
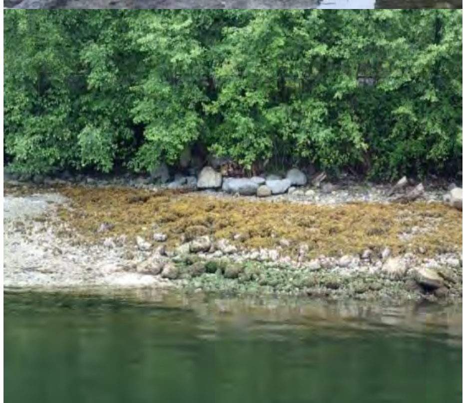
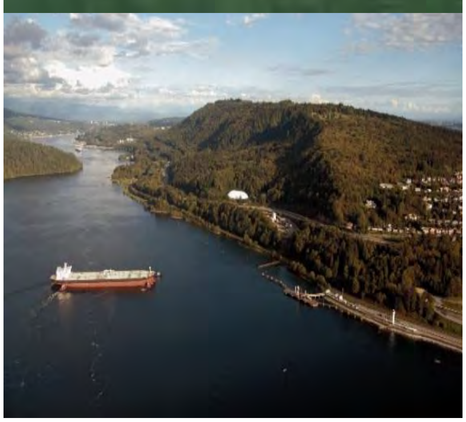

DETAILED QUANTITATIVE ECOLOGICAL RISK ASSESSMENT FOR LOADING ACCIDENTS AND MARINE SPILLS

Technical Report for the Trans Mountain Pipeline ULC

Trans Mountain Expansion Project

May 2014

Document #SREP-NEB-TERA-00002

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EXECUTIVE SUMMARY

Crude oil spills cause harm in the environment, and large oil spills have potential to cause great harm. However, technical analysis of potential accident scenarios for the proposed Trans Mountain Expansion Project (TMEP) helps to put the potential for harm into perspective. The following analysis is based conservatively upon oil spill fate and transport modeling that allows crude oil to spread and disperse in the environment without taking credit for oil spill emergency response activities, such as booming, skimming, or application of dispersants, that could mitigate the net environmental effects of an accidental oil spill.

Operational spills at the Westridge Marine Terminal (WMT), should they occur, will be mitigated through the use of protective booming at the WMT and around vessels being loaded. The protective booming reduces the chance of spilled oil interacting with the broader environment. As shown in this report, small oil spills (*e.g.*, 10 m^3 of Cold Lake Winter Blend (CLWB)) that remain confined within the protective boom and are quickly remediated would have very minor environmental effects in terms of exposures to fish and wildlife. In the unlikely event of a credible worst case (CWC) spill (*i.e.*, 160 m^3 of CLWB of which 32 m^3 is assumed to escape from the protective boom), and that some of the crude oil were to escape from the protective boom, there would be greater environmental effects. However, technical analysis indicates that the potential for mortality of fish or harm to developing fish eggs and embryos is very low. The potential for harm to wildlife caused by inhalation of hydrocarbon vapours is also very low. Some birds and semi-aquatic mammals would likely be oiled, and they could potentially die as a result of that oil exposure, but the numbers would likely be small, and population-level effects would be unlikely. Several kilometres of shoreline would likely be oiled, but the level of oiling would generally be light, and biological effects would be moderate with recovery expected within two to five years. Hydrocarbon deposition to subtidal sediments and effects on subtidal sediment communities would be very slight.

The chronic effects of residual polycyclic aromatic hydrocarbons (PAH) compounds near the WMT would be moderate, with the greatest effects likely to be observed in the intertidal zone, with bioaccumulation of PAHs by intertidal organisms. The predicted total polycyclic aromatic hydrocarbons (TPAH) concentrations in intertidal biota and sediments would not be expected to cause chronic adverse effects to highly exposed ecological receptors such as shorebirds or sea ducks.

This technical analysis of oil spill effects and recovery at the WMT is supported and validated by the findings of studies conducted following accidental damage to a pipeline in Burnaby in 2007, as a result of which approximately 100 m^3 of crude oil entered Burrard Inlet via storm drains near the WMT.

The results of the comparison between observed and measured effects in Burrard Inlet resulting from the 2007 pipeline rupture, and the predictions of the detailed quantitative ecological risk assessment (DQERA) for a CWC loading accident at WMT, provides the following validation of the ecological risk assessment:

- While the 2007 spill of 100 m^3 included a timely spill response and mitigation, the observed dispersion of the oil in 2007 was similar to that predicted by the oil spill modelling used in the DQERA for the hypothetical unmitigated spill of 32 m^3.

- The observed effects of surface oiling on fish and wildlife after the 2007 spill were consistent with, and similar to those predicted by the DQERA (*i.e.*, no fish kill, and limited effects from direct oiling on birds and marine mammals).

- The predictions of the DQERA for the extent and intensity of shoreline oiling are very similar to the shoreline oiling and effects observed after the spill in 2007.

- The observed recovery of shoreline biota is consistent with predictions provided in the DQERA (*i.e.*, less than 2 years in lightly oiled areas, and 2-5 years in more heavily oiled areas).

- The measured hydrocarbon and PAH concentrations in surface water provide good alignment with the predictions of the DQERA.

- When compared, the total parent PAH concentrations in mussel, crabs, and clams tissue predicted by the DQERA for the WMT CWC scenario were slightly higher or within the range of levels measured following the 2007 release. This comparison suggests that Exposure Point Concentrations (EPC) values for mussel, crabs and clams tissue PAH concentrations used in the DQERA to evaluate food chain effects are reasonable.

Marine transportation accidents have the potential to release larger volumes of crude oil. An accident scenario based upon the powered grounding of a vessel at Arachne Reef in the Gulf Islands was evaluated, and consideration was given to the release of crude oil from a single tank (*i.e.*, 8,500 m^3 of CLWB) or a CWC spill of 16,500 m^3 of CLWB resulting from damage to two tanks within the vessel. Both the smaller and the CWC spill volumes have qualitatively similar potential to cause adverse environmental effects, although the smaller spill would generally be expected to cause less extensive and less severe effects.

Despite the large volume of oil represented by the CWC spill at Arachne Reef, the potential for mortality of fish was found to be modest. Only one small area was identified as experiencing this potential, which was presumed to be the result of onshore winds driving crude oil into shallow water with wave action. Higher potential was identified for dissolved PAHs to harm developing fish eggs and embryos, with predicted maximum 24-hour average TPAH concentrations between 1 and 10 µg/L affecting an area of 1,962 km^2 in Haro Strait and part of Juan de Fuca Strait. While this is clearly a large area of habitat, it represents only a portion of the 12,249 km^2 of marine habitat within the regional study area (RSA). The predicted TPAH exposure is transient, and affects predominantly surface water layers. In order for this exposure to cause effects on developing fish eggs and embryos, they would have to be present at the same time and place, and in a sensitive developmental stage (*e.g.*, the first 24-hours of development). Therefore, while biological effects of TPAH exposure are possible, they are not certain to occur and would not affect all species or all developmental stages. As a result of the multiple factors that could influence the outcome, it is concluded that effects on developing fish eggs and embryos are possible, but not likely to result in effects on fish at the population level.

The potential for the CWC spill at Arachne Reef to cause harm to wildlife receptors as a result of the inhalation of vapours was found to be low. Oiling of wildlife receptors (particularly birds and furred mammals) was identified as a greater risk, with about 520 km^2 of marine habitat predicted to experience exposure to a surface oil slick with thickness greater than 10 µm at some time within 15 days of the spill. A larger area would experience exposure to surface sheens less than 10 µm thick, but these thin sheens have low potential to cause harm to wildlife receptors. Certainly thousands of marine birds and potentially hundreds of semi-aquatic mammals like otters could be exposed to oiling, and if so, many of these would die as a result of that exposure. However, the potential for exposure would vary seasonally and by species in the context of the actual timing, location and size of any accidental oil spill. Importantly, the predicted area of effects is modest in comparison with the total available habitat within the RSA, indicating that the effects of an oil spill on dispersed populations would not necessarily be catastrophic.

Other marine mammals such as pinnipeds and cetaceans are more tolerant of exposure to crude oil than marine birds and furred mammals. While there is potential for harm to populations of these species, experience at other locations suggests that the potential is low, and that harm is usually limited to a few highly exposed individuals. In the case of the endangered southern resident killer whale population, effects on an individual (*i.e.*, death or injury) would represent an effect at the population level. Such an outcome is unlikely, however, due primarily to the low probability of an accident that would result in a

CWC spill, and secondarily to the relatively low exposures (as detailed in this report) resulting from inhalation or ingestion of hydrocarbons in the event of an accident.

The CWC spill at Arachne Reef would have conspicuous effects on shorelines, and it is estimated that approximately 650 km of shoreline, out of approximately 4,400 km of shoreline within the RSA, would experience oiling. Different shoreline types vary in their capacity to retain oil, but much of the shoreline within the RSA is rock, which is impermeable and has low retention capacity. Very little of the shoreline is high-energy boulder or cobble beach with high penetration potential and which could sequester oil. Although there would be harm to intertidal communities exposed to crude oil and shoreline cleanup activities, these shorelines would generally be expected to recover within two to five years.

The chronic effects of residual PAH compounds following the CWC spill at Arachne Reef would vary according to location and the degree of exposure. The greatest effects would likely be observed in the intertidal zone, with bioaccumulation of PAHs by intertidal organisms. The predicted TPAH concentrations in intertidal biota and sediments would be sufficient to cause chronic effects on highly exposed receptors such as shorebirds, sea ducks and herons, at heavily oiled locations during the first year following the accident. However, continued weathering and natural recovery processes, assisted by shoreline cleanup activities, would reduce this potential to a sublethal level at most locations within one to two years.

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DEFINITIONS AND ACRONYM LIST

Definition/Acronym	Full Name
AIRA	Aleutian Islands Risk Assessment
Avoidance	a means to prevent a potential negative effect through routing/siting of the project, changes to project design or construction timing
BBL	Barrel, a unit of measure for oil, equal to 158.987 litres
BC	British Columbia
BC CDC	British Columbia Conservation Data Centre
BC MCA	British Columbia Marine Conservation Analysis
BSD	Blue Sac Disease
BTEX	Benzene, Toluene, Ethylbenzene and Xylenes
CCME	Canadian Council of Ministers of the Environment
CEA Act	*Canadian Environmental Assessment Act, 2012*
CEA Agency	Canadian Environmental Assessment Agency
CEPA	*Canadian Environmental Protection Act, 1999*
CLWB	Cold Lake Winter Blend
Compensation	a means intended to compensate unavoidable and potentially significant or unacceptable effects any may consist of offsets (no net loss), research, education programs, and financial compensation (considered only when all other options have been exhausted)
COPC	Chemical(s) of Potential Concern
COSEWIC	Committee on the Status of Endangered Wildlife in Canada
CPCN	Certificate of Public Convenience & Necessity
CWS	Canadian Wildlife Service
dibit	diluted bitumen
DFO	Fisheries and Oceans Canada
DNV	Det Norske Veritas
DQERA	Detailed Quantitative Ecological Risk Assessment
DWT	Dead Weight Tons

Definition/Acronym	Full Name
EBA	EBA Engineering Consultants Ltd., operating as EBA, A Tetra Tech Company
Element	a technical discipline or discrete component of the biophysical or human environment identified in the NEB Filing Manual.
EPC	Exposure Point Concentration
EPP	Environmental Protection Plan
ERA	Ecological Risk Assessment
ESA	Environmental and Socio-economic Assessment
EVOS	Exxon Valdez Oil Spill
EVOSTC	Exxon Valdez Oil Spill Trustee Council
GIS	geographic information system
HHRA	Human Health Risk Assessment
IBA	Important Bird Area
Indicator	a biophysical, social, or economic property or variable that society considers to be important and is assessed to predict Project-related changes and focus the effects assessment on key issues. One or more indicators are selected to describe the present and predicted future condition of an element. Societal views are understood by the assessment team through published information such as management plans and engagement with regulators, public, Aboriginal, and other interested groups.
ISGOTT	International Safety Guide for Oil Tankers and Terminals
KMC	Kinder Morgan Canada Inc.
MAH	Monocyclic Aromatic Hydrocarbon
MCTS	Marine Communication and Traffic Services
Measurement Endpoint	one or more 'measurement endpoints' are identified for each indicator to allow quantitative or qualitative measurement of potential Project effects. The degree of change in these measurable parameters is used to characterize and evaluate the magnitude of Project-related environmental and socio-economic effects. A selection of the measurement endpoints may also be the focus of monitoring and follow-up programs, where applicable.
Mitigation measures	mean measures for the elimination, reduction or control of a project’s adverse environmental effects, including restitution for any damage to the environment caused by such effects through replacement, restoration, compensation or any other means.
NEB	National Energy Board
NEB Act	*National Energy Board Act*
NOAA	National Oceanic and Atmospheric Administration
PAH	Polycyclic Aromatic Hydrocarbon

Definition/Acronym	**Full Name**
PMV	Port Metro Vancouver
RSA (Regional Study Area)	The area extending beyond the Local Study Area boundary where the direct and indirect influence of other activities could overlap with project-specific effects and cause cumulative effects on the environmental or socio-economic indicator.
PQERA	Preliminary Quantitative Ecological Risk Assessment
SARA	*Species At Risk Act*
SOA	Special Operating Area
Stantec	Stantec Consulting Ltd.
Supplemental studies	Supplemental studies - studies to be conducted post submission of the application to confirm the effects assessment conclusions and gather site-specific information for the implementation of mitigation from the Project-specific environmental protection plans
TEX	Toluene, Ethylbenzene, Xylenes
the Project	the Trans Mountain Expansion Project
TLM	Target Lipid Model
TMEP	Trans Mountain Expansion Project
TSS	Total Suspended Solids
Trans Mountain	Trans Mountain Pipeline ULC
US EPA	United States Environmental Protection Agency
US NFWF	United States National Fish & Wildlife Foundation
US NRC	United States National Research Council
WCMRC	Western Canada Marine Response Corporation
WMT	Westridge Marine Terminal
YVR	Vancouver International Airport

1.0 INTRODUCTION

1.1 Project Overview

Trans Mountain Pipeline ULC (Trans Mountain) is a Canadian corporation with its head office located in Calgary, Alberta. Trans Mountain is a general partner of Trans Mountain Pipeline L.P., which is operated by Kinder Morgan Canada Inc. (KMC), and is fully owned by Kinder Morgan Energy Partners, L.P. Trans Mountain is the holder of the National Energy Board (NEB) certificates for the Trans Mountain pipeline system (TMPL system).

The TMPL system commenced operations 60 years ago and now transports a range of crude oil and petroleum products from Western Canada to locations in central and southwestern British Columbia (BC), Washington State and offshore. The TMPL system currently supplies much of the crude oil and refined products used in BC. The TMPL system is operated and maintained by staff located at Trans Mountain's regional and local offices in Alberta (Edmonton, Edson, and Jasper) and BC (Clearwater, Kamloops, Hope, Abbotsford, and Burnaby).

The TMPL system has an operating capacity of approximately 47,690 m^3/d (300,000 bbl/d) using 23 active pump stations and 40 petroleum storage tanks. The expansion will increase the capacity to 141,500 m^3/d (890,000 bbl/d).

The proposed expansion will comprise the following:

- Pipeline segments that complete a twinning (or "looping") of the pipeline in Alberta and BC with about 987 km of new buried pipeline.
- New and modified facilities, including pump stations and tanks.
- Three new berths at the Westridge Marine Terminal in Burnaby, BC, each capable of handling Aframax class vessels.

The expansion has been developed in response to requests for service from Western Canadian oil producers and West Coast refiners for increased pipeline capacity in support of growing oil production and access to growing West Coast and offshore markets. NEB decision RH-001-2012 reinforces market support for the expansion and provides Trans Mountain the necessary economic conditions to proceed with design, consultation, and regulatory applications.

An Application has been made pursuant to Section 52 of the *National Energy Board Act* (*NEB Act*) for the proposed Trans Mountain Expansion Project (referred to as "TMEP" or "the Project"). The NEB will undertake a detailed review and hold a Public Hearing to determine if it is in the public interest to recommend a Certificate of Public Convenience and Necessity (CPCN) for construction and operation of the Project. Subject to the outcome of the NEB Hearing process, Trans Mountain plans to begin construction in 2016 and go into service in 2017.

Trans Mountain has embarked on an extensive program to engage Aboriginal communities and to consult with landowners, government agencies (*e.g.,* regulators and municipalities), stakeholders, and the general public. Information on the Project is also available at www.transmountain.com.

While Trans Mountain does not own or operate the vessels calling at the Westridge Marine Terminal, it is responsible for ensuring the safety of the terminal operations. In addition to Trans Mountain's own screening process and terminal procedures, all vessels calling at Westridge must operate according to rules established by the International Maritime Organization, Transport Canada, the Pacific Pilotage Authority, and Port Metro Vancouver. Although Trans Mountain is not responsible for vessel operations, it is an active member in the maritime community and works with BC maritime agencies to promote best

practices and facilitate improvements to ensure the safety and efficiency of tanker traffic in the Salish Sea. Trans Mountain is a member of the Western Canada Marine Response Corporation (WCMRC), and works closely with WCMRC and other members to ensure that WCMRC remains capable of responding to spills from vessels loading or unloading product or transporting it within their area of jurisdiction.

Currently, in a typical month, five vessels are loaded with heavy crude oil (diluted bitumen) or synthetic crude oil at the terminal. The expanded system will be capable of serving 34 Aframax class vessels per month, with actual demand driven by market conditions. The maximum size of vessels (Aframax class) served at the terminal will not change as part of the Project. Similarly, the future cargo will continue to be crude oil, primarily diluted bitumen or synthetic crude oil. Of the 141,500 m^3/d (890,000 bbl/d) capacity of the expanded system, up to 100,200 m^3/d (630,000 bbl/d) may be delivered to the Westridge Marine Terminal for shipment.

In addition to tanker traffic, the terminal typically loads two to three barges with oil per month and receives one or two barges of jet fuel per month for shipment on a separate pipeline system that serves Vancouver International Airport (YVR). Barge activity is not expected to change as a result of the expansion.

1.2 Context of this Detailed Quantitative Ecological Risk Assessment

This Detailed Quantitative Ecological Risk Assessment (DQERA) is being provided as supplemental information to the Section 52 Application that was submitted by TMEP to the NEB on December 16, 2013. This supplementary task is focused on providing additional detailed evaluation of the acute and chronic toxic effects associated with hypothetical crude oil spills to the marine environment including:

- spills resulting from loading accidents at the Westridge Marine Terminal (WMT)
- spills resulting from a vessel accident during marine transportation.

This DQERA evaluates the potential risk to ecological health resulting from hypothetical spills at the specific locations noted above. The nature of the hypothetical spills (location and release volume) has been based on failure/risk analysis completed by Det Norske Veritas (DNV 2013). The Ecological Risk Assessment is based on the results of crude oil spill fate and transport modelling completed by EBA Engineering Consultants Ltd. (EBA 2013). The crude oil spill scenarios presented here consider both a credible worst case spill and a smaller spill for each location. In this case, Trans Mountain has selected Cold Lake Winter Blend (CLWB) diluted bitumen as a representative crude oil for the purposes of evaluating effects of hypothetical spills.

1.3 Scope of the DQERA

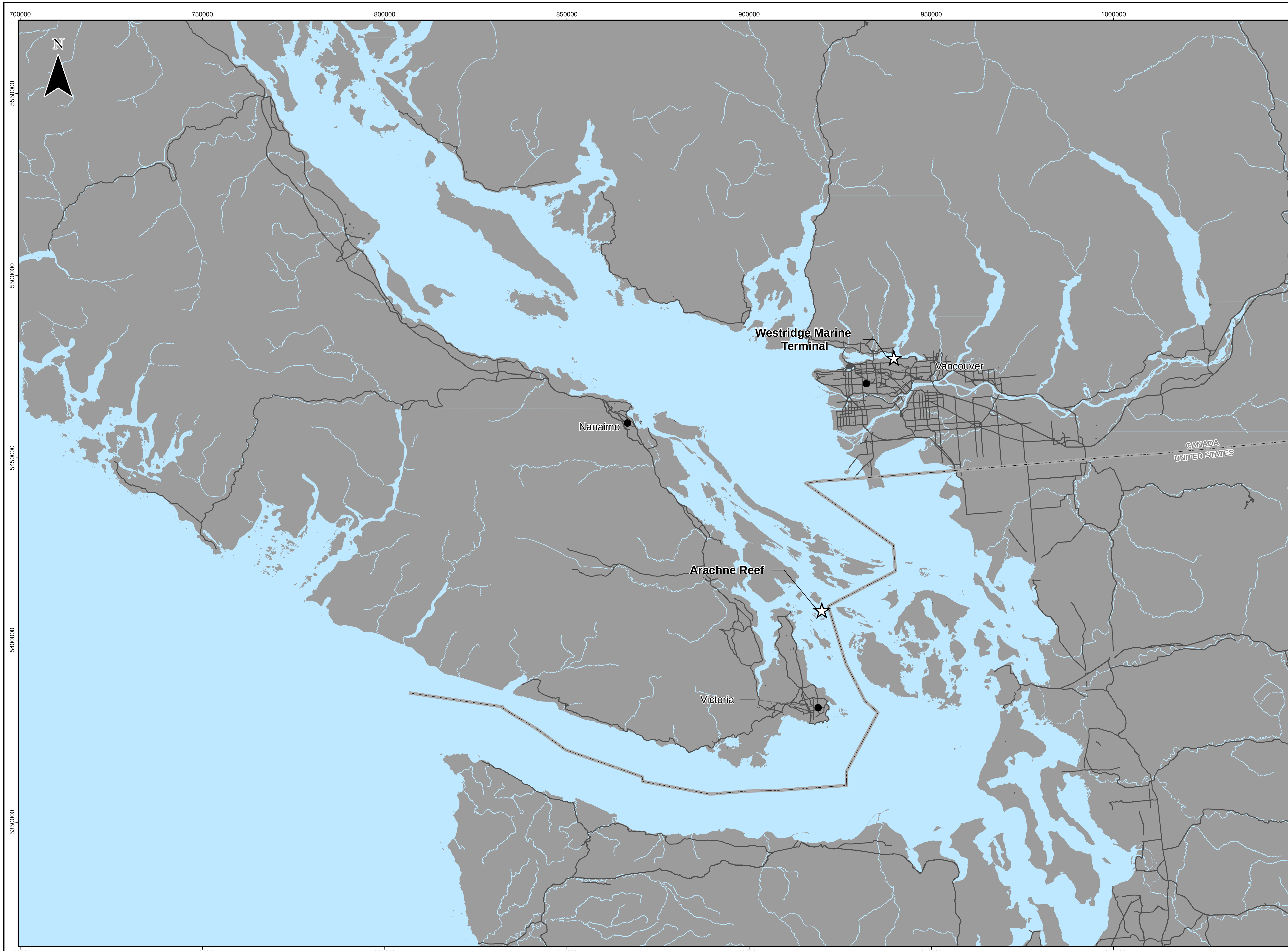
This DQERA evaluates the toxicologically-induced changes in health of ecological receptors that might be exposed to chemicals of potential concern (COPC) from hypothetical spills of CLWB at two specific locations including the WMT in Burrard Inlet and Arachne Reef, located in the Gulf Islands of the Strait of Georgia (Figure 1.1).

TRANS MOUNTAIN

FIGURE: 1.1

Location of Hypothetical Spills

WESTRIDGE MARINE TERMINAL AND ARACHNE REEF



Hypothetical Spill Location

Canada/United States Border

Road

Regional Study Area

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Stantec

Although there is no reason to believe that there are any errors associated with the data used to generate this product or in the product itself, users of these data are advised that errors in the data may be present.

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1.4 Objectives

The specific objectives of the DQERA are to:

- Evaluate the potential health risks and toxicological effects to representative ecological receptors from exposure to hypothetical spills of a crude oil.
- Consider both a credible worse case (CWC) spill, and a smaller spill at each location.
- Support Human Health Risk Assessment (HHRA, Intrinsik 2014), as required.
- Further inform the Environmental and Socio-Economic Assessment (ESA) document and support the NEB Application filing process.

1.5 Regulatory Standards

The NEB Filing Manual does not outline specific requirements or methodologies to evaluate spills and malfunctions. However, on September 10, 2013, the NEB issued “Filing Requirements Related to the Potential Environmental and Socio-Economic Effects of Increased Marine Shipping Activities, Trans Mountain Project”, which provided guidance on a number of factors including the selection of locations where accidents and malfunctions should be evaluated, and the spill scenarios that should be developed, and the factors that should be included in these assessments. Therefore, the general methodologies utilized in this DQERA follow the accepted guidance published by National standards and regulatory authorities, including the Canadian Council of Ministers of Environment (*i.e.,* CCME 1996, 1997) and the United States Environmental Protection Agency (US EPA 1998).

Preliminary Quantitative Ecological Risk Assessments (PQERA) were completed previously in support of the Application and are presented in Volumes 7 and 8B of the Application. The PQERA studies presented an effects assessment consistent with the approach used for the Aleutian Islands Risk Assessment (AIRA, ERM 2011) and discussed the range of potential effects to various ecological resources by considering the probability of exposure to predicted surface oil slicks and affected aquatic and shoreline habitats within the study area.

While using spill scenarios considered in the PQERA, this DQERA goes beyond the scope of the previous assessments to evaluate the toxicological effects of exposure to COPC in various environmental media which is predicted by detailed deterministic modelling of specific but representative spills.

1.6 Organization of the DQERA Report

This DQERA for hypothetical crude oil spills is organized into the sections described in Table 1.1.

Table 1.1 Organization of the DQERA Report

Report Section	Content
Executive Summary	A non-technical summary of key findings to assist the reader in quickly understanding the most important aspects of this DQERA.
Section 1 – Introduction	An introductory section that provides an overview of the Project and describes the context, scope and objectives of the DQERA in the Environmental and Socio-economic Assessment (ESA) process. Also introduces regulatory standards used in the DQERA.
Section 2 – Consultation and Engagement	A description of the regulatory and stakeholder consultation and engagement process.
Section 3 – Ecological Risk Assessment Framework and Methodology	A description of ERA framework and methods used in the DQERA.
Section 4 – DQERA for Crude Oil Spills from Hypothetical Loading Accidents at the Westridge Marine Terminal	This section provides the results of the potential environmental effects resulting from spills occurring at the WMT. Effects are evaluated for a range of receptors and from exposure to chemical components of crude oil which are dissolved in the water column, deposited on shorelines and entrained in underlying sediment. Both short term (acute) and long term (chronic) effects are evaluated.
Section 5 – DQERA for Crude Oil Spills from Hypothetical Marine Transportation Accident at Arachne Reef	This section provides the results of potential environmental effects resulting from spills occurring along the marine transportation route near Arachne Reef in the Gulf Islands. Effects are evaluated for a range of receptors and from exposure to chemical components of crude oil which are dissolved in the water column, deposited on shorelines and entrained in underlying sediment. Both short term (acute) and long term (chronic) effects are evaluated.
Section 6 – Chronic Effects Assessment for Ingestion of Hydrocarbons by Mammals And Birds	This section evaluates the level of wildlife exposure to residual hydrocarbons via ingestion pathways and the effects of direct contact between wildlife and residual crude oil in the aftermath of an accidental oil spill.
Section 7 – Certainty and Confidence	Certainty and confidence in the predictions made in the ecological risk assessment.
Section 8 – Summary and Conclusions	Summary of the potential environmental effects arising from potential accidental oil spill scenarios described in the report.
Section 9 – Closure	Closure statement.
Section 10 – References	A list of references cited throughout the DQERA.

2.0 CONSULTATION AND ENGAGEMENT

Trans Mountain and its consultants have conducted a number of engagement activities to inform Aboriginal communities, stakeholders, the public and regulatory authorities about the approach to assessing potential environmental and socio-economic effects of the Project, and to seek input throughout the Project planning process.

2.1 Public Consultation, Aboriginal Engagement and Landowner Relations

Trans Mountain has implemented and continues to conduct open, extensive and thorough public consultation and Aboriginal engagement programs. These programs were designed to reflect the unique nature of the Project as well as the diverse and varied communities along the proposed pipeline and marine corridors. These programs were based on Aboriginal communities, landowner and stakeholder groups' interests and inputs, knowledge levels, time and preferred methods of engagement. In order to build relationships for the long-term, these programs were based on the principles of accountability, communication, local focus, mutual benefit, relationship building, respect, responsiveness, shared process, sustainability, timeliness, and transparency.

Feedback related to marine transportation that was raised through various Aboriginal engagement and public consultation activities including public open houses, ESA Workshops, and one-on-one meetings, is summarized below and was considered in the development of this technical report, and the description of effects from marine transportation spills in Volume 8A AND Volume 7 – TR7-1, the Ecological Risk Assessment of Westridge Marine Terminal Spills. The concerns that were identified through this process can be encapsulated as a general concern about effects of crude oil spills on water, fish and wildlife.

In addition, concerns related to crude oil spills in the marine environment (*e.g.,* spill response times and proportion of product that can be recovered from the water and from shorelines; WCMRC equipment locations and response capacity; liability regime in Canada in the event of a spill; and ability to fund the cost of a spill) were also raised and detailed information on marine spills is provided in Volume 8A.

The full descriptions of the public consultation, Aboriginal engagement and landowner relations programs are located in Volumes 3A, 3B and 3C, respectively. Section 3.0 of Volume 8A summarizes the consultation and engagement activities that have focused on identifying and assessing potential issues and concerns related to spills during marine transportation which may be affected by the construction and operation of the Project. Information collected through the public consultation, Aboriginal engagement and consultation programs for the Project was considered in the development of this technical report, and the assessment of marine transportation spills in Volume 8A.

2.2 Regulatory Consultation

Regulatory consultation with the applicable subject matter experts was conducted to present and discuss the proposed assessment methods and approaches for the various ERA studies. Consultation was completed in two phases with various expert groups including 1) consultation on the selection of ecological receptors for the ERA studies, and 2) consultation on the proposed oil spill fate modelling and methods for assessing hypothetical spills.

Consultation on the selection of Key Indicators for the ESA, and receptors for the ERA, was completed in conjunction with the other ESA disciplines during a meeting held on April 16, 2013. The TMEP project team met with representatives from Environment Canada including members of the Canadian Wildlife Service (CWS) and the Environmental Assessment Office, as well as one external advisor to CWS. No specific comments or concerns were identified by the regulators during the consultation sessions, or through subsequent follow-up discussions.

A consultation session was also held on May 24, 2013, in Vancouver, with representatives of Transport Canada, Envrionment Canada, Fisheries and Oceans Canada and Port Metro Vancouver.

3.0 ECOLOGICAL RISK ASSESSMENT FRAMEWORK AND METHODOLOGY

3.1 Overview

This DQERA builds on the results of the Preliminary Quantitative Ecological Risk Assessments (PQERA) which were completed to evaluate the potential effects of loading accidents at the Westridge Marine terminal, and for accidental releases at various locations along the marine transportation route. The results of the PQERA are summarized in the NEB Application; Volume 7 (Section 8 for loading accidents) and Volume 8 (Section 5.6.2.2, 5.6.2.3 and 5.6.2.4 for hypothetical spills at the Strait of Georgia, Race Rocks, and Arachne Reef (Gulf Islands) respectively. Further details are provided in the Technical Reports TR 7A-1 Ecological Risk Assessment of Westridge Marine Terminal Spills, and TR 8B-7 Ecological Risk Assessment of Marine Transportation Spills.

The PQERA reports presented an effects assessment consistent with the approach used for the Aleutian Islands Risk Assessment (AIRA, ERM 2011) and discussed the range of potential effects to various ecological resources by considering the probability of exposure to predicted surface oil slicks and affected aquatic and shoreline habitats within the study area. These assessments were based on stochastic oil spill modelling completed for each of four seasons including winter (January to March), spring (April to June), summer (July to September) and fall (October to December). Each set of stochastic modelling results considered season specific behaviour (wind direction and speed, temperature, etc.), trajectories, and oil fate. Potential effects were evaluated by overlaying GIS data layers containing information on biological resources, sensitive habitats and other areas of ecological importance.

The potential consequences in terms of negative environmental effects from crude oil exposure from each spill scenario were evaluated for four ecological receptor group/habitat combinations including the following:

- Shoreline and Near Shore Habitats
- Marine Fish and Supporting Habitat
- Marine Birds and Supporting Habitat
- Marine Mammals and Supporting Habitat.

These four ecological receptor groups broadly represent the range of marine resources within each study area comprising ecological resources and supporting habitat, including water, sediment and air quality. Each of the four ecological receptor groups presented a variety of habitats and/or individual receptor types of differing sensitivity to crude oil exposure. The potential ecological consequences of crude oil exposure at any given location were considered to be defined by the overlap of the likelihood of crude oil presence in the event of an accidental spill, and the sensitivity of ecological habitat or receptors that may be present at that location.

The primary focus of this DQERA is the quantification of toxicological induced changes in the health of marine ecological receptors from exposure COPC associated with hypothetical spills of CLWB resulting from a loading accident at WMT, and from a tanker accident at Arachne Reef during marine transportation, as predicted by 3-dimensional deterministic modelling of specific, but representative spills.

The DQERA was conducted according to accepted methodologies and guidance published by regulatory authorities, including the Canadian Council of Ministers of Environment (CCME 1996, 1997) and the United States Environmental Protection Agency (US EPA 1998). In addition the particular requirements of the NEB were followed (see Section 1.5).

The DQERA has followed a standard risk assessment protocol that is composed of the following steps:

- problem formulation
- exposure assessment
- hazard assessment
- risk characterization
- discussion of certainty and confidence in the predictions.

The terminology and methodology has followed the guidance provided by CCME (1996). The framework and methodology for the DQERA are described in Figure 3.1 and in the following sub-sections.

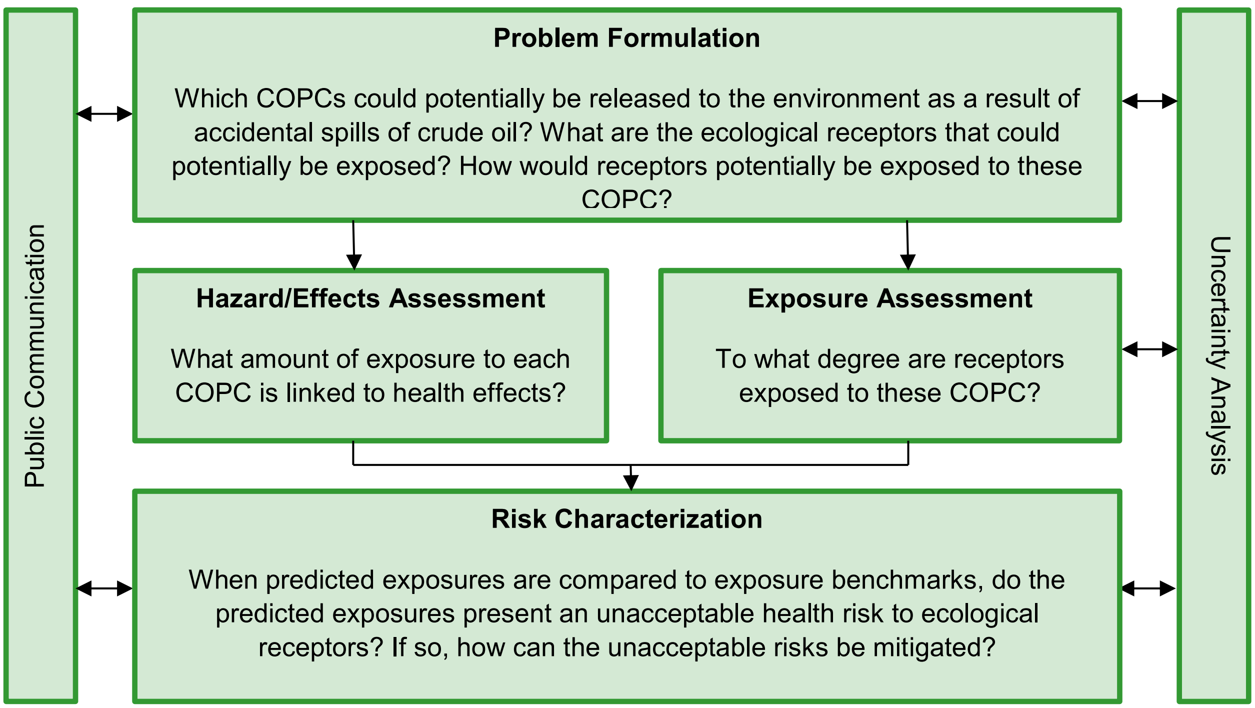


Figure 3.1 Conceptual Diagram of the DQERA Framework

3.2 Problem Formulation

The problem formulation stage is an information gathering and interpretation stage that focuses the study on areas of primary concern for the Project. Problem formulation defines the nature and scope of the work to be conducted, and enables practical boundaries to be placed on the overall scope of work, so that the DQERA is directed at the key areas and issues of concern. The assembled data provides information regarding the general characteristics of the study area, the crude oil products being considered, the identification of credible release points and release mechanisms for the crude oil, potential ecological receptors and any other specific areas or issues of concern to be addressed.

The key components of the problem formulation step include:

- characterization of the geographic areas where the DQERA is being conducted
- identification of representative crude oil products being assessed, which individual COPC are present, and mechanisms of release to the environment
- identification of exposure media and pathways
- identification and characterization of representative ecological receptors.

The outcome of these components forms the basis of the DQERA.

3.2.1 Regional Study Areas for the DQERA

The spatial boundaries of this DQERA include the geographic extent where potential exposures are expected to be measurable from hypothetical spills at each location (*i.e.,* the modelling domain for the crude oil spill trajectory model). The Regional Study Area (RSA) is described as the area of ecological relevance where exposure to COPC could potentially result from accidental spills at WMT and Arachne Reef. These areas are effectively established by the physical limits of the modelling domain for the crude oil spill modelling.

Regional Study Areas for hypothetical spills WMT and Arachne Reef have been described previously in Volume 7 and Volume 8C of the Application, respectively. A general description of the RSA for each location for the purposes of the DQERA is also summarized in Sections 4.2.1 and 5.2.2.

3.2.2 Release Scenarios Considered in the DQERA

Det Norske Veritas (DNV) was retained by Trans Mountain to conduct a marine quantitative risk assessment (QRA) of the effects of the increased tanker traffic resulting from the Project as part of the Marine Terminal Systems and Transshipment (TERMPOL) review process. The QRA examined the probability of certain types of incidents occurring en route to the marine terminal or during marine terminal transshipment operations, and the likelihood of an oil spill accident with an uncontrolled release of product. The outcome of the QRA provided the probability for accidental oil discharges of varying potential amounts of product including a credible worst case spill scenario. That in turn has provided input to oil spill modelling for selected scenarios in support of both the ecological and human health risk assessments.

The potential spill scenarios evaluated by DNV and considered in the preliminary ecological risk assessments are discussed in Volume 7 and Volume 8B of the Application. A description of the spill scenarios and release volumes for each spill location considered in the DQERA is also summarized in Sections 4.2.3 and 5.2.1

3.2.3 Selection of a Representative Crude Oil Product

A sample of the representative diluted bitumen to be evaluated in the DQERA (*i.e.,* CLWB) was provided by Trans Mountain. The sample was subsequently subjected to detailed physical and chemical analyses in order to gain an understanding of the particular hydrocarbon fractions present, as well as the individual COPCs present for the Project. CLWB was selected because it is already transported by Trans Mountain, and is expected to remain a major product transported by the new line. In addition, the diluent in CLWB is condensate (a light hydrocarbon mixture derived from natural gas liquids). As such the CLWB was considered to be a conservative choice for both the ecological and human health risk assessments. This is because the volatile and relatively water-soluble hydrocarbons associated with the condensate would present a higher level of risk due to inhalation of volatiles, and exposure to dissolved hydrocarbons than would synthetic oil, which is also used as a diluent, but contains fewer volatile and less water soluble constituents.

3.2.4 Physical Properties of Cold Lake Winter Blend

Example physical properties of CLWB are listed in Table 3.1. Note that all transported hydrocarbons will meet Trans Mountain pipeline quality specifications as outlined in TMEP Petroleum Tariff Rules (KMC 2013).

Table 3.1 Physical Properties for Cold Lake Winter Blend Diluted Bitumen

Physical Property	Units	Cold Lake Winter Blend Diluted Bitumen
Interfacial Tension	dyne/cm	42
Absolute Density @ 15°C	kg/m^3	926
Measured Relative Density @ 15°C	N/A	0.9268
American Petroleum Institute (API) Gravity @ 15°C	N/A	21.2
Closed Cup Flash Point	°C	<-35
Pour Point	°C	-33
Viscosity @ 5°C – kinematic	cSt	542.1
Viscosity @ 10°C – kinematic	cSt	371.2
Viscosity @ 15°C – kinematic	cSt	261.6
Viscosity @ 30°C – kinematic	cSt	105.9
Viscosity @ 40°C – kinematic	cSt	64.09
Viscosity @ 60°C – kinematic	cSt	28.63
Gas Equivalency Factor	m^3 gas/ m^3 liquid	86.6

Source:
Analysis performed by Maxxam Analytics, with the exception of viscosity at 5, 10, and 15°C, which were calculated from the measured values at higher temperature following ASTM D-341 by C. Brown (pers. comm. 2013).

3.2.5 Chemical Properties of Cold Lake Winter Blend

The majority of the chemical analysis carried out on the sample of CLWB was done by Maxxam Analytics, with confirmatory analysis for selected test groups carried out by Research and Productivity Council (RPC, Fredericton, New Brunswick). The following analytical packages were included:

- trace elements
- petroleum hydrocarbons
- polycyclic aromatic hydrocarbons (PAH)
- alkylated PAH
- pentachlorophenol and phenol
- volatile organic compounds (VOC)

- alkylated mono aromatic hydrocarbons (MAH).

Table 3.2 provides a full list of trace elements and organic compounds analyzed in the CLWB. Copies of the original laboratory certificates are provided in Appendix A of the Qualitative Ecological Risk Assessment of Pipeline Spills – Technical Report, Volume 7.

Table 3.2 Chemical Constituents of Cold Lake Winter Blend Diluted Bitumen

Analyte	Concentration (mg/kg)	Analyte	Concentration (mg/kg)
Metals			
Aluminum	<1	Mercury	0.021
Barium	<1	Molybdenum	5
Beryllium	<1	Nickel	46.8
Boron	1	Phosphorus	0.8
Cadmium	<1	Potassium	1
Calcium	2	Silicon	2
Chromium	<1	Silver	<1
Cobalt	<1	Sodium	12
Copper	<1	Strontium	<1
Iron	3	Sulphur	37,100
Lead	<1	Tin	<1
Lithium	<1	Titanium	1
Magnesium	<1	Vanadium	135
Manganese	<1	Zinc	<1
Sulfur Compounds			
Hydrogen Sulphide (H_2S)	<0.5	n-Propanethiol	<0.5
Carbonyl Sulphide	<0.5	Thiophene/sec-Butanethiol	2.9
Methanethiol	<0.5	Diethyl Sulphide	<0.5
Ethanethiol	1.1	Iso-Butanethiol	<0.5
Dimethyl Sulphide	1.7	n-Butanethiol	0.5
Carbon Disulphide	<0.5	Dimethyl Disulphide	<0.5
Iso-Propanethiol	2.5	n-Pentanethiol	<0.5
t-Butanethiol	<0.5	n-Hexanethiol	0.5
Methyl Ethyl Sulphide	0.9	n-Heptanethiol	<0.5
SAPA (Saturates, Aromatics, Polars, Asphaltenes)			
Saturates	318,000	Polars	398,000
Aromatics	203,000	Asphaltenes	80,000
Summary Composition			
Methane	<100	n-Butane	5,100
Ethane	<100	Iso-Pentane	31,600
Propane	400	n-Pentane	34,200
Iso-Butane	1,000		
BTEX (Benzene, Toluene, Ethylbenzene, Xylenes)			
Benzene	1,800	Ethylbenzene	470
Toluene	3,900	Xylenes	3,500
PHCs (Petroleum Hydrocarbons)			
F1 (C_6-C_{10}) – BTEX	110,000	Aliphatics >C_{21}-C_{34}	60,000
F2 (C_{10}-C_{16})	82,000	Aliphatics >C_{34}-C_{50}	23,000
F3 (C_{16}-C_{34})	260,000	Aromatics >C_8-C_{10}	<6,000
F4 (C_{34}-C_{50})	110,000	Aromatics >C_{10}-C_{12}	4,100
Aliphatics C_6-C_8	55,000	Aromatics >C_{12}-C_{16}	22,000
Aliphatics >C_8-C_{10}	20,000	Aromatics >C_{16}-C_{21}	47,000
Aliphatics >C_{10}-C_{12}	16,000	Aromatics >C_{21}-C_{34}	120,000
Aliphatics >C_{12}-C_{16}	40,000	Aromatics >C_{34}-C_{50}	77,000
Aliphatics >C_{16}-C_{21}	46,000		
SVOCs – PAHs (Semi Volatile Organic Compounds – PAHs)			
Acenaphthene	12	Fluoranthene	7.3
C_1-Acenaphthene	<4.5	C_1-fluoranthene/pyrene	75
Acenaphthylene	<4.5	C_2-fluoranthene/pyrene	200
Acridine	39	C_3-fluoranthene/pyrene	340

Table 3.2 Chemical Constituents of Cold Lake Winter Blend Diluted Bitumen

Analyte	Concentration (mg/kg)	Analyte	Concentration (mg/kg)
Anthracene	6.6	C_4-fluoranthene/pyrene	170
Benzo(a)anthracene	5.6	Fluorene	21
C_1-benzo(a)anthracene/chrysene	59	C_1-Fluorene	150
C_2-benzo(a)anthracene/chrysene	230	C_2-Fluorene	300
C_3-benzo(a)anthracene/chrysene	110	C_3-Fluorene	770
C_4-benzo(a)anthracene/chrysene	37	Indeno(1,2,3-cd)pyrene	<4.5
Benzo(b&j)fluoranthene	6.7	2-Methylnaphthalene	80
Benzo(k)fluoranthene	<4.5	Naphthalene	34
C_1-benzo(b,j,k)fluoranthene/benzo(a)pyrene	21	C_1-Naphthalene	160
C_2-benzo(b,j,k)fluoranthene/benzo(a)pyrene	37	C_2-Naphthalene	600
Benzo(g,h,i)perylene	4.8	C_3-Naphthalene	780
Benzo(c)phenanthrene	<4.5	C_4-Naphthalene	810
Benzo(a)pyrene	5.8	Phenanthrene	63
Benzo(e)pyrene	5.1	C_1-phenanthrene/anthracene	310
Biphenyl	7.3	C_2-phenanthrene/anthracene	550
C_1-biphenyl	50	C_3-phenanthrene/anthracene	660
C_2-biphenyl	84	C_4-phenanthrene/anthracene	230
Chrysene	8.6	Perylene	9
Dibenz(a,h)anthracene	<4.5	Pyrene	<13
Dibenzothiophene	44	Quinoline	<8.9
C_1-dibenzothiophene	330	Retene	43
C_2-dibenzothiophene	910		
C_3-dibenzothiophene	700		
C_4-dibenzothiophene	440		
SVOCs – Phenols (Semi Volatile Organic Compounds – Phenols)			
Cresols	<16	2,4-dinitrophenol	<43
Phenol	<8.1	2,6-dichlorophenol	<8.5
3 & 4-chlorophenol	<21	2-chlorophenol	<4.3
2,3,5,6-tetrachlorophenol	<4.3	2-methylphenol	<8.7
2,3,4,6-tetrachlorophenol	<4.3	2-nitrophenol	<43
2,4,5-trichlorophenol	<4.3	3 & 4-methylphenol	16
2,4,6-trichlorophenol	<4.3	4,6-dinitro-2-methylphenol	<43
2,3,5-trichlorophenol	<4.3	4-chloro-3-methylphenol	<4.3
2,3,4-trichlorophenol	<4.3	4-nitrophenol	<43
2,4-dichlorophenol	<6.3	Pentachlorophenol	<4.3
2,4-dimethylphenol	29		
VOCs (Volatile Organic Compounds)			
Bromodichloromethane	<150	Methyl methacrylate	<200
Bromoform	<250	Methyl-tert-butylether (MTBE)	<150
Bromomethane	<100	Styrene	<100
Carbon tetrachloride	<100	1,1,1,2-tetrachloroethane	<500
Chlorobenzene	<100	1,1,2,2-tetrachloroethane	<250
Chlorodibromomethane	<100	Tetrachloroethene	<100
Chloroethane	<100	1,2,3-trichlorobenzene	<200
Chloroform	<100	1,2,4-trichlorobenzene	<200
Chloromethane	<150	1,3,5-trichlorobenzene	<200
1,2-dibromoethane	<100	1,1,1-trichloroethane	<100
1,2-dichlorobenzene	<100	1,1,2-trichloroethane	<100
1,3-dichlorobenzene	<100	Trichloroethene	<50
1,4-dichlorobenzene	<100	Trichlorofluoromethane	<100
1,1-dichloroethane	<100	1,2,4-trimethylbenzene	300
1,2-dichloroethane	<100	1,3,5-trimethylbenzene	<2,500
1,1-dichloroethene	<100	Vinyl chloride	<50
cis-1,2-dichloroethene	<100	neo-Hexane	<100
trans-1,2-dichloroethene	<100	Methylcyclopentane	8,000
Dichloromethane	<150	Cyclohexane	10,000

Table 3.2 Chemical Constituents of Cold Lake Winter Blend Diluted Bitumen

Analyte	Concentration (mg/kg)	Analyte	Concentration (mg/kg)
1,2-dichloropropane	<100	Methylcyclohexane	10,500
cis-1,3-dichloropropene	<100		
trans-1,3-dichloropropene	<100		

Source: Analysis performed by Maxxam Analytics.
Note: Values prefaced by "<" were not detected. The numeric value denotes the laboratory reportable detection limit.

Both parent and alkylated PAHs were detected in CLWB with highest concentrations encountered for dibenzothiophene and its alkylated species followed very closely by naphthalene and its alkylated species. Alkylated species generally exhibited higher concentrations than the parent species, which is typical of petroleum products. The total PAH concentration including all parent and alkylated species is 9,436 mg/kg for CLWB (concentrations less than the detection limit were assumed to be zero, consistent with Yang *et al.* 2011). The total PAH concentration for CLWB is within the range encountered in other Alberta petroleum products, such as Wabasca Heavy (ΣPAH = 1,762 mg/kg; Yang *et al.* 2011) and Alberta Sweet Mixed Blend (ΣPAH = 10,694 mg/kg; Yang *et al.* 2011).

Non-petroleum compounds in crude oils, such as metals, are seldom of environmental concern as primary pollutants. For example, after the discharge of an estimated 160 to 340 million gallons of crude oil during the 1991 Gulf War, trace metal concentrations in oiled intertidal and sub-tidal sediments remained within the background range (Fowler *et al.* 1993 in Hugenin *et al.* 1996). Similarly, the US EPA (2011) concluded in response to a crude oil spill into the Yellowstone River in Montana that metal concentrations in the spilled oil were present only at very low levels, and as such were unlikely to pose any threat to human life or the environment. Likewise, Anderson (2006) concluded that there was no post-spill evidence of an increase in water or sediment metal concentrations at Wabamun Lake, Alberta, following a spill of Bunker "C" fuel oil and pole treating oil in 2005.

As indicated in Table 3.2, the measured concentrations of trace metals in CLWB are generally very low (<1 mg/kg), with the exception of nickel (46.8 mg/kg) and vanadium (135 mg/kg). However, these values are similar to the average crustal abundance for these elements (Ni 32 ± 4 ppm, V 106 ± 7 ppm, Hu and Goa 2008), and it is believed that these trace metals are likely to remain largely associated with the diluted bitumen following a spill. Therefore, the DQERA focuses on the environmental effects of hydrocarbons (*i.e.,* crude oil and its hydrocarbon constituents) released into the marine environment.

3.2.6 Pseudo-Component Approach for Deterministic Oil Spill Modelling

Models used in the deterministic fate evaluation (SPILLCALC and H3D) have been developed over many years to include as much information as possible to simulate the fate and effects of oil spills in a realistic manner. However, there are limits to the complexity of processes that can be modelled, as well as gaps in knowledge regarding the environment that is affected and the behavior of organisms and ecosystems. For the deterministic modelling and the DQERA, the "pseudo-component" approach of Payne *et al.* (1984) is used, such that chemicals in the oil mixture are grouped by physical-chemical properties, and the resulting component category behaves as if it were a single chemical with characteristics typical of the chemical group. These pseudo-components are meant to be representative of analytes demonstrating similar properties. For example, toluene, ethylbenzene, and xylenes (TEX), which represent a relatively narrow range of molecular weights, vapor pressures, Log K_{OW} and solubilities, have been grouped together as pseudo-component AR2. Benzene, which is typically considered alongside TEX, was segregated into another pseudo-component, AR1, due to its relatively higher vapor pressure. The pseudo-components used for the oil spill modelling of Cold Lake Winter Blend are presented in Table 3.3 below. As the basis of both the DQERA and the HHRA are analyte-specific toxicity benchmarks, the

pseudo-components are broken out again to estimate concentrations of individual analytes in the various media.

Table 3.3 Pseudo-Components Used in the Oil Spill Modelling

Pseudo-Component	Description
VOL	Volatiles
AR1	Benzene
AR2	Toluene, Ethylbenzene, Xylenes
AR3	Aromatics >C_8-C_{10}
AR4	Aromatics >C_{10}-C_{12}
AR5	Aromatics >C_{12}-C_{16}
AR6	Aromatics >C_{16}-C_{21}
AR7	Aromatics >C_{21}-C_{34}
AL1	Aliphatics C_6-C_8
AL2	Aliphatics >C_8-C_{10}
AL3	Aliphatics >C_{10}-C_{12}
AL4	Aliphatics >C_{12}-C_{16}
AL5	Aliphatics >C_{16}-C_{21}
AL6	Aliphatics >C_{21}-C_{34}
RES1	F4 (>C_{34}-C_{50})
RES2	Resins
RES3	Asphaltenes

3.2.7 Oil Spill Trajectory and Fate Modelling

Crude oil spill modelling simulations were completed by EBA to support the various ERA and HHRA studies, and to inform the oil spill response planning for the Project. The simulations were based on the hypothetical spill scenarios developed by DNV as described previously.

Two numerical models were used for the EBA study: H3D, a three dimensional circulation model calibrated and validated in the area of study, to generate surface currents; and SPILLCALC, EBA's proprietary spill model, to simulate the movement and weathering of the oil slick resulting from the spill. Oil trajectories, weathering and shore contact were computed by the modelling system, SPILLCALC and the models are run in both stochastic and deterministic modes.

Stochastic simulations were performed for a complete annual cycle to take into consideration seasonal variations in winds and currents, with hypothetical accidental releases of CLWB at the spill site being initiated every six hours of each day throughout the year. All hypothetical spill simulations were allowed to run unmitigated for up to 15 days, with between 360 and 720 individual simulations being performed for each seasonal suite. For modelling purposes, and to be conservative, no consideration was given to possible mitigation, such as oil spill response activities. In the assessment of the initial damage to the environment that would result from an oil spill. For recovery assessment, however, it was assumed that oiled shorelines would be subject to SCAT (shoreline cleanup and assessment techniques), and that oil recovery operations properly executed would accelerate the natural recovery process. Details of the stochastic modelling completed by EBA are provided in EBA (2013) Modeling the Fate and Behaviour of Marine Oil Spills for the Trans Mountain Expansion Project – TERMPOL Report, Volume 8C.

Subsequent to the completion of the stochastic oil spill modeling, two individual simulations (representing both a credible worst case and a smaller spill) from both the Westridge and the Arachne Reef spill locations were selected for more comprehensive 3-D deterministic modeling, which would compute the fate of the oil in the water column, in the air, on the water surface and stranded on shorelines. The fate of individual pseudo-components was also tracked for all media. A primary purpose of these simulations was to provide information on the potential toxic effects of the spill.

The objective of the selection process was to identify representative scenarios for each location that was realistic, while tending to be conservative from both the ecological and human health perspective. Four deterministic simulations were conducted including at the Arachne Reef site (CWC and a smaller spill) and at the Westridge Marine Terminal site (CWC and a smaller spill). A more detailed description of each scenario selected for detailed 3-D modelling by EBA is provided in Sections 4.1.2 and 5.2.1.

The airborne transport of the evaporated portion of each pseudo-component was also modelled for each spill scenario using CALPUFF. CALPUFF is an advanced, multi-layered, multi-species, non-steady-state Gaussian puff air dispersion modeling system that can simulate the effects of time- and space-varying meteorological conditions on pollutant transport and is recommended by the B.C. Ministry of Environment for long-range transport and for short-range transport in complex, non-steady state meteorological conditions found in complex terrain and coastal situations.

Time and spatially variant evaporative flux area sources were produced at hourly intervals by SPILLCALC and written to a series of files as the emission inputs for CALPUFF. Of the 17 pseudo-components, 11 were shown to readily evaporate into the air. CALPUFF then tracked the transport of these fluxes calculating the concentration and transport time to each receptor grid point, set up at ground level over hourly intervals (EBA 2013).

3.3 DQERA Technical Approach

Sections 3.1 and 3.2 provided a high-level overview of the technical approach to the DQERA. The following sections will provide further details regarding the specific approaches that are used to characterize and evaluate acute and chronic risks to a variety of ecological receptors, including water and sediment quality, shoreline habitat, marine life (algae, invertebrates, fish), seabirds and marine mammals.

3.3.1 Assessment and Measurement Endpoints

Suter (1993) defined assessment endpoints as being explicit expressions of environmental values or characteristics to be protected at a site, reflecting societal and ecological values. In practice, assessment endpoints are usually broad statements articulating the overall goals of a risk assessment. For this risk assessment, endpoints are maintenance of:

- marine plant, invertebrate and fish communities, such that productivity and usability of such resources are not diminished
- intertidal communities, such that productivity and usability of such resources are not diminished
- marine and semi-aquatic mammal populations, at levels similar to pre-spill levels
- piscivorous birds, shorebird and seabird populations, at levels similar to pre-spill levels

The information needed to deal directly with the higher-level assessment endpoints is often difficult to generate and rarely available. Therefore, measurement endpoints are used to bridge the gap.

Measurement endpoints are simpler and more clearly defined measurable responses to stressors, related to the assessment endpoints, and intended to provide a basis for assessing risk potential for the assessment endpoint. They may be defined in terms of an unacceptable level of effect to ecological receptors, such as a certain relative percent decrease in survival, growth or reproduction of ecological populations (Suter 1993). As part of a weight-of-evidence approach, one or more measurement endpoints may be used for each assessment endpoint. Measurement endpoints can also be used as a starting point in the development of follow-up or environmental effects monitoring programs. The following are the measurement endpoints considered in the DQERA:

- For aquatic community-level resources, including marine plants and fish, concentrations of COPC in water following a hypothetical spill should not exceed levels that could acutely impair the survival, growth or reproduction of a sensitive species. A sensitive species is defined as being the 5th percentile species in a species sensitivity distribution.

- For sub-tidal sediment community-level resources, including marine benthic invertebrates, concentrations of COPC in sub-tidal sediment following a spill example should not exceed levels that could chronically impair the survival, growth or reproduction of a sensitive species. A sensitive species is defined as being the 5th percentile species in a species sensitivity distribution.

- For intertidal sediment community-level resources, including algae and invertebrates, exposure to COPC arising from environmental effects of spills should not exceed levels that could impair community diversity, biomass or productivity.

- For mammalian and avian receptors, exposures to COPC arising from the environmental effects of spills should not exceed levels that could acutely impair survival, growth or reproduction.

The goal is to identify potential risks to marine biota at the community or population level rather than at the individual level. Risk assessment calculations focus on the exposure of a hypothetical individual that is maximally exposed to COPC within a defined geographic area. In actuality, not all members of a population would experience the same level of exposure, and hypothetical changes in individual health would not necessarily manifest themselves as changes in individual or population health.

3.3.2 Selection of Ecological Receptors for Acute Effects Assessment

The initial effects of an oil spill are the most conspicuous, and have the greatest potential to cause acute toxicity or mortality to individual organisms. Acute effects are considered to be those that result from direct exposure to hydrocarbons, and are manifest within a short period of time following such exposure. For additional context, the highest concentrations of volatile hydrocarbons in air, and dissolved hydrocarbons in water, are typically observed within the first 24 to 48 hours following an oil spill. The length of time that oil can remain floating on the water surface, or is available to strand along shorelines is more variable, and scales according to the characteristics of the receiving environment. The time-frame of environmental fate and transport modeling used to support this assessment reflects those characteristics, such that oil has undergone initial stranding within Burrard Inlet for 60 hours following the initiation of the hypothetical spill, and oil has undergone initial stranding in the Strait of Georgia and Juan de Fuca Strait within 10 to 15 days of the initiation of the hypothetical spills at Arachne Reef. The acute effects assessment corresponds to these time frames.

With the large number of habitats and wildlife species found in and around the RSA it is not practical to assess each individual habitat or species. After reviewing the range of habitats and species expected within the RSA, receptors were selected based on the following factors:

- Selected habitats should focus on those that are likely to be highly exposed to oiling in the event of an accidental oil spill, or which are of critical importance to species of interest.

- Indicator species should be indigenous to the area, and likely to have high exposure to COPC due to their habitat preferences and home range or residency.

- Indicator species should be representative of various trophic levels and feeding guilds in the marine ecosystem.

- Consideration should be given to indicator species having cultural (*e.g.,* traditional use), economic (*e.g.,* recreational or commercial harvest) or social (*e.g.,* species at risk) significance.

Key habitats and trophic levels within the RSA have been represented in the selection of ecological receptors. Each selected receptor is considered to be representative of other species belonging to the same guild of species or occupying a similar position in the food web. Therefore, results of the risk characterization step for a selected receptor can be used to make inferences about risk to other similar species. Using these criteria, the ecological receptors assessed in this DQERA are expected to provide adequate and conservative representation of the faunal and floral diversity in each RSA.

3.3.2.1 Identification and Selection of Habitat Types as Marine Ecological Receptors

In the event of an oil spill, whether at the Westridge Terminal, elsewhere in Burrard Inlet, in the Strait of Georgia, or in the Juan de Fuca Strait, it is reasonable to assume that floating oil will initially spread on the surface of the water, and that some or most of that oil will subsequently strand along shorelines. At the same time, some of the crude oil constituents will be subject to evaporation (giving rise to concentrations of volatile hydrocarbons in air), and will dissolve into the water column. Some crude oil may also be transported to sediments, either as a result of the sorption of dissolved hydrocarbon constituents to sinking particles in the water column, or as a result of oil weathering and interactions with biota or mineral particles resulting in the formation of oil-rich aggregates that sink. Some of the crude oil or its constituents in any of these environments will also be subject to processes of degradation, including but not limited to biodegradation and photodegradation.

Habitat types that are expected to be exposed to crude oil in the event of a spill include:

- The atmosphere, where organisms that breathe at or near the water-air interface may be exposed to potentially toxic concentrations of volatile organic substances in air.
- The surface of the water, which can be further subdivided into different habitat units depending upon water depth or proximity to land, which can determine the types and abundance of sea birds or marine mammals occupying the habitat.
- The water column, which can be exposed to crude oil and its constituents as dissolved hydrocarbons, finely dispersed droplets that will remain submerged for some appreciable time, or larger droplets or globules of oil that may be temporarily overwashed by water, but which will re-surface within a short period of time. The water column may be stratified vertically or horizontally by salinity or thermal profiles, and concentrations of crude oil and its constituents will vary in response to such stratification.
- Shorelines, which can have various characteristics (*e.g.,* grain size, slope, and degree of exposure to wave action) that will affect both the interactions of oil with the shoreline, and the types of marine and intertidal flora and fauna that would be exposed to such crude oil.
- Sub-tidal and deeper sediments may be exposed to crude oil that has sunk or become associated with sinking particles, leading to exposure of epibenthic and infaunal species.

3.3.2.2 Identification and Selection of Community-Level Marine Ecological Receptors

The primary exposure pathway for some flora and fauna may be from direct contact with a single abiotic environmental medium (*e.g.,* marine plants, invertebrates, or fish exposed to COPC in water or sediment). Therefore, toxicity benchmarks are commonly derived that relate COPC concentrations in these media (*i.e.,* water or sediment) to toxicological effects thresholds for organisms that reside in or rely on that medium. Additionally, these benchmarks are typically generated using toxicity data for not one, but many species that reside in and rely on that medium, with the intent of protecting sensitive species, and all life stages.

Key community level receptors evaluated in this DQERA include:

- Aquatic species (including algae, invertebrates and fish) exposed to dissolved hydrocarbons in the water column, with non-polar narcosis as a mode of toxic action.
- Birds and mammals exposed to crude oil on the surface of the water, leading to harmful effects on these species as a result of either hypothermia (caused by loss of insulative characteristics of fur or feathers) or ingestion of crude oil as a result of grooming or other behaviours following such exposure.
- Shoreline and intertidal communities (including eelgrass and kelp beds) exposed to crude oil as a result of the stranding of such oil along the shoreline following an accident or spill, and subsequent oil spill response or cleanup activities, with resulting harm to algal, intertidal invertebrate, or intertidal fish species.
- Benthic invertebrate and demersal fish species exposed to crude oil as a result of processes that indirectly or directly lead to the sinking of oil and its incorporation into subtidal or deepwater sediments.

3.3.3 Selection of Ecological Receptors for Chronic Effects Assessment

Following the acute phase of an oil spill, when there is potential for overt effects due to direct contact with oil or temporary high concentrations of hydrocarbon constituents in air or water, concern shifts to chronic effects of oil exposure that might arise from low-level exposure to lingering oil or PAH exposures. This chronic exposure assessment focuses on oral ingestion of hydrocarbons in food or other media (*e.g.,* sediment) and the effects that such chronic exposure may have on the health of exposed wildlife species.

Selection of wildlife species for this assessment therefore considers a number of factors, in addition to the primary consideration of whether receptors may be highly exposed to COPCs by virtue of their preferred habitats and dietary requirements. These other factors include but are not limited to considerations of traditional use by Aboriginal people, commercial and/or recreational harvest, and protections that may be extended to various species under provincial or federal legislation.

3.3.3.1 Identification of Traditional Use Species

As presented in Section 5, Volume 8B of the Application, TERA on behalf of Trans Mountain undertook the Traditional Marine Resources Use (TMRU) study for the Project. The TMRU was initiated in 2012 with the objective to determine the extent and general nature of each community's current use of marine resources, identify existing concerns and potential effects of the project on marine resource use, provide traditional knowledge information for the assessment of effects and recommend appropriate mitigation measures to address the concerns relative to traditional marine resource use.

Of the 27 marine and inlet Aboriginal communities engaged on the Project with Trans Mountain, the following 21 communities have been identified as having an interest in the Project or having interests potentially affected by the increased Project-related marine vessel traffic:

- Esquimalt Nation
- Cowichan Tribes
- Halalt First Nation
- Hwlitsum First Nation
- Pacheedaht First Nation
- Penelakut First Nation
- Semiahmoo First Nation
- Stz'uminus First Nation (Chemainus)

- Lyackson First Nation
- Malahat First Nation
- Pauquachin First Nation
- Scia'new Indian Band (Beecher Bay)
- Tsartlip First Nation
- Tsawout First Nation
- Tsawwassen First Nation
- Tseycum First Nation
- Katzie First Nation
- Kwikwetlem First Nation
- Musqueam Indian Band
- Squamish Nation
- Tsleil-Waututh Nation.

The TMRU considered various assessment indicators and measurement endpoints related to the assessment traditional marine resource use within the region, including subsistence and traditional activities such as hunting, fishing, harvesting and gathering within the marine and inter-tidal environments of the Project area.

Species of importance to Aboriginal communities for traditional and subsistence purpose that were identified through the TMRU include, but are not limited, various species of invertebrates and crustaceans (*i.e.,* barnacles, urchin, clams, mussels, scallops, oysters, octopus, prawn and crab), fish (*i.e.,* salmon, halibut, cod, dogfish, herring, red snapper, shiners, sturgeon, and euchalon), marine mammals (*i.e.,* grey and killer whales, Steller sea lion, seals, dolphins and porpoises) and waterfowl (*i.e.,* ducks and geese). Also of importance to the Aboriginal communities involved with the TRMU study were various species of seaweed, and other marine plants.

3.3.3.2 Identification of Commercial Harvest Species

The commercial fishing and aquaculture industry in coastal waters of BC contribute nearly $845.3 million and $614 million, respectively, in wholesale value to the provincial economy in 2011. The commercial fishing industry within the RSA includes salmon, groundfish (*e.g.,* Pacific hake rockfish, cod, sole, and halibut), Dungeness crab, prawn, shrimp, and herring, as well as other species to a lesser extent (Section 6, Volume 8B of the Application).

Permanent or temporary closures of a commercial fishery are managed by DFO to protect sensitive habitat areas or fish species. Several permanent fishery closures exist within the Marine RSA including, but not limited to: nine Rockfish Conservation Areas, three marine reserves and three navigational closures.

In additional to the thriving commercial fishery within the Marine RSA, there is an active aquaculture industry. The aquaculture industry is mainly located in the protected inshore waters along the east coast of Vancouver Island and the Gulf Islands where Atlantic salmon, Pacific Salmon, shellfish such as clams, oysters and scallops as well as sablefish, trout and sturgeon represent over half the aquaculture production in Canada (BC MCA 2011).

The Marine RSA also supports various recreational and tourism related industries with a focus on marine resources. Whale watching for killer whales or Orcas, as well as minke whales, humpback whale along with sea lions, porpoises and seals make use of the Marine RSA. Commercial sport fishing and recreational fishing for species including Pacific salmon, giant halibut, bottom fish (*e.g.,* rockfish, lingcod and halibut), along with shellfish (*e.g.,* clam, crab and sea urchin, squid, octopus, sea cucumber) are also active throughout the region.

3.3.3.3 *Identification of Species at Risk*

Species at risk or of conservation concern are defined as wildlife species listed in Schedule 1 of the Canadian *Species At Risk Act* (*SARA*) as extirpated, endangered, threatened or of special concern; species that are red-listed (extirpated, endangered or threatened) or blue-listed (of special concern) by the BC CDC; or listed as endangered, threatened or of special concern by COSEWIC. British Columbia has no stand-alone endangered species legislation; however, red-listed vertebrate species may be legally designated as endangered or threatened under the provincial *Wildlife Act*.

Within the two study areas, 14 species are listed as species at risk or of conservation concern (see Table 3.4).

Table 3.4 Federal and Provincial Listed Species at Risk Potentially Present within the Study Areas

Species	Relevant Distribution and Seasonal Timing	Conservation Status		
		***SARA* – Schedule 1**	**BC CDC**	**COSEWIC**
Sockeye salmon	Fry emerge in spring, rear in freshwater lakes for 1 to 3 years, and then migrate to the ocean for another 2 to 3 years before returning to their natal stream to spawn (DFO 2001, Hart 1973).	No Status	Yellow	Endangered
Coho salmon	Mature coho salmon migrate to their natal streams from October to December to spawn (DFO 2012a, DFO 2001). Juvenile coho remain in their spawning stream for 1 to 2 years before migrating to marine waters in the spring (DFO 2012a, DFO 2001).	No Status	Yellow	Endangered
Chinook salmon	Chinook salmon populations are categorized based on two major life-cycle types: stream and ocean (DFO 2001). Stream-type chinook typically spend 1 to 2 years in fresh water before migrating to marine waters, while ocean-type chinook typically spend no more than 90 days in fresh water before migrating to sea (DFO 2012a, DFO 2001). Spawning times for chinook vary among stocks. They are often referred to as “spring salmon” because they spawn earlier than other Pacific salmon species. Chinook generally migrate upstream to the middle to upper regions of large rivers in British Columbia from the spring through fall to spawn (DFO 2012a, DFO 2001, Hart 1973). These upstream migrations can be as far as 1,500 km inland (DFO 2012a). The majority of chinook salmon in British Columbia come from the Fraser River watershed where spawning occurs from August to December (DFO 2001). Fry emerge in the spring.	No Status	Yellow	Threatened
Quillback rockfish	Quillback rockfish mate from November to February and are born between March and July with a subsequent pelagic larval phase lasting 1 to 2 months (COSEWIC 2009). Quillback rockfish range from the Gulf of Alaska to southern California (COSEWIC 2009; Lamb and Edgell 2010). They occur in depths ranging from 16 to 182 m but are most common between 50 to 100 m (COSEWIC 2009, DFO 2012b, DFO 2006).	No Status	No Status	Threatened
Cassin’s auklet	Cassin’s auklet is found on islands from the Baja California peninsula to the Aleutian Islands, Alaska. The center of population is BC, where, in 1980, an estimated 2 million birds occurred in the Scott Island group and 1.1 million on Triangle Island. Wintering populations move south, frequenting waters off the continental shelf edge. Breeding primarily occurs along the coast of British Columbia. This auklet nests in shallow burrows, which the birds excavate, and also in rock crevices or under trees or logs. During the nonbreeding season, the birds spend most of their time at sea, with southern populations likely moving north and northern ones moving south to the central portion of its Pacific range. It is most abundant in waters of the continental shelf.	No status	Blue	No status

Table 3.4 Federal and Provincial Listed Species at Risk Potentially Present within the Study Areas

Species	Relevant Distribution and Seasonal Timing	Conservation Status		
		SARA – Schedule 1	**BC CDC**	**COSEWIC**
Surf scoters	Surf scoters are medium-distance migrants that are widely distributed along the entire BC coastline, especially during spring migration. The Strait of Georgia and Burrard Inlet are particularly important winter and spring staging grounds. Southward migration from inland breeding areas occurs from late August to October (BC CDC 2013) and is usually at night (Butler and Savard 1985). Large aggregations occur from a few hundred to several thousand individuals. Wintering surf scoters usually forage within 1 km of the shore (Vermeer 1981). Non-breeding habitat includes sheltered freshwater and marine bays, harbours and lagoons. At these sites, birds prefer shallow marine waters, less than 10 m deep, with substrates of pebbles and sand (Goudie *et al.* 1994, Campbell *et al.* 1990). This species rarely uses estuaries except during migration (Campbell *et al.* 1990; Savard *et al.*1998). Large numbers forage near steep shores of fjords where food resources (*e.g.,* mollusks) are abundant on submarine rocky walls (Vermeer 1981; Vermeer and Bourne 1984). Surf scoters eat aquatic invertebrates on its breeding grounds and marine mollusks in spring, fall, and winter (Savard *et al.* 1998).	No status	Blue	No status
humpback whale *Megaptera novaeangliae*	Relatively common and abundant, especially during summer and fall. Some presence year-round. Use area primarily for foraging. Numbers have been increasing in this area in recent years. Food sources include capelin, herring, and a variety of other small fishes, krill and other crustaceans. The Humpback Whale (North Pacific population) was recently reassessed and downgraded from "Threatened" to " Special Concern" under *SARA*.	Special Concern Schedule 1	Blue	Special Concern
killer whale – southern resident ecotype *Orcinus orca*	Common and regular sightings, particularly during summer and fall. RSA coincident with majority of the identified critical habitat for this species. The range of the southern resident population extends from Haida Gwaii, BC to Monterey Bay, California (COSEWIC 2008). The transboundary area between BC and Washington State in the United States, which includes the southern portion of the Strait of Georgia, the southern Gulf Islands, Boundary Pass, Haro Strait and Juan de Fuca Strait, has been designated as "critical habitat" under *SARA* for the southern resident population (DFO 2008, 2009, 2011). Some members of the population remain in the same general area in winter and spring but others seem range over much greater distances. Some have been reported as far south as California, and as far north as Haida Gwaii. Food sources include principally chinook and chum salmon (summer and fall). Little is known about the diet during the winter and spring. Other food sources include fish and cephalopods (squid). Some coho salmon. Some bottom fish, possibly ling cod, kelp greenling and sablefish.	Endangered Schedule 1	Red	Endangered
fin whale *Balaenoptera physalus*	Rare sightings in Juan de Fuca Strait. Unlikely presence, given historical distribution and preferred habitat (*i.e.,* primarily offshore). May occasionally use western-most portion of RSA for foraging. Food sources include small invertebrates, schooling fishes and squids. In North Pacific, diet is dominated by euphausiids (70%) followed by copepods (25%), with some fish (incl. herring) and squid.	Threatened Schedule 1	Red	Threatened
killer whale – Bigg's (previously transient) ecotype *Orcinus orca*	Regular sightings but not common. Present year round primarily for hunting. Wide-ranging, hunt and breed throughout large area. Food sources include marine mammals - particularly harbour seals, porpoises and sea lions. Also known to attack and kill baleen whales and minke whales.	Threatened Schedule 1	Red	Threatened

Table 3.4 Federal and Provincial Listed Species at Risk Potentially Present within the Study Areas

Species	Relevant Distribution and Seasonal Timing	Conservation Status		
		SARA – Schedule 1	**BC CDC**	**COSEWIC**
grey whale *Eschrichtius robustus*	Fairly common but not generally abundant. Most common to western Vancouver Island, some whales remain resident throughout summer to forage. May also be observed at other times of year during migration. Food sources include benthic invertebrates (epibenthic and infaunal amphipods, sand shrimp, ghost shrimp, small clams), planktonic invertebrates (mysid shrimps, planktonic crab larvae), and herring spawn and larvae.	Special Concern Schedule 1	Blue	Special Concern
harbour porpoise *Phocoena phocoena*	Common, use area for foraging and calving. Likely year round residents. Most commonly found in shallow (<200 m) nearshore areas. Food sources include epipelagic and mesopelagic cephalopods and fish, such as market squid, herring, sand lance and hake and large zooplankton such as euphausiids during weaning.	Special Concern Schedule 1	Blue	Special Concern
Steller sea lion *Eumetopias jubatus monteriensis*	Common. Year-round presence. Peak numbers in RSA during fall and winter. No rookeries (pupping areas) in RSA. One major year-round haulout (Carmanah Point) and numerous major winter haulouts, including one at Race Rocks, which is protected within an MPA. Use area to forage and haul out (*e.g.,* rest, socialize). Food sources include over 50 species of fish and invertebrates; preferred prey – small or medium sized schooling fishes such as herring, hake, sandlance, salmon, dogfish, eulachon and sardines. Bottom fish such as rockfish, flounder and skate. Also squid and octopus, crabs, mussels, clams and other invertebrates.	Special Concern Schedule 1	Blue	Special Concern
sea otter *Enhydra lutris*	Occur along much of the west coast of Vancouver Island and along a small section of the central British Columbia coast. Typically found in exposed coastal (<50 m depths) areas with shallow rocky reefs; however, in the winter, they may move to more sheltered areas within their home ranges. Food sources include a variety of invertebrates, including bivalves, snails, urchins, chitons, crabs, and sea stars. Also known to eat demersal fish species.	Special Concern Schedule 1	Blue	Special Concern

3.3.3.4 Selection of Marine Ecological Receptor Species

The following avian and mammalian species are receptors considered in this DQERA, taking into consideration factors of likely exposure to crude oil in the event of a spill, traditional use, commercial and recreational harvest, conservation status, and ability to represent other species belonging to the same or similar ecological guilds.

Black Oystercatcher

The black oystercatcher (*Haematopus bachmani*) is a conspicuous shorebird of western North America, ranging from the Aleutian Islands to the coast of Baja California. It prefers rocky shorelines with quiet embayments and forages on marine invertebrates in the intertidal zone, particularly molluscs such as mussels (50% of diet), clams (25% of diet) and gastropods (25% of diet), although other invertebrates such as chitons, crabs, isopods and barnacles are also consumed (Carney *et al.* 2013). The black oystercatcher is the largest North American shorebird, weighing about 590 g. From this body weight, a daily food ingestion rate of approximately 44.2 g dry matter per day is estimated using Nagy's (1987) equation for seabirds, as recommended in the Wildlife Exposure Factors Handbook (US EPA 1993a). This value is adjusted for moisture content, assuming that marine invertebrates (soft tissue, exclusive of shell) have a moisture content of 85%, to give a food ingestion rate of 295 g wet weight/d (exclusive of shell). The intertidal zone sediment ingestion rate of the black oystercatcher is estimated from the food ingestion rate to be 4.03 g dw/d, assuming that its prey

(primarily mussels, clams, winkles and limpets) have an average dry sediment content equivalent to 9.12% of their tissue dry mass.

Pigeon Guillemot

The pigeon guillemot (*Cepphus columba*) is an alcid bird found in coastal areas of British Columbia, ranging from Siberia and Alaska to California. They breed along rocky shores, cliffs and islands, usually in small colonies. Rather like puffins, they lay their eggs in rocky cavities or disused burrows of other seabirds. Outside of the breeding season, the birds migrate out to sea, with birds from various parts of the Pacific coast congregating in the coastal waters of British Columbia.

Pigeon gullemots dive for food, swimming underwater to feed on benthic prey, which is usually obtained close to shore. They mainly eat fish including sculpins, sandfish, capelin, small cod and crabs. The average body weight ranges from 450 to 550 g (Cornell Lab of Ornithology 2014). From an assumed body weight of 500 g, a daily food ingestion rate of approximately 39.3 g dry matter per day is estimated using Nagy's (1987) equation for seabirds, as recommended in the Wildlife Exposure Factors Handbook (US EPA 1993a). This value is adjusted for moisture content, assuming that marine fish have a moisture content of 75%, to give a food ingestion rate of approximately 160 g wet weight/d. The sub-tidal zone sediment ingestion rate of the pigeon guillemot is estimated from the food ingestion rate to be 0.393 g/d, assuming that its prey (small fish) have an average dry sediment content equivalent to 1% of their tissue wet mass.

Surf Scoter

The surf scoter (*Melanitta perspicillata*) is one of three North American species of scoter and is a sea duck. Surf scoters inhabit Atlantic and Pacific coastal environments during the fall, winter and early spring, but migrate to freshwater wetlands, primarily in the boreal forest, to reproduce. Some non-breeding individuals, however, may remain in coastal environments throughout the year. On wintering grounds, surf scoter prefer shallow coastal areas less than 10 m deep with a variety of hard and soft bottom substrates (Savard *et al.* 1998). Surf scoter are frequently found in large mixed-species flocks, but outnumber other scoters along steep rocky shores and fjords in British Columbia (Savard *et al.* 1998).

The surf scoter is a medium size sea duck species, and individuals collected from coastal British Columbia sites had a mean mass of 1,025 g for females and 1,153 g for males (Savard *et al.* 1998). Like most sea ducks, its diet in marine environments consists primarily of bivalve molluscs, with preference for mussels in rocky intertidal areas, and clams in soft bottom areas (Lewis *et al.* 2007a, 2007b). For a short period of time prior to migration in the spring, large groups of scoters congregate on herring spawning sites to feed on deposited roe (Lewis *et al.* 2007b). Surf scoter will remain in an area provided there is sufficient food, but migrate to capitalize on seasonally abundant food sources. In freshwater environments, surf scoter feed upon a variety of aquatic invertebrates (Savard *et al.* 1998).

In the marine environment, the surf scoter is modelled as consuming 90% molluscs, 5% fish (roe) and 5% plant material (Savard *et al.* 1998). With a modelled body weight of approximately 1.1 kg (1,100 g), allometric models estimate that the surf scoter will consume 313 g wet-weight food/day (US EPA 1993a) and will ingest 5.91 g of dry intertidal zone sediment per day.

Bald Eagle

The bald eagle (*Haliaeetus leucocephalus*) is the second largest bird of prey found in North America, and the largest found in Canada (Stocek 1992). Adult birds are readily identified by their striking appearance, characterized by dark brown body plumage contrasting sharply with white head and tail plumage (Buehler 2000). The bald eagle's range is restricted to North America, where it prefers sea coasts, lakeshores or riverine habitat that possesses suitable nesting trees in which to breed. The majority of Canada's breeding population is found along the coastline of British Columbia, although the northern boreal forests of Alberta, Saskatchewan, Manitoba and Ontario also support substantial breeding populations. Cape Breton and Newfoundland support the majority of the Atlantic breeding population (Stocek 1992). In the autumn, central Canadian breeding populations migrate to the west-central and southwestern United States, returning in late winter or early spring. Pacific and Atlantic populations may remain in their breeding habitat year-round if their fishing areas do not freeze over (Stocek 1992).

Female bald eagles are up to 25% larger than males, and birds from northern latitudes (Canada and Alaska) are larger than their counterparts in the southeastern and southwestern United States (Buehler 2000). The typical body mass of the bald eagle ranges from 3 kg to 6.3 kg (Buehler 2000), although masses of 7 kg have been recorded (Stocek 1992). Immature eagles grow rapidly owing to a voracious appetite.

Bald eagles are opportunistic feeders, taking live prey when available but preferring to scavenge carrion or to pirate freshly killed prey from other predators (Stocek 1992; US EPA 1993b). Their preferred food items include fish, aquatic birds and mammals; however, choice of prey is site-specific and may vary widely across their range (Buehler 2000). Adult birds are more likely to hunt and kill food items whereas immature birds are more prone to obtaining food through scavenging and piracy (Stocek 1992). Coastal bald eagle populations are modelled as consuming 95% marine fish and 5% mammals, while inland populations are modelled as consuming 45% terrestrial vertebrates (mammals and birds) and 55% freshwater fish.

For the coastal bald eagle, US EPA (1993a) provides a food ingestion rate of 0.12 g/gbw-day for free-flying adult birds. Based on a body weight of 4.5 kg this provides a total food ingestion rate of 540 g/d. Assuming 95% of this is marine fish, the daily fish ingestion rate is 513 g (wet weight). Associated with this fish ingestion there will be a marine sediment ingestion rate of 1.47 g/day.

Glaucous-winged Gull

The glaucous-winged gull (*Larus glaucescens*) is a medium- to large-sized seabird, similar to a herring gull, weighing between 0.73 and 1.69 kg, with an average value of about 1.1 kg (US EPA 1993b). Glaucous-winged gulls feed on almost anything, including fish, squid, crustaceans, molluscs, worms, insects, small mammals and birds, duck and gull eggs and chicks, amphibians, and garbage. They have a foraging range that may vary from approximately 300 to 785,000 ha (US EPA 1993b). They consume approximately 250 g of wet weight food per day.

The glaucous-winged gull's diet is modelled as including 7.5% soil invertebrates, 15% terrestrial mammals, 7.5% marine invertebrates, and 70% fish. Based on its consumption of these foods, glaucous-winged gull is estimated to incidentally ingest approximately 0.36 g/day of dry soil, 0.47 g/day of dry intertidal sediment and 0.49 g/day of dry interal and subtidal sediment. Due to its strong association with

water and tendency to scavenge, the glaucous-winged gull would be highly exposed to hydrocarbons in water, shoreline sediments, fish or carrion during and following an oil spill event.

Great Blue Heron

Great blue heron (*Ardea herodias*) is a large wading bird (greater than 1 m tall), weighing approximately 2.2 kg (US EPA 1993b). They primarily inhabit aquatic and marine areas, spending most of their time foraging for fish in shallow waters of lakes, rivers, streams, or estuarine and sheltered coastal areas.

Great blue heron feed predominantly on small fish. Based on US EPA (1993b), its diet is modelled as fish (95%), invertebrates (5%). Adults consume approximately 0.4 kg of wet weight food per day. Based on its consumption of these foods, great blue heron is estimated to incidentally ingest approximately 1.58 g/day of dry sediment from the intertidal zone. Due to its association with water, the great blue heron would be highly exposed to hydrocarbons in water, shoreline sediment and invertebrates during and following an oil spill event.

Double-crested Cormorant

The double-crested cormorant (*Phalacrocorax auritus*) can be found both in coastal and inland areas areas, and is widely distributed in North America, from the Aleutian Islands to Mexico, and from Florida to the Maritime provinces of Canada. Double-crested cormorants are colonial waterbirds that seek water bodies big enough to support their diet, of mainly fish. However, they may roost and form breeding colonies on smaller lagoons or ponds, and then fly considerable distances a feeding area. In addition to fishing waters, cormorants need perching areas for the time that they spend resting each day. After fishing, cormorants retire to high, airy perches (often rocks, wires, or tops of trees) to dry off and digest .

The double-crested cormorant dives to find its prey. It mainly eats fish, which are caught while diving underwater, but will sometimes also eat amphibians and crustaceans. They are relatively large birds, ranging in weight from 1.2 to 2.5 kg. From an assumed body weight of 1,800 g, a daily food ingestion rate of approximately 97 g dry matter per day is estimated using Nagy's (1987) equation for seabirds, as recommended in the Wildlife Exposure Factors Handbook (US EPA 1993a). This value is adjusted for moisture content, assuming that marine fish have a moisture content of 75%, to give a food ingestion rate of approximately 390 g wet weight/d. The subtidal and intertidal zone sediment ingestion rate of the double-crested cormorant is estimated from the food ingestion rate to be 0.97 g/d, assuming that its prey (small fish) have an average dry sediment content equivalent to 1% of their tissue dry mass.

Coastal-dwelling Mink

The mink (*Neovison vison)* weighs approximately 850 g. It is a medium-sized member of the weasel family and is the most abundant and widely distributed carnivorous mammal in North America (US EPA 1993b). Mink are found throughout the continental portion of Canada (including Newfoundland), except in the most barren portions of northwestern Quebec and eastern Nunavut.

Mink are active year-round and are associated with aquatic habitats such as rivers, streams, lakes, ditches, swamps, marshes, coastal shorelines and backwater areas (US EPA 1993b). Home ranges vary considerably but are in the range of 7.8 ha to 380 ha

(US EPA 1993b). The mink feeds extensively on small mammals, fish, amphibians and crustaceans, as well as birds, reptiles and insects depending on the season (US EPA 1993b). Mink consume approximately 0.22 kg of wet weight food per day. The diet of coastal-dwelling mink is modelled as including 55% small mammals, 35% fish and 10% marine benthic invertebrates.

Based on its consumption of a variety of foods, the coastal-dwelling mink is estimated to incidentally ingest approximately 0.78 g/d of dry sediment from the intertidal zone. As the coastal-dwelling mink is expected to meet dietary water requirements by seeking freshwater sources, direct seawater ingestion is not considered for this receptor.

Harbour Seal

The harbour seal (*Phoca vitulina*) is the seal most commonly found along temperate and Arctic marine coastlines of the Northern Hemisphere, including the coastline and inlets of British Columbia. Harbor seals weigh about 11 kg at birth and can double their weight during a suckling period of about one month. They can reach 1.5 to 1.8 m in length. The average weight for adults is about 180 pounds (82 kg); males are somewhat larger than females and can weigh up to 130 kg (Alaska Department of Fish and Game 2014). The mean body weight for an adult harbour seal is assumed to be approximately 80 kg (US EPA 1993b).

Harbor seals inhabit a variety of environments and are able to tolerate a wide range of temperatures and water. In its eastern (Atlantic) range, the harbor seal inhabits inlets, islets, reefs, and sandbars; in its western (Pacific) range, preferred habitats include tidal mud flats, sand bars, shoals, river deltas, estuaries, bays, coastal rocks, and offshore islets, even ranging up rivers into freshwater areas in search of food (US EPA 1993b). Habitats used for haul-outs include cobble and sand beaches, tidal mud flats, offshore rocks and reefs, glacial and sea ice, and man-made objects such as piers and log booms, where they are protected from adverse weather conditions and predation, with access to a foraging area.

Harbor seals feed primarily on seasonally available prey including fish such pollock, Pacific cod, capelin, eulachon, Pacific herring, sandlance, Pacific salmon, sculpin, flatfish (*e.g.,* flounder and sole), octopus, and squid (Alaska Department of Fish and Game 2014). The food ingestion rate is reported to range from 0.05 to 0.13 g wet weight/g body weight per day (US EPA 1993b), and a value of 0.1 is assumed here, giving a total diet of approximately 8 kg wet weight per day. This diet is assumed to comprise 75% fish and 25% squid or other mollusks. These foods (on a dry weight basis) are assumed to contain 1% dry sediment, giving an estimated subtidal sediment ingestion rate of approximately 20 g/day.

Steller Sea Lion

The Steller sea lion (*Eumetopias jubatus*) is the predominant species of sea lion in Canada. It is the largest species of sea lion in the world (DFO 2007a) and has a range extending from the Kuril Islands and the Sea of Okhotsk in Russia, to the Gulf of Alaska in the north, and down to the west coast of North American to Año Nuevo Island off central California. The California sea lion (*Zalophus californianus*) native to the coast of California and Mexico including Baja California and the Tres Marias Islands also occasionally migrates north to British Columbia (Mate 1978; Price 2002). The Steller sea lion is listed as a species of special concern under *SARA*, COSEWIC and

the BC Ministry of Environment. In 1970, it was placed under protection through the Federal *Fisheries Act* as populations had declined substantially as a result of hunting, accidental mortality from fishing nets and pollution. To date, populations are recovering (DFO 2007a; Environment Canada 2008a).

Steller sea lions exhibit extreme sexual dimorphism. Males are commonly twice the size of females. Adult females typically weigh 200 to 300 kg, whereas adult males typically weigh 400 to 800 kg (Environment Canada 2008a). The average female weight has been estimated as 263 kg and the average male weight as 566 kg (Gonder 2000). Steller sea lions have a limited range within Canada and are found in three main breeding areas, or rookeries, on the coast of British Columbia: the Scott Islands, Cape St. James and offshore from the Banks Islands, in the northern British Columbia mainland. When not breeding, haulout sites are distributed throughout the coastal waters of British Columbia and Washington State, including the Gulf and San Juan Islands (Jefferies *et al.* 2000). This species is not considered migratory (Environment Canada 2008a).

The Steller sea lion diet consists mainly of pelagic fish including salmon, herring, sand lance and sardines, but it will occasionally hunt benthic species such as rockfish, flounder and skate. Cephalopods such as squid and octopus are also important parts of its diet (Gonder 2000; Environment Canada 2008a). When not hunting in open water, the Steller sea lion can be seen loafing on rocky shorelines, often in groups (Gonder 2000).

Allometric models indicate that a 566 kg male Steller sea lion consumes approximately 50 kg of wet weight food per day (US EPA 1993a). The Steller sea lion is modelled as consuming 90% fish and 10% marine invertebrates (squid and octopus). Based on its consumption of these foods, Steller sea lions are estimated to incidentally ingest 126 g of dry sediment per day.

Harbour Porpoise

Differences in the skull morphology of harbour porpoises from the Atlantic and Pacific oceans has led to sub-specific separation. In Canada, members of the Pacific subspecies (*Phocoena phocoena vomerina*) inhabit the coastal waters of British Columbia (Environment Canada 2006a). They are small, sexually dimorphic cetaceans. Adult females have a mean length of approximately 1.6 m and weigh 60 kg, whereas adult males are approximately 1.45 m in length and weigh 50 kg (CMS 2003). An average body weight of 55 kg is assumed here.

Although colouration may vary, the harbour porpoise typically has a dark grey dorsal side that transitions to a white ventral side at the mid-flank (Hammond and Masi 2000). A dark stripe extends from the mouth to the flippers (CMS 2003). Both the Atlantic and Pacific subspecies are designated as species of special concern by COSEWIC (*P. p. vomerina* is listed as a Schedule 1 species under the *SARA*). Incidental catch of the harbour porpoise in fishing gear (*e.g.,* gill nets) is a major cause of mortality and the major reason for these designations. Great white sharks, killer whales and occasionally bottlenose dolphins will prey on harbour porpoises.

The harbour porpoise is found on the continental shelves of temperate North America, generally inhabiting oceanic depths of less than 150 m (Environment Canada 2006b; IMMA 1998). Some seasonal migration is exhibited, as the harbour porpoise moves north and inshore during the summer and spends winters further offshore in southern waters (following movements of prey) (CMS 2003). In the Bay of Fundy, for example, these animals may cover areas upwards of 11,000 km^2 over the course of one month. However, much of their foraging effort is usually focused on an area less than 300 km^2.

The harbour porpoise diet consists mainly of small (shorter than 40 cm), schooling fish (*e.g.,* herring and sand lance) and squid, although squid are reportedly more abundant in the diet of Pacific Ocean porpoises than those in the Atlantic (Santos and Pierce 2003; Environment Canada 2006b). Harbour porpoises consume these food items (assumed to comprise 95% fish and squid and 5% benthic invertebrates) at a rate of approximately 4.32 kg wet weight per day (estimated using the allometric equation provided in Innes *et al.* 1987). Based on its diet of pelagic fish and squid, the rate of dry marine sediment ingestion for the harbour porpoise is estimated to be 15.2 g/d.

Killer Whale

Killer whales, also known as orcas (*Orcinus orca*) are the largest member of the dolphin family. They are easily identified by their tall triangular dorsal fin and their distinctive black and white colouration. The maximum recorded body lengths are 9.0 m for males and 7.7 m for females. Clark *et al.* (2000) give average body mass values for adult male (4,088 kg) and female (2,441 kg) killer whales. The maximum recorded mass was 6,600 kg for a 7.65 m male, and 4,700 kg for a 6.58 m female. Longevity is approximately 80 years for females and 40-50 years for males, respectively.

Killer whales occur in all oceans of the world, although they are most common in highly productive areas. The southern resident killer whale is one of five distinct populations which have been identified in Canadian waters. Populations in Canada include one on the east coast (the northwestern Atlantic and eastern Arctic population) and four distinct populations on the west coast (including northern resident, southern resident, west coast transient, and offshore populations). Each population differs morphologically, genetically and behaviourally and they do not associate with each other. The southern resident population is of greatest concern in the context of this ERA as these animals are generally found around southern Vancouver Island in summer and fall, although the animals may range widely at other times of year.

During the summer and fall, the distribution of both northern and southern resident killer whales is closely linked to that of chinook salmon. The southern resident population is known to range from Haida Gwaii in northern BC, to Monterey Bay in California. Members of a killer whale population may be spread over hundreds of kilometres at any given time. In 2008, Critical Habitat for resident killer whales was designated for two areas in British Columbia under the Canadian *Species At Risk Act*. For southern residents this area includes Juan de Fuca and Haro Straits, Boundary Pass, the waters surrounding the southern Gulf Islands, and a part of southern Georgia Strait off the mouth of the Fraser River.

In Canada, killer whales have iconic status with aboriginal people and the public, and have been extended legal protection under the *Marine Mammal Regulations* of the *Fisheries Act*. There were 70 southern residents in 1974. In 2001, the southern resident population was listed as Endangered under the *SARA*. Section 32 (1) of the *SARA* outlines the general prohibition that no person shall kill, harm, harass, capture or take an individual of a wildlife species that is listed as an extirpated, endangered or threatened species (as defined in Schedule 1 of the *SARA*). Therefore, whereas most wildlife species are managed at a population level, the *SARA* has the effect of extending individual protection to species and/or populations such as the southern resident killer whale that are deemed to be extripated, endangered or threatened. While the southern resident population increased fairly steadily through the mid-1990s, it has been declining since then. In 2006, there were 87 individuals in the southern resident killer whale population. The principal anthropogenic threats to northeastern Pacific populations of killer whales are disturbance (physical and acoustic), prey depletion, and contaminants. These threats may act synergistically. Oil spills, collisions with vessels, interactions with commercial fisheries and climate change also may affect killer whales.

Killer whales show little or no tendency to avoid oil spills. During the EVOS members of the transient AT1 population and the Resident AB pod were seen surfacing in oil slicks immediately following the spill in 1989. As reported by the Exxon Valdez Oil Spill Trustees Council (EVOSTC 2010) and Matkin *et al.* (2008), both groups experienced mortality in the months following oil exposure. Deaths were potentially due to the inhalation of petroleum vapours, or from feeding on oiled seals or contaminated fish (EVOSTC 2010). Mortality continued in the following year because mothers died leaving orphaned calves that subsequently died (EVOSTC 2010). The mortality rate for the AB pod was reported to be 19% in 1989 and 21% in 1990, compared to an expected natural mortality rate of about 2.5% (EVOSTC 2010, Matkin *et al.* 2008). However, the AB pod was in conflict with the commercial longline fishery of Prince William Sound prior to the EVOS. Matkin *et al.* (1999) documented bullet wounds on 10 whales in the pod, 5 of which subsequently died. Between 1985 and 1986, 6 whales were lost from AB pod. The mortality rate in the AB pod was therefore higher than normal even before the oil spill. Matkin et al. (1999) described the social structure of the matriarchal pods, noting that the loss of an important matriarch can affect a pod for some years thereafter. Thus, the loss of key matriarchs from the 1986 shootings, and from the 1989 EVOS event, may have resulted in a continuing of AB pod members (Harwell and Gentile 2006). Harwell and Gentile (2006) concluded that the population reduction of the AB pod in the immediate aftermath of the EVOS was ecologically significant, and was most likely caused by exposure to crude oil, which exacerbated ongoing effects on that pod as a result of recent conflict with fishermen.

Both northern and southern resident killer whales forage selectively for salmon. Their preferred prey is chinook salmon from May through September, despite the low abundance of this species relative to others. During the summer months, chinook salmon make up roughly 80 percent of the whales' diet (NOAA 2013). During October, chum salmon are abundant and dominate the diet, although Chinook are taken as well. From November through April, the diet is not well known. Chinook salmon appear to be preferred by resident killer whales due to their large size, high fat content, and year-round availability in coastal waters. The Fraser River is the primary spawning grounds for chinook salmon in Canada.

Assuming an adult female killer whale has a body mass of 2,440 kg, and also that it has an energy requirement of approximately 60 kcal/kg bw/d (Williams *et al.* 2004), its daily energy requirement will be in the range of 146,400 kcal/d. Taking the energy content of chinook and other salmon to be approximately 1.79 kcal/g wet mass, the daily fish ingestion rate for the whale will be approximately 81.8 kg ww/d. This represents 3.35% of the whale's body weight, and is consistent with feeding rates for killer whales in captivity (2% to 4% of their body mass per day). It is further estimated that the moisture content of salmon (the primary prey of resident killer whales) is 73.4%, giving a dry-food ingestion rate of 21.76 kg dw/d. Fish typically contain about 1% dry sediment on a dry weight tissue basis, as part of their gut contents. Therefore the killer whale is estimated to ingest approximately 218 g dry sediment each day as part of its diet.

Humpback Whale

The humpback whale (*Megaptera novaeangliae*) is a baleen whale of the family Balaenopteridae. They can reach lengths of 13 to 14 m and are recognizable by their variable black and white colouration and long pectoral flippers that can be up to 25% of the body length. The average adult mass is on average 34,000 kg, up to a maximum of approximately 45,000 kg.

The north Pacific humpback is one of two distinct populations that have been identified in Canada, the other being the western north Atlantic population. Humpback whales are found in marine waters from the tropics to sub-polar waters. In the Pacific region of Canada, their range extends the length of the British Columbia coastline and includes both offshore waters and coastal inlets. Humpback whales are in Canadian waters

primarily for summer feeding, although they have been confirmed to be present in low numbers throughout the year.

Humpbacks have cultural significance to coastal First Nations, and have historically been hunted for subsistence. Commercial whaling had seriously depleted all populations of humpback until the species was given legal protection in 1966, after which a general increase in abundance has been noted. Humpback whales in British Columbia were recently estimated to have an annual population growth of 4.1% compared to an estimate of 4.9% for the entire North Pacific.

Humpback whales are protected throughout most of their global range under the International Convention for the Regulation of Whaling, and the Convention on International Trade in Endangered Species of Wild Flora and Fauna. In Canada, the Humpback whale is legally protected under the *Fisheries Act* (*Marine Mammal Regulations*). The North Pacific population was until recently listed as “threatened” under the *SARA*, but this classification was downgraded to “special concern” as a result of sustained population growth. Humpback whales continue to be subject to noise disturbance, habitat degradation on the breeding grounds in southern waters, entanglement in fishing gear, and ship strikes.

Humpbacks are baleen whales, meaning that they filter their food through baleen plates. Their diet consists primarily of crustacean zooplankton (particularly euphausiids (*i.e.,* krill) followed by copepods, and small schooling fish (*e.g.,* Pacific herring, capelin, sand lance, Pacific sardine, juvenile salmonids, Pacific cod, mackerel, and anchovy), pteropods, and some cephalopods (*e.g.,* squid). In Frederick Sound, Alaska, euphausiids (*Thysanoessa spinifera* and *Euphausia pacifica*) were observed to make up 50-80% of the diet (COSEWIC 2011). It is estimated they eat an average of about 1,000 to 1,500 kg of krill and small fish per day during the summer months (NOAA 2014). For exposure and modeling purposes, it is assumed that a humpback whale weighing 40,000 kg consumes 750 kg ww of small crustaceans and 500 kg ww of small fish per day, and incidentally consumes 2.38 kg of dry sediment per day.

3.4 Exposure and Toxicity Assessment

The purpose of the exposure and toxicity assessment steps are to evaluate data related to the crude oil product, ecological receptors and exposure pathways identified during the problem formulation phase; and to evaluate whether such exposures (usually expressed in terms of the daily ingested dose from all relevant media) may be in a range that can credibly be linked to health effects.

Using site-specific data and a series of conservative assumptions, the exposure assessment predicts the behaviour and distribution of COPC in the environment, and the extent to which ecological receptors would be exposed via exposure scenarios and pathways identified in the problem formulation. The magnitude of exposure depends on the interaction(s) of a number of parameters, including:

- concentrations of COPC in various environmental media following a hypothetical spill
- physical-chemical characteristics of the COPC, which affect their environmental fate and transport and determine such factors as efficiency of absorption into the body and rate of metabolic breakdown or excretion
- influence of site-specific environmental characteristics (*e.g.,* shoreline geology, sediment type, topography, hydrology, and hydrogeology on the COPC behaviour within environmental media)
- physiological and behavioural characteristics of the receptors.

Separate exposure assessments are conducted for each hypothetical spill scenario. Exposure assessments for specific scenarios have been selected from stochastic crude oil spill modelling and are based on the properties of the representative hydrocarbons, and an assumed release volume for each scenario. Details of the specific scenarios evaluated are discussed in Section 4.2.3 and 5.2.1.

A conservative default assumption is that COPC are fully bioavailable in all exposure media. All substances discharged to the intertidal and sub-tidal sediment, as well as those dissolved within the water column related to the hypothetical spill examples, are assumed to be completely (*i.e.,* 100%) bioavailable upon release into the marine environment.

3.4.1 Evaluation of Exposure to Hydrocarbons in the Water Column

The deterministic modelling completed by EBA provided pseudo-component concentrations for each grid cell of the water column within the model domain, and for selected time intervals (*i.e.,* hourly values for the full length of the model simulation, up to 60 hours for the WMT, and 240 to 360 hours for spill scenarios at Arachne Reef). Grid sizes for evaluating effects in the water column were based on 125 m x 125 m for the release scenarios at Westridge Marine Terminal, and a 975 m x 975 m grid for both scenarios at Arachne Reef.

Two assessments were carried out for hydrocarbons in the water column, the first considering total hydrocarbon concentrations and using a non-polar narcosis endpoint to evaluate the potential for mortality of marine biota (including algae, invertebrates and fish), and the second evaluating the potential for exposure to PAHs to induce deformities or cause mortality of developing fish eggs and embryos (the Blue Sac Disease endpoint). The technical basis for evaluating these two endpoints is described in the following sections.

3.4.1.1 Narcosis-Based Toxicity Benchmarks for Aquatic Organisms

The principle of narcosis as a toxicological mode of action for organic compounds has long been understood. The toxic modes of action for organic compounds were classified by Verhaar *et al.* (1992). Type 1 or “baseline” non-polar narcotic chemicals are broadly defined as all non-ionic organic chemicals with a similar mode of toxic action (*i.e.,* narcosis), that do not interact with specific receptors in an organism, and are not reactive when considering overall acute effects. Non-polar narcosis is the mechanism most relevant to oil spills, as it includes the BTEX group (benzene, toluene, ethyl-benzene and xylenes, also known as mono-aromatic hydrocarbons or MAH), PAHs (polycyclic aromatic hydrocarbons), and the balance of the aliphatic and aromatic chemical groups comprising PHC (petroleum hydrocarbons) generally.

Di Toro *et al.* (2000) and Di Toro and McGrath (2000) applied the non-polar narcosis theory to develop water and sediment quality criteria for PAHs, a component of crude oils and other hydrocarbon mixtures. On the basis that narcotic chemicals affect a target class of lipids within the whole organism, Di Toro’s model has been named the Target Lipid Model (TLM). The TLM has subsequently been validated for both a wide range of aquatic organisms (algae to amphibians), and a wide range of hydrocarbon compounds, including complex mixtures such as gasoline (McGrath *et al.* 2005) and crude oils (Di Toro *et al.* 2007).

Based on the assumptions that hydrocarbon concentrations in aquatic organisms equilibrate with concentrations in the environment; that at a molecular level, narcotic substances have similar effects on biological functions; and that the site of toxic action is in the lipids associated with biological membranes, a critical contaminant concentration (C^*_L with units of μmol hydrocarbon/g lipid or octanol) can be defined, representing the level of exposure that identifies with acute or chronic toxicity. Some differential toxicity is observed for certain classes of hydrocarbons (such as PAHs), and this is addressed using chemical class correction factors.

From C^*_L, and from considerations of multi-component organic compound solubility in water (or sediment pore water), it is possible to develop models that can account for the toxicity of hydrocarbon mixtures in water and sediment, for a broad range of aquatic biota. This was done and validated by Di Toro *et al.* (2000), Di Toro and McGrath (2000), McGrath *et al.* (2005), McGrath and Di Toro (2006, 2009) and Di Toro *et al.* (2007). McGrath and Di Toro (2009) provide a summary of values

of C^*_L based on acute lethality for 47 species of aquatic organisms including green algae, protozoa, crustaceans, mollusks, insects, fish, and amphibians. Values of C^*_L for individual species range from 24.5 µmol/g octanol (*Onchorhynchus gorbuscha/* pink salmon) to 500 µmol/g octanol (*Ankistrodesmus falcatus*/ a green alga). There is no notable differentiation between freshwater and saltwater species sensitivity, or phylogenetic grouping, and the more sensitive species (C^*_L <50 µmol/g octanol) include representatives of fish, crustaceans, insects, and algae.

Di Toro *et al.* (2000) estimated the acute to chronic ratio (ACR) for the toxicity of non-polar narcotic chemicals as having a value of 5.09. Subsequent investigations (*e.g.,* McGrath and Di Toro, 2006; 2009) have suggested slight modifications to this value for certain classes of chemicals.

Following a US EPA protocol (Stephan *et al.*, 1985), Di Toro *et al.* (2000) identified the 5th percentile value of C^*_L as being representative of a sensitive species. This endpoint was then used as the basis for calculating a final acute value (FAV) and final chronic value (FCV = FAV/ACR) for hydrocarbons. The FAV is an estimate of the concentration of the material that would be lethal to 50% of exposed individuals of a sensitive species in an acute toxicity test. The final chronic value would be lethal to 50% of exposed individuals of a sensitive species under chronic exposure conditions.

Having established that Type 1 narcotic substances can be treated as a class with additive toxicity (subject to some chemical class-specific correction factors as outlined above), Di Toro *et al.* (2000) showed how the toxicity of individual non-polar narcotic substances in water (denoted by the subscripts $_w$ (water) and $_j$ (chemical identity)) can be converted into Toxic Units (TU), which can be summed explicitly. These TU are defined so that when an individual TU or $\Sigma\, TU_j \geq 1$, the toxicity endpoint associated with the $C^*_{W,j}$ is expected.

$$TU_{W,j} = C_{W,j} / C^*_{W,j}$$

where TU are toxic units, the $C_{W,j}$ are the water column concentrations (mmol/L or mg/L) of the narcotic substances, and $C^*_{W,j}$ are the critical concentrations of those compounds (mmol/L or mg/L) at which some specific toxic response may be observed. For example, a saturated solution of benzene in water has a concentration of approximately 2,000 mg/L. The FAV LC_{50} for benzene in water is 29.5 mg/L. Therefore, a saturated solution of benzene in water contains 2,000/29.5 ≈ 68 TU, where the toxic endpoint is lethality to a sensitive species.

Predicting the toxicity of oils requires that the toxicity of mixtures of oil components (individual compounds or groups of compounds) be predicted. Since they are additive, it follows that:

$$TU_{mixture} = \Sigma\, TU_{W,j}$$

Therefore, the potential toxicity of a hydrocarbon mixture can be estimated on the basis of the sum of the toxicity (toxic units) of its dissolved component concentrations in water. The level of protection involved in this calculation can be selected by choosing an appropriate value of C^*_L, as in Table 3.5.

Table 3.5 Estimated Values of the FAV, FCV and HC_5 Concentrations for Selected MAH and PAH Compounds in Water

Chemical	Log K_{ow} (L/kg)	Molecular weight (g/mole)	FAV (mg/L) using C^*_L = 32 µmol/g octanol)	FCV (mg/L) using C^*_L = 8.36 µmol/g octanol)
Benzene	1.943	78.11	29.5	7.71
Toluene	2.438	92.14	12.0	3.13
o-Xylene	2.946	106.17	4.62	1.21
Ethylbenzene	3.006	106.17	4.06	1.06
m-Xylene	3.032	106.17	3.84	1.00
p-Xylene	3.051	106.17	3.68	0.96

Table 3.5 Estimated Values of the FAV, FCV and HC_5 Concentrations for Selected MAH and PAH Compounds in Water

Chemical	Log K_{ow} (L/kg)	Molecular weight (g/mole)	FAV (mg/L) using C^*_L = 32 µmol/g octanol)	FCV (mg/L) using C^*_L = 8.36 µmol/g octanol)
Naphthalene	3.256	128.19	1.63	0.43
Phenanthrene	4.584	178.23	0.13	0.034
Chrysene	5.782	228.29	0.013	0.003

Note:
The original estimate of FAV was 35.3 µmol/g octanol (Di Toro *et al.* 2000). Based on more additional data (McGrath and Di Toro 2009), this value is updated to approximately 32 µmol/g octanol. The FCV is estimated as FAV/ACR, and the most recent value for ACR for baseline narcotic chemicals has been revised from 5.09 (Di Toro *et al.* 2000) to 3.83 (McGrath and Di Toro 2009). The FCV (formerly 6.94 µmol/g octanol) is therefore revised to be 32/3.83 = 8.36 µmol/g octanol. Values of Log K_{ow} and molecular weight are from McGrath and Di Toro (2009).

Some important assumptions and limitations of the TLM must be kept in mind in the context of its proposed use as a tool in the evaluation of environmental effects of oil spills.

- Non-polar narcosis is conceptualized as a reversible mode of toxic action. If not killed or severely impaired by narcosis, exposed organisms should be able to recover from exposures when ambient concentrations of narcotic substances in water are reduced. This is also consistent with what is known about the rapid metabolism and/or excretion of absorbed hydrocarbons by most aquatic organisms (with the exception of certain taxa such as mollusks that lack enzyme pathways needed to metabolize PAHs).

- Lipids within an exposed organism are treated as if they were a single pool, and lipid normalization of tissue residue data is performed on this basis. It is likely, however, that there are several relevant pools of lipids within an organism (including as a minimum the target lipid, as well as other lipid pools such as storage lipids that have different composition and perfusion rates. Polar lipids include membrane-associated lipids (*e.g.,* phospholipids, free fatty acids, lipoproteins), whereas lipids associated with energy storage are predominantly non-polar. Thus, it is not certain that the target lipid equilibrates with narcotic toxicants at the same rate as other lipid pools in the body, and critical lipid concentrations may be underestimated if the target lipid equilibrates more rapidly than these other pools (McCarty *et al.*, 2013; McElroy *et al.*, 2010). On the other hand, fish having high lipid content have been observed to have greater resistance to hydrophobic chemicals than fish having lower lipid content, and it has been suggested that non-polar storage lipids could serve as a buffer by absorbing toxic chemicals at a site where toxicity is not expressed (Lassiter and Hallam 1990).

- Non-polar narcosis does not address the issue of chemicals that may have a specific mode of action, while also potentially possessing a narcotic effect of lower potency. Examples of this include phototoxicity, where exposure to ultraviolet light can cause certain PAH compounds or their metabolites to exert much higher toxicity than the parent compound as a narcotic substance; and effects of certain PAH compounds on developing eggs resulting in a constellation of effects such as yolk sac edemas, pericardial edemas, craniofacial malformations, and hemorrhaging, collectively referred to as "blue sac disease". McGrath and Di Toro (2009) reviewed reported effects concentrations for the latter syndrome, concluding that the predictions of the TLM and the observed effects concentrations for BSD fall in similar ranges.

- Various authors have noted that lines predicting the toxicity of hydrophobic organic chemicals cross the aqueous solubility line for those compounds at Log K_{ow} values between about 5.5 and 6.5. Similarly, models predicting bioconcentration of organic chemicals from water generally peak at Log K_{ow} values around 6, declining at higher values (Arnot and Gobas 2006). While some of this effect may be due to the extreme insolubility of such hydrophobic substances, the role of steric

factors that impede the movement of large molecules across biological membranes is not excluded. Based on McGrath and Di Toro (2009), a Log K_{ow} cutoff of 6.4 is applied here for MAH and PAH compounds, although a lower cutoff of Log K_{ow} = 5.5 is applied to aliphatic hydrocarbons where steric effects caused by molecular size may hinder absorption relative to the PAHs.

In the deterministic oil spill modelling, the diluted bitumen (CLWB) was described using 17 individual pseudocomponents (fractions), each one defining a smaller group of hydrocarbon compounds having similar characteristics. Following the principles outlined above, critical hydrocarbon concentrations in water were calculated for each pseudocomponent having a Log K_{ow} value less than 6.4 (aromatic compounds) or 5.5 (aliphatic compounds) for a sensitive (5th percentile) marine receptor species. As a result, only 10 of the pseudocomponents merit consideration as potentially bioavailable and toxic fractions; the remaining 7 pseudocomponents represent hydrocarbon fractions that have low bioavailability and are insufficiently soluble in water to represent a credible risk of acute lethality to aquatic life.

The predicted values are summarized in Table 3.6, and represent concentrations that would result in mortality of 50% of exposed individuals of a hypothetical sensitive species (representative of salmonid fish, crustaceans, and sensitive algal species) over a period of 96 hours. Using GIS, the predicted hydrocarbon concentrations in each cell of the model domain were compared to these benchmark values, and the results were summed to obtain an integrated quotient in the form of toxic units (TU). A running average value was then calculated, and the maximum 96-hour average TU value was selected for graphical presentation. A potential for direct mortality of aquatic organisms, including fish and invertebrates, was identified in grid squares where the 96-hour running average value exceeded unity.

Table 3.6 Final Acute Value Toxicity Benchmarks for Selected Hydrocarbon Pseudo-components

Pseudo-Component	Description	MW[2] (g/mol)	Narcotic Chemical Class[3]	Log Narcotic Chemical Class Correction Factor	Log K_{ow}[4]	Final Acute Value (FAV, mg/L)
VOL	Volatiles	70.8	Aliphatics	0.000	2.89	4.43
AR1	Benzene	78.1	MAHs	-0.109	2.13	19.7
AR2	Toluene, Ethylbenxene, Xylenes	99.2	MAHs	-0.109	2.93	4.42
AR3	Aromatics >C_8-C_{10}[1]	120.0	MAHs	-0.109	3.59	1.30
AR4	Aromatics >C_{10}-C_{12}	130.0	PAHs	-0.352	3.79	0.52
AR5	Aromatics >C_{12}-C_{16}	150.0	PAHs	-0.352	4.09	0.32
AR6	Aromatics >C_{16}-C_{21}	190.0	PAHs	-0.352	4.59	0.14
AR7	Aromatics >C_{21}-C_{34}	240.0	PAHs	-0.352	5.49	0.025
AL1	Aliphatics C_6-C_8	100.0	Aliphatics	0.000	3.99	0.59
AL2	Aliphatics >C_8-C_{10}	130.0	Aliphatics	0.000	4.89	0.11

Notes:

1 The ranges in this column represent the number of carbon atoms in the hydrocarbon molecule (*e.g.,* aromatics >C_8-C_{10} would be molecules containing at least one benzene ring, with specifically 9 or 10 carbon atoms.

2 MW stands for Molecular Weight.

3 Three different classes of hydrocarbon chemicals are indicated in this column: aliphatics are hydrocarbon molecules comprising chains of carbon atoms. MAHs (monoaromatic hydrocarbons) are hydrocarbon molecules containing a single benzene (aromatic) ring structure. PAHs (polycyclic aromatic hydrocarbons) are hydrocarbon molecules containing more than one aromatic ring structure.

4 Log K_{ow} stands for the logarithm of the mean octanol-water partition coefficient for a particular pseudocomponent, an indicator of both the relative water solubility of that hydrocarbon fraction, and its tendency to partition into the lipid phase of exposed organisms.

3.4.1.2 Endpoints for Exposure of Fish to PAHs

Chronic exposure to polycyclic aromatic hydrocarbons (PAHs), including alkyl PAHs, can cause a variety of effects on fish, some of which may simply be indicators of exposure (*e.g.,* changes in cytochrome activity or enzyme activation), and some of which may be:

- indicators of reduced fitness or long-term adverse effects (*e.g.,* sub-lethal responses such as changes in developing fish embryos broadly designated blue sac disease (BSD) or tumour induction in adult fish), or
- are clearly adverse effects (*e.g.,* acute lethality or reduced hatching success in exposed eggs).

Wu *et al.* (2012) note that the PAHs causing toxicity and those causing induction of EROD or CYP1A enzyme systems may not be the same and that such induction may not be a part of the primary mechanism of embryotoxicity.

This risk assessment considers the effects of hydrocarbons, including PAHs, on developing fish eggs and fry, and the risk assessment identifies realistic screening benchmarks for total PAH (TPAH) concentrations in water that are associated with reduced hatching success or mortality in developing fish eggs and fry.

Blue sac disease

Blue sac disease refers to a suite of morphological abnormalities of fish embryos that may be associated with chemical, physical, or thermal shock. Common symptoms include ocular, yolk sac, and pericardial edema, hemorrhaging, circulatory abnormalities, and spinal and craniofacial deformities (Fallahtafti *et al.* 2011). Pericardial edema appears to be the most sensitive morphological indicator of embryonic fish exposure to crude oil (Marty *et al.* 1997a; Carls *et al.* 1999). However, the development of symptoms associated with BSD does not necessarily result in mortality of fish embryos. The effects of PAH may range in severity from a complete cessation of embryo development and death before feeding begins to minor reductions in growth (Hodson *et al.* 2011).

The sublethal effects caused by BSD may influence survival rates by compromising the ability of fish larvae to escape predators and to successfully capture prey (Carls *et al.* 1999; Fallahtafti *et al.* 2011). In some cases, fish exhibit effects consistent with premature hatching: increased yolk volume and hepatocellular glycogen, increased apoptosis of gonadal cells, and less food in the GI tract (Marty *et al.* 1997b). These effects would likely decrease the chances of marine survival because swimming ability decreases as the yolk is absorbed, and it is lowest just before complete absorption, making it difficult to avoid predation (Marty *et al.* 1997b). In addition, premature fish had not yet initiated exogenous feeding (Marty *et al.* 1997b). Premature hatching (2-7 days early) was also observed in early life stages of fish exposed to natural oil sands and was attributed to rupture of the hatching glands (Colavecchi *et al.* 2006).

Several studies have linked BSD in the early life stages of fish with crude oil exposure and/or exposure to PAHs. Many of these studies were initiated as a consequence of the 1989 Exxon Valdez oil spill (EVOS) in Prince William Sound (PWS), Alaska. However, there has been considerable controversy regarding the sensitivity of fish eggs and embryos to PAH exposure.

The precise mechanisms leading to PAH-associated malformation and sublethal effects in fish early life stages are unknown (Incardona *et al.* 2004). Further, it remains unclear whether individual compounds act through separate mechanisms or share a common mode of action (Incardona *et al.* 2004). For example, while narcosis is often thought to be the primary mode of toxicity for low molecular weight (LMW) PAHs, work by Incardona *et al.* (2004) indicated that although three-ring PAHs disrupted cardiac function, they did not affect neuronal function. Doses of phenanthrene and dibenzothiophene that stopped circulation did not immobilize the embryos nor did they cause insensitivity to mechanosensory stimulation, both of which are indicators of narcosis (Incardona *et al.* 2004). Further, this cardiotoxicity appeared to be exerted by a pathway that does not require AhR activation (Incardona *et al.* 2009). This finding supports an earlier hypothesis that while multi-ring PAHs may require activation through the CYP1A enzyme system, some PAHs may be directly toxic (Carls *et al.* 1999).

Some workers have suggested that, CYP1A induction is an important contributor to PAH toxicity and the occurrence of signs of BSD. Brinkworth *et al.* (2003) hypothesized that prolonged CYP1A activity induced by continuous exposure to retene (an alkyl phenanthrene) may increase the formation of reactive oxygen species (ROS), leading to edema (decreased cellular integrity of blood vessels). This hypothesis fits observed retene toxicity, in which BSD appeared first and most often as vascular hemorrhaging (Brinkworth *et al.* 2003). They further suggest that the observed lag between CYP1A induction and the subsequent appearance of BSD indicates that oxidative stress defences must be depleted before oxidative stress occurs (Brinkworth *et al.* 2003). For example, juvenile trout, which are resistant to oxidative stress relative to larval stages, may be able to maintain high antioxidant levels through feeding, while larvae have a finite supply transferred maternally (Brinkworth *et al.* 2003).

In cases where BSD results in mortality in embryos, one cause may be poor utilization of yolk metabolites and subsequent starvation due to a breakdown of vitelline vessels supplying the embryo (Billiard *et al.* 1999 in Brinkworth *et al.* 2003). In larvae, edema accounted for the majority of mortality and was accompanied by decreased blood flow to the tissues, interference with nervous system function, and increased energy expenditure (Carls *et al.* 1999). In cases of severe pericardial and yolk sac edema, blood flow can be reduced to such an extent that tissues become necrotic and fish die (Marty *et al.* 1997a).

Common indicators of toxicity, such as mortality and hatching success, were found not to be particularly sensitive endpoints (Fallahtafti *et al.* 2011). In terms of mortality, the difference between EC_{50} and LC_{50} values ranged from 5- to 50-fold, and hatching success was not affected at PAH metabolite concentrations that caused other embryonic effects (*e.g.,* lethality both pre- and post-hatch) (Fallahtafti *et al.* 2011).

Toxicity Benchmarks for Bulk Oil from the Exxon Valdez Oil Spill

Marty *et al.* (1997b) monitored the development of pink salmon (*Oncorhynchus gorbuscha*) incubated in water flowing through gravel contaminated with weathered Prudhoe Bay crude oil (PBCO, approximating 11-day old crude oil). Larvae were sampled before, at, and 13 days post-emergence for histopathology and CYP1A activity. The uptake of PAHs by fish tissue was mediated by PAH dissolution in water, with biological effects observed at peak aqueous TPAH concentrations of 4.4 µg/L. As in herring (Carls *et al.* 1999), effects included CYP1A induction, yolk sac edema, premature emergence, and increased mortality. While CYP1A induction was correlated with exposure, it was not a conclusive indicator of oil exposure, because differences in staining were much less distinct after 13 days (likely due to rapid metabolism of PAHs according to Marty *et al.* 1997b). In addition, tissues of the control fish also displayed elevated CYP1A activity, suggesting that another inducing compound may have been present in the experimental system (Marty *et al.* 1997b).

Heintz *et al.* (1999) evaluated effects on pink salmon embryos exposed to artificially weathered Alaska North Slope crude oil or oil that had weathered on gravel for one year. They reported lethal effects (*i.e.,* increased mortality) at initial aqueous TPAH concentrations as low as 1.0 µg/L. However, this and other related studies were subsequently criticized by Brannon *et al.* (2001) on the basis that methodological artifacts associated with sample timing and sampling methods resulted in shock that was a major cause of embryo mortality, which was mistakenly interpreted as an effect of oil exposure. Subsequent work by Brannon *et al.* (2006a, 2007) identified toxicity in pink salmon embryos corresponding to a TPAH concentration in water of approximately 5 µg/L and TPAH concentrations in oiled gravels of approximately 5 mg/kg. Critical tissue concentrations in pink salmon eggs were identified as lying in the range of 7.1 mg/kg (Brannon *et al.* 2006a).

Both Rice *et al.* (2001) and Brannon *et al.* (2001) published reports seeking to resolve differing findings with respect to the sensitivity of pink salmon eggs and larvae to spilled oil. The following summarizes areas of dispute regarding the different findings (Rice *et al.* 2001), including:

- Both sets of researchers agreed that some nearshore habitats remained contaminated a year after the spill, but small amounts of oil were bioavailable to fry, and fry growth was not affected significantly in 1990 or later (Rice *et al.* 2001).

- Studies conducted by researchers associated with the Auke Bay Laboratory in Alaska (including Rice and co-workers) found that embryos incubating in some oiled streams affected by the EVOS continued to display elevated embryo mortality through 1994 (Rice *et al.* 2001). It was hypothesized that high molecular weight PAH in weathered oil was leaching from oiled stream banks into salmon redds. In contrast, studies by Brannon and co-workers found no evidence of in-stream oil or increased embryo mortality. Brannon *et al.* (2001) were critical of the study designs used by researchers from the Auke Bay Laboratory, which they argued introduced artifacts into the data that were misinterpreted as representing oil effects. Later laboratory studies by Trestee scientists confirmed that early sampling time was indeed related to increased risk of shock injury (Thedinga *et al.* 2005; Carls and Thedinga 2010.)

- Laboratory studies carried out by Rice and co-workers (Heintz *et al.* 1999) found that salmon embryos were sensitive to weathered crude oil concentrations in the range of 1 µg/L (Rice *et al.* 2001). Brannon *et al.* (2008) subsequently argued that the presence of oil droplets in the experimental apparatus invalidated the dissolved TPAH concentrations reported by Heintz *et al.* (1999) and that actual exposure concentrations were higher, due to the presence of and contact between oil droplets and eggs.

- Notwithstanding disagreement about the magnitude or persistence of oil spill effects on salmon embryos in oiled streams, both sets of researchers concluded that long-term damage in the pink salmon population in PWS as a whole was not evident, and that the population collapse of 1992 and 1993 (coinciding with the collapse of Pacific herring) was not directly linked to oil toxicity (Rice *et al.* 2001). Brannon *et al.* (2006b), evaluating 16 years of adult pink salmon data beginning in 1989, concluded that no effects of oiling were detected in the analysis of spawner density or recruits per spawner in spill-affected streams.

Toxicity Benchmarks for Hydrocarbon Mixtures other than EVOS

Colavecchia *et al.* (2006) examined the influence of natural oil sands on the early development of white sucker (*Catostomus commersoni*). Sediment-associated PAH in natural oil sands from two rivers (TPAH concentrations of 250 to 360 µg/g) were composed primarily of alkylated phenanthrene/anthracenes, fluoranthene/pyrenes, and benz[a]anthracene/chrysenes. Sediments from wastewater ponds had average TPAH concentrations of 1.3 µg/kg and were composed primarily of alkylated fluoranthene/pyrenes and benz[a]anthrancene/chrysenes. Groups of white sucker eggs exposed to natural PAH inputs exhibited mortality, premature hatching, reduced growth and larval malformations, the most common of which (pericardial and yolk sac edema, hemorrhage, and spinal defects) are associated with BSD (Colavecchia *et al.* 2006). Juvenile fish exposed to natural oil sands and to wastewater pond sediments for 96 hours exhibited significantly increased ethoxyresorufin-O-deethylase (EROD; a proxy for CYP1A induction) activity. Cumulative mortality of fish larvae (to 21 days post-hatch) was significantly increased for organisms exposed to TPAH concentrations greater than 100 µg/L. Natural oil sands sediments from Ells River were most toxic to white sucker embryos. The PAH composition in natural oil sands sediments from Ells River differed from other sites in that it had larger quantities of both parent and alkyl phenanthrenes/anthracenes (Colavecchia *et al.* 2006).

Results consistent with those of Brannon *et al.* (2006a, 2007) were published by Wu *et al.* (2012). Rainbow trout (*Onchrhynchus mykiss*) embryos were exposed to chemically dispersed and undispersed crude oils. Although the crude oils had differing overall toxicity, the toxicity was largely explained by the concentration of TPAH. In addition, while addition of dispersants increased the overall toxicity of crude

oils, the toxicity was again well explained on the basis of the increased dissolved TPAH concentrations. Lethality endpoints for rainbow trout embryos appeared at aqueous TPAH concentrations greater than about 7.5 µg/L.

Toxicity Benchmarks for Individual PAH Compounds

Much of the most recent research on BSD has focused on the role of individual PAHs, particularly alkyl and tricyclic PAHs, in the onset of BSD. Alkyl PAHs induced BSD and can be up to ten times more toxic than their non-alkylated counterparts (Fallahtafti *et al.* 2011; Turcotte *et al.* 2011), possibly due to an increased binding affinity for the AhR receptor (Billiard *et al.* 2002 in Brinkworth *et al.* 2003).

Brinkworth *et al.* (2003) exposed early life history stages of rainbow trout to dissolved retene (7-isopropyl-1-methylphenanthrene) in an attempt to understand 1) whether there is an early life stage that is particularly sensitive to retene toxicity and 2) whether CYP1A induction is a forerunner of BSD. Tissue retene concentrations were elevated in embryos just after fertilization (possibly a reflection of storage in yolk lipids), but decreased as the fish developed, likely as a result of more efficient metabolic and/or excretory systems (Brinkworth *et al.* 2003). However, retene continued to be taken up by fish post-hatch. Expression of CYP1A, which was first observed during organogenesis, increased steadily until the swim-up stage, suggesting that CYP1A induction was part of the metabolism and excretion of retene. Other signs of BSD (*e.g.,* hemorrhaging, yolk sac edema, and mortality) arose one week after the increase of CYP activity post-hatch, suggesting a relationship between CYP induction and the onset of BSD (Brinkworth *et al.* 2003). In 100% of cases, CYP induction was followed by signs of BSD. The observed prevalence of BSD did not exceed 10% until post-hatch, at which time it increased to 60 to 100% in fish exposed to retene concentrations between approximately 9.0 µg/L and 32 µg/L for 1 to 2 weeks. Based on the level of CYP1A expression and the prevalence of BSD, Brinkworth *et al.* (2003) suggest that the larval stage is most sensitive to retene toxicity. It remains unclear whether this is because embryonic fish are resistant to retene toxicity or because toxic effects experienced by embryos are not expressed until the larval stage.

Incardona *et al.* (2004) exposed zebrafish embryos (pre-hatching) to individual non-alkylated PAHs in an attempt to understand the mechanisms of toxicity leading to signs of BSD. Exposure to 3-ring PAHs (*e.g.,* dibenzothiophene, phenanthrene) alone reproduced the most pronounced effects of crude oil exposure, and induced signs of BSD (*e.g.,* cardiac dysfunction, edema, spinal curvature, craniofacial deformities). These compounds appeared to directly affect cardiac conduction, which then led to effects on cardiac morphogenesis, kidney development, neural tube structure, and craniofacial formation (Incardona *et al.* 2004). While cardiac rhythm returned to normal 48 to 72 hours after exposure ceased, heart morphology had changed (Incardona *et al.* 2004). Conversely, exposure to the 4-ring PAH pyrene resulted in a different syndrome, with effects including anemia, vascular defects, and neuronal cell death, which are similar to the effects observed for strong aryl hydrocarbon receptor (AhR) ligands such as 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) (reviewed in Incardona *et al.* 2004). The effects resulting from exposure to 3-ring PAHs were indistinguishable from those resulting from genetic disruptions to heart development, suggesting that defects associated with PAH exposure result from cardiac dysfunction (Incardona *et al.* 2004).

Fallahtafti *et al.* (2011) examined the role of hydroxylation (*i.e.,* metabolism) in the toxicity of methylphenanthrene (MP) to early life stages of Japanese medaka (*Oryzias latipes*). Metabolism enhanced alkyl PAH toxicity, with some phenols exhibiting a 4-fold increase in toxicity relative to their non-hydroxylated counterparts. Effects were similar in phenol-type metabolites with hydroxylation on the ring distal to the alkylated ring (1MP7, 1MP8) and the non-hydroxylated reference compound (1-MP), and included circulatory effects and edema (Fallahtafti *et al.* 2011). Conversely, exposure to phenol-type metabolites with the hydroxylation on the ring proximal to the alkylated ring (1MP4, 1MP9) and with the OH group on the chain as opposed to the ring (OH-MP, a benzylic alcohol) caused circulatory effects with only minor edema (which was suggested by Carls *et al.* 1999 as the most important indicator of

PAH toxicity). Further, the first group (1MP7, 1MP8) was 5 to 8 times more toxic than the second group (1MP4, 1MP9, and OH-MP) and 2 to 3 times more toxic than the reference, non-hydroxylated compound, which was attributed to structural differences between metabolites (Fallahtafti *et al.* 2011).

Marine Water Quality Guidelines for PAH Compounds

Water quality guidelines from the province of British Columbia, and as presented by the Canadian Council of Ministers of the Environment, all date from before 2000 and all focus on individual PAH compounds, without a framework to integrate PAH mixtures. These guidelines are therefore of limited utility, since PAH compounds rarely occur in isolation. Water quality standards for PAHs and other hydrocarbons from U.S. jurisdictions are similarly limited and dated (*e.g.,* US EPA 1986).

Di Toro *et al.* (2000) followed a process designed to establish water quality criteria by the US EPA and applied this process to individual compounds, while developing an overall framework to evaluate the toxicity of PAHs and other hydrocarbons as complex mixtures. Final chronic values (FCV) derived by Di Toro *et al.* (2000) vary according to Log K_{ow} values for different PAHs. For pure naphthalene, an FCV of 320 µg/L is presented. This decreases for phenanthrene (FCV = 32.4 µg/L), and decreases again for fluoranthene (FCV = 12.2 µg/L). Assuming a mixture of alkylated PAH compounds having a mean Log K_{ow} value of 5.0 and a mean molecular weight around 200 g/mole, the formulation presented by Di Toro *et al.* (2000) would suggest an FCV for TPAH in the range of 14.3 µg/L. French McCay (2002) identified narcosis-based incipient LC_{50} values (*i.e.,* LC_{50} values based on indefinite exposure) of 5 µg/L TPAH for sensitive species and 50 µg/L TPAH for average species exposed to mixtures of mono-cyclic aromatic hydrocarbons and PAHs.

Toxicity Benchmarks for PAH Exposure of Aquatic Receptors

The chronic toxicity of hydrocarbons to aquatic receptors is evaluated in two key ways. Conceptually, exposure for fish, fish eggs, and benthic invertebrates was considered to be primarily to hydrocarbons present in dissolved form. Hydrocarbons are hydrophobic and partition strongly between water and other available non-polar media, including sediment organic matter, and living organisms. Uptake of hydrocarbons from water by living organisms is considered to be regulated primarily by equilibrium exchange processes between water and lipids, and to take place across permeable or vascular surfaces, such as gills or egg membranes. Once inside the organism, hydrocarbons become part of the generalized lipid pool, and may or may not be metabolized. Because some invertebrates (such as molluscs) lack enzyme systems capable of rapidly metabolizing PAHs, it is assumed that hydrocarbons can be accumulated by benthic invertebrates. In contrast, vertebrate species are capable of metabolizing and excreting most hydrocarbon compounds, and bioaccumulation is less pronounced.

Based on information reviewed here, and with particular reference to studies by Brannon *et al.* (2006a), McIntosh *et al.* (2010) and Wu *et al.* (2012), it is concluded that the earliest and least severe symptoms of BSD in fish embryos exposed to TPAH occur at concentrations around 1 µg/L. Mortality of developing salmonid eggs and embryos may occur if the dissolved TPAH concentration in their incubation habitat exceeds 10 µg/L during chronic exposure. Not all species or life stages are equally sensitive. Fish eggs appear to be most sensitive during the first 24 hours following fertilization. At other times during the incubation period or for free-swimming larval fish, short-term exposures to concentrations of TPAH as high as 100 µg/L may not cause adverse effects.

3.4.2 Evaluation of Wildlife Exposure to COPC in Air

In the deterministic oil spill modelling, the diluted bitumen (CLWB) was described using 17 individual pseudocomponents, each one defining a group or class of hydrocarbon compounds having similar characteristics (*i.e.,* aliphatic or aromatic compounds, of similar molecular weight, hydrophobicity, and vapour pressure. Following the principles outlined above, inhalation LC_{50} values were calculated for each pseudocomponent based on their characteristic vapour pressures.

Non-polar narcosis as a mechanism to explain the toxicity of hydrocarbon vapours is highly relevant to oil spills as it provides an integrated model that predicts the toxicity of a wide variety of petroleum hydrocarbon compounds, such as light aliphatic compounds, BTEX and PAHs. Many quantitative structure-activity relationship (QSAR) models have been developed over the last few decades to predict non-polar narcosis endpoints, including lethality and sub-lethal effects. However, these had not generally been extended or refined for consideration of the inhalation pathway prior to the work of Veith *et al.* (2009).

Veith *et al.* (2009) used QSAR-based experimental and reporting criteria to develop a QSAR model to predict non-polar narcosis endpoints for mammalian inhalation. This work included review of the inhalation toxicity literature, and compilation of a database of inhalation endpoints that met the QSAR-based criteria, focusing on 4-hour exposure for a single species of rodent (*i.e.,* rat). Studies for which symptoms were not consistent with non-polar narcosis, as well as studies which considered reactive chemicals or chemicals with known receptor-mediate toxicity were not considered. The final list of compounds included a variety of chemical classes; *i.e.,* aliphatic and aromatic compounds (which are predominant components of petroleum hydrocarbons), as well as halogenated compounds, alcohols and ketones.

Central to the QSAR model developed by Veith *et al.* (2009) is the Ferguson principle, which states that the ratio of the partial vapor pressure needed to produce narcosis to that of the pure chemical should be the same for all non-polar narcotics. Therefore, under steady-state conditions, the partial vapor pressure required for non-polar narcosis should be directly proportional to the partial vapor pressure of the pure chemical. Where the partial vapor pressure required for non-polar narcosis can be converted into a concentration needed to cause lethality (*i.e.,* the LC_{50}), a relationship linking the LC_{50} to the vapour pressure of the pure chemical can be derived.

Based on the compiled database of 4-hour inhalation LC_{50} values for the rat, Veith *et al.* (2009) derived the following linear relationship correlating LC_{50} to the vapor pressure of the pure chemical (with an r^2 value of 0.91):

$$\log LC_{50} = 0.69 \log VP + 1.54$$

where LC_{50} is the chemical concentration in air (mmol/m^3) causing lethality in 50% of exposed test animals over the defined duration of 4 hours, and VP is the vapor pressure of the chemical (mmHg). For the present work, the equation of Veith *et al.* (2009) was adapted to use different units of LC_{50} (mol/m^3) and vapor pressure (Pa) as follows:

$$\log LC_{50} = 0.69 \log VP - 2.94$$

The predicted values are summarized in Table 3.7, and represent concentrations in air that would be expected to result in mortality of 50% of exposed individual mammals. The linear relationship developed by Veith *et al.* (2009) correlating 4-hour inhalation LC_{50} values for the rat to the vapor pressure of the pure chemical is expected to be applicable to other mammalian species. According to Paterson and Mackay (1989) inhalation LC_{50} may be related to a critical concentration in tissue such as the brain. Solubilities of volatile organic compounds in tissue exhibit both water and fat solubilities with solubility in water predominating for the alcohols and solubility in lipids predominating for alkanes (Paterson and Mackay 1989). As such, it would be expected that species with similar brain lipid content would demonstrate similar inhalation LC_{50}. Literature has shown that brain lipid content is of 11% for rats (Pratt *et al.* 1969) and 11.5% for the grey whale (Varanasi *et al.* 1993). Therefore it is expected that the LC_{50} presented below, would be applicable to the rat (original test species) and to other mammalian species of similar brain lipid content such as whales.

Using a GIS the predicted hydrocarbon-in-air concentrations at the water-air interface in each cell of the model domain can be compared to these benchmark values. For each pseudo-component, the predicted concentration in air is divided by the predicted LC_{50} value, to calculate the number of toxic units (TU) present in air. The TU are then summed across the pseudo-components to estimate the toxicity of the hydrocarbon mixture in air.

Table 3.7 Mammalian Inhalation Toxicity Benchmarks for Selected Hydrocarbon Pseudo-Components

Pseudo-Component	Description [1]	MW [2] (g/mol)	Vapor Pressure (Pa)	LC_{50} [3] (µg/m³)
VOL	Volatiles	70.8	9.98E+04	2.34E+08
AR1	Benzene	78.1	1.27E+04	6.23E+07
AR2	Toluene, Ethylbenzene, Xylenes	99.2	2.47E+03	2.55E+07
AR3	Aromatics $>C_8$-C_{10}[1]	120.0	1.27E+03	1.95E+07
AR4	Aromatics $>C_{10}$-C_{12}	130.0	4.14E+00	4.07E+05
AR5	Aromatics $>C_{12}$-C_{16}	150.0	8.72E-03	6.68E+03
AR6	Aromatics $>C_{16}$-C_{21}	190.0	2.13E-05	--- [4]
AR7	Aromatics $>C_{21}$-C_{34}	240.0	9.16E-08	--- [4,5]
AL1	Aliphatics C_6-C_8	100.0	6.38E+03	4.96E+07
AL2	Aliphatics $>C_8$-C_{10}	130.0	6.38E+02	1.32E+07
AL3	Aliphatics $>C_{10}$-C_{12}	160.0	6.38E+01	3.31E+06
AL4	Aliphatics $>C_{12}$-C_{16}	200.0	4.86E+00	7.00E+05
AL5	Aliphatics $>C_{16}$-C_{21}	270.0	1.11E-01	6.98E+04
AL6	Aliphatics $>C_{21}$-C_{34}	390.0	2.59E-06	--- [4,5]
RES1	F4 ($>C_{34}$-C_{50})	570.0	1.00E-10	--- [4,5]
RES2	Resins	825.0	1.00E-10	--- [4,5]
RES3	Asphaltenes	1599.0	1.00E-10	--- [4,5]

Notes:

1. The ranges in this column represent the number of carbon atoms in the hydrocarbon molecule (*e.g.,* aromatics $>C_8$-C_{10} would be molecules containing at least one benzene ring, with specifically 9 or 10 carbon atoms.
2. MW stands for Molecular Weight.
3. LC_{50} stands for lethal concentration (50%).
4. Did not readily evaporate in modelling completed as part of this study (EBA 2013).
5. Non-volatile organic compounds according to Kelly *et al.* (1994).

3.4.3 Evaluation of Exposure to Surface Water Oiling

Potential acute environmental effects on wildlife (*i.e.,* the air-breathing vertebrates: birds, mammals and reptiles) in the spill-affected area are evaluated based on probability of encounter with floating oil, and the amount of oil likely accumulated on an individual animal. This analysis is based on the methodology outlined by French McCay (2009).

Jenssen and Ekker (1991a,b) studied eiders exposed to oil on their feathers at varying doses, finding that metabolism was affected above 20 mL of (crude) oil. However, their review of the literature revealed that exposure to considerably more oil (200-500 ml) is required for significant and potentially lethal effects. Following French McCay (2009), 350 ml is assumed to be a lethal exposure for many wildlife species. Assuming a swimming animal has a width of 15 cm, it would need to swim through 230 m of oil of 10 µm thickness, or 2.3 km of oil at 1 µm thickness, to obtain a dose of 350 mL. This distance spent in oil need not be in a straight line. If an animal swims 10 m/min (0.17 m/sec), 230 m would be covered in 23 min (or 2.3 km would be covered in about 3.8 hours). Thus, a slick thickness of 10 µm is assumed as a threshold thickness for oiling mortality (French McCay 2009). Those animals oiled above a threshold lethal dose are assumed to die, given the low probability for timely capture, treatment and rehabilitation.

The likelihood of encounter with oil would be different for each wildlife type depending on its behavior (*e.g.,* fraction of its time spent feeding in aquatic systems, likelihood of scavenging oil-contaminated food,

movement behaviours). Terrestrial mammals and birds that do not feed in aqueous habitats would likely avoid or not contact oil, except for those attracted to carrion (*e.g.,* bears, foxes, coyotes, wolverines, bald eagles). Semi-aquatic and aquatic mammals such as mink or otter would have a higher potential for exposure to oiling of fur, due to their more aquatic habits. Seabirds would have high potential to be exposed to oil in an oil slick, and are known to be sensitive to such exposures. More fully marine mammals such as seals and whales, which rely on blubber rather than fur for insulation, are relatively insensitive to external oil exposure.

3.4.4 Evaluation for Damage to and Recovery of Oiled Shoreline and Intertidal Communities

The Exxon Valdez oil spill (EVOS) accident of 1989 resulted in the release of some 41,000 m^3 of Alaska North Slope crude oil into PWS and outer coastal areas of the Gulf of Alaska (GOA). Much of the spilled oil (approximately 19,000 m^3, Coats *et al.* 1999) impinged on shoreline and became stranded in intertidal habitat (Spies *et al.* 1996). Shorelines in PWS proximal to the spill site received fresher and more acutely toxic oil, whereas shorelines farther away and in the GOA received more weathered oil of lower acute toxicity. Oil coated rock surfaces and penetrated into cobble-gravel beaches, although most of the oil that stranded did so in the upper half of the intertidal zone, above the zone of greatest biological productivity. Cleanup efforts added to the biological damage caused by the oil, and a practice known as "high-pressure hot-water washing" was particularly damaging. In retrospect, it is important to distinguish between adverse effects caused by oiling, and those caused by cleanup practices that are now discredited.

Due to the nature of the event, and the time required for study design and mobilization in the context of a massive emergency response action, pre-spill biological data describing PWS and the GOA were limited, and insufficient to provide baseline against which spill effects or recovery could be evaluated.

In the aftermath of the EVOS, four major studies were set up to evaluate effects and recovery of the intertidal community. These included:

- the Shoreline Ecology Program (SEP) focusing on PWS, funded by Exxon
- the Gulf of Alaska Study, focusing on the GOA, funded by Exxon
- the Coastal Habitat Injury Assessment (CHIA) funded by the Exxon Valdez Oil Spill Trustee Council (EVOSTC)
- the Hazardous Materials (Hazmat) study funded by the National Oceanographic and Atmospheric Administration (NOAA).

Each of these studies was independent, and while they had similar overall objectives, the specific study design approaches that were followed differed. As a result, the results of the four studies, while generally in agreement, varied in details, and conclusions about the degree of damage, as well as the time-course and degree of recovery, also differed. The following discussion of methods for use in the current DQERA is based upon all four studies, and seeks to integrate their findings, while focusing on effects of spilled oil, rather than the often confounded effects of aggressive cleanup techniques, particularly high-pressure hot-water washing.

3.4.4.1 Definitions of Recovery

Various teams working on the damage and recovery assessments of the EVOS adopted different definitions for recovery. As the definition of recovery usually guided study design, the various definitions and their implications became very important in the assessment process.

The SEP (funded by Exxon) assumed that recovery of a biological resource can be defined in terms of specific derived parameters (statistical variables) related to community structure. Two sub-sets of parameters and methods were used. Univariate statistics were applied to community metrics including the number of individuals present, the number of species present, and the community diversity. Recovery was then considered complete when negative effects were no longer present. This was deemed to be the case when the means of the derived parameters did not exhibit a statistically significant difference from values obtained at unoiled reference locations (often based on analysis of covariance (ANCOVA), with initial oiling as the main effect, and concomitant variables such as wave exposure, grain size, or TOC as covariates). In addition, multivariate analysis (De-trended Correspondence Analysis) was used to develop a confidence envelope around the community structure of unoiled reference sites, and recovery was deemed to have occurred when the oiled sites occupied space within the confidence envelope (Page *et al.* 1995).

The NOAA/Hazmat study (Coats *et al.* 1999, Skalski *et al.* 2001) summarized Exxon's definition of recovery as "the reestablishment of a healthy biological community characteristic of the area", whereas the State and Federal resource trustees felt that "recovery would occur when the Sound looks as it would have if the spill had not occurred". Noting that no pre-spill baseline information was available for the intertidal communities exposed to spilled oil from the Exxon Valdez, Skalski *et al.* (2001) concluded that recovery could not be based on the notion of a return to intertidal population levels and composition prior to the spill, because the pre-spill levels were unknown. Coats *et al.* (1999) and Skalski *et al.* (2001) proposed that recovery can be considered complete when impacted intertidal populations eventually begin to track or parallel the control site profiles. Under this scenario, the statistical tests of recovery are equivalent to tests for parallelism. Currents that direct oil to certain shorelines may also be responsible for the distribution of larvae and nutrients. Because these site differences cannot be randomized across treatment designations, recovery assessments based on the direct comparison of mean population levels are statistically untenable (Skalski *et al.* 2001). Differences in population or community levels between reference and exposed sites, therefore, are not a consideration in assessing recovery: only the relative patterns of the temporal trends are of interest. This approach recognizes that reference and exposed sites need not be identical to begin with, and may not be identical when recovery is complete, but that they should show similar responses to external stressors over time.

The EVOSTC (2010) still considers the intertidal community to be "recovering" from the effects of the oil spill. To understand this perspective, it is necessary to first evaluate the recovery objective as articulated by the EVOSTC (2010): "intertidal communities will have recovered when such important species as *Fucus* (marine algae/seaweed) have been re-established at sheltered rocky sites, clams and mussels at soft or mixed sediment beaches are not contaminated by residual oil, the differences in community composition and organism abundance on oiled and unoiled shorelines are no longer apparent after taking into account geographic differences, and the intertidal and nearshore habitats provide adequate, uncontaminated food supplies for predators and subsistence users". In the case of the EVOSTC, the principal barrier to recovery for the intertidal community appears to be the presence of small amounts of "lingering oil" in certain types of habitat within the intertidal zone.

3.4.4.2 Effects of Spilled Oil in Prince William Sound

In 1989, survey teams found oil on about 16% (783 km) of the approximately 5,000 km of shoreline in PWS (Neff *et al.* 1995). The extent of oiling declined substantially between 1989 and 1992, so that in 1991 only about 96 km of shoreline was deemed to remain oiled, and in 1992 only about 10 km remained oiled. Most of the oil was found in the biologically least productive upper intertidal and supratidal zones. The shoreline characteristics in the western sound are predominantly steep bedrock cliffs interspersed with deep bays and pocket beaches with boulder, cobble, and occasionally sandy sediments (Neff *et al.* 1995).

During 1989, cleanup efforts focused on removing the bulk of oil from the shorelines as quickly as possible (Neff *et al.* 1995). Cleanup activities included manual pick-up of oil with sorbent pads, low- and high-pressure washing with cold and hot (up to 60°C) water, mechanical tilling and removal of oiled sediments, and bioremediation (Harrison 1991), with cleanup efforts focusing on the most heavily oiled upper and middle intertidal zones. However, as the damaging effects of high-pressure warm-water washing became known, this practice was discontinued in subsequent years in favour of less damaging methods (Neff *et al.* 1995).

In 1989, shoreline surveys covered 5,500 km in PWS and the GOA. Oiling intensity was classified (Neff *et al.* 1995) as heavy, moderate, light or very light, and also by the width of shoreline affected (>6 m, 3 to 6 m, or <3 m). Not all shorelines in western PWS were oiled. The most heavily oiled shorelines faced east or north, and the distribution of oil on shorelines, which reflected prevailing winds, currents and exposure, may have also biased the allocation of oiling to shoreline sections that differed from nearby un-oiled sections that otherwise would have served as reference areas.

Rates of oil removal from shorelines varied because of differences in wave exposure, as well as the intensity of cleanup effort. Surface oiling tended to decrease most rapidly on lightly oiled shores, exposed shores, and shores that received intensive cleanup shortly after the spill (Neff *et al.* 1995). Overall, however, natural processes such as storms during the winters of 1989-90 and 1990-91 played a large role in the overall recovery from oiling. By the spring of 1990, most of the coastal segments in PWS that were initially heavily oiled were reduced to the Narrow, Very Light, or No Oil categories (Neff *et al.* 1995). Two types of residual oil were also important after 1989. Subsurface oil confined to small isolated deposits in the upper intertidal and supratidal zones became known as “sequestered” deposits. Oil in these deposits has been shown to persist in a relatively unweathered form on a timescale of decades. Oil was also trapped in lower intertidal areas where mussel beds created conditions such that oil could become trapped beneath the surface layer of the mussel bed, providing a longer-term source of residual oil exposure for such mussels. Mussel beds were often left undisturbed following the EVOS because of concern about further damaging this important food resource for wildlife species.

The SEP study of effects and recovery of the intertidal community (funded by Exxon) was designed so that the results could be extrapolated to the entire spill zone in PWS (Boehm *et al.* 1995). This required the development of a stratified random sampling (SRS) approach as a basis for spatial generalization. The program comprised 64 randomly chosen study sites representing four major habitat types and four levels of oil exposure, as well as periodic sampling of 12 subjectively chosen “fixed” sites. Sampling followed the sediment quality triad approach, with measurements and observations of oiling levels, sediment toxicity, and biological community status.

Both observations and chemical measurements showed that exposure to hydrocarbons decreased rapidly with time after the initial oiling. Mean TPAH levels were 200 mg/kg in the upper intertidal (although much lower in the middle and lower intertidal areas), and these decreased by approximately two orders of magnitude in the first two years. Toxicity to marine amphipods was generally observed only at TPAH concentrations greater than 4 mg/kg in both 1990 and 1991 (Boehm *et al.* 1995).

The biological assessment of PWS beaches following the EVOS presented considerable challenges. Four main habitat types (exposed bedrock/rubble; sheltered bedrock/rubble; boulder/cobble and pebble/gravel), each with different elevational sub-habitats, as well as four oiling levels (heavy, moderate, light and none). However, the characteristics of such shorelines tend to be dominated by “patch dynamics”, and the high biological diversity means that community assessments result in a very large number of species being recorded, with most occurring in only a few samples (Gilfillan *et al.* 1995a). This results in an extremely high level of baseline variability (or “noise”) against which a signal associated with oil spill effects must be detected. The wide variety of physical substrates present (from bedrock to sand and silt) also means that multiple sampling methods must be applied (*e.g.,* quadrat-based, boulder scrapes, core samples, etc.) further complicating inter-comparison and analysis.

The SEP did not report results for 1989, so estimates of the magnitude of adverse effects associated with oiling intensity are not available. The SEP also did not differentiate between habitats that were subject to aggressive cleaning, so effects integrate both effects of spilled oil, and effects of oil spill recovery activities. The SEP concluded that although spill effects were still evident in the summer of 1990, most of the shorelines had already recovered by that time (*i.e.,* statistically significant differences were rarely detected between shorelines that had been oil-exposed in 1989 and shorelines that had not). By 1991 the oil was dramatically reduced in concentration on most PWS shorelines. Toxicological data showed that sediments were toxic for a period of several months to one year. Univariate and multivariate analyses showed that 73% to 91% of the initially affected shorelines had recovered by the summer of 1990 (Gilfillan *et al.* 1995a).

A second major study of oil effects in PWS was that reported by Coats *et al.* (1999) and Skalski *et al.* (2001) under the patronage of the U.S. National Oceanographic and Atmospheric Administration (NOAA), Hazardous Materials Respond Division (HazMat). This study used a stratified random sampling design to select sampling locations along transects in the upper, middle and lower tidal levels. The upper level was established in the lichen zone, above where rockweed (*Fucus* spp.) prevails. The middle transect was located in the *Fucus* zone, and the lower transect was positioned just below the *Fucus* zone, near mean low water. Epibiotic sampling was conducted within replicate quadrats established along a transect within each elevational zone. These included counts of individual organisms and estimates of species coverage, as well as samples of infauna taken using corers in the lower intertidal transect only. Three categories of sites were considered, including unoiled sites, sites that were oiled but not treated, and sites that were oiled and treated with high-pressure hot-water washing. The same sites were visited repeatedly between 1989 and 1997 to document the magnitude of oil and cleanup effects, as well as the recovery process (Skalski *et al.* 2001).

The NOAA/HazMat study (Coats *et al.* 1999, Skalski *et al.* 2001) concluded that the greatest initial oil coverage occurred along the middle intertidal zone in 1989, but oil presence in the upper intertidal increased between 1989 and 1990. In both zones, oil was substantially gone by 1992, with traces (<1% coverage) remaining in the upper intertidal zone until about 1995. Mean rockweed coverage varied between approximately 30% and 70% at unoiled sites, and was initially reduced to about 25% coverage at oiled sites, and to about 2% coverage at sites that were both oiled and cleaned with high-pressure hot-water washing. Areas treated either way recovered to have coverage similar to reference sites by 1992, but a subsequent decline in cover at treated sites by 1995 was attributed by some workers to a simultaneous senescence of rockweed that formed a single-aged cohort after beaches were subjected to pressure washing. In this sense, oil exposure alone caused significant, but not catastrophic decline in the rockweed (*Fucus*) community, from which recovery was rapid unless damage was exacerbated by pressure washing.

The abundance of infauna was initially lower at hot-water washed sites than at sites that were oiled but not washed, and both showed reduced abundance when compared to reference sites (Coats *et al.* 1999). After two years of low infaunal abundance in 1990 and 1991, a steep rise in infaunal abundance occurred and was complete by 1992. The taxa whose abundance increased the most included crustaceans (*Cumella vulgaris*), gastropods (*Fartulum occidentale*) and polychates (*Eteone longa*, *Ophelia limcina*, *Syllis alternata*, *Fabriciola berkeleyi* and *Laphania boecki*). After the initial recovery, infaunal abundance at washed sites remained lower than at untreated sites, and the community composition was distinctly different (Coats *et al.* 1999). Although it is possible that this reflected an a priori difference between the washed and unwashed sites, the likely explanation is that hot-water pressure-washing resulted in the erosion of fine sediments (including organic matter) from the intertidal zone, and that the washed areas could not support the same species assemblage or biomass pending the recovery of the fine sediment fraction. No lingering effect was observed in the oiled but unwashed sites, where recovery had occurred by 1992 (Coats *et al.* 1999).

Recovery of the intertidal epifauna (living on top of rock surfaces, instead of within the sediment matrix) was assessed using similar methods to the infauna. The epifauna were further categorized as sessile (living fixed to the rock) or motile (able to move around). The abundance of motile invertebrates was apparently reduced by about half at oiled but untreated sites during 1989, but recovery was complete by 1991 (Coats *et al.* 1999, Skalski *et al.* 2001). At sites that were oiled and treated, the initial reduction in invertebrate abundance was about two orders of magnitude, but recovery also occurred by 1991. These effects were most pronounced (Coats *et al.* 1999, Skalski *et al.* 2001) for limpets (*Lottiidae*) and periwinkles (*Littorina* spp.). There was no statistically significant evidence of effects on barnacles or mussels, but this evaluation was potentially confounded by the difficulty of distinguishing recently dead from live animals (Coats *et al.* 1999). In the lower intertidal zone, effects were also much larger at oiled sites that were treated with high-pressure hot-water washing. Again a reduction in abundance by about one half appears to apply to sites that were oiled but not treated, whereas reductions by a factor of 10 or more would apply to sites that were oiled and subsequently treated (Skalski *et al.* 2001). Regardless of the magnitude of initial effects, recovery of abundance (as measured by the parallelism test) occurred between 1991 and 1992 (Coats *et al.* 1999, Skalski *et al.* 2001).

Peterson (2001) reviewed the acute, chronic, and indirect effects of the EVOS on marine biota. His synthesis noted that (with the exception of the NOAA/HazMat study) many of the major studies of oil spill response and recovery were confounded in that it was not possible to separate the effects of cleanup from the effects of oil exposure. In addition, all of the studies lacked pre-spill information and used contrasts of oiled and unoiled shoreline segments to assess effects and recovery of the spill. A potential bias, in that oil exposed shores tended to be those exposed to greater current flux, also affected comparisons of oiled and unoiled shorelines (Peterson 2001). The EVOS was not a controlled experiment. Oil quantity and quality varied among strata (shoreline segments as well as different elevations in the intertidal zone), and biota also varied among strata.

Peterson (2001) compared the effects of the EVOS with other spills with respect to "greening" of the shoreline (a reflection of the colonization by annual or ephemeral algal species of rock space cleared of *Fucus*). The rapid proliferation of such algae is facilitated by the elimination of many of the grazer species (such as gastropods). There was, however, only limited evidence of "greening" following the EVOS, and speculative explanations for this include a lack of observation; or alternatively lower mortality of grazers than had occurred after other oil spills, such as the Torrey Canyon spill, where more toxic detergents and dispersants were used to clean shorelines.

Peterson (2001) concluded that much of the immediate loss of intertidal invertebrate fauna was a consequence of hot-water high-pressure washing. He judged that toxicity may have played a role, along with the physical effects of smothering under a layer of oil. However, indirect effects (such as desiccation or increased visibility to predators once the protective layer of *Fucus* was removed) may also have been involved.

3.4.4.3 Effects of Spilled Oil in the Gulf of Alaska

Another 1,300 km of the approximately 10,000 km shoreline of the western GOA was also oiled to some degree (Neff *et al.* 1995). Forty-eight sites in the GOA region were sampled in 1989 to assess the effects of the EVOS, and selected sites were re-sampled during 1990 to evaluate recovery (Gilfillan *et al.* 1995b). Owing to the greater distance and travel time from the spill location, shoreline oiling was generally lighter, and oil was more weathered, in the GOA when compared to PWS, and biological communities experienced lower levels of effects, which tended to be more localized, in keeping with the patchy and discontinuous nature of oiling in the GOA. Adverse effects were generally limited to middle- and upper-intertidal areas (Gilfillan *et al.* 1995b).

Field investigations in 1989 included sediment or rock wipe samples for hydrocarbon analysis, percent cover observations of macroscopic flora and fauna, removals of macroscopic flora and fauna for

community evaluation, analysis of sediment toxicity using amphipod tests, and analysis of mussel tissues for hydrocarbons. Similar evaluations were repeated in 1990. The sites were not chosen randomly, and statistical comparisons were undermined by sub-optimal distribution of oil (all sand sites were oiled in some degree, and all mud habitats were unoiled; Gilfillan *et al.* 1995b). As a result, statistical power was not high.

Notwithstanding the statistical limitations, it was found that mean hydrocarbon concentrations in GOA sediments were 10 to 100 times lower than in PWS, with the highest mean concentrations being found in the splash and upper intertidal zones (Gilfillan *et al.* 1995b). Oiling levels were very uneven (patchy) in the GOA. Oil in the splash zone and upper intertidal was also found to be more weathered in the GOA than in PWS.

Analysis of mussel tissues showed lower concentrations in mussels from the GOA than in PWS, and comparison of mussel tissue to sediment PAH concentrations showed lower bioavailability of PAHs in 1990 than in 1989 (Gilfillan *et al.* 1995b). Consistent with the lower loadings and greater weathering, sediments in the GOA were much less toxic to amphipods in 1989 than sediments in PWS (Gilfillan *et al.* 1995b). Mortality of amphipods in oiled GOA sediments was not significantly different than in control sediments.

With respect to faunal abundance (biological cover), significant decreases were observed at moderately to heavily oiled bedrock and boulder/cobble sites. Organism abundance and species richness showed consistent declines with increased oiling. However, effects on diversity were not as marked (Gilfillan *et al.* 1995b).

3.4.4.4 Environmental Effects of Lingering Oil from the EVOS

In addition to recovery, the EVOSTC (2010) also focus on “lingering oil” that is still present in some intertidal areas within the spill zone. Studies have shown that some beaches contain pockets of lingering oil, which has been sequestered and protected from weathering deep in porous beach substrates. These pockets of lingering oil represent a very small fraction of the total amount of spilled oil, and the environmental effects of this oil are very small or negligible in the broader context of PWS and the Gulf of Alaska. Approximately 350 km of shoreline was considered to be heavily oiled, and both the spill and the subsequent cleanup activities had substantial effects on the flora and fauna of the intertidal zone. The intertidal sediments captured approximately 40 to 45% of the spilled oil, and most of these beaches were cleaned using a variety of methods. By 1992, approximately 10 km of beaches remained uncleaned. Most of the oil that initially washed up on beaches was dispersed back into the ocean during the three years following the spill, so that by the end of 1992 only about 2% of the initial volume of spilled oil remained on the beaches of PWS (Wolfe *et al.* 1994).

Boehm *et al.* (2008) systematically collected 678 intertidal sediment samples from areas that were most heavily oiled in 1989, and a further 66 samples from two sites that were known to be active otter foraging sites. Only 19 of the 744 pits contained heavy oil residue, and 34 contained moderate oil residue. Where oil was found, it was typically highly weathered. Boehm *et al.* (2008) noted that previous studies have shown no significant bioavailability of PAHs in buried and sequestered oil residues, and that important amounts of sequestered oil are extremely patchy and uncommon. Unweathered residues are rare, and generally confined to the upper intertidal zone, removed from areas of biological productivity. These residues remain because they are sequestered and, as a result, are not bioavailable or released to the environment at levels that pose a risk to biota or humans using the shoreline (Boehm *et al.* 2008).

In addition to oil that was sequestered in porous beach substrates following the EVOS, some crude oil was sequestered in mussel beds, among the mats of mussel byssus, cobbles and fine sediment (Boehm *et al.* 1996). Ebert and Lees (1996) as cited by Peterson (2001) reported data from mid-intertidal mussel beds that showed higher levels of PAHs in oiled and hot water washed areas, than in oiled but

untreated areas, which were themselves higher than reference areas. Again, high-pressure hot-water washing appears to have created conditions that were in some ways worse than if oil had been left alone on the beaches. The presence of crude oil within mussel beds created conditions that led to increased bioaccumulation of PAHs by mussels, with the potential for higher levels of exposure for birds and mammals, such as black oystercatcher, harlequin duck, and otter, that consumed the mussels. Such exposure was confirmed for several consumer species through cytochrome P450 analysis (EVOSTC 2010), although the extrapolation of biological effects from such exposure biomarker data is highly uncertain.

Boehm *et al.* (1996) found that areas of residual oil trapped in mussel beds comprised less than 3% of the available mussels in two areas that were heavily oiled following the EVOS, and concluded that the estimated PAH dosage to consumer organisms was one to three orders of magnitude lower than doses known to cause sublethal effects in surrogate species. This conclusion was independently examined and confirmed by Harwell *et al.* (2010).

3.4.4.5 Estimating Crude Oil Retention and Benchmarks for Effects on the Intertidal Zone

As part of the support for spill modeling associated with the assessment of accidental spills of diluted bitumen, Coastal & Ocean Resources (2013) prepared a shoreline GIS dataset, and then provided algorithms for estimating initial oil retention, should a spill of diluted bitumen reach the shoreline. Shoreline attribute datasets from British Columbia and Washington State were combined in GIS, and selected attributes were attached to each shore unit. The combined dataset extended from northern Vancouver Island to Grays Harbour in Washington State and includes Puget Sound and the Strait of Georgia. Selected attributes included coastal class (summarizing the overall character of the shore unit), degree of exposure to waves, unit length, width and area of the intertidal zone, the shoreline type and upper intertidal substrate type, and an oil residence index (ORI). Areas of saltmarsh were also identified from British Columbia and Washington State ShoreZone datasets. A total of 15,911 km of shoreline was classified, with unit lengths subdivided to <100 m, resulting in 172,000 unique shoreline segments for modeling.

Within the GIS, the data were reduced to thirteen “spill shore types” representing combinations of substrate type and wave exposure (Coastal & Ocean Resources 2013). One important assumption incorporated through the spill shore types was that low energy shorelines almost always have a fine subsurface substrate (mud or sand), even when the surface substrate is pebble, cobble or boulder. As a result, penetration of crude oil on low energy shorelines will be limited, due to the barrier provided by the underlying fine substrate. On high energy shorelines, coarse substrates may extend to greater depth, increasing permeability and the potential for long-term retention of stranded crude oil (Coastal & Ocean Resources 2013).

The spill shore types as defined for the Project by Coastal & Ocean Resources (2013) are described in Table 3.8, along with information on their prevalence within the overall study area, and their potential to retain stranded diluted bitumen. Oil retention depends upon a number of factors, and Coastal & Ocean Resources (2013) describe the assumptions and underlying data that were used in order to derive estimates of the initial shoreline oil retention capacity (L/m^2). Oil penetration and retention into sediments is substantially related to oil viscosity (Coastal & Ocean Resources (2013). Diluted bitumen, particularly once weathered, has relatively high viscosity (ranging from 2,000 to 100,000 cP, and even higher should it emulsify; Belore 2010). Oil adhesion values were assumed to be similar to those of Bunker C or IFO-180 type oils (Coastal & Ocean Resources 2013).

Based on the information reviewed from the EVOS, the following biological effects classes and effect durations were assumed for shoreline communities (intertidal and subtidal algae and invertebrates, see Table 3.9). At low levels of initial oiling (<10% of initial oiling capacity), effects on intertidal algal and invertebrate communities may be very light, and would be difficult to discern using statistical methods due

to the high level of variability found in the intertidal zones. Such areas would be judged to have low levels of effect magnitude, and recovery would occur rapidly. At higher levels of initial oiling intensity, greater effect magnitude will be observed. Time to recovery may also vary by habitat type, with those habitats having greater potential to retain oil (generally, sites having low wave exposure and more porous substrates) having longer effect duration and time to recovery.

To summarize the effects of shoreline oiling in a north-Pacific environment on intertidal flora and fauna, the intensity of oiling after the EVOS was spatially variable, ranging from heavy and wide slicks, to thin and discontinuous oiling. Most of the oil affected the upper intertidal zone, with much lower levels of oil deposition on sediments in the lower intertidal and subtidal areas. Effects of oil alone caused mortality of algae and invertebrates, in the intertidal zone, but not the more severe damage that was associated with shoreline clean up activities, notably the high-pressure warm-water washing technique. Although the most serious effects were not well documented during 1989, most researchers documented rapid recovery, already underway in 1990, so that the intertidal zone was deemed to be largely recovered by 1991 or 1992. Effects that have been characterized as playing out over longer periods of time (such as subsequent simultaneous senescence of even-aged *Fucus* stands that colonized rock surfaces after aggressive cleanup efforts) appear to be largely attributable to now-discredited remedial actions such as hot-water high-pressure washing.

Table 3.8 Shore Types with Total Oil Retention Estimates as Defined for the Project

Site Exposure	Upper Intertidal Substrate	Spill Shore Type	% Total Shoreline Length	Initial Oil Retention Capacity (L/m^2)
Low	Rock	Rock, low energy: assumed to be impermeable.	23%	5
	Rock with pebble or cobble veneer	Rock with veneer, low energy; a discontinuous veneer of pebble, cobble or boulder over rock.	8%	6
	Pebble veneer	Pebble veneer over sand; a single layer of pebbles overlying sand, typical of low energy shorelines; stranded oil may attach to pebble but sand in subsurface limits penetration.	13%	9
	Cobble or boulder veneer	Coarse veneer over sand; a single layer of cobbles or boulders overlying sand; sand limits subsurface penetration.	14%	15
	Sand or mud	Sand or mud which typically has high water content and limits viscous oil penetration.	9%	15
	Rip-Rap	Coarse boulders or sometimes concrete rubble that is commonly used as shore protection.	4%	35
	Marsh	Marsh.	12%	13
	Wood	Wood bulkheads, generally assumed to be pilings and therefore somewhat porous.	0%	11
High	Rock	Impermeable rock surfaces; joint and fracture patterns may allow some oil retention.	8%	5
	Rock with coarse veneer	Boulder and cobble overlying bedrock creates potential for stranded oil retention.	1%	45
	Boulder, cobble beaches (also includes some rip-rap sections	Coarse boulder or cobble beaches assumed to have high penetration potential; may include coarse beaches associated with rock platforms; although high energy, penetration may result in lengthy persistence.	2%	65
	Sand with pebble, cobble or boulder	Combinations of sand and various forms of gravel (pebble, cobble, boulder); and matrix is assumed to minimize penetration.	2%	8
	Sand	High energy sand beaches; sand will limit viscous oil penetration; sand is likely to be highly mobile so has the potential to bury stranded oil.	5%	8

Source:
Compiled from Coastal & Ocean Resources (2013).

Table 3.9 Biological Effect Magnitude and Duration Associated with Initial Oiling Intensity

Site Exposure	% of Maximum Initial Oil Retention Capacity	Biological Effect Magnitude (% Initial Loss of Community)*	Time to Recover from Biological Effect (years)*
Low	>90 - 100	90	2 – 5
	>50 - 90	50	2 – 5
	>10 - 50	25	2
	>0 - 10	10	<1
High	>90 - 100	90	2 – 3
	>50 - 90	50	2 – 3
	>10 - 50	20	1 – 2
	>0 - 10	10	<1

Note:
* Assumes appropriate remedial actions are taken, and inappropriate shoreline cleaning methods are not employed.

EVOSTC (2010) acknowledge that “by 1991, in the lower and middle intertidal zones, algal coverage and invertebrate abundances on oiled rocky shores had returned to conditions similar to those observed in unoiled areas. However, large fluctuations in the algal coverage in the oiled areas caused a subsequent alteration in community structure. The *Fucus* canopy was initially eliminated in most of the areas that underwent extensive cleaning, thereby removing the protection provided by this alga to intertidal organisms from predation, desiccation and abrasion. This early eradication of *Fucus* led to instability of this alga's subsequent populations because the single-aged stands present after re-colonization of the habitat were susceptible to large synchronous die-offs. Until a broader distribution of mixed-aged stands is established, this cycle may continue for many generations. Meanwhile, full recovery of *Fucus* is crucial for the recovery of intertidal communities at oiled sites, because many intertidal organisms depend on the shelter this seaweed provides”. This evaluation makes it clear that the ongoing effects of the oil spill post-1991 were primarily associated with cleanup effects, and not effects of the spilled oil alone.

Initial effects on the flora and fauna of the intertidal zone would occur at all tidal levels and in all types of habitats, although the heaviest oiling would be expected in the upper intertidal zone. Dominant species of algae and invertebrates including rockweed, limpets, barnacles, mussels, periwinkles, polychaete and oligochaete worms would be directly affected.

The degree of injury would be correlated with the intensity of oiling (taking into consideration the amount of oil loading and the degree of coverage). Within one to two years on exposed shorelines, and within two to five years on protected shorelines, the algal coverage and invertebrate abundance on oiled shorelines would be expected to return to conditions similar to those observed in unoiled areas.

3.4.5 Sediment Quality Guidelines for Petroleum Hydrocarbons

There are few published regulatory guidelines for petroleum hydrocarbons in sediment. This is due in part to the complexity of hydrocarbon chemistry, which can significantly affect toxicity. This section will explore theoretical and empirical measures of hydrocarbon toxicity in freshwater and marine sediments sediment, in order to facilitate the identification of benchmarks below which adverse effects are unlikely.

Routine analytical laboratory detection limits for petroleum hydrocarbons in sediment typically range from 2.5 to 15 mg/kg, as follows:

- C_6 - C_{10} (less BTEX): 2.5 – 10 mg/kg
- $>C_{10}$ - C_{16}: 5 – 10 mg/kg
- $>C_{16}$ - C_{21}: 5 – 10 mg/kg

- >C_{21} - <C_{32}: 5 – 15 mg/kg
- >C_{34}: 10 – 15 mg/kg
- Modified TPH: 15 mg/kg.

On this basis (and assuming that crude oil constituents reaching sediment would tend to be weathered and depleted in BTEX and light aliphatic and polycyclic aromatic hydrocarbon constituents), total petroleum hydrocarbon concentrations in sediment that are below about 15 mg/kg are typically around or at the detection limit, or are non-detectable. Marine sediments in urban areas typically have a measurable background concentration of petroleum hydrocarbons. This background is derived from a number of sources, including but not limited to runoff from urban land areas (*e.g.,* runoff from roads and parking lots), losses of hydrocarbons such as fuels and lubricating oils used by marine shipping, and spills. In the vicinity of New York Harbour, the median TPH concentration was 1,360 mg/kg, with a range of 25 to 12,600 mg/kg, and the predominant source appeared to be motor oil (Brownawell *et al.* 2007), presumably from stormwater runoff.

The State of Massachusetts (Battelle 2007) published benchmarks for petroleum hydrocarbon fractions in sediment, based on the non-polar narcosis model, and assuming equilibrium partitioning between hydrocarbons and sediment organic carbon. These benchmarks, which are intended to protect a sensitive species from chronic effects, can be compared (Table 3.10) to values previously developed by Verbruggen (2004), and values calculated independently following the model developed by Di Toro and McGrath (2000). All three sets of values are based upon the equilibrium partitioning of TPH fractions between sediment organic carbon (f_{oc}, assumed to be 1% of dry weight) and sediment pore water, and assume a non-polar narcosis mode of action for petroleum hydrocarbons. However, all three sets of benchmarks reference different toxicity databases, and differ in the approach that they take in order to achieve the endpoints that they seek to protect.

MADEP (Battelle 2007) calculated LC_{50} values for representative chemicals and then used an application factor to convert the mean LC_{50} value to a chronic value representative of a sensitive species. Verbruggen (2004) sought to protect the 50th percentile species from very slight effects (for example, a 10% reduction in body mass). Di Toro *et al.* (2000) developed a toxicity model for hydrocarbons based on the 5th percentile species in an acute toxicity sensitivity distribution, and then used an application factor to estimate a chronic value.

Table 3.10 Summary of Battelle (2007), Verbruggen (2004), and the Target Lipid Model (TLM, following Di Toro *et al.* 2000 and Di Toro and McGrath 2000) Effects Benchmarks for TPH (mg/kg dry sediment, normalized to 1% sediment organic carbon)

Battelle (2007) (Chronic)		Verbruggen (2004) (Chronic)		TLM (Chronic)		TLM (Acute)	
Aliphatic		**Aliphatic**		**Aliphatic**		**Aliphatic**	
C_5-C_8	15.9	C_5-C_6	1.6	C_7-C_8	9.8	C_7-C_8	50
C_9-C_{12}	27.2	C_7-C_8	1.5	C_9-C_{10}	14	C_9-C_{10}	70
C_{13}-C_{18}	55.4	C_9-C_{10}	1.4	C_{11}-C_{12}	18	C_{11}-C_{12}	94
C_{19}-C_{36}	98.8	C_{11}-C_{12}	2.6	C_{13}-C_{16}	26	C_{13}-C_{16}	130
		C_{13}-C_{16}	28	C_{17}-C_{21}	NA[1]	C_{17}-C_{21}	NA[1]
				C_{22}-C_{34}	NA[1]	C_{22}-C_{34}	NA[1]
Aromatic		**Aromatic**		**Aromatic**		**Aromatic**	
C_6-C_8	5.3	C_5-C_7	3.9	C_6-C_8	<10	C_6-C_8	<50
C_9-C_{12}	2.3	C_8	4.4	C_9-C_{10}	11	C_9-C_{10}	58
C_{13}-C_{16}	1.3	C_9-C_{10}	4.9	C_{11}-C_{12}	13	C_{11}-C_{12}	64
C_{16}-C_{36}	0.4	C_{11}-C_{12}	5.6	C_{13}-C_{16}	15	C_{13}-C_{16}	76
		C_{13}-C_{16}	6.8	C_{17}-C_{21}	20	C_{17}-C_{21}	100
		C_{17}-C_{21}	8.8	C_{22}-C_{34}	27	C_{22}-C_{34}	140
		C_{22}-C_{35}	20				
Sum of TPH[2]	**206.6**	**Sum of TPH[2]**	**89.5**	**Sum of TPH[2]**	**163.8**	**Sum of TPH[2]**	**832**

Notes:

1 The aliphatic TPH fractions heavier than C_{16} are considered to be insufficiently soluble to cause toxicity.

2 In principle, the benchmarks above should be treated individually, and HQ values summed, rather than summing the individual hydrocarbon fraction benchmarks.

In addition to theoretical investigations, some field studies exist that can help inform the development of benchmarks for petroleum hydrocabons in sediment. Field studies may be based upon gradients of exposure to hydrocarbons that are present in the environment due to historical spills or ongoing chronic releases; or may be based upon experimental additions of hydrocarbons to test plots or other experimental units.

Nance (1991) studied natural benthic invertebrate assemblages exposed to a gradient of weathered crude oil in New Bayou, Texas, a tidal estuary containing a discharge point for produced water from oil and gas production activities. Local TPH accumulations of more than 10,000 mg/kg were observed in sediments, although most sampling stations had TPH concentrations less than 1,000 mg/kg. A sediment TPH concentration of 2,500 mg/kg was found to reflect the average value needed to depress population abundance, which was markedly depressed in the vicinity of the produced water outfall, although a zone of stimulated abundance was found both upstream and downstream from the discharge point. Community diversity showed a similar threshold concentration for effects of TPH. Overall, Nance (1991) concluded that areas characterized as within the zone of depression had an average sediment TPH concentration above 2,000 mg/kg. Moderate depression effects were observed at TPH concentrations between 2,000 and 3,500 mg/kg, while major depression effects were observed at those stations that averaged TPH concentrations above 5,000 mg/kg. At low concentrations, the presence of hydrocarbons can provide an energy subsidy, resulting the development of enhanced microbial communities that in turn provide food and energy for higher trophic level consumers. Nance (1991) estimated that within New Bayou, the zone of stimulation (based upon the abundance of benthic invertebrates) was approximately five times larger than the zone of depression, and that the benthic gain (again based upon abundance of benthic invertebrates) overshadowed the benthic loss by a factor of about 2.2.

Rozas *et al.* (2000) seasonally sampled fish and benthic invertebrates exposed to weathered TPH originating from spills of gasoline, home heating oil and crude oil present in salt marshes of upper Galveston Bay, Texas. Concentrations of TPH were generally low (approximately 75% of samples contained TPH concentrations <200 mg/kg), although TPH concentrations up to 7,833 mg/kg were

measured. They found potential relationships between sediment TPH and abundance for very few species of fish or invertebrates. This lack of significant results was considered notable given that at least three significant results would be expected by chance alone among the 63 tests performed. Of 30 abundant taxa examined in fall, only one species (marsh grass shrimp) showed a significant negative relationship with sediment TPH. In contrast, significant positive relationships were found between infaunal densities and TPH concentration for total annelids, total oligochaetes, and *Streblospio benedicti*. In spring, 33 taxa were examined and significant negative relationships between abundance and sediment TPH concentration were found for four taxa (including two life stages of the brackish grass shrimp, and two species of annelid), and positive relationships were found for one polychaete, the mollusk *Geukensia demissa*, and total mollusks. It was concluded that background levels (generally <500 mg/kg) of weathered TPH in marsh sediments did not affect habitat use by most estuarine organisms.

Pettigrove and Hoffmann (2005) added synthetic motor oil to clean sediments to simulate hydrocarbon pollution in urban streams, and to study the effects of high molecular weight ($>C_{16}$) hydrocarbons on benthic invertebrate communities. They found that threshold effects (depressed abundance of sensitive species) began at a concentration of 860 mg/kg, and that TPH concentrations ranging from 1,858 to 14,266 mg/kg resulted in a significant reduction in the total numbers of taxa and abundance. Based upon these results, they concluded that low level (860 mg/kg) TPH-polluted sediments might increase the abundance of opportunistic species, whereas TPH concentrations between 860 and 1,870 mg/kg are likely to reduce the abundance of TPH pollution-sensitive taxa. TPH concentrations greater than 1,870 mg/kg were considered likely to lead to more substantial losses in species presence and abundance. Pettigrove and Hoffmann (2005) also hypothesized that TPH concentrations between 860 and 1,870 mg/kg would severely affect predatory organisms that directly or indirectly rely on benthic invertebrates as a source of food. A TPH concentration of 840 mg/kg was proposed as an interim guideline value to indicate possible ecological impairment.

Anson *et al.* (2008) furthered the work of Pettigrove and Hoffman (2005) by testing a broader range of hydrocarbon products under similar conditions. They concluded that the proposed guideline of 840 mg/kg remained valid. Some naturally occurring organic wetland sediments that they sampled contained substances that resembled TPH, although they did not appear to cause adverse effects on the benthic invertebrate community. Anson *et al.* (2008) recommended that sediments found to exceed the benchmark value of 840 mg/kg should be further tested to separate potentially benign biogenic sources of hydrocarbons from potentially detrimental anthropogenic TPH.

3.4.6 Summary

Total petroleum hydrocarbon concentrations less than about 10 mg/kg in sediment will typically report as “not detectable” in routine laboratory analysis. Urban sediments can be expected to have a detectable background level of contamination, due to storm water runoff washing small individual deposits of oil and grease into storm drainage systems, and thence to receiving water bodies, in addition to marine-sourced pollution. Such background concentrations may range from <100 to >1,000 mg/kg.

Benthic invertebrate communities are subject to high levels of natural variation in terms of spatial patchiness, that is reflected in highly variable values for abundance and community composition. Integrative indices of community composition, such as the number of taxa present and community diversity, tend to be more stable, but can be somewhat insensitive as effect indicators. Notwithstanding these limitations, effects of hydrocarbon contamination on benthic invertebrate communities can be expected to affect sensitive species at concentrations starting around 200 mg/kg, and to have effects on community productivity at concentrations exceeding 500 mg/kg. Petroleum hydrocarbon concentrations greater than 2,000 mg/kg in sediment should be considered indicative of serious contamination, such that both acute effects, and serious effects on community productivity might be expected to occur.

In the present analysis, petroleum hydrocarbon deposition to sediment was estimated as a mass per unit area (*i.e.,* g/m^2). Assuming that oil deposited to sediment is initially mixed into the surface 1-cm layer of sediment, and that sediment in this layer has a bulk density of between 1.0 and 1.2, then the surface 1-cm layer of one square metre would have a volume of 0.01 m^2, and a wet mass of 10 to 12 kg. The following relationships would then apply:

- Oil deposition to sediment of <0.1 g/m^2 would result in a surface sediment TPH concentration of <10 mg/kg, and would functionally be non-detectable.
- Oil deposition to sediment of 1 g/m^2 would result in a surface sediment TPH concentration of <100 mg/kg, and would be unlikely to result in detectable biological effects.
- Oil deposition to sediment of 5 g/m^2 would result in a surface sediment TPH concentration of <500 mg/kg, which could result in adverse effects to a limited number of sensitive species, but could increase overall community productivity due to the enrichment effect.
- Oil deposition to sediment of 20 g/m^2 would result in a surface sediment TPH concentration of <2,000 mg/kg, which would be expected to cause reduced community diversity, biomass and productivity.
- Oil loadings to sediment of greater than 20 g/m^2 would be expected to cause progressively more serious reductions in community diversity, biomass and productivity.

3.5 Chronic Exposure Assessment for Mammals and Birds

Birds and mammals living in the marine environment may be chronically exposed to the effects of low levels of crude oil or PAHs following an oil spill. Such exposures, which are assumed to be dominated by oral exposure due to consumption of prey or sediment containing traces of hydrocarbons, are evaluated here at two points in time. The first corresponds to the end of the acute phase of the spill, when the oil is relatively fresh and concentrations are relatively high. The second point represents a time one to two years after the spill, when the oil is more weathered, and concentrations have diminished.

The selection of an assessment time point that is one to two years after the spill reflects the fact that a hypothetical oil spill could occur at any time. Crude oil may be expected to weather more quickly during warm weather than during cold weather periods. However, a spill that occurred in the spring would undergo one winter and two summers within 16 months of the event, whereas a spill that occurred in the fall would undergo two winters and only one summer during an equivalent period of time. The selection of an assessment time of one to two years after a spill is based upon one year of model weathering, but is intended to reflect the reality that oil will weather more rapidly during some periods of the year.

For the hypothetical spill at the WMT, therefore, the potential for chronic effects to mammals or birds caused by chronic oral ingestion of crude oil or PAHs is evaluated starting about 5 days after the spill initiation, and again after one to two years.

For the hypothetical spills at Arachne Reef, the potential for chronic effects to mammals or birds caused by chronic oral ingestion of crude oil or PAHs is evaluated starting about two weeks after the spill initiation, and again after one to two years.

3.5.1 Selection of Exposure Point Concentration (EPC) Values for Chronic Effects Assessment

For the deterministic large spill scenario at the WMT, the greatest exposure to spilled crude oil occurs within an area bounded to the west by the Second Narrows Bridge, and to the east by the entrances to Indian Arm and Port Moody Arm, respectively. The potential for chronic exposure of mammal and bird

receptors is therefore evaluated within this zone, where exposure to spilled crude oil and PAH residues will be greatest.

The following processes were followed in order to determine appropriately conservative EPC values on which to base the exposure assessment for mammals and birds:

- For organisms that are primarily exposed to COPCs in the water column, including pelagic fish and squid, and mussels attached to rocks or other substrates emergent from the sediment, the maximum 24-hour average narcosis TU and TPAH concentrations were identified. These concentrations were used to estimate bioaccumulation of hydrocarbons by pelagic organisms as described in Section 3.5.2 below.

- For organisms that are primarily exposed to COPCs associated with intertidal sediment, including bivalves, gastropods, crustaceans, and intertidal zone fish, the impingement of crude oil onto porous beach substrates (including mud, sand, gravel and cobble beach units) was estimated, and the 90th percentile value was selected (taking into consideration the length of the units, the maximum oil holding capacity of the units, and degree to which the units were estimated to be oiled). It was assumed that oil spill response activities would result in cleaning of beaches, so if the 90th percentile value exceeded a value of 1 L of crude oil per square metre of intertidal sediment, this value was capped at a maximum value of 1 L/m^2. A model (see Appendix A) was then used to simulate the weathering of crude oil in intertidal beach sediments, and the residual concentrations of crude oil pseudo-components and PAHs in sediment interstitial water, and on the sediment organic carbon fraction, in addition to whole sediment, were calculated.

- For organisms that are primarily exposed to COPCs associated with subtidal sediment, including bivalves, crustaceans, and demersal fish, the deposition of crude oil onto subtidal sediment (assumed to be sandy or muddy sediment) was estimated, and the 90th percentile value (g/m^2) was selected. A model (see Appendix A) was then used to simulate the weathering and burial of crude oil in subtidal sediments, and the residual concentrations of crude oil pseudo-components and PAHs in sediment interstitial water, and on the sediment organic carbon fraction, in addition to whole sediment, were calculated.

- Concentrations of bioavailable pseudo-components of the crude oil, and PAHs, in the tissues of exposed fish, mollusks and crustaceans were estimated following methods that are described in Section 3.5.2, below.

A similar process was followed to evaluate the chronic effects of the large oil spill scenario at Arachne Reef. However, owing to the less confined nature of the marine environment at Arachne Reef, with the potential for crude oil to disperse into a wider variety of marine habitats, three separate assessment locations were selected, as follows.

- A primary assessment location was selected in the vicinity of the Gulf/San Juan Islands, given the high potential for shoreline and intertidal zone oiling, and proximity to the hypothetical spill site. An area having a radius of 5 km was selected between Arachne Reef and San Juan Island, and the expected crude oil and PAH concentrations in water, intertidal sediment and subtidal sediment were estimated as described above for the WMT.

- A second assessment location was selected for the large oil spill at Arachne Reef by identifying the location of the highest 24-hour exposure to crude oil pseudo-components in water. For the large hypothetical spill scenario this occurred near the Fraser River Delta. A semi-circle having a radius of 5 km was established around this point, and the expected crude oil and PAH concentrations in water, intertidal sediment and subtidal sediment were estimated as described above for the WMT.

- A third assessment location was selected for the large oil spill at Arachne Reef by identifying the area of the greatest potential deposition of crude oil constituents to subtidal sediment. For the large hypothetical spill scenario this occurred in the Juan de Fuca Strait south of San Juan Island. A semi-circle having a radius of 5 km was established around this point, and the expected crude oil and PAH concentrations in water, intertidal sediment and subtidal sediment were estimated as described above for the WMT.

3.5.2 Bioaccumulation of Hydrocarbons by Fish and Invertebrates

The bioaccumulation of hydrocarbons including PAHs by marine invertebrates and fish has been the subject of considerable empirical and theoretical investigation (Parkerton *et al.* 1993, Thomann and Komlos 1999, Di Toro *et al.* 2000, Di Toro and McGrath 2000, Arnot and Gobas 2004, Parkerton and Connolly 2013). It was recognized early that sediment, and in particular sediment organic matter, plays a key role as a reservoir and source of hydrophobic organic contaminants to freshwater and marine biota (Allen 1986). It was also recognized that the pelagic and demersal or benthic food webs, although interconnected, reflect different degrees of exposure to the sediment reservoir, which results in differential exposure to and accumulation of PAHs in the tissues of marine fish and invertebrates (Llobet *et al.* 2006).

The following sections provide a simplified approach to predict wildlife exposure to hydrocarbon fractions and PAH compounds in the tissues of mollusks, crustaceans and fish living in the marine environment during and after a hypothetical oil spill. The approach considers pelagic and demersal/benthic food webs separately, with exposure of pelagic biota occurring briefly, followed by a depuration phase, and exposure of demersal/benthic biota occurring chronically, with tissue concentrations treated as being in equilibrium with the sediment reservoir of hydrocarbons. In the chronic ERA for wildlife receptors, the individual receptors are assumed to consume a variety of food types selected from either or both of the pelagic or the demersal/benthic food webs.

3.5.2.1 Bioaccumulation of Hydrocarbons from Water by Pelagic Organisms

Bioaccumulation and retention of dissolved hydrocarbons was evaluated in selected pelagic fish and invertebrate species including salmon, herring, cod, squid, krill and mussels. Although sessile and not pelagic, mussels were assumed to be attached to rocky or other hard substrates above the bottom, and exposed primarily to dissolved hydrocarbons in the water column. Accumulation of hydrocarbons from sediment by sediment-associated mollusks such as clams or whelks is addressed separately below. The assumed characteristics of the "pelagic" species are presented in Table 3.11.

Table 3.11 Assumed Characteristics of Aquatic Species Exposed to Dissolved Hydrocarbons in Water

Species	Body Weight (kg wet)	Biota Lipid Content (unitless)	Biota Water Content (unitless)	Biota Non-Lipid Content [1] (unitless)
Herring	0.1 [2]	0.110 [2]	0.700 [3]	0.190
Salmon	1.4 [2]	0.068 [2]	0.734 [3]	0.198
Squid	0.038 [2]	0.025 [2]	0.785 [3]	0.190
Krill	0.0001 [4]	0.020 [5]	0.830 [6]	0.150
Mussels	0.01 [7,8]	0.017 [3]	0.828 [3]	0.155

Notes:
1 Biota Non-Lipid Content = 1 – Biota Lipid Content – Biota Water Content (Arnot and Gobas 2004).
2 Iverson *et al.* (2002)
3 NOAA (1987)
4 Nicol and Endo (1999)
5 Taatjes and Cass (2014)
6 Mauchline (1980) in Wilding (2007)
7 Hurlburt and Hurlburt (1974)
8 Zagata *et al.* (2008)

Accumulation of hydrocarbons by fish and invertebrates present or suspended within the water column (*i.e.,* the pelagic food web) is based upon a model developed by Arnot and Gobas (2004) for the bioaccumulation of hydrophobic organic chemicals from the dissolved phase in aquatic ecosystems. In this analysis, petroleum hydrocarbons (specifically the pseudo-components evaluated in this DQERA, subject to some limitations; and the suite of parent and alkylated PAH compounds) are assumed to be typical hydrophobic organic compounds.

The key limitation for the pseudo-components is that the higher molecular weight hydrocarbons, represented by pseudo-components AL3, AL4, AL5, AL6, RES1, RES2, and RES3 are assumed to have negligible bioavailability due primarily to their very low solubility in water, and these were excluded from further analysis. This is consistent with observations by Niimi and Oliver (1988) that absorption of organic chemicals ingested by fish is limited by high molecular weight and/or large molecular volume, and by Di Toro *et al.* (2000) that bioaccumulation and toxicity of hydrocarbons in water is limited by low solubility for compounds having log K_{ow} values greater than about 6.

The model of Arnot and Gobas (2004) considers site-specific attributes, chemical properties and characteristics of aquatic species to derive bioconcentration factors as well as uptake and elimination rate constants. Several uptake and elimination routes are included within the model. For the purpose of this DQERA, uptake of dissolved hydrocarbons from water through the gill, as well as metabolism/excretion and growth dilution were considered.

It is not necessary to distinguish between metabolism and excretion of hydrocarbons for the purposes of the model, as they both represent “loss” processes which can be treated separately or combined with equal facility. However, some workers (Di Toro *et al.* 2000) have speculated that metabolism of PAH compounds may simply transform compounds into metabolites that could be less, equally, or more toxic than the parent compound. In addition, whereas parent PAH compounds appear to be quite readily metabolized, alkylated PAH compounds appear to be less readily metabolized with rates of metabolic breakdown declining as the degree of alkylation and log K_{ow} values increase. Therefore, although PAHs are known to be subject to metabolic breakdown, it is conservative in this approach to predicting hydrocarbon accumulation by pelagic organisms to assume that this process is negligible, and to consider only excretion of hydrocarbons.

Accumulation of hydrocarbons by aquatic species was conservatively assumed to reach steady-state within the 24-hour period of greatest exposure. Steady-state bioconcentration factors for each species were evaluated for each bioavailable pseudo-component or PAH compound following an approach developed by Arnot and Gobas (2004) using the following equation:

$$K_{BW} = f_{lipid} \cdot K_{OW} + f_{non\text{-}lipid\ organic\ matter} \cdot \beta \cdot K_{OW} + f_{water\ content}$$

where K_{BW} is a biota-water partition coefficient (L/kg wet weight tissue), f_{lipid} is the lipid fraction of the organism (unitless), K_{OW} is the octanol-water partition coefficient (L/kg), $f_{non\text{-}lipid\ organic\ matter}$ is the fraction of the organism that is non-lipid organic matter (NLOM; unitless), β (unitless) is the proportionality constant of the sorption capacity of NLOM to that of octanol, and $f_{water\ content}$ is the fraction of the organism that is water (L water/kg tissue, or unitless assuming that 1 L water is equal to 1 kg).

The whole-body concentration of each pseudo-component or PAH compound in each aquatic species ($C_{pelagic}$; mg/kg wet weight) for this first time-step can then be calculated as:

$$C_{pelagic} = K_{BW} \cdot C_{water}$$

where C_{water} is the COPC (either a hydrocarbon pseudo-component or a PAH compound) dissolved concentration in water (mg/L). As noted above this is conservatively estimated to be the

maximum 24-hour average concentration for TPH or TPAH anywhere within the oil spill fate and transport model domain.

With the exception of mussels, the rapid accumulation phase is followed by a period of quite rapid hydrocarbon depuration, so that tissue concentrations decrease over time. This depuration phase represents the effects of both excretion of hydrocarbons and growth dilution, but not metabolic conversion of hydrocarbons to simpler molecules, as explained above. Because mussels are known to have a limited capacity to metabolize or excrete PAHs (Yender *et al.* 2002), they were conservatively evaluated assuming accumulation to steady-state, followed by growth dilution only, (*i.e.,* without subsequent metabolism or depuration of hydrocarbons).

For other (non-mussel) aquatic species, depuration rate constants (k_{out}; h^{-1}) were evaluated for each hydrocarbon type and species as presented in Arnot and Gobas (2004):

$$k_{out} = (E_W \cdot G_V/W_B)/\ K_{BW}$$

where E_W = gill chemical exchange efficiency (unitless), G_V = ventilation rate (L/h), and W_B = wet weight of the organism (kg).

Changes in tissue COPC concentration due to growth dilution were evaluated using a growth rate constant for aquatic species (k_G, h^{-1}) in cool aquatic environments (around 10°C) from Thomann *et al.* (1992) as presented in Arnot and Gobas (2004):

$$k_G = 0.0005 \cdot W_B^{-0.2} \cdot 0.0417$$

Since bioaccumulation is only assumed to occur at the first time-step, subsequent changes to tissue hydrocarbon concentrations are related to either depuration or growth dilution. Change in tissue concentrations are then evaluated as:

$$dC_{pelagic}/dt = -\ (k_{out} + k_G) \cdot C_{pelagic}$$

For pelagic organisms, therefore, exposure to hydrocarbon COPCs is conceptualized as occurring during a brief period when dissolved hydrocarbons are present in the water column. It is conservatively assumed that the exposed biota come to equilibrium with the dissolved hydrocarbons at the location and during the period of the maximum 24-hour average concentration. Hydrocarbons are subsequently depurated, and the organisms grow, resulting in a gradual decline in the tissue concentrations following the initial exposure. This pattern is consistent with the process and patterns of observed hydrocarbon concentrations in the tissues of pelagic organisms following exposure to spilled hydrocarbons.

3.5.2.2 Bioaccumulation of Hydrocarbons from Sediment by Demersal and Benthic Organisms

In contrast to water, spilled hydrocarbons can accumulate in sediments and persist for long periods of time. Accumulation of hydrocarbons by benthic invertebrates present within intertidal and sub-tidal sediment, or demersal fish living in contact with and feeding primarily on benthic infauna, is based on the Biota to Sediment Bioaccumulation Factor (BSAF) concept. Aquatic species evaluated using this approach include flatfish and other fish such as sculpins living in contact with the subtidal sediment; forage fish such as juvenile salmon, sticklebacks and sculpins associated with the intertidal zone and marsh areas; crabs and clams exposed to subtidal or intertidal sediment; clams and mussels exposed to intertidal sediment, and gastropods (*e.g.,* winkles, whelks and limpets) exposed to intertidal sediment. Assumed characteristics of these species are presented in Table 3.12.

Table 3.12 Assumed Characteristics of Aquatic Species Associated with Sediment

Species	Sediment Type	Biota Lipid Content (unitless)	Biota Water Content (unitless)
Flatfish[1]	Subtidal	0.010[3]	0.850[4]
Forage fish[2]	Intertidal	0.050[3]	0.770[4]
Crabs	Subtidal and Intertidal	0.009[4]	0.781[4]
Clams	Subtidal and Intertidal	0.032[4]	0.788[4]
Gastropods	Intertidal	0.036[4]	0.792[5]

Notes:
1 Flatfish are intended to represent bottom dwelling fish such as flounders and sculpins.
2 Forage fish are intended to represent small fish such as juvenile salmon, sticklebacks, and sculpins living in marsh areas or in the intertidal zone.
3 Iverson *et al.* (2002)
4 NOAA (1987)
5 USDA (2011)

Marine sediment to benthic invertebrate uptake factors (UP_{MSBI}) for organic compounds were obtained from a BSAF equation derived by Di Toro and McGrath (2000):

$$BSAF = C_L/C_{S,OC}$$

where C_L is the concentration of a chemical in the lipid of a benthic organism (mmol/kg lipid) and $C_{S,OC}$ is the concentration of the same chemical in the organic carbon fraction of the sediment (mmol/kg organic carbon). Using additional equations presented by Di Toro and McGrath (2000), the equation can be simplified to:

$$BSAF = 10^{(-0.038 \log(Kow)-0.00028)}$$

BSAF can be normalized according to the organic carbon content of the marine sediment (f_{oc}) and lipid content of the invertebrates (f_{lipid}) by incorporating an adjustment factor (f_{lipid}/f_{oc}). Lipid fractions in benthic invertebrates are presented in Table 3.12.

Bioavailability and organism physiology have been identified as the two important variables having a major effect on bioaccumulation of hydrocarbons by aquatic fauna. Of the total environmental concentration, only the bioavailable fraction can enter the organism. Physiological factors, including lipid levels and the rates of uptake and elimination (metabolism, diffusion and excretion) also determine the body burden (Meador *et al.* 1995). Also applied to this evaluation, therefore, was a metabolic factor (MF), which depends upon the type of species being evaluated, as well as the chemical class (*e.g.,* PAHs, PCBs), and to some extent the hydrophobicity of chemicals within a chemical class.

The MF depends upon the type of organism because some species (notably mollusks) have very limited enzymatic capability to metabolize PAHs, whereas others (*e.g.,* fish and higher vertebrates) are highly capable in this regard. Crustaceans appear to have greater capacity to metabolize PAHs than mollusks, but less than fish. Chemical class effects are important because some types of chemicals (such as PCBs or DDT) are very stable and resistant to metabolic conversion, whereas others (such as the PAHs) are more amenable to metabolic breakdown pathways. Within the class of hydrocarbon compounds, individual chemicals also have variable resistance to metabolic breakdown. The monoaromatic substances are readily metabolized by vertebrates, although higher molecular weight PAHs, and higher-alkylated PAHs tend to be more resistant to metabolic breakdown.

The metabolic factor (Table 3.13) is assigned values that range between 0.05 (for chemicals that are expected to be readily metabolized or excreted) and 1.0 (for chemicals that are not expected to be metabolized or excreted). The choice of the metabolic factor depends on the COPC and and reflects professional judgement concerning the chemical type, log K_{ow} value, and organism type, based on

previously collected or published empirical data (*e.g.,* JDAC 2002). Concentrations of PAHs in fish tissues are generally lower than concentrations in invertebrates under similar exposure regimes (Van Geest *et al.* 2011), and much lower than equilibrium theory would suggest (van der Oost *et al.* 2003). The capacity of crustaceans to metabolize PAHs is generally found to be intermediate between that of mollusks and fish. High molecular weight PAHs, and more highly alkylated PAHs that tend to have higher Log K_{ow} values are more persistent than PAHs having lower Log K_{ow} values.

Table 3.13 Metabolic Factors (MF) Assumed for Breakdown and Excretion of PAHs by Benthic/Demersal Mollusks, Crustaceans and Fish

Log K_{ow} of COPC	Mollusks	Crustaceans	Fish
<3.0	0.5	0.1	0.05
3.0 to <5.0	0.75	0.25	0.1
≥5.0	1.0	1.0	1.0

The final component of the bioaccumulation model for benthic biota is a dry weight to wet weight conversion factor. The final UP_{MSBI} equation used in this ERA was:

$$UP_{MSBI} = MF \cdot (\text{dry weight/wet weight}) \cdot (f_{lipid}/f_{oc}) \cdot 10^{(-0.038\log(Kow)-0.00028)}$$

Equilibrium uptake factors for representative marine organisms exposed to the demersal/benthic food webs are summarized in Table 3.14. The concentration of a COPC in the wet tissue of a receptor organism is estimated by multiplying the sediment COPC concentration (mg/kg dry weight) by the appropriate UP_{MBSI} value.

Table 3.14 Equilibrium Uptake Factors (UPMSBI) of Aquatic Species Associated with Sediment, Based on an Assumed Sediment Organic Carbon Fraction (f_{oc}) of 0.01

Parameter	UP_{MSBI}				
	Flatfish	Forage Fish	Crabs	Clams	Gastropods
Pseudo-Components					
VOL	0.0058	0.045	0.015	0.26	0.29
AR1	0.0062	0.048	0.016	0.28	0.31
AR2	0.0058	0.044	0.015	0.26	0.29
AR3	0.011	0.084	0.036	0.37	0.41
AR4	0.011	0.083	0.035	0.37	0.40
AR5	0.010	0.080	0.034	0.36	0.39
AR6	0.010	0.077	0.0330	0.34	0.38
AR7	0.093	0.71	0.1218	0.42	0.46
AL1	0.011	0.081	0.035	0.36	0.40
AL2	0.0098	0.075	0.032	0.33	0.37
AL3	0.090	0.69	0.12	0.41	0.45
AL4	0.081	0.62	0.11	0.36	0.40
AL5	0.067	0.51	0.088	0.30	0.33
AL6	0.046	0.36	0.061	0.21	0.23
RES1	0.036	0.28	0.048	0.16	0.18
RES2	0.065	0.50	0.085	0.29	0.32
RES3	0.033	0.25	0.043	0.15	0.17
Polycyclic Aromatic Hydrocarbons					
Acenaphthene	0.011	0.082	0.035	0.36	0.40
Acridine	0.011	0.085	0.037	0.38	0.42
Anthracene	0.010	0.077	0.033	0.34	0.38
Benzo(a)anthracene	0.089	0.69	0.12	0.40	0.45
C_1-benzo(a)anthracene/chrysene	0.088	0.68	0.12	0.40	0.44
C_2-benzo(a)anthracene/chrysene	0.084	0.65	0.11	0.38	0.42
C_3-benzo(a)anthracene/chrysene	0.081	0.62	0.11	0.36	0.40

Table 3.14 Equilibrium Uptake Factors (UPMSBI) of Aquatic Species Associated with Sediment, Based on an Assumed Sediment Organic Carbon Fraction (f_{oc}) of 0.01

Parameter	UP_{MSBI}				
	Flatfish	Forage Fish	Crabs	Clams	Gastropods
C_4-benzo(a)anthracene/chrysene	0.078	0.60	0.10	0.35	0.39
Benzo(b&j)fluoranthene	0.090	0.69	0.12	0.41	0.45
C_1-benzo(b,j,k)fluoranthene/benzo(a)pyrene	0.084	0.64	0.11	0.38	0.42
C_2-benzo(b,j,k)fluoranthene/benzo(a)pyrene	0.080	0.61	0.11	0.36	0.40
Benzo(g,h,i)perylene	0.085	0.65	0.11	0.38	0.42
Benzo(a)pyrene	0.088	0.68	0.12	0.40	0.44
Benzo(e)pyrene	0.085	0.65	0.11	0.39	0.43
Biphenyl	0.011	0.082	0.035	0.36	0.40
C_1-biphenyl	0.010	0.077	0.033	0.34	0.37
C_2-biphenyl	0.096	0.74	0.13	0.43	0.48
Chrysene	0.092	0.70	0.12	0.42	0.46
Dibenz(a,h)anthracene	0.083	0.64	0.11	0.38	0.41
Dibenzothiophene	0.010	0.078	0.034	0.35	0.38
C_1-dibenzothiophene	0.0098	0.075	0.032	0.33	0.37
C_2-dibenzothiophene	0.094	0.72	0.12	0.42	0.47
C_3-dibenzothiophene	0.089	0.68	0.12	0.40	0.45
C_4-dibenzothiophene	0.086	0.66	0.11	0.39	0.43
Fluoranthene	0.095	0.73	0.12	0.43	0.47
C_1-fluoranthene/pyrene	0.093	0.71	0.12	0.42	0.46
C_2-fluoranthene/pyrene	0.088	0.68	0.12	0.40	0.44
C_3-fluoranthene/pyrene	0.084	0.65	0.11	0.38	0.42
C_4-fluoranthene/pyrene	0.080	0.62	0.11	0.36	0.40
Fluorene	0.010	0.080	0.034	0.35	0.39
C_1-Fluorene	0.0097	0.074	0.032	0.33	0.36
C_2-Fluorene	0.096	0.73	0.13	0.43	0.48
C_3-Fluorene	0.095	0.73	0.12	0.43	0.47
Indeno(1,2,3-cd)pyrene	0.083	0.64	0.11	0.38	0.42
Naphthalene	0.011	0.086	0.037	0.38	0.42
C_1-Naphthalene	0.011	0.082	0.035	0.36	0.40
C_2-Naphthalene	0.010	0.078	0.034	0.35	0.38
C_3-Naphthalene	0.0098	0.075	0.032	0.33	0.37
C_4-Naphthalene	0.094	0.72	0.12	0.42	0.47
Phenanthrene	0.010	0.077	0.033	0.34	0.38
C_1-phenanthrene/anthracene	0.096	0.74	0.13	0.43	0.48
C_2-phenanthrene/anthracene	0.094	0.72	0.12	0.42	0.47
C_3-phenanthrene/anthracene	0.089	0.68	0.12	0.40	0.44
C_4-phenanthrene/anthracene	0.085	0.65	0.11	0.38	0.42
Perylene	0.087	0.67	0.11	0.39	0.43

Notes:
These values of UP_{MSBI} are calculated assuming a sediment organic carbon content of 1% (f_{oc} = 0.01). In sediments with higher values of f_{oc}, the UP_{MSBI} values would be proportionally reduced. Differences between flatfish and forage fish reflect different assumptions about relative lipid and moisture contents.

3.5.3 Hazard Assessment

The hazard assessment (or toxicity assessment) stage selects toxicity reference values (TRVs) which are essentially dose or exposure limits for toxic effects to receptors, for each COPC. The toxicity of a COPC depends on the amount taken into the body (the dose) and the duration of exposure (*i.e.,* the length of time the receptor is exposed). For each COPC, there is a specific dose and duration of exposure necessary to produce a toxic environmental effect in a given receptor (this is referred to as the dose-response relationship of a COPC). The toxicity of a COPC is dependent on:

- inherent properties that cause a biochemical or physiological response at the site of action

- ability of the COPC to reach the site of action
- unique sensitivities associated with the species being tested, its life-stage or interactions with other environmental or physiological conditions.

The dose-response principle is central to the DQERA method for assessing risks to birds and mammals.

3.5.3.1 Toxicity Benchmarks for Exposure of Wildlife Receptors to Petroleum Hydrocarbons

Hazard index and HQ values calculated for ecological receptors depend directly upon the selection of the oral-based toxicity reference value (TRV). The toxicological database in support of a TRV preferably includes a number of chronic or multi-generational exposure studies involving exposure of relevant test species (*i.e.,* the ecological receptor of interest or a phylogenetically similar species) to appropriate chemical forms of the substance of interest. Ideally, one or more relevant biological endpoints such as growth, reproductive effects, or survival are measured in the study. Databases that meet this requirement are available for some chemicals, but in many cases available toxicity data are limited to studies conducted with a limited variety of laboratory animals.

Toxicity Reference Values for this ERA are based on dose response studies, typically conducted with laboratory animals where the lowest observed adverse effects level (LOAEL) or no observed adverse effects level (NOAEL) has been quantified. The TRVs used in this risk assessment were determined from studies in which endpoints were derived from the administered dose, rather than the absorbed dose. This is a conservative approach because compounds are often administered in a more bioavailable form than would be found in the environment.

The preferred toxicity measure used for derivation of TRVs in this ERA is the LOAEL since this value corresponds to the onset of toxicologically induced responses in test animals. However, in the absence of a suitable LOAEL, more conservative NOAEL-based TRV can be used. Generally, LOAEL used for TRV derivation are based on long-term growth or survival, or sub-lethal reproductive effects determined from chronic exposure studies. As such, these endpoints are relevant to the maintenance of wildlife populations. The LOAEL represents a threshold dose at which adverse outcomes are likely to become evident. This threshold is considered an appropriate endpoint for ERA since TRVs are used as the denominator in the HQ calculation, and HQ values equal to or greater than one may be considered indicative of potential adverse environmental effects. Hazard quotients calculated with NOAEL-based TRVs are more conservative since NOAEL refer to a threshold at which no relevant toxicological effects from COPC exposure are observed.

Numerous sources were reviewed to obtain the most relevant TRVs for ecological receptors. Information sources reviewed include, but are not limited to:

- Oak Ridge National Laboratory Toxicity Benchmarks for Wildlife (Sample *et al.* 1996)
- US EPA Ecological Soil Screening (EcoSSL) documents
- Agency for Toxic Substances and Disease Registry (ATSDR)
- *Canadian Environmental Protection Act* (*CEPA*), Priority Substance List Assessment Reports
- primary scientific literature.

Uptake and depuration of various petroleum hydrocarbons by molluscs and fish have been shown in numerous studies. Due to the lack of a rapid detoxification system, molluscs are unable to readily metabolize aromatic hydrocarbons, and moderate accumulation can occur. Bioaccumulation of petroleum

hydrocarbons in higher organisms, such as fish, is found to be low due to their metabolic elimination and detoxification mechanisms. PAHs are not accumulated by fish through dietary exposure because of the combined effects of poor absorption efficiencies and rapid elimination times. The Log K_{ow} of the C_{20} cycloalkane fraction and $>C_{20}$ fractions is greater than 9.0. At this Log K_{ow} there are no empirically observed bioconcentration or bioaccumulation factor values recorded for any species of invertebrates or vertebrates, likely as a result of very low bioavailability and thus poor dietary assimilation efficiency. There is no evidence that petroleum hydrocarbons biomagnify up food chains (Environment Canada and Health Canada 2011).

In this DQERA study, the potentially harmful effects of chronic exposure to petroleum hydrocarbons are evaluated in several ways. For some compounds, which are recognized to be important components of crude oil (*e.g.,* the BTEX compounds, certain PAH compounds, and the Canada-Wide Standard petroleum hydrocarbon fractions), sufficient toxicological data exists to determine whether toxicity is likely to occur due to exposure to individual compounds. Some compounds (notably the F4 fraction under the Canada-Wide Standards fractionation scheme, and asphaltenes) are assumed to have negligible bioavailability, and hence are often not considered to be active contributors to the potential toxicity of hydrocarbon mixtures. However, there are gaps in the toxicological database, and data are lacking for many of the identified petroleum hydrocarbon compounds. For this reason, the following approach has been taken to evaluate the chronic toxicity of petroleum hydrocarbons derived from crude oil.

In Canada (CCME 2008), emphasis has been placed on exposure pathways based on direct contact between plant roots or soil invertebrates and the contaminated soils. This emphasis is based on the need to preserve the principal ecological functions performed by the soil resources. Less emphasis has been placed on the estimation of contamination in soils beyond which wildlife or domesticated animals might be at risk. The relative lack of emphasis on terrestrial vertebrate animals such as mammalian or avian vertebrates is probably acceptable for petroleum hydrocarbon release sites as most petroleum hydrocarbons are readily metabolized by vertebrates, modified into a more readily excretable form, and thus do not tend to accumulate in tissues. In addition, petroleum hydrocarbons are not readily absorbed into and accumulated by plant tissues. The net result is that consumption of either plants or other animals (as opposed to soil ingestion) does not tend to constitute the major component of exposure for petroleum hydrocarbons in wildlife and livestock.

The Canada-Wide Standard for Petroleum Hydrocarbons in soil (CCME 2008) provides benchmark values for the protection of soil invertebrates and plants exposed to hydrocarbons. For the purpose of the standard, it is assumed that free-phase petroleum hydrocarbon (which can cause harm to wildlife receptors as a result of direct ingestion from preening of feathers or fur) is not present. To develop benchmark values protective of plants and soil invertebrates, available toxicity data were standardized at a 25^{th} percent effects level (*e.g.,* determine what concentration of petroleum hydrocarbons results in a 25% reduction in invertebrate survival or plant growth). The 25^{th} percentile of the species sensitivity distribution for test results was then used as the guideline for agricultural or residential land use. The Tier 1 guidelines are intended to be generally applicable and to apply to both weathered and fresh hydrocarbons; in general the guidelines were developed using fresh hydrocarbons including but not limited to gasoline and crude oil, and it is acknowledged that the toxicity of hydrocarbon mixtures may decrease as weathering progresses. The Canada-Wide Standard guidelines for hydrocarbons in soil (taking the lower of the values for fine or coarse-grained agricultural soil for the agricultural/residential land use) are as follows: F1, 210 mg/kg dry soil; F2, 150 mg/kg dry soil; F3, 300 mg/kg dry soil; F4, 2,800 mg/kg dry soil.

Mammals at risk from oil-related injuries include those that frequent water including river otter and mink. These animals spend much time in water, have high site fidelity, and rely on fur to maintain thermoregulation. Other terrestrial mammals of concern include those associated with water bodies and

riparian habitat such as bear and moose. These animals are likely to be affected by the consumption of oiled food times, as well as by direct contact and habitat degradation (Hugenin *et al.* 1996).

Cattle will voluntarily ingest large doses of petroleum substances (Coppock *et al.* 1996). In such acute poisoning cases the lung is the target organ. Chemical pneumonia results when droplets of oil are inhaled. Another cause of chemical pneumonia is aspiration of hydrocarbons during vomiting, regurgitation or eructation. Lung lesions have been reported following the voluntary ingestion of petroleum by cattle. Such lesions have been reported in cattle given 20 to 60 mL crude oil/kg body weight, and in sheep after a 1-day exposure to water contaminated with natural gas condensate, which also caused reddening and hemorrhage in the digestive tract. Kidneys can also be target organs of petroleum hydrocarbon toxicosis (Coppock *et al.* 1996).

CCME (2008) derive a value for the toxicity of petroleum hydrocarbons (with a focus on fresh crude oil) to livestock. A lowest documented effects dose of 2.1 g fresh crude/kg body weight/day is reported. This is divided by an uncertainty factor of 10 to obtain a Daily Threshold Effects Dose of 210 mg/kg body weight/day, which is then used to estimate the allowable concentration of whole fresh crude oil ingested in livestock drinking water. However, it is noted that the value for weathered crude oil could be 3.7 times higher, due to the lower toxicity of weathered crude oil (CCME 2008). A value of approximately 0.78 g/kg body weight/day could then be appropriate for chronic ingestion of weathered crude oil.

Other studies evaluating the toxicity of fresh or weathered crude oil to mammals are presented in Table 3.15. These studies include cattle, rats and ferrets, and therefore represent herbivores, omnivores and carnivores. The results from these studies are highly consistent with respect to the dose ranges that have been shown to have adverse effects on mammals. Based on the available information it is concluded that mammals are generally quite tolerant of exposure to un-weathered or weathered crude oil. Adverse effects are considered unlikely at dose levels less than 0.5 g/kg body weight/day.

There is a general lack of literature describing the oral toxicity of crude oil or PAH exposure to marine mammals including seals, sea lions, or whales. However, it is reasonable to assume that the sensitivity of those species groups is generally similar to the sensitivity of other mammals. The TRVs for marine mammals exposed to crude oil are therefore also expected to be in the range of 0.5 kg body weight/day.

Table 3.15 Toxicological Benchmarks for Mammalian Receptors Exposed to Crude Oil

Study Design	Effects	Benchmark Derivation	Reference
Domestic cattle (8/group) were administered single oral doses of Pembina Cardium crude oil at 16.7, 33.4 and 67.4 g/kg body weight. Cattle were sacrificed at days 7 or 30 for pathological and histological examination, as well as measurement of a suite of enzyme activities.	No cattle died as a result of being dosed with crude oil. Cattle treated with 16.7 mg/kg body weight showed minimal signs of intoxication; cattle treated with higher doses exhibited tremors, nystagmus, regurgitation and vomiting, myoclonic seizures, depression, locomotor abnormalities and pulmonary distress. On day 7, cattle in exposed groups showed alteration in enzyme activities in liver, kidney and lung tissues. Cattle sacrificed on day 30 showed few statistically significant differences from control animals, and reduced differences in cytochrome P-450 activity.	Acute toxicity greater than 67.4 g/kg body weight.	Khan *et al.* (1996)

Table 3.15 Toxicological Benchmarks for Mammalian Receptors Exposed to Crude Oil

Study Design	Effects	Benchmark Derivation	Reference
Sprague-Dawley rats were given doses of 0.25, 0.50 or 1.25 mL/kg Pembina Cardium crude oil, or 1.25 mL/kg commercial diesel fuel, on days 1, 3, 5 and 8 of the study, and were sacrificed on day 10. Tissue and blood samples were tested for a suite of enzyme activities, hematology and blood chemistry, and pathological examination.	No rats died as a result of being dosed with crude oil or diesel fuel, and there was no sign of distress of intoxication in the exposed animals. Dose-dependent changes were observed in levels of a suite of enzyme indicators (including EROD). The only significant systemic change was a small increase in the liver somatic index of rats exposed to the highest dose of crude oil or diesel fuel.	No sign of distress or intoxication in rats exposed to crude oil at up to 1.25 mL/kg body weight.	Khan *et al.* (2001)
The toxicity of naturally weathered Exxon Valdez crude oil was tested in a battery of acute and subchronic tests using European ferrets (Mustela putorius). Young adult male and female ferrets were administered oil at a dose of 0.5, 1.0 or 5.0 g/kg body weight once daily for three days. Prior to the study and at termination, blood samples were taken for chemistry and enzyme testing, and the animals were weighed. At study termination (day 5) the animals were subject to gross necropsy examination and selected tissues were taken for histological examination.	No mortality of ferrets occurred as a consequence of being administered crude oil. No grossly observable signs of toxicity were noted. No effects on mean body weight were detected. No grossly observable signs of toxicity were noted during postmortem examination of the ferrets. Microscopic examination of tissues did not reveal lesions considered to be related to oil exposure. Significantly lower mean spleen to body weight ratios and raw spleen weights were noted in all female treatment groups. No other organ weight differences were noted. With the exception of lower mean serum albumin concentrations in the 5 g/kg female dose group, no significant differences among clinical chemistry parameters were noted. No significant differences in the hematological parameters were noted in any group.	Subacute toxicity of crude oil to European ferrets is >5 g/kg body weight/day. Acute toxicity of three unweathered crude oil samples to mice ranged from >10 to 16 g/kg body weight.	Stubblefield *et al.* (1995a) Smith *et al.* (1980)

Alcids (the family of web-footed diving birds with short legs and wings that include the auks, murres, and puffins) are considered to be the most vulnerable of bird groups to oil, due to their tendency to form large flocks and to spend much time floating on offshore waters. Large scale mortality to eggs is also likely due to their tendency to form large breeding colonies making them vulnerable in the event of an accident. Among waterfowl, populations of dabbling ducks are generally less exposed because they tend not to form large colonies. Therefore, both adult mortality and effects on eggs are less likely (at the population level) due to dispersion. Direct mortality rates for shorebirds are generally low because they spend less time in the water. Raptors become oiled primarily via consumption of oiled prey or carrion, and oil exposure can cause reproductive effects (*e.g.,* oiling of eggs, as well as nest disturbance caused by shoreline cleanup operations). Wading birds generally experience low mortality because they wade in shallow, sheltered waters to feed. However, plumage can become contaminated due to wading through oiled vegetation or exposure to oil slicks, and indirect effects can occur due to loss of prey resulting in starvation, or shifting to alternative foraging sites (Hugenin *et al.* 1996).

Chronic low levels of oil pollution may have adverse effects on aquatic bird populations. Small amounts of oil applied to the external surface of bird eggs are toxic. Single oral doses of oil have been demonstrated to cause lipid pneumonia, gastrointestinal irritation, and fatty livers. Pathological responses of birds examined after fatal exposure to Bunker C oil included enteritis, hepatic fatty changes, and renal tubular nephrosis (Szaro *et al.* 1978).

Other studies evaluating the toxicity of fresh or weathered crude oil to birds are presented in Table 3.16. These studies focus on mallard ducks, waterfowl which would be highly exposed to hydrocarbons in the event of a spill. Based on the available information it is concluded that birds are generally quite tolerant of exposure to un-weathered or weathered crude oil. The lowest reported adverse effects on a reproductive endpoint are identified at a dose of approximately 0.2 g/kg body weight/day.

Table 3.16 Toxicological Benchmarks for Avian Receptors Exposed to Crude Oil

Study Design	Effects	Benchmark Derivation	Reference
The toxicity of naturally weathered Exxon Valdez crude oil was tested in a battery of acute and subchronic tests using mallard ducks. Adult ducks were tested with an acute oral dose of 5 g/kg body weight and observed for up to 14 days following testing (acute oral toxicity); Five-day old ducklings were tested by feeding them a diet containing weathered crude oil at a concentration of 50 g/kg diet (subacute dietary toxicity) for five days, followed by a 3-day observation period on uncontaminated ration; food avoidance was tested using ducklings offered diet containing 0, 1.25, 2,5, 5, 10 and 20 g/kg diet for five days, followed by a 3-day observation period on uncontaminated ration; and ducks (16 weeks old) were fed a diet containing 0, 10, 30 or 100 g/kg weathered crude oil for 14 days.	No adult ducks died following single doses of 5 g/kg body weight; no grossly observable signs of toxicity were noted, and there were no significant effects on feed consumption or body weight. No treatment related abnormalities were noted on postmortem examination. No mortality or observable sign of toxicity was noted in ducklings fed crude oil in their diet at 50 g/kg body weight. Food consumption was not affected, and not significant differences in body weight or growth were noted. Post-mortem examination showed no evidence of systemic toxicity. Ducklings did not avoid food containing crude oil at up to 20 g/kg. No significant differences in body weight or growth were found, and no consistent grossly observable lesions were noted in postmortem examination. No mortalities or grossly observable signs of toxicity were noted in 14-day exposure to dietary concentrations of up to 100 g/kg diet. No significant treatment related differences in clinical blood chemistry was noted between treatment and control birds. No consistent or substantive differences were noted among the histological appearance of the kidney, thymus, brain or bone marrow of high dose birds when compared to control birds. Spleens and livers of high dose birds were found to show some minor changes when compared to control birds.	Weathered Exxon Valdez crude oil presented little potential for acute toxicity to wildlife species from oral ingestion. Lethal concentrations and no-observed adverse effect levels were greater than the maximum tested doses (>5 g/kg body weight in the oral study, and >50 g/kg diet in the subacute dietary tests). LD50 values for refined hydrocarbon products were reported to range from 7 to 20 mL/kg body weight.	Stubblefield *et al.* (1995a) Hartung and Hunt (1966) as cited by Stubblefield *et al.* (1995a).
A one-generation reproductive toxicity study and a direct eggshell application toxicity study were conducted using naturally weathered crude oil obtained following the EVOS. Mallard ducks, 16 weeks of age, were exposed to dietary concentrations of 0, 0.2, 2 and 20 g/kg diet for 22 weeks. Eggs laid between weeks 12 and 22 of exposure were incubated and hatched. Mallard eggs were also treated with either weathered crude oil or Vaseline (a non-toxic control) to determine the extent of coverage causing reduced viability.	No deaths of ducks occurred that were attributed to crude oil exposure. All surviving birds appeared healthy throughout the study, and no signs of toxicity were noted. No statistically significant differences in growth of birds or food consumption were noted. Consumption of diets containing crude oil at 20 g/kg feed resulted in changes in clinical chemistry parameters (i.e., serum phosphorus, total protein, albumin, bilirubin and calcium), reductions in eggshell thickness and strength (although the viability of embryos was not affected), and suggested liver and spleen weight changes. No significant effects were noted at dietary concentrations of 0.2 or 2 g/kg feed. Long-term ingestion of weathered crude oil at dietary concentrations of 20 g/kg feed may result in reduced egg fitness. Application of weathered crude oil to areas of up to 33% of the shell area had no appreciable effect on embryo survival, suggesting that not only is it substantially less toxic than unweathered crude oil, but that is not as effective as a shell sealant as Vaseline, which caused effects when 17% or greater of the egg shell was covered. However, the severity of effects based on dietary exposure to weathered crude oil was considerably less than has been reported in studies using unweathered crude oil.	Weathered crude oil is substantially less toxic to mallard ducks than unweathered crude oil. Ingestion of a diet containing weathered crude oil at 20 g/kg caused reductions in eggshell thickness and strength, which could result in reduced hatching success of ducklings.	Stubblefield *et al.* (1995b)

Table 3.16 Toxicological Benchmarks for Avian Receptors Exposed to Crude Oil

Study Design	Effects	Benchmark Derivation	Reference
Fresh South Louisiana crude oil was fed to mallard ducklings at concentrations of 0.025, 0.25, 2.5 and 5% of diet from hatching to 8 weeks of age.	Growth was depressed in birds receiving a diet containing 5% oil but there was not oil-related mortality. Diets containing as low as 0.25% oil caused behavioural response. Liver hypertrophy and splenic atrophy were evident in birds fed 2.5% or 5% oil. Some biochemical effects were noted, and tubular inflammation and degeneration in the kidney were noted in birds fed the 5% diet. High concentrations of oil in the diet impaired development of the wings and flight feathers and caused stunting.	Exposure to fresh crude oil over an 8 week period caused impaired development of mallard ducklings at a dose of 0.824 g/kg body weight/day.	Szaro *et al.* (1978)
Fresh South Louisiana crude oil was fed to mallard ducks at concentrations of 0.25 and 2.5% of diet for 26 weeks.	No birds died during the study, nor were body weights significantly depressed. Oviduct weight was greatly reduced on necropsy in ducks on the 2.5% diet, and was also significantly reduced in ducks on the 0.25% diet. Egg production was lower in ducks fed oil in the diet, however, the hatchability of eggs was not significantly different, and there was no effect on eggshell thickness. No significant effects were observed on liver weight, although spleen weight was reduced on the 2.5% diet.	Exposure to fresh crude oil over a 26 week period resulted in reduced egg production. The reduction was about 14% in ducks fed 0.25% oil in diet, and was accompanied by reduced oviduct weight. The 0.25% diet equates to a dose of approximately 0.2 g/kg body weight/day.	Coon and Dieter (1981)

Risk to mammals and birds is evaluated on the basis of their exposure to the individual BTEX compounds, as well as to the Canada-Wide Standard petroleum hydrocarbon fractions, and individual (un-substituted) PAH compounds. The toxicity of these individual components is considered to be additive. In addition, to capture the potential toxicity of alkylated MAH and PAH compounds, as well as other hydrocarbon substances not otherwise accounted for, the total hydrocarbon mixture is summed to represent the total exposure of wildlife receptors to weathered crude oil, and the toxicity of the summed fractions will be compared to the empirically derived benchmark values described above.

3.5.3.2 Toxicity Benchmarks for Exposure of Wildlife Receptors to Polcyclic Aromatic Hydrocarbons (PAHs)

Wildlife toxicity reference values (TRVs) are estimates of an exposure concentration (*e.g.,* mg/L in water) or ingested dose rate (*e.g.,* mg/kg body weight/day) that is unlikely to cause adverse effects on an exposed receptor organism. They are specific for chemicals, and should be as specific as possible for the receptor species. In practice, TRVs are generally applied to “mammals” or “birds” based on available information for surrogate species, due to the lack of data for most relevant receptor species. The chronic TRV is defined as the dose above which ecologically relevant effects might occur to wildlife species upon chronic exposure, and below which it is reasonable to conclude that effects would not occur (US EPA 2005).

While toxicological data are available for a number of individual PAH compounds (*e.g.,* naphthalene, phenanthrene, benzo(a)pyrene), they are not available in explicit form for most alkyl PAH compounds. In addition, many of the studies that are available are of limited utility due to details of study design and other factors. The US EPA (2007) conducted a screening evaluation of PAH toxicity to wildlife and other ecological receptors as part of the ecological soil screening level (EcoSSL) process. Within this process, PAHs were divided into low (3 or fewer ring) and high molecular weight (4 or more ring) classes. An extensive literature search identified 5,478 papers that contained PAH toxicity data potentially suitable for developing mammalian or avian TRVs. Of these, only 46 studies met acceptance criteria, and only two

contained data for avian species. This was judged by US EPA (2007) to be insufficient to support the development of a TRV for avian receptors.

Harwell *et al.* (2012) reviewed the US EPA (2007) EcoSSL report for PAHs and concluded that the avian toxicity data were not suitable as a basis for deriving TRVs to evaluate the exposure of harlequin duck to residual PAHs from the EVOS. This was due to limitations regarding the species tested, as well as the limited chemical representation (singular low- and high-molecular weight compounds), which did not support extrapolation to a TPAH mixture. Upon further review, however, they concluded that the studies of Stubblefield *et al.* (1995a, b) could provide such a basis. Using the studies by Stubblefield, Harwell *et al.* (2012) derived NOAEL-based TRVs of 2.00 and 2.14 mg/kg body weight/day and LOAEL-based TRVs of 19.56 and 22.01 mg/kg body weight/day, for male and female mallard ducks, respectively.

Harwell *et al.* further reviewed studies that had been identified by US EPA (2007) to determine whether data were available that could be compared with the proposed TRVs. Studies appropriate for comparison included work involving Cassin’s auklet, black oystercatcher, Leach’s storm petrel, sanderling, common murre, wedge-tailed shearwater, black-legged kittiwake, herring gull and Atlantic puffin. Although these studies did not individually provide information from which a TRV could be derived, they did not contra-indicate the TRV derived by Harwell *et al.* (2012). On this basis Harwell *et al.* (2012) concluded that the NOAEL and LOAEL TRVs calculated using data for mallard duck exposed to weathered Exxon Valdez crude oil provide conservative estimates for TRVs that can be applied to other species.

Avian NOAEL and LOAEL TRVs for ingestion of TPAH of 2.0 and 20 mg/kg body weight/day are adopted for use with seabirds and shorebirds in the present DQERA.

For mammals, Harwell *et al.* (2010) re-evaluated the mammalian data reported by US EPA (2007). The EcoSSL report stated that the best use of the available data was to calculate total doses of low- or high-molecular weight PAHs. However, Harwell *et al.* (2010) were able to present the data in terms of TPAH exposure, and selected a geometric 95% lower confidence interval as the TRV representing exposures based on both the NOAEL and the LOAEL. The selected TRV values for TPAH exposure for the NOAEL and LOAEL were 51.8 and 83.6 mg/kg body weight/day, respectively, and these values are adopted for mammalian exposures in the present DQERA.

3.6 Risk Characterization

The purpose of risk characterization is to evaluate the evidence linking COPC with adverse environmental effects by combining information from the exposure and hazard assessments. The potential for adverse environmental effects was quantified by comparing the concentration or dose of a substance that can be tolerated, or below which adverse environmental effects are not expected (*i.e.,* benchmark), to the expected EPC. The quotient of the two ([COPC concentration]/[benchmark]) is referred to as a hazard quotient (HQ). Acute and/or chronic hazard indices (HI) are derived for chemicals with similar modes of action and target organs by summing the HQ of individual COPC. This methodology follows that of Di Toro and co-workers (Di Toro *et al.* 2000, Di Toro and McGrath 2000). An acute or chronic HI value less than 1.0 indicates that the exposure concentration is less than the threshold of toxicity for the chemical class evaluated. Given the conservative approach to the estimation of exposure and selection of benchmarks, a chronic HI less than 1 is not expected to result in adverse effects and no further assessment is required. Since the acute to chronic ratio for toxicity of non-polar narcotic substances has a value of approximately 5 (Di Toro *et al.* 2000), a chronic HI value greater than 5 may also be used as an indicator of potential acute effects.

3.6.1 Hazard Quotients and Hazard Indices

A hazard quotient (HQ) is derived by dividing the exposure point concentration in environmental media (*e.g.,* soil, sediment, water), or the dose of a COPC ingested by a receptor organism as a result of its

exposure to foods as well as other environmental media, by a benchmark value representing a safe concentration or dose. If a hazard quotient value is less than unity, there is no meaningful risk present. Where chemical substances have additive interactions (as is generally assumed to be the case for hydrocarbon exposure), the hazard quotient values for two or more substances may be summed to estimate a hazard index (HI) for a mixture of substances having similar chemical structure, mode of toxic action, and target tissue or organ. If a hazard index value is less than unity, there is no meaningful risk present.

When used in risk assessment, toxicity benchmark values may be derived from regulatory standards (for example, the Canada-Wide Standard for TPH fractions), or may be derived from toxicological studies for individual compounds (such as benzene). In either case uncertainty factors or safety factors may be applied to results that have been selected to represent the lower range of concentrations demonstrated to cause a toxicological response in one or more receptor species. As a result, while it is reasonable to believe that an adverse effect is not likely to occur as long as a particular benchmark value is not exceeded (or the HI/HQ value is less than 1), it does not follow that an adverse effect is likely to occur if the benchmark value is exceeded (or the HI/HQ value is greater than 1). Rather, an HI or HQ value exceeding 1 should be interpreted as being indicative of the potential for an adverse effect to occur. A variety of factors, including the nature of the adverse effects identified in the derivation of the toxicological or regulatory benchmark value, as well as a more rigorous analysis of the nature and extent of conservatism built into the underlying components of the assessment, should be considered.

3.6.2 Chemical Interactions

Risk assessments are complicated by the fact that most toxicological studies are conducted using a single chemical whereas environmental exposures generally involve more than one contaminant. Calculating a HQ for exposure to mixture of COPC is problematic because all COPC do not have the same modes of action, target endpoints or magnitudes of toxicity. Chemicals in a mixture may interact in four general ways to elicit a response:

- Non-interacting – chemicals do not produce a response in combination with each other; the toxicity of the mixture is the same as the toxicity of the most toxic component of the mixture.
- Additive – chemicals have similar targets and modes of action but do not interact; the hazard for exposure to the mixture is simply the sum of hazards for the individual chemicals.
- Synergistic – there is a positive interaction among the chemicals such that the response is greater than would be expected if the chemicals acted independently or in an additive manner.
- Antagonistic – there is a negative interaction among the chemicals such that the response is less than would be expected if the chemicals acted independently or in an additive manner.

There are chemical classes that have similar modes of action and target organs (*i.e.,* they act in an additive manner), and in these cases, an appropriate characterization of risk is achieved by summing the HQ for each compound. HQ for BTEX and TPH are summed to derive a single HI. The PAH substances are also treated as a class, but are not included with BTEX and TPH since measures of TPH also include the specific PAH compounds.

3.7 Discussion of Uncertainty and Confidence

Limitations associated with the administrative boundaries and uncertainties of the risk assessment, in addition to conservative assumptions used in the modelling, are identified and discussed to provide perspective on the certainty and confidence that should be placed on the predictions.

This ERA step includes a qualitative assessment of the level of confidence that can be placed in the analysis and results. Risk assessments normally include elements of uncertainty, and these uncertainties are addressed by incorporating conservative assumptions (*i.e.,* assumptions that are likely to over-state rather than under-state the actual adversity of outcomes) into the analysis. Discussion of certainty and confidence in the analysis is provided in order to put these considerations into context, and to demonstrate that the conclusions are either not sensitive to key assumptions, or that the assumptions used are conservative.

4.0 DQERA FOR A CRUDE OIL SPILL FROM A HYPOTHETICAL LOADING ACCIDENT AT THE WESTRIDGE MARINE TERMINAL

4.1 Overview of Westridge Marine Terminal Operations

The WMT is situated on the south shore of Burrard Inlet, in Vancouver Harbour. The existing facility performs two primary functions: loading crude oil onto tankers and barges for transportation to the United States and elsewhere; and offloading jet fuel for onshore pipeline transport to the Vancouver Airport.

Vessels calling at the WMT follow all applicable national and international rules and regulations and follow procedures as recommended in the latest version of the International Safety Guide for Oil Tankers and Terminals (ISGOTT). All vessels arriving at the terminal will be assisted by tugs during berthing operations. Once a tanker has been assisted to its assigned berth by the tugs, the vessel's mooring lines will be secured by trained terminal personnel. A containment boom will also be deployed prior to, and throughout, all oil loading operations, including loading arm connection, cargo loading and disconnection procedures.

The vessel loading process at WMT is a closed system, with oil loading via loading arms and displaced vapour being collected and transmitted to onshore processing facilities via the vapour piping system. After loading operations are completed, the terminal personnel drain and disconnect the loading arms and vapour line in accordance with written terminal procedures. Once final departure procedures and documentation have been completed, Pilots then board the vessel, tugs are made fast, the gangway is removed, and the ship's main propulsion and steering system are tested for operational readiness. In coordination with the ship's crew, the shore-side mooring crew will then release the mooring hooks as directed by the vessel master and mooring lines will be taken aboard the vessel, after which it will be ready for departure with the assistance of the tugs.

Additional details of the loading operations and procedures at the WMT are provided in ESA Volume 4C Section 6.1.10.

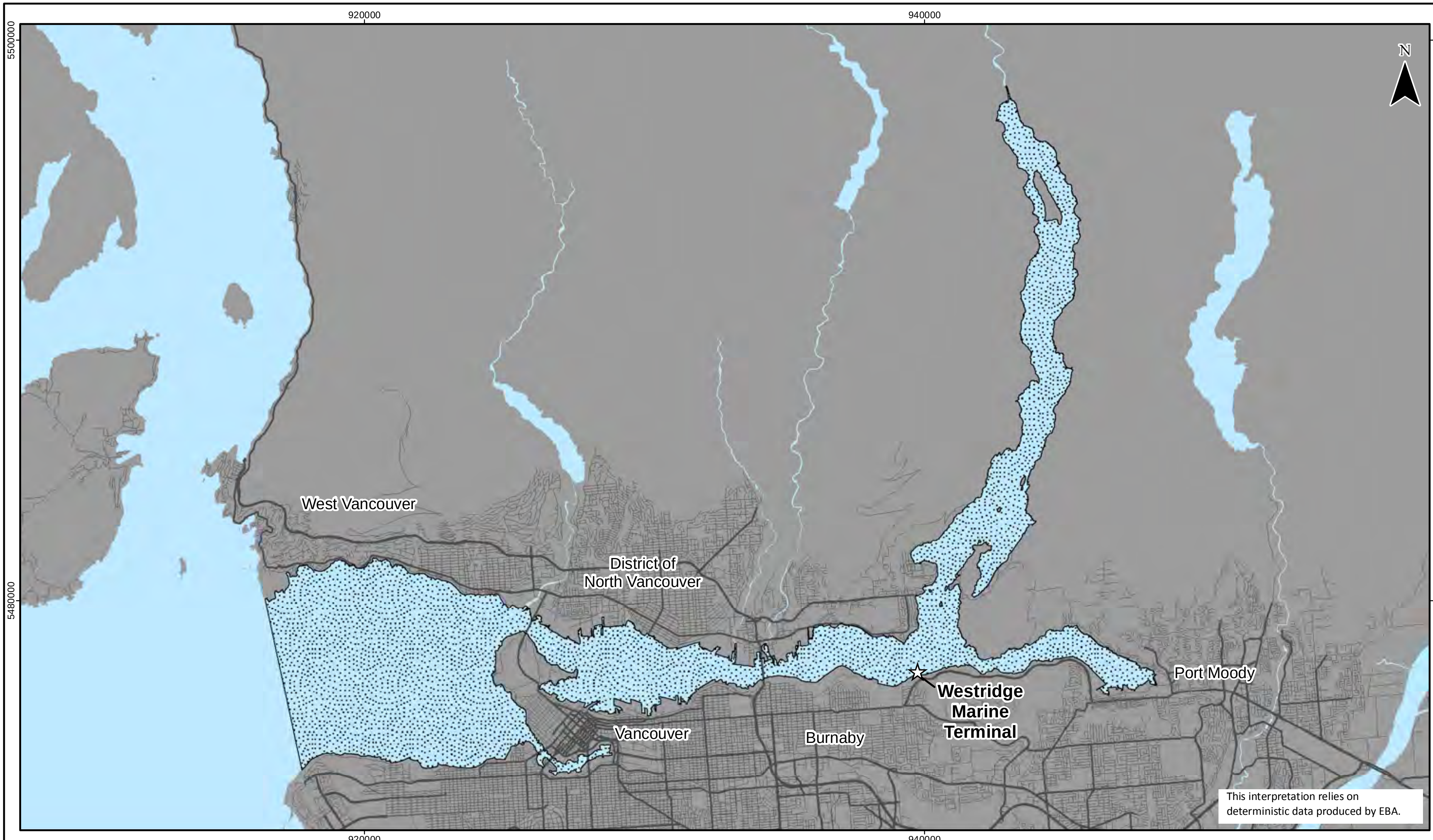
4.2 Problem Formulation

An overview of the problem formulation stage of the DQERA is provided in Section 3.2. The key objectives of this stage of the risk assessment include the following:

- characterization of the geographic areas where the DQERA is being conducted
- identification of representative crude oil products being assessed, which individual COPC are present, and mechanisms of release to the environment
- identification of exposure media and pathways
- identification and characterization of representative ecological receptors.

4.2.1 Spatial Boundaries of the Assessment

Spatial boundaries of the DQERA for crude oil spills originating from the WMT include the geographic extent where potential effects are expected to be measurable (*i.e.,* the modelling domain of for the deterministic crude oil spill model). The Regional Study Area (RSA) is defined as the area of ecological relevance where effects could potentially result from spills. This area is effectively established by the physical limits of modelling domain for the deterministic crude oil spill modelling and includes the area of Vancouver Harbour, and Burrard Inlet east of the First Narrows, including Indian Arm and Port Moody Arm. The Regional Study Area for spills originating at the WMT is shown in Figure 4.1.



Stantec

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ALL LOCATIONS APPROXIMATE

MAP NUMBER 123110494_016D		PAGE SHEET 1 OF 1
DATE Apr 2014	TERA REF. REF	REVISION A
SCALE 1:200,000	PAGE SIZE 8.5 x 11	DISCIPLINE ERA
DRAWN HW	CHECKED PM/AS	DESIGN MS/HW

☆ Westridge Marine Terminal

Regional Study Area Boundary

Road

TRANS MOUNTAIN

FIGURE: 4.1

Regional Study Area for Hypothetical Spills Originating at the Westridge Marine Terminal

4.2.2 Crude Oil Products Selected for Assessment

Trans Mountain has selected Cold Lake Winter Blend (CLWB) as a representative product for the purposes of evaluating environmental effects of hypothetical crude oil spills. Rationale for the selection of CLWB and a summary of the physical and chemical properties of the product are provided in Sections 3.2.3, 3.2.4 and 3.2.5.

4.2.3 Hypothetical Spill Scenarios Considered in the Assessment

This DQERA evaluates the potential environmental effects resulting from two spill scenarios originating from hypothetical loading accidents at the WMT. The nature of the hypothetical spills has been based on failure/risk analysis completed by Det Norske Veritas (DNV 2013). The DQERA is based on the results of crude oil spill fate and transport modelling completed by EBA Engineering Consultants Ltd. (EBA 2013). The spill scenarios presented here consider both a credible worst case (CWC) spill, and a smaller spill.

The CWC spill at this location was assessed assuming a hypothetical release volume of 160 m^3. At 160 m^3, this spill is larger than the credible worst case spill resulting from a rupture of a loading arm (*i.e.,* 103 m^3) as predicted by DNV. It is, however, substantially smaller than the capacity (over 1,500 m^3) of the precautionary boom that will be deployed around each berth while any cargo transfer activities are taking place. While this larger spill volume could be retained within the pre-deployed boom, for the purposes of the oil spill modelling, 20% of the spilled oil (*i.e.,* 32 m^3) is assumed to escape the containment boom. This condition was chosen to ensure a conservative approach to the assessment of spill response requirements at the site, and does not reflect Trans Mountain's expectation for performance of the precautionary boom which will be in place to fully contain any such release at the WMT. The credible worst case oil spill volume resulting from this scenario has been calculated by Det Norske Veritas (DNV 2013) to be 103 m^3, and is deemed to be a low-probability event, with likelihood of occurring once every 234 years.

The smaller spill volume of 10 m^3 was based on a hypothetical leak in a loading arm. The spill volume was estimated by DNV based upon global statistics for spills attributed to loading accidents at similar facilities, and from historical incident reports from the WMT. The smaller spill is assumed to be completely retained within the precautionary boom.

4.2.4 Selection of Spill Scenarios for 3-D Deterministic Modelling

EBA (2013) completed stochastic oil spill modeling for the simulated 160 m^3 CWC spill of CLWB, with 80% (128 m^3) contained within the boom and 20% (32 m^3) escaping from confinement and spreading into Burrard Inlet. All four seasons were modeled stochastically, including winter (January – March), spring (April – June), summer (July – September) and fall (October – December). The stochastic models relied upon weather, currents and tide data for the years 2011 and 2012. A model simulation was initiated every three hours, each run being independent of the others (except to the extent that the weather, tide and hydrological data are continuous records). This simulation approach resulted in approximately 736 simulations for each season. The full set of approximately 2,900 individual independent simulations covers the expected fate and behavior of spilled oil over the course of a full year of weather and tidal observations, and in this sense the stochastic data set is as realistic as possible. Environmental conditions in 2011 and 2012 were analyzed and compared to a 30 year wind record: the selected period (October 2011 – September 2012) was confirmed to be representative of wind speeds and wind directions.

Subsequent to the completion of the stochastic oil spill modeling, one of the 2,900 individual simulations was selected for 3-D deterministic modeling. The 3-D modelling is used to compute the fate of the oil in the water column, on the water surface and in the air. The objective of the selection process was to identify a representative scenario that was realistic, while tending to be conservative from both the

ecological and human health risk perspective. Selection of the stochastic simulation for deterministic oil spill modeling proceeded step-wise, as outlined below.

First, consideration was given to the four seasons that were modeled stochastically. It was determined that the summer season would be the focus of the deterministic modeling as warmer water and air temperatures would facilitate more rapid dissolution and/or volatilization of lighter hydrocarbons into water or air, respectively. At the same time, generally lower wind speeds during the summer would result in less wave action (hence, less vertical mixing of the water column, and higher concentrations of dissolved hydrocarbons in the surface water layer), as well as less dilution of vapours in air. Summer season conditions are considered conservative because increased hydrocarbon concentrations in water and air would increase risk to people and organisms relative to colder ambient conditions. In addition, people and a wider variety of organisms are more likely to be in close proximity to a spill at the WMT during the summer months.

Second, consideration was given to the length of shoreline oiled, as oil spill effects on shorelines are among the more obvious and profound environmental effects of spills (both with respect to people and organisms). To this end, the median length of shoreline oiled as a result of the spill was determined based on the 736 summer stochastic simulations. The specific simulations resulting in a length of oiled shoreline most closely approximating this median value were then identified and examined. Twenty (20) simulations meeting this criterion were identified and brought forward.

The median length of shoreline oiled was identified as a selection criterion in order to balance: i) the need to address potential risks to aquatic organisms, which are primarily driven by the amount of oil dissolved water; ii) the need to address potential risks to shoreline habitat, shorebirds and terrestrial wildlife species, which are primarily driven by the length of shoreline oiled; and iii) the need to address the potential risks that might be presented to humans on shore and on the water, from chemical exposures associated with an oil spill to the Burrard Inlet. Beyond the selection criteria discussed above, a number of additional considerations factored into the selection of the single simulation from the stochastic modeling for deterministic oil spill modeling. These additional criteria were:

- the maximum thickness of the oil modelled on water during the simulation
- the time elapsed to first contact with the shoreline
- the exposure duration for the oil on water
- the distribution of total hydrocarbon between water, shore and air (*i.e.,* the mass balance).

Thus, as the third step of the selection process, each of the 20 stochastic simulations identified above was ranked (*e.g.,* high, medium, low) according to how well the selection criteria were satisfied. Higher weighting was given to those simulations that demonstrated greater thickness of the oil reaching the shoreline, shorter time to first contact with the shoreline, longer exposure time on water, and higher percentage of hydrocarbon in air.

As the final step of the selection process, the oil spill modelling outputs for the two preferred simulations were compared to the outputs for the summer-season stochastic modeling to evaluate which was more "typical" of spills during that season. On the basis of this final visual comparison, the deterministic simulation initiated based on the weather, current and tidal patterns from 10:00 pm on August 21 was selected as a "reasonable, yet conservative" scenario based on a credible worst case spill volume.

It is important to emphasize that the "time" and "date" of the hypothetical spill selected for the DQERA are by themselves inconsequential, and simply provide a particular set of weather, tide and hydrological

conditions prevailing during the simulation. The environmental conditions for this date and time were also used to model the fate of oil resulting from the smaller (10 m^3) spill.

4.2.5 COPC Concentrations in Representative Hydrocarbons

The fate and environmental effects of spilled crude oil depend upon the characteristics of the oil. For these simulations, samples of CLWB diluted bitumen were obtained and analyzed to characterize the hydrocarbon composition. A set of 17 pseudo-components was developed, suitable both for environmental fate modeling and toxicological assessment. The pseduocomponent approach allows for a simplified yet realistic description of the oil by breaking it down into a set of fractions, each with defined physical and chmical characteristics. Concentrations of COPC in fresh CLWB are discussed and presented in Section 3.2.5. The EBA oil spill fate modelling provided predictions of the distribution and thickness of floating oil, the mass of bulk oil stranded on shorelines, and the pseudocomponent concentrations dissolved in the water column, and dispersed in the air above the spilled oil. These data were evaluated for various time intervals (*i.e.,* hourly for the 60-hour duration of the simulation) to determine concentrations of COPC in the various media throughout the simulation and for consideration in the DQERA.

Potential environmental effects to ecological receptors from exposure to the COPC in various media were evaluated as outlined in methods presented in Section 3. The potential effects from exposure to COPC for various ecological receptors are discussed presented in the following sections.

4.3 Exposure Assessment and Effects Characterization, 10 m^3 Spill

As it has been assumed that the hypothetical spill of 10 m^3 of CLWB would be completely retained within the containment boom, no modelling of surface oiling or shoreline oiling was completed. However, the potential for oil constituents to dissolve in water, or to volatilize to air was recognized, and these exposure pathways were modeled for the smaller spill. In addition, the potential for small quantities of oil to be deposited to sediment was also recognized and simulated.

Owing to the small size of this spill simulation, and because of the consideration that will be given to potential bioaccumulation and food-chain effects of hydrocarbon exposure for the larger (160 m^3) spill scenario, a detailed evaluation of food-chain effects corresponding to the smaller spill was not undertaken.

4.3.1 Potential Narcotic Effects on Marine Biota from Exposure to Hydrocarbons in the Water Column

Hydrocarbon compounds dissolved in the water column can be accumulated by marine organisms and cause narcosis, potentially leading to death in a short period of time. The narcosis endpoint is the primary cause of acute toxicity for a broad range of marine receptor organisms (*e.g.,* algae, invertebrates and fish) when exposed to high concentrations of hydrocarbon compounds including those originating from crude and refined oils (Di Toro *et al.* 2000, French McCay 2009).

The deterministic modelling completed by EBA (2013) provided pseudo-component concentrations for each grid cell of the water column within the model domain, and for hourly time intervals for the full length of the model simulation, up to 60 hours for hypothetical spills at the WMT. Grid sizes for evaluating effects of hydrocarbon in the water column were based on a surface grid size of 125 m x 125 m (EBA 2013), and were also stratified according to depth in the water column.

The potential for acute effects to marine biota swimming or suspended in the water column (including but not limited to plants, invertebrates and fish) resulting from exposure to hydrocarbons dissolved in the water column were evaluated using the non-polar narcosis endpoint. Using GIS, the predicted concentrations of each hydrocarbon pseudo-component in each cell of the full 3-D model domain were

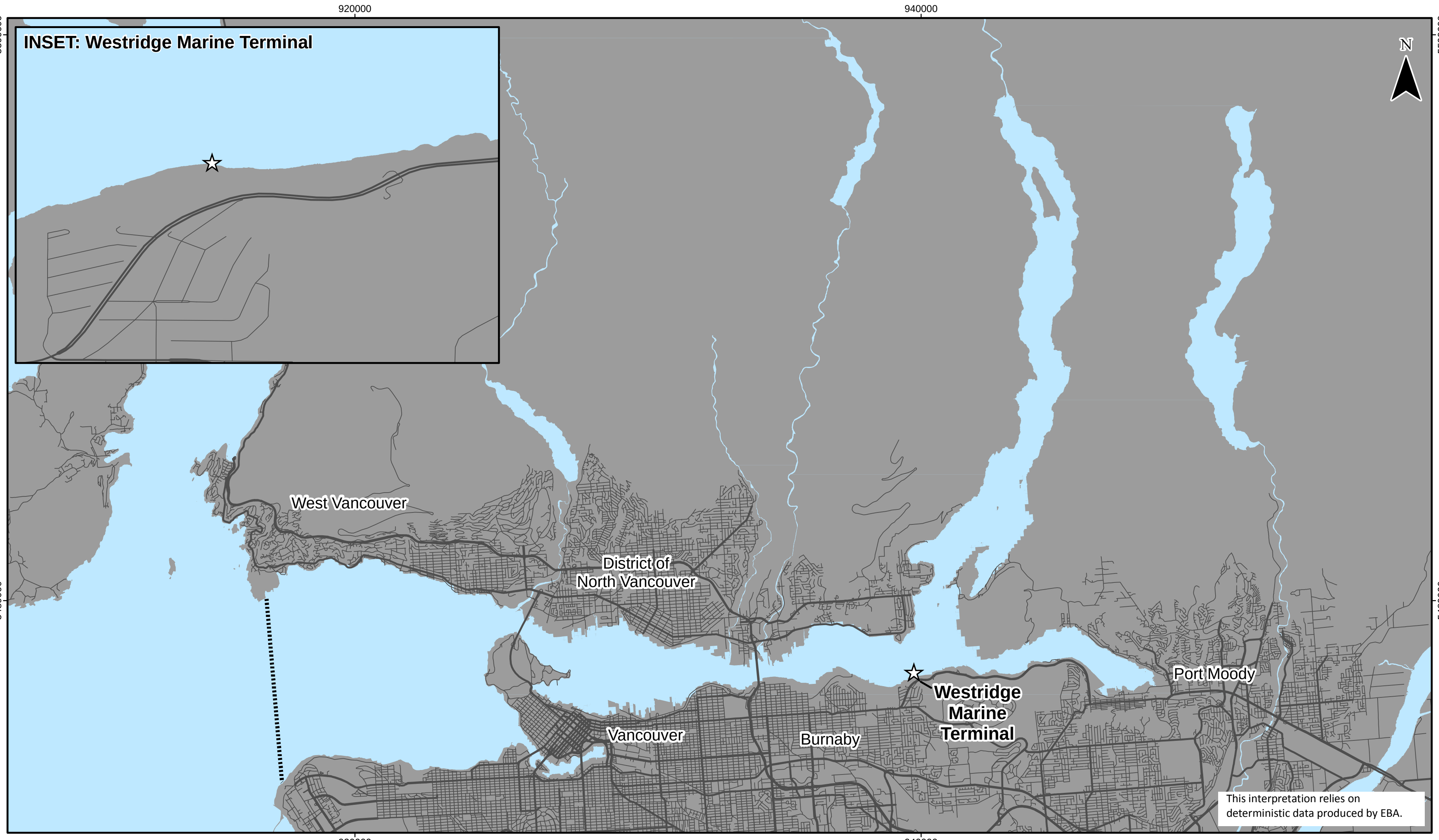
compared to these benchmark values, and the results within each cell were summed to obtain an integrated hazard quotient in the form of toxic units (TU). A running average value was then calculated, and the maximum 60-hour average TU value at any depth below each grid square was selected for presentation. This 60-hour average TU value was conservatively compared to the 96-hour TU value of 1 which would correspond to 50% mortality of a sensitive (5^{th} percentile on the species sensitivity distribution) species. A potential for direct mortality of marine organisms, including fish and invertebrates, was identified in grid squares where the 60-hour average TU value exceeded unity. In addition, the maximum instantaneous TU value at any location was also considered, to ensure that extreme values did not occur that would lead to fish mortality in a much shorter period of time than the 60-hour average exposure might indicate.

Figure 4.2 presents the mapped results of the narcosis TU values averaged over the 60 hours period of maximum exposure within each grid square for the smaller (10 m^3) spill at the WMT. Based on the results, predicted 60-hour average TU values do not exceed the minimum threshold contour of 0.05, and the highest 60-hour average narcosis TU value is calculated to be 0.0261 TU. The highest instantaneous TU value at any location was slightly less than 0.31. As the maximum 60-hour average TU value does not exceed 1 for any grid cell, no fish kill or other acute narcotic effects to marine biota exposed to dissolved hydrocarbons in the water column would be expected to result from the release of 10 m^3 of CLWB.

4.3.2 Potential Developmental Effects from Exposure to TPAH in the Water Column

Short-duration exposure to low concentrations of dissolved PAH compounds, particularly 3- and 4-ring PAHs and alkyl PAHs, can lead to adverse effects on developing fish eggs and embryos, a syndrome characterised as blue sac disease (BSD, Heintz *et al.* 1999, Carls *et al.* 1999, McIntosh *et al.* 2010, Incardona *et al.* 2013). This syndrome is characterized by cardiotoxicity, edema of the pericardial space or yolk sac, and craniofacial or spinal malformations in developing embryos, with newly fertilized eggs demonstrating the highest level of sensitivity. As detailed in Section 3, the lowest severity of effects (usually non-lethal) has been identified at TPAH concentrations around 1 µ/L, with increasing severity as exposure concentrations increase between 10 and 100 µg/L. Recent experimental findings (Wu *et al.* 2012, Martin *et al.* 2014) tend to show that exposure to water-accommodated fractions of TPAH alone may not be sufficient to induce BSD (depending upon the viscosity of the oil and its tendency to disperse as a function of wind and wave action), although the enhanced dissolution of hydrocarbons provided by chemical dispersant use increases exposure and the likelihood of subsequent BSD symptoms.

For the purposes of evaluating effects from exposure to TPAH in the water column, the concentrations of PAH compounds present within each grid cell and hourly time interval were calculated from the predicted pseudo-component concentrations provided by the EBA model output. Individual PAH concentrations (with the exception of naphthalene, which comprises only about 3% of the TPAH in CLWB, and is not known to induce BSD) were summed to calculate a TPAH concentration in water. The maximum 24-hour average TPAH concentrations in any water layer beneath each grid square (based on hourly time interval data) were projected to the water surface and mapped. The potential for adverse effects on developing fish eggs and embryos was identified in grid squares where the maximum 24-hour average value exceeded toxicity benchmarks for TPAH as described above (*i.e.*, a threshold of 1 µg/L for non-lethal developmental effects, with expectation of increasing severity of effects including lethality at higher concentrations). The maximum absolute hourly TPAH concentration in water at any location was also recorded.



km
0 1.5 3 4.5 6

ALL LOCATIONS APPROXIMATE

MAP NUMBER 123110494_003D		PAGE SHEET 1 OF 1
DATE Apr 2014	TERA REF. REF	REVISION A
SCALE 1:200,000	PAGE SIZE 8.5 x 11	DISCIPLINE ERA
DRAWN HW	CHECKED PM/AS	DESIGN MS/HW

☆ Westridge Marine Terminal

Regional Study Area Boundary

Road

Narcosis Toxic Units for 60 hour Exposure*

- 0 - 0.05
- > 0.05 - 0.25
- > 0.25 - 0.5
- > 0.5 - 0.75
- > 0.75 - 1.0
- > 1.0

*NOTE: Maximum Value is 0.0261

FIGURE: 4.2

Narcosis Toxic Units for 60-Hour Exposure (Projected to the top layer)

WESTRIDGE MARINE TERMINAL DETERMINISTIC SIMULATION 10 m³ SPILL