.

5.1.15 Bearded seal

Bearded seals are unlikely to be impacted by the physical effects of dispersants or dispersed oil (Section 3.1), because they use subcutaneous blubber to regulate body heat. Although slight surface oiling of seal fur may occur after oil is dispersed into the water column, the oil is expected to be dilute (Section 2) and less likely to stick to fur (CDC and ATSDR, 2010; Lessard and Demarco, 2000) than oil alone.

Bearded seals live near sea ice and maintain holes through which they can breathe or haul out to rest, pup, or molt (Cameron et al., 2010). Oil under ice could pool in breathing holes and affect seals that surface to breathe, or coat seals as they move in and out of the holes. Heavy coating of seal fur may result in localized irritation (Section 3.1). Surfacing in untreated oil poses a greater threat to bearded seal, as oil could be inhaled (volatile components) or aspirated (vapors and liquid oils) (Section 3.1), leading to various systemic impacts or death. The removal of oil from the ocean's surface by chemical dispersion should reduce the likelihood of such impacts.

Bearded seals primarily feed on large epibenthic invertebrates, bivalves, and benthic fish under sea ice (Cameron et al., 2010). These species are unlikely to be exposed to oil under baseline conditions as adults, but may be exposed to toxic levels at early life stages. As shown in Sections 3.3 and 3.4 and Figures 8 and 9, dispersants reduce the toxicity of crude oil to early life stages of aquatic species in general, although some species (e.g., bivalves) are more sensitive to dispersed oil than to oil alone (Attachment B-1). It is not expected that the application of dispersants will significantly impact adult benthic invertebrates (Section 4), nor will dispersants increase toxicity to sensitive life stages of benthic invertebrates relative to baseline conditions. Therefore, indirect impacts on bearded seals are unlikely.

Ingestion of dispersed oil is possible among bearded seals as they feed in the shallow water column, but they are not expected to ingest large volumes of oil in this way, since oil concentrations decrease rapidly over time and throughout the water column after chemical dispersion (Section 2). Ingestion of oil in the shallow water column (as deep as 10 m) may increase due to dispersion, but ingestion results in less severe impacts on mammals than does inhalation (Section 3.1). Ingestion of PAHs is not expected to be a major source of PAH body burdens in marine mammals, because mammals are known to effectively metabolize and excrete PAHs (Albers and Loughlin, 2003). Ingested hydrocarbons are unlikely to accumulate or magnify in bearded seals as a result of chemical dispersion; exposures to PAHs are likely to be acute rather than chronic due to dilution (Section 2.1) and biodegradation of oil and PAHs after chemical dispersion (Section 2.2). Acute exposures to PAHs have been linked to various effects on wildlife in PWS after EVOS, although toxicity is noted as secondary to the physical impacts caused by fouling (e.g., hypothermia) (Albers and Loughlin, 2003). It is not clear whether such exposures caused by the chemical dispersion of oil would result in reduced survival, growth, or reproduction.

Based on the rationale presented in this section, bearded seals are not anticipated to be significantly impacted, either directly or indirectly, by chemical dispersion. Rather, under most circumstances, the removal of oil from the ocean's surface will benefit bearded seals, eliminating the most impactful routes of exposure and reducing toxicity to the planktonic base of the food web (i.e., early life stages of prey species).

In the unlikely event that BMPs fail, and the implementation of the Unified Plan fails to protect marine mammals, bearded seals may be adversely impacted by the application of dispersants. Potential impacts on bearded seals in a worst-case scenario are provided in the main text of the BA.

5.2 BIRDS

As discussed in Section 3.1, bird species are at particular risk of exposure to baseline oiling, and are especially susceptible to the physical impacts of oiling.

5.2.1 Short-tailed albatross

Dispersants, if applied inappropriately, could result in severe impacts on the short-tailed albatross (Duerr et al., 2011). BMPs dictate monitoring for bird presence and avoiding the application of dispersants directly to birds on water or in flight; Butler et al. (1988) indicate that such BMPs are unlikely to be ignored. If BMPs are implemented and dispersants are not applied directly to short-tailed albatross, the impacts of surface oiling (Section 3.1) would assumedly be reduced. The reduced concentration, volume, and areal extent of an oil slick would limit the likelihood of exposure of birds found over open water.

Although embryotoxicity has been observed in response to dispersants and dispersed oil (Finch et al., 2012; Wooten et al., 2012; Albers, 1979 and Albers and Gay, 1982, both cited in Wooten et al., 2012), it is not clear whether short-tailed albatross oiled in Alaska waters transfer oil to their nestlings in Japan or Taiwan (USFWS, 2008). Since oiling is expected to lessen after dispersion (Section 2, Section 3.1; CDC and ATSDR, 2010; Lessard and Demarco, 2000), it is unlikely that dispersed oil would be transferred from Alaska waters to nestlings in Asia.

Short-tailed albatross feed mostly at the surface, diving from either the air or an on-water position for shallow fish (e.g., bonito [*Sarda* sp.], flying fish [*Exocoetidae* sp.], and sardines [*Clupeidae* sp.]) and invertebrates (i.e., squid, shrimp) (Hasegawa and DeGange, 1982; Tickell, 1975, 2000; all cited in USFWS, 2008). Since the prey of the short-tailed albatross reside in the shallow ocean, they are susceptible to exposure to oil and dispersed oil. Based on the analyses presented in Sections 3.3 and 3.4, dispersants can reduce the toxicity of oil to these species relative to baseline conditions

(Figures 8 and 9). Thus, it is unlikely that dispersants will have adverse indirect effects on the short-tailed albatross.

While PAHs are known to increase in concentration in dispersed oil plumes relative to baseline conditions (Ramachandran et al., 2004), acute toxicity is generally not increased (Sections 3.3 and 3.4, Figures 8 and 9). Furthermore, the uptake and trophic transfer of PAHs to fish is limited by their efficient metabolisms (Wolfe et al., 2001; Logan, 2007; Payne et al., 2003). Long-term uptake is likely limited by the acute nature of dispersed oil plume exposure, given natural transport mechanisms, rapid dilution, and increased rates of biodegradation (Section 2). Alterations to the bioavailability of PAHs caused by oil dispersion will not likely increase the body burden of PAHs in short-tailed albatross, since exposures to increased PAHs will be acute rather than chronic; chronic exposures tend to result in increased body burdens over time (Albers and Loughlin, 2003). Acute exposures to PAHs have been linked to various effects on wildlife in PWS after EVOS, although toxicity is noted as secondary to the physical impacts caused by fouling (e.g., hypothermia) (Albers and Loughlin, 2003), which may be reduced by chemical dispersion (CDC and ATSDR, 2010; Lessard and Demarco, 2000). It is unclear whether PAH exposures in bird species would result in reduced survival, growth, or reproduction (Section 6.3.3).

Ingestion, aspiration, and inhalation of oil by short-tailed albatross during flight, feeding, and preening are all likely to be much greater under baseline conditions (Sections 2 and 3.1). The removal of oil from the ocean's surface will effectively reduce the volume, concentration, and areal extent (i.e., likelihood of encounter) of oil to which this species will be exposed.

Based on the rationale presented in this section, short-tailed albatross is not anticipated to be significantly impacted, either directly or indirectly, by chemical dispersion. Rather, under most circumstances, the removal of oil from the ocean's surface will benefit short-tailed albatross by eliminating the most impactful routes of exposure and reducing toxicity to the planktonic base of the food web (i.e., early life stages of prey species), as well as adult prey species of fish and invertebrates.

In the unlikely event that BMPs fail, and the implementation of the Unified Plan fails to protect marine birds, short-tailed albatrosses may be adversely impacted by the application of dispersants. Potential impacts on short-tailed albatrosses in a worst-case scenario are provided in the main text of the BA.

5.2.2 Spectacled eider

Dispersants, if applied inappropriately, could result in severe impacts on the spectacled eider (Duerr et al., 2011; Jenssen and Ekker, 1991a, b). BMPs dictate monitoring for bird presence and avoiding the application of dispersants directly to birds on water or in flight; Butler et al. (1988) indicate that such BMPs are unlikely to be ignored. If BMPs are implemented and dispersants are not applied directly to spectacled eider, the impact of surface oiling (Section 3.1) would assumedly be reduced. The reduced concentration, volume, and areal extent of an oil slick would limit the likelihood of exposure of birds found over open water. This is particularly important for spectacled eider, which congregate in very limited areas (i.e., wintering habitat), many of which are listed as critical habitat (66 FR 9146, 2001).

Critical habitat for spectacled eider includes vegetated intertidal habitat on the Yukon-Kuskokwim (Y-K) Delta, shallow (between 5 and 15 m) marine waters in Norton Sound, and relatively deep waters (as deep as 75 m) between St. Matthew and St. Lawrence Islands in the Bering Sea. Although physical impacts would likely be most pronounced in wintering habitat (i.e., Bering Sea) due to low temperatures and the cooling effect of water (Section 3.1), baseline oiling effects on habitat would likely be greatest in the molting and breeding areas, where shorelines might trap oil and slowly release it over time (Peterson et al., 2003). The application of dispersants to an oil spill on the open ocean before it reaches these critical habitats would likely reduce the extent of oiling (Sections 2 and 3.1) and the long-term impacts on the benthic community (Section 3.1).

Embryotoxicity in birds has been observed in response to dispersants and dispersed oil (Finch et al., 2012; Wooten et al., 2012; Albers, 1979 and Albers and Gay, 1982, both cited in Wooten et al., 2012). Since oiling is expected to lessen after dispersion (Section 2, Section 3.1; CDC and ATSDR, 2010; Lessard and Demarco, 2000), it is less likely that oil would be transferred from nesting eiders to nestlings. This assumes that dispersants are applied at a distance from eider populations and critical habitat, in accordance with BMPs (Alaska Clean Seas, 2010).

Spectacled eider mostly feed on benthic invertebrates (Petersen et al., 1999) in shallow waters during much of the year, although they move to deeper waters in winter. As their prey base is generally within the upper 15 m of the water column, some exposure of prey to dispersed oil may occur, and early life stages of prey may be exposed to both oil and dispersed oil. The application of chemical dispersant is expected to decrease the toxicity to the overall planktonic community (including sensitive life stages of prey), so such an application is not expected to have adverse impacts on eider prey overall. Certain sensitive prey species (e.g., bivalve larvae) may be at greater risk of chemical toxicity (Figures 3 through 7), so indirect impacts may occur at times when eider diets are primarily composed of bivalve tissues (May through July) (USFWS, 1996). Invertebrate larvae have been shown to be particularly sensitive to dispersants and dispersed oil (Attachment B-1). However, impacts on benthic communities are anticipated to be short-term and of low magnitude (Mageau et al., 1987; Cross and Martin, 1987; Cross and Thomson, 1987); mass mortality has not occurred in field observations with dispersed oil. Still, long-term reproduction in bivalves may be inhibited by oil dispersion (Cross and Thomson, 1987), which may impact foraging by eiders. The potential for reduced populations of sensitive bivalves suggests that indirect impacts at the local scale are possible, as are indirect impacts at the individual eider level.

While PAHs are known to increase in bioavailability in dispersed oil plumes relative to baseline conditions (Section 2), toxicity is generally not increased (Sections 3.3 and 3.4, Figures 8 and 9). Furthermore, the uptake and trophic transfer of PAHs to fish is limited by their efficient metabolisms (Wolfe et al., 2001; Logan, 2007). Alterations to the bioavailability of PAHs caused by dispersed oil will not likely increase the body burden of PAHs in spectacled eider over time (Albers and Loughlin, 2003). The exposure of spectacled eider to PAHs after chemical dispersion is likely to be acute rather than chronic (due to dilution and degradation of oil components after chemical dispersion) (Sections 2.1 and 2.2), so body burdens are likely to decrease over time as dissolved PAH concentrations in the environment, which were increased as a result of chemical dispersion, are metabolized and excreted by spectacled eider. The uptake of PAHs in diet is also expected to decrease over time, as PAHs and other oil components are depurated and degraded in prey tissues (e.g., bivalves) (Humphrey et al., 1987). It should be noted that chemical dispersant application is not intended for shallow, nearshore habitats where eider are likely to be feeding on invertebrates, so exposures to dispersed oil are likely to occur after dilution and biodegradation have already begun to decrease the concentration of oil components in the water column. It is not clear whether sublethal impacts resulting from short-term PAH exposures (enhanced by chemical dispersion) would result in reduced survival, growth, or reproduction in bird species (Section 6.3.3).

Ingestion, aspiration, and inhalation of oil by spectacled eider during flight, feeding, and preening are all likely to be much greater under baseline conditions (Sections 2 and 3.1). The removal of oil from the ocean's surface will effectively reduce the volume, concentration, and areal extent (i.e., likelihood of encounter) of oil to which this species will be exposed.

Based on the rationale presented in this section, spectacled eider may be significantly impacted, either directly or indirectly, by chemical dispersion. Although, the removal of oil from the ocean surface will benefit spectacled eider by eliminating the most impactful routes of exposure to oil, their prey, which is at times limited to more sensitive species, could be impacted by chemical dispersion of oil close to nearshore habitats (although dispersion is not intended for use within nearshore habitats).

5.2.3 Steller's eider

Dispersants, if applied inappropriately, could result in severe impacts on the Steller's eider (Duerr et al., 2011; Jenssen and Ekker, 1991a, b). BMPs dictate monitoring for bird presence and avoiding applying dispersants directly to birds on water or in flight; Butler et al. (1988) indicate that such BMPs are unlikely to be ignored. If BMPs are implemented and dispersants are not directly applied to Steller's eider, the impact of surface oiling (Section 3.1) would assumedly be reduced. The reduced concentration, volume, and areal extent of an oil slick would limit the likelihood of exposure of birds found over open water. This is particularly important for Steller's eider, which congregate in very limited areas (i.e., critical breeding habitat) (66 FR 9146, 2001). Also, Steller's eider molt on water and are flightless for approximately three weeks during the late summer (between July and October) (Petersen, 1981; as cited in USFWS, 2002), during which time oiling could result in significant impacts (Section 3.1); this is based on the assumption that post-molt plumage is more sensitive to oil than fully developed plumage. Dispersant application would reduce the amount (i.e., concentration, volume, and areal extent) of oil that enters Steller's eider critical habitat (Section 3.4.2.3.1 of the BA) and the time that the oil remains on the surface (Section 2).

Critical habitat for Steller's eider includes vegetated intertidal areas on the Y-K Delta, open marine waters up to 9 m deep, and associated eelgrass beds and the benthic invertebrate communities in that area; additional habitat can be found along the Aleutian Islands. Impacts are most likely to occur in the southern critical habitat along the Aleutian Islands, due to the prevalence of spills in that area (Appendix D to the BA). However, baseline oiling effects on habitat are likely to be greatest in the breeding and nesting areas on the Y-K Delta and near Barrow, Alaska (USFWS, 2002); oil on the shorelines and forage habitat of these areas could result in significant oiling of nesting birds and nestlings, as well as chronic exposures of the benthic community to oil trapped in sediment along the intertidal shoreline (Peterson et al., 2003; Cross and Thomson, 1987). The application of dispersants to an oil spill on the open ocean before it reaches these critical habitats would likely reduce the extent of oiling (Sections 2 and 3.1) and the long-term impacts to the benthic community (Peterson et al., 2003; Cross and Thomson, 1987). The application of dispersants in shallow, nearshore habitats is not an approved use, so dispersed oil that moves into Steller's eider critical habitat will already have begun to dilute and biodegrade (Sections 2.1 and 2.2).

Embryotoxicity in birds has been observed in response to dispersants and dispersed oil (Finch et al., 2012; Wooten et al., 2012; Albers, 1979 and Albers and Gay, 1982, both cited in Wooten et al., 2012). Since oiling is expected to lessen after dispersion (Section B2, Section B3.1; CDC and ATSDR, 2010; Lessard and Demarco, 2000), it is less likely that oil would be transferred from nesting eiders to nestlings. This assumes that dispersants are applied at a distance from eider populations and critical habitat in accordance with BMPs (Alaska Clean Seas, 2010).

Steller's eider mostly feed on benthic invertebrates (Petersen, 1981; as cited in USFWS, 2002) in shallow waters during much of the year. Their prey base generally resides in shallow waters, based on where they congregate (Section 3.4.2.3.1 of the BA), indicating that some exposure to dispersed oil may occur. Early life stages of prey may be exposed to both oil and dispersed oil. The application of chemical dispersant is expected to decrease the toxicity to the overall planktonic community (including sensitive life stages of prey), so such an application is not expected to have adverse impacts to Steller's eider prey overall. However, larvae of certain invertebrate species have been shown to be particularly sensitive to dispersants and dispersed oil

(Attachment B-1, Figures 3 through 7). Impacts on benthic communities tend to be short-term and of low magnitude (Mageau et al., 1987; Cross and Martin, 1987; Cross and Thomson, 1987), and mass mortality has not occurred in field observations with dispersed oil. Still, long-term reproduction in bivalves may be inhibited by oil dispersion (Cross and Thomson, 1987), which may impact foraging by eiders. The potential for reduced populations of sensitive bivalves suggests that indirect impacts at the local scale are possible, as are indirect impacts at the individual eider level.

While PAHs are known to increase in concentration in dispersed oil plumes relative to baseline conditions (Section 2), toxicity is generally not increased (Sections 3.3 and 3.4, Figures 8 and 9). Furthermore, uptake and trophic transfer of PAHs to fish is limited by their efficient metabolisms (Wolfe et al., 2001). Alterations to the bioavailability of PAHs caused by oil dispersion will not likely increase the body burden of PAHs in Steller's eider over time (Albers and Loughlin, 2003). The exposure of Steller's eider to PAHs after chemical dispersion is likely to be acute rather than chronic (due to dilution and degradation of oil components after chemical dispersion) (Sections 2.1 and 2.2), so body burdens are likely to decrease over time as dissolved PAH concentrations in the environment, which were increased as a result of chemical dispersion, are metabolized and excreted by Steller's eider. The uptake of PAHs in diet is also expected to decrease over time, as PAHs and other oil components are depurated and degraded in prey tissues (e.g., bivalves) (Humphrey et al., 1987).

Acute exposures to PAHs have been linked to various effects on wildlife in PWS after EVOS, although toxicity is noted as secondary to the physical impacts caused by fouling (e.g., hypothermia) (Albers and Loughlin, 2003), which may be reduced by the application of chemical dispersant (Lessard and Demarco, 2000; CDC and ATSDR, 2010). It is not clear whether sublethal impacts resulting from short-term PAH exposures (enhanced by chemical dispersion) would result in reduced survival, growth, or reproduction in bird species (Section 6.3.3).

Ingestion, aspiration, and inhalation of oil by Steller's eider during flight, feeding, and preening are all likely to be much greater under baseline conditions (Sections 2 and 3.1). The removal of oil from the ocean's surface will effectively reduce the volume, concentration, and areal extent (i.e., likelihood of encounter) of oil to which this species will be exposed.

Based on the rationale presented in this section, Steller's eider may be significantly impacted, either directly or indirectly, by chemical dispersion. Although, the removal of oil from the ocean's surface will benefit Steller's eider by eliminating the most impactful routes of exposure to oil, their prey, which is at times limited to more sensitive species, could be impacted by chemical dispersion of oil close to nearshore habitats (although dispersion is not intended for use within nearshore habitats).

5.2.4 Kittlitz's murrelet

Dispersants, if applied inappropriately, could result in severe impacts on the Kittlitz's murrelet (Duerr et al., 2011; Jenssen and Ekker, 1991a, b). BMPs dictate monitoring for bird presence and avoiding the application of dispersants directly to birds on water or in flight; Butler et al. (1988) indicate that such BMPs are unlikely to be ignored. It is expected that the reduced concentration, volume, and areal extent of an oil slick resulting from dispersant application in open water would limit the likelihood of exposure of birds found in the nearshore environment, or in polynyas and glacial meltwaters (Sections 2 and 3.1; Day et al., 1999; Day et al., 2011).

Embryotoxicity in birds has been observed in response to dispersants and dispersed oil (Finch et al., 2012; Wooten et al., 2012; Albers, 1979 and Albers and Gay, 1982, both cited in Wooten et al., 2012). Since oiling is expected to lessen after dispersion (Sections 2 and 3.1; CDC and ATSDR, 2010; Lessard and Demarco, 2000), it is less likely that oil would be transferred from nesting murrelets to nestlings. This assumes that dispersants are applied at a distance from Kittlitz's murrelet populations in accordance with BMPs (Alaska Clean Seas, 2010). Nesting habitat is typically removed from areas where such applications might occur, in coarse, rocky, and uneven ground or skree (USFWS, 2006); these features are associated with glaciated (or formerly glaciated) habitats on alpine terrain (van Pelt and Piatt, 2003). To a lesser extent, Kittlitz's murrelet nest in crevasses of cliffs, potentially near the coast (Day et al., 1999); dispersants and dispersed oil are unlikely to encounter these hidden nesting areas.

Kittlitz's murrelet mostly feed by diving after schooling fish (e.g., capelin, sand lance [*Ammodytidae* sp.], herring, and juvenile walleye) (Day et al., 1999), but may switch seasonally to feed on what is available (Hobson et al., 1994; as cited in USFWS, 2011b; Day et al., 1999; Day and Nigro, 2000; Day et al., 2011). Kittlitz's murrelet is predominately piscivorous, but they will also feed on crustaceans such as euphausiids (Hobson et al., 1994; as cited in USFWS, 2011b) (Hobson et al., 1994; as cited in USFWS, 2011b; Day et al., 1999; Day and Nigro, 2000; Day et al., 2011). Exposure of murrelet prey species to both oil and dispersed oil may occur due to the shallow depths at which murrelet feed (i.e., nearshore and shallow offshore) (Day et al., 1999; Day and Nigro, 2000; Day et al., 2011). The application of chemical dispersant is expected to decrease toxicity to the overall planktonic community (including sensitive life stages of prey) (Sections 3.3 and 3.4, Figures 8 and 9), and dispersants are expected to protect nearshore habitats and shorelines (Fingas, 2008) that support Kittlitz's murrelet and its prey (Day et al., 1999; Day and Nigro, 2000; Day et al., 2011). One notable exception may be spawning species that could potentially be impacted by oil or dispersed oil (Section 5.3.4); it is possible that oil is less toxic to embryonic or larval herring species than dispersed oil, although the long-term impacts of shoreline and vegetation oiling (Peterson et al., 2003) may be more lasting (Humphrey et al., 1987).

While PAHs are known to increase in concentration in dispersed oil plumes relative to baseline conditions (Section 2), toxicity is generally not increased (Sections 3.3 and 3.4, Figures 8 and 9). Furthermore, the uptake and trophic transfer of PAHs to fish is limited by their efficient metabolisms (Wolfe et al., 2001; Logan, 2007). Alterations to the bioavailability of PAHs caused by oil dispersion will not likely increase the body burden of PAHs in Kittlitz's murrelet over time (Albers and Loughlin, 2003). The exposure of Kittlitz's murrelet to PAHs after chemical dispersion is likely to be acute rather than chronic (due to dilution and degradation of oil components after chemical dispersion) (Sections 2.1 and 2.2), so body burdens are likely to decrease over time as dissolved PAH concentrations in the environment, which were increased as a result of chemical dispersion, are metabolized and excreted by Kittlitz's murrelet. The uptake of PAHs in diet is also expected to decrease over time, as PAHs and other oil components are depurated and degraded in prey tissues (e.g., fish) (Wolfe et al., 2001; Wolfe et al., 1998; Logan, 2007).

Acute exposures to PAHs have been linked to various effects on wildlife in PWS after EVOS, although toxicity is noted as secondary to the physical impacts caused by fouling (e.g., hypothermia) (Albers and Loughlin, 2003), which may be reduced by the application of chemical dispersant (Lessard and Demarco, 2000; CDC and ATSDR, 2010). It is not clear whether sublethal impacts resulting from short-term PAH exposures (enhanced by chemical dispersion) would result in reduced survival, growth, or reproduction in bird species (Section 6.3.3).

Ingestion, aspiration, and inhalation of oil by Kittlitz's murrelet during flight, feeding, and preening are all likely to be much greater under baseline conditions (Sections 2 and 3.1). The removal of oil from the ocean's surface will effectively reduce the volume, concentration, and areal extent (i.e., likelihood of encounter) of oil to which this species will be exposed (Sections 2 and 3).

Based on the rationale presented in this section, Kittlitz's murrelet is not anticipated to be significantly impacted, either directly or indirectly, by chemical dispersion. Rather, under most circumstances, the removal of oil from the ocean's surface will benefit Kittlitz's murrelet by eliminating the most impactful routes of exposure to oil and reducing toxicity to the planktonic base of the food web (i.e., early life stages of prey species, winter forage) (Day et al., 1999; Day and Nigro, 2000; Day et al., 2011).

In the unlikely event that BMPs fail, and the implementation of the Unified Plan fails to protect marine birds, Kittlitz's murrelets may be adversely impacted by the application of dispersants. Potential impacts on Kittlitz's murrelets in a worst-case scenario are provided in the main text of the BA.

5.2.5 Yellow-billed loon

Dispersants, if applied inappropriately, could result in severe impacts on yellow-billed loons (Duerr et al., 2011; Jenssen and Ekker, 1991a, b). BMPs dictate monitoring for bird presence and avoiding the application of dispersants directly to birds on water or in flight; Butler et al. (1988) indicate that such BMPs are unlikely to be ignored. This is particularly true due to the fact that yellow-billed loon tend to be found in the uplands near permanent freshwater lakes (Earnst et al., 2006).

Exposures of yellow-billed loon to dispersants or dispersed oil are very unlikely during warm seasons, when they inhabit upland areas, but this species winters in coastal areas of the Aleutian Islands, Gulf of Alaska (GOA), Prince William Sound (PWS), Cook Inlet, Southeast Alaska (74 FR 12932, 2009), and particularly in Southeast Alaska south of Kodiak Island (North, 1994).[41] Although many spills have occurred in these areas since 1995 (Appendix D to the BA, Section 3.1.1 of the BA), the majority occurred during summer months. Crude oil was rarely spilled in these areas, although two crude oil spills have occurred in Cook Inlet during winter (Section 3.1.1). Oil spilled in loon habitat that is allowed to reach the coastal nearshore environment, particularly protected embayments less than 30 m deep (Strann and Østnes, 2007; as cited in USFWS, 2010b), could result in exposure and serious physical impacts. The reduced concentration, volume, and areal extent of an oil slick resulting from dispersant application in open water would limit the likelihood of exposure of birds found in the nearshore environment (Sections 2 and 3.1).

Yellow-billed loon migrate north in spring to breeding and nesting areas, particularly on the North Slope; on the way, loon stop periodically in groups in melting polynyas (2010b). Oiling in polynyas may be concentrated and cause serious harm to yellow-billed loon. It is expected that dispersion will reduce the exposure of this species to oil in polynyas (CDC and ATSDR, 2010; Lessard and Demarco, 2000), since the oil is removed quickly and effectively from the surface (Section 2.1).

Embryotoxicity in birds has been observed in response to dispersants and dispersed oil (Finch et al., 2012; Wooten et al., 2012; Albers, 1979 and Albers and Gay, 1982, both cited in Wooten et al., 2012). Since oiling is expected to lessen after dispersion (Section 2, Section 3.1;CDC and ATSDR, 2010; Lessard and Demarco, 2000), it is less likely that oil would be transferred from nesting loons to nestlings. Nesting generally occurs in the uplands, away from oiling, so direct application of dispersants to nests is unlikely.

Yellow-billed loon mostly feed by diving after small fish (e.g., stickleback [*Gasterosteidae* sp.] and least cisco [*Coregonus sardinella*]) and invertebrates (Earnst et al., 2006; North and Ryan, 1989; North, 1994; USFWS, 2010b). Exposure of loon prey to both oil and dispersed oil may occur due to the shallow depths at which loon feed (i.e., shallow coastal nearshore) (Strann and Østnes, 2007; as cited in USFWS, 2010b). The application of chemical dispersant is expected to decrease toxicity to the overall planktonic community (including sensitive life stages of prey) (Sections 3.3 and 3.4, Figures 8 and 9), and to protect nearshore habitats and shorelines (Fingas, 2008) that support yellow-billed loon and its prey. One notable exception may be spawning

[41] Southeast Alaska has been the site of frequent releases of diesel fuel (Appendix D), although diesel fuel is not a substance that is likely to be dispersed due to its volatility.

species that could potentially be impacted by oil or dispersed oil (Section 5.3.4); it is possible that oil is less toxic to embryonic or larval herring species than dispersed oil (Section 5.3.4), although the long-term impacts of shoreline and vegetation oiling (Peterson et al., 2003) may be more lasting (Humphrey et al., 1987; Section 2).

While PAHs are known to increase in concentration in dispersed oil plumes relative to baseline conditions (Section 2), toxicity is generally not increased (Sections 3.3 and 3.4, Figures 8 and 9). Furthermore, the uptake and trophic transfer of PAHs to fish is limited by their efficient metabolisms (Wolfe et al., 2001). Alterations to the bioavailability of PAHs caused by oil dispersion will not likely increase the body burden of PAHs in yellow-billed loon over time (Albers and Loughlin, 2003). The exposure of yellow-billed loon to PAHs after chemical dispersion is likely to be acute rather than chronic (due to dilution and degradation of oil components after chemical dispersion) (Sections 2.1 and 2.2), so body burdens are likely to decrease over time as dissolved PAH concentrations in the environment, which were increased as a result of chemical dispersion, are metabolized and excreted. The uptake of PAHs in diet is also expected to decrease over time, as PAHs and other oil components are depurated and degraded in prey tissues (e.g., fish, bivalves, and other macroinvertebrates) (Wolfe et al., 2001; Wolfe et al., 1998; Logan, 2007; Humphrey et al., 1987).

Acute exposures to PAHs have been linked to various effects on wildlife in PWS after EVOS, although toxicity is noted as secondary to the physical impacts caused by fouling (e.g., hypothermia) (Albers and Loughlin, 2003), which may be reduced by the application of chemical dispersant (Lessard and Demarco, 2000; CDC and ATSDR, 2010). It is not clear whether sublethal impacts resulting from short-term PAH exposures (enhanced by chemical dispersion) would result in reduced survival, growth, or reproduction in bird species (Section 6.3.3).

Ingestion, aspiration, and inhalation of oil by yellow-billed loon during flight, feeding, and preening are all likely to be much greater under baseline conditions (Sections 2 and 3.1). The removal of oil from the ocean's surface will effectively reduce the volume, concentration, and areal extent (i.e., likelihood of encounter) of oil to which this species will be exposed.

Based on the rationale presented in this section, yellow-billed loon are not anticipated to be significantly impacted, either directly or indirectly, by chemical dispersion. Rather, under most circumstances, the removal of oil from the ocean's surface will benefit yellow-billed loon by eliminating the most impactful routes of exposure to oil and reducing toxicity of oil to the planktonic base of the food web (i.e., early life stages of prey species, winter forage) (Strann and Østnes, 2007; as cited in USFWS, 2010b).

In the unlikely event that BMPs fail, and the implementation of the Unified Plan fails to protect marine birds, yellow-billed loons may be adversely impacted by the application of dispersants. Potential impacts on yellow-billed loons in a worst-case scenario are provided in the main text of the BA.

5.3 FISH

5.3.1 Chinook salmon, all ESUs

Non-spawning adult and juvenile Chinook salmon may be found in Alaska, offshore or in coastal areas, living relatively deep in the water column (i.e., 30 to 70 m) (NMFS, 2005; Healey, 1991). It is unlikely that this species will be exposed to oil under baseline conditions. It is possible that dispersed oil will reach depths at which Chinook salmon are present, but it will be dilute, particularly at or beyond 10 m deep (Section 2).

Since Chinook salmon are among the most insensitive species to have been tested in exposures to oil and dispersed oil (Figures 4 through 6; Attachment B-1), it is likely that this species is particularly resilient, even as juveniles, relative to the entire aquatic community. Sensitive life stages of this salmonid are not found in Alaska, and thus cannot be exposed to dispersants or dispersed oil.

The larvae of salmon prey may be found in the upper water column during certain times of the year, and may be exposed to both concentrated oil and dispersed oil. Based on the assessment in Sections 3.2 through 3.4, it is likely that the toxicity of oil to Chinook salmon and its prey will decrease after dispersant application.

Fish species are able to efficiently metabolize and excrete PAHs (Payne et al., 2003; Wolfe et al., 2001; Logan, 2007), so the markedly increased dissolved PAHs in the water column resulting from chemical dispersion (Ramachandran et al., 2004) do not biomagnify in fish tissues and transfer to higher trophic levels (i.e., piscivorous salmonids) (Payne et al., 2003; Wolfe et al., 2001; Logan, 2007). The toxicity of PAHs to early-life-stage fish species is addressed indirectly in Sections 3.2.4 through 3.2.5.3 (given that PAHs are a component of the oil and dispersed oil used in toxicity tests), and uncertainties involved with the analysis of PAH toxicity in fish are provided in Sections 6.2 (general analytical uncertainties) and 6.3.2 (specific to fish). For example, it is unclear whether sublethal impacts caused by increased PAH exposures after chemical dispersion would lead to decreased survival, growth, or reproduction in juvenile and adult salmon species.

Due to the relatively low expected exposure of Chinook salmon, their insensitivity to dispersed oil as adults and juveniles, and the low likelihood that their prey population will be impacted (relative to the baseline condition), Chinook salmon are not anticipated to be negatively impacted by the application of dispersants in Alaska waters.

In the unlikely event that BMPs fail, and the implementation of the Unified Plan fails to protect species of salmon, Chinook salmon may be adversely impacted by the application of dispersants. Potential impacts on Chinook salmon in a worst-case scenario are provided in the main text of the BA.

5.3.2 Coho salmon, Lower Columbia River ESU

Non-spawning adult and juvenile coho salmon may be found in Alaska, offshore or in coastal areas (Morris et al., 2007; Favorite, 1965), living relatively deep in the water column. It is unlikely that this species will be exposed to oil under baseline conditions. It is possible that dispersed oil will reach depths at which coho salmon are present, but it will be dilute, particularly at or beyond 10 m (Section 2).

Coho salmon appear to be highly sensitive to oil alone, although it is unknown whether they are sensitive to dispersants alone or dispersed oil (Attachment B-1). Based on the genus geometric mean LC50 values for *Oncorhynchus* sp., this group is relatively insensitive to dispersed oil and dispersants, Corexit® 9500 in particular (Figure 4). It is therefore likely that coho salmon are less sensitive to dispersed oil than to oil alone, based on the general trend in the whole community (Sections 3.3 and 3.4, Figures 8 and 9) and the relative sensitivity of Chinook salmon (Sections 3.2 and 5.3.1).

The larvae of salmon prey may be found in the upper water column during certain times of the year, and may be exposed to both concentrated oil and dispersed oil. Based on the assessment in Sections 3.2 through 3.4, it is likely that the toxicity of oil to coho salmon and its prey will decrease after dispersant application.

Fish species are able to efficiently metabolize and excrete PAHs (Douben, 2003; Wolfe et al., 2001), so the markedly increased dissolved PAHs in the water column resulting from chemical dispersion (Ramachandran et al., 2004) do not biomagnify in fish tissues and transfer to higher trophic levels (i.e., piscivorous salmonids) (Payne et al., 2003; Wolfe et al., 2001; Logan, 2007). The toxicity of PAHs to early-life-stages of various fish species is addressed indirectly in Sections 3.2.4 through 3.2.5.3 (given that PAHs are a component of the oil and dispersed oil used in toxicity tests), and uncertainties involved with the analysis of PAH toxicity in fish are provided in Sections 6.2 (general analytical uncertainties) and 6.3.2 (specific to fish). For example, it is unclear whether sublethal impacts caused by increased PAH exposures after chemical dispersion would lead to decreased survival, growth, or reproduction in juvenile and adult salmon species.

Due to the relatively low expected exposure of coho salmon, their insensitivity to dispersed oil as adults and juveniles, and the low likelihood that their prey population will be impacted (relative to the baseline condition), coho salmon are not anticipated to be negatively impacted by the application of dispersants in Alaska waters.

In the unlikely event that BMPs fail, and the implementation of the Unified Plan fails to protect species of salmon, coho salmon may be adversely impacted by the application of dispersants. Potential impacts on coho salmon in a worst-case scenario are provided in the main text of the BA.

5.3.3 Steelhead trout, all DPS

Non-spawning adult and juvenile steelhead trout may be found in Alaska, offshore or in coastal areas (Sheppard, 1972; as cited in Laufle et al., 1986; Burgner et al., 1992; as cited in McKinnell et al., 1997); they live relatively deep in the water column, where they feed on benthic species (ADF&G, 2012; NOAA, 2011). It is unlikely that this species will be exposed to oil under baseline conditions. It is possible that dispersed oil will reach depths at which steelhead trout are present, but it will be very dilute, particularly at or beyond 10 m deep (Section 2.1).

Rainbow trout (which are not a genetically different species from steelhead trout) appear to be highly insensitive to dispersants alone, although it is unknown whether they are sensitive to oil alone or dispersed oil (Attachment B-1). Based on the genus geometric mean LC50 values for *Oncorhynchus* sp., this group is relatively insensitive to dispersed oil (Attachment B-1), but moderately sensitive to oil alone. It is likely that steelhead trout are less sensitive to dispersed oil than to oil alone, based on the general trend in the whole community (Sections 3.3 and 3.4, Figures 8 and 9) and the relative sensitivities of related salmonids (Sections 3.2 and 5.3.1).

The larvae of salmon prey may be found in the upper water column during certain times of the year, and may be exposed to both concentrated oil and dispersed oil. Based on the assessment in Sections 3.2 through 3.4, it is likely that the toxicity of oil to steelhead trout and its prey will decrease after dispersant application.

Fish species are able to efficiently metabolize and excrete PAHs (Douben, 2003; Wolfe et al., 2001), so the markedly increased dissolved PAHs in the water column resulting from chemical dispersion (Ramachandran et al., 2004) do not biomagnify in fish tissues and transfer to higher trophic levels (i.e., piscivorous salmonids) (Payne et al., 2003; Wolfe et al., 2001; Logan, 2007). The toxicity of PAHs to early-life-stage fish species is addressed indirectly in Sections 3.2.4 through 3.2.5.3 (given that PAHs are a component of the oil and dispersed oil used in toxicity tests), and uncertainties involved with the analysis of PAH toxicity in fish are provided in Sections 6.2 (general analytical uncertainties) and 6.3.2 (specific to fish). For example, it is unclear whether sublethal impacts caused by increased PAH exposures after chemical dispersion would lead to decreased survival, growth, or reproduction in juvenile and adult salmon species.

Due to the relatively low expected exposure of steelhead trout, their insensitivity to dispersed oil as adults and juveniles, and the low likelihood that their prey population will be impacted (relative to the baseline condition), steelhead trout are not anticipated to be negatively impacted by the application of dispersants in Alaska waters.

In the unlikely event that BMPs fail, and the implementation of the Unified Plan fails to protect species of salmonids, steelhead trout may be adversely impacted by the application of dispersants. Potential impacts on steelhead trout in a worst-case scenario are provided in the main text of the BA.

5.3.4 Pacific herring

Pacific herring are found throughout Alaska waters seasonally (Mecklenburg et al., 2002), and are important prey for many larger species of fish, birds, and marine mammals. They live throughout the water column a depth of 400 m (NOAA Fisheries, 2013), and therefore may be exposed to dispersed oil when in the upper 10 m (Section 2). In Southeast Alaska, spawning generally occurs in nearshore environments with organic, semi-protected, and partially mobile substrate (NMFS, 2007), such as eelgrass or kelp. These areas are also highly susceptible to oiling (Peterson et al., 2003), (consistent with baseline conditions) so chemical dispersants may practicably be used to protect such habitats (Fingas, 2008).

Toxicity testing indicates that Pacific herring is particularly sensitive to oil alone (Rice et al., 1979; cited in Barron et al., 2013). Lee et al. (2011b) showed that although oil was slightly more toxic to Pacific herring than dispersed oil, both were highly toxic at low, ecologically relevant (Section 2) concentrations and at short exposure durations (i.e., 6 hours). This indicates that the application of chemical dispersants may cause significant mortality in embryonic herring (Section 3.2), even if dilution occurs fairly rapidly (Section 2.1).

Furthermore, the potential for localized mortality in small, sensitive zooplankton exists and may be enhanced after chemical dispersion (Sections 6.2, 6.3.1, and 6.4). At early life stages, larval Pacific herring are relatively immobile and graze on zooplankton in the upper water column. A reduction in the prey base of a larval species of fish, one that cannot move to an area not impacted by chemical dispersion (e.g., Pacific herring), could result in reduced growth and fitness. It is possible, therefore, that chemical dispersion will result in indirect adverse impacts on Pacific herring. The enhancement of toxicity to sensitive, shallow-dwelling invertebrates is a point of uncertainty discussed at more length in Sections 6.2, 6.3.1, and 6.4.

Based on the toxicity evaluation presented in Sections 3.2 through 3.4, Pacific herring are at particular risk for significant, direct, individual-level impacts (i.e., reduced survival, growth, or reproduction) resulting from the application of dispersants. The risk of acute toxicity to Pacific herring assumes that oil has not been dispersed to non-toxic concentrations prior to moving into the nearshore environment, and that dispersants will not be sprayed in the nearshore environment, where herring are known to spawn and rear (NOAA, 2012a). Although it is possible that dispersants could mitigate toxicity to herring (at early life stages) by limiting the concentration, volume, and areal extent of surface oiling (Section 2), the potential for significant toxicity remains with the oiling of shorelines, submerged aquatic vegetation (i.e., spawning substrate), and intertidal sediments (Fingas, 2008). In addition, toxicity may be increased by the redistribution of oil into the water column under foreseeable circumstances.

5.4 MARINE REPTILES

All marine reptiles are considered "accidental or uncommon" in Alaska, and as such will be treated in a similar manner in this section. The assumption is that sea turtles are very rarely found in Alaska waters, which precludes them from exposure to chemical dispersants. This section is therefore intended to describe a worst-case scenario, in which turtles would be found to be present in or near the area of a spill response when dispersants were applied or soon thereafter.

The potential for oiling of marine reptiles to occur in Alaska is remote due to their uncommon (or accidental) presence so far north. The likelihood of dispersants or dispersed oil coming into contact with these species in Alaska is equally remote.

Marine turtles feed on a variety of species, from plants and algae (Bjorndal, 1997) to tunicates, cnidarians, and other pelagic invertebrates (Bjorndal, 1997; NMFS and USFWS, 2007; Kopitsky et al., 2005) or shallow-water invertebrates (Witherington, 2002). The early life stages of these prey species and the mature forms of algae and shallow-dwelling invertebrates may be found in the upper water column during certain times of the day or year, and may be exposed to both concentrated and dispersed oil. Based on the assessment in Sections 3.2 through 3.4, it is likely that the toxicity of oil to marine turtle prey will decrease after dispersant application.

All marine reptiles must surface to breathe, so exposure to both oil and dispersed oil is possible. Little is known about the toxicity of oil and dispersed oil to marine reptiles, although it can be assumed that systemic impacts related to inhalation, aspiration, ingestion, and dermal contact are similar to those of other groups (Section 3.1). As mentioned in Section 3.1.1.2[42], it is expected that dispersion will remove a large amount of oil (i.e., volume, concentration, and areal extent) from the surface (Section 2), where marine reptiles surface to breathe. The redistribution of oil through chemical dispersion will likely result in mitigated acute impacts on marine reptiles, changing the route of exposure from predominately inhalation, aspiration, and dermal contact at the surface to ingestion and dermal contact with dilute oil in the water column.

Although dissolved PAHs in the water column are expected to increase after chemical dispersion (Ramachandran et al., 2004), it is unlikely that sea turtles will accumulate sufficient PAHs to cause acute impacts. Long-term impacts will assumedly be mitigated by the rapid decrease in ambient concentrations over time (Section 2). Therefore, chronic exposures to increased PAHs are unlikely. Marine reptiles have efficient mechanisms for metabolizing and excreting PAHs (Albers and Loughlin, 2003), which should prevent the accumulation of PAHs in their tissues over time.

[42] The discussion of marine reptiles is in Section 3.1.1.5, but the discussion of decreased risk of inhalation is in the analogous section for birds, Section 3.1.1.2.

Exposure to PAHs through the food web is possible, as PAHs bioaccumulate in invertebrates (Wolfe et al., 1998), which are prey items of several marine reptiles. However, prolonged uptake (e.g., chronic inputs) of LPAHs from invertebrates to reptiles as a result of chemical dispersion is unlikely, due to the rapid depuration of those chemicals in invertebrates (and fish) (Wolfe et al., 2001; Wolfe et al., 1998), as well as the relatively short time (~1 year) required to return to baseline tissue concentrations in other benthic species (Humphrey et al., 1987). Conversely, HPAHs may remain in invertebrate tissues for longer periods of time. Impacts related to PAH exposure are a point of uncertainty, in that individual-level impacts (i.e., reduced survival, growth, or reproduction) are not clearly defined for marine reptiles (Section 6.3.5).

Based on the rationale provided above, the application of chemical dispersants in accordance with associated BMPs will not adversely impact marine reptiles in Alaska. Specific BMPs relevant to marine reptiles include monitoring for their presence, and not applying dispersants when and where turtles are present. It should be noted again that marine reptiles are uncommon in Alaska waters, so the likelihood of encountering such species during any response action is low.

6 Uncertainty Analysis

There are various points of uncertainty that have been stated throughout this appendix that will be summarized in this section.

6.1 Sea Conditions, Spill Conditions, and Expected Spill Responses

No two spills are expected to be alike, considering the complex nature of the environment into which oil is spilled, the expansive area of the State of Alaska, and the various potential sources of oil (e.g., oil tanker, oil platform, marine fueling station, etc.). Therefore, it is impossible to accurately predict the response actions that will be applied and the efficacy of those actions. For example, the use of dispersants would not be effective under many conditions, nor would it be practical under all conditions (Nedwed, 2012).

Assuming that conditions are such that dispersants are approved for use on a given spill, it is impossible to know in advance the effectiveness of the dispersant due to changing sea conditions (e.g., wind and wave energy, tides), the presence of sea ice, salinity differences, and various other conditions. Furthermore, it is impossible to know in advance whether BMPs will be entirely successful in mitigating damages to listed or candidate fish and wildlife species.

6.2 Calculation of the HC5

The HC5s derived for use in this BA are representative of only Corexit® 9500 or Corexit® 9527, the only two dispersants currently available for use (i.e., stockpiled) in Alaska. However, Corexit® 9527 is no longer being manufactured, so the model created here will become obsolete once those stockpiles are exhausted. It is assumed that Corexit® 9500 will be used once Corexit® 9527 ceases to be available for emergency responses. Few toxicity data are available to evaluate other dispersant formulations that could be approved for use by the Alaska Regional Response Team (ARRT) in the future.

The majority of studies used to derive the HC5s were based on continuous exposure scenarios. As discussed, the resulting LC50s were generally lower than those derived from spiked exposures. Because a geometric mean LC50 was used to represent a given species or genera, spiked data were, in some cases, combined with continuous exposure data. Although spiked exposures are expected to provide a more realistic simulation of dispersants in the field (i.e., surface application), the HC5s derived are more representative of continuous exposures. For these reasons, the HC5s may overestimate toxicity as it relates to a field application, and can thus be seen as protective (over a short time period).

Although only early-life-stage fish species were used in developing the SSDs, there were various invertebrates included in the SSDs for which the life stage was uncertain.

Because life stage is important in driving the sensitivity of invertebrates (as well as most species in general), the sensitivity of certain taxa may be slightly overestimated.

The toxicity data largely represent either temperate or warm-water species (as opposed to Arctic species), which may not react in the same way as species in Alaska. Tests of Corexit® 9500-dispersed oil using arctic species have shown that they are somewhat less sensitive than non-Arctic species (Figure 6). However, this result was likely affected by a difference in exposure regimes: Toxicity tests using Arctic species mostly applied spiked exposures, whereas toxicity tests using temperate species used primarily continuous exposures (i.e., static, flow through, or renewal) (Attachment B-1). Because spiked exposures tend to result in increased LC50 values, regardless of species, the apparent insensitivity of Arctic species shown in Figure 6 may be an artifact of the exposure method.

It is assumed that the distributions of toxicity values are representative of all water column species in a given aquatic habitat, even though the true number of species is limited (i.e., the water column does not contain every species at a given location). The species used for each model are considered surrogates for all fish, aquatic plants, and invertebrates that may be affected in a field application of dispersants.

Most importantly, the analysis presented above, which uses acute laboratory data, does not incorporate two very important sources of uncertainty. Although sublethal and chronic impacts are discussed in a cursory way in Section 3.2, such impacts are not incorporated into the determination of the HC5s. PAHs are thought to be the most toxic component of oil, and chemical dispersants generally increase the exposure of planktonic species to PAHs by making PAHs more bioavailable (Ramachandran et al., 2004; Yamada et al., 2003; Milinkovitch et al., 2011a; Lee, 2013). Sublethal effects may occur at much lower exposure concentrations than the HC5s (Smit et al., 2009), and such effects may have lasting impacts on plankton.

Also of great importance is the fact that traditional laboratory testing of aquatic toxicity is conducted in chambers without UV light in order to control for photodegradation of PAHs or other similarly degraded toxicants. But PAHs are known to be up to 1,000 times more toxic when exposed to UV light (Barron and Ka'aihue, 2001). In the shallow ocean, solar irradiance is ubiquitous; furthermore, there can be extreme light conditions in the State of Alaska, depending on the time of year (i.e., midnight sun or polar day phenomena). For these reasons, it can be assumed that an ecologically relevant exposure to PAHs, made more bioavailable by the application of dispersants (Ramachandran et al., 2004), will occur in conjunction with photo-enhanced toxicity, particularly in species of invertebrates and larval fish that are translucent (Barron et al., 2008).

6.3 PAH TOXICITY

6.3.1 Invertebrates

The analysis of the toxicity of oil and dispersed oil (including PAHs as a component of both) presented in Section 3.3 clearly shows that dispersed oil is less toxic than oil alone. Although several authors have shown the opposite to be true (Attachment B-1; Section 3.4.1), the magnitude of differences in toxicity observed across all studies demonstrates that in general, dispersed oil is less toxic to aquatic species than oil alone (Section 3.3); the magnitude of differences across studies is presented visually in Figures 8 and 9. In addition, toxicity is shown to decrease in general after dispersant application (Section 3.3), even though PAHs have been shown to increase in solution as well as in tissues of various species (i.e., taken up from the water column) (Ramachandran et al., 2004). Therefore, the analysis addresses the acute toxicity of PAHs in solution, in a laboratory study, after chemical dispersant application.

There are various potential reasons for uncertainty in drawing conclusions about the likelihood of impacts of dispersed oil on planktonic species when using acute toxicity data. Based on the uncertainties identified in Section 6.2, it is possible that dispersed oil will have an impact on plankton, more so than the analysis presented in Section 3.3 (based on acute toxicity) would suggest.

6.3.2 Fish

A major point of uncertainty in the analyses provided in this appendix has to do with the use of surrogate fish species in the estimation of impacts on fish. For example, the fish included in the SSD presented in Section 3.3 include many taxa that are not found in Alaska waters and that are not protected under ESA.

Oil, particularly the toxic component PAHs in oil (Barron, 2012; Milinkovitch et al., 2011a; Roy et al., 1999; Brannon et al., 2006; Carls et al., 1999, 2000; Meador, 2003; Payne et al., 2003), has various sublethal impacts on fish species (Stige et al., 2011; ITOPF, 2011). Metabolites of PAHs are often more toxic than their parent compounds, so adverse impacts on fish are most likely to occur after accumulation and metabolism of parent compounds, but before excretion (Payne et al., 2003). Payne et al. (2003) provide a concise review of the historically reported sublethal impacts of PAHs on fish (e.g., Chinook salmon, rainbow trout, and herring), including genotoxicity, immunotoxicity, histopathological impacts (e.g., hepatic lesions), behavioral impacts, and reproductive impacts. Such impacts may result in reduced fitness, leading to the death of individuals. A clear example of this impact is provided by Claireaux et al. (2013), who showed that European sea bass (*Dicentrarchus labrax*) exposed to oil and dispersed oil were more susceptible to normal environmental perturbations than those that were not exposed to oil or dispersed oil. To test this, both chemically exposed and control fish were placed in a chamber that became hypoxic for a time and, subsequently, very warm for a time; the fish were then transferred to the field for monitoring of growth and survival. Those fish exposed (after exposure to oil or dispersed oil) to low dissolved oxygen and high temperatures had a significantly higher rate of mortality or a significantly lower rate of growth than the control fish, suggesting that their fitness was compromised by chemical exposure (Claireaux et al., 2013).

Another important consideration for fish, particularly unpigmented, early-life-stage fish that reside in the upper water column (e.g., Pacific herring), is the possibility of photo-enhanced toxicity; this is discussed in Section 6.1. Similarly to invertebrates, the potential for acute mortality in prey fish species or larval life stages of ESA-listed Pacific herring under natural lighting conditions may be underestimated by the analyses presented in Section 3.3.

Although dermal exposures of fish may increase after chemical dispersion, it is not clear how dermal exposures to dispersed oil will impact the survival, growth, or reproduction of fish. It is possible that topical lesions may occur based on studies with PAHs (Logan, 2007), however a clear link between topical lesions and reduced growth, survival, and/or reproduction in fish species has not been established.

6.3.3 Birds

Although contact of bird species with oil may be greatly diminished by the application of chemical dispersants, the increase of PAHs in the water column may impact various species of birds, particularly those that feed on invertebrates. Invertebrates are known to accumulate more PAHs in their lipids due to less efficient PAH metabolisms, so birds that feed on invertebrates are likely to be exposed to greater concentrations of dietary PAHs after chemical dispersion than if the chemicals had not been applied. Spectacled and Steller's eiders are known to selectively consume bivalves, which have been shown to accumulate significant amounts of oil after chemical dispersion (Michel and Henry Jr, 1997; Lemiere et al., 2005). Short-tailed albatross selectively consume squid, which may also accumulate PAHs; little or no data is available for accumulation in squid, but squid are invertebrates, and invertebrates tend to have less efficient PAH metabolisms (Meador, 2003). In lieu of direct exposure data for bird species, data from rats exposed to oil-contaminated mussel tissue were used. The rats experienced increased genetic liver damage (Lemiere et al., 2005), even though they assumedly have efficient PAH metabolisms (Albers and Loughlin, 2003), so such impacts may also be observable in birds that selectively consume invertebrates. Although fish accumulate PAHs to a lesser degree than do invertebrates, the trophic transfer of PAH metabolites stored in fish tissues to piscivorous birds (e.g., Kittlitz's murrelet, yellow-billed loon, short-tailed albatross) may also occur, resulting in PAH-related toxicity in those birds. HPAHs are more likely to be transferred in this way, as fish metabolize and depurate HPAHs at a slower rate than LPAHs (Payne et al., 2003; Wolfe et al., 2001).

Direct impacts on birds caused by exposure to dispersants or dispersed oil are generally extrapolated from non-ESA listed species, and may have been extrapolated from studies with non-bird species (e.g., Norway rats). For these reasons, conclusions made about potential direct impacts of dispersants alone or dispersed oil are uncertain.

6.3.4 Mammals

Toxicity caused by PAHs is generally associated with highly toxic metabolites (Albers and Loughlin, 2003), so the transfer of metabolites (rather than parent PAHs) through diet may result in some toxicity (Albers and Loughlin, 2003). Similarly, metabolism of parent molecules (taken up through direct contact) to toxic metabolites is generally expected to be a source of sublethal toxicity in mammals (Albers and Loughlin, 2003), although perhaps less relevant for more mutagenic HPAHs that concentrate in tissues of prey species. It is difficult to predict the level of toxicity in mammals due to PAH uptake, because previous studies have not directly investigated impacts on listed species related to PAHs alone (Albers and Loughlin, 2003); furthermore, it is not clear whether deceased marine mammals found with high concentrations of PAHs in tissues were chronically exposed to PAHs, nor is it clear to what concentrations they were exposed, what the source of the PAHs was, or whether they were exposed to various chemicals in addition to petrogenic PAHs (Albers and Loughlin, 2003). More importantly, it is not clear whether PAH uptake resulting from a chemical dispersant application will cause individual-level impacts (e.g., reduced survival, growth, or reproduction) in ESA-listed mammals. Given the expected difference in chemical exposures between mammals chronically exposed in contaminated waterways (e.g., beluga in St. Lawrence estuary) (Albers and Loughlin, 2003) and those exposed in a rapidly diluting and degrading oil plumes (Section 2), it is reasonable to assume that toxic responses will differ in the latter circumstance. In other words, the exposures of mammals to dispersed oil plumes is expected to be acute rather than chronic, and noted impacts in the literature tend to reflect chronic rather than acute exposures. Conversely, acute exposures noted in marine mammals exposed during and after EVOS resulted in high levels of PAH uptake; mortalities in Northern sea otter were attributed to hypothermia (a physical effect of oiling) rather than toxicity (a secondary effect) (Albers and Loughlin, 2003), and brain lesions noted in harbor seals[43] exposed to the same oil spill were not causally linked to PAH exposures (Albers and Loughlin, 2003). Therefore, there is a lack of directly relatable toxicity data for ESA-listed species regarding PAH exposures for relevant durations to accurately predict the likelihood of PAH impacts, particularly at the individual level (e.g., reduced survival, growth, or reproduction).

Given that PAH metabolites are known to impact mammalian species (Albers and Loughlin, 2003; Lemiere et al., 2005), and that dispersants increase the bioavailability

[43] Harbor seals were alive at the time of sampling (Albers and Loughlin, 2003).

of these chemicals to various species (including prey), the use of chemical dispersants may cause sublethal impacts in some mammals. It is expected that chemical dispersants will cause the uptake of PAHs in some mammal diets to increase; this is particularly true of those that selectively consume longer-lived invertebrates (e.g., Pacific walrus, northern sea otter, some baleen whales, and bearded seal), which accumulate higher concentrations of PAHs.[44] However, it is uncertain whether the increase in PAHs in invertebrate tissues will be over a large enough area and for a sufficiently long duration to cause reduced survival, growth, or reproduction in marine mammals that consume contaminated invertebrates. For example, bivalves on shorelines impacted by dispersed oil depurated or metabolized hydrocarbons over the period of year (Mageau et al., 1987), returning to the pre-spill condition (i.e., lower tissue concentration) after about 1 year; bivalves on shorelines impacted by untreated oil continued to take up hydrocarbons for a longer period of time (Humphrey et al., 1987). Chemical dispersion has been shown to increase the rate of depuration of LPAHs in both larval topsmelt (Wolfe et al., 2001) and a rotifer (Wolfe et al., 1998), suggesting that internalization of PAHs and the subsequent transfer to higher trophic levels of LPAHs can be mitigated by chemical dispersion.

Mammals that selectively feed on fish (e.g., Steller sea lion, some baleen and most toothed whales, and ringed seal) or other mammals (e.g., polar bear) are likely to accumulate PAHs through their diet, but they may accumulate lower concentrations due to the more efficient metabolic activity in fish and mammals.

Direct impacts on mammals caused by exposure to dispersants or dispersed oil are generally extrapolated from non-ESA listed species (e.g., Norway rats). For these reasons, conclusions made about potential impacts of dispersants alone or dispersed oil are uncertain.

Dermal exposures to dispersed oil may result in topical lesions in fish species (Logan, 2007) and possibly mammals as well; however, it is unclear how such lesions could result in reduced growth, reproduction, or survival. Dermal exposures are likely to be reduced by chemical dispersion, as fouling is expected to decrease (CDC and ATSDR, 2010; Lessard and Demarco, 2000).

6.3.5 Reptiles

As with birds and mammals, the likelihood of sublethal impacts on marine reptiles caused by the increased dissolution of PAHs into the water column and concomitant increase in PAH concentrations in prey tissues is uncertain. Reptile species tend to be little studied toxicologically, so it is exceedingly difficult to extrapolate impacts from previous studies. However, as reptiles are very rare in Alaska waters, it is unlikely that

[44] Note that sea otter, baleen whales, and bearded seal will also feed on finfish species if available, assuming that it is energetically favorable to forage on those fish species.

any adverse impact on marine reptiles will occur as a result of chemical dispersant application.

It is possible that dermal exposures will occur in marine reptiles, but dermal exposures are likely to be reduced by chemical dispersion (CDC and ATSDR, 2010; Lessard and Demarco, 2000).

6.4 Indirect Impacts of Dispersed Oil Toxicity

Given the discussion in Section 6.3, it is uncertain whether planktonic species will be significantly impacted by dispersed oil relative to oil alone due to the increased solubility and uptake of PAHs in the upper water column. Planktonic species that are immobile (aside from moving with ocean currents) have the greatest potential to be impacted (Barron and Ka'aihue, 2001). However, it is unclear whether the mortality of plankton in the vicinity of a treated oil spill will result in significant, indirect impacts on wildlife. For example, cetaceans are known to feed over large areas and may not be impacted by a localized mortality of sensitive plankton. Although many sensitive species may be killed during an oil spill or after chemical dispersion, the biomass contained within a planktonic community may remain much the same over time (Varela et al., 2006); therefore, the resource for non-selectively feeding species such as baleen whales may not be reduced.

In terms of duration, it is likely that the planktonic community within a given area will be replaced with new members as the ocean mixes and currents recharge a degraded area with previously unexposed planktonic individuals. Planktonic species impacted in the Gulf of Mexico during DHOS recuperated to pre-spill conditions within a matter of weeks to months (Abbriano et al., 2011). It was suggested that the rate of recruitment into impacted areas was due to various potential factors, including rapid reproduction, the ability of some species to selectively avoid oil droplets in water, and the circulation and mixing of the ocean; dispersion and degradation were also cited as potential reasons for this rapid recovery (Abbriano et al., 2011). Impacts on the prey base (i.e., available food rather than specific individuals or taxa) are therefore unlikely to persist.

6.5 Toxicity of Dispersant Components and Degradates/Metabolites

The analyses of dispersant toxicity presented in Sections 3.1 through 4.3 do not include a specific discussion of the individual component chemicals within dispersant mixtures. It is unclear, based on the analyses presented in this appendix, what the toxicities of these individual components are. However, the conceptual model for the application of chemical dispersants assessed in this appendix does not include individual components, applied singly or in mixtures, other than the original formulation (i.e., Corexit® 9500 or Corexit® 9527). Therefore, it is not deemed necessary to assess individual dispersant components. Similarly, individual components of oil are not directly assessed, though some emphasis is placed on PAHs as a group of chemicals found in oil.

There is a general paucity of data regarding the toxicity and fate and transport of the degradates or metabolites (created primarily via biodegradation) of chemical dispersant component chemicals (Table 2). It is not clear whether such resultant products will be more or less toxic than or equally toxic to parent chemicals in chemical dispersants. The assessment of chemical toxicity of chemical dispersants alone does not directly address this uncertainty, rather discussing the toxicity of the parent components as a mixture.

7 Conclusion

Based on the analyses of toxicity, fate, and transport, as well as the likelihood of exposure of ESA-protected or candidate species, many species will not be adversely impacted by chemical dispersion at the individual level (i.e., reduced survival, reproduction, or growth) relative to baseline oiling. This conclusion assumes that the Unified Plan (which is specifically structured to provide for the protection of sensitive wildlife) will be implemented in accordance with all appropriate BMPs. For ESA-listed birds, mammals, and reptiles, this conclusion contains a degree of uncertainty, as discussed in Sections 6.3.3, 6.3.4, and 6.3.5, respectively. However, several species have been specifically identified as being at direct or indirect risk for adverse impacts related to oil exposures enhanced by chemical dispersion. Steller's and spectacled eiders, Pacific walrus, and Pacific herring may all be impacted by the application of chemical dispersants, even if most BMPs are observed. Only Pacific herring is expected to be directly impacted, whereas Steller's and spectacled eiders and Pacific walrus are expected to be indirectly impacted; this conclusion is primarily based on the reliance of eiders and walrus on bivalves as prey, and the fact that bivalves are known to be highly sensitive to dispersants and dispersed oil (Section 3.3; Attachment B-1). Similarly, Pacific herring are known to be highly sensitive to dispersants and dispersed oil, and they are found in Alaskan waters during all times of the year and in the nearshore coastal areas during early life stages (when herring are most sensitive).

In the unlikely event that BMPs are not implemented, or that such practices fail to be protective of sensitive species (i.e., a worst-case scenario), chemical dispersants may impact any species other than Aleutian shield fern and Eskimo curlew, which are terrestrial species that would not be exposed to chemical dispersants, and sea turtles, which are extremely rare in Alaskan waters. For example, the inadvertent spraying of chemical dispersants on or very near individual birds (any species) or Northern sea otter may result in the loss of thermoregulation, leading to hypothermia and death. If spraying were to occur near individual marine mammals, dermal exposures could result in sublethal impacts, such as irritation of skin, eyes, and mucous membranes. Similarly, inhalation and aspiration of recently sprayed dispersants by marine birds and mammals could result in irritated lung tissue and impaired breathing (as well as affected diving and foraging behavior).

Chemical dispersion will likely increase the bioavailability of dissolved PAHs in the water column over a short period of time (i.e., prior to dilution and biodegradation [Section 2]), possibly resulting in sublethal impacts on all species (excepting Aleutian shield fern, Eskimo curlew, and marine reptiles). It is unclear whether sublethal impacts (e.g., lesions) will result in significant effects on ESA-listed or candidate species (Section 6.3). It is also possible that increased exposure to dissolved PAHs among shallow-dwelling planktonic species (i.e., invertebrates and fish) will result in alterations to the food web, potentially causing indirect impacts on ESA-listed or candidate species (as well as direct impacts on early life stage Pacific herring, should the dispersed oil reach the coastal areas). Although the analysis provided in this appendix supports the conclusion that chemical dispersion will reduce the overall toxicity of oil in the water column (Figures 8 and 9), it is possible that the analysis underestimates the risk to the aquatic community (e.g., early life stages of invertebrate and fish species) from PAH exposures, which may become more toxic under natural conditions (Barron and Ka'aihue 2001; Barron et al. 2008).

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Attachment B-1. Toxicity Data

Table of Contents

Acronyms

ALC	Arabian light crude oil
AMC	Arabian medium crude oil
ANS	Alaska North Slope crude oil
BSC	Bass Strait crude oil
BSD	blue sac disease
CIC	Cook Inlet crude oil
DOR	dispersant-to-oil ratio
EC50	concentration that causes a non-lethal effect in 50% of an exposed population
EPA	US Environmental Protection Agency
EROD	ethoxyresorufin-O-deethylase
KCO	Kuwait crude oil
LC50	concentration that is lethal to 50% of an exposed population
MESA	medium South American fuel oil
MFO	medium fuel oil
NPL	National Priorities List
NRC	National Research Council
PBCO	Prudhoe Bay crude oil
ppm	parts per million
SLC	Sweet Louisiana Crude oil
SSD	species sensitivity distribution
TTV	threshold toxicity value
VCO	Venezuelan medium crude oil
WQC	water quality criteria

Introduction

This attachment presents the currently available toxicity data from published literature on chemical dispersants (Tables 1 and 2), crude oil (Table 3), and chemically dispersed oil (Tables 4 and 5). These data (with some exceptions identified in the tables) were used to create chemical-specific species sensitivity distributions (SSDs) for current-use chemical dispersants (i.e., Corexit® EC9527A and Corexit® EC9500A, hereafter referred to as Corexit® 9527 and Corexit® 9500, respectively), crude oil alone, and crude oil dispersed by those chemicals. From the SSDs, hazardous concentrations (HC5) were calculated, and these values were compared. The raw data and the calculations of SSDs and HC5 values are discussed at length in Appendix B.

Table 1. Available median lethal toxicity values (LC50) for current-use and NPL-listed chemical dispersants

Dispersant Chemical	Latin Name	Life Stage	Duration (h)	Range of LC50s (ppm)	Source(s)
BP 1100-X	*Penaeus monodon*	post-larval	24	4,351 – 7,207	Bussarawit (1994)
BP 1100-X	*Penaeus monodon*	post-larval	48	2,818 – 4,598	Bussarawit (1994)
BP 1100-X	*Penaeus monodon*	post-larval	96	1,253 – 2,044	Bussarawit (1994)
Corexit 9500	*Allorchestes compressa*	adult	96	3.5	Gulec et al. (1997)
Corexit 9500	*Americamysis bahia*	neonate	48	42	Hemmer et al. (2010)
Corexit 9500	*Americamysis bahia*	neonate	48	5.4	Hemmer et al. (2011)
Corexit 9500	*Americamysis bahia*	nr	48	32.2	Inchcape (1995)
Corexit 9500	*Americamysis bahia*	nr	96	31.4 – 35.9	George-Ares and Clark (2000); Fuller and Bonner (2001); Clark et al. (2001); Rhoton et al. (2001)
Corexit 9500	*Americamysis bahia*	non-embryo	48 – I96	20.9	Edwards et al. (2003) as cited in Barron et al. (2013)
Corexit 9500	Americamysis bahia	non-embryo	48 – I96	32	Fuller et al. (2004)
Corexit 9500	*Americamysis bahia*	adult	nr	37.20	Wetzel and Van Fleet (2001)
Corexit 9500	*Americamysis bahia*	nr	SD	500 – 1,305	Coelho and Aurand (1997); Fuller and Bonner (2001); Clark et al. (2001); Rhoton et al. (2001)
Corexit 9500	*Americamysis bahia*	nr	SD	>789	Coelho and Aurand (1997); Fuller and Bonner (2001); Clark et al. (2001); Rhoton et al. (2001)
Corexit 9500	*Americamysis bahia*	adult	SD	1,038	Wetzel and Van Fleet (2001)
Corexit 9500	*Artemia salina*	nr	48	21	George-Ares and Clark (2000)
Corexit 9500	*Atherinosoma microstoma*	juvenile	96	50	Marine and Freshwater Resources Institute (1998)
Corexit 9500	*Brachydanio rerio*[a]	nr	24	>400	George-Ares and Clark (2000)
Corexit 9500	*Chionoecetes bairdi*	larvae	96	5.6	Rhoton et al. (2001)
Corexit 9500	*Chionoecetes bairdi*	larvae	SD	355	Rhoton et al. (2001)
Corexit 9500	*Crassostrea virginica*	non-embryo	48 – I96	167	Liu (2003) as cited in Barron et al. (2013)
Corexit 9500	*Cyprinodon variegatus*	larvae	96	170 – 193	Fuller and Bonner (2001)
Corexit 9500	*Cyprinodon variegatus*	non-embryo	48 – I96	180	Fuller et al. (2004)
Corexit 9500	*Cyprinodon variegatus*	larvae	SD	593 – 750	Fuller and Bonner (2001)
Corexit 9500	*Eurytemora affinis*	adult	96	5.2	Wright and Coehlo (1996)

Table 1. Available median lethal toxicity values (LC50) for current-use and NPL-listed chemical dispersants, cont.

Dispersant Chemical	Latin Name	Life Stage	Duration (h)	Range of LC50s (ppm)	Source(s)
Corexit 9500	*Fundulus grandis*	non-embryo	48 – I96	172.6	Liu (2003) as cited in Barron et al. (2013)
Corexit 9500	*Fundulus heteroclitus*	nr	96	25.2	Nalco (2005)
Corexit 9500	*Fundulus heteroclitus*	nr	96	140	George-Ares and Clark (2000)
Corexit 9500	*Haliotis rufescens*	embryo	SD	12.8 – 19.7	Singer et al. (1996)
Corexit 9500	*Holmesimysis costata*	juvenile	SD	158 – 245	Singer et al. (1996)
Corexit 9500	*Hydra viridissima*	non-budding	96	160	Mitchell and Holdway (2000)
Corexit 9500	*Lates calcarifer*	juvenile	96	143	Marine and Freshwater Resources Institute (1998)
Corexit 9500	*Litopenaeus setiferus*	non-embryo	48 – I96	31.1	Liu (2003) as cited in Barron et al. (2013)
Corexit 9500	*Macquaria novemaculeata*	larvae	96	19.8	Gulec and Holdway (2000)
Corexit 9500	*Menidia beryllina*	larvae	96	130	Hemmer et al. (2010)
Corexit 9500	*Menidia beryllina*	larvae	96	7.6	Hemmer et al. (2011)
Corexit 9500	*Menidia beryllina*	larvae	96	25.2 – 85.4	Inchcape (1995); (Fuller and Bonner, 2001); Rhoton et al. (2001)
Corexit 9500	*Menidia beryllina*	nr	96	140	Nalco (2005)
Corexit 9500	*Menidia beryllina*	non-embryo	48 – I96	79.3	Edwards et al. (2003) as cited in Barron et al. (2013)
Corexit 9500	*Menidia beryllina*	non-embryo	48 – I96	79	Fuller et al. (2004)
Corexit 9500	*Menidia beryllina*	juvenile	nr	85.1	Wetzel and Van Fleet (2001)
Corexit 9500	*Menidia beryllina*	larvae	SD	40.7 – 116.6	Fuller and Bonner (2001); Rhoton et al. (2001)
Corexit 9500	*Menidia beryllina*	larvae	SD	205	Fuller and Bonner (2001);; Rhoton et al. (2001)
Corexit 9500	*Menidia beryllina*	juvenile	SD	21.6	Wetzel and Van Fleet (2001)
Corexit 9500	*Oncorhynchus mykiss*[a]	nr	96	354	George-Ares and Clark (2000)
Corexit 9500	*Palaemon serenus*	nr	96	83.1	Gulec and Holdway (2000)
Corexit 9500	*Palaemonetes varians*	nr	6	8,103	Beaupoil and Nedelec (1994)
Corexit 9500	*Penaeus monodon*	larvae	96	48	Marine and Freshwater Resources Institute (1998)
Corexit 9500	*Polinices conicus*	nr	24	42.3	Gulec et al. (1997)
Corexit 9500	*Sarotherodon mozambicus*	nr	96	150	George-Ares and Clark (2000)
Corexit 9500	Sciaenops ocellatus	juvenile	SD	744	Wetzel and Van Fleet (2001)
Corexit 9500	*Scophthalamus maximus*	nr	96	75	Nalco (2005)

Table 1. Available median lethal toxicity values (LC50) for current-use and NPL-listed chemical dispersants, cont.

Dispersant Chemical	Latin Name	Life Stage	Duration (h)	Range of LC50s (ppm)	Source(s)
Corexit 9500	*Scophthalmus maximus*	yolk-sac larvae	48	74.7	George-Ares and Clark (2000); Clark et al. (2001)
Corexit 9500	*Scophthalmus maximus*	yolk-sac larvae	SD	>1,055	George-Ares and Clark (2000); Clark et al. (2001)
Corexit 9500	*Skeletonema costatum*	nr	72	20	Norwegian Institute for Water Research (1994)
Corexit 9500	*Tigriopus japonicus*	larvae	96	10	Lee et al. (2013)
Corexit 9527	*Allorchestes compressa*	nr	96	3	Gulec et al. (1997)
Corexit 9527	*Americamysis bahia*	nr	96	19 – 34	Bricino et al. (1992);George-Ares et al. (1999);Exxon Biomedical (1993a); Pace and Clark (1993)
Corexit 9527	*Americamysis bahia*	nr	96	29.2	Bricino et al. (1992); George-Ares et al. (1999);Exxon Biomedical (1993a); Pace and Clark (1993)
Corexit 9527	*Americamysis bahia*	nr	48	24.1 – 29.2	Inchcape (1995); Clark et al. (2001)
Corexit 9527	*Americamysis bahia*	nr	SD	>1,014	Pace et al. (1995); Clark et al. (2001)
Corexit 9527	*Anonyx laticoxae*	nr	96	>140	Foy (1982)
Corexit 9527	*Anonyx nugax*	nr	96	97 – 111	Foy (1982)
Corexit 9527	*Argopecten irradians*	nr	6	200	Ordzie and Garofalo (1981)
Corexit 9527	*Argopecten irradians*	nr	6	1,800	Ordzie and Garofalo (1981)
Corexit 9527	*Argopecten irradians*	nr	6	2,500	Ordzie and Garofalo (1981)
Corexit 9527	*Artemia salina*	nr	48	53 – 84	Bricino et al. (1992)
Corexit 9527	*Artemia* sp.	larvae	48	52 – 104	Wells et al. (1982)
Corexit 9527	*Artemia* sp.	larvae	48	42 – 72	Wells et al. (1982)
Corexit 9527	*Atherinops affinis*	larvae	96	25.5 – 40.6	Singer et al. (1990); Singer et al. (1991)
Corexit 9527	*Atherinops affinis*	larvae	SD	59.2 – 104	Singer et al. (1991)
Corexit 9527	*Boeckosimus edwardsi*	nr	96	>80	Foy (1982)
Corexit 9527	*Boeckosimus sp.*	nr	96	>175	Foy (1982)
Corexit 9527	*Brevoortia tyrannus*	embryo-larval	48	42.4	Fucik et al. (1995)
Corexit 9527	*Callinectes sapidus*	larvae	96	77.9 – 81.2	Fucik et al. (1995)
Corexit 9527	*Chlamydomonas reinhardti*	nr	4	575	Norland et al. (1978)

Table 1. Available median lethal toxicity values (LC50) for current-use and NPL-listed chemical dispersants, cont.

Dispersant Chemical	Latin Name	Life Stage	Duration (h)	Range of LC50s (ppm)	Source(s)
Corexit 9527	*Corophium volutator*	non-embryo	48 – l96	159	Scarlett et al. (2005)
Corexit 9527	*Crassostrea gigas*	embryos	48	3.1	George-Ares and Clark (2000); Clark et al. (2001)
Corexit 9527	*Crassostrea gigas*	embryos	SD	13.9	George-Ares and Clark (2000); Clark et al. (2001)
Corexit 9527	*Cyprinodon variegatus*	nr	96	74 – 152	Bricino et al. (1992)
Corexit 9527	*Daphnia magna*[a]	larvae	48	75	Bobra et al. (1989)
Corexit 9527	*Fundulus heteroclitus*	nr	96	81	Nalco (2010)
Corexit 9527	*Fundulus heteroclitus*	nr	96	99 – 124	Bricino et al. (1992)
Corexit 9527	*Gammarus oceanicus*	juvenile	96	>80	Foy (1982)
Corexit 9527	*Gnorimosphaeroma oregonensis*	nr	96	>1,000	Duval et al. (1982)
Corexit 9527	*Haliotis rufescens*	embryos	48	1.6 – 2.2	Singer et al. (1990); Singer et al. (1991)
Corexit 9527	*Haliotis rufescens*	embryos	SD	13.6 – 18.1	Singer et al. (1991)
Corexit 9527	*Holmesimysis costata*	nr	96	15.3	Coelho and Aurand (1997)
Corexit 9527	*Holmesimysis costata*	nr	96	2.4 – 10.1	Pace and Clark (1993); Exxon Biomedical (1993b, c); Clark et al. (2001)
Corexit 9527	*Holmesimysis costata*	juvenile	96	4.3 – 7.3	Singer et al. (1990); Singer et al. (1991)
Corexit 9527	*Holmesimysis costata*	nr	SD	195	George-Ares and Clark (2000); Clark et al. (2001)
Corexit 9527	*Holmesimysis costata*	juvenile	SD	120 – 163	Singer et al. (1991)
Corexit 9527	*Hydra viridissima*[a]	non-budding	96	230	Mitchell and Holdway (2000)
Corexit 9527	*Leiostomus xanthurus*	embryo-larval	48	27.4	Fucik et al. (1995)
Corexit 9527	*Leiostomus xanthurus*	embryos	48	61.2 – 62.3	Slade (1982)
Corexit 9527	*Macquaria novemaculeata*	larvae	96	14.3	Gulec and Holdway (2000)
Corexit 9527	*Macrobrachium rosenbergii*	embryo-larval	288	80.4	Law (1995)
Corexit 9527	*Macrocystis pyrifera*	zoospores	SD	86.6 – 102	Singer et al. (1991)
Corexit 9527	*Menidia beryllina*	larvae	96	14.6 – 57	Bricino et al. (1992); Fucik et al. (1995); Pace and Clark (1993); Inchcape (1995); Exxon Biomedical (1993d); Clark et al. (2001)
Corexit 9527	*Menidia beryllina*	larvae	96	52.3	Bricino et al. (1992); Fucik et al. (1995); Pace and Clark (1993); Inchcape (1995); Exxon Biomedical (1993d); Clark et al. (2001)
Corexit 9527	*Menidia beryllina*	larvae	96	>100	Fucik et al. (1995)

Table 1. Available median lethal toxicity values (LC50) for current-use and NPL-listed chemical dispersants, cont.

Dispersant Chemical	Latin Name	Life Stage	Duration (h)	Range of LC50s (ppm)	Source(s)
Corexit 9527	*Menidia beryllina*	nr	96	14.57	Nalco (2010)
Corexit 9527	*Menidia beryllina*	embryos	SD	58.3	George-Ares and Clark (2000);Clark et al. (2001)
Corexit 9527	*Myoxocephalus quadricornis*	nr	96	<40	Foy (1982)
Corexit 9527	*Oncorhynchus mykiss*[a]	juvenile	96	260	Doe and Wells (1978)
Corexit 9527	*Oncorhynchus mykiss*[a]	nr	96	96 – 293	Wells and Doe (1976)
Corexit 9527	*Onisimus litoralis*	nr	96	80 – 160	Foy (1982)
Corexit 9527	*Oryzias latipes*	nr	24	130 – 150	George-Ares and Clark (2000)
Corexit 9527	*Oryzias latipes*	nr	24	400	George-Ares and Clark (2000)
Corexit 9527	*Palaemon serenus*	nr	96	49.4	Gulec and Holdway (2000)
Corexit 9527	*Palaemonetes pugio*	nr	96	640	NRC (1989)
Corexit 9527	*Palaemonetes pugio*	nr	96	840	NRC (1989)
Corexit 9527	*Penaeus monodon*	post-larval	24	355 – 623	Bussarawit (1994)
Corexit 9527	*Penaeus monodon*	post-larval	48	120 – 213	Bussarawit (1994)
Corexit 9527	*Penaeus monodon*	post-larval	96	32 – 55	Bussarawit (1994)
Corexit 9527	*Penaeus monodon*	nr	96	35 – 45	Fucik et al. (1995)
Corexit 9527	*Penaeus setiferus*	post-larval	96	11.9	Fucik et al. (1995)
Corexit 9527	*Penaeus vannemai*	nr	96	35 – 45	Fucik et al. (1995)
Corexit 9527	*Phyllospora comosa*	nr	48	30	Burridge and Shir (1995)
Corexit 9527	*Pimephales promelas*	nr	96	201	Nalco (2010)
Corexit 9527	*Platichthys flesus*	350-g juvenile	96	100	Baklien et al. (1986)
Corexit 9527	*Protothaca stamiea*	nr	96	100	Hartwick et al. (1982)
Corexit 9527	*Pseudocalanus minutus*	adult	48	8.5 – 35.5	Wells et al. (1982)
Corexit 9527	*Pseudocalanus minutus*	nr	48	8 – 12	Wells et al. (1982)
Corexit 9527	*Pseudocalanus minutus*	adult	96	5 – 24.8	Wells et al. (1982)
Corexit 9527	*Pseudocalanus minutus*	nr	96	5 – 25	Wells et al. (1982)
Corexit 9527	*Sciaenops ocellatus*	embryo-larval	48	52.6	Fucik et al. (1995)
Corexit 9527	*Scophthalamus maximus*	nr	96	50	Nalco (2010)
Corexit 9527	*Scophthalamus maximus*	nr	72	9.4	Nalco (2010)

Table 1. Available median lethal toxicity values (LC50) for current-use and NPL-listed chemical dispersants, cont.

Dispersant Chemical	Latin Name	Life Stage	Duration (h)	Range of LC50s (ppm)	Source(s)
Corexit 9527	*Thalassia testudinum*	nr	96	200	Baca and Getter (1984)

[a] Freshwater species.

LC50 – concentration that is lethal to 50% of an exposed population

nr – not reported

NPL – National Priorities List

NRC – National Research Council

ppm – parts per million

SD – spiked concentration, declining exposure

Table 2. Available sublethal toxicity values for current-use chemical dispersants

Dispersant Chemical	Latin Name	Life Stage	Duration (h)	Endpoint	Range of LC50s (ppm)	Source(s)
Corexit 9500	*Haliotis rufescens*	embryos	48	NOEC	0.7	Aquatic Testing Laboratories (1994) as cited in NRC (2005)
Corexit 9500	*Haliotis rufescens*	nr	SD	NOEC	5.7 – 9.7	Singer et al. (1996)
Corexit 9500	*Holmesimysis costata*	nr	SD	NOEC	41.4 – 142	Singer et al. (1996)
Corexit 9500	*Hydra viridissima*	nr	168	NOEC	13	Mitchell and Holdway (2000)
Corexit 9500	*Phyllospora comosa*	zygotes	48	EC50, not specified	0.7	Burridge and Shir (1995)
Corexit 9500	*Skeletonema costatum*	nr	72	EC50, not specified	20	Norwegian Institute for Water Research (1994)
Corexit 9500	*Vibrio fischeri*	na	0.25	reduced bioluminescence	104 – 242	Fuller and Bonner (2001)
Corexit 9527	*Artemia* sp.	larvae	48	time to molt	42 – 72	Wells et al. (1982)
Corexit 9527	*Haliotis rufescens*	embryos	48	abnormal growth	1.6 – 2.2	Singer et al. (1990); Singer et al. (1991)
Corexit 9527	*Haliotis rufescens*	embryos	SD	abnormal growth	13.6 – 18.1	Singer et al. (1991)
Corexit 9527	*Hydra viridissima*	nr	168	NOEC	< 15	Mitchell and Holdway (2000)
Corexit 9527	*Macrobrachium rosenbergii*	embryo-larval	288	hatching	80.4	Law (1995)
Corexit 9527	*Macrocystis pyrifera*	zoospores	48	NOEC	1.3 – 2.1	Singer et al. (1990); Singer et al. (1991)
Corexit 9527	*Macrocystis pyrifera*	zoospores	SD	IC50, not specified	86.6 – 102	Singer et al. (1991)
Corexit 9527	*Macrocystis pyrifera*	zoospores	SD	NOEC	12.2 – 16.4	Singer et al. (1991)
Corexit 9527	*Polinices conicus*	nr	24	EC50, not specified	33.8	Gulec et al. (1997)
Corexit 9527	*Skeletonema costatum*	nr	72	biomass production	9.4	Nalco (2010)
Corexit 9527	*Vibrio fischeri*	na	0.25	reduced bioluminescence	4.9 – 12.8	George-Ares et al. (1999); Exxon Biomedical (1993a)[a]

Sources: NRC (2005) and George-Ares and Clark (2000)

Note: sublethal toxicity values were not used in further calculations.
EC50 – concentration that causes a non-lethal effect in 50% of an exposed population
IC50 – concentration required for 50% inhibition of a normal process (equivalent to an EC50)
NOEC – no-observed-effect concentration
nr – not reported
NRC – National Research Council
ppm – parts per million
SD – spiked concentration, declining exposure

Table 3. Available median lethal toxicity values (LC50) for crude oil

Oil Type	Weathered (Y/N)	Latin Name	Type of Exposure	Life Stage	Duration (h)	Oil LC50 (ppm TPH)	Source
ALC	Y	*Menidia beryllina*	static (75% renewal), sealed	early-life stage	96	4.9	Fuller and Bonner (2001) as cited in NRC (2005)
ALC	Y	*Menidia beryllina*	spiked	larval	96	32.3	Fuller and Bonner (2001) as cited in NRC (2005)
AMC	N	*Americamysis bahia*	static (75% renewal), sealed	larval	96	0.56	Fuller and Bonner (2001) as cited in NRC (2005)
AMC	N	*Americamysis bahia*	spiked	larval	96	26.1	Fuller and Bonner (2001) as cited in NRC (2005)
AMC	Y	*Cyprinodon variegatus*	static (75% renewal), sealed	larval	96	3.9	Fuller and Bonner (2001) as cited in NRC (2005)
AMC	Y	*Cyprinodon variegatus*	spiked	larval	96	6.1	Fuller and Bonner (2001) as cited in NRC (2005)
ANS	N	*Americamysis bahia*	flow-through	larval	96	2.61	Rhoton et al. (2001) as cited in NRC (2005)
ANS	N	*Americamysis bahia*	spiked	larval	96	8.21	Rhoton et al. (2001) as cited in NRC (2005)
ANS	N	*Boreogadus saida*	spiked	<1 year	96	1.2	McFarlin et al. (2011)
ANS	N	*Calanus glacialis*	spiked	nr	96	2.4	McFarlin et al. (2011)
ANS	N	*Fundulus grandis*	static	non-embryo	96	7.8	Liu (2003) as cited in Barron et al. (2013)
ANS	N	*Litopenaeus setiferus*	static	non-embryo	96	6.59	Liu et al. (2006)
ANS	Y	*Menidia beryllina*	flow-through	larval	96	0.79	Rhoton et al. (2001) as cited in NRC (2005)
ANS	N	*Menidia beryllina*	flow-through	larval	96	15.59	Rhoton et al. (2001) as cited in NRC (2005)
ANS	N	*Menidia beryllina*	spiked	larval	96	26.36	Rhoton et al. (2001) as cited in NRC (2005)
ANS	N	*Myoxocephalus sp.*	spiked	larvae	96	1.6	McFarlin et al. (2011)
BSC	N	*Allorchestes compressa*	static (60% renewal)	nr	96	311,000	Gulec et al. (1997)
BSC	N	*Hydra viridissima*[a]	static	nr	96	0.7	Mitchell and Holdway (2000)
BSC	N	*Macquaria novemaculeata*	static (50% renewal)	larval	96	465000	Gulec and Holdway (2000)
BSC	N	*Melonotaenia fluviatilis*[a]	static, daily renewal	embryo	96	1.28	Pollino and Holdway (2002)
BSC	N	*Octopus pallidus*	semi-static	hatchling	48	0.39	Long and Holdway (2002)
BSC	N	*Palaemon serenus*	static (50% renewal)	nr	96	258,000	Gulec and Holdway (2000)
Bunker C	N	*Americamysis almyra*	nr	nr	48 – 96[b]	0.9	Malins 1977 as cited in Barron et al. (2013)
Bunker C	nr	*Capitella capitata*	nr	nr	48 – 96[b]	0.9	Malins 1977 as cited in Barron et al. (2013)
Bunker C	nr	*Cyprinodon variegatus*	nr	nr	96	3.1	Malins 1977 as cited in Barron et al. (2013)
Bunker C	nr	*Farfantepenaeus aztecus*	nr	nr	48 – 96[b]	1.9	Malins 1977 as cited in Barron et al. (2013)
Bunker C	nr	*Fundulus similis*	nr	nr	96	1.69	Malins 1977 as cited in Barron et al. (2013)
Bunker C	nr	*Menidia beryllina*	nr	nr	96	1.9	Malins 1977 as cited in Barron et al. (2013)
Bunker C	nr	*Neanthes arenaceodentata*	nr	nr	48 – 96[b]	3.6	Malins 1977 as cited in Barron et al. (2013)
Bunker C	nr	*Pagurus longicarpus*	nr	nr	48 – 96[b]	0.42	Malins 1977 as cited in Barron et al. (2013)
Bunker C	nr	*Palaemonetes pugio*	nr	nr	48 – 96[b]	2.6	Malins 1977 as cited in Barron et al. (2013)
Bunker C	nr	*Spiochaetopterus costarum*	nr	nr	48 – 96[b]	4.92	Malins 1977 as cited in Barron et al. (2013)
CIC	nr	*Aulorhynchus flavidus*	nr	nr	96	1.34	Malins 1977 as cited in Barron et al. (2013)
CIC	nr	*Aulorhynchus flavidus*	nr	nr	96	2.55	Rice et al 1979 as cited in Barron et al. (2013)
CIC	nr	*Chlamys hastata*	nr	nr	48 – 96[b]	2	Moles 1998 as cited in Barron et al. (2013)
CIC	nr	*Chlamys hastata*	nr	nr	48 – 96[b]	3.94	Rice et al 1979 as cited in Barron et al. (2013)

Table 3. Available median lethal toxicity values (LC50) for crude oil, cont.

Oil Type	Weathered (Y/N)	Latin Name	Type of Exposure	Life Stage	Duration (h)	Oil LC50 (ppm TPH)	Source
CIC	nr	*Clupea pallasii*	nr	nr	96	1.22	Rice et al 1979 as cited in Barron et al. (2013)
CIC	nr	*Crangon alaskensis*	nr	nr	48 – 96 [b]	0.87	Rice et al 1979 as cited in Barron et al. (2013)
CIC	nr	*Eleginus gracilis*	nr	nr	48 – 96 [b]	2.28	Malins 1977 as cited in Barron et al. (2013)
CIC	nr	*Eualus fabricii*	nr	nr	48 – 96 [b]	1.46	Malins 1977 as cited in Barron et al. (2013)
CIC	nr	*Eualus suckleyi*	nr	nr	48 – 96 [b]	3.94	Rice et al 1979 as cited in Barron et al. (2013)
CIC	nr	*Myoxocephalus polyacanthocephalus*	nr	nr	96	3.82	Rice et al 1979 as cited in Barron et al. (2013)
CIC	nr	*Notoacmea scutum*	nr	nr	48 – 96 [b]	3.65	Malins 1977 as cited in Barron et al. (2013)
CIC	nr	*Notoacmea scutum*	nr	nr	48 – 96 [b]	8.18	Rice et al 1979 as cited in Barron et al. (2013)
CIC	nr	*Oncorhynchus gorbuscha*	nr	nr	96	1.2	Moles 1998 as cited in Barron et al. (2013)
CIC	nr	*Oncorhynchus gorbuscha*	nr	nr	96	1.47	Malins 1977 as cited in Barron et al. (2013)
CIC	nr	*Oncorhynchus gorbuscha*	nr	nr	96	1.5	Rice et al 1979 as cited in Barron et al. (2013)
CIC	nr	*Pagurus hirsutiusculus*	nr	nr	48 – 96 [b]	3.1	Malins 1977 as cited in Barron et al. (2013)
CIC	nr	*Pandalus borealis*	nr	nr	48 – 96 [b]	4.94	Rice et al 1979 as cited in Barron et al. (2013)
CIC	nr	*Pandalus danae*	nr	nr	48 – 96 [b]	0.81	Malins 1977 as cited in Barron et al. (2013)
CIC	nr	*Pandalus goniurus*	nr	nr	48 – 96 [b]	1.85	Malins 1977 as cited in Barron et al. (2013)
CIC	nr	*Pandalus hypsinotus*	nr	nr	48 – 96 [b]	1.4	Moles 1998 as cited in Barron et al. (2013)
CIC	nr	*Paralithodes camtschaticus*	nr	nr	48 – 96 [b]	1.5	Moles 1998as cited in Barron et al. (2013)
CIC	nr	*Paralithodes camtschaticus*	nr	nr	48 – 96 [b]	3.69	Rice et al 1979 as cited in Barron et al. (2013)
CIC	nr	*Platichthys stellatus*	nr	nr	96	1.8	Moles 1998 as cited in Barron et al. (2013)
CIC	nr	*Salvelinus malma*	nr	nr	96	1.54	Malins 1977 as cited in Barron et al. (2013)
CIC	nr	*Salvelinus malma*	nr	nr	96	1.55	Rice et al 1979 as cited in Barron et al. (2013)
CIC	nr	*Theragra chalcogramma*	nr	nr	48 – 96 [b]	1.73	Rice et al 1979 as cited in Barron et al. (2013)
Ecofisk	N	*Platichthys flesus*	constant	350-g juvenile	96	75	Baklien et al. (1986)
Iranian heavy crude	N	*Tigriopus japonicus*	static	juvenile	96	124.3	Lee et al. (2013)
KFO	N	*Americamysis bahia*	constant	nr	96	0.63	Clark et al. (2001)
KFO	N	*Americamysis bahia*	static daily renewal, sealed	nr	96	0.78	Pace et al. (1995) as cited in NRC (2005)
KFO	N	*Holmesimysis costata*	constant	nr	96	0.1	Clark et al. (2001)
KFO	Y	*Menidia beryllina*	constant	nr	96	0.14	Clark et al. (2001)
KFO	N	*Menidia beryllina*	constant	nr	96	0.97	Clark et al. (2001)
No. 2 fuel oil	nr	*Americamysis almyra*	nr	nr	48 – 96 [b]	0.9	Malins 1977 as cited in Barron et al. (2013)
No. 2 fuel oil	N	*Americamysis bahia*	static daily renewal	eggs	48	16.12	EPA (1995)
No. 2 fuel oil	nr	*Capitella capitata*	nr	nr	48 – 96 [b]	2.3	Malins 1977 as cited in Barron et al. (2013)
No. 2 fuel oil	nr	*Chlamys rubida*	nr	nr	48 – 96 [b]	0.8	Malins 1977 as cited in Barron et al. (2013)
No. 2 fuel oil	nr	*Crangon alaskensis*	nr	nr	48 – 96 [b]	0.36	Rice et al 1979 as cited in Barron et al. (2013)
No. 2 fuel oil	nr	*Cryptochiton stelleri*	nr	nr	48 – 96 [b]	1.24	Malins 1977 as cited in Barron et al. (2013)
No. 2 fuel oil	nr	*Cyprinodon variegatus*	nr	nr	96	6.3	Malins 1977 as cited in Barron et al. (2013)

Table 3. Available median lethal toxicity values (LC50) for crude oil, cont.

Oil Type	Weathered (Y/N)	Latin Name	Type of Exposure	Life Stage	Duration (h)	Oil LC50 (ppm TPH)	Source
No. 2 fuel oil	nr	*Eualus fabricii*	nr	nr	48 – 96 [b]	0.53	Malins 1977 as cited in Barron et al. (2013)
No. 2 fuel oil	nr	*Eualus suckleyi*	nr	nr	48 – 96 [b]	0.59	Rice et al 1979 as cited in Barron et al. (2013)
No. 2 fuel oil	nr	*Fundulus similis*	nr	nr	96	3.9	Malins 1977 as cited in Barron et al. (2013)
No. 2 fuel oil	nr	*Katharina tunicata*	nr	nr	48 – 96 [b]	0.44	Malins 1977 as cited in Barron et al. (2013)
No. 2 fuel oil	nr	*Menidia beryllina*	nr	nr	96	3.9	Malins 1977 as cited in Barron et al. (2013)
No. 2 fuel oil	N	*Menidia beryllina*	nr	nr	96	10.72	EPA (1995)
No. 2 fuel oil	nr	*Myoxocephalus polyacanthocephalus*	nr	nr	96	1.31	Rice et al 1979 as cited in Barron et al. (2013)
No. 2 fuel oil	nr	*Neanthes arenaceodentata*	nr	nr	48 – 96 [b]	2.6	Malins 1977 as cited in Barron et al. (2013)
No. 2 fuel oil	nr	*Notoacmea scutum*	nr	nr	48 – 96 [b]	5.04	Malins 1977 as cited in Barron et al. (2013)
No. 2 fuel oil	nr	*Oncorhynchus gorbuscha*	nr	nr	96	0.54	Rice et al 1979 as cited in Barron et al. (2013)
No. 2 fuel oil	nr	*Oncorhynchus gorbuscha*	nr	nr	96	0.81	Malins 1977 as cited in Barron et al. (2013)
No. 2 fuel oil	nr	*Palaemonetes pugio*	nr	nr	48 – 96 [b]	3.5	Malins 1977 as cited in Barron et al. (2013)
No. 2 fuel oil	nr	*Pandalus borealis*	nr	nr	48 – 96 [b]	0.21	Malins 1977 as cited in Barron et al. (2013)
No. 2 fuel oil	nr	*Pandalus danae*	nr	nr	48 – 96 [b]	0.8	Malins 1977 as cited in Barron et al. (2013)
No. 2 fuel oil	nr	*Pandalus goniurus*	nr	nr	48 – 96 [b]	1.69	Malins 1977 as cited in Barron et al. (2013)
No. 2 fuel oil	nr	*Paralithodes camtschaticus*	nr	nr	48 – 96 [b]	0.81	Rice et al 1979 as cited in Barron et al. (2013)
No. 2 fuel oil	nr	*Paralithodes camtschaticus*	nr	nr	48 – 96 [b]	5.1	Malins 1977 as cited in Barron et al. (2013)
No. 2 fuel oil	nr	*Salvelinus malma*	nr	nr	96	0.15	Rice et al 1979 as cited in Barron et al. (2013)
No. 2 fuel oil	nr	*Salvelinus malma*	nr	nr	96	2.29	Malins 1977 as cited in Barron et al. (2013)
No. 2 fuel oil	nr	*Xenacanthomysis pseudomacropsis*	nr	nr	48 – 96 [b]	2.31	Rice et al 1979 as cited in Barron et al. (2013)
Norman Wells Crude	Y	*Daphnia magna*	static	larval	48	4	Bobra et al. (1989)
Norman Wells Crude	N	*Daphnia magna*	static	larval	48	10	Bobra et al. (1989)
PBCO	N	*Atherinops affinis*	spiked	early-life stage	96	9.35	Singer et al. (2001) as cited in NRC (2005)
PBCO	nr	*Chlamys rubida*	nr	nr	48 – 96 [b]	2.07	Malins 1977 as cited in Barron et al. (2013)
PBCO	nr	*Cottus cognatus*	nr	nr	48 – 96 [b]	3	Moles et al. 1979 as cited in Barron et al. (2013)
PBCO	nr	*Eualus fabricii*	nr	nr	48 – 96 [b]	1.94	Malins 1977 as cited in Barron et al. (2013)
PBCO	Y	*Holmesimysis costata*	spiked	nr	96	0.951	Singer et al. (2001) as cited in NRC (2005)
PBCO	N	*Holmesimysis costata*	spiked	early-life stage	96	14.23	Singer et al. (2001) as cited in NRC (2005)
PBCO	N	*Menidia beryllina*	spiked	larval	96	11.83	Singer et al. (2001) as cited in NRC (2005)
PBCO	N	*Menidia beryllina*	flow-through	larval	96	14.81	Rhoton et al. (2001) as cited in NRC (2005)
PBCO	nr	*Oncorhynchus gorbuscha*	nr	nr	96	1.41	Malins 1977 as cited in Barron et al. (2013)
PBCO	nr	*Oncorhynchus gorbuscha*	nr	nr	96	3.73	Moles et al. 1979 as cited in Barron et al. (2013)
PBCO	nr	*Oncorhynchus kisutch*	nr	nr	96	1.45	Moles et al. 1979 as cited in Barron et al. (2013)
PBCO	nr	*Oncorhynchus nerka*	nr	nr	96	1.05	Moles et al. 1979 as cited in Barron et al. (2013)
PBCO	nr	*Oncorhynchus tshawytscha*	nr	nr	96	1.47	Moles et al. 1979 as cited in Barron et al. (2013)
PBCO	nr	*Pandalus borealis*	nr	nr	48 – 96 [b]	2.11	Malins 1977 as cited in Barron et al. (2013)

Table 3. Available median lethal toxicity values (LC50) for crude oil, cont.

Oil Type	Weathered (Y/N)	Latin Name	Type of Exposure	Life Stage	Duration (h)	Oil LC50 (ppm TPH)	Source
PBCO	nr	*Pandalus goniurus*	nr	nr	48 – 96 [b]	1.26	Malins 1977 as cited in Barron et al. (2013)
PBCO	nr	*Pandalus hypsinotus*	nr	nr	48 – 96 [b]	1.96	Malins 1977 as cited in Barron et al. (2013)
PBCO	nr	*Paralithodes camtschaticus*	nr	nr	48 – 96 [b]	2.35	Malins 1977 as cited in Barron et al. (2013)
PBCO	nr	*Salvelinus alpinus*	nr	nr	96	2.17	Moles et al 1979 as cited in Barron et al. (2013)
PBCO	nr	*Salvelinus malma*	nr	nr	96	1.1	Malins 1977 as cited in Barron et al. (2013)
PBCO	nr	*Salvelinus malma*	nr	nr	96	1.25	Moles et al 1979 as cited in Barron et al. (2013)
PBCO	nr	*Thymallus arcticus*	nr	nr	48 – 96 [b]	2.04	Moles et al 1979 as cited in Barron et al. (2013)
PBCO	N	*Oncorhynchus tshawytscha*	constant	juvenile	96	6.2	Van Scoy et al. (2010)
PBCO	N	*Oncorhynchus tshawytscha*	constant	juvenile	96	7.46	Lin et al. (2009)
SLC	nr	*Americamysis almyra*	nr	nr	48 – 96 [b]	8.7	Malins 1977 as cited in Barron et al. (2013)
SLC	N	*Americamysis bahia*	nr	nr	48	2.7	Hemmer et al. (2011)
SLC	nr	*Capitella capitata*	nr	nr	48 – 96 [b]	12	Malins 1977 as cited in Barron et al. (2013)
SLC	nr	*Cyprinodon variegatus*	nr	nr	96	19.8	Malins 1977 as cited in Barron et al. (2013)
SLC	N	*Fundulus grandis*	static	non-embryo	96	8.3	Liu et al 2003 as cited in Barron et al. (2013)
SLC	nr	*Fundulus similis*	nr	nr	96	16.8	Malins 1977 as cited in Barron et al. (2013)
SLC	nr	*Leander tenuicornis*	nr	nr	48 – 96 [b]	6	Malins 1977 as cited in Barron et al. (2013)
SLC	N	*Litopenaeus setiferus*	static	non-embryo	96	6.5	Liu et al 2003 as cited in Barron et al. (2013)
SLC	N	*Menidia beryllina*	nr	nr	96	3.5	Hemmer et al. (2011)
SLC	nr	*Menidia beryllina*	nr	nr	96	5.5	Malins 1977 as cited in Barron et al. (2013)
SLC	nr	*Neanthes arenaceodentata*	nr	nr	48 – 96 [b]	12	Malins 1977 as cited in Barron et al. (2013)
SLC	nr	*Palaemonetes pugio*	nr	nr	48 – 96 [b]	10.7	Malins 1977 as cited in Barron et al. (2013)
SLC	nr	*Platynereis dumerilii*	nr	nr	48 – 96 [b]	9.5	Malins 1977 as cited in Barron et al. (2013)
VCO	N	*Americamysis bahia*	static (90% renewal), sealed	larval	96	0.15	Wetzel and Van Fleet (2001)
VCO	N	*Americamysis bahia*	spiked	larval	96	0.59	Wetzel and Van Fleet (2001)
VCO	N	*Menidia beryllina*	spiked	larval	96	0.63	Wetzel and Van Fleet (2001)
VCO	N	*Sciaenops ocellatus*	spiked	larval	96	0.85	Wetzel and Van Fleet (2001)

Primary sources: NRC (2005), George-Ares and Clark (2000), and Barron et al. (2013) (supplemental material)

[a] Freshwater species.

[b] Exact durations were not reported by Barron et al. (2013), but the acceptability criterion for invertebrate species tests was reported as between 48 and 96 hours.

ALC – Arabian light crude oil
AMC – Arabian medium fuel oil
ANS – Alaska North Slope crude oil
BSC – Bass Strait crude oil
CIC – Cook Inlet crude oil
EPA – US Environmental Protection Agency
KFO – Kuwait fuel oil
LC50 – concentration that is lethal to 50% of an exposed population
nr – not reported
PBCO – Prudhoe Bay crude oil
ppm – parts per million
SLC – Sweet Louisiana Crude oil
TPH – total petroleum hydrocarbons
VCO – Venezuelan medium crude oil

Table 4. Available median lethal toxicity values (LC50) for oil and oil dispersed by current-use and NPL-listed chemical dispersants

Dispersant Chemical	Oil Type	Weathered (Y/N)	DOR	Latin Name	Type of Exposure	Life Stage	Duration (h)	Oil-only LC50 (ppm TPH)	Dispersed Oil LC50 (ppm TPH)	Relative Toxicity[a]	Source
Corexit 9500	BSC	N	1:10	*Allorchestes compressa*	static (60% renewal)	nr	96	311,000	14.8	more toxic	Gulec et al. (1997)
Corexit 9500	AMC	N	1:10	*Americamysis bahia*	static (75% renewal), sealed	larval	96	0.56 – 0.67	0.64 – 0.65	less toxic	Fuller and Bonner (2001) as cited in NRC (2005)
Corexit 9500	AMC	N	1:10	*Americamysis bahia*	spiked	larval	96	26.1 – 83.1	56.5 – 60.8	less toxic	Fuller and Bonner (2001) as cited in NRC (2005)
Corexit 9500	ANS	N	1:10	*Americamysis bahia*	continuous	larval	96	2.61	1.4	more toxic	Rhoton et al. (2001) as cited in NRC (2005)
Corexit 9500	ANS	N	1:10	*Americamysis bahia*	spiked	larval	96	8.21	5.08	more toxic	Rhoton et al. (2001) as cited in NRC (2005)
Corexit 9500	Forties	N	1:10	*Americamysis bahia*	constant	nr	96	--	0.42	na	Clark et al. (2001)
Corexit 9500	Forties	N	1:10	*Americamysis bahia*	spiked	nr	96	--	15.3	na	Clark et al. (2001)
Corexit 9500	No. 2 fuel oil	N	1:10	*Americamysis bahia*	static daily renewal	eggs	48	16.12	3.4	more toxic	EPA (1995)
Corexit 9500	PBCO	N	1:10	*Americamysis bahia*	spiked	larval	96	>6.86	15.9	na	Wetzel and Van Fleet (2001)
Corexit 9500	VCO	N	1:10	*Americamysis bahia*	spiked	larval	96	0.59 – 0.89	10.2 – 18.1	less toxic	Wetzel and Van Fleet (2001)
Corexit 9500	VCO	N	1:10	*Americamysis bahia*	static (90% renewal), sealed	larval	96	0.15 – 0.4	0.5 – 0.53	less toxic	Wetzel and Van Fleet (2001)
Corexit 9500	VCO	Y	1:10	*Americamysis bahia*	spiked	larval	96	> 0.63 – > 0.83	72.6 – 120.8	na	Wetzel and Van Fleet (2001)
Corexit 9500	PBCO	N	1:10	*Atherinops affinis*	spiked	early-life stage	96	9.35 – 12.13	7.27 – 17.7	more toxic	Singer et al. (2001) as cited in NRC (2005)
Corexit 9500	PBCO	Y	1:10	*Atherinops affinis*	spiked	nr	96	> 1.45 – > 1.60	16.86 – 18.06	na	Singer et al. (2001) as cited in NRC (2005)
Corexit 9500	ANS	N	1:20	*Boreogadus saida*	spiked	< 1 year	96	1.2	45	less toxic	
Corexit 9500	ANS	N	1:20	*Boreogadus saida*	spiked	< 1 year	96	2	46	less toxic	McFarlin et al. (2011)
Corexit 9500	ANS	N	1:20	*Boreogadus saida*	spiked	< 1 year	96	1.5	80	less toxic	McFarlin et al. (2011)
Corexit 9500	ANS	N	1:20	*Boreogadus saida*	spiked	< 1 year	96	--	50	na	McFarlin et al. (2011)
Corexit 9500	ANS	N	1:20	*Calanus glacialis*	spiked	nr	96	4	14	less toxic	McFarlin et al. (2011)
Corexit 9500	ANS	N	1:20	*Calanus glacialis*	spiked	nr	96	2.4	15	less toxic	McFarlin et al. (2011)
Corexit 9500	ANS	N	1:20	*Calanus glacialis*	spiked	nr	96	5	16	less toxic	McFarlin et al. (2011)
Corexit 9500	ANS	N	1:20	*Calanus glacialis*	spiked	nr	96	3.3	18	less toxic	McFarlin et al. (2011)
Corexit 9500	ANS	N	1:20	*Calanus glacialis*	spiked	nr	96	> 1.0	30	na	McFarlin et al. (2011)
Corexit 9500	ANS	N	1:20	*Calanus glacialis*	spiked	nr	96	> 5.5	30	na	McFarlin et al. (2011)
Corexit 9500	ANS	N	1:20	*Calanus glacialis*	spiked	nr	96	4	37	less toxic	McFarlin et al. (2011)
Corexit 9500	ANS	N	1:20	*Calanus glacialis*	spiked	nr	96	> 1.0	50	na	McFarlin et al. (2011)
Corexit 9500	ANS	N	1:20	*Calanus glacialis*	spiked	nr	96	> 0.8	75	na	McFarlin et al. (2011)
Corexit 9500	ANS	N	1:20	*Calanus glacialis*	spiked	nr	96	> 0.9	75	na	McFarlin et al. (2011)
Corexit 9500	ANS	N	1:20	*Calanus glacialis*	spiked	nr	96	> 0.8	79	na	McFarlin et al. (2011)
Corexit 9500	ALC	N	1:25	*Clupea harengus*	static daily renewal	embryos	336	--	4.33	na	Lee et al. (2011)
Corexit 9500	ANS	N	1:25	*Clupea harengus*	static daily renewal	embryos	336	--	2.03	na	Lee et al. (2011)
Corexit 9500	ANS	N	1:25	*Clupea pallasi*	static daily renewal	embryos	336	--	1.94	na	Lee et al. (2011)
Corexit 9500	MESA	N	1:25	*Clupea pallasi*	static daily renewal	embryos	336	--	1.75	na	Lee et al. (2011)

Table 4. Available median lethal toxicity values (LC50) for oil and oil dispersed by current-use and NPL-listed chemical dispersants, cont.

Dispersant Chemical	Oil Type	Weathered (Y/N)	DOR	Latin Name	Type of Exposure	Life Stage	Duration (h)	Oil-only LC50 (ppm TPH)	Dispersed Oil LC50 (ppm TPH)	Relative Toxicity[a]	Source
Corexit 9500	Forties	N	1:10	*Crassostrea gigas*	constant	larval	48	--	0.81	na	Clark et al. (2001)
Corexit 9500	Forties	N	1:10	*Crassostrea gigas*	spiked	larval	48	--	3.99	na	Clark et al. (2001)
Corexit 9500	AMC	Y	1:10	*Cyprinodon variegatus*	spiked	larval	96	> 5.7 – 6.1	31.9 – 39.5	na	Fuller and Bonner (2001) as cited in NRC (2005)
Corexit 9500	AMC	Y	1:10	*Cyprinodon variegatus*	static (75% renewal), sealed	larval	96	3.9 – 4.2	>9.7 – 10.8	na	Fuller and Bonner (2001) as cited in NRC (2005)
Corexit 9500	PBCO	Y	1:10	*Holmesimysis costata*	spiked	nr	96	0.951 – 1.03	5.72 – 33.27	less toxic	Singer et al. (2001) as cited in NRC (2005)
Corexit 9500	BSC	N	1:29	*Hydra viridissima*[b]	static	larval	96	0.7	7.2	less toxic	Mitchell and Holdway (2000)
Corexit 9500	ANS	N	1:20	*Litopenaeus setiferus*	static	non-embryo	96	6.59	7.5	less toxic	Liu et al. (2006)
Corexit 9500	BSC	N	1:10	*Macquaria novemaculeata*	static (50% renewal)	larval	96	465000	14.1	more toxic	Gulec and Holdway (2000)
Corexit 9500	BSC	N	1:50	*Melonotaenia fluviatilis*[b]	static, daily renewal	embryo	24	4.48	2.26	more toxic	Pollino and Holdway (2002)
Corexit 9500	BSC	N	1:50	*Melonotaenia fluviatilis*[b]	static, daily renewal	embryo	48	3.38	1.94	more toxic	Pollino and Holdway (2002)
Corexit 9500	BSC	N	1:50	*Melonotaenia fluviatilis*[b]	static, daily renewal	embryo	72	2.1	1.67	more toxic	Pollino and Holdway (2002)
Corexit 9500	BSC	N	1:50	*Melonotaenia fluviatilis*[b]	static, daily renewal	embryo	96	1.28	1.37	less toxic	Pollino and Holdway (2002)
Corexit 9500	ALC	Y	1:10	*Menidia beryllina*	spiked	larval	96	> 14.5 – 32.3	24.9 – 36.9	na	Fuller and Bonner (2001) as cited in NRC (2005)
Corexit 9500	ALC	Y	1:10	*Menidia beryllina*	static (75% renewal), sealed	early-life stage	96	4.9 – 5.5	1.5 – 2.5	more toxic	Fuller and Bonner (2001) as cited in NRC (2005)
Corexit 9500	ANS	Y	1:10	*Menidia beryllina*	continuous	larval	96	0.79	0.65	more toxic	Rhoton et al. (2001) as cited in NRC (2005)
Corexit 9500	ANS	N	1:10	*Menidia beryllina*	continuous	larval	96	15.59	12.42	more toxic	Rhoton et al. (2001) as cited in NRC (2005)
Corexit 9500	ANS	N	1:10	*Menidia beryllina*	spiked	larval	96	26.36	12.22	more toxic	Rhoton et al. (2001) as cited in NRC (2005)
Corexit 9500	ANS	Y	1:10	*Menidia beryllina*	spiked	larval	96	> 1.13	18.89	na	Rhoton et al. (2001) as cited in NRC (2005)
Corexit 9500	Forties	N	1:10	*Menidia beryllina*	constant	nr	96	--	0.49	na	Clark et al. (2001)
Corexit 9500	Forties	N	1:10	*Menidia beryllina*	spiked	early-life stage	96	--	9.05	na	Clark et al. (2001)
Corexit 9500	PBCO	N	1:10	*Menidia beryllina*	continuous	larval	96	14.81	4.57	more toxic	Rhoton et al. (2001) as cited in NRC (2005)
Corexit 9500	PBCO	N	1:10	*Menidia beryllina*	spiked	larval	96	> 19.86	12.29	more toxic	Rhoton et al. (2001) as cited in NRC (2005)
Corexit 9500	PBCO	Y	1:10	*Menidia beryllina*	spiked	larval	96	--	20.28	na	Singer et al. (2001) as cited in NRC (2005)
Corexit 9500	PBCO	N	1:10	*Menidia beryllina*	spiked	larval	96	11.83	32.47	less toxic	Singer et al. (2001) as cited in NRC (2005)
Corexit 9500	PBCO	N	1:10	*Menidia beryllina*	spiked	larval	96	> 6.86	18.1	na	Wetzel and Van Fleet (2001)
Corexit 9500	VCO	N	1:10	*Menidia beryllina*	spiked	larval	96	0.63	2.84	less toxic	Wetzel and Van Fleet (2001)
Corexit 9500	VCO	Y	1:10	*Menidia beryllina*	spiked	larval	96	> 1.06	30.8	na	Wetzel and Van Fleet (2001)
Corexit 9500	VCO	N	1:10	*Menidia beryllina*	static (90% renewal), sealed	larval	96	<0.11	0.68	less toxic	Wetzel and Van Fleet (2001)
Corexit 9500	ANS	N	1:20	*Myoxocephalus* sp.	spiked	larvae	96	> 1.4	18	na	McFarlin et al. (2011)
Corexit 9500	ANS	N	1:20	*Myoxocephalus* sp.	spiked	larvae	96	1.6	17	less toxic	McFarlin et al. (2011)
Corexit 9500	ANS	N	1:20	*Myoxocephalus* sp.	spiked	larvae	96	3	29	less toxic	McFarlin et al. (2011)

Table 4. Available median lethal toxicity values (LC50) for oil and oil dispersed by current-use and NPL-listed chemical dispersants, cont.

Dispersant Chemical	Oil Type	Weathered (Y/N)	DOR	Latin Name	Type of Exposure	Life Stage	Duration (h)	Oil-only LC50 (ppm TPH)	Dispersed Oil LC50 (ppm TPH)	Relative Toxicity[a]	Source
Corexit 9500	ANS	N	1:20	*Myoxocephalus* sp.	spiked	larvae	96	3.3	46	less toxic	McFarlin et al. (2011)
Corexit 9500	PCBO	N	1:10	*Oncorhynchus tshawytscha*	constant	juvenile	96	6.2 – 9.9	37 – 60.5	less toxic	Van Scoy et al. (2010)
Corexit 9500	PCBO	N	1:10	*Oncorhynchus tshawytscha*	constant	juvenile	96	7.46	155.93	less toxic	Lin et al. (2009)
Corexit 9500	BSC	N	1:10	*Palaemon serenus*	static (50% renewal)	nr	96	258000	3.6	more toxic	Gulec and Holdway (2000)
Corexit 9500	VCO	N	1:10	*Sciaenops ocellatus*	spiked	larval	96	0.85	4.23	less toxic	Wetzel and Van Fleet (2001)
Corexit 9500	Forties	N	1:10	*Scophthalamus maximus*	constant	nr	48	0.35	0.44	less toxic	Clark et al. (2001)
Corexit 9500	Forties	N	1:10	*Scophthalamus maximus*	spiked	nr	48	> 1.33	48.6	na	Clark et al. (2001)
Corexit 9500	Iranian heavy crude	N	1:10	*Tigriopus japonicus*	static	juvenile	96	124.3	10.7	more toxic	Lee et al. (2013)
Corexit 9527	BSC	N	1:10	*Allorchestes compressa*	static (60% renewal)	nr	96	311000	16.2	more toxic	(Gulec et al., 1997) as cited in NRC (2005)
Corexit 9527	KCO	Y	1:10	*Americamysis bahia*	constant	nr	96	--	0.11	na	Clark et al. (2001)
Corexit 9527	KCO	N	1:10	*Americamysis bahia*	constant	nr	96	0.63	0.65	less toxic	Clark et al. (2001)
Corexit 9527	KCO	N	1:10	*Americamysis bahia*	spiked	nr	96	> 2.93	17.2	na	Clark et al. (2001)
Corexit 9527	KCO	Y	1:10	*Americamysis bahia*	spiked	nr	96	> 0.17	111	na	Clark et al. (2001)
Corexit 9527	KCO	N	1:10	*Americamysis bahia*	spiked	nr	96	> 2.9	17.7	na	Pace et al. (1995) as cited in NRC (2005)
Corexit 9527	KCO	N	1:10	*Americamysis bahia*	static daily renewal, sealed	nr	96	0.78	0.98	less toxic	Pace et al. (1995) as cited in NRC (2005)
Corexit 9527	PBCO	N	1:10	*Atherinops affinis*	spiked	early-life stage	96	16.34 – 40.2	28.6 – 74.73	less toxic	Singer et al. (1998) as cited in NRC (2005)
Corexit 9527	ANS	Y	1:25	*Clupea pallasi*	static	larval	24	0.045	0.199	less toxic	Barron et al. (2004) as cited in NRC (2005)
Corexit 9527	KCO	N	1:10	*Crassostrea gigas*	constant	larval	48	--	0.5	na	Clark et al. (2001)
Corexit 9527	KCO	N	1:10	*Crassostrea gigas*	spiked	larval	48	--	1.92	na	Clark et al. (2001)
Corexit 9527	MFO	N	1:10	*Crassostrea gigas*	constant	larval	48	> 1.14	0.53	more toxic	Clark et al. (2001)
Corexit 9527	MFO	N	1:10	*Crassostrea gigas*	spiked	larval	48	> 1.83	2.28	na	Clark et al. (2001)
Corexit 9527	Norman Wells crude	N	1:20	*Daphnia magna*	static	larval	48	10	14	less toxic	Bobra et al. (1989)
Corexit 9527	Norman Wells crude	Y	1:20	*Daphnia magna*	static	larval	48	4	15	less toxic	Bobra et al. (1989)
Corexit 9527	Norman Wells crude	Y	1:20	*Daphnia magna*	static	larval	48	> 0.2	17	na	Bobra et al. (1989)
Corexit 9527	KCO	N	1:10	*Holmesimysis costata*	constant	nr	96	0.1	0.17	less toxic	Clark et al. (2001)
Corexit 9527	KCO	N	1:10	*Holmesimysis costata*	spiked	nr	96	> 2.76	1.8	more toxic	Clark et al. (2001)
Corexit 9527	PBCO	N	1:10	*Holmesimysis costata*	spiked	juvenile	96	> 25.45 – > 34.68	10.54 – 10.83	more toxic	Singer et al. (1998) as cited in NRC (2005)
Corexit 9527	PBCO	N	1:10	*Holmesimysis costata*	spiked	early-life stage	96	14.23 – > 17.5	9.46 – 14.4	more toxic	Singer et al. (2001) as cited in NRC (2005)
Corexit 9527	BSC	N	1:29	*Hydra viridissima*[b]	static	nr	96	0.7	9	less toxic	Mitchell and Holdway (2000)
Corexit 9527	BSC	N	1:10	*Macquaria novemaculeata*	static (50% renewal)	larval	96	465000	28.5	more toxic	Gulec and Holdway (2000)

Table 4. Available median lethal toxicity values (LC50) for oil and oil dispersed by current-use and NPL-listed chemical dispersants, cont.

Dispersant Chemical	Oil Type	Weathered (Y/N)	DOR	Latin Name	Type of Exposure	Life Stage	Duration (h)	Oil-only LC50 (ppm TPH)	Dispersed Oil LC50 (ppm TPH)	Relative Toxicity[a]	Source
Corexit 9527	BSC	N	1:50	*Melonotaenia fluviatilis*[b]	static, daily renewal	embryo	48	3.38	2.92	more toxic	Pollino and Holdway (2002)
Corexit 9527	BSC	N	1:50	*Melonotaenia fluviatilis*[b]	static, daily renewal	embryo	72	2.1	1.25	more toxic	Pollino and Holdway (2002)
Corexit 9527	BSC	N	1:50	*Melonotaenia fluviatilis*[b]	static, daily renewal	embryo	96	1.28	0.74	more toxic	Pollino and Holdway (2002)
Corexit 9527	KCO	N	1:10	*Menidia beryllina*	constant	nr	96	0.97	0.55	more toxic	Clark et al. (2001)
Corexit 9527	KCO	Y	1:10	*Menidia beryllina*	constant	nr	96	0.14	1.09	less toxic	Clark et al. (2001)
Corexit 9527	KCO	N	1:10	*Menidia beryllina*	spiked	nr	96	> 1.32	6.45	na	Clark et al. (2001)
Corexit 9527	KCO	Y	1:10	*Menidia beryllina*	spiked	nr	96	> 0.66	10.9	na	Clark et al. (2001)
Corexit 9527	BSC	N	1:50	*Octopus pallidus*	semi-static	hatchling	24	0.51	3.11	less toxic	Long and Holdway (2002)
Corexit 9527	BSC	N	1:50	*Octopus pallidus*	semi-static	hatchling	48	0.39	1.8	less toxic	Long and Holdway (2002)
Corexit 9527	BSC	N	1:10	*Palaemon serenus*	static (50% renewal)	nr	96	258000	8.1	more toxic	Gulec and Holdway (2000)
Corexit 9527	Ecofisk	N	1:1	*Platichthys flesus*	constant	350-g juvenile	96	75	--	more toxic	Baklien et al. (1986)
Corexit 9527	KCO	N	1:10	*Scophthalamus maximus*	constant	nr	48	--	2	na	Clark et al. (2001)
Corexit 9527	KCO	N	1:10	*Scophthalamus maximus*	spiked	nr	48	--	16.5	na	Clark et al. (2001)
Norchem OSD-570	Diesel oil	N	1:10	*Balanus amphitrite*	static	larval	24	--	505	na	Wu et al. (1997)
Norchem OSD-570	Diesel oil	N	1:10	*Balanus amphitrite*	static	larval	48	--	71	na	Wu et al. (1997)
nr	Middle East crude oil	N	nr	*Paleamon elegans*	static	nr	24	83.5	1.1	more toxic	Unsal (1991) as cited in NRC (2005)
Omniclean	No. 2 fuel oil	N	1:1 to 1:10	*Cyprinodon variegatus*	static	larval	96	94	80 – 165	more toxic	Adams et al. (1999) as cited in NRC (2005)
Vecom B-1425	Diesel oil	N	1:10	*Balanus amphitrite*	static	larval	24	--	514	na	Wu et al. (1997)
Vecom B-1425	Diesel oil	N	1:10	*Balanus amphitrite*	static	larval	48	--	48	na	Wu et al. (1997)

Primary sources: NRC (2005) and George-Ares and Clark (2000)

[a] Relative toxicity indicates whether the mixture of dispersant and crude oil is more or less toxic than crude oil alone. The determination is based on the lowest available, comparable LC50 values for both crude oil and dispersed oil reported in the study. Comparable data are bounded LC50 values or unbounded LC50 ranges that exclude the other bounded LC50 value or unbounded range.

[b] Freshwater species.

AMC – Arabian medium crude
ANS – Alaska North Slope crude oil
BSC – Bass Strait crude oil
DOR – dispersant-to-oil ratio
KCO – Kuwait crude oil
LC50 – lethal concentration for 50 % of the organisms tested
MESA – medium South American crude oil
MFO – medium fuel oil
NPL – National Priorities List
na – not applicable
nr – not reported
NRC – National Research Council
PBCO – Prudhoe Bay crude oil
VCO – Venezuelan medium crude oil

Table 5. Available sublethal toxicity values for oil and oil dispersed by current-use and NPL-listed chemical dispersants

Dispersant Chemical	Oil Type	Weathered (Y/N)	DOR	Latin Name	Type of Exposure	Life Stage	Duration (h)	Endpoint	Oil EC50 (ppm TPH)	Dispersed Oil EC50 (ppm TPH)	Source(s)
Corexit 9500	ANS	N	nr	*Oncorhynchus mykiss*	static daily renewal	embryo	528	BSD Index	> 0.362	> 0.606	Wu et al. (2012)
Corexit 9500	Federated	N	nr	*Oncorhynchus mykiss*	static daily renewal	embryo	528	BSD Index	> 0.508	> 0.589	Wu et al. (2012)
Corexit 9500	MESA	N	nr	*Oncorhynchus mykiss*	static daily renewal	embryo	528	BSD Index	> 0.895	> 0.506	Wu et al. (2012)
Corexit 9500	Scotian light	N	nr	*Oncorhynchus mykiss*	static daily renewal	embryo	528	BSD Index	> 1.744	> 5.369	Wu et al. (2012)
Corexit 9500	ANS	N	nr	*Oncorhynchus mykiss*	static daily renewal	nr	528	chronic mortality	> 0.362	0.764	Wu et al. (2012)
Corexit 9500	Federated	N	nr	*Oncorhynchus mykiss*	static daily renewal	nr	528	chronic mortality	> 0.508	0.714	Wu et al. (2012)
Corexit 9500	MESA	N	nr	*Oncorhynchus mykiss*	static daily renewal	nr	528	chronic mortality	0.880	0.614	Wu et al. (2012)
Corexit 9500	Scotian light	N	nr	*Oncorhynchus mykiss*	static daily renewal	nr	528	chronic mortality	> 1.744	3.281	Wu et al. (2012)
Corexit 9500	ANS	N	nr	*Oncorhynchus mykiss*	static daily renewal	embryo	528	EROD activity (CYP1A induction)	> 0.362	0.500	Wu et al. (2012)
Corexit 9500	Federated	N	nr	*Oncorhynchus mykiss*	static daily renewal	embryo	528	EROD activity (CYP1A induction)	0.293	> 0.589	Wu et al. (2012)
Corexit 9500	MESA	N	nr	*Oncorhynchus mykiss*	static daily renewal	embryo	528	EROD activity (CYP1A induction)	0.735	0.517	Wu et al. (2012)
Corexit 9500	Mesa sour crude	Y	1:20	*Oncorhynchus mykiss*	static daily renewal	juvenile	48	EROD activity (CYP1A induction)	1.06E-05	1.00E-07	Ramachandran et al. (2004)
Corexit 9500	Scotian light	N	1:50	*Oncorhynchus mykiss*	static daily renewal	juvenile	48	EROD activity (CYP1A induction)	3.90E-05	6.60E-06	Ramachandran et al. (2004)
Corexit 9500	Scotian light	N	nr	*Oncorhynchus mykiss*	static daily renewal	embryo	528	EROD activity (CYP1A induction)	> 1.744	2.415	Wu et al. (2012)
Corexit 9500	Terra Nova	N	1:20	*Oncorhynchus mykiss*	static daily renewal	juvenile	48	EROD activity (CYP1A induction)	3.35E-04	3.00E-07	Ramachandran et al. (2004)
Corexit 9500	ANS	N	nr	*Oncorhynchus mykiss*	static daily renewal	embryo	528	percentage normal	0.133	0.226	Wu et al. (2012)
Corexit 9500	Federated	N	nr	*Oncorhynchus mykiss*	static daily renewal	embryo	528	percentage normal	0.072	0.053	Wu et al. (2012)
Corexit 9500	MESA	N	nr	*Oncorhynchus mykiss*	static daily renewal	embryo	528	percentage normal	0.657	0.157	Wu et al. (2012)
Corexit 9500	Scotian light	N	nr	*Oncorhynchus mykiss*	static daily renewal	embryo	528	percentage normal	1.440	1.168	Wu et al. (2012)
Corexit 9500	BSC	N	1:29	*Hydra viridissima*	static renewal	adult	168	population growth rate	> 0.6	4	Mitchell and Holdway (2000)
Corexit 9500	ANS	N	nr	*Oncorhynchus mykiss*	static daily renewal	embryo	528	ratio of yolk weight to fish weight	> 0.362	> 1.015	Wu et al. (2012)
Corexit 9500	Federated	N	nr	*Oncorhynchus mykiss*	static daily renewal	embryo	528	ratio of yolk weight to fish weight	> 0.508	> 1.218	Wu et al. (2012)
Corexit 9500	MESA	N	nr	*Oncorhynchus mykiss*	static daily renewal	embryo	528	ratio of yolk weight to fish weight	0.823	0.777	Wu et al. (2012)
Corexit 9500	Scotian light	N	nr	*Oncorhynchus mykiss*	static daily renewal	embryo	528	ratio of yolk weight to fish weight	> 1.744	> 3.996	Wu et al. (2012)
Corexit 9500	ANS	N	1:25	*Clupea harengus*	static	embryos	2.4	reduced hatch	nr	11.08	Lee et al. (2011)
Corexit 9500	ANS	N	1:25	*Clupea harengus*	static	embryos	8	reduced hatch	nr	3.07	Lee et al. (2011)
Corexit 9500	ANS	N	1:25	*Clupea harengus*	static	embryos	24	reduced hatch	nr	0.49	Lee et al. (2011)
Corexit 9500	ANS	N	1:25	*Clupea harengus*	static	embryos	336	reduced hatch	nr	<0.25	Lee et al. (2011)
Corexit 9500	Arabian light	N	1:25	*Clupea harengus*	static	embryos	2.4	reduced hatch	nr	18	Lee et al. (2011)
Corexit 9500	Arabian light	N	1:25	*Clupea harengus*	static	embryos	8	reduced hatch	nr	2.21	Lee et al. (2011)
Corexit 9500	Arabian light	N	1:25	*Clupea harengus*	static	embryos	24	reduced hatch	nr	1.94	Lee et al. (2011)
Corexit 9500	Arabian light	N	1:25	*Clupea harengus*	static	embryos	336	reduced hatch	nr	<0.37	Lee et al. (2011)
Corexit 9500	ANS	N	nr	*Oncorhynchus mykiss*	static daily renewal	embryo	528	severity index	> 0.362	0.663	Wu et al. (2012)
Corexit 9500	Federated	N	nr	*Oncorhynchus mykiss*	static daily renewal	embryo	528	severity index	0.506	0.619	Wu et al. (2012)
Corexit 9500	MESA	N	nr	*Oncorhynchus mykiss*	static daily renewal	embryo	528	severity index	0.826	0.560	Wu et al. (2012)
Corexit 9500	Scotian light	N	nr	*Oncorhynchus mykiss*	static daily renewal	embryo	528	severity index	> 1.744	2.577	Wu et al. (2012)

Dispersant Chemical	Oil Type	Weathered (Y/N)	DOR	Latin Name	Type of Exposure	Life Stage	Duration (h)	Endpoint	Oil EC50 (ppm TPH)	Dispersed Oil EC50 (ppm TPH)	Source(s)
Corexit 9527	PBCO	N	1:10	*Haliotis refescens*	spike-flow through	adult	48	abnormal larval growth	> 33.58 to > 46.99	17.81 to 32.7	Singer et al. (1998)
Corexit 9527	PBCO	N	1:10	*Atherinops affinis*	spike-flow through	adult	96	initial narcosis	16.34 to 40.2	> 62.22 to > 140.97	Singer et al. (1998)
Corexit 9527	PBCO	N	1:10	*Holmesimysis costata*	spiked-flow through	adult	96	initial narcosis	11.1 to 15.9	111.07 to 48.03	Singer et al. (1998)
Corexit 9527	BSC	N	1:29	*Hydra viridissima*	static renewal	adult	168	population growth rate	> 0.6	0.6	Mitchell and Holdway (2000)
Norchem OSD-570	Diesel oil	N	1:10	*Balanus amphitrite*	static	larvae	24	phototaxis inhibition	nr	400	Wu et al. (1997)
Norchem OSD-570	Diesel oil	N	1:10	*Balanus amphitrite*	static	larvae	48	phototaxis inhibition	nr	80	Wu et al. (1997)
Omniclean	No. 2 fuel oil	N	1:1 – 1:10	*Cyprinodon variegatus*	static	< 24h fry	168	early life stage biomass production	nr	25	Singer et al. (1998)
Vecom B-1425	Diesel oil	N	1:10	*Balanus amphitrite*	static	larvae	24	phototaxis inhibition	nr	400	Wu et al. (1997)
Vecom B-1426	Diesel oil	N	1:10	*Balanus amphitrite*	static	larvae	48	phototaxis inhibition	nr	60	Wu et al. (1997)

Primary sources: NRC (2005) and George-Ares and Clark (2000)

ANS – Alaska North Slope crude oil
BSC – Bass Strait crude oil
BSD – blue sac disease
DOR – dispersant to oil ratio
EC50 – concentration that causes a non-lethal effect in 50% of an exposed population
EROD – ethoxyresorufin-O-deethylase
MESA – medium South American crude oil
NPL – National Priorities List
nr – not reported
NRC – National Research Council
PBCO – Prudhoe Bay crude oil
ppm – parts per million
TPH – total petroleum hydrocarbons

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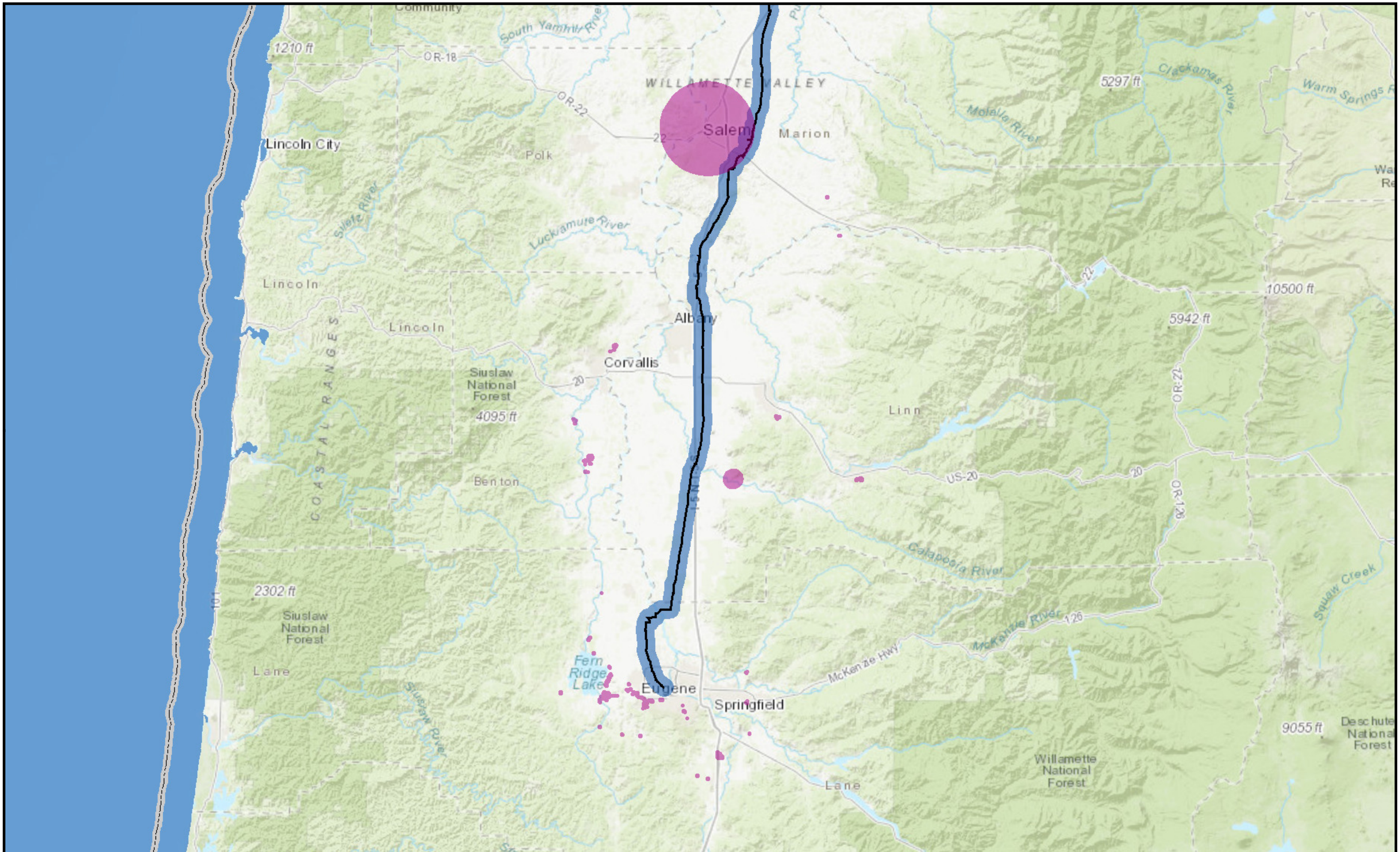
APPENDIX C. MAPS TO SUPPORT EVALUATION OF DISCOUNTABLE EFFECTS



Bradshaw's Desert-Parsley
Petroleum Pipeline
Action Area
State

Data Sources:
ORBIC 2017;
EPA2018;
ESRI 2014

Figure C-1
Bradshaw's Desert-Parsley Observations
Oregon

0 5 10 20
Miles

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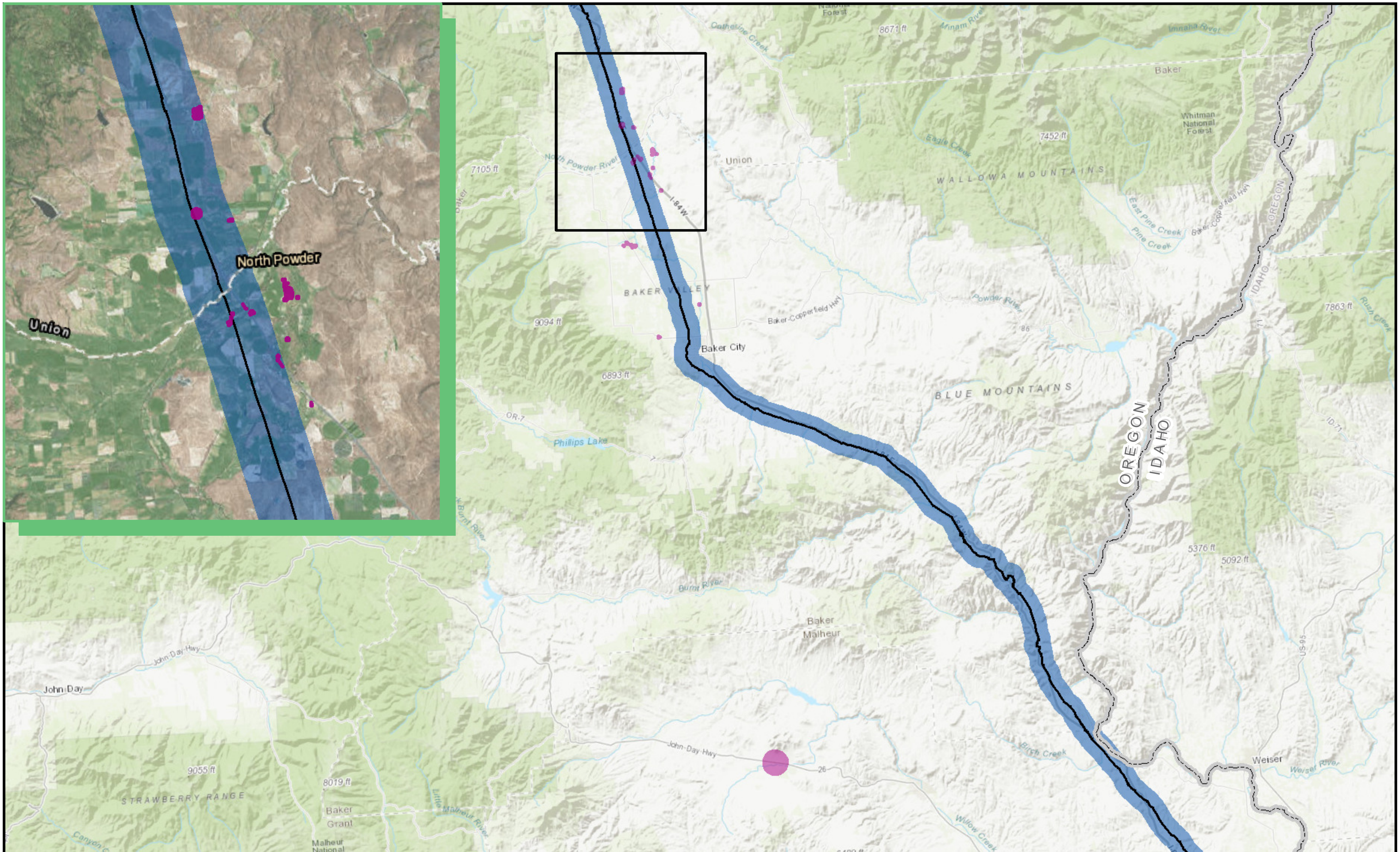

Howell's spectacular thelypody

Petroleum Pipeline

Action Area

State

N W E S

Data Sources:
ORBIC 2017;
EPA2018;
ESRI 2014

Figure C-2
Howell's Spectacular Thelypody Observations
Oregon

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Miles

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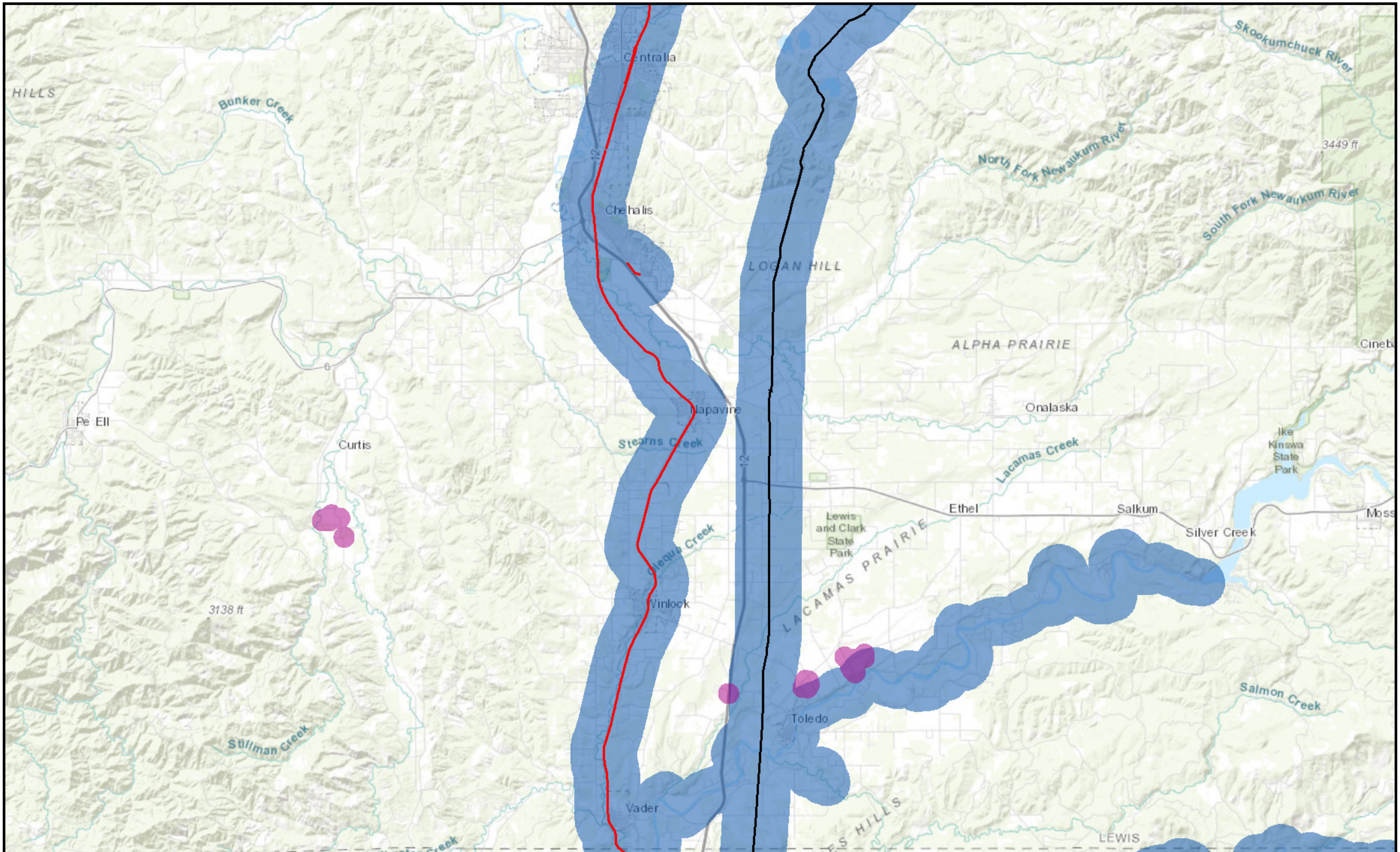


Kincaid's lupine

Petroleum Pipeline

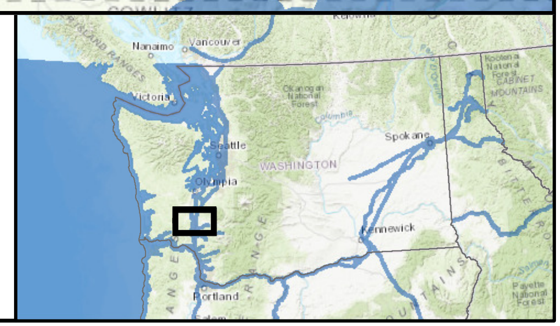
Action Area

State

Data Sources:
WDNR 2017;
ESRI 2014

Figure C-3
Kincaid's Lupine Observations Washington

0 2 4 8
Miles



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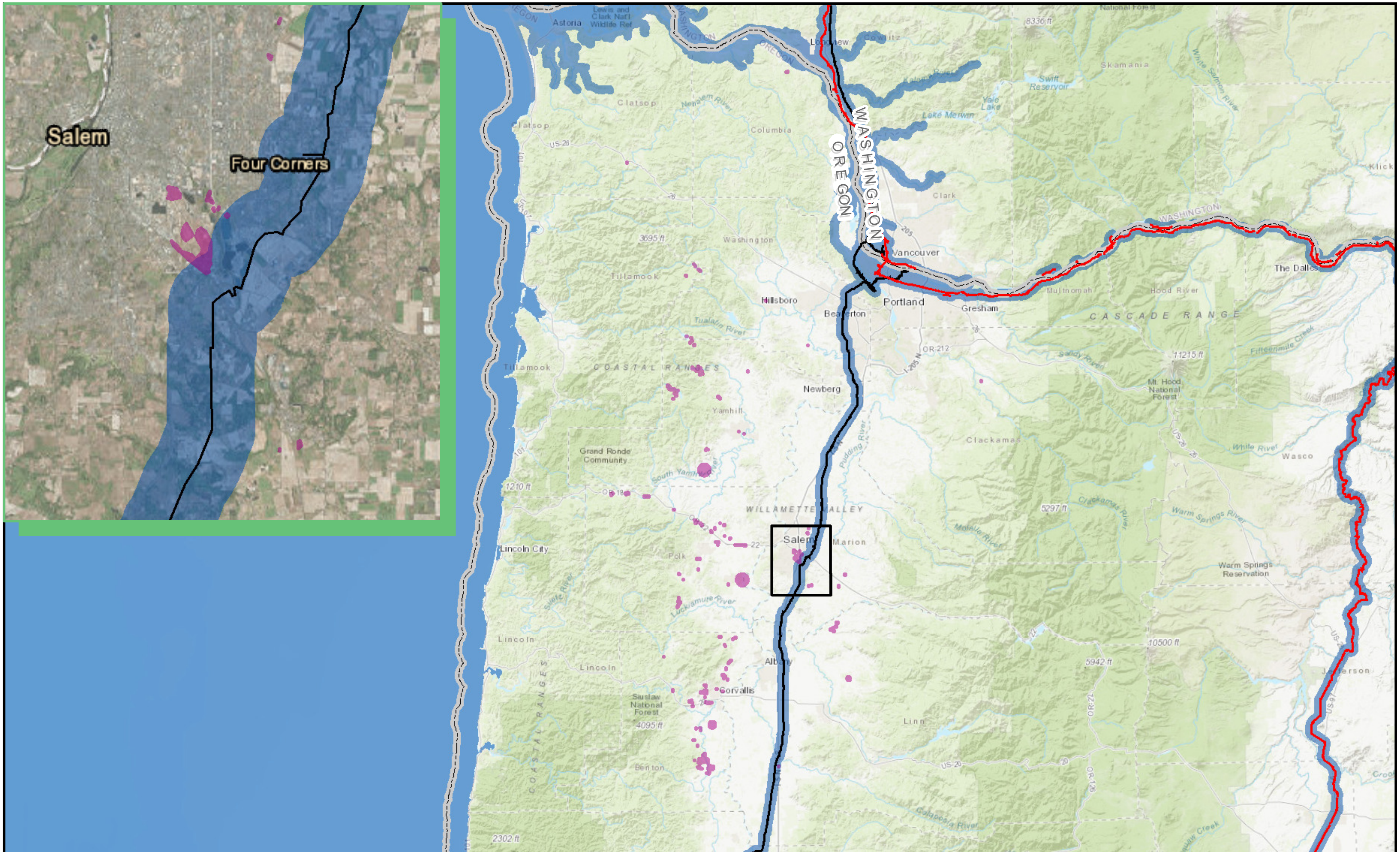

Nelson's checkermallow

Petroleum Pipeline

Action Area

State

Data Sources:
ORBIC 2017;
EPA2018;
ESRI 2014

Figure C-4
Nelson's Checkermallow Observations
Oregon

0 10 20 40
Miles

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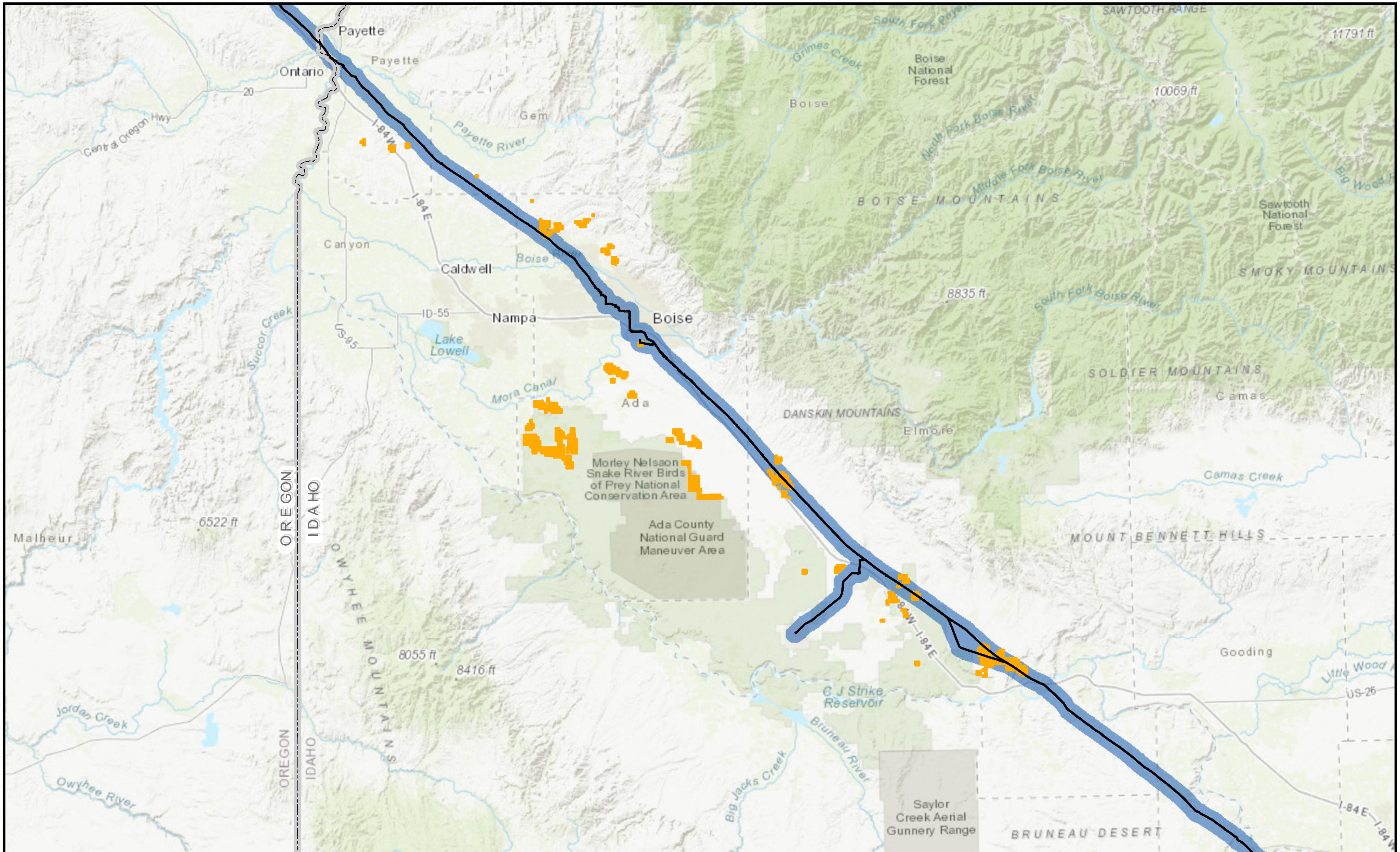

N
W
E
S

Petroleum Pipeline
Slickspot Peppergrass Critical Habitat
Action Area
State

Data Sources:
USFWS 2014;
EPA 2017; ESRI 2014

Figure C-5
Slickspot Peppergrass
Critical Habitat

0 5 10 20
Miles

M:\Boulder\EPA_USCG\USFWS_SlickspotPeppergrass.mxd 5/7/2018

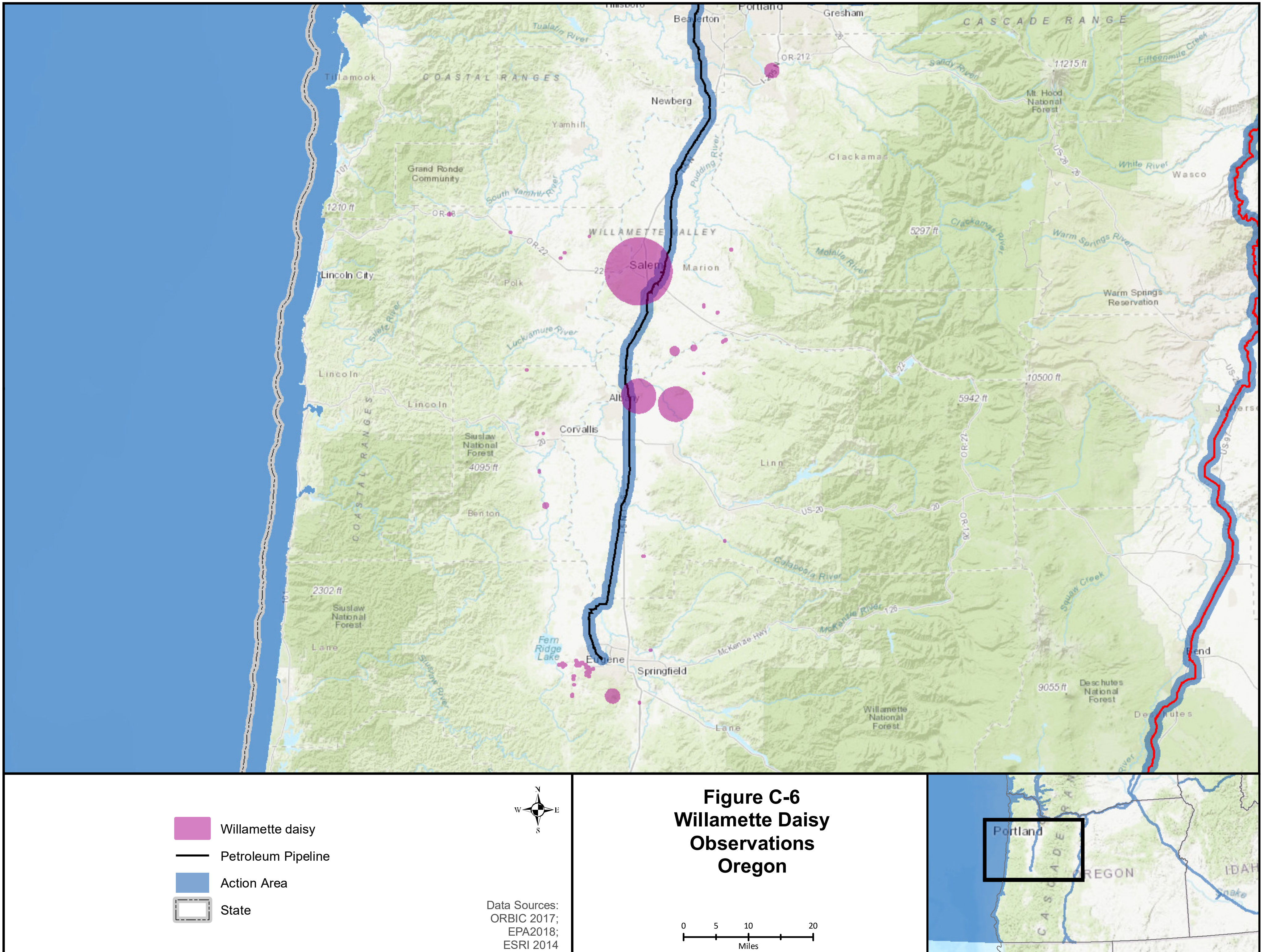

Figure C-6
Willamette Daisy Observations Oregon

Data Sources:
ORBIC 2017;
EPA2018;
ESRI 2014

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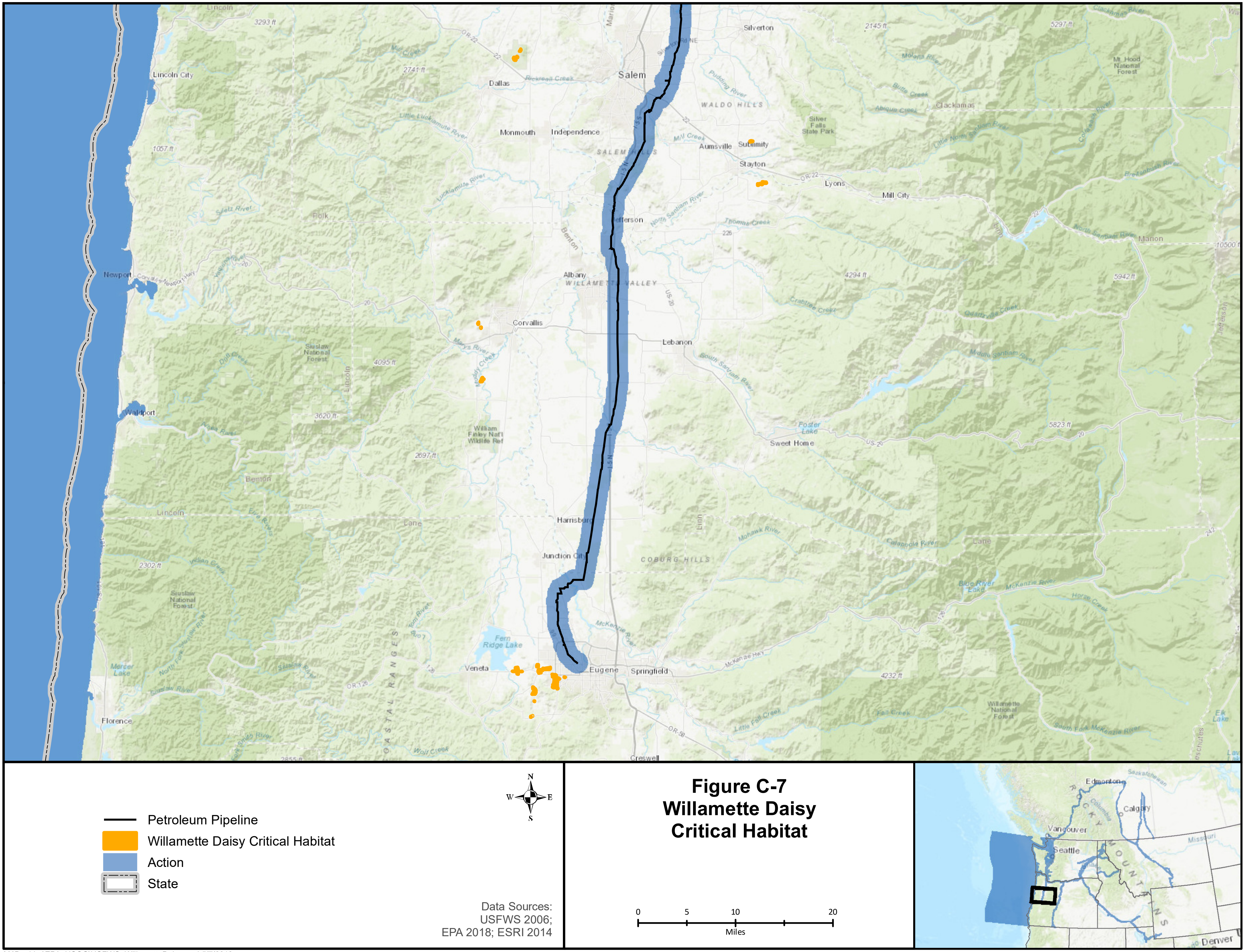

Figure C-7
Willamette Daisy Critical Habitat

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APPENDIX D. PLANT OBSERVATIONS

Applegate's milkvetch (*Astragalus applegatei*)

Feature/ Object ID	Estimated reporting Accuracy	Last Observed	PRES_CODE	Element occurrence Rank	Notes
8	High	2013	Extant	Verified extant (viability not assessed)	Small population on eastern edge of buffer
12	Low	1937	Historical	Historical	
79	High	1994	Historical	Historical	
83	Medium	1983	Historical	Historical	Historical popluation on Northern Pacific Santa Fe rail line rail yard?
151	Very High	2007		Poor estimated viability	Near Hwy 97 with access for response
173	High	1992		Poor estimated viability	Near Hwy 97 with access for response
181	Very High	2008		Excellent estimated viability	Population 0.6 miles from rail line in urban area with access for response
238	Very High	2008		Fair or poor estimated viability	0.6 miles from rail line, Hwy 97 provides access for response
306	Very High	2008		Good estimated viability	
307	Very High	2007		Failed to find	0.6 miles from rail line, Hwy 97 provides access for response
321	Very High	2010		Excellent estimated viability	
322	Very High	2008		Good or fair estimated viability	Found on grounds of Klamath Regional Airport
323	Very High	2008		Poor estimated viability	
458	Very High	2010		Fair estimated viability	Located on west side of Klamath River 0.4 miles from Union Pacific
564	High	2008		Fair or poor estimated viability	Found on grounds of Klamath Regional Airport
803	High	2008		Failed to find	

Data from ORBIC database

Highlighting indicates observations that overlap with the Action Area.

Washington: Not in the Action Area

Idaho: Not in the Action Area

Golden Paintbrush (*Castilleja levisecta*)

Feature/ Object ID	State	Estimated reporting Accuracy	Last Observed	Modified Date	Creation Date	Occurrence/Presence	Precision	Notes
49	Oregon	Low	5/10/1922	10/5/2016	10/12/1992	Extirpated	ND	
129	Oregon	Very Low	5/6/1916	10/5/2016	1/16/1985	Extirpated	ND	Overlaps with pipeline in densely populated urban area of Salem
186	Oregon	Medium	5/14/1938	10/5/2016	1/16/1985	Extirpated	ND	
357	Oregon	Very High	6/27/1905	12/11/2016	3/8/2006	Extirpated	ND	
358	Oregon	Very High	6/27/1905	12/8/2016	3/8/2006	Extirpated	ND	
359	Oregon	Very High	6/27/1905	12/8/2016	3/8/2006	Extirpated	ND	
360	Oregon	Very High	6/27/1905	12/8/2016	3/9/2006	Extirpated	ND	
47	Washington	Medium	2008	ND	ND	Excellent	Location is precise	Direct overlap with rail line in rural area
408	Washington	Medium	2008	ND	ND	Good	Location is precise	
2015	Washington	High	2008	ND	ND	Fair/Poor	Location is precise	

Data from ORBIC database and WDNR Natural Heritage database.

Note: this species is being reintroduced into Washington and is present in Thurston and Island Counties.

Highlighting indicates observations that overlap with the Action Area.

Presence in Idaho is unknown

Spaldings Catchfly (*Silene spaldingii*)						
Feature/ Object ID	Precision Code	Estimated reporting Accuracy	Last Observed	PRES_CODE	Element occurrence Rank	Notes
571	Location is precise	High	2013	Extant	Verified extant	
1790	Location is precise	Medium	2013	Extant	Verified extant	
2704	Location is precise	High	2013	Extant	Poor	
1364	Location is precise	High	2013	Extant	Fair/Poor	
2859	Location is precise	High	2013	Extant	Fair	
4075	Location is precise	High	2012	Extant	Excellent/Good	
3318	Location is precise	High	2011	Extant	Good	
917	Location is precise	High	2010	Extant	Verified extant	
2133	Location is precise	High	2010	Extant	Verified extant	
2297	Location is precise	Medium	2010	Extant	Verified extant	
2990	Location is precise	High	2010	Extant	Verified extant	
4596	Location is precise	High	2010	Extant	Verified extant	
4630	Location is precise	High	2010	Extant	Verified extant	On the edge and witin the 1 mile buffer of the rail line south of the town of Sprague. Town is between the rail line and the plants. If a spill were to occur in this location the staging area would be established within a developed area associated with the town of Sprague. These plants would not be affected.
4749	Location is precise	Medium	2010	Extant	Verified extant	
5070	Location is precise	High	2010	Extant	Verified extant	
2097	Location is precise	Medium	2010	Extant	Good	
3645	Location is precise	High	2010	Extant	Excellent	Many polygons (19) of plants locations are withn this buffer to the rail line. The closest plants are 0.25 miles from the rail line. There are roads inluding the Sprague Highway Rd NE, Lake Valley Loop Rd., Fish Trap Rd, Jack Brown Rd. N. between the plants and the rail ine to allow for movement of spill reponse vehicles . All of the plants within this buffer are within 10 miles of the town of Sprague. If a spill were to occur in this area a staging area would be expected to be established in Sprague where the supplies and infrastructure would be availabe to responders.
2166	Location is precise	High	2009	Extant	Verified extant	
1445	Location is precise	High	2008	Extant	Good	
2135	Location is precise	High	2007	Extant	Verified extant	
2569	Location is precise	High	2007	Extant	Verified extant	
3716	Location is precise	High	2007	Extant	Verified extant	
1970	Location is precise	High	2005	Extant	Verified extant	
5353	Location is precise	High	2004	Extant	Verified extant	
4338	Location is precise	High	2002	Extant	Verified extant	
1765	Location is precise	Medium	2002	Extant	Good/Fair	
55	**Location is precise**	**High**	**2002**	**Extant**	**Fair**	These plants are located directly on the pipeline. If a spill were to occur at this location these plants would be affected. South Ladd Rd runs through the plant polygon and so it would be possible to access the spill but it's likely that responders would need to go off road in some locations if the spill were extensive.

Spaldings Catchfly (*Silene spaldingii*)

Feature/ Object ID	Precision Code	Estimated reporting Accuracy	Last Observed	PRES_CODE	Element occurrence Rank	Notes
821	Location is precise	High	2001	Extant	Verified extant	
954	Location is precise	High	2001	Extant	Verified extant	
3496	Location is precise	High	2001	Extant	Verified extant	
3871	Location is precise	High	2001	Extant	Verified extant	
9	Location is precise	Medium	2000	Extant	Good/Fair	
3187	Location is precise	High	2000	Extant	Failed to Find	
545	Location is precise	High	1996	Extant	Verified extant	
3447	Location is precise	Medium	1995	Extant	Verified extant	
3501	Location is precise	High	1995	Extant	Verified extant	
486	Location is precise	High	1995	Extant	Poor	These plants are located at the edge of 1 mile buffer buffer of the pipeline adjacent to W. Thorpe Road. The Fairchild Airforce Base is close by and could provide the infrastructure for a response action. These plants are 0.75 miles from the pipleine and it's unlikely that they would be affected by a spill response. Since plants have been identified in this location suitable habitat exists, however if this habitat were occupied by the plants we assume that they would have been identified and included in the polygon. While a response action may affect suitable habitat, to the best of our knowledge it is not occupied by plants at this time. If habitat is affected we don't anticipate that it would be destroyed and that recovery would take place over time to functional habitat.
757	Location is precise	Medium	1995	Extant	Poor	
1764	Location is precise	Medium	1995	Extant	Poor	
1125	Location is precise	Medium	1995	Extant	Good/Fair	
2851	Location is precise	High	1995	Extant	Fair/Poor	On edge of the 1 mile buffer for the pipeline overlapping S. Strangland Rd. These plants are 0.8 mile from the pipeline any response to a pipeline spill wouldn't not affect these plants due to the distance.
1183	Location is precise	High	1993	Extant	Verified extant	On edge of the 1 mile buffer for the railline which parallels Sprague Highway Rd E. These plants are located close to the town of Sprague which would provide the infrastructure for a response action. The plants themselves are too far from the railline to be affected, however unoccupied suitable habitat (assuming it exists) may be affected).
3747	Location is precise	High	1990	Extant	Poor	
4135	Location is precise	High	1982	Extant	Extirpated?	
1446	Location is precise	High	1981	Extant	Failed to Find	
3211	Location is precise	Medium	1981	Extant	Failed to Find	

Data from WDNR Natural heritage Database

Highlighting indicates observations that overlap with the Action Area.

Oregon: Not in the Action Area

Idaho: Not in the Action Area

Bradshaw's lomatium (*Lomatium bradshawii*)

Feature/ Object ID	State	Estimated reporting Accuracy	Last Observed	Modified Date	Element occurrence Rank	Notes
23	Oregon	Very High	6/27/1905	12/8/2016	Possibly fair estimated viability	
34	Oregon	High	5/11/2006	12/8/2016	Good estimated viability	
38	Oregon		5/1/2006	10/5/2016	Excellent estimated viability	
95	Oregon	Unknown	5/3/1916	12/8/2016	Historical	Overlaps with Pipeline in densely populated urban area of Salem and Four Corners
154	Oregon	High	6/15/2006	12/8/2016	Possibly good estimated viability	
177	Oregon	Very High	4/14/2006	12/8/2016	Poor estimated viability	
203	Oregon	Very High	4/26/2004	12/8/2016	Excellent estimated viability	
216	Oregon	Very High	5/15/1991	12/8/2016	Poor estimated viability	
295	Oregon	Low	6/14/1905	12/8/2016	Verified extant (viability not assessed)	
412	Oregon	High	5/5/2006	12/8/2016	Excellent estimated viability	
591	Oregon		6/27/1905	10/5/2016	Poor estimated viability	

Not in Action Area in Washington or Idaho

NLAA - Discountable this species is only present in one location within the Action Area and that observation is historical.

Howell's Spectacular (*Thelypodium howellii)*								
Feature/ Object ID	State	Estimated reporting Accuracy	Last Observed	Modified Date	Creation Date	Occurrence/Presence	Precision	Notes
50	Oregon	Very High	7/2/2009	10/5/2016	9/26/1990	Fair or poor estimated viability	ND	Narrow strip of plant growth to the northeast of highway 84 and the Old Oregon Trail Highway. Approximatley 0.5 miles from pipeline
75	Oregon	Very High	7/2/2009	10/5/2016	2/24/1992	Good estimated viability	ND	
80	Oregon	Very High	7/8/1997	10/5/2016	2/15/1990	Good or fair estimated viability	ND	
87	Oregon	Very High	5/31/1990	10/5/2016	3/31/1991	Possibly extirpated	ND	
103	Oregon	Very High	6/1/1996	10/5/2016	10/12/1992	Possibly excellent estimated viability	ND	Present at the eastern edge of the 1 mile buffer 0.85 miles from the pipeline
117	Oregon	High	7/2/2009	10/5/2016	7/11/1999	Fair estimated viability	ND	Present at the eastern edge of the 1 mile buffer 0.85 miles from the pipeline. Plants boarder Highwy 84 and the Old Oregon Trail Highway.
120	Oregon	High	6/9/2008	10/5/2016	10/13/1992	Excellent estimated viability	ND	
172	Oregon	High	6/18/2005	10/5/2016	3/31/1991	Failed to find	ND	Plants documents along the La Grande-Baker Highway near the pipeline, in a rural area.
187	Oregon	Medium	197--	10/5/2016	1/6/1987	Possibly extirpated	ND	Polygon centered on I 84 adjacent to pipeline
215	Oregon	High	6/19/1995	10/5/2016	2/15/1990	Failed to find	ND	
249	Oregon	Low	8/16/1969	10/5/2016	10/14/1992	Extirpated	ND	
266	Oregon	Very High	7/9/1997	10/5/2016	3/26/1991	Fair or poor estimated viability	ND	Located on the east edge of the buffer adjected to Bagwell Rd. and Olson Rd east of I84, pipeline on the west side of Highway I84 in this location.
380	Oregon	High	7/7/1997	10/5/2016	8/6/1996	Poor estimated viability	ND	
413	Oregon	High	4/19/1989	10/5/2016	5/9/1999	Failed to find	ND	

The plants have been documented in a rural area of easern Oregon growing along and in the vicinity of I 84 the Old Oregon Highway. The most recent observation (hig or very high repodting accuracy) is 2009 with plants having fair to poor estimated viability. The basis for a NLAA would be the

Kincaid's Lupine (*Lupinus oreganus*)

Feature/ Object ID		Estimated reporting Accuracy	Last Observed	Modified Date	Creation Date	Occurrence/Presence	Element Occurrence Ranking	Precision	Notes
5	Oregon	High	6/14/1905	12/8/2016	1/22/1998	Verified extant (viability not assessed)		ND	
7	Oregon	Very High	6/10/2009	12/11/2016	2/19/1992	Good or fair estimated viability		ND	
11	Oregon	Low	5/22/1979	12/8/2016	9/10/1990	Historical		ND	
13	Oregon	High	5/14/1991	12/8/2016	11/18/1992	Fair estimated viability		ND	
25	Oregon	Very High	4/1/1990	12/11/2016	11/19/1992	Possibly good estimated viability		ND	
26	Oregon	Very High	6/1/2007	12/8/2016	3/4/1998	Verified extant (viability not assessed)		ND	
30	Oregon	High	5/22/1993	12/8/2016	2/4/1994	Excellent estimated viability		ND	
31	Oregon	Low	7/19/1916	10/5/2016	9/10/1990	Historical		ND	
46	Oregon	Very High	6/3/2015	12/11/2016	10/19/2000	Good estimated viability		ND	
66	Oregon	High	6/3/1999	12/8/2016	1/26/2000	Poor estimated viability		ND	
67	Oregon	High	6/30/1992	12/8/2016	4/13/1993	Fair estimated viability		ND	
69	Oregon		6/14/1905	10/5/2016	1/22/1998	Verified extant (viability not assessed)		ND	
90	Oregon	High	1991-05	12/8/2016	2/19/1992	Poor estimated viability		ND	
93	Oregon	High	2009-pre	12/8/2016	9/10/1990	Poor estimated viability		ND	
100	Oregon	High	6/13/1905	12/8/2016	1/18/1998	Verified extant (viability not assessed)		ND	
126	Oregon		5/13/2015	10/10/2016	10/20/1992	Fair estimated viability		ND	
130	Oregon	High	6/28/1905	12/8/2016	1/18/1998	Verified extant (viability not assessed)		ND	
133	Oregon	Very High	6/11/2009	12/8/2016	1/22/1998	Verified extant (viability not assessed)		ND	
146	Oregon		5/8/2007	10/5/2016	3/4/1998	Verified extant (viability not assessed)		ND	
157	Oregon	High	6/15/1905	12/8/2016	1/22/1998	Verified extant (viability not assessed)		ND	
160	Oregon		5/11/2009	10/5/2016	10/9/1991	Excellent estimated viability		ND	
162	Oregon	Very High	3/17/2009	12/8/2016	12/12/1990	Verified extant (viability not assessed)		ND	
164	Oregon	High	6/14/1905	12/8/2016	1/22/1998	Verified extant (viability not assessed)		ND	
165	Oregon	Very High	5/28/1992	12/8/2016	1/24/1993	Excellent or good estimated viability		ND	
169	Oregon		4/23/2007	10/5/2016	3/4/1998	Verified extant (viability not assessed)		ND	
170	Oregon	Very High	6/14/1905	12/8/2016	3/4/1998	Fair estimated viability		ND	
182	Oregon	High	6/11/2009	12/8/2016	1/22/1998	Verified extant (viability not assessed)		ND	
183	Oregon	High	7/7/1990	12/8/2016	6/11/1991	Poor estimated viability		ND	
195	Oregon	Very High	5/20/1995	12/8/2016	1/18/1998	Verified extant (viability not assessed)		ND	
197	Oregon	Very High	6/28/1905	12/11/2016	3/9/1995	Verified extant (viability not assessed)		ND	
199	Oregon	Unknown	5/31/1916	12/8/2016	10/24/1989	Historical		ND	The observations are located within Salem, Keizer and Four Corners an urbanized area in Oregon. Due to the amount of development it's likely there very little suitable habitat exists for this species which is probably why observations are historical.
224	Oregon	High	5/27/1979	12/8/2016	2/19/1992	Verified extant (viability not assessed)		ND	
229	Oregon	High	6/14/1905	12/8/2016	1/18/1998	Verified extant (viability not assessed)		ND	
230	Oregon	High	6/14/1905	12/8/2016	1/18/1998	Verified extant (viability not assessed)		ND	
231	Oregon	Very High	1992-	12/11/2016	2/19/1992	Good estimated viability		ND	
233	Oregon	Very High	6/20/2014	12/8/2016	3/1/1993	Good estimated viability		ND	
234	Oregon	Medium	5/21/1992	12/8/2016	7/16/1996	Verified extant (viability not assessed)		ND	
242	Oregon	High	6/28/1905	12/8/2016	2/1/1999	Good estimated viability		ND	
252	Oregon	High	5/27/2015	12/8/2016	2/1/1999	Good or fair estimated viability		ND	
257	Oregon	Medium	5/30/2006	10/5/2016	1/18/1998	Verified extant (viability not assessed)		ND	
262	Oregon	High	6/28/1905	12/8/2016	2/1/1999	Excellent estimated viability		ND	
285	Oregon	High	7/3/1905	12/8/2016	11/19/1991	Excellent estimated viability		ND	
286	Oregon	Very High	6/14/1989	12/8/2016	9/9/1991	Excellent or good estimated viability		ND	
292	Oregon	High	6/14/1905	12/8/2016	1/18/1998	Verified extant (viability not assessed)		ND	
303	Oregon	Medium	6/24/1941	12/8/2016	6/21/2004	Historical		ND	
308	Oregon	High	6/28/1905	12/8/2016	11/4/2007	Possibly excellent estimated viability		ND	
315	Oregon	High	5/30/2006	12/8/2016	4/11/2007	Verified extant (viability not assessed)		ND	
316	Oregon	Medium	6/27/1905	10/5/2016	4/11/2007	Verified extant (viability not assessed)		ND	
317	Oregon		6/8/2006	10/5/2016	4/15/2007	Verified extant (viability not assessed)		ND	
318	Oregon	Medium	6/15/2005	12/8/2016	4/15/2007	Verified extant (viability not assessed)		ND	
319	Oregon	Low	6/8/2005	12/8/2016	4/15/2007	Verified extant (viability not assessed)		ND	
320	Oregon	Very High	6/8/2005	12/11/2016	4/15/2007	Verified extant (viability not assessed)		ND	
337	Oregon	Very High	7/2/2010	12/11/2016	2/4/2007	Excellent estimated viability		ND	
343	Oregon	High	6/26/1905	12/8/2016	1/27/2015	Verified extant (viability not assessed)		ND	
361	Oregon	High	7/21/2000	12/8/2016	5/4/2005	Poor estimated viability		ND	
364	Oregon	High	2006?	12/8/2016	7/27/2009	Verified extant (viability not assessed)		ND	

Kincaid's Lupine (*Lupinus oreganus*)									
Feature/ Object ID		Estimated reporting Accuracy	Last Observed	Modified Date	Creation Date	Occurrence/Presence	Element Occurrence Ranking	Precision	Notes
366	Oregon		6/13/1905	10/5/2016	2/18/1990	Verified extant (viability not assessed)		ND	
367	Oregon	Very High	6/18/2009	12/8/2016	7/2/2008	Possibly fair estimated viability		ND	
368	Oregon	Very High	2009-pre	12/8/2016	8/4/2009	Verified extant (viability not assessed)		ND	
370	Oregon		5/24/2007	10/5/2016	7/27/2009	Verified extant (viability not assessed)		ND	
372	Oregon	Medium	5/29/2006	12/8/2016	1/28/2007	Poor estimated viability		ND	
373	Oregon	High	5/21/2007	12/8/2016	7/27/2009	Verified extant (viability not assessed)		ND	
375	Oregon	High	6/27/1905	12/8/2016	11/5/2007	Possibly excellent estimated viability		ND	
376	Oregon		5/9/2007	10/5/2016	7/27/2009	Verified extant (viability not assessed)		ND	
377	Oregon	High	6/10/1999	12/8/2016	10/6/1999	Poor estimated viability		ND	
378	Oregon	High	6/29/1905	12/8/2016	7/2/2008	Poor estimated viability		ND	
379	Oregon	Very High	2006?	12/8/2016	7/27/2009	Verified extant (viability not assessed)		ND	
382	Oregon		2006?	10/5/2016	7/21/2009	Verified extant (viability not assessed)		ND	
384	Oregon	Very High	6/7/2007	12/11/2016	6/17/2008	Excellent estimated viability		ND	
385	Oregon	Very High	5/21/2007	12/11/2016	7/21/2009	Verified extant (viability not assessed)		ND	
387	Oregon	High	4/7/2010	12/11/2016	3/19/2011	Verified extant (viability not assessed)		ND	
388	Oregon	High	5/14/2007	12/8/2016	7/21/2009	Verified extant (viability not assessed)		ND	
389	Oregon	High	2010-pre	12/8/2016	3/19/2011	Verified extant (viability not assessed)		ND	
391	Oregon		5/21/2007	10/5/2016	7/21/2009	Poor estimated viability		ND	
392	Oregon	Very High	6/8/2005	12/11/2016	12/12/1990	Verified extant (viability not assessed)		ND	
394	Oregon		6/4/2007	10/5/2016	7/21/2009	Verified extant (viability not assessed)		ND	
397	Oregon	High	6/8/2006	12/11/2016	7/21/2009	Verified extant (viability not assessed)		ND	
398	Oregon	High	5/16/2008	12/8/2016	7/21/2009	Verified extant (viability not assessed)		ND	
402	Oregon	Very High	6/18/2009	12/8/2016	7/21/2009	Verified extant (viability not assessed)		ND	
404	Oregon	High	5/1/2007	12/8/2016	7/21/2009	Verified extant (viability not assessed)		ND	
406	Oregon	Very High	5/15/2007	12/8/2016	7/21/2009	Verified extant (viability not assessed)		ND	
407	Oregon	Very High	5/14/2007	12/8/2016	7/22/2009	Verified extant (viability not assessed)		ND	
409	Oregon		5/16/2007	10/5/2016	7/22/2009	Verified extant (viability not assessed)		ND	
411	Oregon	Very High	5/1/2007	12/8/2016	7/22/2009	Verified extant (viability not assessed)		ND	
414	Oregon	High	7/4/2007	12/8/2016	7/22/2009	Verified extant (viability not assessed)		ND	
415	Oregon	Very High	5/25/2007	12/8/2016	7/22/2009	Verified extant (viability not assessed)		ND	
417	Oregon	High	7/2/2007	12/8/2016	7/22/2009	Verified extant (viability not assessed)		ND	
448	Oregon	Very High	7/9/2010	12/8/2016	10/2/2011	Verified extant (viability not assessed)		ND	
451	Oregon	Medium	7/2/1905	12/8/2016	10/12/2011	Verified extant (viability not assessed)		ND	
453	Oregon	Low	5/23/1966	12/8/2016	10/12/2011	Historical		ND	
455	Oregon	Low		12/8/2016	10/13/2011			ND	
460	Oregon	Medium	6/1/2011	10/5/2016	6/2/2011	Excellent or good estimated viability		ND	
463	Oregon		6/21/2010	10/5/2016	7/21/2009	Verified extant (viability not assessed)		ND	
465	Oregon	Very High	6/3/2011	12/11/2016	6/5/2011	Excellent estimated viability		ND	
467	Oregon	High	6/10/2009	12/8/2016	6/7/2011	Verified extant (viability not assessed)		ND	
471	Oregon	Very High	6/2/2006	12/11/2016	10/4/1992	Excellent estimated viability		ND	
473	Oregon	Very High	6/2/2008	12/8/2016	3/26/1991	Verified extant (viability not assessed)		ND	
476	Oregon	Very High	2011-06	12/8/2016	8/4/2011	Verified extant (viability not assessed)		ND	
495	Oregon	Very High	1991-06	12/11/2016	2/19/1992	Excellent estimated viability		ND	
528	Oregon	High	6/11/1990	12/8/2016	6/12/1991	Excellent or good estimated viability		ND	
551	Oregon	High	6/27/1905	12/8/2016	3/1/2007	Good estimated viability		ND	
552	Oregon	Low	6/15/2006	12/8/2016	4/16/2007	Poor estimated viability		ND	
630	Oregon	Very High	6/17/2003	10/5/2016	2/13/2005	Excellent estimated viability		ND	
690	Oregon	High	5/26/2015	12/8/2016	7/10/1996	Fair estimated viability		ND	
696	Oregon	High	6/16/2015	12/8/2016	11/16/2016			ND	
706	Oregon	Medium	6/29/1905	12/8/2016	10/12/2011	Verified extant (viability not assessed)		ND	
729	Oregon	Very High	2011-06	12/8/2016	8/4/2011	Verified extant (viability not assessed)		ND	
733	Oregon	High	6/12/2004	12/8/2016	11/22/2004	Fair estimated viability		ND	
735	Oregon	Very High	5/18/2015	12/8/2016	1/23/2015	Verified extant (viability not assessed)		ND	
747	Oregon	Very High	5/30/2007	12/11/2016	6/16/2008	Good or fair estimated viability		ND	
771	Oregon	Very High	5/12/2015	12/8/2016	2/19/1992	Possibly excellent estimated viability		ND	
773	Oregon	Very High		12/8/2016	7/27/2009			ND	
775	Oregon	Very High	5/7/2007	12/8/2016	6/16/2008	Verified extant (viability not assessed)		ND	
782	Oregon	Very High	6/18/2009	12/8/2016	7/2/2008	Excellent or good estimated viability		ND	
787	Oregon	Very High	6/29/1905	12/8/2016	7/2/2008	Possibly fair estimated viability		ND	

Kincaid's Lupine (*Lupinus oreganus*)

Feature/ Object ID		Estimated reporting Accuracy	Last Observed	Modified Date	Creation Date	Occurrence/Presence	Element Occurrence Ranking	Precision	Notes
3076	Washington	High	7/2/1905	ND	ND	Extant	Good	Location is Precise	
4068	Washington	High	2012	ND	ND	Extant	verified extant	Location is Precise	Located in the Cowlitz Prairie within the 1 mile buffer of the Cowlitz River
4765	Washington	High	2011	ND	ND	Extant	Fair	Location is Precise	Located on the western edge of the 1 mile buffer for the pipeline. Ploygon centers over I5 and Knowles Road in Drewes Prairie.
4954	Washington	High	2011	ND	ND	Extant	Fair	Location is Precise	The location of these plants overlaps with numerous roads with locations for staging areas closer to the pipeline and the Cowlitz River.

Although not in the database, according to USFWS (Gabrielle Robinson) potential habitat for this species is found in the action area. Occurance of this plant are located within the action area in Thurston County.
This species isn't present in Idaho.

NLAA it's unlikely that if a spill response took place for a pipeline spill that the plants would be affected due to the distance from the pipeline.

Nelson's sidalcea (*Sidalcea nelsoniana*)

Feature/ Object ID		Estimated reporting Accuracy	Last Observed	Modified Date	Creation Date	Occurrence/Presence	Element Occurrence Ranking	Precision	Notes
1	Oregon	Low	5/29/1924	12/8/16	9/30/1991	Historical	ND	ND	
2	Oregon	High	7/12/2001	12/8/16	11/4/2001	Poor estimated viability	ND	ND	
3	Oregon	High	1995	12/8/16	10/2/1991	Fair estimated viability	ND	ND	
4	Oregon	High	1995	12/8/16	10/2/1991	Good estimated viability	ND	ND	
18	Oregon	High	7/14/1993	12/8/16	9/30/1991	Poor estimated viability	ND	ND	
27	Oregon	Very High	2005	12/8/16	10/2/1991	Poor estimated viability	ND	ND	
29	Oregon	Very High	2005	12/8/16	7/22/2002	Poor estimated viability	ND	ND	
33	Oregon		6/15/2007	10/5/16	3/21/1991	Excellent estimated viability	ND	ND	
36	Oregon		7/11/2006	10/5/16	10/19/2000	Poor estimated viability	ND	ND	
42	Oregon		2011-summ	10/5/16	12/6/1993	Excellent or good estimated viability	ND	ND	
44	Oregon	High	2009	12/8/16	12/6/1993	Possibly historical	ND	ND	
48	Oregon	Medium	6/7/1947	12/8/16	2/7/1991	Possibly historical	ND	ND	
51	Oregon	High	1995	12/8/16	12/11/1994	Poor estimated viability	ND	ND	
56	Oregon	High	7/8/1985	12/8/16	2/26/1992	Historical	ND	ND	
57	Oregon	High	1999-07	12/8/16	9/12/1999	Possibly historical	ND	ND	
71	Oregon	Very High	1995	12/8/16	10/6/1991	Fair estimated viability	ND	ND	
77	Oregon	Medium	1995	12/8/16	9/30/1991	Verified extant (viability not assessed)	ND	ND	
96	Oregon	Low	6/22/1918	12/8/16	9/30/1991	Possibly extirpated	ND	ND	
98	Oregon	High	1995	12/8/16	9/26/1990	Fair estimated viability	ND	ND	
99	Oregon	High	6/6/2007	12/8/16	10/6/1991	Fair estimated viability	ND	ND	
101	Oregon	High	7/3/2001	12/8/16	7/29/2001	Fair or poor estimated viability	ND	ND	
102	Oregon	Very High	2005	12/11/16	10/25/2001	Excellent estimated viability	ND	ND	
104	Oregon	High	1995	12/8/16	10/22/1992	Poor estimated viability	ND	ND	
106	Oregon	Very High	6/29/2009	12/11/16	10/22/1992	Possibly good estimated viability	ND	ND	
112	Oregon	Very High	1995	12/11/16	10/2/1991	Excellent estimated viability	ND	ND	
115	Oregon	High	6/10/1991	12/8/16	2/19/1992	Poor estimated viability	ND	ND	
116	Oregon	Very High	2005	12/8/16	10/2/1991	Possibly poor estimated viability	ND	ND	
118	Oregon	High	6/6/2007	12/8/16	8/3/1993	Fair estimated viability	ND	ND	
125	Oregon	High	6/30/1993	12/8/16	12/6/1993	Fair or poor estimated viability	ND	ND	
127	Oregon	High	1995	12/8/16	10/28/1992	Poor estimated viability	ND	ND	
131	Oregon	Very High	2009	12/8/16	10/6/1991	Poor estimated viability	ND	ND	
132	Oregon	High	2009	12/8/16	10/9/1991	Poor estimated viability	ND	ND	
137	Oregon	High	1995	12/8/16	10/1/1991	Good or fair estimated viability	ND	ND	
138	Oregon	High	1995	12/8/16	10/1/1991	Possibly good estimated viability	ND	ND	
145	Oregon	High	7/3/2001	12/8/16	9/30/1991	Possibly historical	ND	ND	
149	Oregon	Very High	2005	12/11/16	2/7/1991	Poor estimated viability	ND	ND	
155	Oregon	High	2005	12/8/16	3/25/1991	Fair or poor estimated viability	ND	ND	
156	Oregon	High	1995	12/8/16	10/6/1991	Poor estimated viability	ND	ND	
159	Oregon	Very High	1997-pre	12/8/16	2/15/1998	Verified extant (viability not assessed)	ND	ND	
163	Oregon	High	6/18/2009	12/8/16	7/22/2002	Poor estimated viability	ND	ND	
178	Oregon	High	1995	12/8/16	12/11/1994	Good estimated viability	ND	ND	
179	Oregon		6/14/2007	10/5/16	1/29/1995	Good estimated viability	ND	ND	
180	Oregon	Low	7/2/1985	10/5/16	9/30/1991	Possibly extirpated	ND	ND	
185	Oregon	Very High	7/27/2006	12/11/16	9/30/1991	Excellent estimated viability	ND	ND	These plants are located north west of I5 which is between the pipeline and the plants.
190	Oregon	High	1995	12/8/16	12/6/1993	Poor estimated viability	ND	ND	
193	Oregon	High	6/21/2007	12/8/16	9/30/1991	Poor estimated viability	ND	ND	
194	Oregon	High	1995	12/8/16	2/26/1992	Possibly historical	ND	ND	
200	Oregon		2009	10/5/16	6/1/94	Fair or poor estimated viability	ND	ND	
202	Oregon	High	7/24/1997	12/8/16	7/5/98	Verified extant (viability not assessed)	ND	ND	
205	Oregon		7/11/2006	10/5/16	3/25/91	Possibly good estimated viability	ND	ND	
207	Oregon	Very High	1994	12/8/16	10/22/92	Possibly fair estimated viability	ND	ND	
208	Oregon	High	2009	12/8/16	10/6/91	Poor estimated viability	ND	ND	
217	Oregon	High	1995	12/8/16	10/1/91	Possibly historical	ND	ND	
219	Oregon	Very High	2005	12/8/16	10/11/92	Good estimated viability	ND	ND	
220	Oregon	Very High	6/25/1999	12/11/16	6/11/92	Poor estimated viability	ND	ND	
226	Oregon	Very High	2005	12/11/16	10/2/91	Possibly poor estimated viability	ND	ND	
228	Oregon	Very High	7/3/1989	12/8/16	10/2/91	Historical	ND	ND	
235	Oregon	High	7/5/1994	12/8/16	10/9/91	Poor estimated viability	ND	ND	
236	Oregon	High	1995	12/8/16	6/11/91	Excellent estimated viability	ND	ND	
240	Oregon	High	1995	12/8/16	10/9/91	Fair estimated viability	ND	ND	
254	Oregon	Very High	7/9/2007	12/11/16	10/1/91	Excellent estimated viability	ND	ND	
263	Oregon	High	1995	12/8/16	3/5/91	Extirpated	ND	ND	
265	Oregon	Medium	7/12/1996	12/8/16	3/25/98	Poor estimated viability	ND	ND	

Nelson's sidalcea (*Sidalcea nelsoniana*)

Feature/ Object ID		Estimated reporting Accuracy	Last Observed	Modified Date	Creation Date	Occurrence/Presence	Element Occurrence Ranking	Precision	Notes
273	Oregon	High	6/24/2004	12/8/16	9/30/91	Poor estimated viability	ND	ND	
274	Oregon	Very High	2005	12/8/16	10/6/91	Poor estimated viability	ND	ND	
278	Oregon	High	2005	12/8/16	7/13/03	Poor estimated viability	ND	ND	
280	Oregon	Very High	2005	12/8/16	3/26/91	Fair estimated viability	ND	ND	
288	Oregon	Very High	2005	12/8/16	3/21/91	Good estimated viability	ND	ND	
289	Oregon	High	1997-pre	12/8/16	2/15/98	Verified extant (viability not assessed)	ND	ND	
290	Oregon	High	1995	12/8/16	9/30/91	Poor estimated viability	ND	ND	
293	Oregon		2005	10/5/16	10/2/91	Poor estimated viability	ND	ND	
298	Oregon	Very High	2004-05	12/11/16	5/12/04	Excellent estimated viability	ND	ND	
300	Oregon	High	7/2/2004	12/8/16	11/22/04	Poor estimated viability	ND	ND	These plants are located adjacent to along Courtney Creek which flows under Pacific Hwy, which is next to I5. The pipeline is located within a large cultivated field 0.5 miles west of the plants. If a spill were to occur in this location these plants would not be affected due to their location. A staging area for spill response would not encounter these plants which are located in a riparian area adjacent to the highway, but would be established elsewhere.
301	Oregon	High	6/12/2004	12/11/16	11/22/04	Poor estimated viability	ND	ND	
329	Oregon	Very High	7/11/2006	12/8/16	7/17/06	Good estimated viability	ND	ND	
333	Oregon	Very High	7/18/2006	12/8/16	11/8/06	Fair estimated viability	ND	ND	
334	Oregon	High	7/18/2006	12/11/16	11/8/06	Poor estimated viability	ND	ND	
344	Oregon	High	2005	12/8/16	10/6/91	Poor estimated viability	ND	ND	
345	Oregon	Very High	2005	12/8/16	12/20/05	Poor estimated viability	ND	ND	
346	Oregon	Very High	2005	12/8/16	12/20/05	Good or fair estimated viability	ND	ND	
347	Oregon	Very High	2005	12/8/16	12/20/05	Good or fair estimated viability	ND	ND	
348	Oregon	High	2005	12/8/16	12/21/05	Poor estimated viability	ND	ND	
349	Oregon	High	2005	12/8/16	12/21/05	Fair estimated viability	ND	ND	
350	Oregon	Very High	2005	12/8/16	12/21/05	Poor estimated viability	ND	ND	
356	Oregon	High	6/16/2004	12/8/16	3/20/05	Poor estimated viability	ND	ND	
371	Oregon	High	1995	12/8/16	12/6/93	Possibly historical	ND	ND	
436	Oregon	High	1989	12/8/16	10/6/91	Historical	ND	ND	
461	Oregon	Very High	5/27/2011	10/5/16	6/2/11	Good estimated viability	ND	ND	
475	Oregon		6/4/2004	10/5/16	6/27/04	Verified extant (viability not assessed)	ND	ND	
511	Oregon	High	1995	12/8/16	9/30/91	Extirpated	ND	ND	
533	Oregon	Very High	6/25/2007	12/11/16	9/30/91	Poor estimated viability	ND	ND	
577	Oregon		7/11/2006	10/5/16	10/6/91	Good or fair estimated viability	ND	ND	
662	Oregon	Very High	2005	12/8/16	12/14/05	Poor estimated viability	ND	ND	
664	Oregon	Very High	2005	12/8/16	12/20/05	Poor estimated viability	ND	ND	
665	Oregon	Very High	6/25/2004	12/11/16	3/20/05	Good or fair estimated viability	ND	ND	
715	Oregon	High	6/24/2009	12/8/16	4/28/11	Verified extant (viability not assessed)	ND	ND	
724	Oregon	Very High	2005	12/8/16	7/5/98	Excellent estimated viability	ND	ND	
736	Oregon	High	6/5/2007	12/8/16	11/28/04	Poor estimated viability	ND	ND	
761	Oregon	High	5/7/2004	12/8/16	6/27/04	Poor estimated viability	ND	ND	
762	Oregon	High	6/22/2004	12/8/16	6/27/04	Good or fair estimated viability	ND	ND	
535	Washington	High	2014			Extant	Fair	ND	
2493	Washington	Medium	2014			Extant	Fair/Poor	ND	

These plants are not in Idaho

Based on the observation data for Washington (WDNR NHP) and Oregon (ORBICS) this species is present in the action area in Oregon. The ORBICS database contains approximately 100 observation of this specis in the state two of these are within the action area. According to the USFWS ECOS database the range of the species includes the action area.

Slickspot Peppergrass							
Feature/ Object ID	Estimated reporting Accuracy	Last Observed	Modified Date	Creation Date	Occurrence/Presence	Precision	Notes

E&E has the data for this. This species is in Idaho Only
NLAA No water in the vicintiy
Check the Appendix C map

Ute ladies-tresses - NE							
Feature/ Object ID	Estimated reporting Accuracy	Last Observed	Modified Date	Creation Date	Occurrence/ Presence	Precision	Notes

Not in Action Area

Water Howellia (*Howellia aquatilis*)									
Feature/ Object ID	Estimated reporting Accuracy	Last Observed	State	Modified Date	Creation Date	Occurrence/ Presence	Occurance Rank	Precision	Notes
97	High	2007	Washington	ND	ND	Extant	Good	Location is precise	
157	High	2009	Washington	ND	ND	Extant	Good/Fair	Location is precise	Located on the south east edge of the 1 mile buffer centered on the S. Cheney Plaza Rd. These plants are likely growing around Long Lake and other pond/wetland habitats. This is a rural area. The rail line is approximately 0.5 miles from the outer extent of the polygon. The rail line is adjacent to the town of Cheney and any staging areas would be set up in this urban location and would not affect the plants that are located 0.5 miles from this location.
261	Medium	1999	Washington	ND	ND	Extant	Good	Location is precise	
273	High	2011	Washington	ND	ND	Extant	Fair	Location is precise	The plants are located in an undeveloped area (Dishman Hills) in Spokane. The pipeline runs through the town of Spokane which provides numerous locations for a staging area. A pipeline spill will not affect plants growing outside of town.
303	High	1987	Washington	ND	ND	Extant	Fair	Location is precise	These plants are growing in a area between the S. Cheney-Spokane Rd and the Columbia Plateau Trail. There are two rail line is thin location and one of the two rail lines runs through the polygon of the plants. A spill response in this area may affect plants. Plants grow in small, ephemeral wetlands found within the forested portions of the channeled scablands in eastern Washington. The plants in this polygon have not been observed since 1987 however, the basemap shows what appear to be wet areas so we are assuming that suitable habitat remains in this area.
496	Medium	1993	Washington	ND	ND	Extant	Poor	Location is precise	
631	Medium	2008	Washington	ND	ND	Extant	Fair	Location is precise	
638	Medium	2009	Washington	ND	ND	Extant	Good/Fair	Location is precise	
639	High	2007	Washington	ND	ND	Extant	Good	Location is precise	
778	Medium	1994	Washington	ND	ND	Extant	Good	Location is precise	
812	High	1998	Washington	ND	ND	Extant	Fair	Location is precise	These plants are growing associated with Chambers Lake and Shaver Lake in an area adjacent to Highway 507. There are unnamed roads in this wooded area assumably providing access to the lakes. The pipeline runs along the east side of highway 507. Highway 507 is located between the plants and the pipeline
816	Medium	2007	Washington	ND	ND	Extant	Good	Location is precise	
824	Medium	1987	Washington	ND	ND	Extant	Good/Fair	Location is precise	
852	Low	1997	Washington	ND	ND	Extant	Verified extant	Location w/in 1 mile radius	This is a large polygon which includes FIDs 4817 and 2535. These plant are growing within the 1 mile buffer of a rail line. The polygons closest to the rail line contain plants that were last observed in 2009. These plants are locaed within the town of Cheney where presumadly a staging area would be establised for any reposne action. Therefore, plants in this location would not be affected by the constriction of a staging area.
928	High	2008	Washington	ND	ND	Extant	Good	Location is precise	
948	High	1990	Washington	ND	ND	Extant	Fair	Location is precise	
1123	Medium	1998	Washington	ND	ND	Extant	Good	Location is precise	
1155	Medium	2007	Washington	ND	ND	Extant	Fair	Location is precise	
1207	High	1986	Washington	ND	ND	Extant	Good	Location is precise	
1724	High	1998	Washington	ND	ND	Extant	Fair	Location is precise	These plants are growing associated with Chambers Lake and Shaver Lake in an area adjacent to Highway 507. There are unnamed roads in this wooded area assumably providing access to the lakes. The pipeline runs along the east side of highway 507. Highway 507 is located between the plants and the pipeline
1824	Medium	2008	Washington	ND	ND	Extant	Fair	Location is precise	
1880	High	1987	Washington	ND	ND	Extant	Good/Fair	Location is precise	The polygon for these plants overlaps with FID 303 on the south side. The rail line runs through this polygon, therefore a spill repsonse in this area could affect these plants. The polygon for these plants overlaps directly with FID 4872, 1880 and FID 4915.
1892	High	2009	Washington	ND	ND	Extant	Fair	Location is precise	The polygon for these plants overlaps with FID 303 on the south side. The rail line runs through this polygon, therefore a spill repsonse in this area could affect these plants. The polygon for these plants overlaps directly with FID 4872, 1880 and FID 4915.
2042	High	1998	Washington	ND	ND	Extant	Fair	Location is precise	
2078	Medium	1998	Washington	ND	ND	Extant	Good	Location is precise	
2115	Medium	1987	Washington	ND	ND	Extant	Fair	Location is precise	
2129	High	2007	Washington	ND	ND	Extant	Good/Fair	Location is precise	
2270	High	1993	Washington	ND	ND	Extant	Failed to find	Location is precise	
2378	Medium	2012	Washington	ND	ND	Extant	Fair	Location is precise	
2432	Medium	2009	Washington	ND	ND	Extant	Good	Location is precise	
2535	Medium	2009	Washington	ND	ND	Extant	Good/Fair	Location is precise	

Water Howellia (*Howellia aquatilis*)									
Feature/ Object ID	Estimated reporting Accuracy	Last Observed	State	Modified Date	Creation Date	Occurrence/ Presence	Occurance Rank	Precision	Notes
2597	High	1996	Washington	ND	ND	Extant	Fair	Location is precise	These plants are growing associated with Chambers Lake and Shaver Lake in an area adjacent to Highway 507. There are unnamed roads in this wooded area assumably providing access to the lakes. The pipeline runs along the east side of highway 507. Highway 507 is located between the plants and the pipeline
2649	Medium	1998	Washington	ND	ND	Extant	Good	Location is precise	These plants are growing associated with Chambers Lake and Shaver Lake in an area adjacent to Highway 507. There are unnamed roads in this wooded area assumably providing access to the lakes. The pipeline runs along the east side of highway 507. Highway 507 is located between the plants and the pipeline
2775	High	2007	Washington	ND	ND	Extant	Good/Fair	Location is precise	
2881	Medium	2008	Washington	ND	ND	Extant	Good	Location is precise	
2975	Medium	1998	Washington	ND	ND	Extant	Good	Location is precise	
3063	Medium	1998	Washington	ND	ND	Extant	Fair	Location is precise	
3141	Medium	1986	Washington	ND	ND	Extant	Good	Location is precise	
3223	Medium	1987	Washington	ND	ND	Extant	Good/Fair	Location is precise	
3347	High	2010	Washington	ND	ND	Extant	Good	Location is precise	
3376	Medium	1998	Washington	ND	ND	Extant	Good/Fair	Location is precise	These plants are growing associated with Chambers Lake and Shaver Lake in an area adjacent to Highway 507. There are unnamed roads in this wooded area assumably providing access to the lakes. The pipeline runs along the east side of highway 507. Highway 507 is located between the plants and the pipeline
3401	Medium	2009	Washington	ND	ND	Extant	Fair	Location is precise	
3452	Medium	1998	Washington	ND	ND	Extant	Good	Location is precise	
3533	Medium	1993	Washington	ND	ND	Extant	Poor	Location is precise	
3556	High	2008	Washington	ND	ND	Extant	Fair	Location is precise	
3587	Medium	2007	Washington	ND	ND	Extant	Good	Location is precise	
3696	High	2001	Washington	ND	ND	Extant	Good	Location is precise	These plants are growing associated with Chambers Lake and Shaver Lake in an area adjacent to Highway 507. There are unnamed roads in this wooded area assumably providing access to the lakes. The pipeline runs along the east side of highway 507. Highway 507 is located between the plants and the pipeline
3811	High	2010	Washington	ND	ND	Extant	Fair	Location is precise	
4118	High	2008	Washington	ND	ND	Extant	Good/Fair	Location is precise	
4123	Medium	2009	Washington	ND	ND	Extant	Good/Fair	Location is precise	
4189	High	2009	Washington	ND	ND	Extant	Good/Fair	Location is precise	
4213	Medium	1998	Washington	ND	ND	Extant	Good	Location is precise	These plants are growing associated with Chambers Lake and Shaver Lake in an area adjacent to Highway 507. There are unnamed roads in this wooded area assumably providing access to the lakes. The pipeline runs along the east side of highway 507. Highway 507 is located between the plants and the pipeline
4214	High	2008	Washington	ND	ND	Extant	Fair	Location is precise	
4366	High	2012	Washington	ND	ND	Extant	Verified extant	Location is precise	
4376	Medium	2008	Washington	ND	ND	Extant	Good	Location is precise	
4456	Medium	2013	Washington	ND	ND	Extant	Good	Location is precise	
4474	Medium	1998	Washington	ND	ND	Extant	Good	Location is precise	These plants are growing associated with Chambers Lake and Shaver Lake in an area adjacent to Highway 507. There are unnamed roads in this wooded area assumably providing access to the lakes. The pipeline runs along the east side of highway 507. Highway 507 is located between the plants and the pipeline
4511	High	2009	Washington	ND	ND	Extant	Good	Location is precise	
4578	High	2008	Washington	ND	ND	Extant	Fair	Location is precise	
4672	Medium	1987	Washington	ND	ND	Extant	Fair/Poor	Location is precise	
4716	High	2008	Washington	ND	ND	Extant	Good/Fair	Location is precise	
4817	Medium	2009	Washington	ND	ND	Extant	Fair	Location is precise	
4845	Medium	2011	Washington	ND	ND	Extant	Good	Location is precise	
4872	High	1987	Washington	ND	ND	Extant	Fair	Location is precise	The polygon for these plants overlaps directly with FID 1880 and FID 4915.
4915	High	1987	Washington	ND	ND	Extant	Good/Fair	Location is precise	
4930	High	2008	Washington	ND	ND	Extant	Good/Fair	Location is precise	
5269	High	2008	Washington	ND	ND	Extant	Fair	Location is precise	

Present in Oregon no data in ORBIC
Present in Idaho but not in the action area.

White Bluffs Bladderpod- NE							
Feature/ Object ID	Estimated reporting Accuracy	Last Observed	Modified Date	Creation Date	Occurrence/ Presence	Precision	Notes
	Not in the Action Area						

Western Lily (*Lilium occidentale*)

Feature/ Object ID	Estimated reporting Accuracy	Last Observed	Modified Date	Creation Date	Occurrence/ Presence	Precision	Notes
24		2006	10/5/2016	3/2/1992	Excellent or good estimated viability	ND	These plants are growing with the Sunset Bay State Park on the south side of the Cape Argo Hwy. A spill response along the coast should not affect these plans as the response would occur on the waterward side of the hwy. These plants are located in a wooded area east of a residential area that borders HWY 101. These plants will not be affected by a spill response.
47	Very High	2007	12/8/2016	1/6/2000	Poor estimated viability	ND	
54		2008	10/5/2016	12/20/1992	Fair or poor estimated viability	ND	There as small polygons of these plants within Floras Lake State Natural Area. These plants are in the vicinity of the Cape Blanco State Airport. Two of the 5 small polygons are near or or Airport Road. These plants are within a wooded area and will not be affected by a spill response.
62	Very High	7/7/2009	12/8/2016	3/2/1992	Fair or poor estimated viability	ND	There are two small polygons locatied to the east of highway 101. Spill reposnse along the shoreline is not expected to affect these plants. This plant grows at the edge of sphagnum bogs and in forest ot understory openings at the marging of ephemeral ponds and small channels. It is also found in prairie and scrub near the ocean.
63	High	2008	10/5/2016	10/18/1992	Poor estimated viability	ND	These plants are growing east of highway 101. Spill reponse along the shoreline is not expected to affect these plants. This plant grows at the edge of sphagnum bogs and in forest understory openings at the margins of ephemeral ponds and small channels. It is also found in prairie and scrub near the ocean.
123	High	7/3/2003	12/8/2016	7/18/2002	Fair or poor estimated viability	ND	According to USFWS 2008 5 - Year Review this population is in decline. It was last surveyed in 2008.
167	Very High	2007	12/8/2016	3/1/1994	Fair or poor estimated viability	ND	
175	High	1988	12/8/2016	10/18/1992	Extirpated	ND	According to USFWS 2009 5-Year Review this population is extirpated and was last observed in 2004. This population was located within a dense residential area.
188	High	1988	12/8/2016	4/14/1997	Extirpated	ND	According to USFWS 2009 5-Year Review this population is extirpated and was last observed in 2004
214	High	2007	12/8/2016	12/9/1992	Extirpated	ND	These plants were located within a developed residential area. This area doesn't appear to be suitable habitat due to the surrounding development.
218	High	1988	12/8/2016	4/14/1997	Extirpated	ND	
243		between 1988	10/5/2016	10/19/1992	Poor estimated viability	ND	According to USFWS 2008 5 - Year Review this population is extirpated. Some individuals were observed and surveyed in 2008. These plants were located on a narrow strip adjecent to a roadway, the area in located adjacent to farm fields.
275	High	6/16/1992	12/8/2016	12/8/1992	Poor estimated viability	ND	
276	High	2008	12/8/2016	12/9/1992	Poor estimated viability	ND	
279	High	7/29/2003	12/8/2016	8/3/2003	Poor estimated viability	ND	
296	Low	2008	12/8/2016	10/19/1992	Extirpated	ND	According to USFWS 2009 5-Year Review this population is extirpated and was last observed in 2004
297		2005	10/5/2016	5/19/1993	Poor estimated viability	ND	
309	Very High	2007	12/8/2016	11/5/2007	Poor estimated viability	ND	
310	High	1994	12/8/2016	11/5/2007	Verified extant (viability not assessed)	ND	
311	Very High	2003	12/8/2016	11/5/2007	Poor estimated viability	ND	
363	High	2008	12/8/2016	6/17/2008	Poor estimated viability	ND	These plants are loacted in a narrow area adjacent to farm fields. A spill response in the marine environment along the shoreline will not affect these plants.

Western Lily (*Lilium occidentale*)							
Feature/ Object ID	**Estimated reporting Accuracy**	**Last Observed**	**Modified Date**	**Creation Date**	**Occurrence/ Presence**	**Precision**	**Notes**
365	High	2008	12/8/2016	6/17/2008	Excellent or good estimated viability	ND	These plants are located in what appears to be an open area far from a roadway and the beach. A spill response in this area will not affect these plants.
443	High	7/7/2009	12/8/2016	10/14/1992	Good or fair estimated viability	ND	
445	Very High	2008	12/11/2016	4/8/1992	Good or fair estimated viability	ND	According to USFWS 2009 5-year review the popualtion was last surveyed in 2002 and last observed in 2008. The plants appear to be in areas on the top of cliffs.
447	Very Low	2008	10/5/2016	10/24/2012	Poor estimated viability	ND	
492	High	1988	12/8/2016	4/14/1997	Extirpated	ND	
544		2008.00	10/5/2016	12/20/1992	Fair or poor estimated viability	ND	These plants are located within a wooded area near the airport. A spill response along the shoreline will not affect these plants.
545	Very High	7/14/2004	12/8/2016	11/6/2007	Poor estimated viability	ND	These plants are located in an open area on a peninsula between 2 embayments. This area could be used for staging to access the shorelin, Iso these plants could be affected.
700	High	2004	12/8/2016	3/6/1995	Poor estimated viability	ND	These plants are located in an area next to what appears to be a lake between a cranberry (assumuing) farm and the shoreline. There are no roads through this area to provide access and staging and therefore these plants won't be affected by a spill response in this area.
766		5/1/05	10/5/2016	10/18/1992	Fair or poor estimated viability	ND	These plants are scattered throughout an undeveloped area east of highway 101 about 0.2 miles east of highway 101. There is one group iof plants located directly adjacent to the highway which could be affect should a staging are be establied in this location. According to USFWS 2009 these plants were last surveyed in 2005 and are declining.

NLAA - due to location and proximity to potential staging areas. Two locations where plants have been observed could be affected by a spill response but the likelihood of a spill in this location is remote.
Not located in Washington
Check location is in Idaho.

Willamette Daisy- NLAA

Feature/Object ID	Estimated reporting Accuracy	Last Observed	Modified Date	Creation Date	Occurrence/Presence	Precision	Notes
15	Unknown	6/13/1924	8-Dec-16	10-Oct-84	Historical		This is a highly urbanized and develped area. Plants are liokely extirpated.
17	Low	1894-07-11	5-Oct-16	10-Oct-84	Historical		The location of these plants is very large most of the area appears to be farm fields. There is a pipline running throug this large polygon. Its likely tht a staging area would be established within the urbanized area close by.
35	Very High	1987-06	8-Dec-16	25-Dec-91	Poor estimated viability		
37	High	5/24/1987	8-Dec-16	11-Jun-91	Poor estimated viability		
52	High	1993	8-Dec-16	18-Jan-98	Poor estimated viability		
55	High	7/18/1991	8-Dec-16	19-Feb-92	Fair estimated viability		
60	Medium	6/3/1989	8-Dec-16	12-Oct-92	Fair or poor estimated viability		
85	Very High	7/3/2006	8-Dec-16	15-Feb-90	Fair or poor estimated viability		
86	Low	1915-07	5-Oct-16	10-Oct-84	Historical		
92	Low	6/23/1934	8-Dec-16	10-Oct-84	Historical		
97	High	4/28/1992	8-Dec-16	10-Feb-93	Poor estimated viability		
107		6/11/2015	14-Nov-16	15-Feb-90	Excellent estimated viability		
109	High	7/6/2006	8-Dec-16	10-Sep-92	Good estimated viability		
110	High	6/9/1992	8-Dec-16	22-Oct-92	Poor estimated viability		
124	Low	1894-06	5-Oct-16	10-Oct-84	Possibly historical		These plants were located in what is now a highly urbanized area along the columbia river.
144	High	6/27/2006	5-Oct-16	6-Jul-93	Fair estimated viability		
153	Very High	4/10/2009	11-Dec-16	26-Sep-90	Excellent estimated viability		
161	Medium	1980-06	8-Dec-16	15-Feb-90	Good estimated viability		
168		6/16/2006	5-Oct-16	12-Oct-92	Good estimated viability		
191	Very High	6/3/1992	8-Dec-16	10-Sep-92	Poor estimated viability		
206	Low	6/28/1922	8-Dec-16	14-May-90	Possibly historical		
227	High	6/15/1996	8-Dec-16	22-May-97	Poor estimated viability		
271	High	5/29/1992	8-Dec-16	10-Sep-92	Poor estimated viability		
291	Very High	6/16/2015	11-Dec-16	24-Feb-92	Fair estimated viability		
299	High	2009	8-Dec-16	21-Apr-04	Poor estimated viability		
324	High	6/27/2002	8-Dec-16	1-Jun-05	Fair estimated viability		
338	High	2006-sum	8-Dec-16	5-Feb-07	Poor estimated viability		
355		7/2/2003	5-Oct-16	31-May-05	Excellent, good, or fair estimated viability		
369	High	6/19/2007	5-Oct-16	27-May-03	Good estimated viability		
383	High	6/12/1986	8-Dec-16	26-Sep-90	Fair estimated viability		
393	High	2008-summer	8-Dec-16	19-Aug-10	Verified extant (viability not assessed)		
431	High	7/10/1986	8-Dec-16	15-Feb-90	Fair estimated viability		
452		6/10/2015	14-Nov-16	15-Feb-90	Good estimated viability		
477	Very High	8/1/2011	8-Dec-16	4-Aug-11	Verified extant (viability not assessed)		
505	High	6/27/1988	8-Dec-16	26-Mar-91	Excellent or good estimated viability		
592	High	6/27/2002	8-Dec-16	31-May-05	Poor estimated viability		
688		6/29/2015	14-Nov-16	19-Nov-91			

Not located in Washington
Check Idaho

Appendix E. California Dispersant Plan Concurrence Letters

UNITED STATES DEPARTMENT OF COMMERCE
National Oceanic and Atmospheric Administration
NATIONAL MARINE FISHERIES SERVICE
West Coast Region
501 West Ocean Boulevard, Suite 4200
Long Beach, California 90802-4213

May 11, 2018

Refer to NMFS No:
WCR 2018-9670

Timothy Holmes
Incident Management & Preparedness Advisor
U.S. Coast Guard, 11th District
Bldg 50-8 CG Island
Alameda, California 94501-5100

Dan Meer
Assistant Director - Superfund
U.S. EPA Region IX
75 Hawthorne Street
San Francisco, California 94105

Re: Endangered Species Act Section 7(a)(2) Concurrence Letter and Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat Response for the Dispersant Preauthorization under Section I of the California Dispersant Plan, Appendix XII of the Regional Response Team IX Regional Contingency Plan

Dear Mr. Holmes and Mr. Meer:

On November 21, 2016, NOAA's National Marine Fisheries Service (NMFS) received final amendments to your second version biological assessment (BA) and request for a written concurrence that the U.S. Coast Guard (USCG) 11th District's and U.S. Environmental Protection Agency (EPA) Region IX's Dispersant Preauthorization under Section I of the California Dispersant Plan (CDP), Appendix XII of the Regional Response Team IX Regional Contingency Plan under the Oil Pollution Act of 1990 is not likely to adversely affect (NLAA) species listed as threatened or endangered or critical habitats designated under the Endangered Species Act (ESA). Additional effects determinations for two species listed in January 2018 were received on May 01, 2018. This response to your requests was prepared by NMFS pursuant to section 7(a)(2) of the ESA, implementing regulations at 50 CFR 402, and agency guidance for preparation of letters of concurrence.

NMFS also reviewed the proposed action for potential effects on essential fish habitat (EFH) designated under the Magnuson-Stevens Fishery Conservation and Management Act (MSA), including conservation measures and any determination you made regarding the potential effects of the action. This review was pursuant to section 305(b) of the MSA, implementing regulations at 50 CFR 600.920, and agency guidance for use of the ESA consultation process to complete EFH consultation.

This letter underwent pre-dissemination review using standards for utility, integrity, and objectivity in compliance with applicable guidelines issued under the Data Quality Act (section 515 of the Treasury and General Government Appropriations Act for Fiscal Year 2001, Public Law 106-554). The concurrence letter will be available through NMFS' Public Consultation Tracking System (https://pcts.nmfs.noaa.gov/pcts-web/hompage.pcts). A complete record of this consultation is on file at the North-Central Coast Office in Santa Rosa, CA.

Proposed Action and Action Area

The federal action is the preauthorized use of four oil spill dispersants listed on the National Oil and Hazardous Substances Pollution Contingency Product Schedule and licensed for use by the State of California (COREXIT EC9527A, COREXIT EC9500A, NOKOMIS 3-AA, and NOKOMIS 3-FA), as delineated in the CDP Section 1, in Federal waters 3-200 nautical miles (nm) from the California shoreline excluding waters within National Marine Sanctuaries, waters within 3 nm of the California-Mexico border and waters within 3 nm of coastal islands. Any other dispersant chemical would require Area Regional Response Team approval for use and is not part of this consultation. The action area includes all areas directly or indirectly affected by the Federal action and therefore includes all marine waters generally within 200 nm of the California shoreline from the borders with Mexico and the State of Oregon. As dispersant will only be applied in the event of an oil spill, the presence of oil in the water in the preauthorized zone is assumed.

Although four dispersants are authorized for use by the State of California, only two are actually stockpiled and available for use, COREXIT EC9500A and EC9527A. There is approximately 43,000 gallons of COREXIT EC9500A stationed at eight locations from Eureka to Long Beach, California, and it represents the most likely dispersant to be used in a preauthorized response. There is approximately 9,550 gallons of COREXIT EC9527 stockpiled in Carpenteria, California (Holmes, pers. com., 2016). This formulation is believed to be more effective on some of the heavier crude oils produced by several of the offshore platforms in this area and it is stored in the event of an appropriate spill. There are no stockpiles of the NOKOMIS products making their utilization during the 96 hour preauthorization window highly unlikely.

USCG regulations require a seven hour response time to an oil spill and all areas off the California coast are considered accessible by the C-130 aircraft stationed in Chandler, Arizona for this purpose. Additional time may be required for dispersant loading at a staging area near the spill and flying to the spill site and this time lag means some water column and/or inhalation impacts from the oil is likely to occur to those organisms present in the spill zone. This response capability makes the C-130 application of one of the COREXIT products the most likely first response (Holmes, pers. com., 2016).

The CDP provides a decision making framework for the preauthorized use of dispersants and the following restrictions on the use of dispersants in the preauthorized zone are included:

- The Pre-Approval Zone Dispersant Use Checklist in the CDP will be utilized.
- Only spilled oil that can be chemically dispersed with the approved dispersants will be treated. Other response options will be implemented for other types of oil or oil products.
- Diesel spills will not be treated with dispersants.
- Only surface applications from aircraft or boats are allowed. No subsurface applications are preapproved.

- If Special Monitoring of Applied Response Technologies (SMART) indicates that dispersant use is ineffective, dispersant use will be stopped.
- Caution will be taken to avoid spraying of marine mammals and sea turtles.
- Applications will only take place during daylight hours and safe sea and air conditions.

The second BA also clarified that pre-approval only applies to dispersant operations expected to be less than 96-hours in duration in accordance with USCG national policy and that the action agencies will initiate emergency consultation with NMFS for all applications of dispersants in order to account for incident specific variables.

Several modifications to the proposed action and second BA have been made through continued informal consultation in order to address and minimize potential interrelated and interdependent effects of the preapproved use of dispersants in the action area. Interrelated and interdependent effects may arise from the use of aircraft and ships to apply the dispersants on an oil spill and include potential ship strikes on, or disturbance of, listed marine mammals and sea turtles. On November 21, 2016, USCG and EPA detailed the following changes and clarifications to the proposed action in the second BA by letter to NMFS and USFWS:

1. The Federal On-Scene Coordinator (FOSC) will establish a minimum horizontal, no-spray buffer of 100 meters (328 feet) from observed congregations of fish or brown sea nettles, rafting flocks of birds, marine mammals or sea turtles in the water and/or marine mammal haul-out areas to minimize potential contact with dispersant spray drift, and noise and vessel disturbance. Incident specific buffers will be based on dispersant drift spray models. According to a dispersant spray drift model produced for the USCG (AMOG Consulting 2016), this will likely result in larger no-spray buffer zones that vary by incident due to several variables that will be examined in individual consultations.
2. Protected species observers will be present on aircraft and vessels associated with dispersant application or transiting the action area to engage in the response.
3. All vessels will have personnel assigned with wildlife spotting as their primary duty.
4. Wildlife spotters, whether on vessels or aircraft, will function to record data on protected species within the spill area and will advise the dispersant spotter and spray aircraft or vessels of sites within the operational area where wildlife have been spotted. Wildlife spotters can direct a suspension of spraying if animals are within the buffer area.
5. Vessels involved in dispersant spraying operations will not exceed 10 knots (11.5 miles per hour) in speed when marine mammals or sea turtles are observed in the area.
6. At a minimum, tier 1 SMART monitoring will be performed, and tier 2 and 3 monitoring conducted as appropriate. Incident specific emergency Section 7 consultations may require additional monitoring.
7. To lessen the potential for ship strikes, vessels will avoid close approach to whales, pinnipeds and sea turtles by instituting a 100 yard (300 feet) in-water buffer. If a vessel is approached by one of these species, and it is safe to do so, the vessel will disengage its props until the animal(s) has clearly moved more than 100 yards (300 feet) from the response vessel.
8. Vessels involved with dispersant spray operations will maintain a distance of 200 meters (656 feet) from observed killer whales (orcas).
9. Restricted use zones of 400 meters (1312 feet) will be established around high concentrations of marine mammals or sea turtles (e.g. feeding areas, migration pathways,

haul-outs or rookeries) for dispersant planes and vessels, or at distances established as part of an emergency consultation with NMFS.

On November 16, 2016, the action agencies further modified the proposed action in order to protect the marbled murrelet, a bird species managed by the U.S. Fish and Wildlife Service. The action agencies clarified that no dispersant applications will be preauthorized within 3-5 nm from the Oregon border to the southern Monterey County, CA border between March 24th and September 15th of every year.

Action Agency's Effects Determination
The action agencies have determined the potential impacts resulting from the Dispersant Preauthorization under Section I of the CDP, Appendix XII of the Regional Response Team IX Regional Contingency Plan may affect, but are not likely to adversely affect (NLAA) all of the species and their designated critical habitats that may occur in the action area and that are presented below:

Sacramento River winter-run Chinook salmon ESU (*Oncorhynchus tshawytscha*)
endangered (June 28, 2005, 70 FR 37160)
Central Valley spring-run Chinook salmon ESU (*Oncorhynchus tshawytscha*)
threatened (June 28, 2005, 70 FR 37160)
California Coastal Chinook salmon ESU (*Oncorhynchus tshawytscha*)
threatened (June 28, 2005, 70 FR 37160)
Southern California steelhead DPS (*Oncorhynchus mykiss*)
endangered (January 5, 2006, 71 FR 834)
California Central Valley steelhead DPS (*Oncorhynchus mykiss*)
threatened (January 5, 2006, 70 FR 37160)
Northern California Coast steelhead DPS (*Oncorhynchus mykiss*)
threatened (January 5, 2006, 70 FR 37160)
Central California Coast steelhead DPS (*Oncorhynchus mykiss*)
threatened (January 5, 2006, 70 FR 37160)
South-Central California Coast steelhead DPS (*Oncorhynchus mykiss*)
threatened (January 5, 2006, 70 FR 37160)
Southern Oregon/Northern California Coast coho salmon ESU (*Oncorhynchus kisutch*)
threatened (June 28, 2005, 70 FR 37160)
Central California Coast coho salmon ESU (*Oncorhynchus kisutch*)
endangered (June 28, 2005, 70 FR 37160)
Southern DPS of North American green sturgeon (*Acipenser medirostris*)
threatened (April 7, 2006, 71 FR 17757)
critical habitat (October 9, 2009, 74 FR 52300)
Pacific eulachon/smelt southern DPS (*Thaleichthys pacificus*)
threatened (March 18, 2010, 75 FR 13012)
Eastern Pacific Scalloped hammerhead shark DPS (*Sphyrna lewini*)
endangered (July 3, 2014, 79 FR 38213)
Fin whale (*Balaenoptera physalus*)
endangered (December 2, 1970, 35 FR 18319)
Blue whale (*Balaenoptera musculus*)
endangered (December 2, 1970, 35 FR 18319)
Humpback whale Central America DPS (*Megaptera novaengliae*)

endangered (September 8, 2016, 81 FR 62259)

Humpback whale Mexico DPS (*Megaptera novaengliae*)
threatened (September 8, 2016, 81 FR 62259)

Sperm whale (*Physeter macrocephalus*)
endangered (December 2, 1970, 35 FR 18319)

Sei whale *(Balaenoptera borealis)*
endangered (December 2, 1970, 35 FR 18319)

North Pacific right whale *(Eubalaena glacialis)*
endangered (March 6, 2008, 73 FR 12024)

Western North Pacific Gray Whale *(Eschrichtius robustus)*
endangered (December 2, 1970, 35 FR 18319)

Southern Resident killer whale (SRKW, *Orcinus orca*)
endangered (November 18, 2005, 70 FR 69903)
critical habitat (November 29, 2006, 71 FR 69054)

Guadalupe fur seal *(Arctocephalus townsendi)*
Threatened (December 16, 1985, 50 FR 51251)

Leatherback sea turtle (*Dermochelys coriacea*)
endangered (June 2, 1970, 35 FR 8491)
critical habitat (January 26, 2012, 77 FR 4170)

Loggerhead sea turtle (*Caretta caretta*)
threatened (July 28, 1978, 43 FR 32800)

Green sea turtle (*Chelonia mydas*)
threatened (July 28, 1978, 43 FR 32800)

Olive Ridley sea turtle (*Lepidochelys olivacea*)
threatened (July 28, 1978, 43 FR 32800)

Black abalone (*Haliotis cracherodii*)
endangered (January 14, 2009, 74 FR 1937)
critical habitat (October 27, 2011, 76 FR 66806)

White abalone (*Haliotis sorenseni*)
endangered (May 29, 2001, 66 FR 29046)

Giant Manta Ray (*Manta birostris)*
threatened (January 22, 2018, 83 FR 2916)

Oceanic Whitetip Shark (*Carcharhinus longimanus*)
threatened (January 30, 2018, 83 FR 4153)

The action agencies NLAA determination for all listed species is based upon their review of the potential direct and indirect effects of the toxicity and exposure scenarios for oil spill dispersants and for dispersed oil in the action area, as presented in the second BA. In late 2016, modifications to the proposed action to address interrelated and interdependent effects from the use of vessels and aircraft to transport and apply dispersants in accordance with the CDP were made by the action agencies as a result of numerous meetings and conversations with NMFS.

For all of the whale and turtle species, the action agencies determined that the potential direct toxicological impacts from the oil spill dispersants and dispersed oil were insignificant and the indirect effects to their prey bases were insignificant or discountable. The modifications to the proposed action (e.g. assigning wildlife spotters to all vessels and aircraft, mandatory in-water buffers and speed controls, etc.) reduced the potential interrelated and interdependent effects to discountable. Potential impacts to critical habitat for the Southern Resident Killer Whale (SRKW) and Leatherback sea turtle were also insignificant.

For the salmonid Evolutionarily Significant Units (ESUs) and Distinct Population Segments (DPSs), the Southern DPS of North American green sturgeon (green sturgeon), the Pacific eulachon/smelt Southern DPS (eulachon), and the Eastern Pacific Scalloped hammerhead shark DPS (Scalloped hammerhead shark), the action agencies determined that the wide ranges and variable water column distribution of these species allows these species to avoid exposure and that the potential direct toxicological effects were insignificant. Furthermore, they determined that potential indirect effects to their prey bases were insignificant and there were no potential interrelated or interdependent effects.

For the pelagic Guadalupe fur seal, their solitary and reclusive natures, as well as their rarity in the action area, made the risk of direct exposure to dispersants or dispersed oil insignificant. Potential indirect effects to their prey base were considered insignificant and the modifications to the proposed action reduced potential interrelated and interdependent effects to discountable levels. The giant manta ray and oceanic whitetip shark are also very rare in the action area, and impacts to their prey bases were considered to be insignificant, so that the action agencies considered the potential for adverse effects to be discountable.

Potential effects to white and black abalone were considered discountable because the habitats they occupy are unlikely to be exposed to dispersants or dispersed oil at problematic concentrations. White abalone are found at depths where surface applications of dispersant or dispersed oil are not expected to be detectable while black abalone only occupy intertidal areas outside of the preauthorization zone. The significant distance between the preapproval zone and black abalone habitat (3 nm or 3.45 miles) will allow sufficient dilution of the dispersants or dispersed oil to remove the potential for effects. The action agencies determined there were no potential interrelated or interdependent effects to these species.

Consultation History

Early consultation between NMFS and the action agencies began on this project in April 2012. A draft BA was produced in June 2012. Technical assistance through a series of meetings and conference calls continued until the official submission of a BA on January 29, 2014. NMFS responded to this submission via a letter on March 10, 2014 asking for additional information and analysis and informing the action agencies that there was not yet sufficient information presented to initiate consultation. This triggered a new series of meetings and scheduled calls that culminated in the BA received on August 28, 2015. This BA contained significant amounts of new information and analysis, but questions concerning interrelated and interdependent effects remained. Following another series of scheduled calls, the action agencies amended the project description via letter dated November 21, 2016 to address potential interrelated and interdependent effects. A final, expected report on modeling dispersant spray drift was submitted on December 12, 2016, but was not reviewed by NMFS until January 2017.

During the time between the submission of the second BA and the decision to amend the project description (approximately 14 months), significant scientific information concerning impacts from the Deepwater Horizon (DWH) oil spill (DWH NRDA Trustees 2016) was released that was not analyzed in the second BA. Shortly before the review of the dispersant spray drift modeling report by NMFS, the oceanic whitetip shark was proposed for ESA listing. It was decided not to address the proposed listing at the time and NMFS agreed to analyze the DWH Natural Resources Damage Assessment (NRDA) Trustees report (2016) and other information released after the second BA as part of its analysis rather than require the action agencies to prepare a third BA. The giant manta ray was then proposed for listing in January 2017.

In late January 2018, both the oceanic whitetip shark and the giant manta ray were listed as threatened under the ESA. NMFS informed the USCG of this development on April 12, 2018 and the action agencies elected to make effects determinations on the two newly species and have them included in this consultation process. The request for concurrence letter was received by NMFS on May 1, 2018.

There is an immense reservoir of scientific journals, books, and gray literature involving oil spills, dispersants and the effects of them, alone and in combination with oil, on a myriad of sea creatures and their habitats. NMFS reviewed many sources as part of this consultation process (see attached bibliography) including four books by the National Research Council (NRC) of the National Academies of Science (NRC 2013, 2003, 2005, 1989), and a recent biological opinion from the Alaska Region (NMFS 2015a) and the BA from that effort (USCG and EPA, 2014). The topic remains an area of active research and interest and the NRC has commissioned another expert committee review effort titled "Evaluation of the Use of Chemical Dispersants in Oil Spill Response" that is expected to conclude sometime in 2019.

ENDANGERED SPECIES ACT

Effects of the Action

Under the ESA, "effects of the action" means the direct and indirect effects of an action on the listed species or critical habitat, together with the effects of other activities that are interrelated or interdependent with that action (50 CFR 402.02). The applicable standard to find that a proposed action is not likely to adversely affect listed species or critical habitat is that all of the effects of the action are expected to be discountable, insignificant, or completely beneficial. Beneficial effects are contemporaneous positive effects without any adverse effects to the species or critical habitat. Insignificant effects relate to the size of the impact and should never reach the scale where take occurs. Discountable effects are those extremely unlikely to occur.

This ESA consultation examines the potential effects of preapproval of four dispersants to oil spills on the surface of the ocean in the action area. Application of any other dispersant or response product, by any means other than ships or aircraft, or outside of the CDP decision making framework discussed earlier are not covered. These types of actions would undergo individual, emergency ESA section 7 consultations.

The baseline for this consultation assumes oil has been spilled in the preapproval zone. A discussion of this altered background condition and the proposed action's effect on it is necessary to present the analysis of the action. Oils are a mixture of thousands of petroleum compounds and other contaminants of varying volatility, water solubility and toxicity (NRC 2005). Most oils spilled on the surface of the action area will spread into a slick, with thickness ranging from several millimeters (mm) to one micrometer (μm) depending on the type of oil and other environmental factors (NRC 1998). Since oil does not spread uniformly, slicks are irregular in shape and thickness. They generally are elongated in the direction of the wind (NRC

2005). Some oils will sink. There is a large variety of crude oils and refined oil products that are transported through the action area off the California coast with some oils identified as readily dispersible and numerous others that are known to not be dispersible.

Wind driven waves will break up an oil slick, producing droplets of various sizes that may be stabilized by natural surfactants, leading to some natural dispersion. Generally, oil droplets are prominent under a slick up to a meter deep in the low parts per million range under natural dispersion scenarios (3-5 ppm up to 1 m depth, 0.03-0.63 ppm 1-2 m deep (NRC 1989)). Smaller droplets are more likely to remain dispersed, while larger droplets are more likely to resurface, but smaller droplets may also coalesce into these larger droplets and reemerge on the surface (NRC 1989).

Movement of the surface slick is generally dictated by the wind in both direction and speed. Lighter molecular weight fractions of the oil (*e.g*. short-chained alkanes, benzene, toluene, ethylbenzene, xylenes and some other two and three ringed Polycyclic Aromatic Hydrocarbons (PAHs) are soluble and they can diffuse away from the surface slick into the waters below the surface layer although many volatilize rapidly as discussed below. They will not coalesce and resurface and these compounds can cause toxic effects (e.g. narcosis) in the water column to exposed biota (NRC 2005). During conditions that slow evaporation rates (*e.g*. night time, cold temperatures) a greater percentage of these more acutely toxic compounds dissolve into the water column where they may impact zooplankton and other near surface life. As oil weathers, the concentration of higher molecular weight PAHs remaining in the slick increase because of the loss of the lighter fraction of hydrocarbons (NRC 2005).

Movement of water below the surface layer may proceed in a different direction based upon the direction of local currents (Zeinstra-Helfrich, *et. al.*, 2015, George-Ares and Clark 2000, Fingas 2014, Mearns *et. al.*, 2001). In the action area, the California current generally moves water from the north to the south, while prevailing winds push the surface waters to the east towards the mainland shore. There are counter currents and gyres in the Southern California area that influence local transport processes as well (Howard, *et. al.*, 2014). Therefore, an oil spill can actually spread in multiple directions and result in a larger contaminated volume of water than is readily evident from just surface observations. A surface slick can serve as a reservoir of oil droplets that undergo natural dispersion as the slick spreads resulting in a prolonged oil exposure event (Carls *et. al.*, 2008).

Evaporation is the most important and rapid of all weathering processes and it can account for the loss of 20-50% of many crude oils and 75% or more of refined petroleum products (NRC 2005). This often leads to a significant loss of the lighter weight, soluble and acutely toxic components of oil and they will not be present to affect organisms in the water column when dispersant applications actually take place. While this may benefit water column organisms, inhalation or aspiration of these compounds by air breathing organisms is also possible during this time and this may cause toxic effects.

Following spreading, evaporation/volatilization and natural dispersion of the spilled oil, wave action may cause some oils to emulsify, forming what is commonly referred to as a mousse. The oil absorbs water and this causes the volume of the spill that must be dealt with to increase dramatically. Mousses are difficult to remove from the ocean by mechanical means or the use of dispersants although some crude oil mousses have been successfully dispersed (NRC 2005).

Ultimately, oil that is introduced into the ocean undergoes some level and form of microbial biodegradation. Biodegradation rates are highly variable based upon the properties of the oil, environmental conditions and the microbes present. In warmer waters, and in waters with natural oil seeps and microbes evolved to take advantage of this carbon source, this process tends to be more rapid than in cooler waters and waters where oil is rare. Microbial growth on open ocean oil slicks is likely to be limited by nutrient availability and may be a slow process relative to the formation of emulsions, which are often very difficult to biodegrade (NRC 2005). Incomplete biodegradation may result in the formation of high molecular weight residues such as "tar balls" or asphaltenes that may sink in open waters and later wash up on shorelines.

Dispersants do not reduce the amount of oil in a spill, but reduce the mass of oil at the surface of the water by forcing the oil into smaller droplets that can be suspended in the water column (NRC 2005). Dispersants also tend to prevent coalescence of oil droplets into sizes that resurface more rapidly (NRC 1989). The goal of dispersant use is to enhance dilution, get the oil off the surface and away from animals susceptible to physical oiling (of skin, fur or feathers) and inhalation impacts from the oil, and prevent its stranding on shore where it may cause chronic exposures to aquatic resources there and in the intertidal zone (Bue *et al.* 1998, Heintz *et al.* 1999, Rice *et al.* 2001, Carls *et al.* 1999). The use of a dispersant is a calculated trade-off of impacts to surface and shoreline resources versus those in the water column (NRC 2005, 1989).

The dispersant effectively encapsulates the oil in a micelle similar in nature to detergents. The nonpolar (hydrophobic) portion of the dispersant attaches to the oil while the polar (hydrophilic) portion remains in the water column (Tjeerdema *et. al.* 2010, Lin *et al.* 2009). Generally, chemically dispersed oil is considered to be confined to the upper 10 meters of the water column due to temperature and density gradients (NRC 2005, 1989, BenKinney *et. al.*, 2011), but this depth is somewhat variable based on environmental conditions. For example, many research and monitoring cruises during the DWH oil spill response found the upper mixed layer there to be ~20m deep based on the conductivity and temperature measurements with a range of 16-29m at times (Grennan *et. al.*, 2015, DWH NRDA Trustees, 2016.) Application of dispersant results in a rapid increase in the mass of oil already in the water column due to natural dispersion, but this also results in rapid dilution of the oil into a greater volume (NRC 2005). This greater dilution rapidly offsets the greater mass in the water column, resulting in concentrations that are lower than those under natural dispersion as the dispersed oil pushes deeper into the water column and becomes subject to subsurface currents influencing transport direction and speeds.

Bejarano *et. al*. (2014b) conducted a recent review of oil spill literature and noted that field trials showed initial high oil concentrations within the top few meters rapidly declining within minutes to hours (≤ 4 hours) to concentrations of 1 ppm or less following dispersant application. This is also evident from monitoring during the DWH response that showed a maximum total petroleum hydrocarbon concentration of 2 ppm at 1m depth approximately 30 minutes after chemical dispersion of a weathered oil slick at the surface (Bejarano *et. al.*, 2013). BenKinney *et. al.* (2011) noted that dispersed oil concentrations at 10m depth were consistent with background concentrations while monitoring aerial dispersant applications during the DWH response. The second BA (USCG and EPA 2015) cites several additional older studies showing similar patterns.

The amount of oil that may reach sensitive shoreline habitats is also reduced by dispersant application and this may mitigate longer term impacts and exposures to some species found there

(Bejarano *et. al.*, 2014b, NRC 2005, 1989, Bue *et al.* 1998, Heintz *et al.* 1999, Rice *et al.* 2001, Carls *et al.* 1999). The mass of oil components in the water column increases and this increases the exposure potential for pelagic and benthic organisms, but not necessarily the toxicity to them. The potential toxicological impacts of dispersant application varies at each spill and depends on the type of oil product spilled, the amount of weathering (evaporation, natural diffusion into the water, ultraviolet (UV) light degradation, etc.) that takes place before dispersant application, the type and life stage of biotic resources present at the site, and natural variables such as temperature, current speeds, UV light intensity, etc.

The toxicity of oil comes from the bioavailability and toxicity of individual hydrocarbons that make up the oil and relates to their solubility in water. Dissolved hydrocarbons, whether chemically or naturally dispersed, may diffuse across gills, skin and other membranes of organisms (NRC 2005). The sensitivity of individual species and life stages is highly variable, but embryonic and larval life stages are usually more sensitive than adults (NRC 2005, DWH NRDA Trustees 2016, Barron *et al.*, 2013, Bejarano *et al.*, 2014, NMFS 2015). Narcosis is a typical form of impact from these exposures and can result from both PAHs and monaromatic or heterocyclic aromatic hydrocarbons (NRC 2005). Other work has shown cardiac toxicity to developing fish embryos (Incardona *et. al.*, 2014, Carls *et. al.*, 1999) resulting in mortality.

Many studies also identified photoenhanced toxicity of PAHs as a potential means of impacting surface and near surface resources exposed to an oil spill (NRC 2005). The DWH NRDA Trustees report (2016) found DWH oil to be ~10-100 times more toxic to invertebrates and larval fish species such as red snapper, mahi-mahi, and bay anchovies. These impacts are most likely to occur to translucent or semi-transparent pelagic larvae and organisms living in shallow water areas that ingest or otherwise absorb some PAHs and where ultraviolet light exposure is greatest. This may include oiled shorelines. This type of impact may not be prominent among opaque organisms (e.g. adult fish, invertebrates, mammals, etc.) or organisms that migrate into the photic zone during the night and retreat to depths during the day (NRC 2005). The effects will occur in the shallow ocean waters whether the oil is naturally or chemically dispersed, but dispersion of an oil slick may reduce the surface area of oil impacting the photic zone and the time it is there.

The NRC (1989) concluded, as shown in the second BA (USCG and EPA 2015), that the acute lethality of dispersed oil is primarily associated with the dissolved oil constituents, and very little with the dispersant itself. The NRC (2005) presented data from many studies to further illustrate that COREXIT 9500 and 9527 are significantly less toxic to multiple species compared to oil and dispersed oil. EPA (2010a, Hemmer *et. al.*, 2011) tested several dispersant formulations during the DWH oil spill response due to the concerns of the public about the volume of COREXIT dispersants being applied. These tests included COREXIT 9500 and the two NOKOMIS products subject to this consultation. The EPA reconfirmed that COREXIT 9500 and the NOKOMIS dispersants were much less toxic than the test oil (Louisiana sweet crude) and the dispersed oil. Numerous other studies have also found that dispersants alone were less toxic than the oils they were tested with (Almeda *et. al.*, 2014, Adams *et. al.*, 2014, Barron *et. al.*, 2013, Coelho *et. al.*, 2011, McFarlin *et. al.*, 2011, Fuller *et. al.*, 2004, DWH NRDA Trustees 2016).

The NRC (2005) further concluded that there was no compelling evidence that chemically dispersed oil is more toxic than physically dispersed oil when the comparisons of toxicity are based upon the measured concentrations of petroleum hydrocarbons in the water column rather than the nominal concentration of oil in water. The NRC (1989) noted that dispersant toxicity thresholds were often reported as nominal concentrations (the total amount of dispersant or oil divided by the total volume of water in the experiment's design) rather than measured concentration of the compounds to which organisms were actually exposed. They (NRC 1989) noted that 2/3 of the literature published prior to 1987 presented nominal concentration data rather than measured concentrations and they concluded that a substantial number of these early studies misinterpreted the toxicity data because of this experimental technique. This is because the bioavailability of the oil components in the dissolved, colloidal and particulate phases may vary (Fuller 1999, Lin *et al.*, 2009) and the nominal concentration method does not allow for differentiation of which forms are bioavailable to the test organism. The encapsulation of the hydrocarbon molecules in a dispersant micelle reduces the toxicity of the oil by making the hydrocarbon generally incapable of diffusing across cell membranes, greatly reducing its bioavailability (Tjeerdema *et. al.*, 2010, Fuller *et. al.* 2004, Lin *et. al.*, 2009). The NRC (2005) determined that the nominal concentration method was no longer generally acceptable for toxicity evaluations involving oil and that standardized protocols (Aurand and Coehlo 2005) were necessary for future work.

To provide further analysis of this point following a number of papers published post-DWH that used the nominal concentration method, Bejarano *et. al.* (2014b) compiled a large number of paired data sets from studies conducting water accommodated fractions (WAF or naturally dispersed) and chemically enhanced water accommodated fractions (CEWAF or chemically dispersed) exposure experiments. It differentiated between the data by experimental design (nominal v. measured concentrations of oil loading) and found that the acute toxicity of CEWAF can be grossly over predicted when using the outdated nominal concentration methods. For the COREXIT products, there were 329 measured WAF-CEWAF paired data points for individual species from 36 independent studies. 89% of this paired data for COREXIT 9527 (n=67) had CEWAF ≤ WAF in toxicity. When CEWAF was determined to be more toxic, it was only between 1.62 and 1.76 fold more toxic, which is within the degree of repeatability for standard acute toxicity testing. However, when nominal concentrations were used, CEWAF was more toxic than WAF in 80% of the paired-data set by 1.1 to >1000 fold.

There are 262 paired records available for COREXIT 9500 in this examination and 78% of measured data points showed CEWAF ≤ WAF in toxicity with most (76%) within threefold of the WAF value. However for the nominal concentration information, 93% of the data had CEWAF as more toxic than WAF by 1.2 to > 1000 fold. The critical review (Bejarano *et al.* 2014b) determined that the nominal concentration method is not a reliable metric of toxicity.

Dispersants also mitigate the toxic effects of oil exposure to water column resources by reducing the duration and concentration of exposure through increased, rapid dilution (NMFS 2015, NRC 2005, 2003, 1998, USCG and EPA 2015, 2014, Bejarano *et al.*, 2014b). This results in another conflict with large portions of the scientific literature (especially older studies but also many recent studies following DWH) regarding the time of exposure and determinations of toxicity based upon experiments with unrealistic exposure scenarios. The environmentally realistic scenario for the use of oil spill dispersants under consideration in the preapproval zone of the CDP will result in an exposure to dispersed oil that will rapidly spike and then dilute as the treated oil disperses deeper into the water column and is advected away from the surface slick (Aurand and Coelho, 2005). As discussed previously, the concentrations to which an organism may be exposed in the water column rapidly dilutes within minutes to hours (≤ 4 hours) to low (≤ 1 ppm) or background levels (Bejarano *et. al.*, 2014b, NRC 2005, 1998, BenKinney *et. al.*, 2011). However, a very large proportion of the studies generate information using traditional toxicological experiment designs, i.e. continuous 24 to 96-hour exposures of organisms to dispersants and dispersed oil, despite these time periods being considered invalid. Longer than realistic exposures lead to overestimates of toxicity.

Clark *et. al.*, (2001) found that spiked exposure conditions were up to 36 times less toxic than constant exposure conditions for COREXIT 9500 and 9527 when tested with three types of oil on five different species. Fuller *et. al.* (2004) found declining exposures of dispersed oil to be clearly less toxic than constant exposures by a factor of nine while, in a paper that compared the results of numerous published data sets, George-Ares and Clark (2000) found that the LC50 values for the most sensitive species in the spiked exposure experiments exceeded the maximum measured COREXIT 9500 and 9527 concentrations in field trials in most cases. Greer *et. al.* (2012) found that pulse exposures of Arabic light crude with COREXIT 9500 were not toxic to Atlantic herring while COREXIT 9500 and Alaska north slope crude resulted in toxicity at concentrations 15 minutes post mixing, but not at 30 or 60 minutes.

Dispersants may also aid in the biodegradation process by greatly increasing the surface area of the spilled oil available to bacteria although the observed rates vary among studies with some even showing the rate of biodegradation initially slows (Abbriano *et al.* 2011, Kleindienst *et al.* 2015, Prince 2015, NRC 2005, Fingas 2014). The COREXIT dispersants themselves are biodegradable (George-Ares and Clark 2000, NRC 2005, Fingas 2014), but no information was found regarding the NOKOMIS products. In general, biodegradation will take place over a matter of weeks to years and may never be complete based upon the type of oil spilled, the microbial community present and a number of environmental factors (Fingas 2014, NRC 2005). The application of dispersants may affect the biodegradation rate, but removing the oil from the surface of the ocean and causing the rapid dilution of the resultant oil droplets in suspension is their intended purpose. Although the information about biodegradation rates are interesting, it does not address a potential impact to ESA listed species under NMFS jurisdiction at this time.

Effects on Listed Species

Cetaceans

There are nine cetacean species and distinct population segments (DPS) analyzed by the action agencies in this consultation. Two are toothed whales (Sperm whale and SRKW) while the seven remaining are baleen whales. Although several of these species are very uncommon off the California coast (e.g. Western North Pacific Gray whales, Carretta *et al*., 2016a), at least one of these species or DPSs may be found in the action area during the entire year, and the second BA (USCG and EPA 2015) presents useful information on locations and timing. We present our analysis of the cetaceans as a group because the exposure scenarios and potential impacts are similar, before analyzing some specific indirect effects.

The action agencies determined that these species may overlap in time and space with pre-approved dispersant applications. We analyzed the potential impacts from dermal exposure, ingestion, baleen contact and inhalation and aspiration as well as impacts to the cetaceans' prey base and from vessel and aircraft operations.

There is little new information available regarding the effects of dispersants or dispersed oil on cetaceans. The majority of available information has been properly analyzed by the action agencies (USCG and EPA 2015). Direct effects to whales from dispersant or dispersed oil are expected to be insignificant or discountable because there is a low probability of dispersants being sprayed onto the whales with the incorporation of the protections detailed earlier (e.g. no spray buffer, protected species observers on aircraft and vessels), their thick epidermis is thought to protect against absorption (DWH NRDA Trustees 2016), the behavior of the cetaceans (i.e. frequent swimming and diving especially during daylight hours when applications take place), as well as the nature of the proposed dispersants themselves.

The dispersants proposed in the CDP are water soluble. Therefore, in the unlikely event that a whale is sprayed, the dispersants are not likely to remain on a listed cetacean except for a very short time. They are likely to make any oil encountered less sticky to the cetaceans (Lessard and DeMarco 2000, Claireaux *et al.* 2013) and may help to minimize observed impacts such as oil sticking to dolphins during the DWH spill (Dias 2017, DWH NRDA Trustees 2016). The potential genotoxic and cytotoxic effects following the 24 hour exposure scenario of skin fibroblast cells to the COREXIT dispersants and dispersed oil presented in a newer study by Wise (2014) are unlikely to occur in a field scenario, and cytotoxic impacts are noted by Judson *et al.* (2010) as a typical response of cells to xenobiotics. The most likely scenario is that of a cetacean surfacing in an oil slick that has been sprayed with dispersant and that the dispersant/dispersed oil mixture would be washed off the whale as it swam through the area or dived again.

Dispersed oil may be less sticky than undispersed oil (Lessard and DeMarco 2000, Claireaux *et al.* 2013) because of the micelle structure of dispersed oil droplets and, for the baleen whales, any oil taken into the whale's mouths during feeding may be less likely to foul their baleen. Just as uncontaminated water is ejected during feeding, water with dispersed oil would be rapidly ejected compared to the observed time for clearing oil fouled baleen with running water (70% within 30 minutes and 95% within 24 hours – Geraci 1990 in USCG and EPA 2015). This should reduce the ingestion of oil and lower the time whales are exposed to oil. Geraci (1990 in USCG and EPA 2015) calculated that 150 gallons of oil would need to be ingested by an adult whale to cause deleterious effects. As presented in the second BA (USCG and EPA 2015), Goldbogen et al. (2007) calculated the potential oil intake by a fin whale feeding in a spill zone still contaminated with 1 ppm hydrocarbons (Bejarano *et al.*, 2013) to be approximately 18 gallons per day. Therefore reducing oil concentrations to this level or lower and preventing prolonged exposure times would help prevent potential ingestion impacts to baleen whales.

While it is speculated that the direct application of dispersants onto a cetacean would cause inflammation of sensitive membranes such as on the eyes or mouth, it is known that volatile hydrocarbons cause this impact to marine mammals (Geraci and St. Aubin 1988, Geraci 1990 in USCG and EPA 2015). By mitigating exposure to volatile hydrocarbons, dispersant use could minimize this impact. No spray buffers reduce the likelihood of direct effects from dispersants to a discountable level.

The purpose of dispersant use is to minimize exposure time to animals on the water's surface such as the listed cetaceans (NRC 2013, 2005, 1998). Benefits may occur from the use of dispersants on an oil slick by reducing the inhalation of oil fumes and ingestion of oil because of the rapidly declining concentrations of oil following dispersion (NRC 2013, 2005, Curd 2011).

Bottlenose dolphins experienced lung damage due to oil vapors as a result of the DWH spill (Smith *et al.* 2017, Kellar *et al*. 2017) as did Killer Whales from the Exxon Valdez oil spill (Matkin et al., 2008). Inhalation of oil fumes or aspiration of aerosols containing oil molecules is a particular threat to cetaceans because they lack the physical structures that filter air taken into the lungs, they exchange 80-90% of their lung volume at a time, and they may hold their breath for extended periods while they dive, allowing for elevated absorption of hydrocarbons onto the lung tissue and into the blood (Takeshita *et. al*., 2017, DWH NRDA Trustees 2016). This type of damage is likely responsible for the increased incidence of lung disease, bacterial pneumonia and reproductive failures found in stranded dolphins during and following the DWH spill (Venn-Watson *et al*. 2015, Colegrove *et al*. 2016, Schwacke *et al*. 2014, DWH NRDA Trustees 2016). COREXIT 9500 was found to cause damage to human and mice lung cells before the body compensated with anti-inflammatory reactions via anti-oxidant production (Lin et al. 2015), but studies specific to marine mammals were not located. This further illustrates the importance of the exposure prevention actions (e.g. wildlife spotters, no spray policies and buffers) agreed to by the action agencies that make the likelihood of a whale being directly sprayed discountable.

As touched upon earlier in discussing dispersants, some zooplankton as well as the larval life stages of some fish species are expected to be impacted by chemically dispersed oil at an increased level compared to physically dispersed oil in the treated footprint of an oil slick, although environmentally realistic exposure times are an important factor not properly considered in some of the experimental designs (Adams *et al.* 2014, Almeda *et al.* 2014, 2013b, Fern *et al.* 2015, Fingas 2014, Incardona *et al.* 2013, Lee *et al.* 2011, Mearns *et al.* 2001, Ortman *et al.* 2012, Prince 2105, Rico-Martinez *et al.* 2013, NRC 2005, Bejarano *et al.* 2014b, Clark *et al.* 2001, Frantzen *et al.* 2015, Georges-Ares and Clark 2000). This impact occurs because the dispersants rapidly force greater amounts of soluble aromatics and PAHs into the water column and oil droplets at sizes that may be consumed by the zooplankton and larval species (NRC 2005, Fingas 2014, Carls *et al.* 2008, Fuller *et al.* 2004). While some invertebrates may bioaccumulate PAHs or hydrocarbons through direct consumption of the droplets, trophic transfer of dispersed oil was not found in experiments by Wolfe *et al*. (2001, 1999, 1998) and vertebrate organisms such as fish and marine mammals have the ability to metabolize and depurate (i.e. eliminate) them (Wolfe *et al.* 2001, 1999, Stein 2010, Varanashi *et al*., 1989).

This increased impact to some species in the water column, targeted or incidentally consumed by feeding cetaceans, is expected to be brief and only in the immediate vicinity of the dispersant application (Varela 2006, Bejarano *et al.*, 2014b, BenKinney *et al.*, 2011). This is minor in comparison to the distribution of the prey and whales, and may potentially be a smaller area than that impacted by a large volume, untreated spill (Carls *et al.*, 2008). Many of the prey species of cetaceans occupy portions of the water column much deeper than will be impacted by dispersant applications and therefore are not expected to be significantly affected. Studies have shown that zooplankton will rapidly recolonize an impacted area (Varela *et al.* 2006, Abbriano *et al.* 2011, Symons and Arnott 2013, NRC 2005). The DWH NRDA Trustees (2016) noted there was not any apparent system-wide population crashes to monitored fish or water column invertebrate species, despite the substantial short-term loss to the water column food web, from the oil spill and dispersant application. Based upon these factors, impacts to zooplankton for actions covered under the CDP are insignificant to baleen whales.

As noted previously, dispersant application will increase the mass of PAHs and other hydrocarbons in the water column up to approximately 10-20m deep for a few hours (Bejarano *et al.* 2014b, 2013, BenKinney *et al.* 2011, DWH NRDA Trustees 2016) to levels that could be a concern for adult fish which do not leave the impacted area (Mearns *et al.* 2001). However, fish species that are prey for the baleen whales such as the Humpback whale DPS's are highly mobile and distributed deeper in the water column than just the top 10-20 meters. It is unlikely that mobile schools of prey fish such as Pacific herring, Northern anchovy, or mackerel will be exposed for prolonged periods of time to dispersants or dispersed oil. As discussed previously in the latter paragraphs of the Effects of the Action section, the toxicity of the dispersed oil to exposed adult fish is likely to be no worse than that of naturally dispersed oil found in the upper water column. Additionally, dispersing a surface slick that can serve as a reservoir of oil droplets that undergo natural dispersion as the slick expands should prevent a prolonged oil exposure event that results in potentially problematic oil concentrations over a longer time and greater surface area and/or volume of water (Carls *et. al.*, 2008). Therefore, the potential effects to baleen whales from impacting their forage fish species is insignificant.

Western North Pacific Gray whales feed on benthic amphipods, often at depths of 50 to 60 m along the continental shelf (Weller 2010). Benthic species at these depths will not be exposed to dispersants or dispersed oil in problematic concentrations. Sperm whales feed on deep dwelling species of cephalopods and fish (NMFS 2010b) often several hundred meters deep. Dispersants and resulting dispersed oil from preapproved applications in the CDP will not be present at these depths to affect these prey species. SRKWs, observed in the action area generally from January to March, feed almost exclusively on salmonids, preferably full grown Chinook salmon, but also take some other species such as quillback rockfish (NMFS 2008). As discussed in a later section, salmonids present in the action area off of California are not expected to be impacted by the action. The other fish species occasionally taken by SRKWs are either at depths not expected to be impacted by preapproved dispersant applications (i.e. rockfish species from 3 – 200 nm from shore) or are mobile, widespread and numerous enough to not be impacted as discussed earlier. Therefore the impacts to prey species on these whale species are discountable and/or insignificant.

The action agencies determined that vessel and aircraft operations may affect but were not likely to adversely affect the ESA listed whale species. Vessel strikes on whales have been documented in the action area (Carretta *et. al*., 2017). In order to mitigate for this effect, the action agencies altered their project description to incorporate several protective practices as described under the Proposed Action section of this letter (e.g. protected species observers with wildlife spotting as their primary duty, minimum 100 yard in-water buffer, maximum speed of 10 knots when marine mammals are observed in the area, etc.). The proposed actions are expected to reduce the likelihood of vessel strikes to a discountable level.

Noise from vessels or aircraft in the response also presents a potential concern for cetacean species but the threat is difficult to quantify. Potential impacts include altering important behavioral patterns, physiological effects such as hearing impairment or stress, and masking critical acoustic cues, and the results of these range from no effect to potentially significant effects on the fitness of marine mammals and their habitat, depending on the context and scale of the noise exposures (Southall et al. 2007, NOAA 2016). Most observations of marine mammal responses to anthropogenic sounds have been limited to short periods, and included the cessation of feeding, resting, or social interactions. Given the many variables involved and complex interaction with sources and animals (i.e., overlap that varies over time, space, and frequency), it has been difficult to link specific behavioral responses to specific sound sources (Southall et al. 2007). More recent controlled exposure studies have illustrated these connections and discerned the importance of more nuanced contextual factors such as the distance of the sound source or the behavioral state of the animal (Southall et al. 2016, Dunlop et al. 2017). Although ship noise may result in negative behavioral, physiological, or auditory effects to cetaceans, it is uncertain whether there are consequences at either individual or population levels. More serious effects are more likely in areas of high whale use or important habitat overlaps with areas of heavy ship traffic and large vessels, such as shipping lanes.

There are only two vessels in California outfitted to spray dispersants and they are limited to use on small spills within short transit distances from their ports (CDP 2008). The largest ship is 65' long (Holmes, pers. com., 2018) and the payloads of the vessels are only 1,000 and 20,000 gallons respectively, limiting their usefulness over the 96 hour preapproval window. There may be a second vessel serving as a spotting platform, or this may be done through air support. The open ocean environment of the action area, the transient nature of dispersant applications and the limited potential use of vessels for dispersant application over the 96 hour preapproval window makes the likelihood of noise impacts from the proposed action discountable.

Harassment of cetaceans by aircraft and vessels is expected to be discountable due to the protective practices adopted by the action agencies and the open ocean environment of the action area which allows cetaceans to move as they please. Should vessels be applying dispersant to a surface slick and listed cetaceans surface or otherwise appear in the immediate area, any accidental harassment they experience would be beneficial by minimizing contact with the oil slick and its associated vapors.

Guadalupe Fur Seals

Relatively little is known about Guadalupe fur seals compared to other fur seal species because they were assumed to be extinct by the late 19th century due to hunting. It is difficult to reconstruct their former range because hunters did not distinguish between Northern and Guadalupe fur seals in their harvest records (Aurioles-Gamboa 2015) but satellite-tagged and stranded Guadalupe fur seals have been found along the entire U.S. west coast and as far as 700 nmi west of California, although most of the population occurs outside of U.S. waters (Carretta *et al*. 2017). During the nonbreeding season from September to May, they are largely at sea foraging. Pups are born between June and August, mostly at Guadalupe Island in Mexico, but there is also a small breeding colony on the easternmost part of the San Benito Islands (also in Mexico). Additionally, since 2008, individual adult females, subadult males and between one and three pups have been observed annually on San Miguel Island at the northern end of the Channel Islands in U.S. waters (NMFS-AFSC, unpublished data). The second B.A. (USCG and EPA 2015) correctly notes that they may be found throughout the preapproval zone although they are expected to be in the southern part of the action are more frequently.

A population estimate of approximately 20,000 individuals was made from direct counts between 2008-2010 in Mexico (Carretta *et al.*, 2017, Aurioles-Gamboa, D. 2015). When at sea, Guadalupe fur seals are presumed to be mostly solitary. Preferred hunting grounds have not been identified.

Guadalupe fur seals depend upon their thick pelage for thermoregulation (Aurioles-Gamboa, D. 2015), but they need to take swims during the heat of the day to protect themselves from heat exhaustion when they are on shore in tropical habitats (NMFS AFSC, undated). As a pinniped with thick pelage, it is expected that exposure to an oil spill could result in impacts to their thermoregulatory performance. When in cooler waters, this compromise could be a threat to their survival.

Dispersants such as those proposed for preapproval were designed to remove oil from the surface waters quickly, thus limiting the exposure of Guadalupe fur seals and other vulnerable animals (e.g. birds, sea otters) to the effects of an oil spill through dermal, inhalation, or ingestion (via grooming) impacts (NRC 2013, 2005). In this capacity the application of a dispersant in the preapproval zone is expected to be a benefit to the Guadalupe fur seal by preventing exposure of their fur, respiratory system and irritable membranes (e.g. eyes) to the oil and fumes, or by rendering the oil less sticky to the Guadalupe fur seals' pelage.

However, it is unknown if accidental overspray of a Guadalupe fur seal with dispersant could result in compromised thermoregulatory performance of their pelage or impacts to their respiratory systems and membranes. Only one study (Duerr *et al.* 2011) was located examining the effects of oil and dispersed oil on marine mammal fur and it is inconclusive. Although the physical structure of sea otter fur did not appear to be altered by the exposures in the experiment, the extractable hydrocarbons results were labeled as preliminary and further analysis has not appeared in the literature. However the preliminary results between oil and dispersed oil were similar and may indicate that dispersed oil is no worse than undispersed oil regarding this impact. Dispersant alone was not tested.

The protective practices detailed earlier in this analysis (wildlife spotters on all vessels and aircraft, no spray zones, etc.) as well as the rarity of the Guadalupe fur seal in the preapproval zone, especially in the cooler waters north of Point Conception, make this possibility extremely unlikely to occur and thus discountable. Operational impacts to the Guadalupe fur seal (i.e. vessel strikes) are also discountable due to the protective measures adopted by the action agencies and the small size and quickness of the species.

Guadalupe fur seals feed mostly on squid species and schooling fish such as mackerel species, anchovies and sardines (Aurioles-Gamboa, D. 2015). Similar to the analysis presented for cetaceans, these species are widely distributed and often found at depths significantly deeper than dispersed oil is expected to penetrate. The impact to these prey species from dispersed oil or dispersant alone will be insignificant.

<u>Sea Turtles</u>

Four species of sea turtles may be found in the preapproval zone and action area for the CDP consultation (leatherback, loggerhead, green, olive ridley sea turtles). Only the leatherback sea turtle has designated critical habitat. The second BA correctly notes that at least one of the four species of sea turtle may be found in the action area during the entire year, although they are more frequent in the warmer waters off Southern California and, in the case of the leatherback sea turtle, the central California coast. All four species are found in Federal waters, but the green sea turtles generally tend to stay closer to shore in state waters.

Like the previously discussed cetaceans and the Guadalupe fur seals, the four species of sea turtles breathe air. Therefore, exposure scenarios for dispersant applications and dispersed oil are similar. The listed sea turtles may be exposed through dermal contact, inhalation, and ingestion and could experience other impacts through effects to their prey base and from vessel and aircraft operations.

There is limited data available regarding the impact of dispersants or dispersed oil to sea turtles. Similar to the analysis for cetaceans and the Guadalupe fur seal in the previous sections, the application of dispersants to an oil slick is expected to benefit sea turtles by reducing the amount of oil on the surface that could stick to them or irritate sensitive membranes such as their eyes, reducing the amount of oil that could be ingested by them, and reducing oil fumes that may be inhaled by them. Average turtle dives last 5-30 minutes and longer dives may last for more than an hour for leatherback sea turtles (Hochscheid 2014) allowing for oil compounds in their lungs time to be absorbed into their blood streams. Recent information generated for the NRDA process for the DWH oil spill clearly shows oiled turtles absorbed PAHs from oil via ingestion and inhalation based on gastrointestinal and lung data (Ylitalo *et al.*, 2017) including a Kemp's ridley sea turtle with an esophagus full of oil. Sea turtles are known to ingest petroleum, perhaps due to mistaking oiled detritus as prey or indiscriminate feeding (Camacho *et al*. 2013), and even very lightly oiled sea turtles recovered during DWH had ~50% occurrence of ingestion (DWH NRDA Trustees 2016). Ylitano *et al.* (2017) examined 492 sea turtles, but found limited data on exposure to dispersants. DOSS (dioctyl sodium sulfosuccinate – a dispersant component) levels were below quantification except in the oil in the esophagus of the aforementioned heavily oiled sea turtle. This indicates that dispersants were either not used in the vicinity of these oiled turtles before they died, or that the dispersant and or dispersed oil was not bioavailable or bioaccumulated by the turtles. This latter hypothesis is in agreement with the research of Wolfe *et al.* (2001, 1999, 1998) which found negligible trophic transfer of petroleum hydrocarbons from invertebrates to vertebrates and that depuration of petroleum hydrocarbons form both vertebrates and invertebrates increased when dispersant was used. This is likely due to the micelle structure of the dispersed oil molecule being absorbed to/by the dispersants and not bioavailable. When this information is considered in conjunction with the rarity of the four sea turtle species in the preapproved application area, the likelihood of direct adverse impacts from dispersants is insignificant.

Impacts to the forage resources of the four sea turtle species is discountable. Green sea turtles are primarily herbivorous but also consume sessile and mobile invertebrates (Lemons et al., 2011). They primarily use resources in shallow, nearshore waters outside of the preapproval zone where any dispersed oil is expected to be diluted to the point it is not detectable or problematic. Olive Ridley sea turtles are pelagic and omnivorous. They are known to dive up to 150m deep to forage on benthic invertebrates. Loggerhead sea turtles found in the action area are typically pelagic juveniles and they are rare off the coast of California except during certain warm water oceanographic conditions. Loggerheads mostly prey on benthic invertebrates, although they also consume some fish and plants. Pelagic red crabs are a favorite prey species. They forage between 0-100m in depth.

Dispersants and dispersed oil from preapproved surface applications in the preapproval zones are not likely to be transported below 10-20 m deep in significant concentrations (NRC 2005, 1998, Bejarano *et al.*, 2014b, BenKinney *et al.*, 2011, DWH NRDA Trustees 2016) leaving much of the forage zones unexposed. Applications in the preapproval zones are unlikely to enter state waters at concentrations that may impact sea turtles or their forage species. Therefore effects to the prey resources of green, loggerhead and Olive Ridley sea turtles are insignificant.

Leatherback sea turtles are the species most likely to be found in the cooler waters north of Southern California. They prey upon scyphomedusae species and their critical habitat designation is based upon eddies and oceanic front areas that produce aggregations of brown sea nettles such as along the central California coast. Little is known about the potential impact of dispersants or dispersed oil to jellyfish species, or to brown sea nettles in particular. One study was conducted following the DWH oil spill examining the impact of Louisiana sweet crude oil on two related scyphozoan species, but this study was unfortunately conducted with exposure durations that are unrealistic to a surface application dispersed oil scenario (16 hour and 6 day exposures) and only the nominal concentration of the whole oil was calculated. Nonetheless, it is interesting to note that the two scyphozoan species showed different tolerances to oil pollution. This means that it cannot be assumed that jellyfish in the same species class will react similarly to dispersed oil or dispersant. In general, jellyfish species seem to be very tolerant of marine conditions with compromised water quality conditions and are found in many urbanized nearshore areas, in increasing numbers, where some petroleum contamination is very likely (Purcell 2012).

In order to add an additional level of protection to leatherback sea turtles and their designated critical habitat, the action agencies agreed to add a minimal horizontal no-spray buffer of 100m to observed aggregations of brown sea nettles even without direct observation of a leatherback sea turtle. As discussed earlier, spill specific variables are likely to increase the size of the buffer. The application of this no spray buffer makes the likelihood of adversely affecting the leatherback sea turtle's prey availability or its designated critical habitat discountable.

The action agencies also determined that operational impacts of oil spill response (i.e. vessel and aircraft operations) could affect the four sea turtle species, but that this was unlikely to result in an adverse effect. No information was found to indicate that aircraft operations affect sea turtles. The use of dedicated wildlife spotters on vessels (and in the aircraft for aerial applications), a minimum 100-yard buffer to be maintained between any vessel operations and sea turtles, and a maximum vessel speed of 10 knots if a sea turtle is observed in the area, reduces the likelihood of colliding with or otherwise impacting the animals to a discountable level.

Abalone

Two species of ESA listed abalone may be found in the action area, black and white abalone, but only the white abalone are found within the preapproval zone. The second BA (USCG and EPA 2015) correctly notes that black abalone only live and spawn in the intertidal and shallow subtidal zones at depths of 6m of less. There is nearly three nautical miles between black abalone and their designated critical habitat and the preapproval zone. Any application of dispersant in the preapproval zone and subsequent dispersed oil is expected rapidly dilute and be nondetectable in black abalone habitat. Potential impacts to black abalone and their designated critical habitat are therefore discountable.

The second BA (USCG and EPA 2015) correctly states that white abalone may be found along the coast of California from Point Conception south to Punta Abreojos, Baja California. They are found at depths between 5-60m, but their habitat (open low relief rocky reefs and boulders) is patchy and therefore so is their distribution. Overharvesting has resulted in their remnant populations being found between 30-60m deep with the highest densities at depths of 40-50m (Butler *et al.*, 2006, Steinhoff *et al.*, 2012).

Generally, surface applications of dispersant to an oil slick and resultant dispersed oil stay in the upper 10-20 m of the water column due to temperature and density gradients (NRC 2005, 1998, DWR NRDA Trustees 2016) and aerial application monitoring during DWH showed concentration at 10m depth were consistent with background concentrations (BenKinney *et al.*, 2011). The same temperature and density gradients are expected to keep viable abalone larvae below this depth during their 3-10 day larval stage. Given the depths of known white abalone habitat and the physical restrictions on potential exposure, adverse effects to white abalone from this action are discountable.

<u>Fish Species</u>

Fifteen fish species are listed under the ESA and found within the action area. A significant amount of information regarding the toxicity of oil, dispersed oil, and the preapproved dispersants alone to fish has already been presented in this document. As noted earlier, dispersant applications to surface oil slicks (the proposed action in the preapproval zone) may increase the toxicity of the upper water column to many invertebrates and larval life stages of fish. This is due to increased dissolved hydrocarbons and the production of more small oil droplets that could be consumed by these species. The level of potential impact is highly variable based upon the substance being dispersed, the presence/absence of vulnerable species and life stages, and the amount of weathering that the spilled oil has already experienced. It is important to remember that many of the oil components that cause acute impacts in the water column (soluble, low molecular weight hydrocarbons and some PAHs) will have already entered the water or volatilized by the time the first dispersant applications occur. Thus the impacts to water column resources will already be occurring. The DWH NRDA Trustees (2016) found that even thin sheens of undispersed oil were extremely toxic to early life stage fish and invertebrates. They also noted potential impacts to several juvenile fish species from DWH oil exposure for reduced growth, immunosuppression and swim performance. In each case, the experiments looked at longer term exposures (1-2 weeks, 4 days and 24 hours respectively) of the juveniles that may be prevented through dispersion of the spilled oil.

During and after DWH, federal and impacted state agencies analyzed more than 8,000 seafood samples including fish, shrimp, oysters, and crabs (Ylitalo *et al*. 2012, Fitzgerald and Gohlke 2014). The samples were tested for seafood safety concerns (*i.e.* edible tissues only), but the results are informative because this large data set found PAHs and dispersant analogs (DOSS) only at low levels or the contaminants, if present, were below detection limits. Multiple fish species from difference trophic levels were tested including red snapper, grouper, and tilefish. The results indicate that the significant oiling and persistence of oiling during DWH, which was much longer than a spill that may be addressed through the preapproval authorities of the proposed action, did not result in the bioaccumulation of PAHs into the fish. This is not surprising because fish are known to metabolize and depurate PAHS (Stein 2010, Varanasi *et al*. 1998).

The ESA listed fish species evaluated do not occur in the preapproval zone as larval species and thus are not particularly vulnerable. The Scalloped hammerhead shark gives live birth to between 1-41 pups in nearshore areas (Miller *et al.*, 2014). Oceanic whitetip sharks have litters of 1-14 live pups (Miller and Klimovich 2016) while giant manta rays give birth to only one pup every two to three years (Young *et. al.*, 2018). Eulachon spawn in fresh water, often in tidal portions of rivers, and the juveniles inhabit estuaries and near shore areas until large enough to swim offshore (NMFS 2017). Green sturgeon spawn in freshwater tributaries of San Francisco Bay (i.e. the Sacramento River) and the juveniles rear in the San Francisco Bay and Delta for several years until moving into the marine environment (NMFS 2002). ESA listed salmonids (three Chinook and two Coho salmon ESUs and five steelhead DPSs) enter the Pacific ocean as juveniles from freshwater spawning streams and rivers in all six area contingency planning areas along the California coast. Given the reproductive strategies of the listed fish species, the chances of adversely affecting their larvae or juvenile lifestages via dispersant applications in preapproval zone is discountable.

Juveniles and adults of these species typically utilize water column depths or locations that are also unlikely to encounter dispersants or dispersed oil at problematic concentrations in addition to being highly mobile so that they may easily leave an impacted area. Scalloped hammerhead sharks may be found in waters south of Point Conception, but they are seldom found in waters cooler than 22^0 C. They are infrequent in California state waters and are more prevalent during El Niño events when ocean temperatures rise (Miller *et al*. 2014). Although known for schooling behavior, this has not been reported in the action area and any Scalloped hammerhead sharks will likely be solitary or in pairs. They are known to occur over continental and insular shelves, and adjacent deep waters at depths of 450 m or more where they frequently prey of benthic organisms such as rays (Miller *et al.* 2014). They have been observed pursuing yellowtail and tuna at shallower depths (LA Times, 2015). Concentrations of dispersant or dispersed oil will be virtually nondetectable at depths in the preapproval zone where Scalloped hammerhead sharks may be found. In nearshore areas, any dispersed oil will be sufficiently diluted from the action area and below 10-20 m so that their preferred prey is not expected to be impacted by the proposed action. Given their mobility, infrequent presence in the action area and tendency to be deep in the water column, the chances of adversely affecting Scalloped hammerhead sharks via dispersant application is discountable.

Miller and Klimovich (2016) presents data on giant manta rays. Southern California waters represent the northern most range of giant manta ray habitat and drift gillnet fishery bycatch data indicates they are only found in low numbers during El Niño events. They utilize large portions of the water column, from feeding in shallow waters (< 10m) to descents of 200-450m in depth in association with the thermocline and prey location. Giant manta rays are filter feeders. Their diet consists of mostly zooplankton although some studies state they also consume small and medium sized fish. Concentrations of dispersant or dispersed oil will be virtually nondetectable at most depths in the preapproval zone where giant manta rays may be found (Bejarano *et al.*, 2014b, 2013, BenKinney *et al.* 2011). In nearshore shallow areas, any dispersed oil will be sufficiently diluted from the application site so that they and their preferred prey is not expected to be impacted by the proposed action. Given their mobility, infrequent presence in the action area, ability of prey resources to avoid or recover from potential impacts and use of a large range of the water column, the chances of adversely affecting giant manta rays via dispersant application is discountable.

Young *et. al.* (2018) presents data on oceanic whitetip sharks. This species is a highly mobile, pelagic species that generally remains offshore in the open ocean. They generally occupy warmer waters near the surface at the outer continental shelf, preferring to be over waters greater than 600 feet deep. They have also been noted to dive to nearly 500 foot depths. While their range may extend to southern California waters, distribution of the species appears to be concentrated in more tropical waters further south. West coast based U.S. drift gillnet fisheries did not record oceanic whitetip sharks in the observed catches from 1990-2015. They are subject to bycatch in other fisheries outside of the action area. This shark feeds primarily on bony fishes and cephalopods but has been known to consume sea birds, other sharks, rays, marine mammals (including scavenging) and garbage. As analyzed earlier, populations of these prey species are not expected to be significantly affected by the use of oil spill dispersants and minimum horizontal, no-spray buffer of 100 meters (328 feet) from observed congregations of fish and rafting flocks of birds, as proposed in the second BA (USCG and EPA 2015) should add further protection. Given their mobility, infrequent presence in the action area, ability of prey resources to avoid potential impacts and use of a large range of the water column, the chances of adversely affecting oceanic whitetip sharks via dispersant application is discountable.

As noted in the second BA (USCG and EPA 2015), green sturgeon adults and subadults reside primarily in coastal marine waters at depths of 100m or less with the majority of time spent between 20-70 m in nearshore waters, bays and estuaries, particularly Willapa Bay, Grays Harbor and the Columbia River Estuary, all far removed from the action area (NMFS 2015b). They are typically found from Monterey Bay northward. Green sturgeon are primarily benthic feeders consuming mostly invertebrates and some small fish and recent data indicates they may make rapid ascents primarily at night (Erichson and Hightower 2007 in NMFS 2015b), presumably following the diel vertical migrations of some prey species.

Designated critical habitat for the green sturgeon within the action area consists of coastal U.S. marine waters within 110 m depth (60 fathoms or 360 feet) from Monterey Bay to the California/Oregon border. Specific primary constituent elements include maintaining migratory corridors, water quality with adequate dissolved oxygen levels and low contaminant concentrations and sufficient food resources such as shrimp, clams, crabs, anchovies, etc. The action agencies determined that approximately 9% of designated critical habitat occurs within the preapproval zone from San Francisco Bay north to the Oregon border.

Approximately 91% of the habitat expected to be routinely occupied by the green sturgeon is outside of the preapproval zone. Within this area and the area that does overlap, green sturgeon are expected to be found primarily along the benthos. As discussed previously, that means that the preapproved surface applications are unlikely to mix to the depths where green sturgeon are found due to temperature and salinity gradients commonly found in marine waters (NRS 2005, 1998). Although green sturgeon may ascend to shallower depths, typically at night, dispersant applications are limited to daylight hours and monitoring shows that dispersed oil rapidly dilutes to low or background concentrations within hours of application (Bejarano *et al.*, 2014b, 2013, BenKinney *et al.* 2011). Areas that are as shallow as potential mixing depths (10-20m) are found close to shore, several nautical miles away from the preapproval zone. Therefore green sturgeon are unlikely to be exposed to dispersants or dispersed oil in problematic concentrations.

Prey resources of the green sturgeon share the same potential exposure scenario and are therefore unlikely to be significantly exposed. In the event of some exposure in the water column that leads to impacts, research and monitoring has shown that invertebrate populations are expected to rapidly recover from impacts (Varela *et al.* 2006, Abbriano *et al*. 2011, Symons and Arnott

2013, NRC 2005). Therefore impacts to the prey resources of green sturgeon are not expected and the overall potential for adverse effects to the species is insignificant.

Only 9% of the designated critical habitat for green sturgeon falls within the preapproval zone and any application of dispersants will only result in temporary water quality impacts on the scale of minutes to hours (Bejarano *et al*. 2014b, 2015, BenKinney *et al*. 2011). Access to migratory corridors will not be affected and impacts to prey resources will be minor and transient and not expected to result in take of green sturgeon. Therefore potential adverse effects to designated critical habitat are discountable.

Eulachon consume plankton in their marine life stage. Although information from specifically studying eulachon is scarce, fisheries bycatch data indicates that they have a large vertical distribution between 10-500 m with most taken around 100 m in depth. They are frequently taken in groundfish and ocean shrimp fisheries at near-benthic depths (NMFS 2017). Like many planktivorous species, their movements are likely part of a diel vertical migration pattern as they follow their prey up in the water column at night.

Similar to the analysis presented for green sturgeon, adverse effects to eulachon from the proposed action are expected to be discountable because the fish themselves are expected to be deep in the water column when preauthorized dispersant applications take place. Their prey resources are unlikely to be significantly impacted because they are mostly found below the mixing layer and footprint for dispersed oil and monitoring shows populations of planktonic organisms have rapidly recovered from typical oil spill situations as noted previously. Therefore adverse effects to eulachon from the proposed action are discountable.

The second BA (USCG and EPA 2015) correctly identifies that the range of steelhead encompasses the entire California coast while Chinook and Coho salmon are generally found from of Monterey Bay northward. Once in the ocean, salmonids may be widely distributed in the action area and throughout the water column depending on temperature, prey availability and the presence of predators. Salmonids smolt in estuaries, entering the ocean as juveniles, and largely stay in coastal waters feeding on zooplankton and larval fish. As they grow into subadults and adults, their range and depth utilization greatly expands (Groot and Margolis 1991, Welch *et al*. 2003) as does the variety of their prey resources (e.g. anchovies, herring, etc.).

An ambient sea water study on Chinook smolts (Lin *et al*. 2009) found that the application of COREXIT 9500 to Prudhoe Bay crude oil significantly reduced the oil's lethal potency by 20 times. A subsequent freshwater study on Chinook pre-smolts found similar results (Van Scoy *et al*. 2010). These studies indicate that ESA listed salmonids in the action area may benefit from dispersant applications because the spilled oil becomes less bioavailable to these lifestages.

Similar to the other ESA listed fish species for which data has already been presented, the preapproved use of dispersants in federal waters is unlikely to result in significant impacts due to the short duration of exposure to dispersants and dispersed oil, the high mobility of salmonids in the action area, the range of depths used by salmonids, and the wide distribution and abundance of their prey species (NMFS 2015). Juvenile salmonids occupying near shore waters during the first months or years at sea are unlikely to be exposed to problematic concentrations of dispersant or dispersed oil 2-3 nmi from the application site due to dilution in the water column and advection in ocean currents. This contrasts with impacts to early life stages of pink salmon in nearshore areas from undispersed crude oil spilled there by the *Exxon Valdez* in Alaska which are well documented (Bue *et al*. 1998, Heintz *et al*. 1999, Rice *et al*. 2001).

There have been several studies conducted specific to salmonids due to their commercial and ecological importance. Exposing adult Chinook salmon to whole and dispersed crude oil in a freshwater experiment did not reduce their homing success or affect the number of days needed for migration (Brannon *et al*. 1986). Similar work conducted on coho salmon in marine waters had the same result (Nakatani and Nevissi, 1991). Earlier work exposed immigrating adult salmon (99% were coho salmon) to a mixture of petroleum hydrocarbons and found that the salmon did not avoid hydrocarbon concentration less than 3.2 ppm (Weber *et al*. 1981). When considered together, these three studies indicate that salmonids migrating from the ocean are unlikely to be deterred by dispersants or dispersed oil, or perhaps even undispersed oil unless it is at higher concentrations than typically found post dispersion in the ocean.

Conclusion
Based on this analysis, NMFS concurs with the USCG and EPA that the proposed action is not likely to adversely affect the subject listed species and designated critical habitats.

Reinitiation of Consultation
Reinitiation of consultation is required and shall be requested by the USCG and EPA, or by NMFS, where discretionary Federal involvement or control over the action has been retained or is authorized by law and (1) new information reveals effects of the action that may affect listed species or critical habitat in a manner or to an extent not previously considered; (2) the identified action is subsequently modified in a manner that causes an effect to the listed species or critical habitat that was not considered in this concurrence letter; or if (3) a new species is listed or critical habitat designated that may be affected by the identified action (50 CFR 402.16).

MAGNUSON-STEVENS FISHERY CONSERVATION AND MANAGEMENT ACT

Under the MSA, this consultation is intended to promote the protection, conservation and enhancement of EFH as necessary to support sustainable fisheries and the managed species' contribution to a healthy ecosystem. For the purposes of the MSA, EFH means "those waters and substrate necessary to fish for spawning, breeding, feeding, or growth to maturity", and includes the associated physical, chemical, and biological properties that are used by fish (50 CFR 600.10), and "adverse effect" means any impact which reduces either the quality or quantity of EFH (50 CFR 600.910(a)). Adverse effects may include direct, indirect, site-specific or habitat-wide impacts, including individual, cumulative, or synergistic consequences of actions.

NMFS determined the proposed action would adversely affect EFH by temporarily increasing the concentration of total petroleum hydrocarbons and PAHs in the upper water column, potentially leading to increased toxicity to some zooplankton and larval lifestages of fish (NRC 2005, DWH NRDA Trustees 2016, Barron *et al.*, 2013, Bejarano *et al.*, 2014, NMFS 2015, Incardona *et. al.*, 2014, Carls *et. al.*, 1999) that are a component of EFH. However, these impacts are expected to be brief due to rapid dilution (NMFS 2015, NRC 2005, 2003, 1998, USCG and EPA 2015, 2014, Bejarano *et al.*, 2014b, BenKinney *et. al.*, 2011) and confined to the upper 10-20m of the water column (NRC 2005, 1998, BenKinney *et. al.*, 2011) leaving a large portion of the photic zone unaffected. Studies of areas impacted by oil spills have shown that zooplankton will rapidly recolonize an impacted area (Varela *et al.* 2006, Abbriano *et al.* 2011, DWH NRDA Trustees 2016). Toxicity may also be lessened due to a decrease in bioavailability to some EFH prey resources (Tjeerdema *et. al.*, 2010, Fuller *et. al.* 2004, Lin *et. al.*, 2009, Bejarano *et. al.*, 2014b) and dispersion may prevent longer term impacts both from surface water exposures and migration of a surface slick into shallow or intertidal waters and the shoreline (Bejarano *et. al.*, 2014b, NRC 2005, 1989, Bue *et al.* 1998, Heintz *et al.* 1999, Rice *et al.* 2001, Carls *et al.* 1999). EFH Habitats of Particular Concern, such as estuaries, submerged aquatic vegetation and shallow rocky reefs, are often found in these nearshore and intertidal areas. Given that the potential adverse effects to EFH from the application of the four dispersants authorized under the CDP are expected to be temporary in nature and that the applications may result in prevention of longer term and more widespread impacts, NMFS is not providing EFH recommendations at this time. The USCG 11th District and U.S. EPA Region IX must reinitiate EFH consultation with NMFS if the proposed action is substantially revised in a way that may adversely affect EFH, or if new information becomes available that affects the basis for NMFS' EFH conservation recommendations (50 CFR 600. 920(l)). This concludes the MSA portion of this consultation.

Please direct questions regarding this letter to Joe Dillon in our Santa Rosa office at (707) 575-6093 or Joseph.J.Dillon@noaa.gov.

Sincerely,

Penny Ruvelas
for
Barry A. Thom
Regional Administrator

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In Reply Refer To:
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Subject: Informal Consultation Regarding Dispersant Preauthorization under the California Dispersant Plan

Dear Mr. Holmes and Meer:

This replies to your letter, dated November 21, 2016, requesting our concurrence on your determination that the proposed preauthorized use of dispersants within Federal waters along the California coastline "is not likely to adversely affect" listed species or critical habitat pursuant to the Endangered Species Act of 1973, as amended (16 U.S.C. 1531 *et seq.*; ESA). The Federal action under consideration is the preauthorized use of dispersants as delineated in the California Dispersant Plan (CDP). The CDP is a part of the Regional Response Team Region IX (RRT9) Regional Contingency Plan and provides a decision and action framework for the preauthorized application of dispersants to an oil spill in areas termed "Pre-Approval Zones." The Pre-Approval Zones are Federal waters 3 to 200 nautical miles (nm) from the California shoreline excluding the following areas: waters within National Marine Sanctuaries; waters within 3 nm of coastal islands; and waters within 3 nm of the California-Mexico border. According to the CDP and confirmed with discussions between our agencies, the application of dispersants in the Pre-Approval Zones have the following parameters in place and are applicable under the proposed action for this consultation:

- The dispersant products must be listed both on the National Oil and Hazardous Substances Pollution Contingency Plan Product Schedule and licensed as a State of California Oil Spill Cleanup Agent. Therefore, the dispersants used in this proposed action are limited to the following four products: COREXIT EC9527A, COREXIT EC9500A, NOKOMIS 3-AA, and NOKOMIS 3-FA.

- The dispersant application method is from aircraft or boat platforms. No subsurface applications are allowed under preauthorization.
- The type of spilled petroleum product is likely to be effectively dispersed by the four dispersant products (via the decision matrix in the CDP). Spilled diesel products are not considered for preapproved dispersant applications.
- The durational limit for preapproved dispersant application is limited to the initial 96 hours of response.
- Dispersant application is at a quantity of no greater than 5 gallons per acre, and at a one (1) to twenty (20) ratio of dispersant to spilled oil.
- Employment of Special Monitoring of Applied Response Technologies (SMART) methods are employed to determine effectiveness of dispersants on spilled oil.
- All other areas and uses of dispersants in California waters are not preauthorized, and will require RRT9 incident specific approval.

The federally listed endangered, threatened, and candidate species considered for this review include the following species that may occur or be impacted by the proposed action:

- Marsh sandwort (*Arenaria paludicola*)
- Marbled murrelet (*Brachyramphus marmoratus*)
- Western snowy plover (*Chardrius nivosus nivosus*)
- Salt marsh bird's beak (*Cordylanthus maritimus maritimus*)
- Southern sea otter (*Enhydra lutris nereis*)
- Tidewater goby (*Eucyclogobius newberryi*)
- Unarmored threespine stickleback (*Gasterosteus aculatus williamsoni*)
- Delta smelt (*Hypomesus transpacificus*)
- Short-tailed albatross (*Phoebastris albatrus*)
- Salt marsh harvest mouse (*Reithrodontomys raviventris*)
- Light-footed clapper rail (*Rallus longirostris levipes*)
- California clapper rail (*Rallus longirostris obsoletus*)
- Gambel's watercress (*Rorippa gambellii*)
- California least tern (*Sterna antillarum browni*)

There are two aspects germane to our section 7 review: effects from chemical toxicity as a result of direct exposure to the dispersants and indirect effects due to changed environmental conditions resulting from use of dispersant products.

First, to assess the effects of toxicity from direct exposure, we relied upon the information provided by RRT9 through the biological assessment (BA) and a review of the scientific literature. The information provided relied upon the best available scientific and commercial information on the acute toxicity of dispersants and chemically dispersed oil, as well as information on their fate and behavior under field conditions. Our review indicates that the selected dispersant products are toxic at high concentrations, but only at levels above what would be encountered above the application rates expected during an emergency response under a preauthorization scenario. The dispersants alone are not as toxic as when combined with oil (chemically dispersed oil). However, the majority of the toxicological effects in exposures to chemically dispersed oil are the result of exposures to oil constituents, and not to the toxicity of dispersants (Clark et al. 2001; Fuller and Bonner 2001; National Research Council 1989, 2003, 2005; Wetzel and Van Fleet 2001). The endpoints used in the analyses were various groups of species that included birds, crustaceans, fish, invertebrates, and plankton. We did not find any contradictory data or information to suggest that some aquatic dependent wildlife groups are toxicologically more sensitive than others. Therefore, we concur that direct exposure and ingestion of the subject dispersant products are not likely to adversely affect listed species.

Secondly, we assessed the indirect effects to listed species as a result changing the baseline conditions (*i.e.,* the oiled environment). We assume the baseline condition for this action is the oiled environment without the application of dispersant products. The application of dispersant products in oiled environments is expected to change the baseline condition by forcing oil into the water column to facilitate biodegradation versus remaining as a floating product or becoming emulsified. Organisms, including federally listed aquatic and aquatic dependent species, are less likely to be exposed as the area containing oil and the dispersant products will be reduced. However, short term exposure to the thicker floating oil deposits and free dispersant product, although not likely to be more toxic than the oil itself, may present new physical barriers for the federally listed species, which could alter the normal behavior of organisms. The amount of free dispersant product that may occur under the Pre-Approval Zone application parameters is not expected to present much risk to listed species. However, results of an yet-to-be published study by the Oiled Wildlife Care Network (Dr. Michael Ziccardi, pers.comm) indicates that dispersant exposures to the plumage of seabirds will affect thermal regulation by altering natural oils of the exposed organisms similar to untreated oil. We believe, based on the best available information, that exposures to any free dispersant product in the water may elicit a detectable negative response in listed bird species that may be in the area of dispersant applications.

In order to avoid any adverse effects to the listed bird species that are likely to be exposed to dispersants under this action (*i.e.*, marbled murrelets), the Environmental Protection Agency and U.S. Coast Guard have agreed to, and adopted, the following conservation measures recommended by the Service during informal consultation as part of the requirements for this action:

- Results from SMART efforts are provided to the Service within the first 24 hours of the application of dispersant.
- If logistically possible, a U.S. Department of the Interior (DOI)/U.S. Department of Commerce (DOC)-approved marine mammal/turtle and/or pelagic/migratory bird observation specialist will accompany the SMART controller observer to determine if marine animals are visible in the area.
- An updated list of DOI/DOC-approved marine mammal/turtle and/or pelagic/migratory bird observation specialists will be provided to California Department of Fish and Wildlife – Office of Spill Prevention & Response for inclusion in revisions to the CDP.
- All dispersant application aircraft will maintain a 1,000-foot (305-meter) horizontal separation from flocks of birds present on the water.

- Scenarios meeting the criteria for applications in the Pre-Approval Zones in the area from the Oregon/California border south to the Monterey County, California line will avoid applications within the initial 3 to 5 nm from shore between 24 March and 15 September. This provides a buffer for marbled murrelets who may be in the area during their breeding period.

This chemical countermeasure approval does not eliminate the need for the responders to consult with our agency on the potential for adverse effects to federally listed species or the potential for adverse modification to federally designated critical habitat from the emergency oil spill response as a whole. We have reviewed the materials forwarded to this office on the dispersant products and conducted some independent literature review. We concur with your determination that federally listed species and critical habitat are not likely to be adversely affected by the conditional preauthorization to use products within RRT9, based on implementation of the aforementioned conservation measures.

This precludes the need for further consultation on this action as required under Section 7 of the Endangered Species Act of 1973, as amended. Should the project be modified or new information indicate endangered species may be affected, consultation should be initiated. This letter provides comments under the authority of and in accordance with provisions of the Endangered Species Act of 1973, as amended (87 Stat. 884, as amended; 16 U.S.C. 1531 *et seq*.).

Should you have further questions, please contact either Kathleen Brubaker of my staff at (707) 822-7201, or Damian K. Higgins of the Pacific Southwest Regional Office at (916) 414-6548.

Sincerely,

Stefanie K Bidasoda

for Bruce Bingham
Field Supervisor

cc:
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