AMAP

ASSESSMENT 2007

Oil and Gas Activities in the Arctic – Effects and Potential Effects

Volume Two

2

Arctic Monitoring and Assessment Programme (AMAP), Oslo, 2010

Assessment 2007: Oil and Gas Activities in the Arctic – Effects and Potential Effects*

Chapter 1
Introduction

Chapter 2
Oil and Gas Activities in the Arctic

Chapter 3
Social and Economic Effects of Oil and Gas Activities in the Arctic

Chapter 4
Sources, Inputs and Concentrations of Petroleum Hydrocarbons, Polycyclic Aromatic Hydrocarbons, and other Contaminants Related to Oil and Gas Activities in the Arctic

Chapter 5
Effects of Oil and Gas Activity on the Environment and Human Health

Chapter 6
Status and vulnerability of Arctic ecosystems

Chapter 7
Scientific Findings and Recommendations

**Bold indicates chapters contained in this volume*

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Glossary

Oil and Gas Industry Conversions

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Preface

This assessment report details the results of the 2007 Assessment of Oil and Gas Activities in the Arctic conducted under the auspices of the Arctic Council and coordinated by the Arctic Monitoring and Assessment Programme (AMAP).

It provides the accessible scientific basis and validation for the statements and recommendations made in the report 'Arctic Oil and Gas 2007' [1] *that was delivered to Arctic Council Ministers in April 2008. It includes extensive background data and references to the scientific literature, and details the sources for figures reproduced in the 'Arctic Oil and Gas 2007' report. Whereas the 'Arctic Oil and Gas 2007' report contains recommendations that specifically focus on actions aimed at improving the Arctic environment, the conclusions and recommendations presented in this report also cover issues of a more scientific nature, such as proposals for filling gaps in knowledge, and recommendations relevant to future monitoring and research work, etc.*

The assessment constitutes a compilation of the prevailing knowledge about oil and gas activities in the Arctic region to the middle of the decade and an evaluation of this information. It was prepared as far as possible in a systematic and uniform manner to provide a comparable knowledge base for the circum-Arctic countries that builds on earlier work and can be extended through continuing work in the future.

The assessment is published in three volumes. This volume, Volume 2, includes Chapters 1, 4, 5 and 7 of the assessment:

Chapter 1 · Introduction

Chapter 4 · Sources, Inputs and Concentrations of Petroleum Hydrocarbons, Polycyclic Aromatic Hydrocarbons, and other Contaminants Related to Oil and Gas Activities in the Arctic

Chapter 5 · Effects of Oil and Gas Activity on the Environment and Human Health

Chapter 7 · Scientific Findings and Recommendations

Chapters 1 and 7 of the assessment are included in all three volumes as they provide important information concerning the content and organization of the material and summarize the overall results of the assessment in case other volumes are not accessible to the reader.

The assessment presented in this report is the responsibility of the scientific experts involved in the preparation of the assessment. Lead countries for this Arctic Oil and Gas Assessment were Norway and the United States. The assessment is based on work conducted by a large number of scientists and experts from the Arctic countries (Canada, Denmark/Greenland/Faroe Islands, Finland, Iceland, Norway, Russia, Sweden, and the United States), together with contributions from indigenous peoples' organizations, from other organizations, and from experts in other countries.

AMAP would like to express its appreciation to all of these experts, who have contributed their time, effort, and data; and especially to the lead experts who coordinated the production of this report, and to referees who provided valuable comments and helped ensure the quality of the report. A list of the main contributors is included in the acknowledgements on page vii of this report. The list is not comprehensive. Specifically, it does not include the many national institutes, laboratories and organizations, and their staff, which have been involved in the various countries. Apologies, and no lesser thanks are given to any individuals unintentionally omitted from the list. Special thanks are due to the lead authors responsible for the preparation of the various chapters of this report.

The support of the Arctic countries is vital to the success of AMAP. AMAP work is essentially based on ongoing activities within the Arctic countries, and the countries also provide the necessary support for most of the experts involved in the preparation of the assessments. In particular, AMAP would like to express its appreciation to Norway and the United States for undertaking a lead role in supporting the Oil and Gas assessment. Special thanks are also offered to Canada, Denmark, Norway, United States and the Nordic Council of Ministers for their financial support to the work of AMAP, and to sponsors of projects that have delivered data for use in this assessment.

The AMAP Working Group that was established to oversee this work, and the Arctic oil and gas assessment expert group are pleased to present its assessment.

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[1] AMAP, 2008. Arctic Oil and Gas 2007. Arctic Monitoring and Assessment Programme (AMAP), Oslo, Norway. xiii+40 pp.

Chapter 1
Introduction

Author: Simon Wilson

1.1. Background

In 1997, the Arctic Monitoring and Assessment Programme (AMAP) presented the results of its first assessment of the pollution status of the Arctic. The reports (AMAP, 1997, 1998a) detailing the results of that assessment included a chapter on 'petroleum hydrocarbons', which described the regional development and transportation of petroleum resources, the environmental fate of petroleum hydrocarbons, and their levels and effects in the Arctic environment.

That first AMAP assessment of petroleum hydrocarbons in the Arctic was prepared at a time when, after a period of intense activity during the 1980s, largely driven by high oil prices following the oil crises of the early 1970s and early 1980s, interest in Arctic oil and gas resources was falling, or was at least being considered a low priority by governments and industry.

Major oil production activities were identified as an issue of 'sub-regional' environmental concern in parts of Western Siberia, and on the North Slope of Alaska where the *Prudhoe Bay* fields had been rapidly developed during the late-1970s and 1980s. As with the Prudhoe Bay development, oil production from fields in the Mackenzie Valley area of Canada were already past their peak by the beginning of the 1990s. Intensive exploration activities in the Canadian Arctic had revealed the presence of substantial quantities of oil, and in particular gas in the Mackenzie Delta/Beaufort Sea area, but the prevailing economic conditions meant that, with the exception of a small amount of oil production from the *Bent Horn* field, these were not commercially exploitable, and discovery wells were therefore capped for possible future production. Offshore, significant exploration activities had been, or were being conducted in the Bering, Beaufort, Norwegian and Barents Seas. Building on its North Sea operations, outside of the Arctic, Norway was just starting production from Norwegian Sea fields, with good prospects of discoveries in the Barents Sea.

Despite the limited extent of Arctic oil and gas development at the time of the first AMAP assessment, two major oil spill events occurring just prior to the publication of the AMAP assessment had focused considerable international attention on the potential threats for environmental impacts associated with oil and gas activities in northern areas. These were the *Exxon Valdez* accident in Prince William Sound in southern Alaska, and the well-publicized 'Komi spill' in Russia, from a pipeline near Usinsk in the lower Pechora Basin.

The first AMAP assessment of petroleum hydrocarbons in the Arctic presented 15 major conclusions, together with the following (main) recommendations:

In regions of existing or developing oil and gas exploitation and transportation in the Arctic:

- *Steps should be taken to harmonize the monitoring of petroleum hydrocarbon levels and effects.*
- *Nautical charts and environmental sensitivity mapping for the Arctic area should be improved as an important counter-measure for oil spills.*
- *Methods and techniques for combating oil spills in ice-covered areas should be developed to reduce damage when spills occur.*

These conclusions and recommendations were reported to Ministers of the eight Arctic countries at the Third Ministerial meeting of the Arctic Environmental Protection Strategy (AEPS) in Tromsø, Norway in 1997.

Work has also been conducted under other Arctic Council Working Groups relating to oil and gas activities in the Arctic, partly in response to these recommendations. This has resulted in reports prepared by the Arctic Council Working Group on Emergency Prevention, Preparedness, and Response (EPPR) on the *Arctic Shoreline Clean-up Assessment Technique (SCAT) Manual* (Owens et al., 2004), the *Arctic Guide for Emergency Prevention, Preparedness and Response* (EPPR, 2008) and the *Circumpolar Map of Resources at Risk from Oil Spills in the Arctic* (EPPR, 2002); and reports prepared by the Working Group on Protection of the Arctic Marine Environment (PAME) on *Arctic Offshore Oil and Gas Guidelines* (PAME, 1997, 2002, 2009), *Arctic Marine Strategic Plan* (PAME, 2004a) and *Guidelines for Transfer of Refined Oil and Oil Products in Arctic Waters* (PAME, 2004b).

1.2. Arctic Council's 2006 assessment of Oil and Gas Activities in the Arctic

In 2002, AMAP proposed to the Arctic Council that an update to its 1997 assessment of Petroleum Hydrocarbons in the Arctic be produced, for delivery in 2006. In the period since the publication of the first AMAP assessment, significant changes have occurred in the global economy with respect to demand for energy, and energy security considerations, which mean that renewed attention is being given to Arctic oil and gas resources. At the same time, assessments of the impacts of climate change (for example, the Arctic Climate Impact Assessment; ACIA, 2004, 2005) were indicating that, under scenarios for the not too distant future, Arctic conditions might be more favorable for resource development, and perhaps more importantly for the associated transportation of resulting production.

Recognizing this situation, and also recognizing that a comprehensive assessment of oil and gas activities in the Arctic should address issues beyond just the potential pollution threats from such development, the Arctic Council therefore requested that relevant working groups, under the lead of AMAP, prepare an assessment of Oil and Gas Activities in the Arctic.

1.2.1. Scope of the assessment

The Arctic Council Ministers (Arctic Council, 2004) directed that this assessment should build on and expand the AMAP assessment completed in 1997, and evaluate four types of impacts or effects associated with oil and gas activities in the Arctic:

- social and economic consequences

- environmental impacts from pollution
- environmental effects from physical impacts and disturbances
- effects on human health

These four components of the assessment constitute the framework for much of the information presented in this assessment report.

The assessment specifically does *not* include the relation between Arctic oil and gas development and global carbon dioxide (CO_2) emissions and greenhouse warming. This topic is addressed in other assessments, for example those by ACIA, the UN Intergovernmental Panel on Climate Change, and national assessments.

Similarly, this assessment focuses on petroleum hydrocarbons associated with oil and gas resource development activities, and not, for example, on use of petroleum products in the Arctic, or petroleum hydrocarbons in a more general sense. Chapters dealing with oil and gas activities (past, present and future), and socio-economic aspects of Arctic oil and gas development are, by definition, limited to addressing oil and gas activities. The chapter dealing with pollution aspects of petroleum hydrocarbons addresses sources associated with oil and gas activities, but includes information on other sources (natural sources, and sources associated with pollution from petroleum products, etc.) for comparative purposes. More information on, for example, polycyclic aromatic hydrocarbons (PAHs) associated with combustion sources can be found in the AMAP assessments on Persistent Organic Pollutants in the Arctic (AMAP, 1998b, 2004). In relation to the 'effects of contaminants', it is generally not possible to isolate effects due to petroleum hydrocarbons released as a result of oil and gas activities, from those released from other natural and anthropogenic sources. However, in connection with effects due to, for example, noise and physical disturbance, the impacts of oil and gas activities can be more readily distinguished and separately considered. Effects on human health are also only considered in this assessment in relation to non-occupational exposures resulting from oil and gas activities.

The possible consequences of increased Arctic oil and gas activity on climate change or other widespread environmental problems, such as ocean acidification or eutrophication is also outside the scope of this assessment.

Finally, the majority of the data presented in this assessment cover the time period up to around 2004/2005 – the latest data available at the time this assessment report was drafted. Some parts of the assessment, however, were subsequently updated to include more recent data where this could readily be included and where this complemented the assessment.

1.2.2. Geographical scope of the assessment

The geographical scope of this assessment is essentially a modified version of the 'AMAP area'. The AMAP area (see Figure 1.1) is a non-formal definition of the Arctic, but is based on several relevant physical and biological definitions, plus political designations, which together delimit an Arctic region that is appropriate for the purposes of AMAP assessments.

The oil and gas assessment area includes the Arctic production areas on the North Slope of Alaska, the Mackenzie Valley, the Norwegian offshore, and the West Siberian and Timan-Pechora basins of northern Russia – some of which have a long history of oil and gas development.

More generally, the assessment covers onshore oil and gas activities:

- in the United States (Alaska), north of the Arctic Circle;
- in Canada, in the petroleum provinces of the Yukon, the Northwest Territories and Nunavut, north of 60° N; and
- in Russia, in the petroleum hydrocarbon basins north of 60° N.

Offshore areas that fall within the assessment area include:

- the Norwegian Sea (the Norwegian continental shelf from 62° N to approximately 70° N, centered on the Haltenbanken area);
- the Barents Sea on the Norwegian-Russian continental shelf, which is a focus of increasing development and an area where marine transport of oil is expected to increase in coming years;
- the Pechora, Kara, Laptev and East Siberian Seas on the Russian shelf;
- on the continental shelf between Russia and the United States, the Bering Sea (the area north of the Aleutian Island chain) and the Chukchi Sea;
- on the US-Canada continental shelf, the Beaufort Sea; and
- the marine areas of the Canadian Arctic Archipelago.

Parts of the assessment area that were not considered in the first AMAP assessment of petroleum hydrocarbons include areas of West Greenland, especially the offshore waters between Greenland and Canada, and the Faroese shelf, where new exploration for oil and gas reserves has been ongoing during the 1990s. All areas around Greenland, Iceland and the Faroe Islands are considered to be within the assessment region.

Chapter 2 of the assessment discusses Arctic oil and gas activities within the above mentioned areas, presenting statistical and descriptive information according to the main oil and gas provinces and basins around the Arctic (see Chapter 2, Figure 2.9). Chapter 3 considers socio-economic aspects of oil and gas development, within certain case study areas (see Chapter 3, Figure 3.3).

Chapter 6 of the assessment considers the status and vulnerability of Arctic ecosystems to oil and gas development according to defined Large Marine Ecosystems (LMEs) (see Figure 1.1), and major terrestrial ecosystems.

Oil and gas resource development is still restricted to certain parts of the Arctic, and in that sense oil and gas remains a sub-regional issue of concern. However, the increasing interest in Arctic oil and gas resources; exploration in new Arctic areas; plans for new pipeline routes in the Arctic; the potential use of Arctic seas for shipping oil and gas; and, not least, the potential impacts of oil and gas related pollution on vulnerable Arctic ecosystems all mean that a circumpolar perspective to Arctic oil and gas development is emerging.

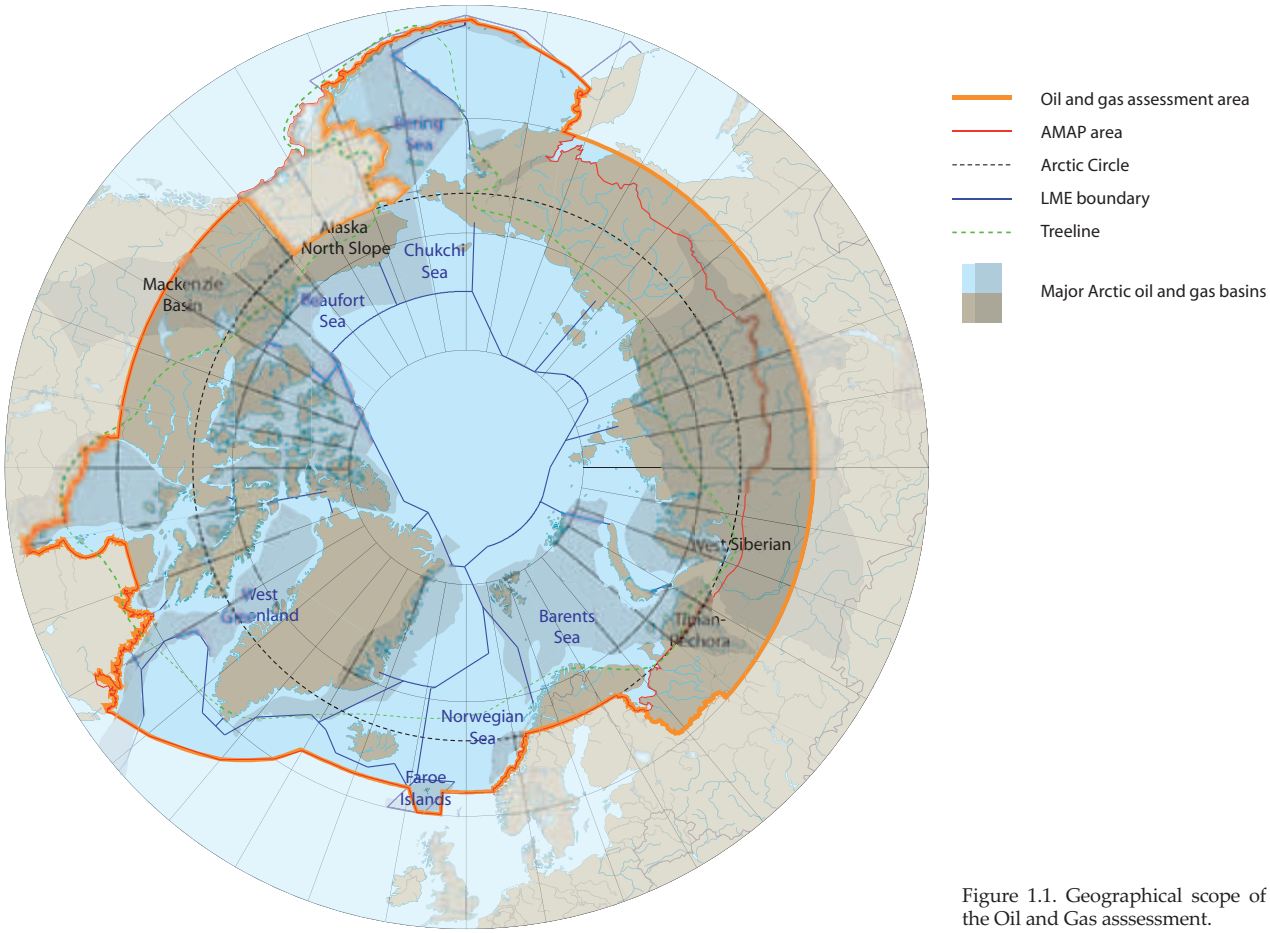

Figure 1.1. Geographical scope of the Oil and Gas asssessment.

1.2.3. Assessment process

For each of the key science chapters (Chapters 2 to 6), one or more countries undertook a 'lead' role, which included the nomination of one or more 'lead authors' for the chapter. The lead country responsibilities were assigned according to Table 1.1.

In order to produce this assessment of oil and gas activities in the Arctic, experts in the various disciplines relevant to each chapter were nominated as lead authors and national experts by the eight Arctic countries. The lead authors, in turn, solicited contributors from the non-Arctic community. An Assessment Steering Group (with membership including, among others, one or more representatives from each participating Arctic Council working group, and the Lead Authors of the chapters) was responsible for the completion of the assessment, reporting directly to the AMAP Working Group and indirectly to all other participating Arctic Council working groups.

The product of this assessment is a fully-validated scientific and technical assessment report (this report) that provides the accessible and fully-referenced basis for statements made in a related overview report. The overview report *Arctic Oil and Gas 2007* (AMAP, 2008) was produced for a wider audience, presenting a concise summary of the results of the assessment, including recommendations addressed to Ministers and decision-makers. The content of the scientific report is the responsibility of the Assessment Steering Group and the lead authors and experts that have been involved in its production (see Acknowledgements). The overview report is prepared under the responsibility of the Arctic Council Working Groups that have been charged with the delivery of the Assessment of Oil and Gas Activities in the Arctic.

This assessment has been subjected to both peer and national review to ensure that it conforms to the highest possible scientific and technical standards with respect to the quality of the material presented.

1.2.4. Readers guide

This assessment is published in three volumes. *Volume I* includes Chapters 2 and 3, providing much of the background that sets the scene for the assessments in other chapters. *Volume II* includes Chapters 4 and 5, the assessments of 'contamination' resulting from oil and gas activities in the Arctic, and the effects of exposure of the environment, biota and humans to this contamination. *Volume III* presents Chapter 6, the assessment of the status and vulnerability of Arctic ecosystems to oil and gas development in the region. Chapters 1 (Introduction) and 7 (Scientific Findings and Recommendations) of the assessment are included in each volume.

A more detailed description of the content and relationship between the different chapters of the assessment is as follows:

Chapter 1, this chapter, sets the stage for the assessment, describes its scope and the processes by which it was accomplished.

Chapter 2 presents statistical and descriptive information characterizing past and current Arctic oil and gas activities, and activities that are likely to occur over the period to 2015 to 2020. These data provide context for assessing effects related to historic activities and provide

Table 1.1. Lead countries for the assessment.

Chapter	Lead
Chapter 2: Oil and Gas Activities in the Arctic	USA and Russia
Chapter 3: Social and Economic Effects of Oil and Gas Activities in the Arctic	USA
Chapter 4: Sources, Inputs and Concentrations of Petroleum Hydrocarbons, Polycyclic Aromatic Hydrocarbons, and Other Contaminants Related to Oil and Gas Activities in the Arctic	Norway and Russia
Chapter 5: Effects of Oil and Gas Activity on the Environment and Human Health	Canada
Chapter 6: Status and Vulnerability of Arctic Ecosystems	Norway

a basis from which to project future levels of activity and effects. In this assessment, the use of the word 'activities' is taken to mean leasing/licensing, seismic and drilling exploration, production drilling and development construction, continuing production operations, all facets of transportation, and eventual decommissioning of facilities. Chapter 2 also presents sections on resource economic drivers for activities, past practices and current best practices and technology, physical impacts and disturbance, and sections on noise from oil and gas activities, oil spill preparedness and response in the Arctic, and monitoring and research programs in each country.

Although Chapter 2 does not include any 'assessment' of the regulatory framework for managing Arctic oil and gas development, this issue is of critical importance for sustainable and environmentally sound development of Arctic oil and gas resources. It was therefore decided that an overview of the existing legal-regulatory systems in the different countries should be prepared for inclusion in this assessment. A reasonably comprehensive referenced review of the main laws and legislation and the implementing regulations, agreements, and procedures for governing oil and gas activities (including, for each country and internationally, preparedness, prevention and response issues and Occupational Health/Safety Regulations) is therefore compiled as an Appendix to the assessment.

Chapter 3 considers the socio-economic strand to the assessment, including the social and economic consequences of the oil and gas activities in the Arctic that are described in Chapter 2. It evaluates historical data and also projects forward as far as possible. It also includes a consideration of the social and economic consequences of environmental effects of pollution and physical impacts and disturbances as examined in Chapters 5 and 6. The intent of Chapter 3 is to provide a comprehensive and balanced view of the positive and negative socio-economic consequences associated with oil and gas development in the Arctic. Chapter 3 includes a series of case studies and mini-case studies that are intended to illustrate diverse situations that exist in the Arctic countries, reflecting different stages in the life cycle of oil and gas activities, differences in political and economic systems, and differences in types of development. Several of these case studies focus on the impacts of oil and gas activities on indigenous population groups in the different countries.

Chapter 4 addresses the pollution strand, identifying sources of contaminant input, environmental concentrations, and contaminant pathways and fates. The information in Chapter 4 builds on information presented in Chapter 2 concerning the petroleum industry, together with available information on other contaminant sources. Chapter 4 also includes a first attempt to quantify a petroleum hydrocarbon budget for the Arctic.

Chapter 5 continues the pollution strand, considering biological effects at the organism level. The chapter comprises two main sections, concerned with effects on terrestrial and aquatic biota, respectively. A third main part of Chapter 5 addresses human health issues, updating and expanding where relevant the information presented in the AMAP Assessments on Human Health (AMAP, 1997, 1998c, 2003). The consideration of human health in this assessment is limited to assessing implications of exposure for health of general populations; occupational health associated with the oil and gas industry is not addressed, although information from occupational exposure is used where relevant to gain possible insight into effects on health of the general population.

Chapter 6 considers vulnerability to, and environmental impacts of oil and gas activities at the levels of species, populations, habitats and ecosystems. The chapter provides brief descriptions of Arctic regional terrestrial and freshwater ecosystems and Large Marine Ecosystems (LMEs) in relation to potential impacts from oil and gas activities. It gives examples of environmental impact assessment and oil spill risk assessment procedures used in several Arctic countries prior to permitting exploration or development. The chapter then assesses the vulnerability of species and populations of plants and animals and of habitats to oil and gas activities, ultimately providing an assessment of vulnerable sites and areas in terrestrial, freshwater, and marine ecosystems. In general, although based on an ecosystem approach, the discussion in Chapter 6 is limited to the direct effects of oil and gas activities, and does not consider potential indirect effects that oil and gas activities may have on other activities in the Arctic, such as commercial fishing or traditional hunting in more localized areas.

Information on certain themes is split between several chapters, to reflect the logical context for presentation of information, for example, the strand on physical impacts and disturbances starts with information on the physical activities (construction work, land use, pipelines, roads, noise etc.) responsible for these impacts/disturbances, presented in Chapter 2, and then goes on to consider their biological effects on organisms in Chapter 5. Consequences for species, populations, habitats and ecosystems are then examined in Chapter 6. Some topics are therefore covered from different perspectives in different chapters, however, section headings and cross-referencing between sections should provide a clear indication of where information on related strands can be found in the respective chapters.

Chapter 7, brings the various strands together to provide an 'overall assessment' of the information presented in Chapters 2 to 6, including a series of conclusions and recommendations based on the science as presented in the assessment. These recommendations will be further considered by the Arctic Council Working Groups, prior to their submission to the Arctic Council Ministers for their consideration in developing a response to the assessment.

References

ACIA, 2004. Impacts of a Warming Arctic: Arctic Climate Impact Assessment. Cambridge University Press.

ACIA, 2005. Arctic Climate Impact Assessment. Cambridge University Press.

AMAP, 1997. Arctic Pollution Issues: A State of the Arctic Environment Report. Arctic Monitoring and Assessment Programme, Oslo, Norway.

AMAP, 1998a. Petroleum hydrocarbons. Chapter 10. In: AMAP Assessment Report: Arctic Pollution Issues. Arctic Monitoring and Assessment Programme, Oslo, Norway.

AMAP, 1998b. Persistent organic pollutants (POPs). Chapter 6. In: AMAP Assessment Report: Arctic Pollution Issues. Arctic Monitoring and Assessment Programme, Oslo, Norway.

AMAP, 1998c. Pollution and human health. Chapter 12. In: AMAP Assessment Report: Arctic Pollution Issues. Arctic Monitoring and Assessment Programme, Oslo, Norway.

AMAP, 2003. AMAP Assessment 2002: Human Health in the Arctic. Arctic Monitoring and Assessment Programme, Oslo, Norway.

AMAP, 2004. AMAP Assessment 2002: Persistent Organic Pollutants in the Arctic. Arctic Monitoring and Assessment Programme, Oslo, Norway.

AMAP, 2008. Arctic Oil and Gas 2007. Arctic Monitoring and Assessment Programme, Oslo, Norway.

Arctic Council, 2004. Senior Arctic Officials Report and Reykjavik Declaration on the occasion of the Fifth Ministerial Meeting of the Arctic Council , 24 November 2004, Reykjavik, Iceland

EPPR, 2002. Circumpolar Map of Resources at Risk from Oil Spills in the Arctic. Emergency Prevention, Preparedness and Response (EPPR) working group of the Arctic Council.

EPPR, 2008. Arctic Guide for Emergency Prevention, Preparedness and Response. Emergency Prevention, Preparedness and Response (EPPR) working group of the Arctic Council.

Owens, E.H. and G.A. Sergy, 2004. The Arctic SCAT Manual: A Field Guide to the Documentation of Oiled Shorelines in Arctic Environments. Environment Canada.

PAME, 1997. Arctic Offshore Oil and Gas Guidelines. Protection of the Arctic Marine Environment (PAME) working group of the Arctic Council.

PAME, 2002. Arctic Offshore Oil and Gas Guidelines. Protection of the Arctic Marine Environment (PAME) working group of the Arctic Council.

PAME, 2004a. Arctic Marine Strategic Plan. Protection of the Arctic Marine Environment (PAME) working group of the Arctic Council.

PAME, 2004b. Guidelines for Transfer of Refined Oil and Oil Products in Arctic Waters (TROOP). Protection of the Arctic Marine Environment (PAME) working group of the Arctic Council.

PAME, 2009. Arctic Offshore Oil and Gas Guidelines. Protection of the Arctic Marine Environment (PAME) working group of the Arctic Council.

Chapter 4

Sources, Inputs and Concentrations of Petroleum Hydrocarbons, Polycyclic Aromatic Hydrocarbons, and other Contaminants Related to Oil and Gas Activities in the Arctic

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4.1. Introduction

This chapter describes the different types and categories of petroleum hydrocarbons and polycyclic aromatic hydrocarbons (PAHs) in the Arctic environment and reviews available information on sources and inputs of these and other chemical contaminants associated with oil and gas activities. Daily operations of the oil and gas industry in the Arctic result in inputs of petroleum hydrocarbons and wastes containing many organic and inorganic contaminants. Although some of these wastes can be problematic on local scales, the emphasis of this chapter is on petroleum hydrocarbons and PAHs because these move through food chains and can spread over wide areas as the result of oil spills. There are also inputs of petroleum hydrocarbons and PAHs from the many small oil spills and accidents, as well as from the general use of oil and gas in the Arctic. This chapter describes these different anthropogenic sources against the background of natural sources of these contaminants.

As part of this assessment, a hydrocarbon budget for the Arctic has been constructed in an attempt to gain a better understanding of the relative importance of different sources of inputs to the total petroleum hydrocarbon load in the Arctic. The environmental fate of oil spills is described, together with the processes leading to the dispersion, removal or burial of petroleum hydrocarbons and PAHs in the Arctic environment. New data on petroleum hydrocarbons in the Arctic that have become available since the previous AMAP assessment on petroleum hydrocarbons (AMAP, 1998) are also presented. The main results of the assessment are drawn together in a series of conclusions at the end of the chapter.

4.2. General properties of petroleum hydrocarbons, PAHs and chemicals used by the oil and gas industry

4.2.1. Crude oil and refined products

Crude oils are complex mixtures of thousands of different chemical compounds. Although hydrocarbons are the most abundant compounds in crude oil, compounds including nitrogen, sulphur and oxygen heterocycles and trace metals are also present in minor and highly variable amounts. Consequently, the chemical and physical properties of different crude oils can be very variable, which in turn influences their environmental fate and effects. Some of

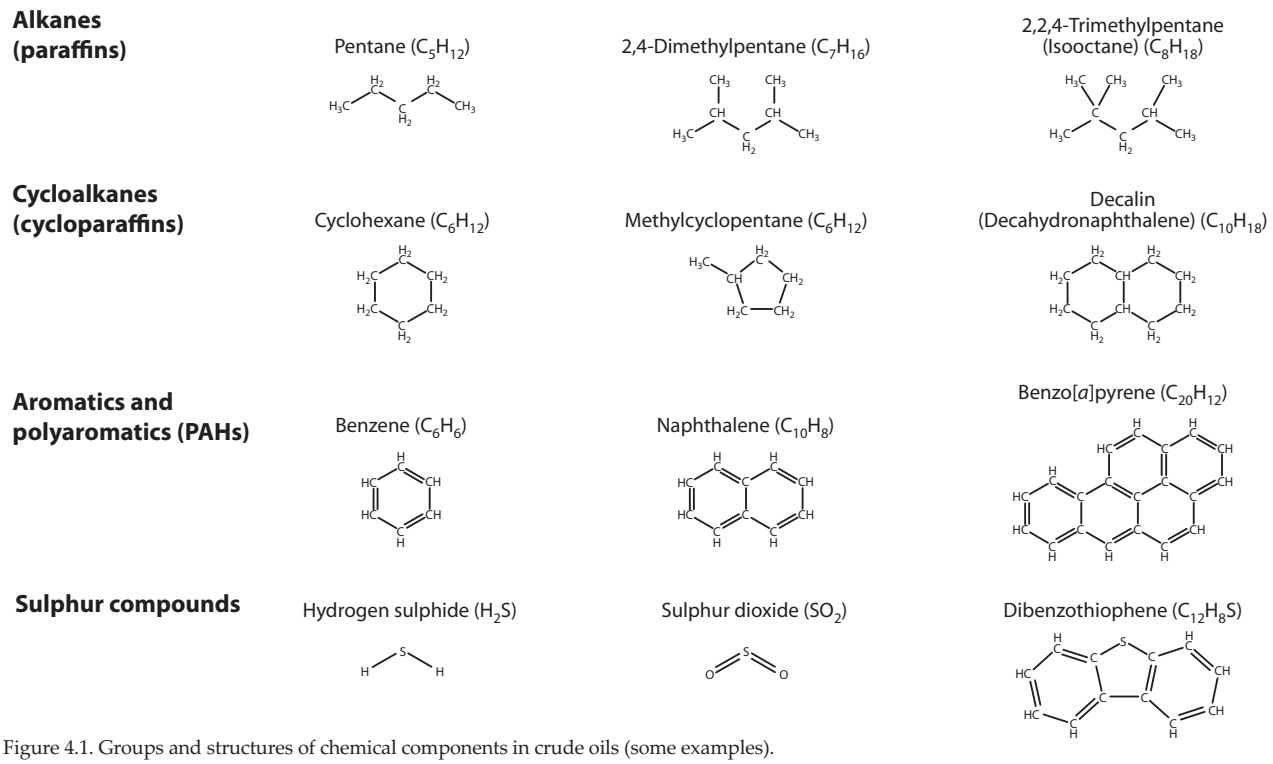


Figure 4.1. Groups and structures of chemical components in crude oils (some examples).

Table 4.1. Properties of some important types of crude oil (Moldestad and Lewis, 2006).

	Composition	Density, kg/m³	Viscosity, cP at 40 °C	Pour point, °C	Wax content, wt %	Evaporation
Paraffinic	Rich in lighter n-alkanes	0.80–0.85	<11	ca. −10 to 6	<5	Medium to high
Waxy	Rich in heavy paraffins (>C_{30})	0.85–0.90	approx. 10 to 100	ca. 6 to 30	>5	medium
Naphthenic	Lacking n-alkanes, rich in naphthenes	0.86–0.95	approx 12 to 1100	<−10	<5	low to medium
Asphalthenic	Rich in asphalthenes (>0.4 wt %), resins	0.90–0.95	approx. 100 to 10000	<−10	<5	low

the main groups of compounds in crude oils are shown in Figure 4.1.

Categorisation of crude oils into paraffinic, waxy, naphthenic, and asphalthenic types is commonly used to compare different crude oils and their weathering behaviour. Some properties of these different types of crude oil are given in Table 4.1 (with examples of chemical structures of some compounds shown in Figure 4.1).

Crude oils are converted into a number of refined oil products at oil refineries. The main products obtained by distillation at atmospheric pressure are distillate fuels, such as gasoline (petrol), kerosene (jet fuel), light gas oil (diesel fuel for cars and trucks), and heavy gas oil (fuel for some marine diesel engines in ships). The terms 'atmospheric residue' or 'long residue' are used to describe the residue from the crude oil that does not boil below 350 °C. 'Heavier' crude oils containing a high proportion of atmospheric residue often go through additional processing. This usually involves distillation under reduced pressure which allows the less volatile oil components to boil at lower temperatures. Vacuum distillates include products such as vacuum gas oil and some grades of lubricating oils.

The main products of oil refining are fuels. Refined products can be divided into four broad categories: distillate fuels, residual fuel oils, intermediate fuel oils, and other refined oil products.

- Distillate fuels: 'light distillates' such as naphtha and gasoline (petrol), 'middle distillates' such as kerosene (jet fuel), and diesel fuels and heavier gas oils.
- Residual fuel oils: the residues remaining after the distillation of the lighter fuels and residues from other conversion processes.
- Intermediate fuel oils: grades of residual marine bunker fuel oil that are produced by blending some of the refinery distillates, or distillate fuels, into the residual fuel oil.
- Other refined oil products: solvents, petroleum coke, lubricants, bitumen and wax.

4.2.2. Chemical composition of crude oil

Petroleum is a mixture of a wide range of hydrocarbons such as alkanes, cycloalkanes, mono-, di- and polycyclic aromatic compounds and other aromatic compounds with molecular masses ranging from 16 for methane to around 10 000 for complex organic molecules. Some of the organic compounds in oil also contain nitrogen, oxygen, and sulphur as well as metals such as vanadium, nickel, iron, aluminium, sodium, calcium, copper and uranium. Examples of oxygen compounds in petroleum include phenols, carboxylic acids, ketones, esters and ethers. The full range of individual chemical compounds in crude oil is not routinely analysed, in part owing to the vast number of isomers of hydrocarbons present. The chemical composition of crude oils from different producing regions and even from within a particular formation can vary widely (NAS, 1985). There can also be large variation in the physical and chemical properties of different crude oils.

4.2.2.1. Hydrocarbons

Hydrocarbons are a class of chemical compounds consisting of the elements carbon and hydrogen. Owing to the ability of carbon atoms to form chemical bonds among themselves and with other elements, an almost limitless number of different structures is possible. Characteristics of the sources of petroleum determine, to a certain extent, the number and type of related molecular structures that are found in a particular crude oil. Fossil hydrocarbons, found in nature as gas, oil, coal, or bituminous shales, span a very wide range of molecular weights and structure types including carbon chains of practically any length, branched chains, cycloparaffins and cycloaromatics in various combinations. In contrast to the multitude of individual compounds in fossil hydrocarbon mixtures, the abundance of hydrocarbon compounds in recent biogenic hydrocarbons is more limited owing to more specific pathways for their biosynthesis, or conversion from food chain precursors.

Alkanes commonly found in oil are saturated hydrocarbons with a general molecular formula of C_nH_{2n+2}. At room temperature, normal alkanes from C_5 to C_{16} are liquid and their boiling points increase with molecular weight. Heptadecane, the C_{17}-alkane, is the first solid alkane at room temperature (melting point 22 °C). Paraffin wax, which is produced from the light fractions of crude oil, contains C_{18} to C_{35} alkanes. Gas oil fractions of highly paraffinic oil may contain, depending on the origin of the oil, up to 50% of normal paraffins. Naphthenic and asphaltic crude oils can contain less than 1% of paraffins. Branched paraffins are also present in petroleum.

Isomers of cyclohexane, cyclopentane and decahydronaphthalene are thermodynamically stable and are commonly found in crude oil. Cycloparaffins, or cycloalkanes, also called naphthenes are saturated hydrocarbons arranged in ring structures, with the general formula C_nH_{2n}.

Alkenes, or olefins, are unsaturated hydrocarbons with the formula C_nH_{2n}. Usually the amount of alkenes in oil is negligible. However, a few samples of crude oil have been shown to contain olefins (Speight, 1999). Similarly, acetylenes, the hydrocarbons containing a triple bond, are, like olefins, only rarely found in crude oil due to the high reactivity of their unsaturated bonds.

All crude oils contain aromatic compounds in relatively large proportions. Benzene and its alkylated homologues, such as toluene and *p*-, *m*-, and *o*-xylenes, are commonly

found. Aromatic hydrocarbons containing a methyl group in the aromatic ring have higher viscosity than other alkylated aromatic hydrocarbons of similar molecular weight (Sanin, 1976). Such aromatic hydrocarbons also have higher density and refractive index, smaller molecular volumes, and higher heats of evaporation. Methylated alkylbenzenes constitute a considerable proportion of middle- and high-boiling oil fractions. For certain oils, they comprise up to 75% of all aromatics present in these fractions (Sanin, 1976).

Aromatic hydrocarbons are slightly more water-soluble than the saturated aliphatic compounds. Benzene, with an aqueous solubility of 1780 ppm is the most water-soluble aromatic hydrocarbon. By comparison, propane, the most water-soluble normal alkane, has an aqueous solubility of 62 ppm; and cyclopentane, the most water-soluble cycloalkane, has a solubility of 156 ppm (Connell and Miller, 1981).

4.2.2.2. PAHs

Polycyclic aromatic hydrocarbons, also known as polyarenes, or polynuclear aromatic hydrocarbons, consist of a variable number of fused aromatic rings often arranged in planar structures. By definition PAHs contain at least three fused rings, although in practice hydrocarbon compounds with two fused rings, such as naphthalene and its alkylated derivatives, are often considered to be PAHs. PAHs exhibit extraordinary structural diversity. The most common types of PAHs are the regular *cata*-condensed and *peri*-condensed alternant hydrocarbons that contain only fused benzoid rings, such as anthracene and pyrene. Other types of PAHs include chain-linked polyarenes (biphenyl type) and cyclophanes, containing fused aromatic rings with sandwich-like structures that are linked together by bridging groups. The term 'parent PAH' refers to those PAH compounds without alkyl substitution.

The general physical characteristics of PAHs are high melting and boiling points, low vapour pressures and low water solubility, which tend to decrease with increasing molecular mass. The compounds are highly lipophilic and can be taken up by biota. Viscosity of PAHs depends on their structure, being lower for methylated compounds than for their analogues containing no methyl groups.

Petroleum is not the only source of PAHs in the environment. They are also formed during incomplete combustion from both natural and anthropogenic sources. PAHs from combustion sources comprise mainly parent compounds. By comparison, petroleum and refined petroleum products contain parent PAHs and a wide range of alkylated PAHs favored by diagenetic processes.

As a consequence of differing physical-chemical properties, there are marked differences in the behaviour of PAHs in the aquatic environment between the low molecular weight compounds such as naphthalene (MW 128) and the high molecular weight compounds such as benzo[*ghi*] perylene (MW 276). The low molecular weight compounds are slightly water-soluble and can be (bio)accumulated in organisms from the 'dissolved' phase, whereas the high molecular weight PAHs have very low water solubility, and preferentially partition to organic and inorganic particulates within the water column. PAHs derived from combustion sources are usually deposited on water surfaces adsorbed onto atmospheric particulates, such as soot particles. Most of the PAHs in water are therefore either taken up by biota or transported to the sediments. Depositional areas are generally regarded as sinks for PAHs.

Table 4.2. Components and functions of drilling chemicals (after Patin, 1999).

Group	Components	Functions
Lubricants	Paraffins, naphthalenes and their derivatives, sulphanol, diesel, mineral oils, graphite, derivatives of fatty acids, lanolin, and others	To reduce friction and heat in the drilling zone and along the drill string
Weighting materials	Barite, calcite, ilmenite and others	To control hydrostatic pressure in the borehole
Viscosifiers	Bentonite and other organophilic clays, carbomethylcellulose, oxymethylcellulose, polyacrylates, lignite, high temperature polymers, starch, xanthan gum, guar gum, and others	To reduce the loss of fluids, control viscosity, and stabilize the well bore
Thinners	Sodium tetraphosphates and other polyphosphates, methylated tannin, chromium lignosulphonates, calcium sulphonate, low-molecular-weight acrylates, polyacrylamides, silicon-organic compounds, and others	To control viscosity and dispersion at different stages to prevent flocculation and excessive thickening of technological fluids
Stabilizers	Polyamides, sulphonated lignites, sulphonated phenolic resins, sodium chloride, granular and fibrous materials, and others	To ensure osmotic balance and stabilization of the well bore
Emulsifiers	Alkylated sulphonates, derivatives of fatty acids, ethers, esters, and others	To form and maintain emulsions during drilling and other technological procedures
Electrolyte pH controllers	Potassium chloride, sodium chloride, gypsum, caustic soda, lime, and others	To maintain the pH of drilling fluid, reduce corrosion, and stabilize emulsions
Corrosion inhibitors	Sodium sulphite, zinc carbonate, ammonium bisulphite, zinc chromate, diammonium phosphate, phosphoxit-7, and others	To prevent corrosion and scaling of pipes, drilling equipment and other machinery
Solvents	Isopropanol, isobutanol, butanol, ethylene glycol, diesel oil, esters, ethers, and others	To prepare solutions of agents and technological fluids
Biocides	Sodium hypochlorite, biguanidine salt, quaternary ammonium salts, aliphatic dialdehydes, oxyalkylated phenols, fatty diamines, thiazolines, carbamates, paraformaldehyde, dichlorophenols, and others	To prevent micro-organism development and microbial degrading of drilling fluid components and other agents

4.2.3. Industrial chemicals

Industrial chemicals and speciality chemical products are widely used at all stages of oil and gas production. The main categories are drilling fluids, production chemicals, injection chemicals, pipeline chemicals, and utility chemicals.

4.2.3.1. Drilling fluids

Drilling of wells for exploration or production requires the use of drilling fluids ('drilling muds'). These fluids lubricate the bit, remove drill cuttings, and support and seal the well hole maintaining wellbore stability. They consist mainly of a base fluid and weight materials to which a number of chemicals are added in smaller quantities (Table 4.2). Base fluids of three main types are used, comprising various oils, 'synthetic' compounds, or water. Synthetic-based drilling fluids are those where the base fluid consists of non-water soluble organic compounds and where neither base fluid nor the additives are of petroleum origin. Weight materials are typically heavy minerals such as finely divided solid material (generally barite, $BaSO_4$), while other drilling fluids are based on heavy salt brines such as sodium chloride (NaCl) or formates (salts of formic acid). Seawater is commonly used when drilling the top portion of a hole from seabed to shallow depth, especially in environmentally sensitive areas.

Depending on the base fluid used, the drilling fluids are referred to as oil-based drilling muds, synthetic-based drilling muds or water-based drilling muds. Drilling fluids and drill cuttings contaminated with drilling fluids are either discharged, injected into a well, or sent to special landfill sites for storage or disposal. Oil-based drilling muds and cuttings with a high hydrocarbon content are treated (for example, incinerated) to 'decontaminate' the material prior to landfill. One of the main functions of the drilling fluid is to transport cuttings uphole. On the platform, shakers separate the cuttings from the fluid. As used fluid can have a significant monetary value, drilling fluids are increasingly recycled wherever possible. Chemicals used for drilling normally account for the largest quantities of chemicals used in oil and gas operations. Other materials, including cement used for well casing, chemicals such as dope for the casing and drill pipe. Additives to the drilling mud may contain environmentally undesirable chemicals, including heavy metals. Most heavy metals in the whole mud remain bound to particulates.

Drilling fluids are not generally standardized, but formulated on a case-by-case basis according to the special needs of the rock to be drilled and the type of drilling to be conducted at a given location. However, the base fluids (oil, synthetic compounds, water) and the weighting agents (e.g., barite, ilmenite) normally account for more than 90% of the fluid by weight. The remainder comprises a wide range of chemicals which are added in small quantities to optimize the function of the fluid in the particular well. These additives include scale inhibitors, corrosion inhibitors, surfactants, foam inhibitors and de-emulsifiers. Like the drilling fluids themselves, additives can also be a significant source of heavy metals (e.g., zinc), petroleum hydrocarbons and other chemicals, depending on the concentrations in the additives and the frequency of their use during drilling. In recent years, more environmentally-friendly substitutes have been found for some of these chemicals.

Well drilling results in considerable emissions of nitrogen and sulphur oxides (NO_x and SO_x) to air from equipment, notably gas or diesel engines used to produce energy to power the drilling operations. Well testing and well clean-up operations can also require the burning of oil, gas and often diesel, but emissions from these activities are normally minor compared to emissions from energy production and flaring.

4.2.3.2. Production chemicals

Production chemicals are used to treat the crude oil and to protect equipment. Scale and corrosion inhibitors, defoamers and de-emulsifiers are normally the most heavily used production chemicals. Other production chemicals typically used in smaller amounts include hydrogen sulphide-scavengers, biocides, hydrate inhibitors, wax inhibitors, asphaltene dispersants, water clarifiers, and gas-drying agents.

4.2.3.3. Injection chemicals

Pressure maintenance in an oil reservoir improves oil recovery and productivity and can be achieved through the injection of water or gas. There are five main types of chemicals used in this process:

- water clarifiers (coagulants and flocculants) to remove particles from injected fluids;
- biocides to eliminate microbial activities;
- chemical scavengers to remove oxygen, carbon dioxide and hydrogen sulphide;
- scale inhibitors; and
- chemicals that increase injectivity.

4.2.3.4. Pipeline chemicals

When transporting oil by pipeline, a number of chemicals are added to the oil to improve the rate and efficiency of throughput and to protect the pipeline. After extraction, oil cools from reservoir temperature (50 to 150 °C) to ambient air or seabed temperatures. On cooling, heavy fractions such as wax ($>C_{18}$) form crystals. Severe deposits of wax may plug both pipelines and production facilities. To remove wax from pipelines, a vessel known as a 'pig' is forced through the pipeline to scrape wax off the walls of the pipe. Pigging is a mechanical solution and is often used in combination with a wax inhibitor. When small amounts of water are present in the oil, corrosion inhibitors are added to the oil stream to prevent corrosion damage to the pipe. Drag reducers are also added to the oil stream to increase pipeline capacity. Drag reducers are high molecular weight polymers which lower turbulence and increase laminar flow in the pipe, thereby lowering the resistance of the oil to flow through the pipe.

When transporting gas and water, hydrates can form and reduce pipeline capacity or block the pipe completely. To prevent this, hydrate inhibitors such as methanol or ethylene glycol are used, sometimes in high doses. The use of cationic surfactants at low doses keeps ice grains in the fluid, preventing them from growing in size and settling. Most commonly used hydrate inhibitors are environmentally friendly; however, high dosage use of methanol and ethylene glycol is also commonly used.

4.2.3.5. Utility chemicals

A range of utility chemicals is used in the oil and gas industry. For example, biocides, drying agents and corrosion inhibitors are often applied when pipelines are prepared for use. These chemicals may subsequently be discharged.

During production, tracers may be added to the production stream to detect leaks. These are not normally hazardous but may not (bio)degrade readily in seawater, and so may persist in the environment if discharged. Detergents are used for well washing, rig washing and general cleaning purposes. Scale dissolvers (acids or chelating agents) are used to dissolve and remove deposits of water-insoluble salts.

Traditionally, chemicals used in the oil and gas industry have been chosen based on technical requirements and financial considerations. Since the mid-1990s, there has been an increasing focus on their environmental properties. This has led to research to identify or develop more environmentally friendly chemicals or practices. Biocides, corrosion inhibitors, emulsifiers, and de-emulsifiers used in the past were often toxic and not readily degradable under normal environmental conditions. There are now chemicals available which are biodegradable and that do not bioaccumulate. For some chemicals there is a clear correlation between technical performance and toxicity because the chemicals act in a similar manner on metal surfaces such as those in pipelines and on biological membranes such as gills and skin. Although chemicals discharged offshore can be rapidly diluted to below threshold levels of toxicity, caution is still required with the use of fat-soluble persistent chemicals which can accumulate in the food web and have the potential to cause long-term damage. This has been one of the major considerations driving the development of biodegradable products to replace traditional polymeric and other persistent chemicals used in the oil and gas industry.

4.2.4. Produced water

Produced water is formation water produced along with oil and gas during hydrocarbon extraction. It may be mixed with injection water and condensation water. The composition and concentration of the naturally-occurring chemical substances in produced water are closely coupled to the geological characteristics of the reservoir, and to the length of time that the field has been in production. The amount of produced water increases with time as the oil and gas are extracted. It may still be profitable to produce oil when the production stream consists of as much as 95% water. There is less produced water associated with gas fields than oil fields.

Examples of some typical compounds found in produced water are shown in Table 4.3.

Naturally-occurring components of particular concern include heavy metals, alkylphenols, PAHs and radioactive substances. Carboxylic acids, mainly propionic acid and acetic acid, dominate among the organic acids. Other major components are benzene, toluene, ethylbenzene and xylene (BTEX), alkylphenols and PAHs. Injected water contains petroleum hydrocarbons, mineral salts, organic acids, heavy metals, suspended particulates, and chemicals added during the production process, such as biocides, corrosion inhibitors and scale inhibitors.

Produced water accounts for the largest volumes of fluids discharged to the environment from oil and gas activities. Onshore, the practice has been abandoned by the United States and all produced water is re-injected into wells. In Russia most produced water is treated and discharged into rivers, lakes, and onto land areas, although Russia is progressively reducing discharges in favour of reusing and re-injecting produced water (Gazprom, 2006, 2007; Surgutneftegas, 2006).

Norway discharges some treated produced water into the Norwegian Sea from offshore oil and gas activities. In most countries, produced water is separated from the produced oil or gas at the production facility. In Russia, this practice is becoming more common and replacing the practice of pumping the corrosive mix of water and petroleum through pipelines to distant facilities for separation. After treatment and separation of oil, for example if produced water meets regulatory limits concerning maximum permissible concentrations of oil, it may be discharged directly to the marine or terrestrial environment. The maximum permissible concentration of oil in water for discharge purposes varies from country to country: in the United States it is 29 ppm average and 42 ppm maximum; for Canada it is 40 ppm and 80 ppm respectively, for Norway it is 30 ppm as of 2006 in accordance with the OSPAR Convention; and in Russia it is 0.05 ppm for discharge into fish habitat and 0.3 ppm for discharge into water bodies used for household water and general use. Various cleaning techniques (e.g., hydrocyclones, filtration, EPCON, C-tour) can be applied to reduce the concentrations of hydrocarbons, metals and chemicals in wastes discharged to the environment. The effects of different technologies vary depending on the characteristics of the actual field and produced water composition. Produced water can also be re-injected into the geological formation,

Table 4.3. Main organic compounds, metals and radioactive elements in produced water from all Norwegian oil and gas production fields for 2006. The total volume of produced water discharged was 144 million m^3.

Substance	Average concentration	Total amount
THC [a]	16.9 mg/L	2 441 t
BTEX [b]	11.3 mg/L	1 644 t
NPD [c]	1.1 mg/L	154 t
PAHs	0.4 mg/l	66 t
Organic acids (<C_6)	241 mg/L	34 800 t
Phenols (C_0–C_4)	3.6 mg/L	530 t
Barium	42 mg/l	6 137 t
Cadmium	0.2 µg/L	30 kg
Copper	5 µg/L	730 kg
Iron	9.5 mg/L	1 370 t
Mercury	0.05 µg/L	7 kg
Lead	2.4 µg/L	348 kg
Zinc	63 µg/L	9 t
^{226}Radium	3.1 Bq/L	460 GBq
^{228}Radium	2.7 Bq/L	390 GBq

[a] Total hydrocarbon content as determined by infrared spectroscopy; [b] benzene, toluene, ethylbenzene and xylenes; [c] naphthalene, phenanthrene, dibenzothiophene and their C_1–C_3 alkylated homologs; [d] EPA 16 PAH priority pollutants. Data for non-radioactive components are from Norwegian State Pollution Control Authority. Data for radioactive components are from the Norwegian Radiation Protection Authority.

Table 4.4. Natural radioactivity in produced water (Jonkers et al., 1997).

Radionuclide	Reported range, Bq/L
^{238}Uranium	0.0003 – 0.1
^{226}Radium	0.002 – 1200
^{210}Lead	0.05 – 190
^{232}Thorium	0.0003 – 0.001
^{228}Radium	0.3 – 180
^{224}Radium	0.5 – 40

often increasing reservoir pressure and thus helping to maintain or increase petroleum production from the reservoir. When re-injection is employed, very small or zero amounts of produced water are normally discharged.

The radioactive constituents in produced water appear to be the same from field to field; however their relative abundance can vary (Table 4.4). Activity concentrations can also vary over time in water from a single field. Radium (Ra) is the most common radioactive component in produced water. Typically, ^{226}Ra and ^{228}Ra constitute more than 90% of the total radioactivity, with the concentration of ^{226}Ra in produced water generally between 100 and 1000 times higher than in seawater.

When produced water with high concentrations of sulphate ions is mixed with seawater, radium may co-precipitate with barium sulphate in an insoluble form. This can also happen if injected seawater mixes with reservoir water in the production process to produce scale deposits consisting mainly of sulphate salts of calcium, barium and strontium, together with radium, inside the production equipment. This can restrict production flow. To reduce the problem with scaling, scale preventing or dissolving chemicals may be added to the production stream. Dissolvers, mainly chelating agents such as EDTA, form complexes with barium and other metals. Scale removed mechanically from the equipment is deposited safely in special facilities on land.

Estimated quantities of produced water from Lukoil's operations in Western Siberia and Timan-Pechora are shown in Table 4.5. TNK-BP reported producing slightly under 600 000 m^3 water a year for their mature wells (BP, 2005). Gazprom, the major producer of gas, reported that in 2005, as in previous years, over 70% of waste water was discharged to surface water bodies (Gazprom, 2006, 2007): 74.9% to surface waters, 19.6% to land areas, and 5.5% re-injected into wells. Total quantities of water discharged decreased between 2001 and 2005 (65.9 million m^3 in 2001, 64 million m^3 in 2002, 61.1 million m^3 in 2003, 48.3 million m^3 in 2004, and 49.1 million m^3 in 2005). Waste-water treatment has improved, mainly due to the commissioning of new treatment facilities and to the modernization of existing facilities.

Table 4.5. Produced water (m^3) from Lukoil operations in Western Siberia and Timan-Pechora (Lukoil, 2004).

	1999	2000	2001	2002	2003
Western Siberia	150 447	192 688	208 144	209 774	220 578
Timan-Pechora	-	17 922	25 623	28 684	28 835

4.3. Sources of petroleum hydrocarbons and PAHs

In nature, all biota are built of organic macromolecules, easily degradable to hydrocarbons. As a consequence, fossil fuels (oil, natural gas, coal etc.) consist mainly of a complex mixture of hydrocarbons degraded from larger molecules – the building blocks of living organisms. Great variability in the chemical structure of hydrocarbons results in major differences in physical properties of fossil fuels, which may be gases, volatile light liquids, viscous heavy liquids, near-solids and solids (such as bitumen, tar, coal). A mixture of these, which occurs as deposits in sedimentary rock, is called petroleum.

Petroleum formation is a long geochemical process consisting of several major steps. Biological macromolecules entering sediments from the external environment will undergo gradual degradation, mostly caused by microbial activity (biodegradation) during the early stages of their burial. This step is called diagenesis. The products of bacterial degradation may undergo further condensation, forming geopolymers, which are the basis of the main products of maturation of organic material: coal, humic material, and kerogen. Certain hydrocarbon compounds undergo only slight transformation, resisting most of the maturation processes, and remain in sediments in an almost unaltered state. Such compounds, termed biomarkers, are used to determine the origin of the organic material.

As the pressure and temperature increase with increasing burial, the next step in the geological maturation of organic material begins. This step is called catagenesis. Geopolymers formed at lower temperatures may be broken into smaller molecules during this step; temperatures rise typically by 30 °C per kilometre depth. At higher temperatures, transformed organic material may, depending on its composition, relative abundance, and pressure, become liquid (oil) or gas ('wet gas' or 'dry gas' – the latter mostly methane and carbon dioxide), the remainder tending towards a complete degradation to graphite. The step subject to the highest temperatures or pressures, when dry gas or graphite are formed, is called metagenesis.

Oil and gas, consisting mostly of relatively light hydrocarbons (i.e., paraffins, mono- and polynuclear aromatics) may in some places leak back to the surface (oil seepages) or be brought to the surface by the oil and gas industry. Oil and gas hydrocarbons, re-entering the surface in this way, will mix with other hydrocarbons present at the surface from both anthropogenic and non-anthropogenic sources. Distinguishing hydrocarbons observed in nature by their origin is a complex task. Each oil has a specific 'hydrocarbon fingerprint', which may be easy to observe in cases of significant oil contamination. However, if the contamination is not acute, the origin may be determined through detailed studies of the precise chemical composition of the hydrocarbons. Chemical analyses of alkanes, biomarkers and PAHs may provide the required information. Studies of PAHs may provide particularly detailed information about sources, dividing these into pyrogenic, petrogenic, biogenic and industrial (which can be both pyrogenic and petrogenic) sources. Box 4.1 explains how analysis of specific hydrocarbon compounds can be used to explain different sources.

Box 4.1. Source identification

Petroleum hydrocarbons

Few petroleum hydrocarbons are unique to petroleum, however many have biogenic or diagenic sources. Robertson (1998) referred to some of the diagnostic parameters allowing an evaluation of petroleum hydrocarbon sources, namely the Carbon Preference Index (CPI), the pristane/phytane ratio, and the alkane compositional ratio.

- The CPI is the ratio of odd-numbered carbon chain n-alkanes to even-numbered carbon chain n-alkanes. Biogenic hydrocarbon mixtures contain much higher concentrations of the odd-numbered n-alkanes (CPI>1) than petroleum hydrocarbons, which have a roughly equal distribution of odd- and even-numbered n-alkane chains (CPI≈1).
- Pristane is a C_{19} isoprenoid alkane which, although present in petroleum, is primarily biogenic in origin. Phytane is a C_{20} isoprenoid alkane, commonly found in petroleum and rarely biogenic. In sediments having no petroleum hydrocarbon input, the ratio is usually much greater than 1.0 (typically 3 to 5). When the value of the ratio is close to 1.0, a strong contribution of petroleum hydrocarbons is indicated.
- The ratio of (nC_{10} to nC_{20}) to (nC_{10} to nC_{34}) is low (approximately 0.01 to 0.1) in 'clean' sediments, but approaches 1.0, depending on the type of petroleum input, in sediments contaminated with petroleum.

Polycyclic aromatic hydrocarbons

There are three major types of PAHs, which differ by their genesis: biogenic, petrogenic and pyrogenic PAHs.

- PAHs of biogenic origin are generated by biological processes or by the early stages of diagenesis in marine sediments (e.g., perylene) (Venkatesan, 1988).
- PAHs of petrogenic origin are formed by low to moderate temperature diagenesis of sediment organic matter to fossil fuels. Approximate percentage of fossil PAHs relative to total PAHs can be counted with the Fossil Fuel Pollution Index (FFPI) (Boehm and Farrington, 1984).

$$FFPI = \left[\frac{N_s + P_s - 1/2(P + C_1P) + D_s}{\Sigma PAH}\right] \times 100\%$$

  In this expression, N_s is the sum of naphthalene and its alkyl homologues; P_s is the sum of phenanthrene and its alkyl homologues; P is phenanthrene; C_1P is methyl phenanthrene; D_s is the sum of dibenzothiophene and its alkyl homologues; ΣPAH is the total PAH concentration (i.e., the sum of two- to six-ring PAHs).
- PAHs of pyrogenic origin are formed as a result of both natural processes (volcanism, hydrothermal or catagenetic transformation of organic matter) and anthropogenic processes (combustion of fossil fuels and recent organic material). The proportions of the less stable parent PAH isomers relative to the more stable isomers (PAH ratios) are temperature dependent (Alberty and Reif, 1988). Increasing PAH ratios often indicates combustion and/or anthropogenic input (Yunker and Macdonald, 1995; Budzinski et al., 1997). The best way to distinguish natural and anthropogenic sources is exhibited by ratios of the principal mass 178, 202, 228 and 276 parent PAHs (Yunker et al., 2002).

The identification of PAH sources in mixtures relies on chemical analyses which generate PAH 'fingerprints' based on the concentrations of individual analytes that are diagnostic of the source (Page et al., 1999). Contributions of the possible sources to total PAH concentration measured in a sample can be evaluated using a non-linear estimation of least-squares source-allocation model that finds the linear combination of source PAH 'fingerprints' that best represents the PAH 'fingerprints' in a sample (Burns et al., 1997).

The goal of non-linear estimation is to minimize a loss function, which is defined as the sum of the squared deviation about the predicted values: Σ(observed value − predicted value)^2.

$$\sum_{i=1}^{k}\left[\sum_{j=1}^{n}\left(s_{i,j}x_j/\Sigma PAH_j\right) - d_i/\Sigma PAH_{\text{sample}}\right]^2$$

In this expression, k is the number of PAH compounds analyzed; n is the number of the tested possible PAH sources; $s_{i,j}$ is the concentration of the ith PAH compound in the jth PAH source; x_j is the fraction of the total PAH in the sample that is due to the jth PAH source; ΣPAH_j is the total PAH concentration of the jth PAH source; d_i is the concentration of the ith sample PAH compounds; and ΣPAH sample is the total PAH concentration in the sample. The computational procedure used to minimize the loss function recognizes that $0 < x_j < 1$ and

$$\sum_{j=1}^{k} x_j = 1$$ (Burns et al., 1997).

Table 4.6. Oils seeps reported in the Alaskan Arctic.

Seep	Latitude	Longitude	Year	Notes	Source
Western Arctic (NPRA)					
Simpson	70.916	−154.616	1909	4 seeps, tar impregnated tundra on peninsula; ~ 11 500 acres mined by locals, tar dunes and domes	Schrader, 1904
	70.9575	−155.3519			
Umiat seep	69.367	−152.133		> 2 seeps along Colville River, light oil sheen in ponds and river	Miller et al., 1959
Fish Creek	70.3	−151.933		Small seep in tundra	Miller et al., 1959
Oily Lake	70.3	−151.15		Northeast of Lookout wells	Miller et al., 1959
Skull Cliff	70.9	−156.8		Seep at base of bluff close to wave beach	Magoon and Claypool, 1981
Kokoloik River			1950	Oil-stained sandstone and asphalt, N flank syncline #10	Chapman and Sable, 1960; USGS PP 303
Omicron Hill	69.38	−160.2	1951	Brown stain in low permeability Cretaceous sandstone along Carbon Creek	Whittington and Stevens, 1951; USGS OFR
Etivluk-Kiligwa			1951	Lower Nanushuk group	Whittington and Stevens, 1951
Lower Nanushk River			1947		Webber and Detterman, 1947; USGS OFR
Atquasuk Lake	70.8	−157.75		Bubbles vigorously in lake adjacent to village	Miller et al., 1959
Pahron		−157.6	1947	In lake north of Meade River near headwaters	Weber et al., 1950
Aupuk	69.55	−157.536	1950	2 seeps in lake 1.75 miles from Aupuk Creek. East end of Aupuk anticline; 300 sq ft area of bubbles	Eberlein et al., 1950
				3 miles west of Aukpuk seep along Colville River	
Central Arctic					
Desolation Creek				Oil stain in Cretaceous (rocks) along several drainages possibly Prudhoe Bay oil	Houseknecht, 2005
Sagwon Bluffs	69.3852	−147.7077	1930	Oil-stained sandstone near Sagavanirktok River	Miller et al., 1959
Eastern Arctic 1002					
South of Kavik	69.6531	−146.7206		Oil-stained sandstone Sagavanirktok	Leffingwell, 1919; Fieldwork 1983
Canning River Formation	69.6533	−146.249		Lightly stained sandstone	Leffingwell, 1919; Fieldwork 1983
Canning River Sagavanirktok	69.6536	−146.2429		Lightly stained sandstone	Leffingwell, 1919; Fieldwork 1983
Katakturuk N	69.871	−145.1793		Oil-stained sandstone turbidite	Leffingwell, 1919; Fieldwork 1983
Katakturuk S	69.7152	−145.4333		Oil-stained sandstone turbidite	Leffingwell, 1919; Fieldwork 1983
Angun Point	61.925	−142.3916		Oil-impregnated tundra and asphalt	Leffingwell, 1919; Fieldwork 1983
Manning Point	70.116	−143.5166		Oil seep into water	Leffingwell, 1919; Fieldwork 1983
Miocene on Jago River	69.9178	−143.3776		Oil-stained slst	Leffingwell, 1919; Fieldwork 1983
Jurassic-Cretaceous at Niguanak	69.9138	−143.3916			
Triassic Sadlerochit (group) at mountain front				Dead oil	Division of Geological and Geophysical Surveys of the Alaska State Department of Natural Resources

4.3.1. Natural seeps

4.3.1.1. Alaska, USA

Since 1909, twenty-six oil seeps (Table 4.6) have been reported in the literature for Arctic Alaska (Banet, 2005). Some of these seeps have been verified and sampled, others have disappeared or never been found again.

While many of these seeps have been studied (Table 4.7), there are no estimates of the volumes of oil seeping at these locations. Although no sources of petroleum seepage have been identified in the American part of the Beaufort Sea or in the Chukchi Sea, it is possible that oil seepage could occur in the Beaufort Sea region because shallow natural gas deposits are common and appear to occur adjacent to high-angle faults that may have served as conduits for gas migration. There is, however, some evidence that submarine oil seepage has occurred in the southeast Bering Sea (Becker and Manen, 1988). Other sources of natural oil in Alaskan waters include coastal peat deposits, erosion of coal and oil shale deposits, and terrestrial oil seeps and peat (Becker and Manen, 1988).

4.3.1.2. Canada

There are four known areas of natural hydrocarbon seeps in the Canadian Arctic: (1) along the Mackenzie River – particularly near Norman Wells (oil and gas) which the Dene Tribe called *Le Gohlini* meaning 'where the oil is' (Janicki, 2001). R.G. McConnell of the Geological Survey described oil seeps along the Mackenzie River when he wrote in 1889 "The Devonian rocks are nearly everywhere more or less petrolifierous and over large areas afford promising indications of the presence of oil in workable quantities"; (2) the lower Mackenzie River valley and delta (oil and gas) (Yunker et al., 2002); (3) the nearshore Beaufort Sea (submarine gas) (Thomas, 1979). This area also contains numerous gas emanations from seabed mud volcanoes (Blasco et al., 2006; Walsh, 2006; Paull et al., 2007; Figure 4.2). The gas vents can be from 5 m wide and 10 m deep (Bennett et al., 2005); and (4) along the coast of Baffin Island at Scott Inlet (submarine oil) (Levy, 1978) and

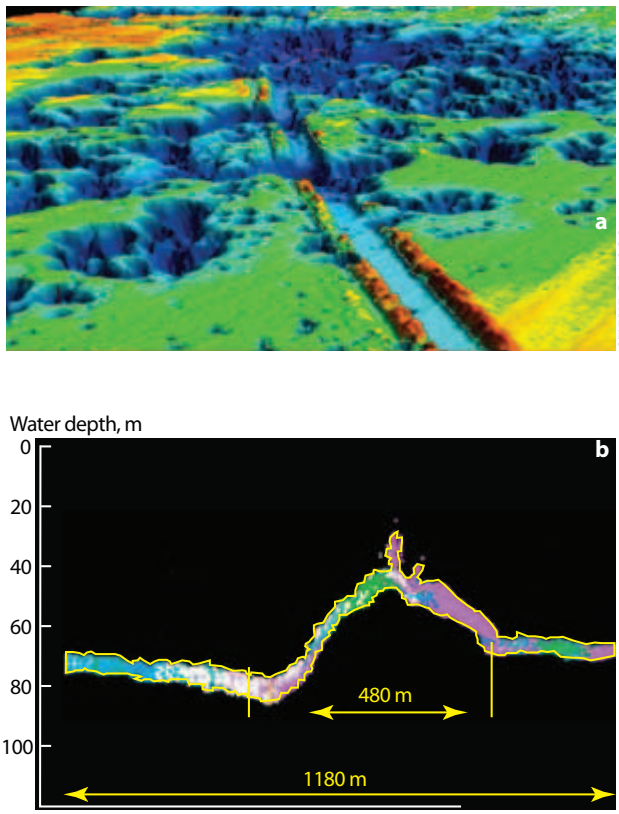


Figure 4.2. Central Canadian Beaufort Shelf, (a) seabed multibeam image of the Kugmallit Bay pockmarks (active methane vents), 8 m water depth; (b) echo-sounder cross-section of the Kopanoar mud volcano with methane venting from the crest, 60 m water depth. Images courtesy of the Geological Survey of Canada.

Table 4.7. Percentage analysis of natural oil seeps in Arctic Alaska (Banet, 2005).

Seep	API Gravity, degrees	Sulphur	^{13}Carbon			Vanadium, ppm	Nickel, ppm	V/(V+Ni), ppm
			Whole, ppt	Aromatic, ppt	Saturated, ppt			
Simpson	18–24	0.22–0.32	-29.9	-28.2	-29	1.2	0.7	0.37
Umiat	30–35	0.04–0.29	-28.2	-26.9	-28.4	0.5	4.5	0.1
Fish Creek	12	0.04						
Skull Cliff	18.4	0.32						
Sagwon Bluffs		0.62		-28.16	-28.98	5.7	11	0.34
South of Kavik		0.59		-27.99	-29.02	<5	<5	
Canning River Can		0.91		-28.55	-29.3	10	9.1	0.52
Canning River Sagavanirktok		1.18		-28.57	-29.52	21	3.3	0.86
Katakturuk N				-28.4	-29.21			
Katakturuk S				-28.78	-29.47			
Angun Point		0.63		-27.77	-28.6	12	9.6	0.56
Manning Point		0.28		-27.16	-28.07	0.8	<0.5	
Miocene on Jago River		0.27		-27.08	-28.12	<0.5	<0.5	0.27
Jurassic at Niguanak		2.07		-28.29	-28.97	0.9	1.3	0.41

API: American Petroleum Institute.

at Buchan Gulf (submarine oil) (Levy and Ehrhardt, 1981). The Scott Inlet seep is about 30 km wide (Bennett et al., 2005).

In addition to oil and gas seeps in the offshore environment, there have been numerous seeps reported on land, for example the seep at Rond Lake near Fort Good Hope, Northwest Territories, where in the 1800s seeped tar was scooped into kegs for trade (Janicki, 2001). No rates of seepage have been measured or estimated for any of these areas. The Smoking Hills, located on Cape Bathurst (Northwest Territories), are the site of burning bituminous shales that have been ignited for centuries along 30 km of sea cliffs. Dense clouds of white smoke consisting of sulphur dioxide, sulphuric acid aerosol, steam and combustion products (including PAHs) are emitted from several fumaroles and carried by prevailing winds across the tundra.

4.3.1.3. Greenland

In the early and mid-1990s geologists discovered significant and fairly extensive oil seeps onshore near the west-central coast in the Disko–Nuussuaq–Svartenhuk Halvø region (Bojesen-Koefoed et al., 1999). Chemical analyses suggest the existence of several petroleum systems in the underlying Cretaceous and Palaeocene marine sediments. Organic geochemical analyses suggest the existence of at least five distinct oil types: (1) a waxy oil which, on the basis of the presence of abundant angiosperm biological markers, is interpreted as having been generated from Palaeocene mudstones; (2) a waxy oil, probably generated from coals and Cretaceous shales; (3) a low to moderately waxy oil containing 28,30-bisnorhopane, and abundant C_{27}-diasteranes and regular steranes, possibly generated from presently unknown Late Cetaceous marine mudstones; (4) a low wax oil of marine, possibly lagoonal/saline lacustrine origin, containing ring-A methylated steranes and a previously unknown series of extended 28-norhopanes; (5) a waxy oil with biological marker characteristics, probably generated from Late Cretaceous mudstones. More recently, age-specific biomarkers from oil seeps show evidence of Jurassic oils in West Greenland (Nation, 2004).

4.3.1.4. Russia

Oil seeps from Arctic Russia were described as early as the sixteenth century on the banks of the Ukhta River in the far northern Timan–Pechora region. Oil seeps and tar sands are common in the Timan–Pechora Basin (Lindquist, 1999).

4.3.2. Oil and gas industry activities and waste discharges

During exploration drilling offshore, accumulation of piles of cuttings on the seabed can lead to a high oxygen demand. The wastes can also contain heavy metals and different types of additives that can be toxic, bioaccumulative and not readily degraded. This may result in local pollution effects. Exploration drilling onshore generates drill cuttings, spent muds and formation fluids that must be stored, cleaned, and disposed of either in waste pits as was the prevailing past practice or in approved disposal sites. Some of these wastes are reused or recycled and some are re-injected in more modern practice. Specific to the production phase are emissions from flaring, venting and production testing, and the potential releases of production chemicals and contaminated production water. In the decommissioning phase, remobilization and spreading of contaminants accumulated during production may constitute a specific threat to the environment. Decommissioning of installations causes minor emissions of exhaust gases and may also create waste problems. When offshore installations, or parts of installations, are decommissioned and removed these operations can result in possible contamination which may negatively affect other activities. Offshore installations left in place may create physical obstructions for shipping and fishing activities, but may also function as an artificial habitat for fish and other marine organisms.

4.3.2.1. Alaska, USA

The *Prudhoe Bay* oil field accounts for most of the North Slope oil and gas reserves and consists of 32 production facilities that have been developed since the first discoveries of oil and gas reserves in 1949. Gas has been produced at only three of these facilities, while oil has been produced at all the rest. Only two oil fields are produced from facilities located in shallow nearshore waters, the rest are located onshore.

Discharges to the environment are regulated by the U.S. Environmental Protection Agency which issued a National Pollutant Discharge Elimination System (NPDES) general permit for effluent discharges associated with oil and gas exploration activities off northern Alaska (USEPA, 2005). This permit allows onshore facilities to discharge certain wastes such as domestic wastewater, grey water, gravel pit and construction dewatering, and storm water (USEPA, 2003). The permit also allows offshore facilities to discharge certain amounts and types of fluids, including non-oil based drilling fluids and drill cuttings, deck drainage, sanitary, domestic, and desalination wastes, muds, cuttings, and test fluids (USEPA, 2006).

4.3.2.1.1. Onshore activities

Exploration activities on the North Slope of Alaska have been mainly conducted using gravel pads. In the past, these pads were designed with surface storage pits to contain drilling muds/cuttings and flare pits. Drilling fluids used in these early wells were composed primarily of clay, and additives used often consisted of Aquagel, tetrasodiumpyrophosphate, calcium chloride, and water. When these wells were drilled, it was common practice to discharge fluids on the tundra and leave drilling fluids in open reserve pits (BLM, 2005). Elevated levels of trace metals in water (zinc and chromium) and sediments (copper, chromium, and lead) have been found in ponds at least as far as 200 m from reserve pits elsewhere on the North Slope (Woodward et al., 1988). That study also found elevated levels of petroleum hydrocarbons in the water and sediments. Waters from the reserve pits and some ponds within 50 m but not at greater distances have been shown to be toxic to a sensitive zooplankton species in bioassays (BLM, 2005).

Records indicate that there are 136 abandoned well sites from these early exploration efforts (Brumbaugh and Porhola, 2004). Government programs began to clean up and decommission some of these sites in the 1970s and 1980s (Schindler, 1983). Studies in the late 1980s and early 1990s on 28 of the well sites for which detailed information was available on the contaminants present showed that none were dangerous but many posed some level of risk to safety, health and the environment (USGS, 1991).

The Bureau of Land Management, U.S. Department of the Interior, reviewed the list of 136 abandoned well sites to determine ownership and well status (Banet, 2006) and found that:

- 39 are uncased core holes that did not penetrate oil and gas zones, have naturally collapsed, and have blended harmlessly into the landscape;
- 33 were either transferred to the North Slope Borough

through the Barrow Gas Field Act of 1984 or were conveyed to the Arctic Slope Regional Corporation and were not evaluated;

- 20 are being used by the U.S. Geological Survey for climatic temperature and permafrost studies;
- 7 are plugged (six at Umiat, Square Lake #1);
- 37 require evaluation in greater detail based on their threat to human health, safety, and the environment if left in their current condition (Brumbaugh and Porhola, 2004). Of these 37 wells, only eight represent any identified potential threat if left in their current condition, mainly due to trash or waste and to a lesser extent low pressure oil or gas (Banet, 2006).

Solid-waste landfills have been associated with virtually all of the drilling sites and Defense Early Warning radar sites (Schindler, 1983). Site investigations have been conducted by the U.S. Department of Defence at all 20 sites located along the Arctic Coastal Plain (USDOD, 1981). Contamination, including petroleum, oils, and lubricants (POLs), polychlorinated biphenols, and pesticides (specifically DDT) were identified at some sites (BLM, 2003). Remedial actions are currently being conducted by the U.S. Army Corps of Engineers to excavate or cap contaminated soils and to remove unsafe structures at these abandoned sites.

Drilling muds and cuttings are no longer stored in surface storage areas on the North Slope. These drilling wastes are reused, recycled, re-injected or transported offsite to an approved disposal facility. Wastes from mud reserve pits are thawed, crushed, slurried and injected through the permafrost and confining layer deep into the earth. Operators are required to document that the injected fluids remain confined within the permitted injection interval and to prevent migration to the overlying groundwater aquifers or to the surface.

In the production phase, wastes generated during Alaskan onshore construction and developmental drilling consist primarily of produced water, drill cuttings and spent muds. These are separated, reused, recycled, treated, or disposed of through onsite injection into an approved disposal well. In addition, domestic wastewater, soil waste, and produced waters generated are injected into a disposal well. Solid wastes, including burnable and recyclable scrap and scrap metal, are transported offsite for disposal at an approved facility.

Even though it is permitted to discharge certain amounts of produced water both to land and to the sea, in practice no produced water is or has ever been routinely discharged to either in the Arctic. All produced water is re-injected into the formations to enhance oil recovery by maintaining formation pressure or to act as water flood (a technique using water to flush sands of oil), everything else is disposed of into deep strata below the permafrost and the water table.

Accidental discharges of produced water have occurred from oil exploration and development activities on the North Slope of Alaska. From 1996 to 2002 there were 240 spills totalling 1 299 664 gallons of 'process water,' which includes both produced water and seawater injected for pressure support (AKDEC, 2003). Most of this was from a single release of 994 000 gallons of process water that occurred on the North Slope on March 17, 1997.

4.3.2.1.2. Offshore activities

For offshore exploration wells, muds and cuttings are generally disposed of by discharge into marine waters deeper than 5 m. For a typical well in the Alaskan Arctic offshore with an average depth of about 2500 m, it is estimated that 425 tons of dry mud will be used of which 80% will be recycled and 20% (85 tons) discharged and 525 tons of dry rock cuttings will be produced and discharged (MMS, 2006). In speculative planning scenarios for possible future activities in outer continental shelf areas in northern Alaska (MMS, 2003, 2006), it is projected that between 6 and 12 exploration and delineation wells could be drilled in the Beaufort Sea in the next 10 to 15 years resulting in possible discharges of 2080 tons of mud and 12 600 tons of cuttings to marine waters and that 7 to 14 exploration wells could be drilled in the Chukchi Sea with 665 to 1330 tons of drilling mud and 4200 to 8400 tons of drill cuttings discharged (MMS, 2006).

Estimates of volatile organic carbon components released into the air during offshore oil production in southeast Alaska vary between 1000 and 8000 t/yr (best estimate, 4000 t/yr) almost exclusively from the mature sub-Arctic Cook Inlet region (NAS, 2003). Unlike regions with a greater degree of urbanization, low-level, chronic releases of petroleum are not significant in Alaska. Erosion of petroleum-bearing rock may play a role at the local scale.

4.3.2.2. Canada

Several petroleum-bearing regions are situated in northern Canada. Development activity in these northern Canadian 'petroleum provinces' (where more than 1500 wells have been drilled) is very much less than the development activity in the western Canada sedimentary basins. Most of the wells were drilled onshore in the mainland Northwest Territories, with fewer drilled in the Mackenzie Delta–Beaufort Sea and Arctic Islands petroleum provinces and in the Yukon Territory. A total of 28 offshore wells have been drilled on the Labrador Shelf. In the Mackenzie/Beaufort Basin extensive drilling took place from the late 1960s to the beginning of the 1990s (see Chapter 2). Eighty-three of the 240 wells in this petroleum province were drilled offshore in the Beaufort Sea.

Since 1990, 168 wells have been drilled in the Canadian Arctic. Most onshore wells are drilled with water-based muds (in the upper section of the wells) and potassium chloride water-based muds (in the deeper sections of the wells). A mud chilling system has been used on an increasingly frequent basis when drilling through the permafrost and hydrate zones. This technique helps to maintain hole stability and provides for a better cement bond for casing strings run over these sections. Wastes are disposed of in reserve pits (sumps). Oil-based muds are now rarely used, but if they are, the muds and cuttings are transported south for disposal.

In the Canadian Arctic, sumps have been built to accommodate at least 4000 m^3 of drilling wastes. The actual volume of waste is typically about 1.3 m^3 for each metre drilled. The wastes comprise drilling fluids, desilter underflow, desander underflow, shaker overflow, rigwash, waste cement and miscellaneous other solids. The hydrocarbon content of the waste comes from formation hydrocarbons and rigwash, which contains various refined lubricating oils and greases. The average size of a sump is approximately 50 m by 50 m wide by 5 m deep; the sumps are usually constructed adjacent to the drilling rig pads. Prior to abandonment, the sumps are covered with the overburden originally excavated to form the sump. Although the construction of the sumps is designed for 'total containment', it is estimated that at least 25% of sumps have leaked (French, 1980) due to one of three problems: non-containment during drilling, melt-out during summer operations, and restoration problems occurring either

during restoration or in subsequent years. Given that the average hydrocarbon content of sump wastes is estimated to be 100–500 µg/g (Hrudey, 1979; French, 1985), and assuming an average density of sump fluids of 1.2 kg/L, then each sump will contain approximately 480–2400 kg of hydrocarbons. Since 1960, in-ground sumps have been used as the waste disposal option for approximately 900 wells in the Canadian Arctic.

All produced water at all production sites in Canada is re-injected into wells dedicated to this purpose.

4.3.2.3. Greenland, Iceland and the Faroe Islands

Greenland, Iceland and the Faroe Islands have no oil extraction or production and little or no oil-related activity, except for the usual consumption of petroleum products. However, exploration drilling for oil has occurred offshore of southwest Greenland and on the Faroese shelf.

4.3.2.4. Norway

In the Norwegian Sea, 196 exploration and 264 production wells have been drilled to date (January 2006). The following fields are on stream: *Draugen*, *Heidrun*, *Njord*, *Norne*, *Åsgard*, *Mikkel*, *Urd* and *Kristin*. A total of 64 exploration wells have been drilled in the Barents Sea to date (January 2006). The *Snøhvit* gas/condensate field in the Barents Sea has been approved for development. Some minor petroleum exploration projects were also undertaken on the Svalbard Archipelago in the 1960s and 1970s.

Norwegian authorities have set a goal for 'zero discharges' from drilling operations on the shelf area of the Lofoten Islands northwards including the Norwegian part of the Barents Sea. An exception is discharge from the top of a borehole. Discharges of produced water will not be permitted in this environmentally sensitive and important fishery area. Discharges do however take place from the fields in the Norwegian Sea. In Norway, drilling and well

Table 4.8. Accumulated discharges from drilling and well chemicals from oil and gas fields in the Norwegian Arctic, 1993–2004.

	Draugen 1993–2004	*Heidrun* 1994–2004	*Norne* 1996–2004	*Njord* 1996–2004	*Åsgard* 1997–2004	*Kristin* 2003–2004	*Snøhvit* 2004	*Urd* 2004
Drilling fluids (t)								
Water-based	3 754	186 009	2 0 174	14 768	52 320	32 659	1 432	3 192
Oil-based	0	0	0	0	0	0	0	0
Synthetic-based	0	1 022	0	0	0	0	0	0
Drilling and well chemicals (t)	37 961	118 777.6	23 601	7 974	66 428	12 184	155	987
Production chemicals (t)	1 195	2 361	3 888	2.3	4 867	0	0	0
Injection chemicals (t)	914.05	18	2.9	0	0	0	0	0
Pipeline chemicals (t)	76	4	0	0	3 639	0.2	0	0
Gas treatment chemicals (t)	29	120	366	0	783	0	0	0
Utility chemicals (t)	29	338	214	1 674	1 124	105	0.03	5.6
Chemicals added to the export flow (t)	0	0	0	377	220	0	0	0
Chemicals from upstream facilities (t)	0	0	0	0	210	0	0	0
Other (t)	0	0	0	0	0	0	0	0
Discharge of contaminants in chemicals (kg)								
Lead	1 109	6 597	1 067	401	3 017	378	0	1.6
Copper	250	2 517	729	247	1 206	706	0.6	3.9
Chromium	42	417	150	49	174	196	1.9	12.9
Cadmium	13	50	68	21	12	2.8	0.06	0.6
Mercury	13	62	63	23	5	0.3	0	0
Zinc	119	1 314	292	160	596	0	0	0
Discharge of additives in chemicals (kg)								
Lead	479	2 875	19	10	132	0.1	0	0
Copper	140	446	105	437	209	8.1	0	0
Zinc	35	93	7	4	102	0	0	0
Mercury	0	5	0	0	0	0	0	0
Chromium	0	6	0	0	0	0	0	0
Cadmium	0	12	0	0	0	0	0	0
Accidental discharge of oil to sea (t)	681.3	24	22	1.7	64	0	0.3	0
Accidental discharge of chemicals to sea (t)	57.1	613	321	124	565	18	0	0.04

operations are the largest sources of chemical discharges offshore (approx. 80%) followed by chemicals used during production and, to a much smaller extent, chemicals used for pipeline treatment. Tables 4.8 and 4.9 summarise the accumulated discharges from drilling and well chemicals, and discharges of chemicals in produced water from oil and gas fields in the Norwegian Arctic.

Discharges of oil-based drilling mud or drill cuttings contaminated with oil-based mud have been prohibited on the Norwegian Shelf since 1991. If oil-based mud is used in drilling operations, residues and cuttings are either taken onshore for treatment or injected into the ground. Within the region of the Lofoten Islands and in the Barents Sea no discharges of cuttings with synthetic-based mud are permitted.

Water-based mud is the preferred drilling fluid on the Norwegian shelf, mainly due to environmental concerns and compliance with regulations. Used fluids and cuttings with remnants of water-based mud may be discharged on the southern part of the Norwegian Sea shelf, but this is not permitted on the Lofoten shelf or in the Barents Sea. More than 95% of the chemicals discharged on the Norwegian shelf are considered to have low environmental impact.

The amounts of different groups of chemical compounds in produced water discharged from installations in the Norwegian Sea sector are shown in Table 4.9. At present, produced water discharges contain approximately 15 mg/L dispersed oil. It is anticipated that the use of improved cleaning techniques in the future will lead to lower concentrations of dispersed oil (and PAHs) in produced water discharges. In general, the composition and concentration of chemical substances in produced water is closely coupled to the geological characteristics of the reservoir, but also to the production history of the field. Metals can sometimes occur in higher concentrations in produced water than in seawater. Given the strong mixing of the water masses on the shelf, however, this is not likely to result in impacts on the environment or ecosystems, with the possible exception of the immediate vicinity of the discharges.

There is limited knowledge about radioactivity in produced water from Norwegian offshore installations, and regular measurements have been made only recently (Figure 4.3). The components appear to be the same from field to

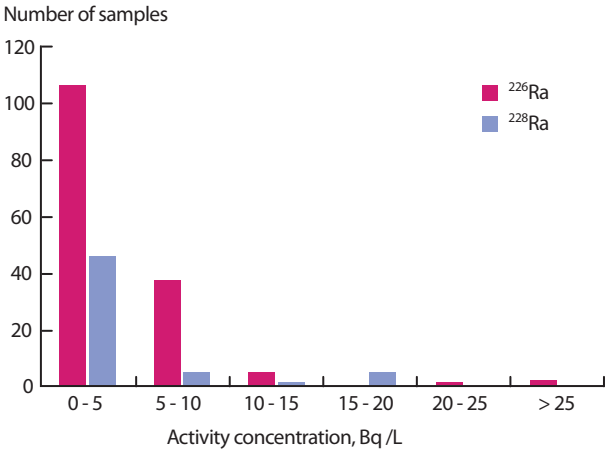


Figure 4.3. Published radium levels in produced water from the Norwegian Shelf, 1995 to 2002 (NRPA, 2005).

Table 4.9. Accumulated discharges of chemical compounds in produced water from oil and gas fields in the Norwegian Arctic, 1993–2004.

	Draugen 1993–2004	*Heidrun* 1994–2004	*Norne* 1996–2004	*Njord* 1996–2004	*Åsgard* 1997–2004	*Kristin* 2003–2004	*Snøhvit* 2004	*Urd* 2004
Oil in produced water (t)	160	473	158	6.8	28	0	0	0
Oil in drainage water (t)	7.5	11	1.17	0	5.3	0.024	0	0
Oil in displacement water (t)	27	0	0	0	0	0	0	0
Sum of oil in water (t)	193	483	159	6.8	33	0.024	0	0
PAH (kg)	879	7 149	4 124	140	1 278	0	0	0
BTEX (BTX until 2001) (kg)	31 300	51 431	150 333	8 937	57 586	0	0	0
Phenols (kg)	1 740	29 200	26 827	4 650	13 489	0	0	0
Alkylphenols (kg)	1 197	11 874	39 913	3 506	8 115	0	0	0
Organic acids (t)	356	2 042	3 220	159	164	0	0	0
Barium discharged with produced water (kg)	52 905	35 803	6 178	31 981	165 502	0	0	0
Cadmium discharged with produced water (kg)	7.9	148.9	5.7	0.8	2	0	0	0
Chromium discharged with produced water (kg)	6.6	62	15	1.1	3.2	0	0	0
Lead discharged with produced water (kg)	10	154	14	2.3	6	0	0	0
Copper discharged with produced water (kg)	7.2	194	20	66	1.9	0	0	0
Mercury discharged with produced water (kg)	0.8	14	1.5	0.4	0.7	0	0	0
Zinc discharged with produced water (kg)	26.4	278	106	62	48	0	0	0
Radionuclides discharged with produced water, ^{226}Radium (million Bq)	23 226	15 505	2 519	1 841	1 073	0	0	0

Table 4.10. Accumulated emissions to air from oil and gas fields in the Norwegian Arctic, 1993–2004.

Emissions, t	*Draugen* 1993–2004	*Heidrun* 1994–2004	*Norne* 1996–2004	*Njord* 1996–2004	*Åsgard* 1997–2004	*Kristin* 2003–2004	*Snøhvit* 2004	***Urd*** 2004
Carbon dioxide	1677970	2882131	2596527	1368946	4957714	61040	1603	9411
Nitrogen oxides	5032	13784	10586	6280	17834	1335	35	206
Sulphur oxides	63	112	145	32	454	8.5	1.1	605
non-methane volatile organic compounds	124479	78648	81986	53278	75258	19	2.5	14
Methane	6849	13495	7708	9460	22250.5	7	0	0

field, but their relative abundance can vary. The activity concentrations may also vary over time in water from a single field. Samples collected in 2003 from all production platforms in the Norwegian sector that discharge produced water, show an average activity concentration of 3.3 Bq/L for ^{226}Ra and 2.8 Bq/L for ^{228}Ra. Almost all samples showed an activity concentration of ^{210}Pb below the detection limit (about 1 Bq/L).

Modelling studies of discharges and the dispersion of produced water indicate that this source could cause a doubling of the activity concentrations of ^{226}Ra in limited areas of the northern North Sea. The major part of the discharged activity is transported towards the Norwegian coast and into the Norwegian Coastal Current where it is transported further northwards into the Arctic (NRPA, 2005).

Accumulated emissions to air from oil and gas fields in the Norwegian Arctic for the period 1993 to 2004 are shown in Table 4.10. Energy production by the offshore petroleum industry is an important source of hydrocarbon-based air pollution. In addition, some emissions from well testing and flaring occur. The most important components are carbon dioxide (CO_2), nitrogen oxides (NO_X), methane (CH_4) and non-methane volatile organic compounds (nmVOCs). Emissions of CO_2 and NO_X are mainly generated by combustion of natural gas in turbines for electricity production and by flaring on the installations. Offloading of oil, particularly at loading buoys on the oil fields, is the main source of nmVOCs. The total offshore oil and gas industry (including activities in the non-Arctic North Sea area) accounts for about 23% of Norway's total NO_X emissions and 28% of its total CO_2 emissions. About 57% of Norway's total nmVOC emissions were generated by oil and gas activities in 2003.

The Norwegian Oil Industry Association recently published a report containing information on the deposition of particle-associated PAHs from combustion processes (mainly flaring) occurring offshore, including all sea areas around Norway (OLF, 2006). Large uncertainties are inherent in these estimates; the best estimate for total PAH emissions from installations on the Norwegian Shelf, including the North Sea, is currently 3.2 to 10.8 t/yr. For comparison, the total discharge of PAHs with produced water (mainly in the North Sea) is estimated at 4.8 t/yr. The total annual emissions of combustion-related PAHs in Norway are estimated at 35 t (sum 6 compounds) or 150 t (sum 18 compounds), most emitted in the North Sea (OLF, 2006).

Removal of offshore installations often requires large-scale marine operations, leading to increased (mostly atmospheric) hydrocarbon emissions. Demolition and recycling also lead to hydrocarbon emissions although these are small compared to emissions during operation. If storage tanks and other parts of the installations are properly cleaned, few discharges to the sea are expected. The volume of waste generated during recycling operations depends largely on the marine growth on the installations. On fields where oil-based or synthetic muds have been used and discharged, disturbance of drill cutting mounds around the installations in connection with removal operations can lead to the spread of drill chemicals. Toppling of substructures and jackets as an alternative to recycling, for instance to create artificial reefs, is likely to cause smaller emissions to the air, but may leak hydrocarbons and other chemicals into the water. Abandoned cables and pipelines, which often contain smaller amounts of heavy metals and other potentially hazardous substances, can also contribute to the contamination of the adjacent seafloor and water.

4.3.2.5. Russia

In the Russian Arctic, much of the oil and gas production is concentrated in the Nenets Autonomous Okrug (NAO), the Yamal-Nenets Autonomous Okrug (YNAO), and in the northern part of the Komi Republic. Hydrocarbon exploration, production, and transportation are among the most important sources of contamination in the NAO. Small abandoned exploration camps, drilling rigs, un-remediated areas of former drilling sites, and traces of caterpillar vehicles in the Bolshezemelskaya tundra are sources of contamination. Accidents including blowouts have also contaminated the NAO (see section 4.4.6.3). At present, there are over 1700 exploratory wells in the NAO, and over 14 000 exploratory wells have been drilled in the West Siberian Basin (north of 60° N) which includes the YNAO.

Oil and gas field facilities include composite structures with production, recharge and intake wells, intra-field and main water, oil and gas pipelines, oil treatment facilities, and gas separators. In the NAO, oil is at present commercially produced at the *Kharyaga*, *Ardalin*, *Varandey*, and *Peschanoozerskoe* fields, and gas at the *Vasilkovskoe* field. Test production in the NAO is presently carried out at the *Musyurshorskoe*, *Toraveiskoe*, *Inzyreiskoe*, *South-Shapkinskoe*, *Cherpayuskoe*, *North-Saremboiskoe*, *Oshkotynskoe*, *Toboiskoe*, *Madseiskoe* fields.

The environment of the YNAO suffers from increasing anthropogenic pressures, the main one being the large-scale extraction of hydrocarbon resources. The development and operation of the largest oil and gas fields, well ties, construction of intra-field and trunk pipelines with infrastructure and social facilities lead to the permanent utilization of lands and result in changes in the natural landscape. In 2004, around 80% of discharges were from stationary pollution

sources. The main polluters were oil and gas industry enterprises: Tyumentransgaz, PO Urengoigazprom, NGDU PO Urengoigazprom and OAO Sibneft-Noyabrskneftegaz and others. Their combined air emissions contributed more than 60% of the total air pollution in the Noyabrsk Region of the YNAO. Crude oil, oil products, phenols and production wastes are discharged into water courses from the drill sites during snowmelt, as a consequence of production storage and transport (Ministry of Natural Resources of the Russian Federation, 2005). Past discharge of untreated and improperly treated waste water from industry in the YNAO amounted to 40 million m^3 in 1999 and 28 million m^3 in 2000.

In the NAO, wastewater discharges into streams and other water bodies amounted to 1.1 million m^3 in 2002. Significant negative environmental effects are associated with industrial and domestic wastes generated in the YNAO, mainly by oil- and gas-related activities. Industries generate about 11 000 t of spent oil products annually, 11% of which are partially used, 19% are decontaminated on-site, 41% are stored at their own storage facilities, and 29% are transported to other facilities for processing. Oil and gas industries contributed 83% of the total industrial wastes generated in the YNAO. Drilling wastes comprise the largest component of industrial wastes (45%), containing about 7% oil and up to 15% organic chemical agents. Sludge pits, where drilling wastes are stored, do not provide safe permanent storage despite measures to seal and protect them with polymeric materials. Pollution occurs as a result of overfilling with waste, damage to containment structures and disturbance of operating conditions (Ministry of Natural Resources of the Russian Federation, 2005).

In Russia, air pollution from oil and gas fields is a consequence of the burning of gas in pits, torches, and flares and the discharge of produced and waste water into surface waters and soil. According to various sources (e.g., Kotkin et al., 1997), between 11 000 and 13 000 t of hydrocarbons are emitted into the air each year in the NAO alone.

A report on flaring from oil and gas activities from satellite imagery indicates that Russian fields flare 50.7 billion m^3 of gas per year – Khanty-Mansiysk at 24.9 billion m^3 and the rest of Russia at 25.8 billion m^3 (Elvidge et al., 2007) (Figure 4.4)

4.3.3. Oil spills

4.3.3.1. Alaska, USA

In the marine environment, inputs of oil from spills from non-tank vessels are individually small, but combined result in spill volumes that are one or two orders of magnitude greater than spills from tankers (NAS, 2003). The *Exxon Valdez* accident (see section 4.4.6.1) constitutes the single largest marine oil spill accident in the vicinity of the Arctic. Other, smaller oil spills have occurred within the Arctic in the past; on both land and at sea, and when not cleaned up the contaminants have persisted, for example in Alaskan soil in some cases for 20 to 30 years after the spill (Margesin and Schinner, 1999).

The number of accidental oil spillages to the tundra in Alaska has increased since the construction of the Trans-Alaska Pipeline System (TAPS) in the 1970s, which was designed and constructed to move oil from the North Slope of Alaska to the southern Alaska ice-free port of Valdez (Panday and Corapcioglu, 1994). The pipeline is approximately 1300 km long, crossing three mountain ranges and 34 rivers and streams including the Yukon, Tanana, and Chena Rivers. After 30 years of operation the average daily throughput has shrunk from a maximum capacity of over 315 000 m^3 of oil in 1988 to 121 000 m^3 of oil in 2006. No catastrophic spills have occurred from the pipeline, not even during the 7.9-magnitude earthquake in November 2002 at Denali Fault (FEMA, 2004).

According to the Alaska Department of Environmental Conservation spill database and older records (BLM, 2003), eight crude oil spills exceeding 80 m^3 associated with onshore or offshore Alaska North Slope oil production occurred between 1985 and 2007. This includes the largest oil spill ever on the North Slope, a 760 m^3 spill from a corroded in-field pipeline in 2006. The largest non-pipeline production facility spill on record is 147 m^3. The average yearly spill rate for 1995 to 2007 is 47 spills a year and the average yearly oil spill volume for this period is 113 m^3 a year, mainly influenced by the large 2006 spill (Table 4.11).

Table 4.11. Crude oil spills on Alaska's North Slope (Alaska Department of Environmental Conservation Spill Database http://www.dec.state.ak.us/spar/perp/search/data.htm).

	Number of spills	Total spill volume, m^3	Average spill volume, m^3
1995	17	0.69	0.04
1996	68	17.48	0.26
1997	61	52.15	0.85
1998	63	21.92	0.35
1999	32	6.42	0.20
2000	57	128.98	2.26
2001	59	109.56	1.86
2002	42	11.23	0.27
2003	54	26.60	0.49
2004	52	9.78	0.19
2005	37	8.71	0.24
2006	42	1063.30	25.32
2007	35	24.71	0.71

Includes crude oil spills from exploration and production facilities, in-field pipelines, and the Trans-Alaska Pipeline System in the Arctic.

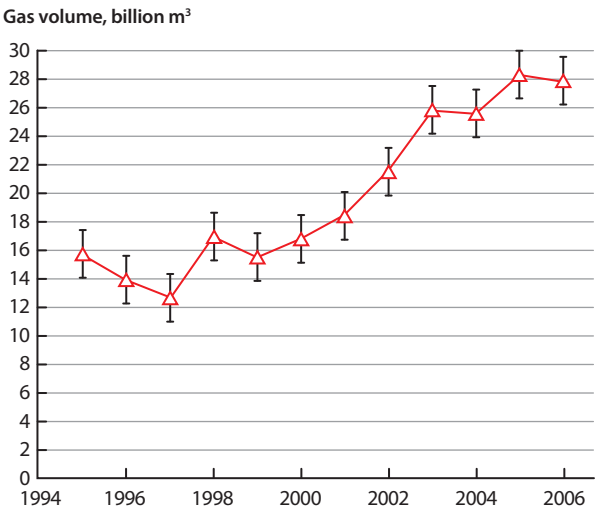


Figure 4.4. Gas flaring in Russia, from satellite imagery (after Elvidge et al., 2007).

Table 4.12. Crude Oil Spills at the Port of Valdez (Alaska Department of Environmental Conservation Spill Database http://www.dec.state.ak.us/spar/perp/search/data.htm).

	Number of spills	Total spill volume, m^3	Average spill volume, m^3
1994	1	31.797	31.797
1995	1	0.004	0.004
1996	6	0.185	0.031
1997	2	0.008	0.004
1998	5	0.019	0.004
1999	3	0.011	0.004
2000	5	0.049	0.010
2001	5	1.185	0.237
2002	4	0.008	0.002
2003	0	0.000	0.000
2004	0	0.000	0.000
2005	0	0.000	0.000
2006	2	<0.001	<0.001
2007	2	<0.001	<0.001

Includes all spills to the marine environment at the port of Valdez from tankers and dockside facilities.

Crude oil spills at the Port of Valdez marine terminal are relatively small, averageing about 0.04 m^3 to the marine environment (Table 4.12) and 0.08 m^3 to land during the period 1995 to 2007.

Eight crude oil spills exceeding 80 m^3 associated with the TAPS including tanker spills in Alaskan waters occurred between 1977 and 2006. Five of these occurred prior to 1982 and were associated with the start-up years of the pipeline. No pipeline spill exceeding 160 m^3 occurred between 1981 and 2000. In an incident of vandalism that occurred outside the Arctic approximately 1080 m^3 of crude oil were released in October, 2001. The 30-year average yearly spill rate for the TAPS, including tanker spills in Alaskan territorial waters is 25 spills a year with an average volume of 36 m^3 of oil released a year (Table 4.13).

The Alaska North Slope rate of small spills (those not exceeding 80 m^3) from facilities, pipelines, and flow lines was approximately 3.9 spills per million m^3 of oil handled. This spill rate consisted of 1.1 small crude oil spills per million m^3 handled and 2.8 small refined product spills per million m^3 handled. Refined oil includes diesel fuel (61% of refined oil spills by frequency and 75% by volume), hydraulic oil (26% by frequency, 10% by volume), engine lubricants (10% by frequency, 3% by volume), aviation fuel, fuel oil, gasoline, grease, transformer oil, and transmission oil. The mean crude oil spill size between 1989 and 2000 was 0.43 m^3; the median was 0.02 m^3. Ninety-nine percent of all spills were less than 0.01 m^3 (Anderson and LaBelle, 2000).

According to the Alaska Department of Environmental Conservation, between 1995 (when the data base was established) and 2005, they received an average of 2301 reports of spills of all types (hazardous substances, process water, non-crude oil, crude oil, other) for all areas of the State, including non-Arctic areas (AKDEC, 2007). Thus, in 2001, there were 234 registered spills on Alaska's North Slope where much of Alaska's oil industry is situated, including three that were large enough for negative effects to be expected on the tundra (Filler and Barnes, 2003).

Table 4.13. Crude oil spills from the Trans-Alaska Pipeline System (Alyeska Pipeline Company, 2007).

	Number of spills	Total spill volume, m^3	Average spill volume, m^3
1977	34	307	9.03
1978	24	2 546	106
1979	43	885	20.6
1980	55	561	10.2
1981	32	240	7.5
1982	30	6.2	0.207
1983	17	0.64	0.038
1984	32	12.4	0.388
1985	31	4.29	0.138
1986	40	6.04	0.151
1987	37	0.64	0.017
1988	35	2.23	0.064
1989	26	40 020[a]	1 540
1990	31	0.95	0.031
1991	54	1.75	0.032
1992	55	3.02	0.055
1993	65	1.27	0.020
1994	44	51.5	1.17
1995	6	0.32	0.053
1996	12	129	10.8
1997	5	0.32	0.064
1998	5	0.08	0.016
1999	8	0.06	0.008
2000	6	0.64	0.107
2001	15	1 090	72.7
2002	9	0.06	0.007
2003	3	0.05	0.017
2004	0	0	0
2005	0	0	0
2006	1	0.06	0.06

Includes all 1300 km of the pipeline, all pipeline facilities, support activities, the marine terminal in Valdez, and tankers while in State waters. [a] The high value in 1989 includes the *Exxon Valdez* tanker spill as it was part of the 'system'.

A significant spill occurred at the *Prudhoe Bay* oil field on March 2, 2006, when a transit line was estimated to have leaked approximately 760 ± 250 m^3 of oil over a 0.79 hectare area due to corrosion. The leak was not discovered by a leak detection system but by a worker who smelled the oil. The source was a 0.75 cm hole in an above ground 1-m diameter crude oil transit pipeline. The hole was discovered in the pipe within a buried culvert (caribou crossing). The clean-up began immediately but was hampered by the cold weather due to short work exposure for clean-up workers, however the clean-up was also aided by the cold because spreading of the viscous oil was restricted. Clean-up operations were completed within two months of the spill and resulted in the recovery of approximately 245 m^3 or roughly 30% of the spilled oil and removal of approximately 10 000 m^3 of oil-contaminated snow and soil, and over 400 m^3 of contaminated gravel (AKDEC, 2006). Soil sampling confirmed that the entire spill site met the clean-up criteria. The site has been backfilled and covered with veggie-mat. Veggie-mat is a 10–15 cm layer

Table 4.14. Summary of diesel fuel and diesel P-50 spills reported in the Canadian Arctic (GNWT Hazardous Materials Spill Database at http://www.e-engine.ca/eps_spillreport/).

	n	Spill size min–max, m^3	Total spill volume, m^3	Average spill volume, m^3	Spills to terrestrial system		Spills to freshwater/ aquatic system		Spills to marine system		Spills associated with oil and gas activities		Volume diesel P-50 spilled	
					m^3	n	m^3	n	m^3	n	m^3	n	m^3	n
1972	6	3–172	428	71	421	5	7	1	0	0	139	2		-
1973	10	u–59	95	9	9	5	61	2	25	3	27	4	8	1
1974	12	u–82	136	11	107	10	29	2	0	0	0.23	1	0.1	1
1975	18	u–68	149	8	65	10	82	7	1.6	1	3.4	2	0.5	3
1976	8	u–318	344	43	343	3	0	0	0.73	3	0.25	2	19	3
1977	10	u–53	150	15	121	5	29	3	u	2	0	0	33	4
1978	277	u–277	322	40	32	3	0.68	1	277	1	0	0	6	2
1979	10	u–23	56	6	55	8	u	1	0.91	1	0.91	1	46	5
1980	52	u–60	114	2	90	10	19	3	4.4	35	6.8	32	48	3
1981	12	u–45	110	9	100	7	2.4	2	7.0	3	8	4	102	8
1982	21	u–75	214	10	210	14	0	0	4.4	5	12	7	37	15
1983	23	u–18	89	4	86	21	0	0	1.1	2	4.4	6	93	13
1984	28	u–25	149	5	147	24	0	0	2.2	2	37	11	205	24
1985	25	u–45	107	4	100	16	6.1	5	0.07	2	2.5	3	460	30
1986	25	u–14	39	2	32	20	0.23	1	7	2	8.4	4	70	16
1987	21	u–32	50	2	57	15	2	2	0.77	2	0.09	1	418	18
1988	23	u–3	12	0.5	11	13	0.91	1	0.21	3	0.69	4	44	14
1989	24	u–30	51	2	51	22	0.12	2	0	0	0.26	3	32	9
1990	26	u–41	118	5	117	16	0.37	3	0.28	2	0.29	3	60	14
1991	32	u–65	147	5	82	29	65	2	0.23	1	0.89	8	3	9
1992	28	u–22	44	2	31	23	13	4	0.41	1	0.51	3	21	23
1993	20	u–10	24	1	24	17	0.39	2	0.045	1	0.045	1	32	17
1994	26	u–40	54	2	53	2.4	0.23	2	0	0	0	0	10	25
1995	23	u–1.3	3.6	0.14	3.6	21	0	0	0	0	0	0	9	15
1996	28	u–3.5	13	0.48	13	23	0.29	2	0	0	0	0	46	18
1997	49	u–10	33	0.68	32	45	1.1	3	0	0	0.04	1	2	10
1998	30	u–35	52	2	51	27	1.4	3	0	0	0	0	9	13
1999	17	1–2.5	8	0.45	6.6	15	0.95	2	0	0	0	0	8	15
2000	68	u–22	46	0.68	46	61	0.84	5	0	0	0.2	1	11	25
2001	62	u–7	18	0.29	18	54	0.29	4	0	0	8.7	7	6	11
2002	96	u–10	22	0.23	22	91	0.23	1	0.16	3	0.67	29	15	10
2003	116	u–63	94	0.82	94	114	0.023	1	0	0	0.24	8	30	24
Total	1226	u–1731	3292	264	2630	749	324	67	334	75	263	148	1884	398

u: Unknown.

of chunks of live, frozen tundra that was taken from a donor site and transplanted to the spill site (AKDEC, 2006).

In December 2004, the M/V *Selendang Ayu* grounded near Unalaska Island between Skan Bay and Spray Cape. The actual amount of fuel spilled is unknown. The total volume of fuel initially on board the vessel was approximately 1690 m^3 of intermediate fuel oil (IFO 380) and 120 m^3 of marine diesel oil. The total estimated amount of all oils released to the environment was 1270 m^3 (1215 m^3 IFO 380 and 55 m^3 marine diesel).

4.3.3.2. Canada

A summary of the accidental discharge of diesel fuel in the Northwest Territories is shown in Table 4.14. During the 1970s and 1980s approximately 3300 m^3 of diesel fuel were reported spilled. Of the total, 80%, 10% and 10% were lost to the terrestrial, freshwater/aquatic and marine environment, respectively. Eight per cent of the total spilled was associated directly with oil and gas activities. It is unclear exactly how much of the fuel spilled was recovered; however, inspection of the data indicates that spill size has steadily decreased with time. Recent data suggest that the magnitude of diesel spills is far lower than in the previous decades. This is due to a greater emphasis today on maintenance and spill prevention. Spills of crude oil (Table 4.15) and diesel P50 (Table 4.14) follow the same pattern. The data indicate that all other hydrocarbon spills (totalling 3500 m^3) have also decreased in size with time.

Two gas blowouts have occurred in the Canadian Arctic. These occurred in 1969 (the Panarctic et al Drake I-67 Well Melville Island) and in 1971 (the Panarctic et al. King Christian Island D-18 Well). In 1976, the Dome et al. Tingmiark K-91 Well in the Canadian Beaufort Sea encountered a high-pressure water flow that came to the surface after the well had been shut down.

Inputs of oil, in all forms, to the Canadian Arctic are very small. The total average annual calculated load to this area, from all sources, during the period 1990 to 1999 was about 2300 t. The dominant anthropogenic source in the area is

Table 4.15. Summary of crude oil spills reported in the Canadian Arctic (GNWT Hazardous Materials Spills Database at http://www.e-engine.ca/eps_spillreport/).

	n	Spill size min–max, m^3	Total volume of spill, m^3	Location			
				Norman Wells, production/ refining	Offshore Beaufort, exploratory drilling	Natural seeps	Other
1973	1	23	23	23	0	0	0
1976	1	4	4	4	0	0	0
1977	1	6	6	6	0	0	0
1978	4	3–40	65	65	0	0	0
1979	3	0.8–1.6	3	3	0	0	0
1980	9	0.17–4.5	19	17	0.36	0	1.2 (mine site)
1981	1	1	1	1	0	0	0
1982	9	0.2–22	33	31	2	0.10 (NormanWells)	0
1983	2	0.023–0.75	0.77	0	0	0.23 (NormanWells	0.75 (MackenzieDelta)
1984	3	1–9.8	23	15	8	0	0
1985	2	1.27– 5.0	6	5	0	1.27 (NormanWells	0
1986	5	0.32–80	88	80	7	0	0.32 (pipeline at Wrigley)
1987	6	0.23–4	16	16	0	0	0
1988	23	0.008–2.3	9.7	5.6	4.1	0	0.064 (Resolute)
1989	11	0.028–3	15	6.9	2.8	0	4.5 (Bent Horn), 0.4 (Mackenzie Delta)
1990	13	0.001–0.6	1.8	1.8	0	0	0
1991	6	0.1–64	108	108	0	0	0
1992	3	0.2–100	101	1	0	0	100 (pipeline at Fort Simpson)
1993	5	0.01–0.2	0.29	0.29	0	0	0
1994	1	4	4	4	0	0	0
1995	2	0.2–0.5	0.7	0.7	0	0	0
1996	1	0.31	0.31	0.31	0	0	0
1997	13	0.04–397	404	404	0	0	0
1998	5	0.02–1	1.7	1.7	0	0	0
1999	3	0.2–5	6.2	6.2	0	0	0
2000	2	0.008–0.2	0.21	0.2	0	0	0.008
2001	1	0.3	0.3	0.3	0	0	0
2002	1	0.2	0.2	0	0	0	0.2
2003	1	37	37	0	0	0	37 (South Slave)
Total	138	0.001–397	978	807	24.26	2.1	144

atmospheric deposition associated with the consumption of petroleum hydrocarbons worldwide. Spills are rare occurrences and are generally associated with tanker traffic in and around Hudson Bay. Land-based sources make a relatively significant contribution to the coastal zone in areas where development has occurred (NAS, 2003). Input of petroleum hydrocarbons to the environment of the Canadian Arctic from oil and gas industrial activities and oil spills constitutes less than 2% of total inputs to the terrestrial environment, about 0.1% of total inputs to the marine environment, or approximately 1% of total inputs to the environment in total. The major sources of petroleum hydrocarbons in the Canadian Arctic are riverine inputs, natural seeps and non-oil and gas-related industrial activities (Thomas and Macdonald, 2005).

4.3.3.3. Greenland

There are no official oil spill statistics in waters surrounding Greenland. Approximately five to ten small oil spills (0.2–0.5 m^3) occur every year (mostly spills of diesel). The largest in the latest ten-year period was 42 m^3 of Arctic Grade fuel oil in 2001. This accident was mainly on land (Lindgren and Lindblom, 2004).

Most spills in Greenland have occurred along the west coast between 62° and 68° N; the most populated area in Greenland. This region is an extremely important centre for fisheries, seabirds and marine mammals. To date, most spills in the region have been small and are related to fuel re-supply to the communities, fishing vessels and other small ships. A small but finite risk to this area has been added with the start of exploratory drilling for crude oil approximately 150 km west of Nuuk in 2000. More exploratory drilling is anticipated

Table 4.16. Total acute oil discharges (m^3) in Norway, 1987–2004.

	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004
Bunker facilities	170	127	106	127	24	218	23	13	12	74	26	154	30	11	13	9	9	84
Industry	128	222	201	98	203	74	66	163	114	392	56	438	181	90	1212	116	90	614
Land transport	31	44	61	27	0	29	26	30	23	27	68	31	53	34	49	23	14	32
Buried fuel tanks	8	66	60	25	17	30	42	76	56	56	48	30	65	60	31	52	43	18
Offshore	267	437	1574	572	336	1017	281	143	184	121	201	216	228	89	89	212	897	94
Ships	192	92	569	475	257	219	227	672	120	91	460	147	223	272	233	141	165	360
Other	0			113	144	135	42	43	57	12	95	13	52	3	21	23	68	30
Total	796	988	2572	1437	981	1723	707	1140	566	773	955	1030	832	560	1648	577	1286	1231

in the offshore area in the coming years. A detailed oil spill sensitivity atlas of Greenland was created recently, which provides information about the areas most vulnerable to oil spills (Mosbech et al., 2000).

Another source of petroleum contamination in Greenland is the Thule Air Force Base. Releases from military planes and other vehicles are a matter of concern, and although no major spills have occurred, elevated concentrations of hydrocarbons such as PAHs have been reported in the Thule area (Glahder et al., 2003).

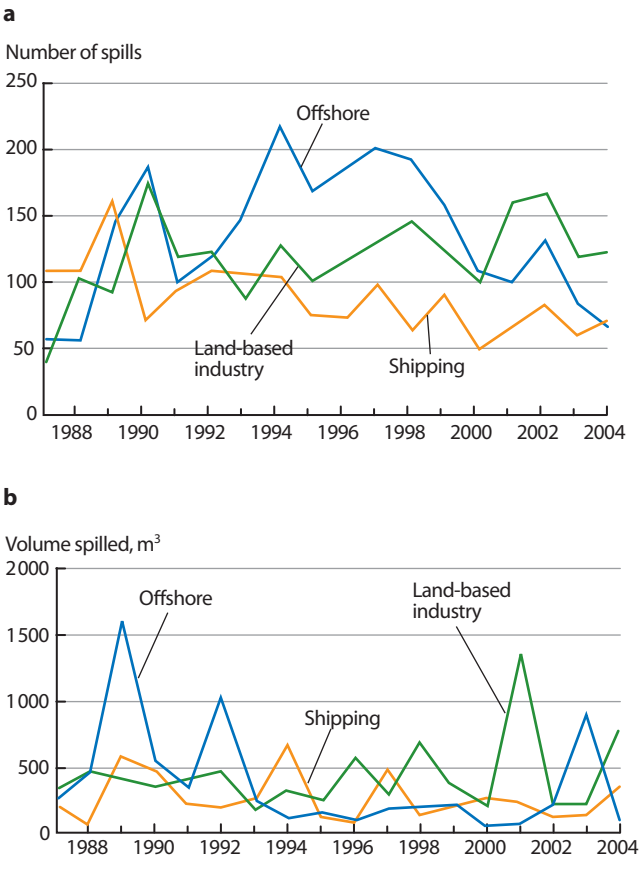


Figure 4.5. Oil spills in the Norwegian environment from land-based industry, offshore activities, and shipping; (a) number of spills, and (b) volume spilled (SFT, 2006).

4.3.3.4. Faroe Islands

The Faroe Islands have no historical statistics of oil spills in the surrounding waters. Prior to 2002, the responsibility for reporting oil spills lay with the Danish defence staff. Since 2002, the Maritime Rescue and Coordination Centre of the Faroe Islands has undertaken oil spill preparedness. Most of the known oil spills have been from diesel fuel. The largest oil spill in the period 2001 to 2003 (27 m^3 of gasoline) was from a foreign fishing ship. An efficient and systematic reporting order is in place (Mosbech et al., 2000).

4.3.3.5. Norway

An overview of oil spills in Norway from 1987 to 2004 is shown in Table 4.16 and Figure 4.5. Most spills are small. The number of accidental oil spills on the Norwegian shelf in 2003 was the highest reported in ten years. There were 131 incidents, one of which (*Draugen* field) accounted for 84% of the total oil spilled in 2003. Of the remaining spills, 74 involved losses of less than 0.05 m^3 of oil. There were also 124 accidental spills of drilling fluids and chemicals, 40 of which exceeded 1 m^3.

4.3.3.6. Russia

According to world statistics, the wreckage rate for tankers of more than 10 000 t is about 2.4%; the wreckage rate for gas carriers is 0.73%. Stranding and collisions were the most common reasons (about 75%) for the incidents (Matishov and Nikitin, 1997). The wreckage rate for supertankers with the potential to spill 120 000 t of oil is very low.

There is a substantial probability of accidental oil spills in areas of oil and gas activities in the Barents Sea and along shipping routes in the Barents and White Seas (Table 4.17). The use of double hulled tankers should result in a decrease in the frequency and volume of emergency oil spills. According to Zhuravel and Mansurov (2005), collisions and stranding of such tankers would lead to spills in fewer than 20% of incidents, with the average size of a spill unlikely to amount to more than 2% of the cargo volume and the maximum size of the spill unlikely to exceed 20% of the cargo volume.

Far less equipment is available for deployment in emergency oil spill situations in the Russian north than is the case in Norway. While Norway has relatively little traffic in ice-covered areas, in most parts of the Russian Barents Sea operations are conducted in ice-covered conditions for about six months a year. There is strong cooperation between oil spill emergency response units from Norway and Russia and Russian oil spill prevention and response authorities are

Table 4.17. Risk estimates of emergency oil spills in the Barents Sea (Zhuravel and Mansurov, 2005).

	Frequency case/year	
	2005	2010
Estimate of goods turnover		
Oil spills, over 7 t	1.05×10^{-2}	3.12×10^{-2}
Oil spills, over 700 t	2.21×10^{-3}	6.55×10^{-3}
Export oil volume estimates including reloading		
Oil spills at sea over 153 m^3	5.87×10^{-2}	1.71×10^{-1}
Oil spills in ports and at terminals over 153 m^3	3.87×10^{-2}	1.13×10^{-1}
Estimates by number of loads in ports and at terminals		
Oil spills in ports and at terminals over 1 t	9.40×10^{-2}	2.28×10^{-1}
Oil spills in ports and at terminals over 10 t	1.88×10^{-2}	4.56×10^{-2}
Oil spills in ports and at terminals over 100 t	3.58×10^{-3}	8.86×10^{-3}

attempting to achieve Norwegian standards for equipment, knowledge and readiness. However, the availability and possibilities for rapid deployment of oil spill response equipment in the event of a spill occurring in the extensive and remote Russian Arctic territories is likely to be limited.

During 2003–2004, a number of accidents were registered at the reloading centres along the Barents and White Seas. Some of these resulted in oil spills. In January 2003, there was an accidental spill into the Dvina River of 1100 t of fuel oil from the coastal storage complex at the military unit in Glukhoye settlement near Arkhangelsk. In September 2003, a ship collision at a mooring location at the Onega offshore reloading terminal resulted in the loss of 54 t of fuel oil into the river. The clean-up operation retrieved only 9 t of fuel oil. In 2004, there was an oil spill of approximately 2000 t of fuel oil in the port of Teriberka. Another spill occurred near the Kolguev Island where nearly 50 t of fuel oil spilled into the sea. Around 200 t of fuel oil remain stranded in the fuel tanks of the dry-cargo ship *Stepan Razin* after it sank at the mouth of Kola Bay. Finally, fuel oil was spilled from the cruiser *Admiral Kuznetsov* at ship-repair factory #35 (Bambulyak and Frantzen, 2005).

Land installations such as oil wells, and oil transportation via pipelines, railways and river barges also represent pollution threats to the Arctic environment. Some oil products enter the Arctic Ocean via the drainage basins of rivers. The previous AMAP assessment (AMAP, 1998) described the chronic spills from pipeline leaks in the Komi Republic in 1994 (see section 4.4.6.2). In 2003, 24 incidents at oil-extracting and processing installations resulted in the contamination of water and land in the Komi Republic. The most extensive accident was a pipeline leak that delivered 800 t of fuel oil into the Ussa River (Bambulyak and Frantzen, 2005). In 2003 and 2004, accidental oil spills occurred at the Musyushor-Sandivey pipeline which polluted the Sandivej and Kolva Rivers. In 2003, approximately 1000 t of oil were discharged into water from 2000 oil spills in Khanty-Mansiysk AR. In 2004, 2800 oil spills were reported in the area (Bambulyak and Frantzen, 2005).

Russia's oil and gas pipeline transport system comprises more than 200 000 km of trunk pipelines, with 48 500 km of these trunk oil pipelines, although there are no trunk oil pipelines in the Arctic area (Zlotnikova et al., 1999; Zhuravel and Mansurov, 2005). Safety of the trunk system, which was low in the 1980s and early 1990s, has recently improved as can be seen in Table 4.18. In 2004–2005 the officially reported rate of spills was at the level of 0.04 per 1000 km. However, there may be some under-reporting of the number of oil spills since 2002 due to established reporting criteria (a pipeline spill of less than 7 t of crude oil is not required to be reported as an emergency situation unless there are human injuries and/or contamination of water bodies). At the same time there are around 135 000 km of oil field and collection pipelines, for which safety remains very questionable. According to different sources, the annual number of oil leaks and spills from these pipelines is of the order of several tens of thousands. In 1999, it was reported that wear and tear of field pipelines was in some places up to 80%, and that their failure rate was two orders of magnitude greater than for trunk pipelines at around 1.5 to 2 breaks per kilometre per year (Zlotnikova et al., 1999).

4.4. Environmental transport and fate of petroleum hydrocarbons and PAHs

4.4.1. River transport

North-flowing rivers are a significant source of petroleum hydrocarbons to the Arctic Ocean. Russian rivers such as the Ob and the Yenisey drain huge and heavily industrialized and intensely cultivated areas to the south leading to a large anthropogenic component to the input of petroleum hydrocarbons. The Mackenzie River, on the other hand, drains a basin with only a sparse human population, and much less industry and agriculture but one that is rich in naturally-occurring petroleum hydrocarbon deposits leading to a high natural petroleum hydrocarbon input to the Arctic Ocean. Much of the hydrocarbon burden of Arctic rivers is deposited initially in the river estuaries. Gordeev (2002) reported that for the Ob and Yenisey, approximately 40% of the dissolved substances and 90% of suspended sediments are deposited in estuarine sediments, the rest entering the open sea. The flux of petroleum hydrocarbons to the Arctic Ocean has a very high seasonal variability, peaking during May to July when the maximum river flows and sediment transport occur.

In regions of high population density and industrial activity, such as occur in Russia within the basins of the large north-flowing rivers, and in Arctic cities such as Murmansk, most petroleum hydrocarbons enter freshwaters via municipal and industrial wastewater discharges, runoff from land, and atmospheric deposition. Occasional isolated oil spills from tankers and pipelines play a lesser role for lakes, rivers and other freshwater bodies. PAHs deposited on soils can also enter freshwaters via leaching. Leaching can also transport

PAHs to deeper soils and to groundwater, from where they can eventually enter river flow to the sea. Leaching rates of PAHs in soils have been estimated at 0.009–0.14 mg/m^2 per year (Wilcke, 2000).

Annual fluxes of 20 PAHs were monitored recently in the Pechora and Yenisey Rivers (AMAP, 2004). Fluxes of several PAHs could not be assessed as their concentrations in water and suspended matter in both rivers were below detection limits. The following PAH compounds were found in detectable amounts and are reported here: acenaphthene, benz[*a*]anthracene, benzo[*b*]fluoranthene, benzo[*e*]pyrene, perylene, benzo[*k*]fluoranthene, benzo[*a*]-pyrene, dibenz[*a,h*] anthracene, indeno[1,2,3-*cd*]pyrene and benzo[*ghi*]perylene. In both rivers, PAH fluxes are dominated by the more soluble 2-ring PAHs (naphthalene, 2-methylnaphthalene, biphenyl) and, to a certain extent, 3-ring PAHs (fluorene, phenanthrene). At the downstream Ust'–Port cross-section of the Yenisey River, the total PAH flux (i.e., the sum of the dissolved and suspended fractions) was 110 t/yr, which is significantly lower than at the upstream Igarka cross-section (440 t/yr). This confirmed an absence of additional PAH sources between the two cross-sections along this part of the river. However, fluxes of some PAHs at the downstream Andeg cross-section of the Pechora River were significantly higher than at the upstream Oksino cross-section. This was true not only for 2- and 3-ring PAHs (such as 2-methylnaphthalene, fluorene and phenanthrene) but also for the heavier PAHs (fluoranthene and pyrene). That resulted in a considerable difference between the total PAH fluxes at Andeg (140 t/yr) and Oksino (96.6 t/yr) and provides additional evidence of local pollution sources between the Oksino and Andeg cross-sections of the Pechora River.

The Mackenzie River is by far the largest single source of sediment to the Arctic Ocean delivering over 124 million t/yr, which is six times the amount discharged by the Lena River, the next largest contributor of sediment to the Arctic Ocean. The amount of sediment discharged by the Mackenzie River accounts for about 55% of total sediment loading to the Arctic Ocean and 99% of the sediment loading from all Canadian rivers (Stein and Macdonald, 2004). In terms of water discharge, the Mackenzie River is fourth largest among the rivers discharging to the Arctic Ocean (Milliman and Meade, 1983), and contributes 90% of the total freshwater input from Canada (Stein and Macdonald, 2004).

The Mackenzie River delta and adjacent shelf area are exceptional because the suspended particulate matter and sediments contain surprisingly high concentrations of PAHs for an area considered 'pristine' (Hites et al., 1980; Yunker et al., 1993). One obvious source of hydrocarbons is the substantial hydrocarbon deposits which occur throughout the Mackenzie River basin. Numerous natural seeps have been known for centuries by indigenous people in the region and have been recorded during the expeditions of various European explorers (GNWT, 2003). Important hydrocarbon-rich areas include: the Athabasca Tar Sands, the Mackenzie River at Norman Wells, and the Liard River.

- The Athabasca oil-sand deposit in north-eastern Alberta contains approximately 206 000 million m3 bitumen (AEUB, 2001). The oil-bearing material is often exposed at the surface and where rivers cut through land formations thereby forming a diffuse source of

Table 4.18. Oil pipeline accidents 1985–2002 (based on Vorobyev et al., 2005, and data from AK Transneft company).

Year	Length, km	Number of accidents		Corrosion	Manufacturing defect	Construction defect	Mechanical damage	Other, incl. illegal intrusion
		total	per 1000 km					
1987	64069	16	0.25	3	3	3	6	1
1988	65866	25	0.38	3	5	10	5	2
1989	66291	17	0.26	2	4	5	4	2
1990	66700	14	0.21	5	2	3	4	0
1991	65350	9	0.14	1	2	4	2	0
1992	48100	10	0.21	0	4	2	4	0
1993	48100	12	0.24	2	1	4	4	1
1994	49600	6	0.12	1	2	1	2	0
1995	47200	7	0.15	2	2	3	0	0
1996	47200	16	0.34	3	1	3	4	5
1987-1996		**132**	**0.23**	**22**	**26**	**38**	**35**	**11**
1997	47200	6	0.13	0	0	0	3	3
1998	47200	18	0.38	3	1	3	7	4
1999	47200	13	0.27	1	1	5	3	3
2000	47200	9	0.19	2	0	0	2	5
2001	48500	11	0.23	0	0	4	0	7
2002	48500	7	0.14	1	0	1	0	5
2003	50 000	18	0.36	0	0	1	1	16
2004	50 000	19	0.38	0	2	0	2	15
2005	50 000	13	0.26	0	2	2	1	8
2006	48 650	18	0.37	1	-	2	-	15
1997-2005		**132**	**0.27**	**8**	**6**	**18**	**19**	**81**

hydrocarbons. In addition, extensive surface mining and refining activities can increase the introduction of hydrocarbons into the environment at point sources. In theory, hydrocarbons have a clear transport pathway to the Arctic Ocean from the tar sands area via the Athabasca River/ Lake Athabasca / Slave River/ Great Slave Lake/ Mackenzie River system.

- Oil production at Norman Wells has occurred since oil was discovered in 1919. It has operated at full capacity since the early 1980s and a small refinery operated from the early 1920s until the mid-1990s. Numerous natural oil seeps have been recorded from Norman Wells downstream to the delta.
- A number of gas fields are in production from which there is a clear pathway of hydrocarbons down the Liard River to the Mackenzie main stem and the Arctic Ocean.

Both n-alkanes and PAHs have a variety of natural and anthropogenic sources. Several studies on the sources and origin of n-alkanes and PAHs have occurred within the Mackenzie River basin. The overall conclusion is that the hydrocarbon content of delta and shelf sediments reflects the combination of multiple sources.

For n-alkanes, the predominant source is biogenic. This is based on the observation that sample analysis showed a well-resolved series of n-alkanes and isoprenoids with a maximum at nC_{27}, a well-defined odd-even predominance, and no unresolved complex mixture (Yunker et al., 1991). Carey et al. (1990) collected suspended sediments from six locations between the confluence of the Mackenzie and Liard Rivers to the delta during summer flow conditions (1985) and during snowmelt input (1986); they found strong evidence for a phytoplankton source of n-alkanes at the Liard River location (1985) and a mixed petrogenic/biogenic source at the other locations. It was clear that the n-alkane profile of the suspended particulates was highly variable and dependent on the relative contribution from overlapping sources at a given time and place. Yunker et al. (1991) estimated the total river input of n-alkanes to the Mackenzie estuary at 440 ± 94 t/yr. The particulate flux exceeded the dissolved flux by around two orders of magnitude. The contribution of n-alkanes from coastal erosion to the overall n-alkane budget was estimated to be less than 10%. The Mackenzie River appears to have a well-defined n-alkane signature which can be used to identify the influence of the river on the hydrocarbon geochemistry of the Beaufort Sea.

For PAHs, the predominant source is petrogenic. This follows from the fact that the alkyl homologues maximize at C_2 or higher for the naphthalenes and at C_1 or C_2 for the phenanthrene/anthracenes and pyrene/fluoranthenes, and that the concentrations of the alkyl homologues exceed the concentrations for the sum of the parent compounds (Shaw et al., 1979; Steinhauer and Boehm, 1992). It appears that the origin of these petrogenic PAHs does not include the Athabasca River system and tar sands because the PAH profiles from there are distinctly different from those of the Mackenzie delta area and are thought to be atmospherically-deposited combustion-related PAHs (Headley et al., 2002). This conclusion is supported by the results of other studies which indicate that only a small proportion (perhaps as little as 3%) of the Athabasca River sediment load can be expected from oil-sand sources along the main stem of the river (Conly et al., 2002) and that Great Slave Lake (and presumably Lake Athabasca) probably act as storage basins for contaminant-enriched sediments (Mudroch et al., 1992). The Mackenzie River clearly contributes almost all river-delivered PAHs given that it accounts for 90% of the freshwater input to the Canadian Arctic coastal zone and that Canadian rivers outside the Mackenzie basin have PAH concentrations that are roughly ten times lower than in the Mackenzie system for parent PAHs and 100 times lower for alkyl PAHs (Yunker et al., 2002).

PAHs from non-petrogenic sources contribute to a lesser extent to the PAH assemblage found in the Mackenzie River particulates and delta sediments. Based on the relative abundance of kinetic and thermodynamic PAH isomers, combustion-related PAHs are detectable but make a minor contribution to the total PAH assemblage (Yunker and Macdonald, 1995). Another contribution is plant detritus. This is based on the presence and distribution of certain higher plant PAHs (e.g., cadalene, pimanthrene, simonellite and retene) in Mackenzie delta suspended particulates and shelf sediments. The contribution appears to be substantial in that the concentrations of higher plant PAHs occur at approximately the same range as parent PAHs. Finally, Yunker and Macdonald (1995) showed that the compositional patterns of PAHs in Mackenzie River particulates, shelf sediments and peat from eroding areas of the Beaufort Sea are distinctly different from those in long-range transported aerosols collected at Alert in the Northwest Territories, indicating that any contribution from long-range atmospheric transport is insignificant.

The total PAH flux from the Mackenzie River is 49 ± 8 t/yr with the particulate flux at least two orders of magnitude greater than the dissolved PAH flux; peat erosion contributes substantially less than 1% of the river particulate flux for most PAHs (Yunker et al., 1991).

In northern Alaska, pond waters away from Prudhoe Bay contain 0.1–0.2 ppb total aromatic hydrocarbons, similar to concentrations in pristine marine waters (Woodward et al., 1988). Hydrocarbons derived from the various sources are detectable as elevated levels of aliphatic and polycyclic aromatic hydrocarbons in Colville River sediment and, downriver, in Harrison Bay sediment (Boehm, 1987). Concentrations of indicator hydrocarbons from these multiple sources are both high and chemically similar to those found in petroleum and this makes it difficult to detect or distinguish non-point source anthropogenic contamination from natural background levels.

4.4.2. Atmospheric transport

Unlike transport by ocean currents which can take years, atmospheric transport takes only a matter of days to deliver petroleum hydrocarbons from the original place of release to a distant receptor. Air transport patterns are highly dependent on season and on the duration and frequency of major weather systems. As well as volatile contaminants the atmosphere can also deliver aerosols containing particle-reactive contaminants such as PAHs. Atmospheric transport appears to be the major pathway for anthropogenic PAHs originating in Eastern Europe to reach the Eurasian Arctic (Wania et al., 1999). Pyrogenic PAHs derived from local, regional and distant forest fires, grass fires, and agricultural-related fires also contribute to atmospheric PAH deposition, but the amount of petrogenic PAH from this source is insignificant. On the North Slope of Alaska pyrogenic PAH signals are significant in tundra soils and form a depositional record of atmospheric fallout from tundra fires (BLM, 2005). Data from the Yamalo-Nenets region showed that 10 000 t of pollutants were emitted to the air in 2004 from 920 million m^3 of gas during flaring.

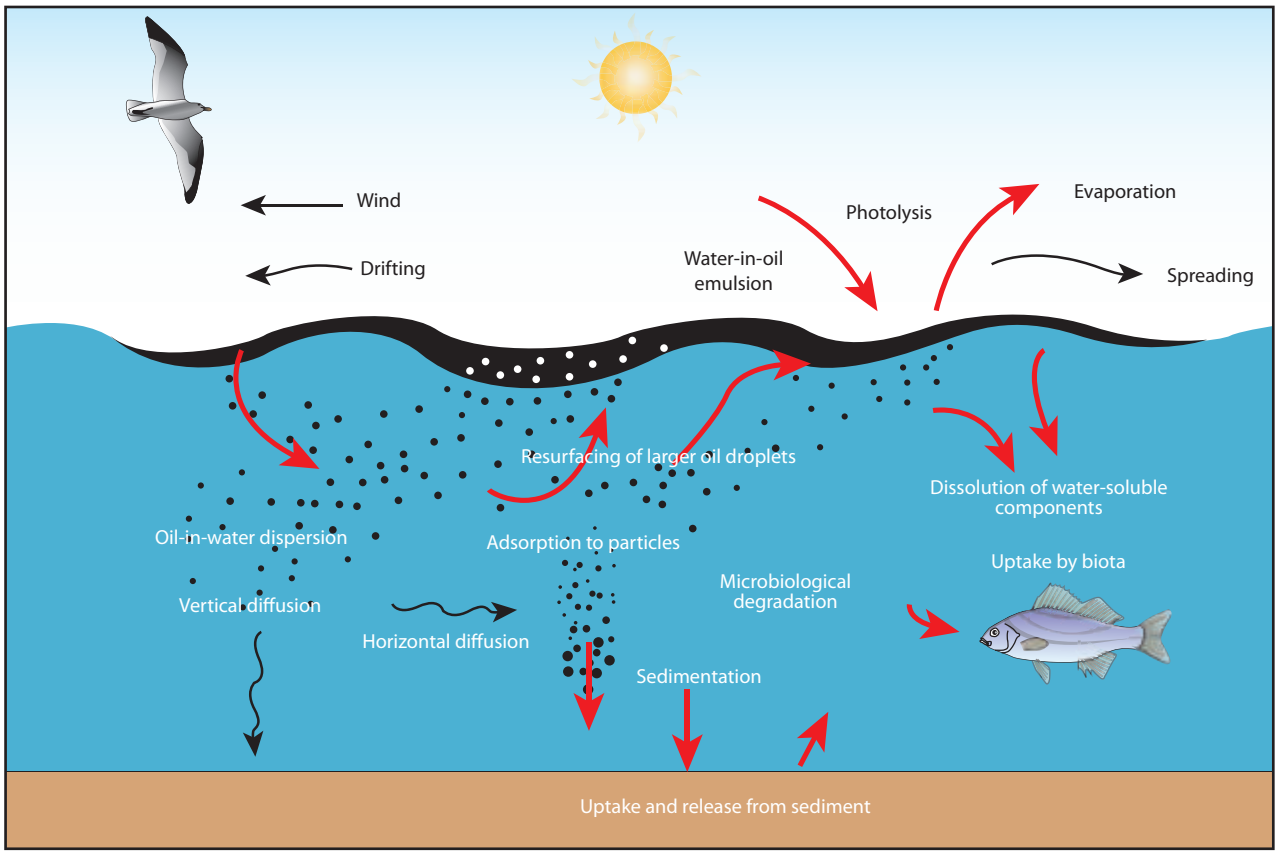


Figure 4.6. Important weathering processes for a marine oil spill (source: SINTEF, Norway).

4.4.3. Environmental fate and behaviour of marine oil spills

4.4.3.1. Weathering of oil

Releases of petroleum into the environment often fall into two categories, namely chronic (continuous) and acute (sudden) releases. An example of a chronic release containing low concentrations of oil (as oil droplets and dissolved components) into the environment is the discharge of produced water from a production unit. An example of an acute release of oil to the marine environment might include an accidental spill from a grounded tanker. Chronic releases may involve large volumes of water with low concentrations of oil (mg/L) or small volume oil spills that persist over longer periods; acute releases may involve large volumes of oil released to the environment (t/km^2) in a single event.

Processes such as dilution and chemical or biological degradation are important for the fate of the petroleum hydrocarbons in continuous (chronic) releases, whereas the fate of acute releases is often governed by processes such as advection and surface spreading, evaporation, emulsification, and natural dispersion. An acute oil spill at sea undergoes several physical and chemical processes, which can change the behaviour of the oil; these changes are important for influencing the magnitude and duration of environmental effects, and for contingency planning and decisions concerning clean-up operations. Some processes lead to removal of the oil from the sea surface while others make the oil spill more persistent (Figure 4.6).

In the case of an oil spill in ice-infested waters the drift and spreading, distribution, and weathering processes affecting the oil spill will differ from those in ice-free waters.

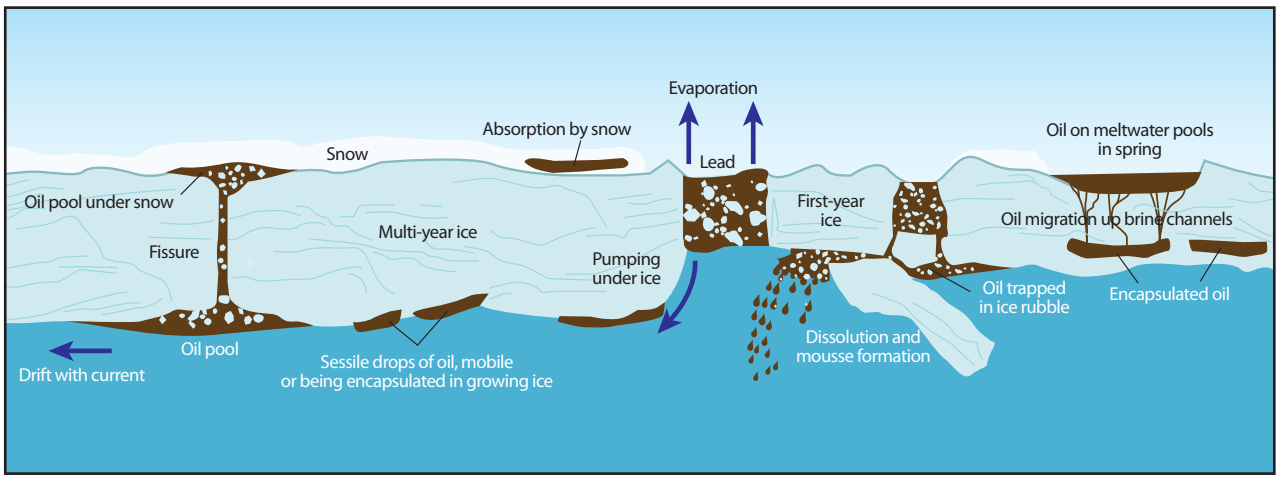


Figure 4.7. Possible distribution for a marine oil spill in ice-infested waters (AMAP, 1998).

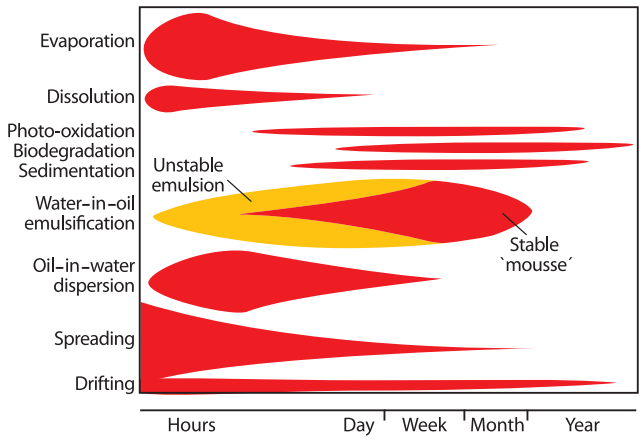


Figure 4.8. Time windows for the most important weathering processes (derived from Mackay et al., 1983).

Ice prevents oil from spreading freely and oil may become trapped or encapsulated within the ice. The presence of ice can also influence the environmental consequences of a spill since areas such as leads and polynyas may have high levels of biological productivity and are important habitats for marine organisms, birds and marine mammals (Figure 4.7).

Weathering of oil slicks at sea is not a homogeneous process for all types of oil. The rate and extent of the weathering processes are highly variable among different types of oil. Several laboratory studies (e.g., Daling and Strøm, 1999; Daling and Brandvik, 2002; Brandvik et al., 1996a; Leirvik, 2005) and field trials (e.g., Brandvik et al., 1996b) have shown that different oil types and environmental conditions may cause different weathering behaviour of an oil spill at sea (Figure 4.8).

Prominent properties of oil influencing its fate are density, viscosity, solubility and volatility. The effect of temperature on each of these properties and the implications for the fate of spilled oil in the Arctic are given in Table 4.19. Generally, cold temperatures slow the change in oil properties leading to an increase in persistence and an increased chance of mousse formation. Depending on the situation, the changes can either improve or reduce the effectiveness of clean-up measures. Reduction in light intensity and the total absence of light for part of the year also reduce the rate of transformation and degradation of oil in the Arctic.

Perhaps the most important factor influencing the fate of oil and the effectiveness of clean-up measures is the presence of ice and snow. A number of reviews have described the behaviour of oil in ice (Mackay, 1978; Dome et al., 1982; Fingas and Hollebone, 2001; NAS, 2003). Different ice conditions dramatically influence the fate and behaviour of oil. Thirty percent ice cover is taken as a point at which the behaviour of oil changes significantly relative to open-water conditions.

4.4.3.2. Spreading

Spreading is one of the most important processes during the early stages of an oil spill in water; the greater the spreading, the greater the rate of weathering of the oil. Initially, the oil spreads as a continuous slick, with the spreading rate influenced by the viscosity, pour point and density of the oil. High viscosity oils spread more slowly and oils with a pour point 10 to 15 °C above sea temperature may experience limited spreading due to their semi-solid character.

The spreading and oil film thickness from a 15 m^3 experimental oil release after 2 to 28 hours of weathering are illustrated in Figure 4.9. After the initial phase, spreading is dominated by physical oceanographic conditions (waves, wind and currents). In most cases, the oil slick will be scattered as 'windows' over a large area in the direction of the wind. Oil thickness will be non-uniform within the slick and will vary from several millimetres in the thick emulsified oil to only a few microns in the 'blueshine' areas. This non-uniform distribution of the slick can limit the effectiveness of clean-up operations at sea. Typically, the thick emulsified oil represents only 5 to 10% of the total slick area, but contains the major part (80–90%) of the volume of the oil slick.

Drift and spreading of oil spills in ice depends strongly on ice conditions. With a marine oil spill in broken, dynamic ice, little is known about the exact behaviour of the oil and the ice–oil interaction. Only a few small-scale basin studies and full-scale experimental releases have been performed. The present rule-of-thumb is that if the ice coverage is less than 30%, then the oil drift/spread will be similar to that typical of spills in open water. If the ice coverage is greater than 30%, then the oil will drift with the ice and the spreading will be significantly reduced. This will also be different in an autumn freeze-up situation where frazil ice is being created continuously between larger ice floes as compared to a spring

Table 4.19. Effect of temperature on various oil properties and the related influence on the fate of the oil.

Property	Effect, 0–25 °C	Influence on fate of oil
Density	870 kg/m^3 (0 °C) to 850 kg/m^3 (25 °C) (approx. 0.1% per °C)	As temperature decreases oil spreads more slowly. There is a greater tendency for oil to enter the water column, and to remain there for longer. There is also a higher chance of interaction with particles and thus a greater chance of sedimentation to the bottom.
Viscosity	Viscosity increases 2- to 4-fold with a decrease in temperature from 25 °C to 0 °C	If oil passes through its pour point at low temperatures it may gel or become semi-solid (the pour point for Beaufort Sea Issungnak crude oil is 5 °C). Gelled oil will persist for longer, but will be easier to clean up.
Solubility	A 25 °C decrease in temperature results in an approx. 4-fold decrease in solubility	Low temperatures result in lower rates of dissolution; water soluble components of oil may persist for longer.
Volatility	A 25 °C decrease in temperature results in an approx. 10-fold decrease in vapour pressure	As temperature falls, vapour pressure decreases with a corresponding decrease in evaporation rates. This leads to a preferential enrichment of non-volatile waxes and asphalthenes making the oil more susceptible to water-in-oil emulsion formation ('chocolate mousse'). Reduced volatility also leads to a delay in changes in those properties of oil which slow down spreading; consequently dispersion/persistence increase.

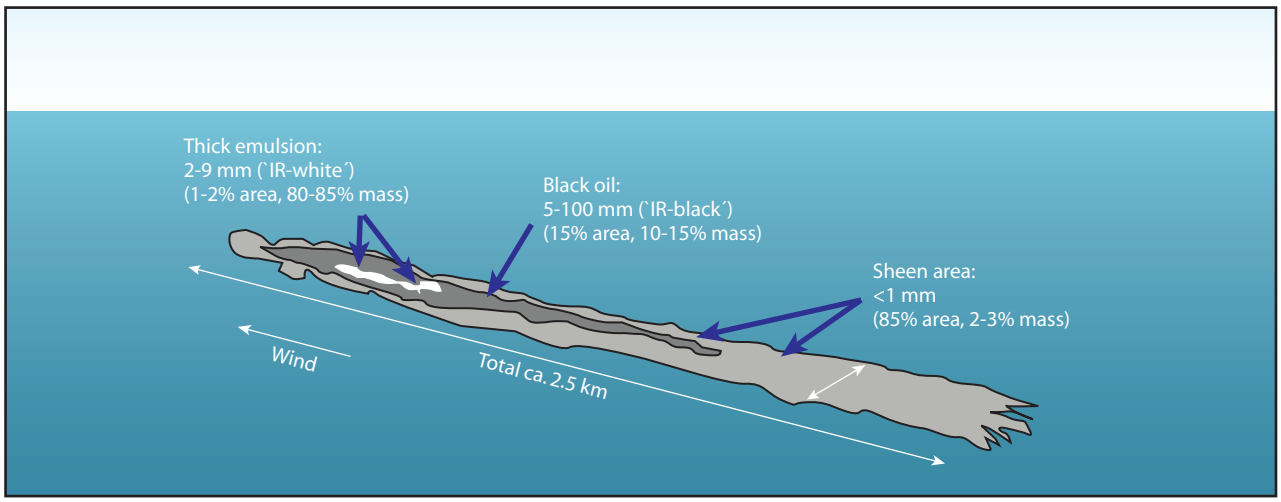


Figure 4.9. Spreading and oil film thickness from a 15 m³ experimental oil release in the Norwegian Sea in 1996. Composite figure based on remote sensing data from a surveillance aircraft used for detecting oil spills (Brandvik et al., 1996b).

melting situation where oil is released from the ice and where larger areas of open water are available. The spreading of two experimental oil releases (27–30 m³ with the same crude oil) is given as film thickness in Figure 4.10a. The first spill was an Arctic experimental oil release in broken, dynamic ice with high ice coverage (70–90%), while the second was a similar release in open temperate water (13 °C). Whereas the open water release of oil spreads out and thins on the sea surface and cannot be detected after three days, the film thickness of oil spilled in high ice cover shows little change after three days.

4.4.3.3. Evaporation

The extent of evaporation from an oil spill is primarily determined by the amount of volatile components in the oil and the ambient temperature in the receiving environment. The greater the proportion of oil components with low boiling points, the greater the degree of evaporation. In the case of a spill in the marine environment, sea temperature and oil film thickness are also important factors. The initial spread of a slick occurs over an expanding area and increases the rate of evaporation. Generally, oil components with a boiling point below 200 °C will evaporate within 24 hours at a water temperature of 13 °C. As surface water temperatures in the Arctic seldom reach 13 °C, evaporation rates will decrease accordingly. The evaporative loss from an oil spill in ice depends strongly on ice conditions. If spreading is prevented by dense ice, evaporation will be reduced due to the increased film thickness. Evaporation of a crude oil on open water and in dense ice is compared in Figure 4.10b In open water the evaporative loss is close to 40%, which is not unusual for light crude oils. In dense ice, the evaporative loss levels out at between 20 and 25% for the same type of oil.

4.4.3.4. Water/oil-emulsification

Oil spilled at sea takes up water to form water-in-oil (w/o) emulsions. This emulsification increases the volume of the oil slick by a factor of two to five (50–80% water). The water uptake causes the formation of viscous and often very stable w/o-emulsions, so-called 'chocolate mousse', which makes the oil much more persistent at the sea surface. The formation of a w/o-emulsion slows other weathering processes, especially evaporation and natural dispersion.

Both the rate and amount of water uptake depend on the composition of the oil and the sea state. The presence of breaking waves (at wind speeds above 5 m/s) is usually regarded as necessary for w/o-emulsification, but water uptake may take place slowly, and to a lesser extent in calmer weather conditions. Ice has a wave damping effect and the reduced energy can dramatically lower w/o-emulsification.

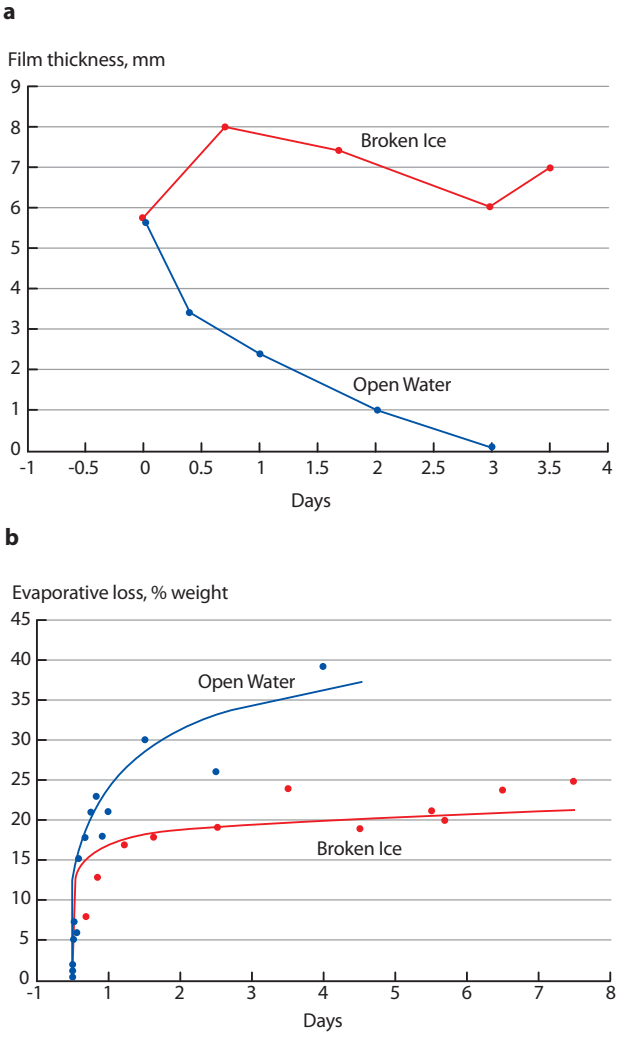


Figure 4.10. Changes in (a) film thickness, and (b) evaporative loss for the experimental crude oil releases as a function of time for both an open-water spill (Haltenbank-89) and an oil-in-ice spill (Marginal Ice Zone 1993) in Norwegian waters (Brandvik et al., 2004).

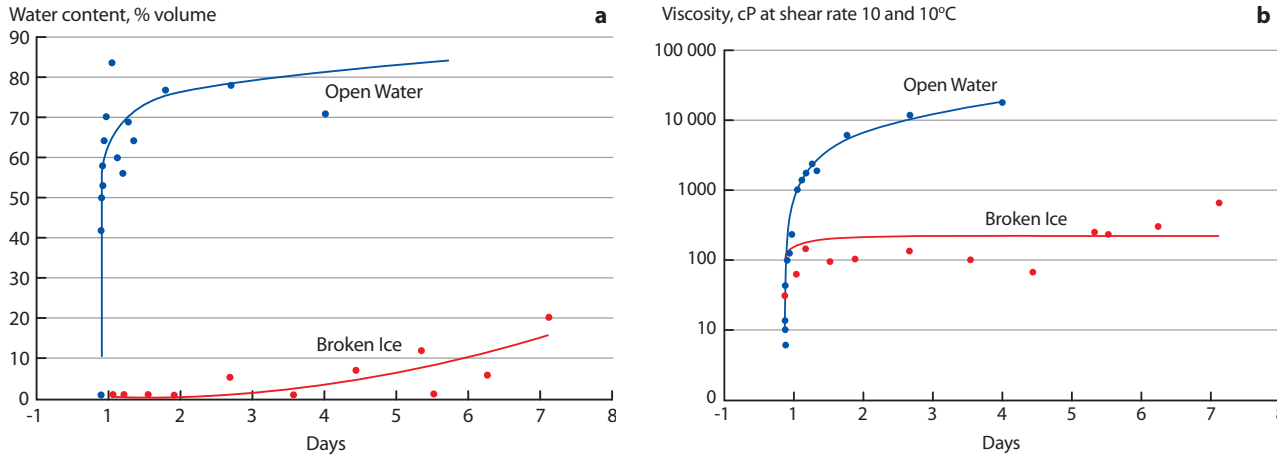


Figure 4.11. Changes in (a) water uptake, and (b) viscosity of the emulsion for the experimental crude oil releases as a function of time for both an open-water spill (Haltenbank-89) and an oil-in-ice spill (Marginal Ice Zone 1993) in Norwegian Arctic waters (Brandvik et al., 2004).

Both higher densities of larger ice sheets and frazil/slushy ice will damp the waves and reduce emulsification.

Figure 4.11 shows the reduced water uptake and the reduced viscosity of the w/o-emulsion due to wave damping with an oil spill in ice. The reduced viscosity of the oil has important implications for oil spill clean-up operations (selection of skimmers, pumps, pumping capacity etc.) and the long-term fate of the spilled oil.

4.4.3.5. Natural dispersion

When a surface oil slick is exposed to breaking waves, energy from the waves will force some of the oil down into the water column. This submerged oil will be divided into droplets by turbulence created by breaking waves. These oil droplets will have a buoyancy dependent upon their droplet size, in accordance with Stoke's Law. The larger droplets have sufficient buoyancy to resurface quickly and re-join the surface slick whereas the smaller droplets take longer to resurface and produce the thin oil tail (the 'sheen') behind the surface oil (see Figure 4.9). The smaller droplets, usually having diameters of less than 70 μm, will be randomly distributed within the water column due to turbulence and their low buoyancy (Lunel, 1995). The size of the droplets formed is, for this reason, an important factor for the rate of natural dispersion. Droplet size is dependent upon the energy input, the viscosity of the surface oil/emulsion and interfacial tension between oil and seawater. The viscosity of the surface oil or emulsion is important for the rate of natural dispersion. This means that an oil spill involving low viscosity oil will have a much higher rate of natural dispersion than one involving an oil type which forms high viscosity emulsions.

Light crude oils, which do not form stable w/o-emulsions, usually have a higher rate of natural dispersion. The *Exxon Valdez* spill in Alaska in 1989 involved a type of crude oil that rapidly forms stable w/o-emulsions, whereas the *Braer* spill near the Shetland Islands in 1992 was a spill of a naphthenic crude oil that does not form stable w/o-emulsions. Most of the crude oil spilled in the *Braer* incident, which was twice the volume of the *Exxon Valdez* spill, naturally dispersed into the water and beach cleaning operations were not required (Thorpe, 1995).

Natural dispersion at sea for different oil types has been predicted using the SINTEF oil weathering model (Figure 4.12).

The natural dispersion of an oil spill in ice can be very limited spatially due to the low energy characteristic of some oil-in-ice spill scenarios, particularly those in areas of high ice coverage.

4.4.3.6. Sedimentation

Sedimentation occurs when the weathering of petroleum is advanced and the lighter fractions have been removed by evaporation and solubilization, or when petroleum is adsorbed onto particulate matter in the water. The latter seems to be the most important process (Connell and Miller, 1981).

A few heavy residual oils have a density greater than 1.00, and will sink in fresh or brackish water, but these situations are very rare. Normally, even very weathered oil will have a density less than the density of seawater (1.025 kg/L at 15 °C, corresponding to a nominal salinity of 35 parts per thousand).

The sinking of oil from an oil spill at sea is often caused by adhesion of oil to sediment particles. This is more likely to happen when oil is released into shallow waters which are rich in suspended particulates during periods of high sea state or in estuaries rich in suspended particulates. A typical such scenario is the grounding of a tanker on a shelving shoreline. The oil-coated sediment particles or the oil droplets with adhering sediment particles will typically be transported away from the shoreline spill location by currents and then sink as they reach lower energy environments. This scenario was the case during the

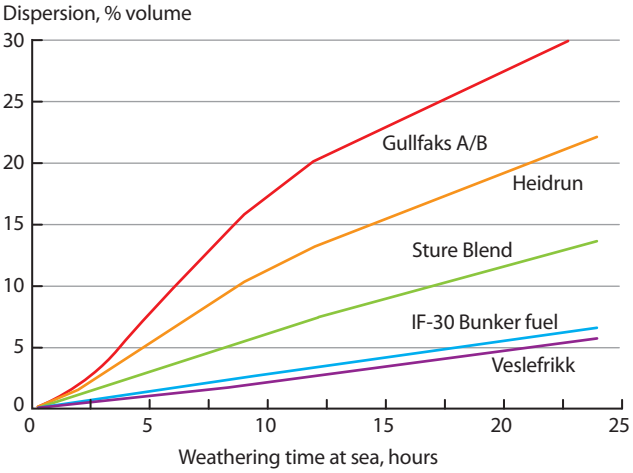


Figure 4.12. Natural dispersion of different oil types as predicted by the SINTEF oil weathering model.

Braer oil spill in 1992, where approximately 30% of the total 85 000 m^3 of crude was estimated to have been deposited on the seabed after adhesion to sediment particles (ESGOSS, 1994).

4.4.3.7. Photo-oxidation

Many of the chemical components in crude oils react with oxygen and the presence of sunlight promotes most of these reactions. Although degradation of oil due to photo-oxidation is not a significant process for the mass balance of an oil slick from an operational point of view, photo-oxidation can influence other weathering processes such as emulsification, due to the creation of polar components with surface-active properties. Some of the photo-oxidation products are also more water soluble.

4.4.3.8. Dissolution

Dissolution is of particular interest because this process increases the bioavailability of the oil components. The rate and extent to which oil components dissolve in seawater depends mainly on the water-soluble fraction (WSF) of the oil. The degree of natural dispersion is also important for the rate of dissolution, although surface spreading and water temperature may have some influence.

Many definitions of the water-soluble fraction exist, but oil components above C_9 are virtually insoluble in seawater whereas lighter compounds, particularly aromatic hydrocarbons such as benzene and toluene, are slightly soluble. These components, however, are also volatile and are thus rapidly lost to the air by evaporation, at rates typically 10 to 1000 times faster than dissolution. Laboratory-prepared WSF solutions of fresh North Sea crude oils having high oil-to-water ratios (> 1:100) might result in dissolved concentrations of 10–50 ppm. The total dissolution potential in the laboratory of soluble component to water is typically 5–15% for fresh North Sea crude oils (Daling, 1983; Daling and Johnsen, 1996). This assumes no evaporation of the light components. Around an oil slick at sea where evaporation and dissolution occur simultaneously and where the oil-to-water ratio is very low, concentrations averaging 2–20 ppb of dissolved oil or BTX (benzene, toluene and xylene) components, have been measured in the upper 10 m (Brandvik et al. 1996b; Strøm-Kristiansen et al., 1996).

Water-soluble components can leak from oil encapsulated in ice. Controlled field experiments with oil encapsulated in first-year ice for up to five months have been performed at Svalbard, Norway. The results show that the concentrations of water-soluble components in the ice decrease with depth, but were still detected even in the bottom section of the ice core. The concentration gradient as a function of time was also measured, the results indicating that water-soluble components had migrated through the porous ice to the underlying water through brine channels. The concentration of water-soluble components in the bottom 20 cm of the ice core decreased from 30 to 6 ppb during the experimental period. Although the concentrations were low, the exposure time was long (nearly four months). This indicates that ice fauna could be exposed to a substantial dose of toxic water-soluble components. The leakage of water-soluble components through the ice is of special interest because these components are highly bioavailable to marine organisms.

4.4.3.9. Oil under first-year ice

If oil is spilled under forming first-year ice such as might result from a subsea pipeline rupture or subsurface blowout, the oil would be rapidly frozen into the ice sheet. Oil will move in the direction of the under-ice currents provided that the current speed is at least 15–25 cm/s (depending on under-ice roughness) and will be trapped by under-ice cavities. If open water leads exist, the oil will be driven downwind to the adjacent ice edge where it can be splashed onto ice/snow, or, as can occur when there is a strong (>0.5 m/s) downward component of water flow at ice edges, oil may be driven down into the water column where it can interact with suspended particles and remain for variable times.

When spring arrives and the ice becomes porous, oil will rise in the many brine channels typical of first-year ice and pool on the surface and then become subject to the usual weathering processes. While this is occurring, the ice breaks up and the oil that cannot be recovered or burned moves to greater distances from the release point. Once the first-year ice has melted, the unweathered oil will form sheens and encounter the same fate as oil spilled onto open water.

4.4.3.10. Oil under multi-year ice

The major difference between multi-year ice and first-year ice is that multi-year ice is thicker and has virtually no brine channels. Thus, upward migration of oil is limited to those parts of the ice containing cracks. It has been estimated that oil accumulations under multi-year ice can be up to ten times thicker than oil accumulations under first-year ice and that upward migration is slower (Dome et al., 1982). Nonetheless, oil will reach the surface within one year and collect in melt pools. It will refreeze in autumn, and become covered with snow until the next thaw cycle begins. An experiment conducted in the Canadian High Arctic indicated that possibly as much as 90% of the oil reached the surface during the first year and was still rising slowly three years later (Dome et al., 1982). As multi-year floes melt, they will leave a thin sheen in their wake together with patches of weathered oil. NAS (2003) suggested that if there are no cracks in the multi-year ice, then the oil may rise through the natural ablation of ice over perhaps a seven- to ten-year period.

4.4.3.11. Oil on sea ice/snow

Oil can occur on sea ice/snow around artificial islands, loading terminals and storage terminals/platforms. If oil spills onto sea ice, it will spread slowly as the air temperatures cool the oil and increase its viscosity. Oil will adhere to snow and the natural roughness of the ice will produce a natural barrier to spreading as will the accumulation of wind-blown snow on the surface of the oil. Under these conditions, mechanical clean-up measures can be very effective.

4.4.4. Microbial degradation

Microbial degradation of petroleum hydrocarbons is an important process by which substantial fractions of spilled oil are weathered or eliminated from the environment. Many bacteria and some fungi have been reported to degrade petroleum hydrocarbons and their derivatives. However, microbial degradation of crude oil in the Arctic appears to proceed slowly. Furthermore, microbial degradation of PAHs, as for other hydrophobic substrates, is believed to be limited by the amounts dissolved in the water phase, with sorbed, crystalline, and non-aqueous phase liquid-dissolved PAHs being unavailable to PAH-degrading organisms (Johnsen

et al., 2005). In cases of massive contamination, PAHs will typically occur as tar droplets with a low surface-to-volume ratio, which further limits bacterial access to the PAHs. In soil, mixing does not occur and the diffusion of molecules may be orders of magnitude lower than in water. Another impediment to bacterial degradation of PAHs in soil is that bacterial cells are excluded from pores smaller than 0.2–0.8 μm and predation reduces bacterial biomass in pores larger than 2 μm. Consequently, a large fraction of PAH-degrading bacteria in soil is expected to be physically separated from sources of PAHs and to depend on diffusive transport of PAHs from their sources to the bacterial cells. Therefore, PAH-degrading populations of bacteria in soil hardly exhibit any growth.

PAH biodegradation occurs at low temperatures in aerobic and anaerobic processes (Eriksson et al., 2003). At 7 °C, PAH degradation by aerobic processes depends on soil type, whereas the extent of PAH removal by anaerobic processes is similar for all soil types. Naphthalene and 3-methylnaphthalene seem to be completely removed during 90-day incubation at 7 °C with both aerobic and anaerobic cultures. Higher PAHs are removed only partially or not at all; the removal rate for total PAHs varies from 11 to 53% for different types of soil for aerobic cultures and from 13 to 39% for anaerobic cultures (Eriksson et al., 2003).

Microbial degradation has been suggested as a means of soil remediation in areas where oil or petroleum fuel spills have occurred. Bioremediation of contaminated soils has been shown to be effective in Canadian Arctic tundra soils (Mohn et al., 2001). After one year of treatment at optimal conditions, concentrations of total petroleum hydrocarbons in soil were reduced from 196 to below 10 mg/kg at one site, and from 2109 to 195 mg/kg at another. Addition of the fertilizers ammonium chloride and sodium phosphate greatly stimulated hydrocarbon removal, indicating that biodegradation was the primary mechanism by which these reductions were achieved. A study of radiolabelled hydrocarbons in microcosms containing soils from sites across the Canadian Arctic showed that the micro-organisms contained in soils substantially removed total petroleum hydrocarbons at 7 °C but mineralization was severely limited by insufficient nitrogen and phosphorus, and also depended on the amount of total carbon and sand in the soils (Mohn and Stewart, 2000). Another study detected the presence of significant populations of cold-adapted hydrocarbon-degrading bacteria in the contaminated and pristine Canadian Arctic soils (Whyte et al., 2002). These studies show that bioremediation of hydrocarbon-contaminated Arctic tundra soils appears to be feasible and may be accelerated by various engineering strategies such as heating or inoculating the soil.

Microbial degradation can also play an important role in the fate of PAHs in the marine environment, being the most important process in the degradation of oils in the oceans. All aquatic environments contain a variety of micro-organisms. Generally, hydrocarbons are transformed by bacteria, yeast or moulds to oxidized products that are more soluble in water than the original compounds. Some studies indicate that up to 80% of benzo[*a*]pyrene may be removed by microbial degradation in some Arctic seas (Tsyban, 1999). The highest activity of microbial populations was observed in the northern Bering Sea and the southern Chukchi Sea. In the upper 45 m of the water column of the northern Bering Sea in summer, bacterio-plankton were shown to be capable of biodegrading between 20 and 90 μg of benzo[*a*]pyrene per square metre per day. The rate of biodegradation of benzo[*a*]pyrene in the East Siberian Sea was lower, only 47–58% of that measured in the northern Bering Sea (Tsyban, 1999).

4.4.5. Chemical degradation

Chemical degradation of PAHs in the Arctic is expected to be slow compared to that in temperate zones. Evaporation that occurs following an oil spill, either in the terrestrial or in the marine environment, leads to the loss of a substantial part of the spilled oil within the first 24 to 48 hours. Evaporated hydrocarbons undergo photochemical transformation in the presence of light and oxygen (direct photolysis) as well as reactions with free hydroxyl or nitrate radicals (•OH and $•NO_3$), ozone, and particulates. The transformation processes in the atmosphere can result in the formation of oxygenated PAHs (aldehydes, ketones, quinones and anhydrides of PAH bicarboxylic acids) and nitro-PAHs, some of which have undesirable health effects. Half-lives of lower molecular weight compounds in the vapour phase are relatively short (a few days), even in the cold climate of the Arctic. According to estimates, degradation times are only about one day for naphthalene and substituted naphthalenes, about two days for biphenyl, acenaphthene, fluorene, phenanthrene and anthracene, and about seven days for higher molecular weight aromatics such as pyrene. Gas-phase PAHs undergo reactions with •OH and $•NO_3$ rapidly and with ozone somewhat more slowly, whereas direct photolysis reactions are more efficient for degradation of compounds attached to particulates. Particle-associated PAHs may also undergo wet and dry deposition, which decontaminates the atmosphere but becomes the source of PAHs in soils and surface waters.

In the terrestrial environment, soils will act as a sink for PAHs and so play an important role in the fate of organic contaminants. Significant input pathways include accidental spills, chronic leaks and atmospheric deposition. PAHs in soils are predominantly sorbed to organic matter, but desorption can occur. The further fate of PAHs can include volatilisation, degradation, leaching, bioaccumulation by soil biota, and sequestration within the mineral and organic matter fractions of the soil. The rate and extent of the various fate processes are controlled by several factors including soil type and the physico-chemical properties of the individual contaminant.

4.4.6. Oil spill case studies

4.4.6.1. The *Exxon Valdez* oil spill

Although just outside the Arctic, the conditions in Prince William Sound, not least the presence of ice, mean that the *Exxon Valdez* spill is an important and relevant source of information in relation to possible consequences of oil spills in the Arctic marine environment. Considerable new information on the long-term environmental impacts of the *Exxon Valdez* spill has become available since the previous AMAP assessment in 1998.

On March 24, 1989, the tanker T/V *Exxon Valdez*, en route from Valdez, Alaska, to Los Angeles, California, ran aground on Bligh Reef in Prince William Sound, Alaska (60° N). The vessel was travelling outside normal shipping lanes in an attempt to avoid ice. Within six hours of the grounding, the *Exxon Valdez* spilled approximately 37 000 t of its 156 000 tonne cargo of North Slope crude oil. This exceeded the amount from all accidental oil spills from all vessels in U.S. waters over the next ten years (1990 to 1999, 19 600 t). The oil would eventually affect more than 2400 km of non-continuous coastline in Alaska, making it the largest oil spill

ever to occur in U.S. waters and one of the largest in the world's history (NAS, 2003).

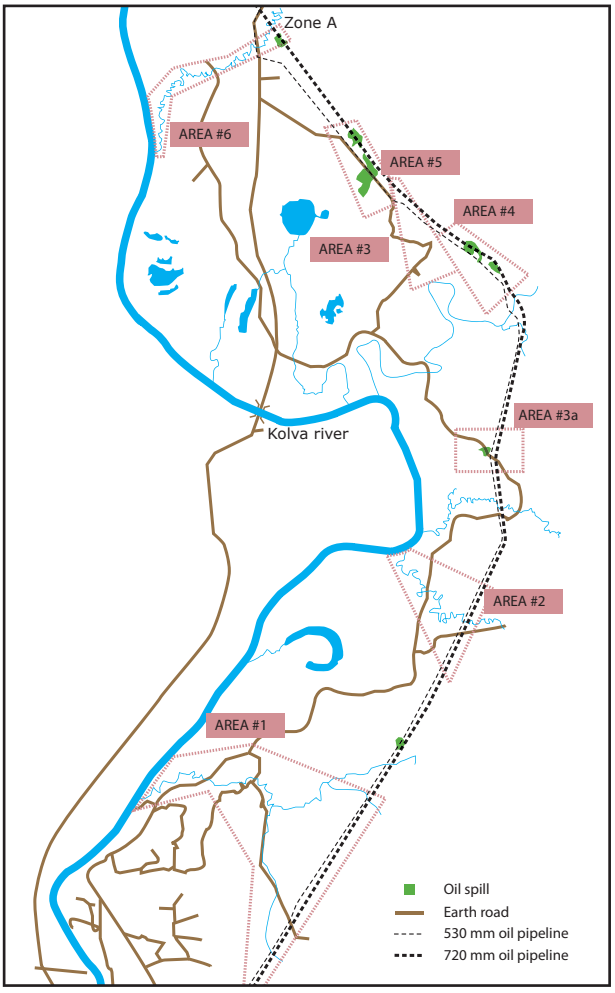
Spreading of the spilled oil was substantial and affected a large area. Around 30% of the released hydrocarbons was dissipated in a very short time due to physical weathering (Margesin and Schinner, 1999). Microbial degradation also occurred: local microbial populations of hydrocarbon degraders increased 10 000-fold despite the cold water temperatures. Two months after the spill, concentrations of microbial oil degraders averaged one million per gram of oiled beach material (Margesin and Schinner, 1999). Large quantities of oil, however, remained for a long time and some for years to come. As of 1991, 41% of the oil remained stranded on intertidal habitats in Prince William Sound. About 25% of the oil was transported out of the Sound, and was stranded primarily along the Kenai Peninsula, lower Cook Inlet, and the Kodiak Archipelago. More than 2000 km of shoreline were oiled, emphasizing the persistence of the spilled oil and the effectiveness of a spring gale and a major coastal current in the transport of a large volume of oil (NAS, 2003). There have been a number of studies which concluded that the toxicity of spilled oil decreased following the spill due to rapid weathering and that the potential for toxicity 13 years after the spill is negligible (Page et al., 2002); such results, however, have been a matter of considerable controversy (Rice et al., 2003).

The response to the *Exxon Valdez* spill involved more personnel and equipment over a longer period of time than any other accidental spill in U.S. history. At the height of the response, more than 11 000 personnel, 1400 vessels, and 85 aircraft were involved in the clean-up. Shoreline clean-up began in April 1989 and continued until September 1989 for the first year of response. The effort continued in 1990 and 1991 with clean-up in the summer months and limited shoreline monitoring during winter. Large volumes of water heated to 60 °C were used to flush oil from almost 30% of the rocky shores and gravel beaches in Prince William Sound. Despite the aggressive clean-up, oil residues persisted for more than 13 years in more sheltered habitats and porous gravel beaches. Monitoring studies have shown that the intensive treatment applied during clean-up operations resulted in delayed recovery of rocky shore intertidal communities (NAS, 2003). Bioremediation techniques were also used, more expansively than ever before. Natural biodegradation has been limited by insufficient nitrogen and phosphorus; oleophilic fertilizer enhanced biodegradation of the oil by around two-fold relative to untreated controls (Margesin and Schinner, 1999). Monitoring the fate and effects of the *Exxon Valdez* oil spill by state and federal agencies continues still.

4.4.6.2. The Komi oil spill

The Komi Republic is one of the main oil-producing regions in northwest Russia. It covers an area of around 417 thousand km^2 (Figure 4.13), has a population (in 1999) of 1.15 million inhabitants and thus an average population density of about three inhabitants per square kilometre. Seventy-five per cent of the population however lives in cities such as the capital Syktyvkar and the northern cities of Usinsk, Vorkuta, Pechora and Inta. Oil resources are located primarily in the Pechora River Basin in the Usinsk, Ukhtinsk and Pechorsk regions. Most of the oil is produced in the Kolva River basin, and is transported from the production wells to the refineries and consumers in liquid form through high pressure pipelines. Consequently, the Komi Republic possesses a vast network of more than 15 000 km of oil pipelines, and about 700 km of these pipelines carry large volumes of oil. There are two major oil and gas pipelines in the Republic: the oil pipeline Usinsk–Ukhta–Yaroslavl, which originates there, and the gas pipeline Siyaniye Severa ('The Shine of the North'), which crosses the region. Most of the pipelines cross bogs and are subject to physical stresses generated by soil and peat displacement, ground subsidence, heaving, and permafrost phenomena.

The Komi Republic has a mainly flat landscape with the Ural Mountains rising in the east. Its main river, the Pechora,



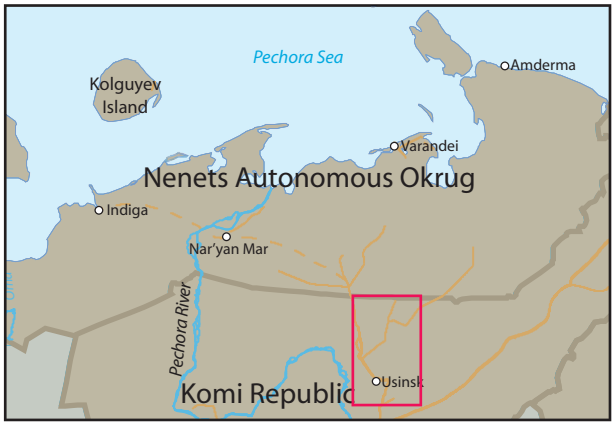


Figure 4.13. Region of the Usinsk spill in the Komi Republic in northwest Russia (see red box on map right).

Table 4.20. The properties of oils from the *Kharyaginsk* ,*Usinsk Devon* and *Usinsk Carbon* fields.

	Kharyaginsk	*Usinsk Devon*	*Usinsk Carbon*
Density at 20 °C (kg/m^3)	0.829	0.837	0.967
Viscosity at 20 °C	5.7 (at 50 °C)	15.17	513.30
Sulphur content (% wt)	0.32	0.61	2.07
Tar content (% wt)	6.0	12.1	29.0
Asphalthenes content (% wt)	0.90	0.68	11.25
Paraffin content (melting point, °C)	21.0 (55)	10.8 (50)	0.4 (48)
Vanadium content (μg/g)	0.9	16.0	100.0
Nickel content (μg/g)	2.0	0.1	-
Coke content (%)	1.72	2.21	12.02
Fractions yield, mass %: 28–200 °C	25.1	25.4	6.6
28–350 °C	50.1	52.2	24.4

is one of the largest rivers in Russia, with flows averaging 4000 m^3/s, and attaining maximum rates of 20 000 m^3/s. The river flows from its source in the Urals to the Pechora Sea, a distance of 1809 km. Its main tributary is the Ussa River, 654 km long, with an average flow of 1300 m^3/s (Yertsev et al., 2000).

The climate of the Komi Republic is moderately continental. The oil exploration zone of the Usinsk region is situated in a swampy area. Summers are cold with low precipitation, low evaporation, and high humidity; frosts can occur in any of the summer months. Winters are long and cold (min. -53 °C), and sub-zero temperatures can last for more than 200 days per year. Snow cover reaches 60 cm depth by the end of winter. The average annual temperature in this territory is -3.2 °C. Average annual precipitation is 420–460 mm. Northern, southern and south-western winds predominate through most of the year at speeds of 4–5 m/s.

Industrial pollution of the Pechora River and its tributaries began in the 1950s. In the mid-1960s an oil refinery discharged more than 20 000 m^3 of waste per day into the Izhma River (a tributary of the Pechora River) and its tributary the Ukhta River. In some locations most fish species disappeared. Other tributaries of the Pechora River (such as the Nibelj, Voi-Vozh, and Yarega Rivers) were polluted by oil products. More than 130 000 m^3 of untreated sewage were dumped into the Vorkuta River each day. By the end of the 1960s these rivers did not meet the requirements specified by the State for ecologically productive waterways.

The 720-mm diameter oil pipeline of Vozey-Golovnye Sooruzheniya owned by Komineft, began operation in 1975 at a working pressure of 4.0 mPa. In 1994, around 16 000 t of oil a day were transported through the pipeline (Muljak and Ivanov, 2004). The oil in the Vozey-Golovnye Sooruzheniya pipeline differs in composition and properties from the oil produced at the local oil fields because the pipeline transports oil from three different oil fields (*Kharyaginsk, Vozeysk* and *Usinsk*) and the proportion of oil from each field changes frequently. In addition, the oil in the pipeline is mixed with ground water, which at the time of the 1994 spill comprised about 50% of the total volume. The properties (Table 4.20) and the hydrocarbon contents (Table 4.21) of these oils can be used to make a rough estimate of oil behaviour during this spill.

During the spring and summer of 1994, inspection of the Vozey-Golovnye Sooruzheniya pipeline located in the Usinsk area of the Komi Republic, revealed an extensive failure: 23 ruptures were identified along a 52-km section of the pipeline located in a remote marshland (Borovinskih, 2004). River water, which had (since 1988) been mixed into the oil to assist in pumping, was identified as a major factor causing corrosion and deterioration of the pipeline. As a result of these ruptures, 280 ha of marshland were contaminated with oil and severely damaged. This included 116 ha of land originally contaminated by the oil spill and 164 ha damaged during the clean-up operation (Figure 4.13). The estimates given for the size of the oil spill vary. According to Muljak and Ivanov (2004), more than 100 000 t of oil leaked from the pipeline. The local newspaper Krasnoye znamya reported on November 1, 1994, that "Oil has got into the Pechora River and is now moving towards the Barents Sea. The polluted water has been transported from the site of the oil spill by the Pechora tributaries (Ussa and Kolva), which join the Pechora River some 600 km from the Pechora estuary. The total amount of oil spilled in the 52 km zone along the pipeline reached

Table 4.21. Grouped hydrocarbon contents (% mass) of the oils from the *Usinsk* field.

Temperature, °C	Petroleum and kerosene fractions			Oily fractions		
	Paraffinic fractions	Naphthenic fractions	Aromatic fractions	Mixtures of paraffinic and naphthenic fractions		Aromatic fractions
				Total	Solid melting point, °C	
28–200	70	21	9	-	-	-
200–300	63	21	16	-	-	-
300–350	-	-	-	77	-	23
350–450	-	-	-	67	12.1 (49)	33
450–490	-	-	-	57	11.0 (56)	43

200 thousand tonnes". Fourteen thousand tonnes were reported by Komi's civil defence department, 60 000 t by the government, while American oil workers at the scene of the spill estimated 200 000 t.

The Komi spill, also known as 'The Usinsk Accident', has been reported as one of the worst oil spills in history. The spill was described in the 1998 AMAP assessment; this review therefore focuses on the post-spill remediation efforts and changes in the environmental situation over the past decade.

Initial containment efforts at the sites of the oil spill, such as siphon dams built of sand, were only marginally successful. And, when the spring thaw occurred in 1995, the dams failed completely.

The clean-up work was divided into three steps: limiting the spread of the spilled oil, collecting the spilled oil, and setting up a bioremediation program (Yertsev et al., 2000). Roads were constructed to the spill sites to transport clean-up equipment. Large quantities of the oil were simply buried in large, clay-lined pits near the collection points. One of the main aims of the clean-up was to prevent the oil reaching the Arctic Ocean.

In 1994, expertise to deal with such large oil pollution events in remote areas was not available in Russia. The complexity of the river and lake networks, the difficult nature of the terrain (bogs and wetland), the Arctic climate and the remoteness of the contaminated zone limited the effectiveness of human intervention. A significant fear was that numerous large rivers (the Kolva, the Ussa and the Pechora) could bring oil pollution to major populated areas, the Pechora River estuary and the marine environment beyond, causing widespread impacts on riverine, estuarine and marine ecosystems which would require additional clean-up technologies. Therefore, the rivers and streams and associated banks were cleaned first to avoid slicks that could be carried to the sea. However, removing oil slicks was difficult due to the boggy terrain. For instance, during the first stage of the operation, which lasted 10 months, 45 km of roads were built to enable equipment to reach the polluted zones. In addition, about 5.5 km of dams and 14 hydro locks were set up to prevent the oil from reaching the main rivers. At least 34 reservoirs were built to receive oil-polluted liquids and soils collected from the areas impacted by the spill (Yertsev et al., 2000). These new structures (roads, dams, reservoirs) further damaged the ecosystem in a habitat where man-made tracks can be visible for more than 50 years. Burning was often considered the best overall solution to remove oil slicks. However, burning of the oil resulted in considerable air pollution and added to the impact on the highly vulnerable tundra soil. In addition, the Arctic climate complicated the collection of oil and oil mixed with water because different collection methods had to be used depending on the temperature. The oil itself was thick and heavy (high-paraffin content) and had to be heated before it could be pumped. Often the oil was frozen in place resembling huge slabs of black candle wax. In many areas, frozen oil was therefore removed with a backhoe. In summer, the heat caused the oil to become fluid again and in many areas the spring and summer meltwater carried oil high into forested areas adjacent to the impacted streams.

Overall, most of the oil pollution was contained inside the impact zone preventing further spreading of the oil into the river network. In 2000, in the region of the Usinsk and Vozeysk Rivers, 102 areas of primary oil contamination were identified (Gubinova, 2004).

During the second stage of the operation, loans provided by the World Bank and other financial institutions allowed advanced clean-up equipment to be brought into the spill area. Improved technological solutions including the construction of dams, hydro locks and collection installations prevented the spread of oil. New methods were also used to absorb oil on the water surface and new clean-up techniques (e.g., aerators and dispersants) were used to collect oil from contaminated river sediments and dissolved hydrocarbons from the water column (Makarova, 2004). Scrapers and dredges were used to remove oil from bogs (Yertsev et al., 2000).

The first assessment of the pollution in the area of the accident was carried out in 1995–1996. In 1999, a much broader assessment of the whole Usinsk province took place including an attempt to estimate the degree of soil pollution and to identify the remaining oil slicks. The preliminary inspection was carried out on an area of 45 135 ha; the observations were conducted along a route 691 km in length (Yertsev et al., 2000). During the inspection of the area, 350 sites were identified as polluted and ecologically damaged by oil and oil-related substances, and an area of 745 ha was designated 'severely impacted'. Based on this account, Komineft developed a corporate program of ecological rehabilitation operations and a response plan for 2000 to 2005 (Yertsev, 2004; Zagvozdkin, 2004). The main clean-up operation in the Usinsk area began in 2000 with the biological rehabilitation of the damaged ecosystem (Zagvozdkin, 2004; Muljak and Ivanov, 2004). A reduction in the size of the impacted area and in the volume of oil remaining in the spill area was noted during the period 1996 to 2003 (Figure 4.14).

Vast technical resources were deployed during the clean-up operation of the Usinsk oil spill in 1994 and the subsequent ecological rehabilitation program. In fact, because of lack of expertise on how to deal with oil pollution in such an ecosystem, the area became the testing ground for various technological innovations dealing with contamination levels, weathering processes, matrix variability (water or land), and soil biological activity (Taskaev and Makarova, 2002; Makarova, 2004).

The indigenous population of the Komi Republic traditionally harvests fish in lakes and rivers, and hunts animals and birds. The Komi region supports one of the largest herds of domestic reindeer in Russia, estimated in the 1980s to number around 65 000 to 120 000. The Pechora River basin has been an important source of fish in the Komi Republic. In the nineteenth century, the Pechora River system provided Russia with 240 t of whitefish and salmon per year. During the period 1950 to 1970, the state fishing industry harvested about 80 000 t of fish per year. During the 1980s, there was a sharp decrease of fish stocks in the region

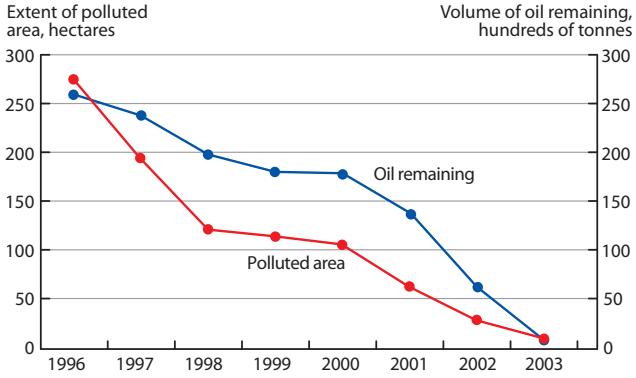


Figure 4.14. Dynamics of the clean-up operation shown as volume of remaining oil and extent of polluted area after the `Usinsk´ accident in 1994.

Table 4.22. Concentrations of some oil hydrocarbons in water samples from the Pechora River in August 1997 (Yertsev et al., 2000).

Components	Concentration, mg/L	
	Location 1	Location 2
Oil hydrocarbons		
C_6-hydrocarbons	0.0006	0.0028
$C_{7–8}$-hydrocarbons	0.0031	0.0028
$C_{9–10}$-hydrocarbons	0.0008	<0.0001
$C_{11–12}$-hydrocarbons	0.0006	<0.0001
$C_{13–14}$-hydrocarbons	0.0140	<0.0001
$C_{15–16}$-hydrocarbons	0.0680	0.0128
$C_{17–20}$-hydrocarbons	0.0700	0.0150
Sum	0.1570	0.0035
Benzene	0.0020	0.0008
Toluene	0.0060	0.0030
Xylenes	0.0090	0.0011
Ethylbenzene	0.0011	<0.0001
Naphthalene	<0.0010	0.0003
Benzo[a]pyrene	$<1 \times 10^{-6}$	$<1 \times 10^{-6}$
Phenol	<0.0005	<0.0005
Cresols	<0.0020	<0.0005
Acetaldehyde	0.0020	0.0005
Acetone	0.0080	0.0021
Carbon tetrachloride	0.0003	0.0004
Chloroform	<0.0010	<0.0010

thought to be the result of over-exploitation and pollution. Many rivers and lakes lost their fishing value and the size of some fish populations fell to critically low levels (Zakharov, 2002). There are 37 species of fish living in the Pechora River, including such commercially important species as Atlantic salmon (*Salmo salar*) (Yertsev et al., 2000). The salmon spawning areas are in the tributaries of the Pechora and Ussa Rivers, which drain the western Ural mountain range. Other important and rare fish also live in the Pechora River system; including a few species registered in the Red Book of Russian endangered species. The Usinsk oil spill increased the impact on fisheries resources in the local streams and waterways in the area affected by the spill and the ensuing clean-up operations (Taskaev et al., 2004). Building dykes and dams in the rivers has changed the composition of the fish fauna. Population sizes of some species have reduced while species formerly unknown in the area have appeared.

To assess the effectiveness of clean-up operations, the ecosystem of the Kolva River was monitored in 1995 for possible changes. Following the spill, oxygen saturation levels decreased and pH increased relative to historical norms but recovered to typical background values within a year or two (Taskaev et al., 2004). Accelerated erosion, however, continued in many streams, which led to higher concentrations of suspended particles on which oil compounds were adsorbed. The diversity indices of plankton communities in the Kolva River fluctuated considerably. Several deformed crustaceans were also observed in the waterways flowing through the polluted area (Taskaev et al., 2004).

In August 1997, concentrations of hydrocarbons were measured at two locations 120 km apart in the Pechora River, each having the same hydrocarbon compositional profile as the oil from the spill (Table 4.22) (Yertsev et al., 2000).

A monitoring study was conducted in the polluted lower Kolva River to estimate dynamic changes in the populations of several species of fish. During 1995 to 1998, the populations of minnows and European grayling decreased 40- and 10-fold, respectively. By 2000, the stocks of both species had increased slightly. As late as 2003, fish caught in the Ussa River and at various locations along the Kolva River had high concentrations of oil in organs and tissues. PAHs found in fish tissues included carcinogenic and mutagenic compounds (Gubinova, 2004).

In 2001–2002, the entire pike population in Lake Shutchye had morphological deformations, including shorter heads, longer lower jaws, and atrophy in various body parts such as fins (Gubinova, 2004).

Despite the emergency response and clean-up operation, there was a sharp change in the ecosystems of the lower Kolva River. The trophic structure of the river ecosystem altered significantly; for example, the density of predators increased substantially. Changes in the structure of soil communities have also been observed since 1998 (Taskaev et al., 2004).

In 2003, there were still 14 ha of polluted and damaged land, which represents 5% of the originally polluted area.

Results from recent monitoring programs (Zakharov, 2002; Muljak and Ivanov, 2004; Popov et al., 2004) indicate that the concentrations of petroleum hydrocarbons in the environment have returned to values measured prior to the accident. But few data have been reported on the long-term effects of the spill on the ecosystem. Consequently, the Komi accident raises unanswered questions about the severity of the chronic effects of oil residues on the health of the entire ecosystem of the Komi Republic.

In October 2004, the State Committee on Ecology at the Federal Assembly endorsed the unprecedented clean-up operations employed in response to the Komi spill and made recommendations for how similar accidental discharges should be dealt with in the future.

Minor oil releases have continued in the Komi region since the spill in 1994. In general, the largest releases take place each year between March and May when snow melts. The total number of releases of oil recorded in 1994, 1995, 1996, and the first half of 1997 was respectively 770, 972, 395 and 137 (Table 4.23).

4.4.6.3. The blowout at the *Kumzha* gas condensate field

4.4.6.3.1. Summary

In 1974, the first well was drilled at the *Kumzha* gas condensate field located in the Pechora River delta in the Nenets Autonomous Okrug. The field held an estimated 10.2 billion m^3 of gas and 3.5 million t of gas condensate. In

Table 4.23. Accidental releases of oil from Komineft Ltd. pipelines in the period January to June 1997.

	January	February	March	April	May	June	Total
Number of releases	17	21	26	26	25	22	137
Contaminated area, m^2	590	960	775	1310	1030	610	5275
Volume of oily liquid, m^3	7.2	13.2	10.4	12.45	10.3	10.9	64.4

November 1979, the first blowout in the field occurred in Well 13. This was a minor blowout which was ended within a few days. A much more severe blowout, however, occurred in Well 9 in November 1980. It took until May 1987 to end the blowout. Prior to stopping the flow, about 2 million m^3 of gas and hundreds of tonnes of condensate were released into the environment every day. The released gas and some of the condensate were mostly burned; condensate that was not burned was contained by a dam. During the event, an underground nuclear detonation was used in an unsuccessful attempt to end the blowout.

4.4.6.3.2. The accident

Drilling of Well 9 began in September 1978 in the pre-anticline zone of the Kumzha reservoir in the Pechora River delta. The purpose for drilling the well was a further assessment of the reservoir's structure and estimation of the oil and gas capacity of this hydrocarbon deposit. The anticline was 1.5 km deep and to reach it drilling at a 26 degree angle was required. The drilling depth was 2300 m, the length of the drill stem reached 2859 m. It took nine months to complete the drilling program. After testing the gas and condensate stream the well was to be capped and contained. However, it soon became evident that the production casing was damaged and that gas was outside the well casing and escaping along three vents at a rate of 2 million m^3 per day. This was clearly an emergency situation. The drill team decided to end the well by pumping a slurry of calcium chloride and cement directly into it. During the operation in late November 1980, mud springs appeared in the snow around the rig; gas appeared outside the casing. The drill team then cut the power and opened the vents to release the pressure and eliminate the mud springs. The flaring gas was burning intensely, but gas was still escaping outside the well casing and ejecting material in its path; however, no major explosion occurred. A rescue team arrived the day after the accident, cleared the well head and the site around the rig and determined that the directional intermediate casing was broken and blocked (Tolkachev, 2000).

Over the next few months, a number of attempts were made to gain control of the situation. Unfortunately, these efforts all failed and the situation at Well 9 remained dire.

Later investigation of the Kumzha Well 9 accident determined that many safety violations had occurred during the drilling, cementing, and testing of the well.

4.4.6.3.3. Use of a nuclear device

After the failure to end the blowout by conventional means, the authorities agreed to detonate a nuclear device (see Box 4.2). Borehole 25 was drilled to the necessary estimated depth, a nuclear device was installed, and the hole was cemented shut. In the town of Naryan-Mar, some 100 km away, all residents were asked to leave their houses and to stay in the streets. On May 25, 1981 an atomic bomb with a power of 37.6 kt of TNT-equivalent was detonated at 1470 m depth (the atomic bomb detonated over Hiroshima was equivalent to about 20 kt TNT). A special radioactivity safety commission visited the site forty minutes after the explosion, and concluded that the radiation levels were within acceptable limits. However, the gas blowout was not under control, and soon the flow of gas resumed. At the surface a 100 m wide crater had been formed by the explosion, and the crater turned into a boiling black mass of water and soil containing sulphur trioxide (Tolkachev, 2000; Pelyushenskii, 2005).

Later investigations indicated that the nuclear explosion would not have succeeded in controlling the blowout as the bomb had not been placed in the correct location to be effective. Borehole 25 had been drilled 260 m from the failed wellhead, but the coordinates and angles provided by the geophysicists for the drilling to intercept Well 9 were incorrect because of errors in well drilling records. Ultimately, these miscalculations predetermined the failure of the nuclear explosion. The absence of reliable data about the exact position of the failed well, the casing, and drilling direction explain why it took so long to handle the emergency at Kumzha (Tolkachev, 2000; Pelyushenskii, 2005).

After the failure of the nuclear explosion to end the blowout, several thousand tonnes of sand and gravel from the upper Pechora River were barged to Kumzha to build a dam 100 m long and 2.6 m high. Over the next few years many attempts were made to terminate the blowout. Working conditions were extremely difficult, as a number of gas and mud springs and huge craters had formed around the original Well 9. Several emergency wells were drilled; Well 26 was drilled toward the centre of where the nuclear explosion had occurred to investigate its consequences and another, Well 27, to make a connection with the drilling string of the initial Well 9. While drilling Well 26, a powerful gas blowout occurred at a depth of about 100 m. The gas caught fire and a flare 20 m high raged for five hours. The rig collapsed (Tolkachev, 2000).

Well 27 was drilled to reach and connect to the failed casing of Well 9. However, due to inaccurate positions of the directions of Well 9, this proved very difficult, and it was decided to stop the ongoing search for the failed shaft of Well 9 and instead to complete the drilling of Well 27 down to the gas layer and then to pump water into it thereby forming hydrates and sealing the well. This drilling continued for several months, but was to no effect and only demonstrated that the failed borehole of Well 9 was at a significant distance from the estimated location. Pumping was subsequently stopped (Tolkachev, 2000).

4.4.6.3.4. Ending the blowout in 1987

After several drill stems connected to Well 27, some metal from the lost casing of Well 9 was found. The rescue team also took advantage of a range of new advanced metal detecting methods to determine the correct position of the well. In March 1987, at a depth of 1003 m, the drill

Box 4.2 Use of underground nuclear explosions for oil and gas exploration, development and emergency response

The Soviet program of peaceful (i.e., unrelated to military activities) underground nuclear explosions began in 1968 after the start of a similar program in the United States. During the U.S. program (1964–1972), 27 underground nuclear explosions were used to enhance the development of gas deposits. During the program in the Soviet Union, 116 nuclear explosions took place: to conduct deep seismic exploration of the earth's crust (39 explosions); to create underground storage reservoirs for hydrocarbons (36 explosions) and toxic industrial wastes (2 explosions); to kill emergency blowouts in gas deposits (5 explosions including the one at Kumzha Well 9); and other applications (32 explosions) (Pelyushenskii, 2005).

reached the casing of failed Well 9. During some unique operations, workers completed repairs to the casing of the failed well consisting of three pipes cemented into each other with a pipe inside. By 25 May 1987, Well 9 was sealed and cemented up to the wellhead. The water-filled crater at the site covered wellheads 5, 9 and 10 of the *Kumzha* field. Gostechnadzor of Russia, estimated that the total costs of the emergency operations at Kumzha reached around US$ 2 million (Tolkachev, 2000; Glotov, 2005). Clearly, the true cost of the accident is many times this figure when ecological damage and loss of production are included.

Status at the site today, Nenetsky State Nature Reserve

The Nenetsky State Nature Reserve was created in 1997. Its territory in the delta of the Pechora River includes the site of Kumzha Well 9 (and three more related sealed wells). Since 2001, the workers of the reserve have reported new gas emissions in connection with Well 9. At the bottom of the crater there were two mud springs still emitting hydrocarbons. In 2005, samples of surface water contained oil at concentrations greatly exceeding the Russian standard for freshwater (Glotov, 2005).

Floodwaters are gradually destroying the protective dam separating the crater from the river. The representatives of the Nenetsky State Nature Reserve are concerned that the dam will fail, and that the condensate collected in the crater will find its way to the river. It was also observed that the concentrations of oil-related chemicals in tissues of valuable whitefish species living in the delta of the Pechora River have increased five-fold compared to concentrations measured in 2003, when they were within the limits of maximum permissible concentration in the Russian Federation. Scientists studying the area suspect that the sharp increase in the contaminant body burdens in local fish is related to leakage of contaminants from the gradually deteriorating protective dam around Well 9, a situation which may ultimately result in the contamination of the Pechora River and estuary (Glotov, 2005; Pelyushenskii, 2005).

The Kumzha site today

4.5. A petroleum hydrocarbon budget for the Arctic

4.5.1. Introduction

This section describes an attempt to construct a petroleum hydrocarbon budget for the Arctic. A schematic for the budget is shown in Figure 4.15.

For the purposes of the budget, petroleum hydrocarbons are defined as the sum of the n-alkanes from nC_{13} to nC_{34} and 24 parent and alkylated PAHs (Table 4.24); these compounds typically comprise around 10% of petroleum (n-alkanes 6% and PAHs 4%) (NAS, 2003). The budget is focused on these compounds for several reasons: (a) from a chemical perspective, oil is a poor quantity to use because there are hundreds/thousands of compounds in 'oil', most of which are never measured in the environment and have many sources, not just petroleum; (b) biota do not bioaccumulate 'oil' but rather specific compounds like the PAHs; (c) diagnostic chemical ratios can be applied to n-alkanes and PAHs to establish a petrogenic source; (d) some PAHs (for example, high molecular weight unsubstituted PAHs and alkylated PAHs) behave relatively conservatively in the environment particularly in cold climates; and (e) some high quality coherent data sets exist for n-alkanes and PAHs.

There are nonetheless some significant problems associated with constructing the budget. The first is the available data. Much of the data reported in the literature are highly aggregated. Consequently, it is not possible to use the relative concentrations of the planktonic n-alkanes (C_{15}, C_{17} and C_{19}) and the higher plant n-alkanes (C_{25}, C_{27}, C_{29}, and C_{31}) to distinguish petrogenic n-alkane inputs from biogenic inputs. Similarly, few PAH data sets include specific parent compounds or alkylated homologues making it impossible to estimate the relative contribution of petrogenic, pyrogenic, diagenetic and biogenic PAHs.

Table 4.24. `Petroleum hydrocarbons´ as defined for the budget.

Petroleum hydrocarbons	= n-alkanes	+ parent PAHs	+ alkylated PAHs
	• nC_{13}–nC_{34}	• phenanthrene • anthracene • fluoranthene • pyrene • benz[*a*]anthracene • chrysene • benzo[*b,j,k*]fluoranthene • benzo[*e*]pyrene • benzo[*a*]pyrene • dibenz[*a,c/a,h*]anthracene • indeno[1,2,3-*cd*]pyrene • benzo[*ghi*]perylene	• 2-methyl naphthalene • 1-methyl naphthalene • 2,6/2,7-dimethyl naphthalene • 1,3/1,7-dimethyl naphthalene • 1,6-dimethyl naphthalene • 1,4/2,3-dimethyl naphthalene • 1,5-dimethyl naphthalene • 1,2-dimethyl phenanthrene • 3-methyl phenanthrene • 2-methyl phenanthrene • 9/4-methyl phenanthrene • 1-methyl phenanthrene

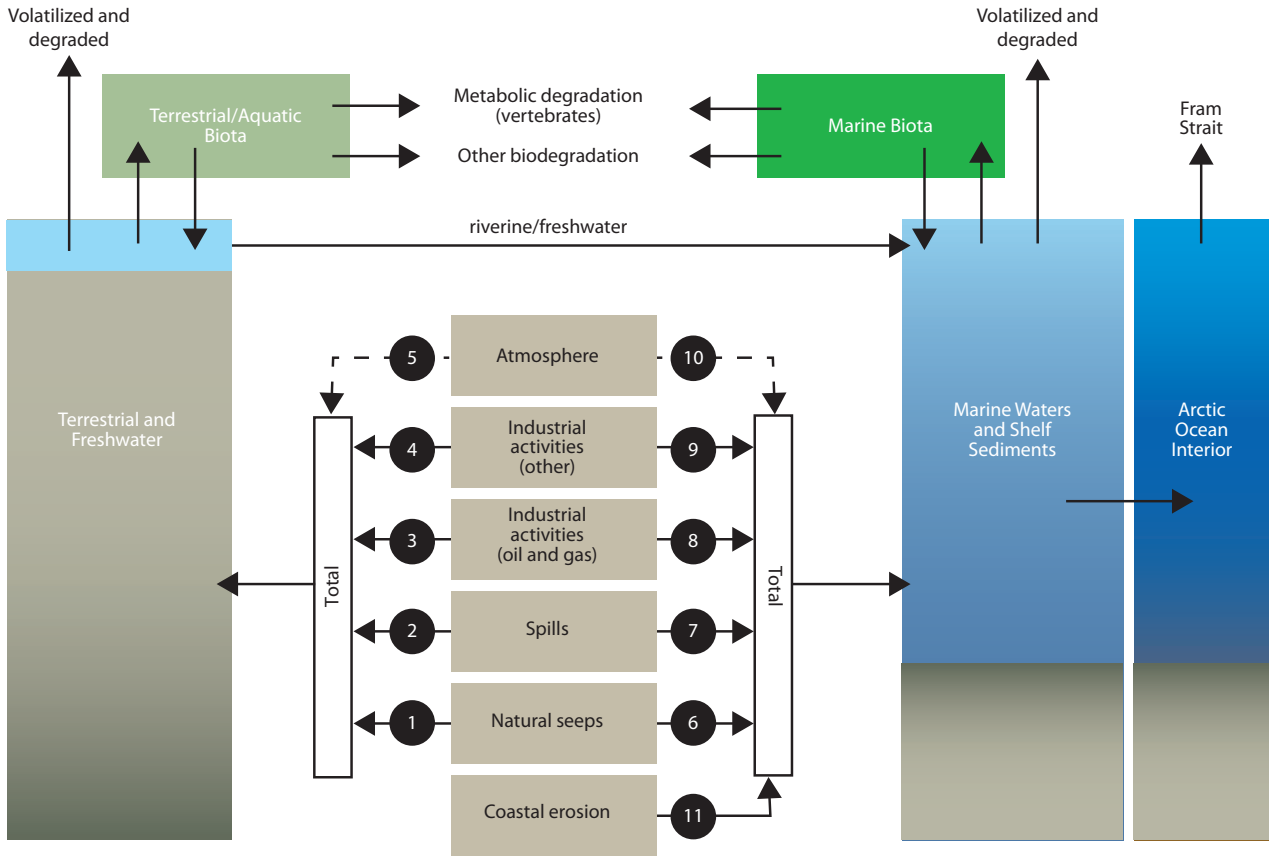


Figure 4.15. A schematic for the Arctic petroleum hydrocarbon budget.

Comparability of data is also limited because of the use of different sample methods for collection and analysis, which can lead to differences of an order of magnitude or more in the reported results; non-standard analyte lists which can lead to reported 'total concentrations' being made up of different individual compounds; lack of supporting quality assurance and quality control documentation; and use of non-specific detection.

Wherever possible, specific data were used to construct the budget. Where such data were not available less specific data were used; consequently, the values in the budget are a blend of data for which the source can be identified and data for which it cannot. It is judged that the values shown are an over-estimate for petroleum hydrocarbons.

Another limitation is the scarcity of data for several environmental compartments and pathways. Important data gaps include n-alkane and PAH assemblages in biota, seeps, spills and freshwater inputs.

- Biota. There are almost no data for n-alkanes and PAHs in plankton, zooplankton, fish and higher vertebrates. And available data are not circumpolar.
- Seeps. There are no quantitative data available that identify the number of seeps and their locations. Information is largely anecdotal.
- Spills. There are limited data on chronic spills from pipelines and terrestrial and marine transport. Many spills, particularly small spills, go unreported or occur in unregulated situations. Cumulative quantities from small and unreported spills could exceed the volume of reported spills.
- Freshwater inputs. Except for the Mackenzie River and several other north-flowing rivers in the Canadian Arctic, there are almost no data for n-alkanes and PAH assemblages in water and suspended particulate matter. This is the case for Russian, Alaskan, Norwegian and Icelandic rivers, and for Greenland runoff and icebergs.

There are several reasons for the lack of data. Firstly, researchers have generally preferred to focus their attention on the highly toxic, persistent and bioaccumulative organochlorine compounds for which there is a comprehensive temporal and spatial database for all environmental compartments in the Arctic. When measurements of compounds in petroleum are made, they generally focus disproportionately on benzo[a]pyrene, which is ubiquitous in the environment and a suspected animal carcinogen. Relatively few measurements of PAH assemblages are made in vertebrates because vertebrates can effectively metabolise PAHs. In addition, sampling large biota such as whales often presents ethical, legal and logistical constraints; limited data are generated on an opportunistic basis, usually from biopsy samples or samples collected from recently dead animals.

In effect, the detailed study of petroleum hydrocarbons in the Arctic environment is in its infancy.

Overall, the budget presented here is a first approximation. With better data and filled data gaps successive budgets will improve. Nonetheless, this budget provides some insight into the relative magnitude of the various inputs of petroleum hydrocarbons to the Arctic environment and the contribution that oil and gas activities in the Arctic make relative to natural and other anthropogenic inputs.

4.5.2. Pathways

4.5.2.1. Riverine/freshwater input

There are multiple inputs of n-alkanes and PAHs to the Arctic environment – some natural and some anthropogenic. And there are a number of diagnostic tools that allow various sources of hydrocarbons (i.e., petrogenic, pyrogenic, diagenetic and biogenic) to be identified and perhaps even quantified. The main requirement is detailed compound-specific analytical data. For the Arctic, this is generally not the case. Nonetheless, this budget (Table 4.25) attempts an estimate of the input of natural petroleum hydrocarbons from river/freshwater runoff into the Arctic Ocean. This largely addresses the issue of petroleum hydrocarbon seeps.

4.5.2.1.1. Alaska, USA

No data were found for n-alkanes and parent/alkyl PAHs in water or suspended particulates for Alaskan rivers. The U.S. Geological Survey (2003) did, however, provide data for the concentrations of parent and alkyl PAHs in sediment cores collected from five lakes (Stickwan, Nolan Creek, Little Medicine, Little Coal and Russian Mission Lakes) in the Yukon River basin. The data clearly show the presence of petrogenic hydrocarbons. Concentrations along the cores suggest a regional phenomenon that has been consistent for at least decades. Consequently, it is concluded that the Yukon River and, by extension, the Alaskan rivers (e.g., Colville) flowing into the Arctic Ocean are analogous to the Mackenzie River for the calculation of petrogenic hydrocarbons (Table 4.25). No estimate has been made for non-petrogenic n-alkanes or PAHs in Alaskan rivers.

4.5.2.1.2. Canada

The Mackenzie River is a good choice for attempting to estimate the quantity of seeped oil in the river discharge because this river drains a large watershed known to be rich in petroleum resources and within which oil seeps have been reported. Also, the watershed is sparsely populated and only moderately industrialized thereby limiting the confounding factor of large local sources of PAHs. Most importantly, however, the data reported by Yunker et al. (2002) for the concentrations of PAH and alkanes in ten relatively small Canadian Arctic rivers provide a means of inferring the contribution of oil seeps to the hydrocarbon loading of the Mackenzie River. Their study showed that the hydrocarbon signatures in the ten small rivers which discharge into Canadian Archipelago waters or Hudson Bay and which together account for about a third of the water discharge from the Mackenzie River, were fundamentally different from those of the Mackenzie River.

The hydrocarbons in the Mackenzie River discharge indicate a predominantly petrogenic source. Specifically, alkylated PAHs (particularly 2- and 3-ring PAHs) exceed the concentrations of parent PAH; alkane profiles lack a distinctive odd-even predominance; and the thermodynamic PAH isomers occur in greater abundance than the less stable kinetic PAH isomers. There is also evidence of a consistent source of 'fresh' petroleum; specifically, (1) naphthalene/phenanthrene ratios of 2.5 for suspended river particulates versus 0.8 for nearshore sediments; (2) pristine/phytane ratios of 0.91 for suspended river particulates versus 1.5 for nearshore sediments; and (3) relatively high naphthalene concentrations in river particulates relative to those in nearshore sediment. Furthermore, PAH ratios and the distributions of PAHs are remarkably similar from year to year in the Mackenzie Delta region based on the analysis of sediment cores from delta lakes. This suggests that regional processes and sources unaffected by anthropogenic activities dominate the source of hydrocarbons. There is no evidence of petroleum inputs from the oil production activities at Norman Wells.

Conversely, the hydrocarbon data from the ten smaller rivers indicate a non-petrogenic source. For example, alkylated PAHs are very minor relative to the parent PAHs in the suspended sediments; alkane profiles are dominated by odd carbon alkanes indicative of a terrestrial plant source; and nC_{17} is often the most dominant alkane (algal source). These data are consistent with the main source of hydrocarbons being detritus from the watershed of rivers draining tundra landscape.

Thus, it is reasonable to assume that the data for the ten small rivers provide a relatively good approximation for the background level of anthropogenic and combustion hydrocarbons in the Canadian Arctic. Therefore, if the concentrations of the predominantly non-petrogenic PAHs are subtracted from the Mackenzie River values, then the result should be a rough estimate of the PAH load of the Mackenzie River derived from petroleum seeps and river bed erosion of hydrocarbon-rich substrates, including bitumens. Using this approach, petroleum seeps account yearly for approximately 16 t of dissolved PAHs and 286 t of particulate PAHs in the Mackenzie River discharge.

Unfortunately, the approach used for PAHs is not valid for the alkane fraction. This is because, while the PAHs are relatively refractory and therefore quasi-conservative, the same cannot be said for the alkanes. Alkane concentrations are highly variable. Large variances can be accounted for merely on the basis of detrital matter, which can contain concentrations of alkanes in the percent range. And there is no systematic relationship between alkanes and PAHs in the available data. But there is a relationship between the relative amounts of PAHs and alkanes in particular types of petroleum. Using an average for the n-alkane and PAH content of petroleum derived from the composition of four types of oil (Prudhoe Bay, South Louisiana, Kuwait and Alberta sweet mixed blend #5; NAS, 1985; Environment Canada, 2006), the amount of n-alkanes in the range nC_{11}–nC_{34} will be about 25% higher than the amount of PAH. On this basis, petroleum seeps on a yearly basis can be estimated to account for around 20 t of dissolved alkanes and 358 t of particulate alkanes in the Mackenzie River discharge.

The inferred total flux of petrogenic PAHs and alkanes from the Mackenzie River is therefore about 670 t/yr. This is equivalent to about 6700 t of crude oil and represents about 1% of the 'best estimate' of worldwide annual rates of natural seepage, which seems reasonable and not excessive, given the vast quantities of petroleum known to exist within the Mackenzie River drainage basin.

4.5.2.1.3. Greenland

Freshwater input to the sea from Greenland is large (Table 4.25) and complicated. Runoff from Greenland is approximately 295 km^3 per year (Box and Bromwich, 2004). It can vary significantly due to volcanic activity (Hanna et al., 2005). Iceberg discharge is estimated at 239 km^3 per year (Reeh et al., 1999), and has been increasing in recent years. There are no published data for petroleum hydrocarbons in freshwater or suspended particulate matter. Using background concentrations for PAHs in snow (50 ng/L) then the estimated total PAH load entering the sea corresponds to 226 t/yr. Assuming that 50% remains in the Arctic region (Baffin Bay

Table 4.25. Arctic petroleum hydrocarbon budget – freshwater inputs.

River	Annual water discharge, km^3/yr	Annual sediment load, 10^6 t/yr	PAHs				Alkanes				Totals		Petrogenic hydrocarbon input				Non-petrogenic input		
			Average water conc., ng/L	Average suspended sediment conc., µg/g	Water flux, t/yr	Suspended sediment flux, t/yr	Average water conc., ng/L	Average suspended sediment conc., µg/g	Water flux, t/yr	Suspended sediment flux, t/yr	PAH flux, t/yr	Alkane flux, t/yr	Hydrocarbon flux, t/yr	PAH, t/yr	Alkanes, t/yr	Total, t/yr	PAH, t/yr	Alkanes, t/yr	Total, t/yr
Alaska																			
Yukon	210	60	?	?	?	69	?	?	?	?	?	?	?	69	86	155	?	?	?
Colville and others	?	33	?	?	?	38	?	?	?	?	?	?	?	38	48	86	?	?	?
Canada																			
Mackenzie	330	124	P: 11 A: 52 T: 63	P: 1.0 A: 1.4 T: 2.4	3.6 17 20.6	124 174 299	110	10	36	1240	319	1276	1595	300	374	674	20	902	922
Ten smaller Arctic rivers	113	0.42	P: 3 A: 19 T: 22	P: 0.093 A: 0.0015 T: 0.095	0.34 2.15 2.49	0.04 0.015 0.055	77	20	8.8	8.4	2.55	17.2	19.75	t/n	t/n	t/n	2.55	17.2	19.8
Greenland																			
Runoff	295	26	50	0.115 [a]	15	3	?	?	?	?	18	?	?	t/n	t/n	t/n	?	?	?
Icebergs	239	50–500	50	0.115 [a]	12	32	?	?	?	?	?	?	?	t/n	t/n	t/n	?	?	?
Iceland																			
Misc. north flowing rivers	?	18	?	?	?	?	?	?	?	?	?	?	?	t/n	t/n	t/n	?	?	?
Norway																			
Rivers flowing to Arctic waters	70	2.6	?	0.0037[b]	?	0.01	?	?	?	?	?	?	?	t/n	t/n	t/n	?	?	?
Svalbard	?	16	?	9 [b]	?	14.4	?	?	?	180	144	180	324	144	180	324	?	?	?
Russia[c]																			
Yenisey	620	4.7	3 [d]	T: 0.21	1.9	1.0	77 [d]	20 [d]	48	94	2.9	142	144.9	t/n	t/n	t/n	2.9	142	144.9
Lena	523	20.7	3 [d]	0.10 [d]	1.6	2.1	77 [d]	10 [d]	40	21	3.7	61	64.7	t/n	t/n	t/n	3.7	61	64.7
Ob	404	15.5	3 [d]	T: 0.012	1.2	0.19	77 [d]	20 [d]	31	310	1.4	341	342.4	t/n	t/n	t/n	1.4	341	342.4
Pechora	131	9.4	3 [d]	0.10 [d]	0.4	0.9	77 [d]	20 [d]	10	188	1.3	198	199.3	t/n	t/n	t/n	1.3	198	199.3
Kolyma	122	10.1	3 [d]	0.10 [d]	0.4	1.0	77 [d]	10 [d]	9	101	1.4	110	111.4	t/n	t/n	t/n	1.4	110	111.4
N. Dvina	110	4.1	3 [d]	0.10 [d]	0.3	0.04	77 [d]	20 [d]	8	82	0.34	88	88.3	t/n	t/n	t/n	0.34	88	88.3
Pyasina	86	3.4	3 [d]	0.10 [d]	0.3	0.03	77 [d]	20 [d]	7	68	0.33	75	75.3	t/n	t/n	t/n	0.33	75	75.3
Other smaller rivers (>20)	898	34.5	3 [d]	0.10 [d]	2.8	3.45	77 [d]	20 [d]	69	690	6.25	759	765.3	tr/neg	tr/neg	tr/neg	6.25	759	765.3
													Totals	551	688	1239	40	2693	2733

P: parent PAH; A: alkylated PAH; T: total; t/n: trace/negligible indicates that the rivers do not show a petrogenic signature for PAHs. Inputs estimated to <10 tonnes per year; [a] As for '[b]' an average for freshwater sediments is used; [b] In the absence of data for direct measurements of hydrocarbons in suspended sediments, an average for uncontaminated nearshore sediments is used; [c] Russian rivers do provide inputs of crude and refined oil products to the Arctic Seas. The greatest part of it appears to come from a large number of industrial and human activities. Therefore, in this budget these inputs are accounted for under the 'spills' pathway; [d] These are extrapolated values because specific Russian data are not yet available. Based on published data for the Ob and Yenisey Rivers, Russian rivers appear to have relatively low concentrations of parent PAHs, and alkyl PAHs are almost absent (Fernandes and Sicre, 1999). Discharge of particulates appears to be similar to that for small Canadian Arctic rivers. Consequently, values reported for 10 smaller rivers in the tables have been used. The exception is the average n-alkane suspended sediment concentration for the Lena and Kolyma Rivers where values for the Mackenzie River are used.

and Davis Strait) and that the other half is exported southward by the East Greenland Current, then approximately 14 t of PAHs will enter the Arctic. No measurements have been made on the n-alkane content of Greenland precipitation or glacial ice, so no estimate is made here for the input of n-alkanes.

Glacial erosion is significant (Hallet et al., 1996; Hasholt, 1996). The annual sediment flux is estimated at 76–526 million t, which includes 50–500 million t from calved icebergs, and 26 million t from glacial meltwater and surging glaciers in north and east Greenland. Floating icebergs can contain up to 160 g of sediment per litre (Hasholt et al., 2005) but the previous estimate of 50–500 million t is based on an average sediment content of only 0.1–0.01% by volume, which was thought by Hasholt et al. (2005) to be more realistic. The concentration of petroleum hydrocarbons in the sediment is thought to be low. This is because the profile of PAHs in coastal sediments has a strong pyrogenic signature and it is assumed that the particulates in glacial ice (the source of these sediments) are derived from long-range transport of pyrogenic hydrocarbons. Hence the input of petrogenic hydrocarbons (Table 4.25) is negligible. The non-petrogenic estimate is based on a PAH content of coastal sediments of 115 ng/g.

4.5.2.1.4. Iceland

Icelandic glacial rivers have very high sediment loads (Thomasson, 1991) because the rivers erode loose, largely volcanic ashes. Volcanic ashes are known to contain a variety of hydrocarbons, including PAHs; some of the PAHs are derived from the volcanic activity while others are from the scavenging of atmospheric PAHs by the ash particles. The PAH assemblages are enriched with parent naphthalenes, phenanthrene and fluorene (Chernova et al., 2001) and thus are likely to be pyrogenic in origin; virtually no data are available for PAHs or petroleum hydrocarbons in the Icelandic environment. For the budget, it is assumed that there is a negligible contribution of petroleum hydrocarbons. Owing to the absence of data no estimate is made for the input of petroleum hydrocarbons associated with the annual sediment transport of 18 million t to the northern coastal areas of Iceland.

4.5.2.1.5. Norway

There are few northward flowing rivers of mainland Norway that discharge into the Arctic Ocean and the total annual sediment discharge is low (2.6 million t). Of this total, 1.1 million t comes from glaciers with the remainder from areas classified as forests. There are no specific data for n-alkanes or parent/alkyl PAHs in river discharges. No petrogenic signal is expected; the non-petrogenic signal should be dominated by higher plant n-alkanes and combustion PAHs. As there are no data available, no estimate is made for the input of petroleum hydrocarbons from mainland Norwegian rivers, but it should be negligible. Based on average coastal sediment PAH concentrations of 37 ng/g (AMAP, 1998), the suspended sediment flux of non-petrogenic PAHs could be a negligible 0.01 t/yr.

Svalbard, in contrast, is an area enriched with petroleum hydrocarbons. The first petroleum seeps were identified by geological surveys as early as 1926. The first oil wells were then drilled on these natural oil seeps. The area is also known for other carboniferous deposits such as coal and peat. The annual suspended sediment load is 16 million t. There are no available data for runoff water concentrations or particulate concentrations of petroleum hydrocarbons. However, the nearshore sediments are enriched with PAHs (8700 ± 5200 ng/g total PAHs; Dahle et al., 2006a) having a high FFPI value (76%). Concentrations are an order of magnitude or more greater than those measured in sediments on the Mackenzie River shelf (Yunker et al., 1996); an area known to receive large quantities of petroleum hydrocarbons. For the budget, the input of hydrocarbons around Svalbard is taken as being substantially petrogenic. It is unclear, however, whether all the hydrocarbons are dominated by natural seeps or by other processes such as erosion of coal deposits, peat or other hydrocarbon sources. Data may exist for a detailed analysis of the hydrocarbon geochemistry of this area, but were unavailable for the budget. The petrogenic input is estimated at 324 t/yr. No estimate is made for the non-petrogenic input.

4.5.2.1.6. Russia

There appear to be no data for dissolved and particulate n-alkanes or parent/alkyl PAHs in rivers upstream from industrial activities. Also, there seems to be evidence from the coastal environment that suggests a very limited petrogenic input. Fernandes and Sicre (1999) for example, showed that sediments offshore of the Ob and Yenisey rivers have prominent anthropogenic and diagenetic signatures. Other evidence that a petrogenic source is weak include: low PAH concentrations together with a low unresolved complex mixture in the hydrocarbon fraction; alkyl phenanthrenes are almost undetectable; the dominant phenanthrenes in suspended particulate matter are probably diagenetic; in the Ob and Yenisey riverine zones, combustion-derived benzofluoranthenes dominate; and the ratios of kinetic/thermodynamic PAH isomers indicate a strong combustion source.

For the budget, it is concluded that the Russian rivers therefore provide negligible natural or seep petroleum hydrocarbons to the Arctic Ocean. In general, the river loads are considered non-petrogenic and without any actual data for the Russian rivers, the input calculations are based on concentrations of relevant dissolved and particulate alkanes and PAHs in the ten small rivers of Arctic Canada (Yunker et al., 2002). The exception is the Lena and Kolyma rivers where the average n-alkane suspended sediment concentrations for the Mackenzie River are used. The petroleum hydrocarbons delivered to the Arctic Ocean by Russian rivers are accounted for in the spill category.

It is surprising that for rivers which flow through vast watersheds containing rich reserves of petroleum hydrocarbons there is not a stronger presence of petroleum in the river water and suspended particulates. It is interesting to note that human activities have significantly altered the riverine outputs of certain Russian rivers. For example, the yearly sediment load of the Yenisey River has decreased from 12.4 to 5.6 million t due to dam construction (Hasholt et al., 2005), and the sediment load of the Kolyma River has doubled in recent years due to gold mining in the catchment area (Bobrovitskaya et al., 2003).

4.5.2.2. Petroleum inputs from natural seeps

Natural petroleum seeps have been noted in the Arctic for centuries. Although there are many reports of seeps around the world, including locations along Arctic coastlines, coastal waters, rivers and lands, direct measurements of seepage volumes and rates are extremely rare. NAS (2003), using extremely broad extrapolations of limited qualitative and semi-quantitative data, estimated the annual global oil seepage rate to the oceans to be within the range 0.2–2 x 10^6 t with a 'best' estimate of 0.6 x 10^6 t. There is no way of knowing how realistic this estimate is. A

corresponding estimate for petroleum seeps on land was not attempted. Within the estimate of 0.6 x 10^6 t/yr for the seepage of petroleum into the global marine environment, NAS (2003) estimated that the annual oil seepage rate for north Alaskan coastal waters was 0.02 x 10^6 t, which seems high given that (a) on a proportional area basis the entire Arctic would receive 2.8% of the global seepage and the 0.02 x 10^6 t/yr amounts to 3.3%; (b) other significant seeps are known in the Arctic (e.g., a contribution from the Mackenzie River); and (c) the huge oil-rich continental shelves of Russia are likely to provide substantial quantities of seeped oil.

For the budget, two very broad assumptions are made: specifically, that (1) total oil seepage into the Arctic Ocean is 0.04 x 10^6 t/yr, or twice the estimate for Alaska alone (this accepts the NAS (2003) estimates as being 'reasonable'); and (2) that the amount of seep oil per year on land is approximately the same as that for the marine environment (which seems reasonable given the vast petroleum reserves in Arctic Russia). On this basis, the amount of petrogenic n-alkanes seeped into the Arctic Ocean and Arctic lands would be 2300 t/yr, and the amount of petrogenic PAHs 1890 t/yr (comprising 190 t parent PAHs, and 1700 t alkylated PAHs).

4.5.2.3. Petroleum hydrocarbon spills

Spills of crude oil and refined products are chronic occurrences in the circumpolar Arctic. Prominent sources are the marine and terrestrial transportation chain, including marine tankers, railway tank cars and river/canal barges; pipelines; and collectively, a wide variety of point sources ranging from leaks at tank farms, fuel transfer points and business/residential sites. Unlike other input pathways, spills of petroleum hydrocarbons are unequivocally petrogenic hydrocarbons.

Databases on spilled oil are generally incomplete and inconsistent, often containing highly subjective assessments of oil spill volumes. Most report only those spills exceeding a certain minimum size occurring at regulated facilities and operations. Yet it is thought that the cumulative amount of spills occurring at unregulated facilities and activities together with spills that go unreported can exceed the amounts reported in the databases.

Table 4.26. Estimates of the average annual quantity (t/yr) of petroleum hydrocarbons spilled in the Arctic.

Country	Type of petroleum hydrocarbons [a,b,c]	Oil spilled in the marine environment			Oil spilled in the terrestrial/aquatic environment				Spills associated with oil exploration and production activities	
		tankers	pipelines	chronic/ incidental	pipelines	rail	barge	chronic/ incidental	marine	terrestrial
Alaska, USA [d]	as oil	n/a	-	-	468	-	-	588	-	98
	n-alkanes	n/a	-	-	27	-	-	35	-	6
	PAH	n/a	-	-	22	-	-	28	-	5
Canada	as oil	n/a	-	10[e]	7	n/a	-	76	<1	23
	n-alkanes	n/a	-	0.6	0.4	n/a	-	5	<1	1.4
	PAH	n/a	-	0.5	0.3	n/a	-	4	<1	1.1
Greenland	as oil	2[f]	-	-	n/a	n/a	n/a	-	n/a	n/a
	n-alkanes	0.1	-	-	n/a	n/a	n/a	-	n/a	n/a
	PAH	0.1	-	-	n/a	n/a	n/a	-	n/a	n/a
Iceland	as oil	85[g]	-	-	-	n/a	n/a	-	n/a	n/a
	n-alkanes	5	-	-	-	n/a	n/a	-	n/a	n/a
	PAH	4	-	-	-	n/a	n/a	-	n/a	n/a
Norway	as oil	combined	-	-	-	n/a	n/a	-	-	-
	n-alkanes	with Russia[h]	-	-	-	n/a	n/a	-	-	-
	PAH		-	-	-	n/a	n/a	-	-	-
Sweden	as oil	n/a	-	-	-	-	-	-	n/a	n/a
	n-alkanes	n/a	-	-	-	-	-	-	n/a	n/a
	PAH	n/a	-	-	-	-	-	-	n/a	n/a
Finland	as oil	n/a	-	-	-	-	-	-	n/a	n/a
	n-alkanes	n/a	-	-	-	-	-	-	n/a	n/a
	PAH	n/a	-	-	-	-	-	-	n/a	n/a
Russia	as oil	2000[j]	-	-	2200	550	715	-	-	-
	n-alkanes	118	-	-	130	32	42	-	-	-
	PAH	94	-	-	103	26	34	-	-	-
Totals	as oil	2087	-	10	2675	550	715	-	= 6037	
	n-alkanes	123	-	0.6	157	32	42	-	= 355	
	PAH	98	-	0.5	125	26	34	-	= 285	

-: No reliable data could be found or none exist; n/a: not applicable; the activity/mode of transport does not occur or is not significant; [a] oil means crude oil or refined products (e.g., fuel oil, diesel, gasoline, etc.); [b] n-alkanes are taken as comprising 5.8% of crude oil; [c] PAHs are taken as comprising 4.7% of crude oil; [d] US spill data from Alaska Oil Spills Database (www.dec.state.ak.us/spar/perp/search/search.asp) and MMS (2000); [e] Canadian spill data from Environment Canada (2006); [f] a spill rate of 100 ppm is applied to annual oil imports to Greenland; [g] a spill rate of 100 ppm is applied to annual oil imports to Iceland; [h] As transport along the Norwegian coast originated in Arctic Russia, the data are accounted for in the Russian value; [j] spill rate is 200 ppm of oil shipped, shipped amounts from Bambulyak and Frantzen (2005) and Frantzen and Bambulyak (2003).

The average amounts of oil hydrocarbons from various sources estimated to be entering the Arctic environment on an annual basis are shown in Table 4.26. It must be emphasized that the amounts shown are only rough estimates because for most entries there are varying degrees of uncertainty (sometimes significant) in the values for amount shipped and average loss rates. There are three important points concerning how the values in Table 4.26 were calculated.

1. The data in the table are not synoptic. They come from many sources which represent long-running annual averages (sometimes highly condensed), or specific data for one or more of the past ten years. Consequently, the data in the table do not represent amounts of oil actually spilled in the Arctic in any given year or that ever might get spilled in any given year.

2. Some estimates are based on extrapolating transported volumes of oil using an average spill loss rate. For example, ESMAP (2003) suggested that the average loss from pipelines in Russia is 55 ppm of the oil shipped/transported. But the rate depends on a number of factors such as diameter of pipe, length of pipeline, age of the pipe and peripherals, and maintenance and monitoring of the pipeline system. In the absence of information required to judge the efficacy of using 55 ppm for the spill rate of oil from pipelines in the Russian Arctic, the rate of 55 ppm was simply adopted for the budget. Furthermore, this average spill rate was also applied to spill losses from railway tank cars and barges in the Russian Arctic for lack of a better value.

3. For tanker transport, the spill loss rate has been declining significantly in recent years. During the 1980s the rate was approximately 1000 ppm of the oil shipped (NAS, 1985). More recent global estimates are in the range of a few hundreds of ppm. The decline in spill rate is attributed to more modern equipment and a heightened awareness of the role of effective management practices in minimizing spills. The spill loss rate should continue to decline in the future as twin-hulled ships are introduced. As with the pipeline estimates, the loss rate for tankers depends on many factors, most under human control, such as age and maintenance of the fleet, adequate use of monitoring systems, compliance with IMO (International Maritime Organization) regulations, and diligent management of loading/offloading procedures and operational discharges. In the budget an average spill loss rate of 100 ppm is used for all tanker estimates except Russia. This is based partly on the performance of Norwegian shipping of oil from the Norwegian Sea by tanker, which employs modern equipment and adheres strictly to observing international regulations. Using the data provided by Bambulyak and Frantzen (2007), Norway's oil spill loss rate at sea is approximately 100 ppm. For Russia, which has a less modern fleet of tankers a spill loss rate of 200 ppm is used in the budget; further justification for using 200 ppm is the complicated chain of transfers that often take place in shipping oil out of the Russian Arctic. For example, in some cases oil starts its journey in a pipeline only to be off-loaded to barges, and then taken to a location where it can be pumped into a second pipeline, which takes it to a coastal oil terminal. There, oil is offloaded to storage tanks or loaded onto small and medium-capacity sea-going tankers. These tankers deliver the oil to another larger loading facility where the oil can be offloaded to storage tanks or placed into pipelines and pumped onto much larger tankers which carry the cargo to its final destination – export markets or refineries in Russia. The transportation infrastructure in northern Russia may remain underdeveloped and inadequate for the safe transport of oil out of the region.

Finally, the lack of data for many of the entries in Table 4.26 is probably not significant to the overall spill input in the budget because the magnitude of these individual inputs is small relative to those for the main Russian inputs, with the possible exception of Russian marine pipeline spills (for which there are no data).

4.5.2.4. Inputs of n-alkanes and PAHs from oil exploration and production

The main routine activities that have the potential to generate n-alkanes and PAHs in the oil exploration and production industry include drilling (exploration, delineation, production – which includes the disposal of drilling muds and cuttings, flaring, and the disposal of produced water) and refinery operations. Inputs from these sources are summarized in Table 4.27. Inputs from flaring and the disposal of produced water and drilling muds and cuttings require some additional description and this is given in the following sections.

4.5.2.4.1. Flaring

When oil is produced, the gases dissolved in the oil (solution gases) are released as the oil is brought to surface pressure. Flaring is the process by which the unwanted gases are disposed of by combustion in the open atmosphere. The solution gases are primarily methane, ethane and propane but can contain small amounts of butane, carbon dioxide, nitrogen, helium, and hydrogen sulphide. When the solution gases are burned, a number of combustion products are formed. In theory, complete combustion results in all of the hydrocarbons in solution gas being oxidized to carbon dioxide and water. In practice, the emissions from flaring depend on the composition of the waste gas and the efficiency of combustion. Factors that affect the efficiency of combustion include the flow rate of the flare gases, wind velocity, ambient turbulence, and the presence of hydrocarbon droplets or water droplets in the flare stream. The efficiency of combustion is almost never 100%; estimates vary from 20 to 99% (Kostiuk et al., 2004). As the conditions for high efficiency (>95%) burns are seldom achieved, a number of products of incomplete combustion are formed. These include unburned hydrocarbons (primarily C_1 to C_3 alkanes), volatile organic compounds, PAH (largely attached to soot particles), and trivial amounts of other organic compounds.

As flares do not lend themselves to conventional emission testing techniques, there are no reliable emission factors (EFs) for PAHs and n-alkanes from flares. Clearly the efficiency of combustion will vary enormously with time at a given flare site depending on the highly variable values of the many parameters affecting combustion. There are however, EFs for various engines that use natural gas as a fuel. For example, the EF for 16 USEPA PAHs (benz[*a*]anthracene, benzo[*a*]pyrene, benzo[*b*]fluoranthene, benzo[*k*]fluoranthene, chrysene, dibenz[*a,h*]anthracene, indeno[1,2,3-*cd*]pyrene, acenapthylene, anthracene, benzo[*g,h,i*]perylene, fluoranthene, fluorene, naphthalene, phenanthrene

Table 4.27. Inputs of n-alkanes and PAHs associated with oil extraction and production.

Category	Onshore, t/yr		Offshore, t/yr		Comments
	n-alkanes	PAH	n-alkanes	PAH	
Drilling					
Exploration/delineation wells	negligible	negligible	1	0.7	Average amount of petroleum hydrocarbon waste per well is 2 tonnes of which 6% is n-alkanes and 4% PAH (predominantly alkylated)
Production wells	negligible	negligible	2.9	1.9	
Produced water					
Norwegian Sea	-	-	11	9	See text, section 4.5.2.4
Russia	3	2.5			See text, section 4.5.2.4
Flaring					
Alaska	negligible	0.02	-	-	See text, section 4.5.2.4
Russia	negligible	1.7	-	-	See text, section 4.5.2.4
Refineries	?	16 (a) 40 (b)	0	0	Total refining capacity in the Arctic is 22 million t/yr. Emission Factor is 0.8 ppm of amount refined (USEPA, 1998): (a) from catalytic cracking (pyrolytic PAH); (b) from process other than catalytic cracking as naphthalene (USEPA, 1998)
Total	3	60.2	14.9	11.6	

and pyrene) from industrial natural gas combustion is 8.9 x 10^{-8} g PAH per m^3 burned. This EF however refers to a relatively efficient burn. For the budget calculation, an EF of 10^{-4} g PAH per m^3 was used to reflect a perceived 1000-fold less effective burn from flares compared to natural gas engines/turbines.

In the Arctic, flaring of waste gas occurs only in Russia and Alaska. In the Norwegian Sea gas flaring is not allowed and can occur only by approval of the Ministry in cases of normal operational safety and for short periods for the purpose of production testing. Waste gases are transported to shore for use as a fuel or in the manufacture of methanol. There is no oil production in the Canadian Arctic and thus no gas flaring.

By far the greatest amount of current flaring in the Arctic occurs in Russia, although the Russian Federation announced in 2007 that Russia will stop the flaring of associate gas during oil production. Estimates for the volume of flared gas in Russia (excluding Khanti-Mansiysk) vary from 8.5 billion m^3 (reported) to 25.8 billion m^3 (detected by satellite) (Elvidge et al., 2007). Part of the difference between the two estimates is that the reported values are for flaring associated with oil production only, whereas the satellite-derived estimate includes flaring from both oil production and condensate stripping activities, which can be very significant. Deriving an estimate of reliable flared gas volumes for the Russian Arctic is also complicated by the fact that data are not reported on a geographical basis.

For the budget calculation, an annual volume of gas flared in the Russian Arctic (all on land) is estimated conservatively to be the average of the 'reported' and 'satellite-derived' estimates, namely 17 billion m^3. Applying the EF of 10^{-4} g PAH emitted per m^3 gas burned, gives a total PAH emission onshore in Arctic Russia of 1.7 t/yr. The corresponding value for Alaska is 0.02 t/yr.

Emissions of n-alkanes (chain length $>C_{12}$) are estimated to be negligible as almost all alkanes in flare emissions are unburned gases in the C_1 to C_3 range. For example, the USEPA (1983) reported that methane, ethane/ethylene, acetylene, propane and propylene accounted for approximately 100% of the hydrocarbon composition of flare emissions.

4.5.2.4.2. Produced water

In the Arctic, produced water is discharged to the environment in the Norwegian Sea and in Russia. All, or almost all, produced water in Alaska is re-injected into the formation.

In Russia, virtually all current oil production is on land. There is no definitive compilation on how much produced water is discharged on land in Russia because data are not reported on a geographical basis, but rather on a corporate or producer basis. It is difficult to extrapolate from one field to another even when production volumes are known because the volume of produced water increases dramatically as reservoirs are depleted; in some cases 6 or 7 m^3 of water are produced for every m^3 of oil.

Gazprom reported that it discharged around 46 million m^3 of produced water to surface water bodies and surface relief in 2005. Using the estimate that about 25% of Russian Arctic oil production can be attributed to Gazprom and assuming that the other producers of oil in the Russian Arctic generate produced water on average at the same rate as Gazprom, then each year about 184 million m^3 of produced water are discharged to surface water bodies and surface relief. The oil producers indicate that prior to discharge the produced water is treated to comply with MPC (Maximum Permissible Concentration) values as outlined in Russian legislation.

The MPC value for oil and oily products (dissolved and emulsified) discharged into water bodies used by fisheries is 0.05 mg/L. For water bodies used for household water use and general use, the MPC is 0.1 mg/L for polysulphide oils and 0.3 mg/L for other oils. It is also known that from time to time the MPC may be exceeded due to accidents or upset conditions (e.g., mechanical failure of oily water separators causing them to exceed normal operating

Table 4.28. Inputs of n-alkanes and PAHs from industrial activities other than oil exploration and production.

Source	Amount combusted	Emission factor	Petrogenic input, t/yr			Non-petrogenic input, t/yr		
			n-alkanes	PAH	Alkyl PAH	n-alkanes	PAH	Alkyl PAH
Combustion of fuel oil/diesel	11.9 x 10^6 t/yr	3.3 ppm [a]	-	-	-	-	39	-
Contribution from unburned portion		200 ppm	213	30	184	-	-	-
Combustion of natural gas	2.7 x 10^{10} m^3	0.0107 kg/ 10^6 m^3 [b]	-	-	-	-	0.3	-
Combustion of bituminous coal	3.7 x 10^6 t/yr	0.91ppm [c]	-	-	-	-	3.4	-
Aluminium smelting	7.13 x 10^4 t/yr [c]	150 ppm[d]	-	-	-	-	10.6	-
Total			213	30	184	-	53.3	

[a] The emission factor for the combustion of fuel oil/diesel is from the USEPA (1998) while the emission factor for the contribution from the unburned portion is from www.me.queensu.ca/courses/MECH435/7; [b] emission factor from Lobscheid and McKone (2004); [c] the amount of aluminium smelted is from www.rual.ru; [d] Emission factor from USEPA (1998).

efficiencies); if so, this results in a fine for the polluter. For the budget calculation, an MPC of 0.3 mg/L was applied.

Using these values, the total oil discharged to surface water bodies and surface relief from produced water is 5.5 t/yr; corresponding to 3 t of n-alkanes and 2.5 t PAH per year.

For the Norwegian Sea, the Norwegian State Pollution Control Authority reported that around 196 t of dispersed oil were discharged to sea in 2006 at six producing fields; corresponding to around 11 t of n-alkanes and 9 t of PAH per year.

4.5.2.4.3. Drilling muds and cuttings

During the period 2003 to 2007, offshore exploration drilling in the Arctic occurred only in the Norwegian Sea, Barents/Pechora Sea and Canadian Beaufort Sea, and offshore production wells were drilled only in the Norwegian Sea and Barents/Pechora Sea. Using data provided by the Norwegian Petroleum Directorate and the National Energy Board of Canada, the annual average was 9 exploration/delineation wells and 24 production wells. For the budget, it was assumed that most of the drilling muds and cuttings generated by these activities was discharged to sea.

Drilling on Arctic lands during this period is estimated to have exceeded 100 wells per year. Input of drilling muds and cuttings to the Arctic terrestrial environment from these activities is considered negligible because most of the wastes are either re-injected downhole, treated and removed, or confined in sumps.

4.5.2.5. Inputs of n-alkanes and PAHs from industrial activities other than oil exploration and production

Inputs of n-alkanes and PAHs from industrial activities other than oil exploration and production are given in Table 4.28. The most prominent sources include combustion of fuel oil, diesel, gasoline and other refined petroleum fuels, to produce power and heat and to fuel engines, and aluminium smelting which is a well known source of PAHs.

For the budget, the total annual amount of oil fuels combusted is considered to be combusted in engines and to be quantitatively equal to the amounts of oil fuels consumed in the Arctic annually, as shown in Table 4.29.

The entries in Table 4.28 are calculated by multiplying the amount combusted by the empirically derived emission factor. In the case of the combustion of oil fuels, both pyrogenic and petrogenic hydrocarbons are emitted to the environment. When oil fuels are burned, virtually all of the n-alkanes and most of the PAHs are decomposed during the combustion process (Westerholm et al., 1988). The exhaust is rich in pyrolytic PAHs. For the budget, the PAHs exhausted from engines are accounted for as pyrogenic PAHs. The exhaust from engines also contains a small amount of unburned fuel. This amount is estimated to be 200 ppm (Mech 435 Applied Combustion Home Page) of which approximately 9% will be n-alkanes and 9% PAHs (alkyl plus parent PAHs). This is accounted for in Table 4.28 as 'contribution from unburned portion'.

Aluminium smelting provides a significant annual PAH input. The value in Table 4.28 is based on one aluminium smelter currently operating in Kandalaksha (near Murmansk). The input of PAHs from this source is expected

Table 4.29. Consumption of oil fuels in the Arctic.

Area	Per capita usage, t/yr	Estimated population in 2000	Total consumption, million t/yr
Arctic USA	9.0	480 000	4.32
Arctic Canada	7.2	100 000	0.72
Greenland	3.6	56 000	0.20
Iceland	2.9	290 000	0.84
Arctic Norway [a]	1.9	380 000	1.10
Arctic Sweden [a]	1.9	264 000	0.50
Arctic Finland [a]	1.9	200 000	0.38
Arctic Russia	1.9	2 000 000	3.80
Total			11.85

[a] no data for Arctic Finland, Sweden and Norway. Consumption assumed to be similar to Russia.

to increase in the near future as an additional production capacity of 300 000 t of aluminium per year is being added to the Kandalaksha smelter and a new aluminium smelter with a capacity of 346 000 t/yr is planned for Fjaroaal, Iceland.

4.5.2.6. Atmospheric inputs

The literature on atmospheric hydrocarbons in the Arctic is sparse. NAS (2003) made an estimate of the atmospheric deposition loadings (wet deposition plus dry aerosol deposition) of petroleum hydrocarbons to the global ocean. Simplifying assumptions include (a) uniform temperature (11°C), (b) uniform annual precipitation (100 cm/yr), (c) average depositional velocities, and (d) average aerosol particle sizes, across the global ocean. Petroleum hydrocarbons were defined as nC_{10}–nC_{33} n-alkanes plus 21 unsubstituted PAHs. The modelling did not take into account the source (pyrogenic versus petrogenic versus biogenic) of the hydrocarbons. The calculated loading to the World Ocean was 57 t/yr. This loading was totally overwhelmed by the calculated volatilization of 3 million t/yr of these compounds, indicating that the global ocean is a net source of hydrocarbons and that hydrocarbon degassing from coastal waters is a major geochemical process. In terms of petrogenic hydrocarbons the volatilization value is a gross overestimate because it ignores the massive quantity of biogenic n-alkanes produced in ocean surface waters. The modelling results showed that 27% of the global deposition occurred in U.S. coastal waters. Clearly, much of the global deposition occurs near locations where hydrocarbons are produced, refined and used; these include heavily industrialized zones.

For the Arctic budget, the global loading estimate is adjusted to account for (a) an additional 50% of the total deposition occurring in the coastal zone of industrialized countries other than the USA; (b) the deposition of the remainder for the proportional area of the Arctic Ocean in relation to that of the World Ocean (2.5%); and (c) to account for the average annual precipitation in the Arctic being 47 cm (ACIA, 2005), rather than 100 cm.

The resulting depositional loading of hydrocarbons to the Arctic marine environment is 92 t/yr and to the Arctic terrestrial environment is 62 t/yr (land represents 40% of the total area of the AMAP-defined Arctic).

One obvious source of non-petrogenic PAHs in the Arctic is forest fires. The 40-year average for the mass of wood consumed in forest fires in Arctic Canada is 25 000 t/yr (Amiro et al., 2001). Using an emission factor for PAH of 360 ppm (USEPA, 1998), this results in an annual PAH input of 9 t to the Canadian Arctic and 18 t to the Arctic as a whole (assuming that fires in Arctic Canada represent half of those occurring in the circumpolar Arctic on average).

4.5.2.7. Inputs from coastal erosion

Yunker et al. (1991) estimated the flux of alkanes and PAHs from shoreline erosion and coastal peat erosion in the Mackenzie River delta area of the Beaufort Sea. The estimate was based on a very detailed analysis of over 700 coastal segments for which coastal retreat rates and peat thickness were computed. The flux of n-alkanes and PAHs for shoreline erosion was estimated at 14.7 and 0.9 t/yr, respectively, and for peat erosion at 28.5 and 0.03 t/yr, respectively.

To extrapolate from the Mackenzie River delta area to the circumpolar Arctic it is assumed that inputs are located predominantly around river deltas and estuaries, and that proportionally similar amounts will occur at each estuary. Thus the total input of (non-petrogenic) n-alkanes and PAH hydrocarbons from coastal retreat will be 147 and 9 t/yr, respectively, and for peat erosion 285 and 0.3 t/yr, respectively.

4.5.3. Biota

A potentially significant removal pathway for petroleum hydrocarbons from the environment is biodegradation. Phytoplankton and bacteria can use petroleum as a carbon source. C_{10} to C_{22} n-alkanes are readily biodegraded; C_{23} to C_{34} n-alkanes generally resist biodegradation. Consequently, plankton-derived n-alkanes readily biodegrade whereas terrestrial plant-derived n-alkanes do not. Two- and three-ring PAHs are more easily degraded than PAHs containing more than three rings or those that are alkyl-substituted.

Experimental laboratory data suggest that microbial degradation of C_{10} to C_{22} n-alkanes is rapid (half-lives of days). For low molecular weight unsubstituted PAHs, half-lives are of the order of four days whereas half-lives for high molecular weight and alkyl-substituted PAHs are much longer. The actual rates of biodegradation are highly variable and depend on many factors including the particular strain of the bacteria population or phytoplankton species, temperature, the composition and concentration of hydrocarbons present in the substrate, and the form of the hydrocarbons (e.g., dissolved, particulate, colloidal), among others (Neff, 1979; NAS, 2003).

If the microbial degradation pathway was a significant process in the Arctic, then the concentrations of petroleum hydrocarbons along sediment cores would be expected to decrease with depth. This does not appear to be the case. Cores obtained from the coastal Beaufort Sea, Alaska, have total petroleum hydrocarbon and PAH profiles that are remarkably consistent with depth (MMS, 2005). Furthermore, if significant biodegradation was occurring in sediments, then an enrichment of the more refractory n-alkanes and PAHs would be expected with depth along sediment cores. The available data for n-alkanes and PAHs in sediment cores do not provide the detailed analytical results needed to determine whether such enrichment has taken place. It is possible also, that the rates of biodegradation in the Arctic environment are simply a lot slower than those measured in laboratory experiments. For the budget it is concluded, therefore, that biodegradation by bacteria is not significant.

Vertebrates (fish, birds and mammals) have a well-demonstrated ability to biodegrade petroleum hydrocarbons using the P450-dependent mixed-function oxidase system (Neff, 1979). Many benthic invertebrates can also metabolize petroleum hydrocarbons, but the rate is usually much slower than in vertebrates. Benthic invertebrates also readily depurate accumulated hydrocarbons largely with little alternation. They thus act as an effective recycling mechanism for petroleum hydrocarbons.

Few unequivocal species-specific measurements for the half-life of various PAHs have been reported. Also, some of the data are difficult to use because there is confusion as to whether the data correspond to metabolic biological half-life or depuration half-life.

It is assumed here that the average metabolic half-life for PAHs in vertebrates in the Arctic is 20 days (some compounds have a half-life of less than a week; for others the half-life is several tens of days). Using the 20-day half-life value and the standing stock values given in Table 4.30, vertebrate metabolism will remove 60 t/yr and 0.018 t/yr of PAHs

Table 4.30. Petroleum hydrocarbon standing stock in marine and terrestrial biota.

Trophic level	No. species	Estimated biomass, t	Source	Average concentration[b] TPH, μg/g ww	Source	Average concentration[b] ΣPAH[c], ng/g ww	Source	Total standing stock	
								TPH, t	ΣPAH, t
Marine									
Phytoplankton	-	1.5×10^9	Stein and McDonald (2004);	?		115	AMAP (1998)	?	175
Zooplankton	-	0.12×10^9	Stein and McDonald (2004);	?		195	AMAP (1998); RCMA (1995); Devon Canada Corporation (2003)	?	14
Benthos	-	6.8×10^8	Cushing and Walsh (1976); Dunbar (1977)	4	AMAP (1998); Devon Canada Corporation (2003); MMS (2005), RCMA (1995)	110	AMAP (1998); Devon Canada Corporation (2003); MMS (2005), RCMA (1995)	2700	74
Fish	13	1.0×10^7	Based on CAFF (2005)	13	AMAP (1998), US Dept Health (1999); Devon Canada Corporation (2003)	94	Eisler (1987); AMAP (1998); Devon Canada Corporation (2003)	130	0.9
Birds	6	0.15×10^5	Based on CAFF (2005)	?		18		?	0.0003
Mammals	11 (whales) 10 (seals) + walrus + polar bear	5.7×10^6	Based on CAFF (2005)	?		1000	Hellou et al. (1991); Marsili et al (2001)	?	5.7
Total marine								**2830**	**270**
Terrestrial									
Phytoplankton		?		?		?		?	?
Zooplankton		?		?		?		?	?
Benthos		?		?		?		?	?
Fish		?		?		18	RCMA (1995); AMAP (1998)	?	?
Birds	18 geese 37 shorebirds	2.4×10^4	Based on CAFF (2005)	?		19	RCMA (1995); AMAP (1998)	?	0.0005
Mammals	6	3.6×10^5	Based on CAFF (2005)	?		4	RCMA (1995); AMAP (1998)	?	0.0014
Total terrestrial								**?**	**0.0019**

[a] biomass estimate is based on census numbers for each species provided in CAFF (2001) multiplied by the average weight for individuals in each species; [b] whenever possible concentrations for muscle or blubber used; [c] ΣPAH usually does not include values for specific compounds. TPH: Total petroleum hydrocarbon.

from the Arctic marine and Arctic terrestrial environments, respectively. These estimates assume a steady-state for the concentrations of PAHs in the biota and ignore any effects related to the movements of species into and out of the AMAP Arctic region.

4.5.4. Projected oil and gas industry input to the overall budget at peak production

It is expected that oil and gas exploration, development, production and transportation will increase dramatically in the next few decades. It is also expected that most of this increase will occur in Russia where vast known reserves will be developed and new fields discovered. On the regional scale of the petroleum hydrocarbon budget, it is interesting to estimate what proportion of total inputs will occur when oil and gas production peaks in the Arctic.

This is a difficult task because of uncertainties in quantifying existing reserves and in predicting the pace of development of infrastructure and uncertainties regarding the success of finding new oil and gas fields, and establishing their size, lifetimes and annual rates of production. To produce an estimate, a number of assumptions have been made, based largely on information presented in Chapter 2 of this assessment and by Zhuravel and Mansurov (2005) and Bambulyak and Frantzen (2007), namely that:

- oil and gas production in the Arctic will peak around 2040 and that 80% of the activity will occur in the Russian Arctic;
- the number of wells drilled at the time of peak production will be 30 offshore and 20 onshore across the entire Arctic;
- oil production at the peak will total 140 million t and 190 million t in the offshore and onshore regions, respectively. This further assumes no production in Greenland or Iceland during the peak year;
- in the peak year, the total oil transported by tanker will be equal to the capacity of current and planned loading terminals in Russia (about 110 million t, including contributions from Varendej, Peschano-Ozerskii, Indiga, Kola Bay, Cape Mishukov, Murmansk, Mohnatkina Pakhta, Lavna Coastal, Vitino, Arkangelsk, Severdvinsk, Onega, Dudinka-Dikson, RPK Gulf of Ob, and the complex at Tiksi);
- the amount of oil transported by pipeline in the Arctic in the peak year will be the sum of onshore production in Russia and offshore/onshore production in Canada and the USA (190 million t);
- the rate of loss of oil from tankers and pipelines is 200 ppm and 55 ppm, respectively. These are probably liberal loss rates because they do not include any technological advances to tankers and pipelines which are inevitable and will reduce the rate of losses from tankers and pipelines in the future, particularly if the price of oil continues to increase;
- crude oil contains approximately 5.8% n-alkanes and 4.7% PAHs on average;
- the annual rate of natural inputs will remain constant; and that
- no adjustment has been made for the density of the various oils.

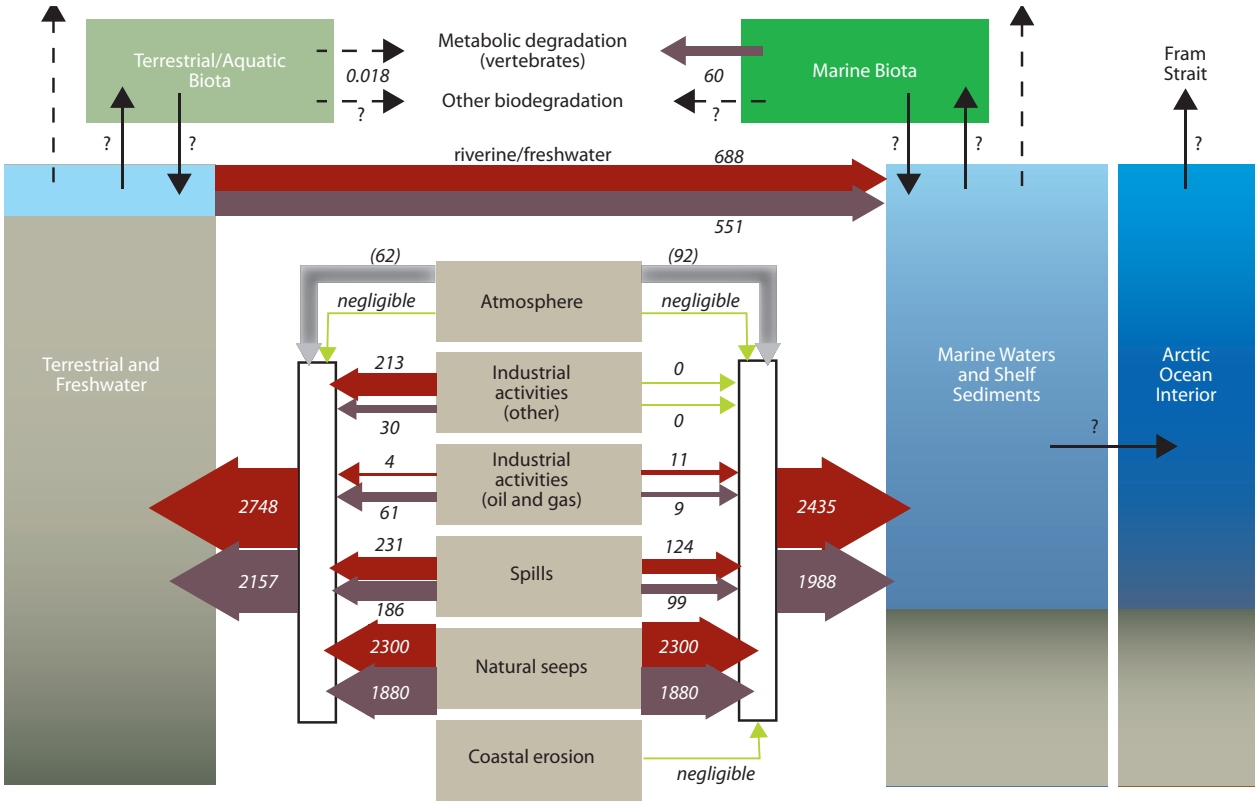


Figure 4.16. Arctic petroleum hydrocarbon budget (t/yr) showing quantified pathways to the extent possible.

The estimate for the total input to the Arctic environment of n-alkanes and PAHs from oil and gas activities including spills in the peak year is 5800 t or approximately 58% of the natural annual input. This compares to the current estimate of approximately 6%. Clearly, when the level of input from the oil and gas industry rivals the natural input on a regional scale there will undoubtedly be many site-specific concerns, particularly because in the years leading up to the peak year there will be significant and increasing chronic pollution issues. The ecological consequences of these inputs and other cumulative stresses, including climate change need to be addressed.

4.5.5. Budget summary

The schematic of the budget, completed to the extent currently possible, is shown in Figure 4.16. Summary input data are given in Table 4.31. In order to complete the budget, priority data gaps need to be filled. These include:

- data for the analyte list shown in Table 4.24 for:
  - wet and dry deposition;
  - dissolved and particulate concentrations in all Russian rivers, the Yukon and Colville River upstream of their estuaries;
  - sediment cores for each continental shelf and for the Arctic Basin;
- estimates for volatilization rates of PAHs and n-alkanes from land and the Arctic Ocean;
- estimates of photo- and chemical oxidation of PAHs in the Arctic;
- spill statistics for tankers and pipelines in Russia;
- rates of biodegradation and metabolic biological half-lives of PAHs and n-alkanes for the various trophic levels in the Arctic; and
- better estimates of biomass for the terrestrial/aquatic Arctic environment.

In its current incomplete form, the budget provides a rough 'snap shot' of the relative magnitude of known inputs of petroleum hydrocarbons to the Arctic environment of the early twenty-first century. At present it indicates that actual estimated inputs from oil and gas exploration, production and refining activities are small compared with natural seeps, spills, and atmospheric deposition. If spills are combined with inputs from oil and gas exploration, production and refining activities in the Arctic, then the inputs are substantial, but still significantly smaller than the estimated inputs from natural seeps; the largest single input of petroleum hydrocarbons to the Arctic environment. Another observation from Table 4.31 is the apparent almost equal inputs of petroleum hydrocarbons into each of the marine and terrestrial/freshwater environments. In addition, it appears that the input of non-petrogenic hydrocarbons is small compared with the input of petrogenic hydrocarbons. Finally, it is currently not possible to estimate the amount of petroleum hydrocarbons that escape the continental shelves and accumulate in the Arctic Basin. This may be substantial depending on the effectiveness of shelf sea ice transporting coastal sediments to the central Arctic Ocean.

4.6. Concentrations of petroleum hydrocarbons and PAHs in the Arctic environment

Direct comparisons of the concentrations reported for petroleum hydrocarbons and PAHs in various studies in the Arctic are often difficult and sometimes impossible to compare quantitatively because of typically wide differences in the sampling methods, analytical methods, instrumentation, quantification, quality control protocols and reporting formats used. Some values, particularly those produced by non-specific techniques such as infra-red, ultraviolet/fluorescence and gravimetry tend to overstate the amount of petroleum hydrocarbons present in a sample. Despite the recommendations made by AMAP in its previous assessment (AMAP, 1998), the lack of consistency among many of the data sets continues to limit severely their usefulness for analyzing spatial and temporal trends, although the recent trend is that more and more data sets do provide coherent, intercomparable data for petroleum hydrocarbons and PAHs.

Even more problematic is the fact that the inherent quality (precision and accuracy) of the analytical data varies widely among the various data sets. This variability is certain to be compounded by other important factors such as sampling representativeness and sampling contamination. Data that are suspect or clearly erroneous have been excluded from the data tables presented in this assessment. However, as adequate details describing the sampling techniques, sample storage, sample processing and sample analysis (calibration, estimation of precision and accuracy, quantification procedures etc.) specific to each data set were not always available to review in detail, no holistic data quality assessment was made of the

Table 4.31. Summary of estimates for petroleum hydrocarbon inputs to the Arctic.

Input pathway	Terrestrial/freshwater environment, t/yr			Marine environment, t/yr		
	n-alkanes	PAHs	Total	n-alkanes	PAHs	Total
Atmosphere	-(?)	- (?)	- (62)	- (?)	- (?)	- (92)
Riverine/freshwater	- (0)	- (0)	- (0)	688 (693)	551 (40)	1239 (733)
Industrial activities (oil and gas)	4 (0)	60 (0)	64 (0)	15 (0)	12 (0)	27 (0)
Industrial activities (non oil and gas)	213 (0)	30 (53)	243 (53)	- (0)	- (0)	- (0)
Spills	231 (0)	186 (0)	417 (0)	124 (0)	99 (0)	223 (0)
Natural seeps	2300 (0)	1880 (0)	4180 (0)	2300 (0)[a]	1880 (0)	4180 (0)
Coastal erosion (coastal soils + peat)	-	-	-	neg. (432)	neg. (9)	neg. (441)
Total	2748 (0)	2156 (53)	4904 (53)	3127 (1125)	2542 (49)	5669 (1266)

neg: Negligible; [a] values in parentheses are comparative values for estimated non-petrogenic inputs.

remaining data. Consequently, the reader should consider the intercomparability of specific data presented here to be more qualitative than quantitative. Nevertheless, while the accuracy and representativeness of any individual data value in certain datasets may be arguable, taken as a whole the data reported can be used to identify the general range of concentrations of petroleum hydrocarbons and PAHs in the various compartments of the Arctic ecosystem. It is also important to note that in recent AMAP studies commissioned as part of this assessment, the issue of intercomparability of data has been addressed and the data from these studies are essentially intercomparable.

4.6.1. Concentrations in air

In the atmosphere, PAHs are distributed between the gas phase and the particulate phase. Phase partitioning largely depends on atmospheric conditions such as temperature and relative humidity. It also depends on the vapour pressure of the individual PAH molecule, which ranges from 10^{-1} to 10^{-10} Pa (Pozzoli et al., 2004). It is therefore possible that the concentrations of the most volatile (low molecular weight) PAHs are underestimated during measurements of the particulate phase due to evaporation losses. The phase partitioning varies widely among individual PAHs. For example, naphthalene, phenanthrene and anthracene occur almost exclusively in the gas phase whereas, benzo[*a*]pyrene and benzo[*ghi*]perylene are primarily found adsorbed onto particulates.

Collection of gas-phase PAHs in the atmosphere is normally achieved using an adsorbent cartridge, typically polymer-based, through which a known air volume is passed. Particle-phase atmospheric PAHs are sampled using filters of different kinds, although recently a new technology of diffusion denuders has been introduced to avoid the problems with artefacts that often arise when filters are used. A diffusion denuder is a tube with an adhesive material-coated inside, on which air particles are adsorbed as the air sample is drawn through the tube. Unfortunately, there is no ideal technology for PAH sample collection because adsorption cartridges and filters as well as diffusion denuders can lead to artefacts, either due to loss of part of the sample or to contamination due to the presence of analyte in the material of which the instrument is made.

The main sources of PAHs in the atmosphere are from combustion processes. PAHs in the atmosphere may also derive from non-combustion processes such as production of creosote and coal tar. PAH emission upon combustion is due either to the pyrosynthesis of PAHs from various precursors in the fuel, or to the survival of PAHs in the starter fuel. Different combustion sources tend to emit different compounds, due to different flame temperatures, oxygen contents, and ratios between pyrosynthetic and originally-surviving PAHs. For instance, PAHs emitted by gasoline-powered cars are mainly formed by pyrosynthesis whereas a significant fraction of those emitted by diesel cars were already present in starter fuel (Pozzoli et al., 2004).

4.6.1.1. Canada

There are few local sources of atmospheric PAH contamination in the Canadian Arctic. In general, the air in the Canadian Arctic is not significantly contaminated with hydrocarbons (Table 4.32). Air in Arctic Canada has been shown to contain three orders of magnitude less PAHs than air in U.K. urban centres and one to two orders of magnitude less than in U.K. rural areas (Prevedouros et al., 2004).

4.6.1.2. Greenland

In general, the Greenland environment is relatively uncontaminated; however long-range transport via water and air is a source of contamination. Ice cores from the Greenland Ice Cap have shown a dramatic increase in PAH concentrations over the last 100 years resulting from the long-range atmospheric transport of PAHs, which corresponds well with the historical record of industrialization and world petroleum production (Macdonald et al., 2000).

4.6.1.3. Norway

Local sources of atmospheric PAH emissions in northern Norway are few and relatively small. Data from the background monitoring station on Zeppelin Mountain at Ny-Ålesund (Svalbard) indicate that in recent years, atmospheric PAH concentrations have been low and stable (Aas et al., 2006). In 2004–2005, the mean value for ΣPAHs was 2.2 ng/m^3. Seasonal fluctuations in concentration are dependent on the yearly transport of air from lower latitudes which is most prevalent during winter and early spring. The burning of wood and fossil fuels at various point sources in this area is also highest during winter. The most volatile PAHs (naphthalene, C_1-naphthalenes, biphenyl) on average made up 48% of total PAH concentrations (38 components analysed). In 2005, the highest PAH value recorded was 14.7 ng/m^3. The sample was collected during a time of significant air inflow from Eastern Europe and Russia.

4.6.1.4. Russia

In the Russian Arctic, only emissions of benzo[*a*]pyrene are routinely documented. In the Murmansk Oblast, the main sources of contamination are located in the area of the Khibiny Massif, which is a natural geochemical anomaly with respect to unique deposits of apatite and nepheline ores, and where one of the world's largest phosphate manufacturing plants is located. These mining operations as well as local heating and power plants use fossil fuels, including raw oil, which leads to the release of PAHs into the atmosphere. The amounts vary greatly from time to time and from place to place. Thus, in 1991 in the town of Apatity, 300 kg of benzo[*a*]pyrene were emitted into atmosphere, but very little is reported to have been emitted since then; in the town of Monchegorsk the emissions of benzo[*a*]pyrene have averaged 20 kg/yr since 1997 (AMAP, 2004).

Table 4.32. Concentrations of PAH in air in the Canadian Arctic (data source: Atmospheric Environment Service, Environment Canada).

	Date	Concentration range, pg/m^3	
		particulate	gas
Alert (82.3° N, 62.3° W)	1992–1996	0.01–3050	8–4650
Tagish (60.3° N, 131.2° W)	1993/1994	0.02–994	64–4280
Little Fox Lake (61.3° N, 135.6° W)	2002/2003	4–102	84–458
Cape Dorset (64.7° N, 61.4° W)	1994 (Mar–Nov)	0.02–461	21–1020

PAH represents the sum of sixteen parent PAHs: acenaphthalene, acenaphthene, fluorene, phenanthrene, anthracene, fluoranthene, pyrene, benz[*a*]anthracene, chrysene, benzo[*b*]fluoranthene, benzo[*k*]fluoranthene, benzo[*e*]pyrene, benzo[*a*]pyrene, indeno [1,2,3-*cd*]pyrene, dibenz[*ah*]anthracene, benzo[*ghi*]perylene.

In the Lovozero area of the Murmansk Oblast, PAH emissions have gradually decreased since the early 1990s, possibly due to changes in the fuel types used. However, after 1998, the amount of PAHs emitted has levelled off (AMAP, 2004).

In the Nenets Autonomous Okrug, a region of oil and gas production, petroleum hydrocarbon fuel types dominate and were responsible for 20% of benzo[*a*]pyrene emissions (in 1995) as compared to only a 7% contribution from coal; this is a much lower ratio than in other regions of Russia (AMAP, 2004). The major source of PAHs, however, is wood burning, which accounts for 72% of benzo[*a*]pyrene emissions. This is of particular concern in relation to possible impacts on human health because firewood is mostly used for domestic heating.

In the Taymir Autonomous Okrug of Russia, the major source of PAH emissions to the atmosphere is the Norilsk industrial complex. Coal combustion dominates PAH emissions for Taymir. Total emissions of four PAH compounds averaged 50 kg/yr. The contributions from defence-related activities, however, have not been included in the assessment, because this information is unavailable (AMAP, 2004).

4.6.2. Concentrations in the terrestrial and freshwater environment

4.6.2.1. Soils

PAH contamination in soil increases in areas affected by emissions from industry and traffic, and domestic heating. For this reason, PAH contamination in Arctic soils is significantly less than that in more densely populated southern latitudes, apart from specific 'hot spots' such as in areas of oil production or leaks, and close to other industrial and contaminated sites. Thus, PAH levels in soils on Svalbard were very low, both within human settlements and on the tundra (Gulinska et al., 2003). Soil samples were collected at three locations in Greenland in 2000 (Riget et al., 2003). Total PAH concentrations were around 110 μg/kg ww, and 42 μg/kg ww for higher molecular weight PAH. Generally, soils in the Arctic currently have PAH concentrations similar to those typical of pre-industrial times, for example benzo[*a*]pyrene in the range 0.8–4.3 μg/kg (Wilcke, 2000).

4.6.2.2. Terrestrial biota

4.6.2.2.1. Greenland

Samples of lichen (*Cetraria nivalis*) were analysed at three locations near the small town of Qaqortoq in southern Greenland in 1999–2000 (Riget et al., 2003). The concentrations were approximately 80 ng/g ww for total PAH and 30 ng/g ww for higher PAHs.

4.6.2.2.2. Russia

Concentrations of PAHs in terrestrial biota were assessed during 1999 to 2004 in four regions across the Russian Arctic (Kola Peninsula, Pechora Basin, Taymir Peninsula and Chukotka) (AMAP, 2004).

RCMA (2005) reported total PAH concentrations (16 compounds) in moss (*Sphagnum* sp.), red bilberry (*Vaccinium vitis-idaea*), and great bilberry (*Vaccinium uliginosum*). Average PAH concentrations in moss from five regions of the Russian Arctic (Bolshezemelskaya Tundra, Yugorsky Peninsula, Yamal Peninsula, Taymir Peninsula and Chukchi Peninsula) varied from 45 ng/g dw (Yugorsky Peninsula) to 907 ng/g dw (Yamal Peninsula); petrogenic naphthalene and phenanthrene were the most abundant compounds in all samples. Average total PAH concentrations in berry samples collected from three regions of the Russian Arctic (Bolshezemelskaya Tundra, Taymir Peninsula and Chukchi Peninsula) were in the range 21–90 ng/g dw. The highest concentrations were in samples from the Chukchi Peninsula. Naphthalene was the most abundant PAH in all the berry samples.

RCMA (2005) reported total PAH concentrations (16 compounds) in muscle and liver of three terrestrial bird species: partridge (*Lagopus lagopus*), goose (*Anser fabialis*), and duck (*Anas acuta*). Concentrations in liver always exceeded concentrations in muscle. In goose liver and duck liver, total PAH concentrations were similar in all regions and varied within the ranges 35–58, and 34–64 ng/g ww, respectively. The highest concentrations were found in partridge, ranging from 114 ng/g ww in the Yugorsky Peninsula region to 214 ng/g ww on the Chukchi Peninsula. Naphthalene and phenanthrene were the most abundant PAH compounds in all bird samples.

In tissue samples from reindeer (*Rangifer tarandus*) and hare (*Lepus timidus*), concentrations of total PAH (16 compounds) were two- to six-fold higher in liver than in muscle (RCMA, 2005). Overall, average PAH concentrations in reindeer liver from four regions of the Russian Arctic (Bolshezemelskaya Tundra, Taymir Peninsula, Yamal Peninsula and Chukchi Peninsula) ranged from 67–319 ng/g ww; the highest concentrations occurred in samples from the Chukchi Peninsula. PAH concentrations in reindeer muscle were about half the concentrations found in bird muscle. For hare liver, the average total PAH concentrations were 75, 114, 138 and 162 ng/g ww for the Yugorsky Peninsula, Taymir Peninsula, Bolshezemelskaya Tundra and Chukchi Peninsula areas, respectively. PAH concentrations in hare muscle were comparable to or higher than those in reindeer muscle. Naphthalene and phenanthrene were the most abundant PAH in all samples.

4.6.2.3. Freshwaters

4.6.2.3.1. Greenland

Precipitation data for total PAHs were measured in the summer and winter of 2001 at three stations in southern Greenland: a background area (Usuk), near Igaliko (a settlement with 40 people), and the town of Qaqortoq (with 3200 inhabitants). Average concentrations were about 50 ng/L, with the lowest values at Usuk and the highest at Qaqortoq. At Qaqortoq, concentrations were two- to three-fold higher than average levels at background sites in France during the summer (Riget et al., 2003).

4.6.2.3.2. Russia

The amounts of petroleum hydrocarbons discharged with wastewater in the Chukchi Autonomous Okrug of Russia has been estimated to average 10 t/yr for the period 1991 to 2000 (AMAP, 2004).

4.6.2.4. Freshwater sediments

4.6.2.4.1. Canada

The Mackenzie River sediments have similar concentrations of alkanes to the neighbouring smaller rivers but

approximately 10-fold higher PAH concentrations, while PAH concentrations in the smaller Canadian rivers are comparable to those in Siberian rivers. The concentrations of most PAH compounds (for which sediment quality guidelines exist [CCME, 1998]) in suspended particulates and sediment samples from the Mackenzie Delta area exceed the sediment quality guideline threshold values (concentrations above which biological effects are expected) under summer flow conditions (Yunker et al., 2002).

4.6.2.4.2. Greenland

Sediment samples collected from a lake in Greenland in 2000 contained around 0.12 μg/g dw total PAHs and 0.04 μg/g dw for the higher molecular weight PAHs, which is comparable to concentrations found in previously analyzed samples (Riget et al., 2003).

4.6.2.4.3. Norway

In 1995 and 2006, the concentrations of PAHs in freshwater sediments were mapped in lakes in northern Norway (Nordland, Troms and Finnmark counties) and Svalbard. In 1995, a total of 22 lakes were investigated and in 2007 a total of 32 lakes were studied. The results revealed that the concentrations of PAH were generally low (<1.0 μg/g dw) in surface sediments from lakes in all areas (Skotvold et al., 1997; Monsen et al., 2007). In both studies the highest PAH concentrations were found in lakes from Nordland County, with up to 7 μg/g dw (ΣPAH minus perylene) in 1995 and up to 6.6 μg/g dw (Σ16 USEPA PAH) in 2006: the lowest concentrations occurred in samples from lakes in Troms and Finnmark counties. The 2007 study reported concentrations between 0.02 and 1.3 μg/g dw (Σ16 USEPA PAH) whereas the 1997 study reported concentrations between 0.1 and 1.0 μg/g dw (ΣPAH minus perylene). In sediment from lakes in Svalbard and in Ellasjøen on Bear Island (Bjørnøya) concentrations varied between 0.9 and 1.2 μg/g dw. The low PAH concentrations in Svalbard lake sediments were confirmed by Rose et al. (2004), who suggested that the source of PAHs in these lakes was the local combustion of coal.

In studies of the Pasvik watercourse (the border river between Norway, Russia and Finland), the concentrations of PAHs in surface sediments varied from 0.2 to 12.9 μg/g dw (Christensen et al., 2007a). The highest concentrations were recorded in Lake Kuetsjarvi (2.9–12.9 μg/g dw) which, for many years, has been the main recipient of discharges from the nickel smelters in Nikel, Russia. The concentrations in the lakes downstream from the smelters (Lake Skrukkebukta and Lake Bjørnvatn) were low, but higher (at 0.6 μg/g dw) than those measured in the upstream lakes (Lake Inari in Finland, Lake Ruskebukta and Lake Tjærebukta). Emissions to water and air from the smelters in Nikel are the most likely source for this pattern of contamination in the Pazvik watercourse.

4.6.2.5. Freshwater biota

4.6.2.5.1. Greenland

Average PAH concentrations in liver from Arctic char (*Salvelinus alpinus*) collected in 2000 from a small lake (Igaliko) and an open fjord (Itilleq), both near Qaqortoq in southern Greenland were 205 and 250 ng/g ww, respectively (Riget et al., 2003).

4.6.2.5.2. Norway

In the muscle tissue of freshwater fish – whitefish (*Coregonus lavaretus*), perch (*Perca fluviatilis*) and Arctic char – from Finnmark the concentrations of total PAH varied from 4 to 14 ng/g ww (Skotvold et al., 1997). Only the most volatile PAHs (from naphthalene to pyrene) were detected.

In Lake Stuorajarvi (Finnmark County) and in the Pazvik watercourse (Norwegian–Russian border area) samples of liver from pike and whitefish were analyzed for PAHs (Christensen et al., 2007b). The concentrations were highest in pike and whitefish from Lake Kuetsjarvi in the Pazvik watercourse (156 ng/g ww and 331 ng/g ww, respectively). The concentrations in pike and whitefish liver from Lake Stuorajavri were 50 and 46 ng/g ww, respectively. The concentrations were higher in samples of whitefish liver from Lake Skrukkebukta (108 ng/g dw) downstream of Lake Kuetsjarvi compared to those measured in whitefish liver from two lakes upstream (34 and 74 ng/g dw) of Lake Kuetsjarvi. Emissions to water and air from the smelters in Nikel (Russia) are the most likely source for this pattern of contamination pattern in the Norwegian–Russian border area.

4.6.2.5.3. Russia

PAHs (five compounds) were measured in muscle tissue of whitefish, Arctic cisco (*C. autumnalis*), broad whitefish (*C. nasus*) and Arctic grayling (*Thymallus arcticus*) collected in 2001 at sites near the Kola Peninsula, east and west Taymir, Chukotka shores, and in the Pechora Basin; the highest concentrations of PAHs (112 ng/g ww) were measured in whitefish from west Taymir (AMAP, 2004).

Total PAH concentrations (16 compounds) were measured in the liver of six freshwater fish species: pike, whitefish, Arctic cisco, chum salmon (*Oncorhynchus keta*), inconnu (*Stenodus nelma*) and perch from various locations in the Russian Arctic sampled in 1999–2004 (RCMA, 2005). Average total PAH concentrations for specific species and areas were as follows: whitefish from the Pur River, Pechora River and small rivers of the Taymir Peninsula, 44, 52, and 103 ng/g ww, respectively; Arctic cisco from the Yamal and Taymir Peninsulas, 68 and 66 ng/g ww, respectively; inconnu, pike and chum salmon from the Chukchi Peninsula, 35, 83, and 53 ng/g ww, respectively; and perch from the Pechora River, 38 ng/g ww (RCMA, 2005).

4.6.3. Concentrations of petroleum hydrocarbons and PAHs in the marine environment

4.6.3.1. Seawater

The main sources of petroleum hydrocarbons entering the marine environment are discharges (typically diffuse discharges) from land, direct discharges to the sea and water courses, and atmospheric inputs. Discharges from land include industrial effluents containing oil, precipitation runoff, waste oils and sewage. Discharges at sea include chronic releases from oil tankers and other vessels, dry-docking, accidents, and offshore oil production. Natural oil seeps also contribute. Atmospheric inputs are mainly via wet and dry deposition. Coastal areas in which there are industrial operations, sea lanes used by oil tankers, and areas of offshore production of oil and gas are therefore locations where concentrations of petroleum hydrocarbons could be expected to be higher than average.

4.6.3.1.1. Canada

Concentrations of alkanes, and parent and alkylated PAHs were measured in 2002 at seven potential offshore drilling areas in the Mackenzie River estuary (Kavik-AXYS, 2004). Samples of several hundred litres each were collected by high volume water sampling/solid phase extraction techniques. The range of alkane concentrations was 6–19 ng/L with the corresponding particulate alkanes in the range 2.5–6.1 ng/L. Dissolved PAH concentrations were 4–6 ng/L for parent compounds and 3.5–11 ng/L for alkylated PAHs. Corresponding ranges of parent PAHs and alkylated PAHs in particulates were 0.5 to 2.6 ng/L and 3.5 to 11.1 ng/L, respectively.

4.6.3.1.2. Norway

In 2004, total hydrocarbon concentrations were measured in the offshore waters of the Norwegian part of the Barents Sea. The results were in the low µg/L range, reaching background and limit of detection values at almost all locations; the exception was a concentration of 23.5 µg/L. The remaining 39 stations had an average concentration of 0.9 µg/L, ranging from 0 to 3.8 µg/L (Boitsov et al., 2007).

In 2005, concentrations of PAHs in seawater samples from eleven locations in the Barents and Norwegian Seas were found to be very low (Boitsov et al., 2007). Concentration ranges for selected PAHs were: acenaphthylene 0–0.06 ng/L, 1-methylphenanthrene 0–0.7 ng/L, benzo[*e*]pyrene 0–0.2 ng/L, benzo[*a*]pyrene 0–0.35 ng/L, perylene 0.08–0.7 ng/L.

4.6.3.1.3. Russia

Concentrations of aliphatic hydrocarbons in surface water were measured after column chromatography on silica gel by infrared spectrophotometry with IR-435 Shimadzu in an expedition aboard the *R/V Akademik Fedorov* in 2000 (Nemirovskaya and Novigatskii, 2003). The expedition went along the Russian Arctic coast and to higher Arctic regions (Mendeleev Rise). Aliphatic hydrocarbon concentrations were in the range 0–33 µg/L (Figure 4.17). The highest levels (up to 33 µg/L) were found in the Pechora Sea and in the Kara Sea areas affected by the Ob and Yenisey River discharges. Low concentrations (0–10 µg/L) were found in water from the Mendeleev Rise region, which is considered one of the least contaminated regions of the Arctic and therefore suitable for measurements of background concentrations. The amounts of dissolved and particulate PAHs and aliphatic hydrocarbons present in snow, ice, and water under ice in the high Arctic regions: Franz-Victoria Trough (80°–82° N, 37°–39° E) and Mendeleev Rise (81°–82° N, 170° E–170° W) were assessed in 1998 and 2000, respectively. Total aliphatic hydrocarbon concentrations (the sum of the dissolved and particulate fractions) in snow (32 µg/L), upper ice core (173 µg/L), and water under ice (32 µg/L) collected in the eastern Arctic Ocean (Franz-Victoria Trough) were much higher than those measured in the western Arctic Ocean (Mendeleev Rise) (11, 12, and 21 µg/L) (Nemirovskaya and Novigatskii, 2003). The similar differences between contamination levels in these two Arctic regions were also typical for PAHs (Nemirovskaya and Novigatskii, 2003). The most abundant PAHs detected were fluoranthene and phenanthrene. The distribution of hydrocarbons in ice is more related to how the ice was formed and the ice drift pattern than the age of the ice. No significant impact of human activity was detected (Nemirovskaya and Novigatskii, 2003).

Sources of petroleum contamination to the Russian Arctic seas include inputs from freshwater systems, long-range transport by sea currents, atmospheric deposition, and contamination linked with human activities in harbours and offshore waters. Petroleum hydrocarbons and PAHs in the Barents Sea and in bays of the Kara Sea can also originate from natural seeps. The Barents and White Seas are areas of intensive human activities such as fishing, and commercial and military shipping, as well as the exploration, production and transportation of hydrocarbons. Human activities in the other seas of the Russian Arctic are presently limited, and therefore the level of contamination is generally low and related to natural sources. However, development of offshore oil and gas exploration in the Gulf of Ob (Kara Sea) is likely to intensify anthropogenic inputs into adjacent ecosystems in future.

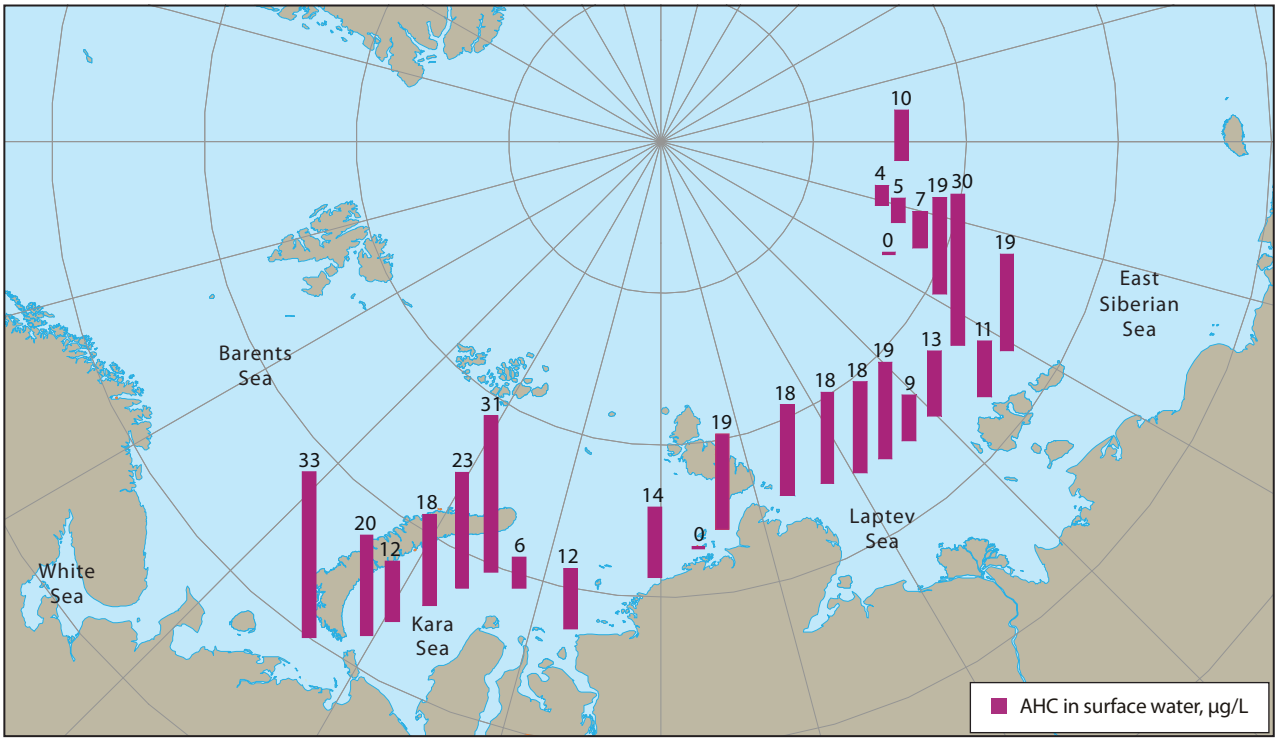


Figure 4.17. Aliphatic hydrocarbons in surface seawater off Russia in 2000 (Nemirovskaya and Novigatskii, 2003).

4.6.3.1.3.1. Barents Sea

In many respects, Atlantic Current inflow determines the hydrological and ecological conditions in the southern part of the Barents Sea. Transboundary transfer of contaminants by ocean currents and the atmospheric transport of aerosols from the northern European industrial centres are believed to have a greater influence on the Barents Sea ecosystem than those of the other seas of the Russian Arctic (Golubeva, 2002; Ilyin et al., 2004).

Industrial activities and municipal utilities emit about 71.5×10^6 m^3 of wastewater per year to Kola Bay, containing about 33 t of petroleum hydrocarbons (Korshenko et al., 2005). In the south-eastern Barents Sea (Pechora Sea) discharges of the Pechora River and industrial installations of Varandej harbour are also sources of contamination. The annual flux of hydrocarbons from the Pechora River to the Barents Sea is about 5400 t (Miskevich, 1996).

Ilyin et al. (2004) reported that in 2001–2004, concentrations of total hydrocarbons in water samples from the central and southern Barents Sea were 0–0.12 mg/L measured by infrared spectrometry. The highest concentrations, exceeding the Russian Federation maximum permissible concentration (MPC) of 0.05 mg/L, were registered in frontal zones and in coastal areas (Ilyin et al., 2004). In the northern Barents Sea (81°–82° N), the average concentration of total hydrocarbons in surface waters was 16 μg/L (range 11–22 μg/L) (Nemirovskaya, 2004). In the mixing zone of Atlantic and Barents Sea water, total hydrocarbon concentrations at the depth of the pycnocline were three- to five-fold higher (up to values exceeding the MPC) than in the surface waters (Nemirovskaya, 2004). In 2002, the total amount of petroleum hydrocarbons in water samples from the south-eastern Barents Sea (Pechora Sea) was 2.0–87.5 μg/L, with an average of 19.0 μg/L (Korshenko et al., 2003). In spring 2003, average concentrations of total hydrocarbons in snow, sea ice, and water under ice collected in the Pechora Sea were respectively 19.4, 15.2 and 19.4 μg/L (Korshenko et al., 2005).

Generally, total concentrations of n-alkanes (sum of nC_{10} to nC_{30}) were in the range 1.1–20.0 μg/L (Ilyin et al., 2004), but in some areas the total concentrations of n-alkanes reached 50–90 μg/L (Plotitsyna et al., 2002; Ilyin et al., 2004). Short-chain alkanes (nC_{12} to nC_{22}) were the most abundant especially in the Murmansk Bank area and in coastal waters. Hydrocarbons of phytogenic and bacterial origin (C_{20}–C_{25}) contributed about 30% of the total. Higher pristane/phytane ratios and lower values of the Carbon Preference Index (CPI, the ratio of odd-numbered to even-numbered carbon chain n-alkanes), indicating stronger petroleum hydrocarbon contamination, were typical for waters of Motovskii Bay and coastal waters of the Kola Peninsula as a whole (Ilyin et al., 2004).

The total concentration of PAHs (sum of 14 PAHs) in surface water samples from the southern Barents Sea were 12–79 ng/L; the highest concentrations were found in coastal waters (Plotitsyna et al., 2002; Ilyin et al., 2004). In surface water samples from the Pechora Sea collected in summer 2002, eight of 24 PAHs analysed were detected and had concentrations in the following ranges: naphthalene <2–18 ng/L, 2-methylnaphthalene 2.1–9.6 ng/L, fluorene 4.9–26.5 ng/L, phenanthrene 2.1–4.6 ng/L, fluoranthene 2.6–5.4 ng/L, pyrene <1–1.6 ng/L, chrysene 0.3–0.78 ng/L and benzo[*b*]fluoranthene <0.2–0.4 ng/L (Korshenko et al., 2003). In spring 2003, average PAH concentrations (sum of 24 compounds) in snow, sea ice, and water under ice collected in the Pechora Sea were 79.3, 50.9 and 107 ng/L, respectively (Korshenko et al., 2005). Higher PAH concentrations occurred in the southern part of the *Prirazlomnoye* oilfield area (Korshenko et al., 2005). PAH concentrations in bottom water samples were higher than those found in surface waters. In the central trench of the Barents Sea, PAH concentrations ranged from 260 to 330 ng/L, and concentrations as high as 420–500 ng/L were found in coastal waters (Ivanov, 2002).

4.6.3.1.3.2. White Sea

River discharges containing contaminants from pulp and paper plants, power stations, municipal utilities, and naval and inland water transport are the most significant sources of contamination in the White Sea. It was estimated that a total of 2244 t of petroleum hydrocarbons entered the White Sea with river fluxes and industrial discharges in 2003 (Korshenko et al., 2005).

The average total hydrocarbon concentration in seawater from the central White Sea, including Dvina, Onega and Kandalaksha Bays, was 12 μg/L (Nemirovskaya, 2004). The highest concentrations were found in Kandalaksha Bay (35–45 μg/L) and Dvina Bay (average 28 μg/L). A CPI value of 0.98 in samples from both bays suggests that the source is predominantly petrogenic (Nemirovskaya, 2004).

4.6.3.1.3.3. Kara Sea

Petrogenic contamination related to ship traffic and petroleum exploration and transportation is significant for the rivers discharging into the Kara Sea. However, recent estimates of the total amounts of petroleum hydrocarbons flowing into the Kara Sea with river discharges are not available.

In 2003, the coastal Kara Sea had an average total hydrocarbon concentration in surface seawater of 60 μg/L (1.2 times the MPC); the highest observed concentration was 406 μg/L (nine times the MPC) (Korshenko et al., 2005).

In Yenisey Bay water, total hydrocarbon concentrations were 8–62 μg/L (Korshenko et al., 2005). PAH concentrations in water samples were 14.3–38.2 ng/L in 2003. Seven of 24 PAH compounds analysed were detected in these samples: naphthalene (6.3–17.2 ng/L), 2-methylnaphthalene (2.7–8.6 ng/L), fluorene (1.3–5.86 ng/L), phenanthrene (1.7–7.9 ng/L), fluoranthene (0.74–3.25 ng/L), pyrene (0.44–1.37 ng/L), and benzo[*b*]fluoranthene (0.17–0.38 ng/L) (Korshenko et al., 2005).

Carroll et al. (2007) reported total PAH concentrations in surface water samples from the Kara Sea, Gulf of Ob and Yenisey Bay of 11.3 to 54.5 ng/L in 2003–2005. Total PAH levels found in the Gulf of Ob were significantly higher than in the other areas investigated. Naphthalene, fluorene, phenanthrene and its alkylated homologues dominated in all samples, representing 88 to 95% of total PAH. Petroleum pollution is the most probable PAH source. Total PAH supplies from the Ob and Yenisey rivers to the Kara Sea in 2003 and 2005 were 15.5 and 11.0 t, respectively (Carroll et al., 2007).

4.6.3.1.3.4. Laptev Sea

Sources of contaminants for the Laptev Sea are poorly studied. The main source of contamination is believed to be freshwater discharge by the Lena River (Lobchenko, 1997).

The average total hydrocarbon concentration in surface waters of the Laptev Sea was 40 μg/L in 1996 (Lobchenko, 1997). The highest concentration (80 μg/L) was found at Guba Buor-Khaya close to Tiksi harbour. In 2003, total hydrocarbon

and PAH concentrations in surface water samples from Khatanga Bay (south-western Laptev Sea) were within the ranges 12 to 52 μg/L and 12 to 30 ng/L, respectively (Korshenko et al., 2005). Concentrations of detected PAH compounds were in the following ranges: naphthalene (4.7–18.2 ng/L), 2-methylnaphthalene (1.2–3.4 ng/L), fluorene (0.8–3.2 ng/L), phenanthrene (3.1–9.4 ng/L), anthracene (0.11–0.84 ng/L), fluoranthene (0.57–1.67 ng/L), pyrene (0.84–6.28 ng/L), and benzo[*b*]fluoranthene (0.22–0.74 ng/L) (Korshenko et al., 2005).

4.6.3.1.3.5. Chukchi Sea

Average concentrations of detected PAH compounds in surface water samples from the Chukchi Sea were: naphthalene 53.8 ng/L, biphenyl 13.4 ng/L, 2-methylnaphthalene 16.5 ng/L, fluorene 10.6 ng/L, phenanthrene 33.5 ng/L, fluoranthene 7.4 ng/L, pyrene 3.9 ng/L, and benzo[*b*]fluoranthene 1.44 ng/L (Korshenko et al., 2003). In the Herald Bank area, the concentration of benzo[*a*]pyrene in surface water (8.3 ng/L) exceeded the MPC for this contaminant in Russia (Korshenko et al., 2003).

4.6.3.2. Marine sediments

4.6.3.2.1. Alaska, USA

For most Alaskan Beaufort Sea stations, the total saturated hydrocarbon concentrations are low, ranging from 0.21 to 16 μg/g dw (Boehm et al., 2001). These hydrocarbons are a mixture of terrestrial plant waxes with lesser amounts of petroleum hydrocarbons. Samples of river sediments and peat have total saturated hydrocarbon values of 5.8 to 36 μg/g dw and 21 to 32 μg/g dw, respectively. Sediments were sampled in the Colville, Kuparuk, and Sagavanirktok Rivers. Peat samples came from areas along the Colville and Kuparuk Rivers. The composition of saturated hydrocarbons in the river and peat samples were similar to the composition in Beaufort Sea surficial sediments. This similarity indicates a common source of saturated hydrocarbons for river sediments and nearshore surficial sediments. The highest total saturated hydrocarbon value for this suite of samples (50 μg/g dw) was found at a station west of West Dock in Prudhoe Bay (Boehm et al., 2001). The sample from this station also contained high concentrations of metals and PAHs thus indicating contamination from an anthropogenic source.

PAH concentrations were within the range of values reported from previous studies in the Beaufort Sea and other areas (Boehm et al., 2001). The PAHs in most of the sediment samples were derived from petrogenic/fossil fuel (petroleum and coal), biogenic (perylene), and pyrogenic sources. The station located west of West Dock had the highest PAH concentration, 2700 μg/g dw.

Boehm et al. (2001) noted an increase in the ratios of pyrogenic to petrogenic PAHs between the samples collected from the same stations in 1989 and 1999; the mean ratios were 0.038 in 1989 and 0.096 in 1999.

The hydrocarbons in sediments sampled in 1997 (Naidu et al., 2001) indicate a mixture of organic matter of marine and terrestrial origin. The total saturated hydrocarbon levels range from about 201 to 12 498 ng/g dw and are largely characteristic of biogenic sources. The low-molecular-weight saturated hydrocarbons are derived mainly from marine sources, and the high-molecular-weight saturated hydrocarbons come mainly from plant waxes in the coastal peats and possibly from coal residues. The PAH assemblages in the sediments are very similar to those observed in coastal peats and river sediments. The concentrations of total PAHs range from about 21 to 2185 ng/g dw.

In Alaska, PAH concentrations in bottom sediments have been recently measured within the framework of the Coastal Environmental Monitoring and Assessment Programme (EMAP), started in 2001. According to a technical report of the survey, which included sampling at 55 stations along the Alaskan coast, total PAH concentrations in sediments varied from about 2 to 840 ng/g dw, with high molecular weight PAH varying from 0 to 188 ng/g dw and low molecular weight PAH varying from 1 to 652 ng/g dw.

4.6.3.2.2. Canada

Kavik-AXYS (2004) reported alkane concentrations in the range 11–14 μg/g dw in sediments from seven potential drill sites in the southern Beaufort Sea. The corresponding ranges for parent and alkylated PAHs were 0.8–1.0 and 1.8–2.2 μg/g dw, respectively. PAH concentrations in other parts of the Canadian Arctic are much lower. For example, in sediments sampled in 1994 in the vicinity of southern Baffin Island, parent PAH concentrations were in the range 0.015–0.03 μg/g dw (DND, 1995). In the central Canadian Arctic at Cambridge Bay parent and alkylated PAH concentrations in 1993 samples had ranges of 0.003–0.15 and <0.001–0.4 μg/g dw (DND, 1994).

4.6.3.2.3. Greenland

Total hydrocarbons have not been regularly measured and assessed in the Greenland environment. The earliest record of hydrocarbon measurements in Greenland sediments dates back to a study by Johansen et al. (1977) who reported concentrations in sediments collected in 1975 from 14 sites offshore of West Greenland between Nuuk and Aasiaat (63°–68° N, 51°30′–57°30′ W). The study included 17 surface sediment samples and the average total hydrocarbon concentration was 0.40 μg/g dw (range 0.06–1.17 μg/g dw).

Jensen et al. (1977) reported total hydrocarbon concentrations for 23 samples (18 stations) collected in 1976 from an area west of Sisimiut (66°–68° N, 56º30′–54° W). The average value was 0.075 μg/g dw (range 0.008–0.61 μg/g dw). These values seem generally lower than those for samples collected in 1975, but some or all of the differences could be due to the high heterogeneity and highly variable total hydrocarbon content of the samples.

Hansen et al. (1978) reported the concentrations of hydrocarbons in the marine environment of Kangerdluarssoruseq (the oil terminal for West Greenland), which included surface sediments sampled in 1976 (2 sites) and 1977 (17 sites). The average concentrations of petroleum hydrocarbons in these two studies were higher than those reported by Jensen et al. (1977), especially the samples from 1976 which had an average value of 2655 μg/g dw (range 2170–3140 μg/g dw). For the 1977 samples an average value of 44.9 μg/g dw (range 1.0–125 μg/g dw) was observed. The higher concentrations reflect the role of Kangerdluarssoruseq as an oil terminal.

Concentrations of PAHs in marine sediments collected from southern Greenland in 1999 and 2000 were about 1100 and 400 ng/g ww, respectively (Riget et al., 2003). An assessment of PAH concentrations in marine sediments from the sea bottom near the onshore open waste dumps situated at the coastline in the immediate vicinity of the sea in West and southern Greenland in 2002 were not above natural background levels for those areas (Asmund, 2007).

In 2004, a new project was initiated in Greenland to assess the background concentrations of PAHs in deep-sea sediments at uncontaminated locations and in areas where natural petroleum seeps were suspected (Mosbech et al., 2007).

Samples were collected in six different areas off south-western Greenland and in two areas off East Greenland. Only samples from the Greenland Sea and one sample from Lysefjord (which was not included in the average) had PAH concentrations considerably higher than background. A comparison of ratios between petrogenic and pyrogenic PAHs indicated that there was no significant petrogenic source in this area and thus no indication of natural petroleum seeps.

A number of studies have been initiated in connection with the opening of an offshore region to hydrocarbon exploration at Disco Bay (West Greenland), including the measurement of hydrocarbons in sediments collected from 18 locations in the area (Mosbech et al., 2007). Total hydrocarbon concentrations ranged from <50 to 580 µg/g dw (geometric mean 83 µg/g dw). Except for a few samples, levels were relatively low and compositional profiles provided only a weak indication of oil seeps or other local petrogenic sources. Concentrations of PAHs (19 compounds) from the Marraat area range from just below 0.1 to >1.5 µg/g dw. The lowest values were measured in sediments from shallow nearshore areas along the coast of Marraat. The highest concentrations were found in sediments from deeper water (200–400 m) and at sites more than 1000 m from the coast. The highest concentrations of PAHs were associated with the more fine-grained material at these locations. The 2- and 3-ring alkylated PAHs were relatively dominant in most sediments indicating that the predominant source of PAHs was petrogenic.

4.6.3.2.4. Faroe Islands

PAH concentrations in sediments in the coastal zone of the Faroe Islands are low (Dam and Danielsen, 2003). PAHs have also been studied in sediments at several sites in the Faroe-Shetland Channel where oil exploration has been undertaken (Hoydal and Dam, 2004) (see Appendix 4.4 for concentrations of naphthalene, phenanthrene and dibenzothiophene).

4.6.3.2.5. Norway

4.6.3.2.5.1. Barents Sea harbours

Studies of PAH concentrations in bottom sediments from four Norwegian harbours (Harstad, Tromsø, Hammerfest and Honningsvåg) were carried out in 1996–1997 (Jørgensen et al., 2000: Dahle et al., 2006b; Figure 4.18). The samples were collected at sites near expected local sources of PAH contamination. Total PAH concentrations in these areas were in the range 0.8–41.5 µg/g dw. Sediments in nearby Kola Bay (Russia) had generally lower PAH concentrations than the northern Norwegian harbours (Killie et al., 1997). However, the Kola Bay samples were collected 2–25 km from local sources in Murmansk Harbour.

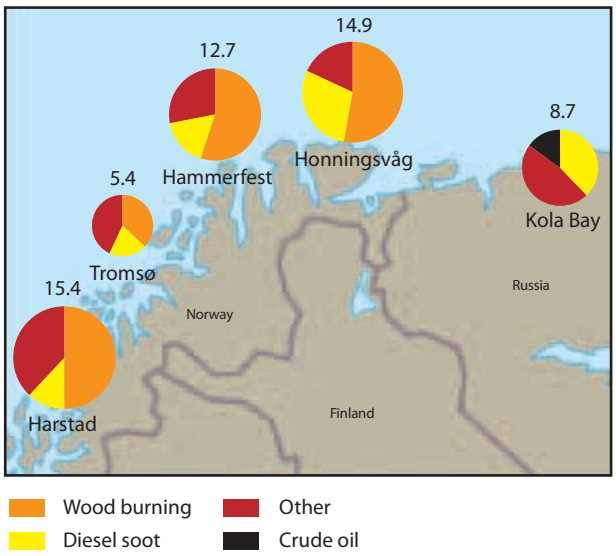


Figure 4.18. Contributions of various sources to total PAH concentrations (mg/kg dw) in bottom sediments from four Norwegian harbours and Kola Bay, Russia (Dahle et al., 2006b).

Pyrogenic PAH compounds predominate in sediment samples from the northern Norwegian harbours. Fluoranthene, pyrene and benzofluoranthenes are the most abundant PAHs, accounting for more than 26% of total PAHs on average. The Fossil Fuel Pollution Index (FFPI; Boehm and Farrington, 1984), which quantifies the approximate percentage of fossil fuel derived PAH to total PAHs, is low; the geometric means of FFPI are from 25 to 38% (Table 4.33). In contrast, in Kola Bay the geometric mean of the FFPI (50%) was significantly higher, and alkylated homologues of dibenzothiophenes and phenanthrene were predominant in the PAH composition.

Table 4.33. Geometric means of PAH concentration, Fossil Fuel Pollution Index values, total toxic benzo[*a*]pyrene-equivalent and PAH isomer pair ratios in bottom sediments from some Norwegian harbours and Kola Bay, Russia.

PAH parameters	Harstad n=13	Tromsø n=18	Hammerfest n=9	Honningsvåg n=7	Kola Bay n=3
PAH concentration (mg/kg dw)					
ΣPAH[a]	15.4	5.4	12.7	14.9	8.7
ΣPyrPAH[b]	9.9	3.1	7.6	7.5	2.8
NPD[c]	4.8	1.9	4.4	6.6	5.5
FFPI (%)	25	30	28	38	51
Total TEQ (mgTEQ/kg dw)	1.41	0.48	1.01	0.84	0.20
Isomer pair ratio					
FLT/202[d]	0.55	0.52	0.55	0.55	0.52
IND/276[e]	0.53	0.51	0.49	0.52	No data

n: Number of samples; FFPI: Fossil Fuel Pollution Index; Total TEQ: total toxic benzo[*a*]pyrene-equivalent; [a]ΣPAH: sum of naphthalene, phenanthrene, dibenzothiophenes and their alkyl-substituted homologues, acenaphthylene, acenaphtene, fluorene, fluoranthene, pyrene, benz[*a*]anthracene, chrysene, benzo[*b+k*]fluoranthene, benzo[*e*]pyrene, benzo[*a*]pyrene, perylene, benzo[*ghi*]perylene, indeno[1,2,3-*cd*]pyrene and dibenzo[*a,h*]anthracene; [b]ΣPyrPAH: sum of PAHs with four- to five-ring hydrocarbons, perylene excluded; [c]NPD: sum of naphthalene, phenanthrene, dibenzothiophenes and their alkyl-substituted homologues; [d]FLT/202: fluoranthene/fluoranthene+pyrene; [e]IND/276: indeno[1,2,3-*cd*]pyrene/ indeno[1,2,3-*cd*]pyrene+benzo[*ghi*]perylene.

The abundance of the less stable kinetic PAHs relative to their more stable thermodynamic isomers (PAH isomer pair ratios), a test applied to infer PAH sources (Yunker et al., 1996), did not differ significantly among the Norwegian harbour samples (Table 4.33) which suggests that PAH contamination in the Norwegian harbours originates from similar sources (Dahle et al., 2006b).

PAH patterns typically produced by burning wood, human habitation pyrogenic PAHs (habitation soot), and diesel soot were found in sediments from all four Norwegian harbours. Data suggested that the identified PAH sources are of local origin, rather than long-range transport. The relatively high contribution of pyrogenic PAH connected to wood burning found in all four Norwegian harbours is evidently linked to the wide use of wood burning stoves to provide heat in Norwegian settlements, whereas this is not typical of the neighbouring Kola Bay area (Figure 4.18). In Kola Bay sediments, the PAHs derive mostly from burning of oil and diesel fuels, presumably associated with shipping and harbour activities, automobiles and power plants. It is estimated that crude oil contributes more than 10% of the total PAH concentrations found in the sediments of Kola Bay (Dahle et al., 2006b).

4.6.3.2.5.2. Sediment monitoring around oil and gas platforms of the Norwegian Sea (Haltenbanken)

The oil companies operating at Haltenbanken undertake environmental monitoring surveys around the oil and gas platforms every third year. The most recent monitoring survey (2006) was organized by the operators Statoil, Hydro, Shell, BP, and Total (DNV, 2007). The survey area and the station grid around the different platforms where sediment sampling was carried out are shown in Figure 4.19. The following fields (number of stations) were included in the investigation: *Urd*

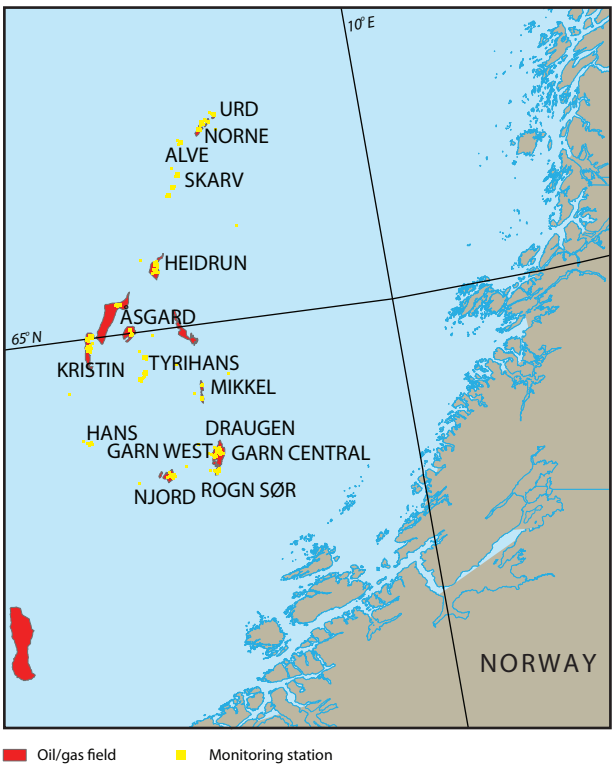


Figure 4.19. Sediment monitoring around oil and gas platforms of the Norwegian Sea (Haltenbacken).

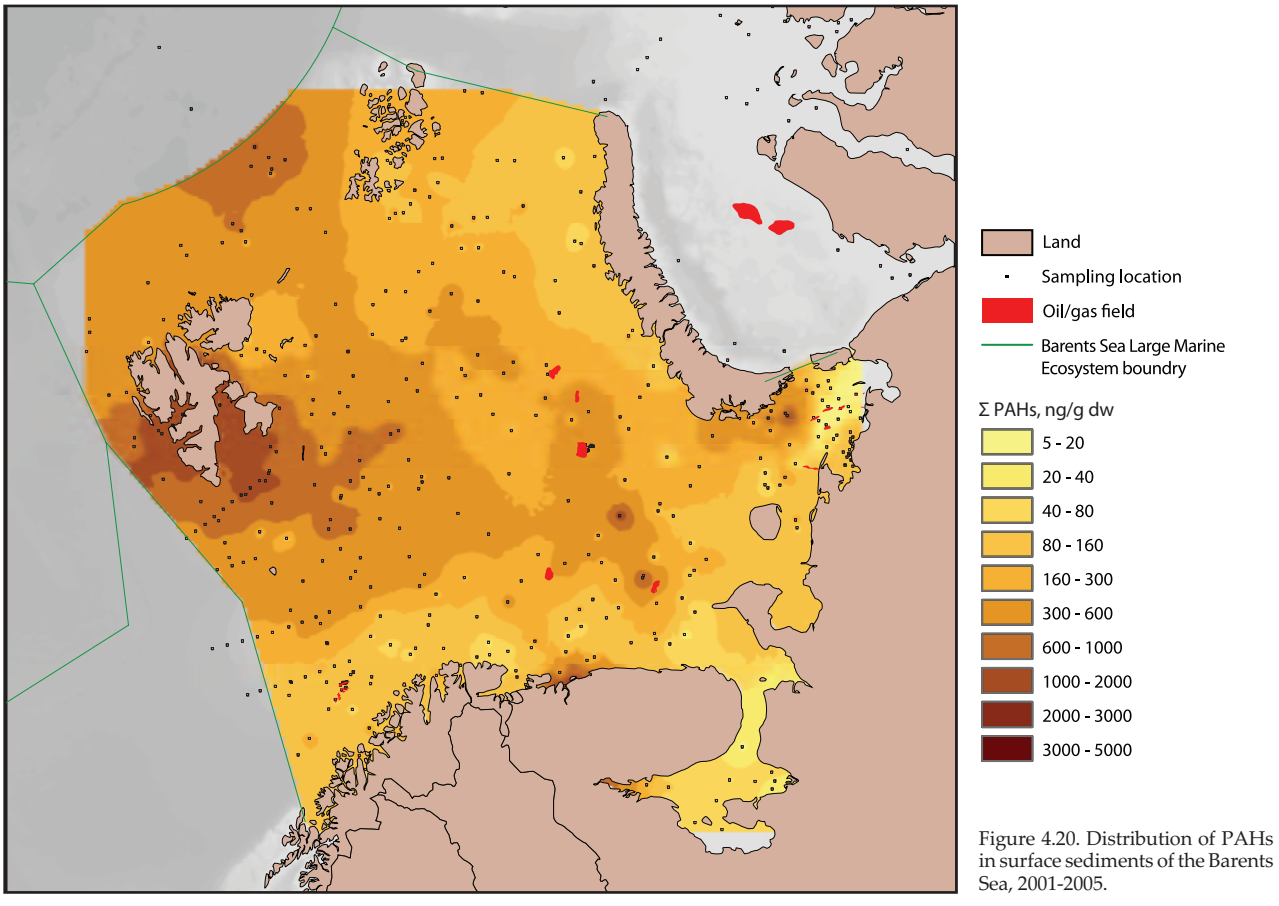


Figure 4.20. Distribution of PAHs in surface sediments of the Barents Sea, 2001-2005.

(8), *Norne* (19), *Heidrun* (20), *Åsgård* (17), *Mikkel* (10), *Kristin* (37), *Njord* (19), *Garn Central* (13), *Garn West* (13), *Draugen* (12), *Rogn Sør* (14), *Alve* (10), *Skarv* (25), *Hans* (14), *Tyrihans* (19). In addition five background stations were included.

The sediments at Haltenbanken are predominantly silt and clay but also contain some fine sand and relatively high amounts of organic material (2.4–8.5% measured as loss on ignition). The total hydrocarbon concentrations in the sediments at Haltenbanken are generally low and vary between 4 and 20 µg/g dw. The concentrations of 2- to 6-ring PAHs are also low. The total hydrocarbon concentrations at the background stations were also low (4–7 µg/g dw). The range of THC concentrations (µg/g dw) in the vicinity of the various oil and gas fields were: *Urd* 4–65, *Norne* 4–2870, *Heidrun* 4–43, *Åsgård* 3–1690, *Mikkel* 3–13, *Kristin* 3–642, *Njord* 4–5250, *Garn Central* 4–7, *Garn West* 5–10, *Draugen* 4–7, *Rogn Sør* 4–6, *Alve* 3–6, *Skarv* 3–8, *Hans* 5–7, and *Tyrihans* 5–9. Elevated concentrations are usually confined to a zone of approximately 250 m around the platforms.

Sediments near the platforms at *Alve*, *Draugen*, *Garn Central*, *Hans* and *Rogn Sør* have no stations where the total hydrocarbon concentrations exceed 10 µg/g dw. Sediments at the *Heidrun*, *Kristin*, *Njord*, *Norne* and *Åsgård* fields, however, have the highest concentrations (maximum values 43–5250 µg/g dw) found in the survey and the pattern of hydrocarbons in the sediments matches that found in the drilling mud base oils used at these fields. The level of enrichment and distribution pattern of total hydrocarbons in the sediments around the various fields vary according to the drilling and waste disposal histories at each field, including factors such as the number of wells and metres drilled and the base fluids used during drilling. The 2006 investigation estimated that 79 km^2 of benthic habitat at Haltenbanken contained sediment concentrations significantly above the simultaneously measured average background concentration of 5.2 µg/g dw. The total area where benthic fauna were affected was estimated to be 14 km^2.

4.6.3.2.5.3. Barents Sea

The spatial distribution of total PAHs in bottom sediments from the Barents Sea is shown in Figure 4.20. The figure is based on data compiled in a special study conducted by AMAP for this assessment and supplied by Akvaplan-niva, the Norwegian Institute of Marine Research (IMR), the Knipovich Polar Research Institute of Marine Fisheries and Oceanography (PINRO) and the All-Russian Research

Table 4.34. PAH concentrations in surface sediment samples (ng/g dw) from the Barents Sea, 2001–2005 (data: Akvaplan-niva, IMR, PINRO and VNIIOkeangeologia)..

	Northwest Barents Sea, n= 22[a] mean ± SD (min – max)	Southeast Barents Sea, n=10[b] mean ± SD (min – max)	Southwest Barents Sea, n=79[c] mean ± SD (min – max)	Northeast Barents Sea, n=9 mean ± SD (min – max)
Naphthalene	37.2 ± 29.0 (2.8 – 88.5)	14.9 ± 16.1 (<10.0 – 57.5)	6.8 ± 7.2 (0.3 – 33.1)	14.2 ± 8.0 (4.6 – 28.6)
C_1-Naphthalenes	178 ± 128 (15.2 – 428)	32.8 ± 24.6 (11.8 – 89.7)	32.0 ± 34.8 (0.5 – 169)	39.5 ± 27.0 (5.6 – 104)
C_2-Naphthalenes	439 ± 323 (41.7 – 911)	24.4 ± 28.2 (3.1 – 96.9)	54.9 ± 73.7 (1.4 – 367)	67.5 ± 52.9 (8.5 – 156)
C_3-Naphthalenes	527 ± 375 (47.9 – 1095)	39.1 ± 80.5 (1.9 – 264)	61.6 ± 87.1 (1.2 – 402)	138 ± 184 (5.0 – 449)
Phenanthrene	89.6 ± 51.3 (17.9 – 236)	31.2 ± 29.1 (8.2 – 92.4)	19.6 ± 20.1 (0.8 – 108)	25.1 ± 20.4 (5.7 – 68.6)
Anthracene	2.5 ± 1.9 (0.3 – 6.4)	2.6 ± 2.3 (0.6 – 7.1)	0.9 ± 0.9 (<0.3 – 5.0)	0.8 ± 0.6 (0.3 – 2.0)
C_1-Phenanthrenes	266 ± 180 (45.0 – 538)	19.9 ± 17.1 (3.0 – 55.0)	36.5 ± 45.7 (1.4 – 184)	44.2 ± 29.3 (9.2 – 97.0)
C_2-Phenanthrenes	341 ± 298 (42.2 – 845)	11.5 ± 19.7 (0.5 – 65.2)	42.2 ± 71.1 (0.3 – 403)	41.0 ± 34.5 (9.7 – 99.6)
C_3-Phenanthrenes	307 ± 252 (51.3 – 749)	6.6 ± 10.6 (<0.7 – 29.1)	32.1 ± 49.4 (0.1 – 300)	38.0 ± 20.7 (8.3 – 67.2)
Dibenzothiophene	11.4 ± 8.5 (1.0 – 28.3)	3.1 ± 3.0 (0.8 – 9.5)	1.9 ± 2.5 (0.1 – 15.6)	2.5 ± 3.0 (0.7 – 8.7)
C_1-Dibenzothiophenes	33.6 ± 21.9 (7.5 – 107)	3.7 ± 3.0 (0.5 – 8.9)	7.3 ± 12.7 (0.1 – 73.7)	9.1 ± 4.7 (3.5 – 16.4)
C_2-Dibenzothiophenes	78.1 ± 60.0 (8.2 – 238)	2.0 ± 3.0 (0.3 – 9.6)	11.9 ± 18.0 (0.1 – 90.3)	8.0 ± 4.4 (2.9 – 14.5)
C_3-Dibenzothiophenes	91.4 ± 105 (<0.7 – 486)	27.2 ± 65.3 (<0.7 – 201)	13.0 ± 25.1 (0.1 – 183)	3.2 ± 2.4 (<0.7 – 6.8)
Acenaphtylene	2.2 ± 3.1 (0.1 – 14.2)	2.5 ± 3.5 (0.3 – 8.9)	0.4 ± 0.5 (<0.3 – 2.1)	2.7 ± 4.2 (0.1 – 11.3)
Acenapthene	4.8 ± 4.9 (0.4 – 21.5)	1.6 ± 2.1 (0.3 – 6.8)	0.9 ± 1.6 (<0.3 – 9.2)	1.5 ± 2.2 (0.1 – 6.7)
Fluorene	34.2 ± 33.0 (1.6 – 98.8)	14.3 ± 16.4 (0.8 – 41.0)	3.5 ± 4.8 (0.1 – 30.8)	6.4 ± 8.2 (0.6 – 19.4)
Fluoranthene	48.4 ± 23.5 (7.0 – 78.9)	32.1 ± 30.5 (6.0 – 89.6)	11.7 ± 11.6 (0.8 – 57.0)	15.2 ± 7.9 (6.2 – 30.8)
Pyrene	43.3 ± 22.1 (12.8 – 87.4)	9.5 ± 7.9 (1.8 – 27.6)	9.4 ± 10.0 (0.7 – 55.2)	11.6 ± 5.2 (6.3 – 20.2)
Benz[*a*]anthracene	13.0 ± 5.2 (4.7 – 24.5)	10.8 ± 12.8 (1.0 – 41.6)	4.3 ± 3.9 (0.2 – 20.1)	5.4 ± 3.2 (2.0 – 11.0)
Chrysene	80.3 ± 57.2 (11.5 – 168)	47.9 ± 48.7 (6.5 – 155)	16.8 ± 15.7 (0.9 – 77.9)	17.8 ± 8.5 (7.4 – 30.5)
Benzo[*b,j,k*]fluoranthene[d]	112 ± 44.1 (29.4 – 204)	125 ± 131 (14.5 – 393)	47.5 ± 32.7 (3.7 – 150)	101 ± 65.8 (20.7 – 214)
Benzo[*e*]pyrene	99.2 ± 58.4 (13.6 – 207)	44.5 ± 52.8 (6.7 – 156)	23.1 ± 20.1 (1.6 – 103)	24.1 ± 12.9 (9.5 – 44.3)
Benzo[*a*]pyrene	25.6 ± 12.8 (5.9 – 56.8)	18.1 ± 22.5 (1.4 – 66.5)	7.4 ± 7.9 (0.6 – 51.3)	10.6 ± 8.1 (4.1 – 27.5)
Perylene	102 ± 147 (11.8 – 700)	63.7 ± 79.7 (2.7 – 227)	38.3 ± 54.4 (2.4 – 374)	185 ± 133 (45.0 – 420)
Benzo[*ghi*]perylene	93.9 ± 51.3 (4.3 – 204)	57.9 ± 77.9 (2.9 – 238)	28.6 ± 22.6 (1.7 – 91.0)	21.2 ± 14.1 (2.5 – 42.5)
Indeno[1,2,3-*cd*]pyrene	38.4 ± 16.7 (5.4 – 68.6)	109 ± 155 (3.3 –466)	28.4 ± 22.3 (1.8 – 126)	19.3 ± 11.8 (3.4 – 40.8)
Dibenz[*a,h*]anthracene	14.8 ± 8.4 (<1.5 – 29.6)	15.8 ± 22.0 (<1.5 – 61.7)	5.3 ± 4.2 (0.2 – 22.9)	3.5 ± 2.4 (<1.0 – 7.3)
Sum PAH	3110 ± 1959 (714 – 6026)	769 ± 697 (137 – 2147)	542 ± 552 (26.9 – 2698)	856 ± 284 (450 – 1138)

[a] Benzo[*b,j,k*]fluoranthene determined in 21 samples only; [b] C_3-dibenzothiophenes determined in 9 samples only; [c] C_1-naphthalenes determined in 78 samples while benzo[*e*]pyrene and perylene determined in 74 samples only; [d] sum of benzo[*b*]fluoranthene, benzo[*j*]fluoranthene and benzo[*k*] fluoranthene.

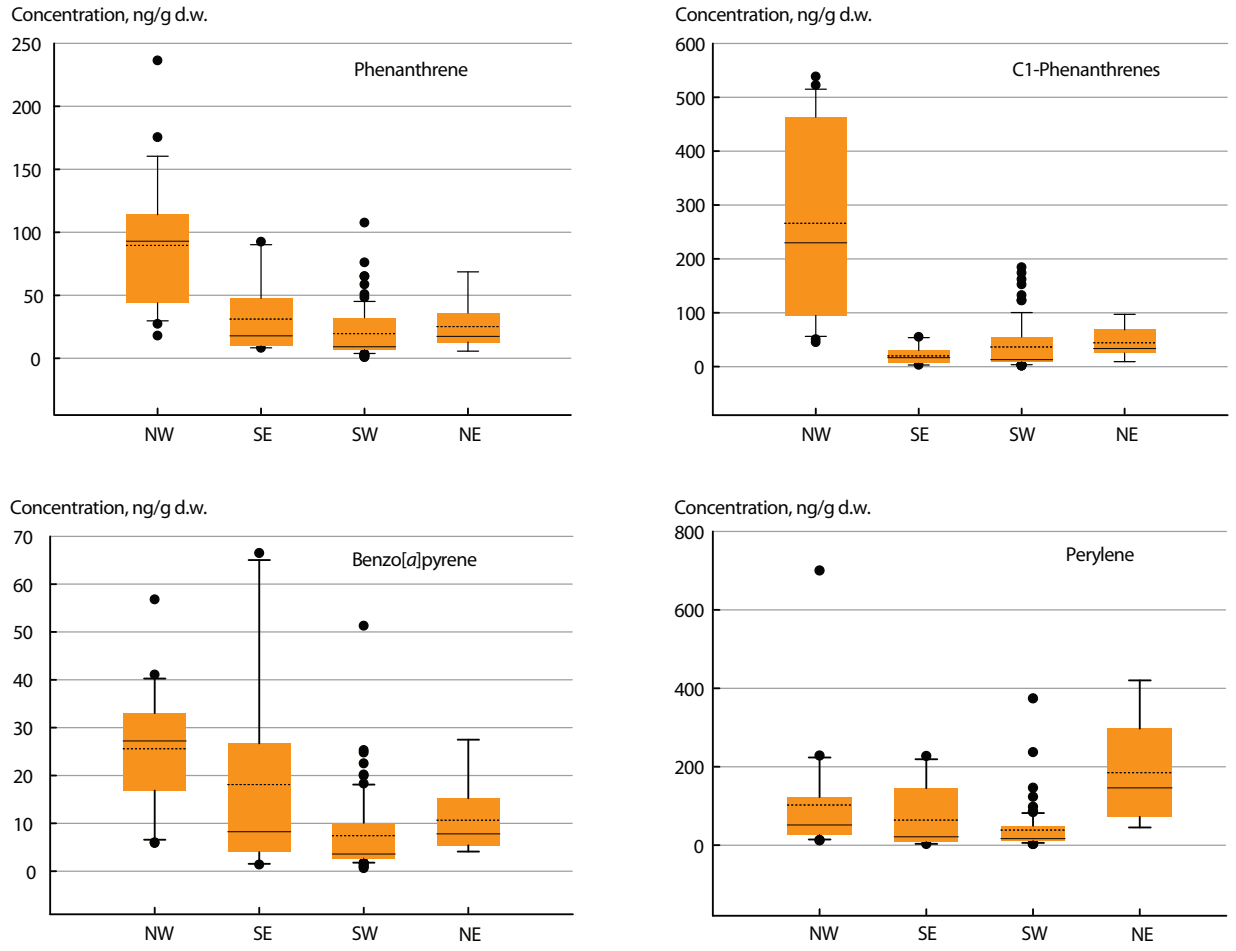


Figure 4.21. Distribution of concentrations of selected PAHs in surface sediment (0-2 cm) samples from the Barents Sea, 2001-2005. Cruises carried out by different institutions (data: Akvaplan-niva, IMR, PINRO and VNIIOkeangeologia).

Institute for Geology and Mineral Resources of the World Ocean (VNIIOkeangeologia) (Savinov et al., 2006; Boitsov et al., 2007). Summary statistics for the combined data set are presented in Table 4.34. and Figure 4.21.

To reveal possible multi-year variations in PAH concentration in surficial sediments, several data sets from the years 2001–2005 have been compared with the data sets collected during the 1990s (Dahle et al., 2008a).

On the basis of the concentrations and composition of the PAHs, the Barents Sea can be divided into five geographical areas: the north-western part, which can be sub-divided between the Svalbard offshore area and the Bear Island Trough area, and the south-western, south-eastern and north-eastern parts of the sea (Dahle et al., 2008a).

In 2001–2005, the highest total PAH concentrations (2698–6026 ng/g dw) were found in Svalbard offshore sediments. Alkyl-substituted homologues of naphthalene and phenanthrene dominated the PAH composition. Comparative studies of bottom sediments from the Svalbard offshore area and soils from West Spitsbergen

Table 4.35. Geometric mean of PAH concentrations (ng/g dw) and PAH molecular mass ratios in bottom sediments from five areas of the Barents Sea collected during the 1990s (1991–1998) and the 2000s (2001–2005).

	Svalbard offshore		Bear Island Trough		SW Barents Sea		SE Barents Sea		NE Barents Sea	
	2000s	1990s	2000s	1990s	2000s	1990s	2000s	1990s	2000s	1990s
	n=13	n=9	n=23	n=14	n=39	n=11	n=14	n=21	n=17	n=17
ΣPAH[a]	4383	3735	1324	1130	178	129	319	169	689	575
ΣPyrPAH[b]	780	598	365	341	88	81	207	96	242	260
NPD[c]	3466	3048	851	691	67	32	88	37	291	184
FLT/202[d]	0.525	0.510	0.539	0.562	0.543	0.567	0.763	0.642	0.520	0.572
IND/276[e]	0.235	0.227	0.403	0.323	0.531	0.585	0.650	0.553	0.450	0.484

n: Number of samples; [a]ΣPAH: sum of naphthalene, phenanthrene, dibenzothiophenes and their alkyl-substituted homologues, acenaphthylene, acenaphtene, fluorine, fluoranthene, pyrene, benz[*a*]anthracene, chrysene, benzo[*b*+*k*]fluoranthene, benzo[*e*]pyrene, benzo[*a*]pyrene, perylene, benzo[*ghi*] perylene, indeno[1,2,3-*cd*]pyrene and dibenzo[*a,h*]anthracene; [b]ΣPyrPAH: sum of PAHs with four- to five-ring hydrocarbons, perylene excluded; [c]NPD sum of naphthalene, phenanthrene, dibenzothiophenes and their alkyl-substituted homologues; [d]FLT/202: fluoranthene/fluoranthene+pyrene. [e]IND/276: indeno[1,2,3-*cd*]pyrene/ indeno[1,2,3-*cd*]pyrene+benzo[*ghi*]perylene.

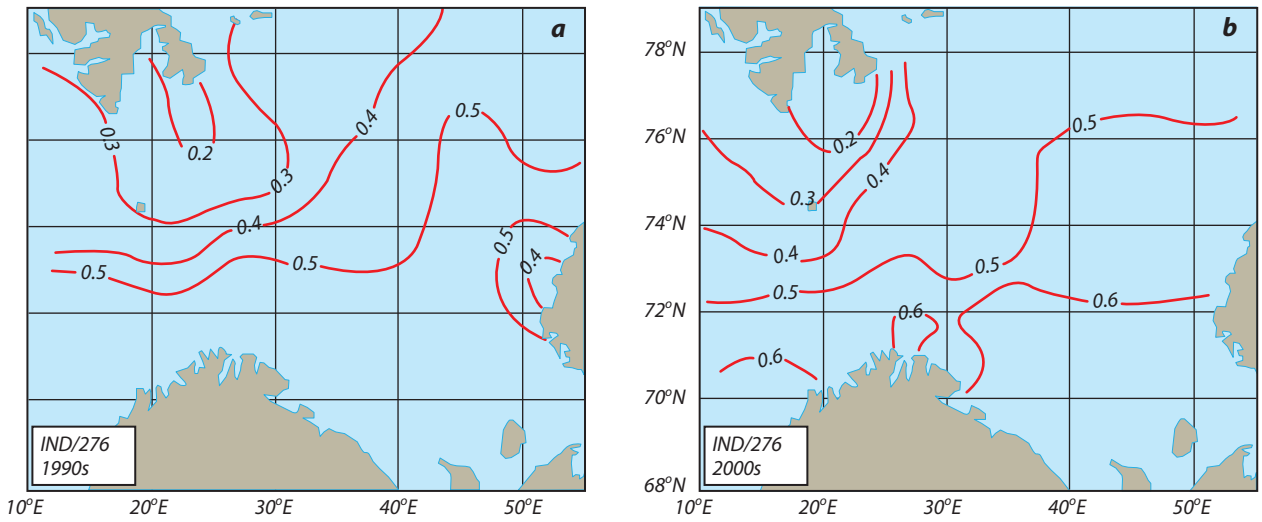


Figure 4.22. Distribution of the IND/276 ratio in bottom sediment samples collected from the Barents Sea in (a) the 1990s and (b) the 2000s.

Island indicated that the predominant source of PAHs is the erosion of coal-bearing bedrock in Svalbard (Dahle et al., 2006a). Although the total concentration of pyrogenic PAHs found in the Svalbard offshore area was high, low PAH ratios (Table 4.35) reveal that these pyrogenic PAHs are mainly produced in natural low-temperature processes. There are no statistically significant differences in PAH concentrations or PAH ratios between samples from the 1991–1998 and 2001–2005 datasets (Table 4.35). There is no oil industry activity on or near Svalbard, but coal mines have been exploited there for decades.

The Bear Island Trough is a deep (400–500 m) area of the Barents Sea strongly influenced by Atlantic Water. Total PAH concentrations in sediments were 900–2200 ng/g dw (2001–2005). The PAH composition is similar to that found on the Svalbard offshore shelf (i.e., predominantly 2- and 3-ring PAH compounds reflecting the strong influence of the weathering of coal-bearing rocks of Svalbard). However, sediments in the Bear Island Trough differ from those at Svalbard in that they contain a higher relative concentration of pyrogenic PAHs. Again, there are no significant differences between the 1991–1998 and 2001–2005 datasets of total PAH concentrations found in this area (Table 4.35).

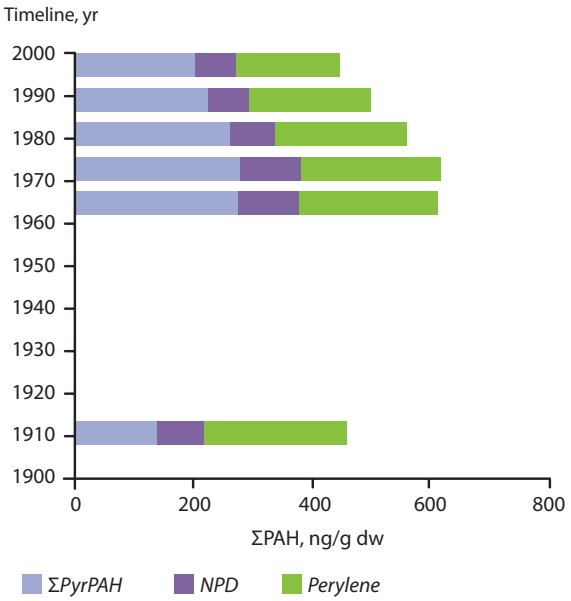


Figure 4.23. Vertical PAH profile in a sediment core from the north-eastern Barents Sea retrieved in 2003 (74°58.0′ N: 46°59.8′ E). Sedimentation rate is 1.2 mm/yr.

In the south-western part of the Barents Sea, total PAH concentrations fall within the range 27–467 ng/g dw. The PAHs have a pyrogenic profile; benzofluoranthenes, indeno[1,2,3-*cd*]pyrene and benzo[*ghi*]perylene being the most abundant compounds. High PAH ratios indicate that most of these pyrogenic PAHs have been produced in high-temperature processes. There are no significant differences in the total concentrations of pyrogenic PAHs and PAH ratios between the 1991–1998 and 2001–2005 datasets (Table 4.35). However, the concentration of 2- and 3-ring PAHs has significantly increased compared with the 1990s (Table 4.35). This increase in concentrations of the petrogenic PAHs may be related to the influx of oil contamination with Atlantic and coastal currents and to increased shipping (including transport of oil) in these waters.

In the south-eastern Barents Sea in 2003, the range of total PAH concentrations is 137–861 ng/g dw. Pyrogenic compounds again dominate the PAH composition; indeno[1,2,3-*cd*]pyrene, benzofluoranthenes and benzo[*ghi*] perylene are the most abundant compounds. PAH ratios are significantly higher compared to the other four areas (Table 4.35). Comparison with the PAH ratios in the south-western Barents Sea indicate that the south-eastern Barents Sea has an influx of combustion or anthropogenic PAHs. Geometric mean concentrations of the sum of pyrogenic PAHs and 2- and 3-ring aromatic hydrocarbons, as well as PAH ratios in the south-eastern part of the Barents Sea in 2003 are significantly higher than those during 1991–1998 (Table 4.35).

The north-eastern part of the Barents Sea includes the eastern part of the Great Bank and the northern basin of the Central Deep. These areas are flushed by both Arctic and Atlantic water. In 2003, total PAH concentrations in the area were 406–1136 ng/g dw, with perylene and benzofluoranthenes the most abundant. PAH ratios are similar to those in the Bear Island Trough suggesting the possible impact of combustion PAH brought to the north-eastern part of the Barents Sea with Atlantic Water. This is also illustrated by the geographical distribution of the indeno[1,2,3-*cd*]pyrene/indeno[1,2,3-*cd*]pyrene + benzo[*ghi*] perylene ratio (IND/276) (Figure 4.22), high values of which occur along the route of the southern branch of the North Cape current

In bottom sediments from the north-eastern part of the Barents Sea, the PAH concentrations found in 2003 were similar to those measured during the 1990s. The limited temporal variability in PAH concentrations is supported by analyses of sediment cores that show only small changes in PAH concentrations during the past 40 years (Figure 4.23) (Dahle et al., 2008a).

4.6.3.2.6. Russia

4.6.3.2.6.1. White Sea

Total hydrocarbon concentrations in bottom sediments from the seaward areas of White Sea bays and from the central White Sea were in the range 4.2–33.2 μg/g dw (Dahle et al., 1998). Concentrations are generally higher in the inner part of the bays and Klyujkov (1997) reported a total hydrocarbon concentration of 330 μg/g dw in sediment from the head of Kandalaksha Bay.

Concentrations of PAHs (sum of the 2- to 6-ring PAHs) in sediments from the open sea and outer parts of the bays were in the range 13–208 ng/g dw in 1994. The highest concentration of PAH was found in Dvina Bay sediments which were dominated by petrogenic PAHs (naphthalenes, phenanthrenes and dibenzothiophenes) (Savinov et al., 2000). In 2006, the highest total PAH concentration (2925 nglg dw) was found in sediments collected close to the Kandalaksha Harbour (Kandalaksha Bay)(Savinova et al., 2007); pyrogenic PAHs were prevalent (Table 4.36).

4.6.3.2.6.2. Kara Sea

Korshenko et al. (2003) reported total hydrocarbon concentrations of 6.7–7.8 μg/g dw in bottom sediments collected from the Kara Sea in 2002. These values are substantially lower than the range of 44–160 μg/g dw for Ob Bay and maximum values of about 84 μg/g dw for the open Kara Sea reported by RCMA (2005) for samples collected in 1999–2005. The lowest concentrations of paraffins (150–400 ng/g dw) reported by Korshenko et al. (2003) were found in sediments from shallow coastal waters. The hydrodynamic environment of this area is defined by the coastal and mixed waters of the Ob-Yenisey current and the eastern Novaya Zemlya currents. In the north-western part of the Kara Sea, the hydrology is defined by the water exchange between the Barents Sea and Arctic Ocean through the St. Anna Trough. In this area, there are higher concentrations of paraffins in the sediments (500–800 ng/g dw). The highest concentrations of total paraffins (1000–7000 ng/g dw) were found in sediments from the central part of the Kara Sea known for its oil and gas structures (Rusanovskaya, Leningradskaya and Obruchevskaya) (Ilyin, 2005). Biogenic n-C_{15}–C_{19} and C_{23}–C_{29} compounds generally predominate in coastal shallow water sediments and in the north-eastern part of the sea, whereas C_{20}–C_{26} paraffins are most abundant in sediments from the central Kara Sea (Ilyin, 2005).

Figure 4.24 shows the distribution of PAH concentrations in sediments of the Kara Sea in the 1990s. Total PAH levels in sediments are generally low, but

Table 4.36. PAH concentrations in surface sediment samples (ng/g dw) from the Kara Sea and White Sea, 2001–2006 (data: Akvaplan-niva and PINRO).

	Southwest Kara Sea, n= 3 mean ± SD (min – max)	Kara Sea – Novaya Zemlya, n=7 mean ± SD (min – max)	Pechora Sea, n=6 mean ± SD (min – max)	White Sea, n=3 mean ± SD (min – max)
Naphthalene	61.9 ± 53.6 (0.1 – 93.5)	17.4 ± 8.2 (6.3 – 31.4)	13.0 ± 14.1 (<0.2 – 34.2)	4.7 ± 8.2 (<10.0 – 14.2)
C_1-naphthalenes	89.4 ± 92.1 (10.4 – 191)	51.1 ± 29.3 (12.8 – 102)	43.5 ± 26.9 (16.5 – 84.1)	15.3 ± 13.7 (5.0 – 30.8)
C_2-naphthalenes	60.6 ± 48.5 (18.8 – 114)	46.7 ± 26.7 (11.0 – 87.0)	15.1 ± 12.6 (4.2 – 33.7)	<5.0 ± 0.0 (<5.0 – <5.0)
C_3-naphthalenes	35.1 ± 26.8 (11.6 – 64.3)	34.2 ± 13.4 (20.5 – 60.9)	9.9 ± 4.4 (2.5 – 14.8)	<10.0 ± 0.0 (<10.0 – <10.0)
Phenanthrene	13.9 ± 5.3 (8.1 – 18.5)	16.5 ± 5.5 (9.1 – 25.2)	27.1 ± 15.4 (8.7 – 49.9)	35.6 ± 57.8 (2.0 – 102.3)
Anthracene	1.2 ± 1.3 (0.4 – 2.7)	0.8 ± 0.6 (0.1 – 1.5)	3.2 ± 1.6 (1.4 – 5.5)	6.9 ± 12.0 (<0.3 – 20.8)
C_1-Phenanthrenes	20.6 ± 5.2 (15.3 – 25.6)	12.3 ± 5.6 (6.1 – 21.9)	11.8 ± 6.7 (4.1 – 23.2)	23.9 ± 41.3 (<0.5 – 71.6)
C_2-Phenanthrenes	25.9 ± 20.0 (11.3 – 48.7)	11.0 ± 9.3 (4.6 – 31.2)	2.1 ± 1.8 (0.8 – 5.6)	12.8 ± 22.1 (<0.5 – 38.4)
C_3-Phenanthrenes	9.9 ± 9.6 (2.7 – 20.8)	2.8 ± 4.3 (0.1 – 12.1)	0.4 ± 0.2 (0.3 – 0.8)	11.4 ± 19.7 (<0.7 – 34.1)
Dibenzothiophene	0.9 ± 0.8 (0.1 – 1.7)	1.0 ± 0.3 (0.4 – 1.2)	2.8 ± 1.9 (1.1 – 6.1)	2.1 ± 3.6 (<0.3 – 6.3)
C_1-Dibenzothiophenes	0.9 ± 0.9 (0.1 – 1.9)	1.7 ± 0.7 (0.3 – 2.4)	2.6 ± 1.8 (1.1 – 6.0)	2.2 ± 3.8 (<0.5 – 6.6)
C_2-Dibenzothiophenes	0.4 ± 0.3 (0.1 – 0.6)	0.9 ± 0.8 (0.1 – 2.5)	0.3 ±0.0 (0.3 – 0.3)	<0.5 ± 0.0 (<0.5 – <0.5)
C_3-Dibenzothiophenes	0.1 ± 0.0 (0.1 – 0.1)	0.1 ± 0.0 (0.1 – 0.1)	55.0 ± 74.4 (1.3 – 192)	<0.7 ± 0.0 (<0.7 – <0.7)
Acenaphtylene	0.3 ± 0.0 (0.3 – 0.4)	0.6 ± 0.3 (0.3 – 1.2)	2.1 ± 2.4 (0.3 – 6.7)	9.5 ± 15.4 (<0.3 – 27.2)
Acenaphthene	0.5 ± 0.4 (0.1 – 0.9)	1.4 ± 2.1 (0.3 – 6.2)	2.8 ± 2.0 (0.3 – 6.4)	2.7 ± 4.7 (<0.3 – 8.2)
Fluorene	3.4 ± 2.0 (2.1 – 5.8)	5.0 ± 4.6 (1.8 – 14.8)	16.3 ± 11.8 (4.4 – 37.9)	3.4 ± 6.0 (<0.3 – 10.3)
Fluoranthene	13.7 ± 2.1 (11.4 – 15.5)	15.3 ± 7.2 (5.8 – 27.5)	51.9 ± 41.8 (13.8 – 124)	71.5 ± 117 (3.7 – 206.5)
Pyrene	7.9 ± 1.1 (6.9 – 9.1)	5.2 ± 4.5 (1.9 – 15.0)	10.0 ± 6.9 (3.6 – 22.8)	59.8 ± 97.7 (3.2 – 172.6)
Benz[*a*]anthracene	2.6 ± 1.0 (1.5 – 3.5)	1.7 ± 1.3 (0.4 – 4.3)	17.6 ± 15.2 (2.5 – 38.1)	46.1± 65.5 (6.7 – 122)
Chrysene	13.3 ± 5.0 (7.5 – 16.3)	8.4 ± 5.1 (2.0 – 16.6)	83.8 ± 66.7 (16.6 – 170)	196 ± 313 (8.7 – 557)
Benzo[*b,j,k*]fluoranthene [a]	35.4 ± 14.2 (19.0 – 44.0)	17.2 ± 15.6 (4.0 – 42.7)	200 ± 217 (33.6 – 614)	229 ± 326 (34.3 – 606)
Benzo[*e*]pyrene	14.6 ± 7.4 (6.3 – 20.5)	7.1 ± 7.8 (1.7 – 22.9)	69.5 ± 77.1 (13.6 – 217)	113 ± 168 (12.7 – 307)
Benzo[*a*]pyrene	2.4 ± 2.2 (0.7 – 4.9)	2.2 ± 3.7 (0.1 – 10.6)	28.4 ± 31.2 (5.1 – 87.1)	55.1 ± 90.8 (<0.5 – 160)
Perylene	86.4 ± 20.6 (70.0 – 110)	10.3 ± 12.9 (0.1 – 32.9)	62.9 ± 73.7 (8.2 – 207)	22.0 ± 35.6 (<0.5 – 63.1)
Benzo[*ghi*]perylene	12.5 ± 6.3 (7.1 – 19.4)	2.2 ± 3.7 (0.1 – 9.3)	102 ± 115 (17.7 – 322)	60.1 ± 104 (<1.5 – 180)
Indeno[1,2,3-*cd*]pyrene	13.2 ± 9.1 (7.4 – 23.7)	2.2 ± 4.4 (0.1 – 11.9)	192 ± 213 (38.8 – 600)	60.1 ± 104 (<1.5 – 180)
Dibenz[*a,h*]anthracene	2.3 ± 2.0 (1.0 – 4.6)	0.3 ± 0.7 (0.1 – 1.9)	30.2 ± 34.0 (5.1 – 95.4)	<1.5 ± 0.0 (<1.5 – <1.5)
Sum PAH	529 ± 187 (323 – 688)	276 ± 32.8 (234 – 324)	1056 ± 889 (309 – 2576)	1044 ± 1629 (90.0 – 2925)

[a] Sum of benzo[*b*]fluoranthene, benzo[*j*]fluoranthene and benzo[*k*]fluoranthene.

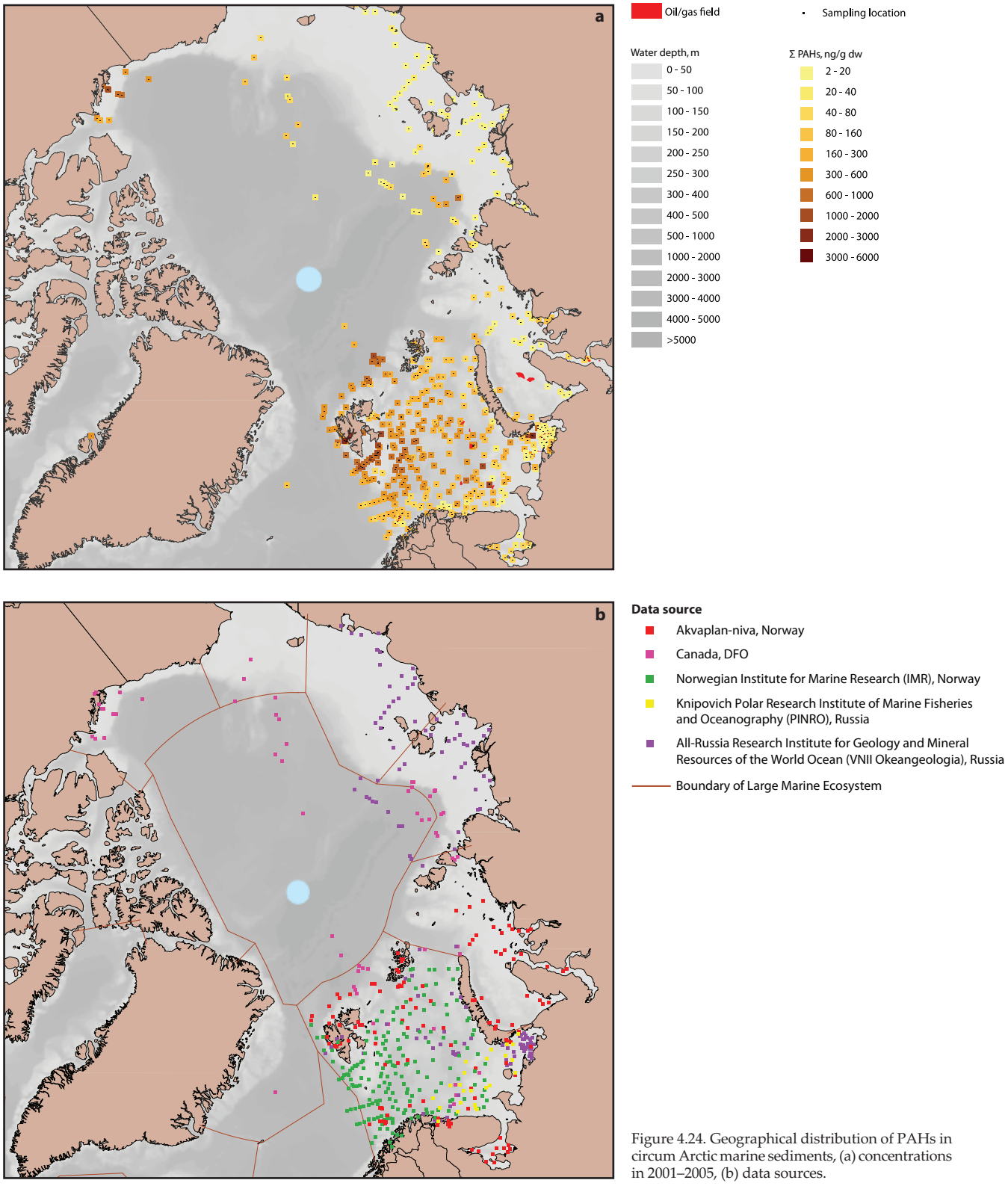


Figure 4.24. Geographical distribution of PAHs in circum Arctic marine sediments, (a) concentrations in 2001–2005, (b) data sources.

some relatively high concentrations have been measured at certain sites. The highest concentrations occur in coastal sediments including the Ob and Yenisey estuaries and decrease towards deeper waters. The most pronounced decrease is seen for perylene; concentrations in marine sediments are only 47–67% of those in the estuarine sediments (Petrova, 2001). Suspended matter in the Ob and Yenisey estuaries and Kara Sea contain low concentrations of PAH (Fernandes and Sicre, 1999). In 2002, Korshenko et al. (2003) reported the following average PAH concentrations (ng/g dw) in Kara Sea bottom sediments: naphthalene, 13.6; 2-methylnaphthalene, 8.4; fluorene 2.8; phenanthrene, 5.2; anthracene, 0.7; fluoranthene, 2.1; pyrene, 0.6; chrysene, 0.8; benzo[*b*]fluoranthene, 1.3; benzo[*k*]fluoranthene, 1.1; and benzo[*a*]pyrene, 0.3. As noted for total hydrocarbon data, concentrations reported by RCMA (2005) for samples collected between 1999 and 2005 are higher than those reported by Korshenko et al. (2003); for example total PAHs in Yenisey gulf sediments were as high as 587 ng/g dw.

Recent measurements of PAHs in the Kara Sea sediments from 2003–2004 (Savinov et al., 2006), indicate that the highest PAH concentration was found in sediments from the Novaya Zemlya Trough. The average PAH concentration (632 ng/g dw) was four times higher than that in sediments from the Kara Sea shelf but similar to that in Yenisey Bay sediments. The Novaya Zemlya Trough is the deepest area of the Kara Sea (>350 m). The grain size of the sediments in the trough is predominantly pelite. According to a vertical PAH profile taken from a sediment core from the southern part of Novaya Zemlya Trough, a two-fold increase in total PAH concentrations occurred during the last decade. Petrogenic PAHs account for most of the increase (Savinov et al., 2006).

Bottom sediments from the bays along the eastern coast of Novaya Zemlya have comparable PAH concentrations, but differ in PAH composition (Table 4.36). Petrogenic PAH compounds are most abundant in sediments from a bay located in the northern part of the southern island of Novaya Zemlya whereas pyrogenic PAHs predominate in the south (Savinov et al., 2006).

4.6.3.2.6.3. Laptev Sea and East Siberian Sea

Total hydrocarbon concentrations in bottom sediments from the coastal areas of the Laptev Sea were in the range 10–180 μg/g dw (Lobchenko, 1997). The highest concentrations were found in Guba Buor Khaya close to Tiksi Harbour (180 μg/g dw) and in Khatanga Bay (86 μg/g dw).

Total PAH concentrations in coastal areas of the Laptev Sea were in the range 13–40 ng/g dw. These concentrations are very similar to the range of 16–60 ng/g dw reported by RCMA (2005). Concentrations of single PAHs (ng/g dw) were: fluorene, 0.2–3.2; phenanthrene, 0.0–10.1; fluoranthene, 0.2–6.2; benzofluoranthene, 3.2–18.3; benzo[*ghi*]perylene, 0.1–12.4; and benzo[*a*]pyrene, 0–0.24 (Lobchenko, 1997).

Other studies of bottom sediments of the eastern Arctic Shelf, report total PAH concentrations of 3–80 ng/g dw (average 40 ng/g dw), which are lower than those in sediments from the western Arctic Shelf (Petrova, 1990, 2001; Yunker et al., 1996; Dahle et al., 2003, 2006a). Low PAH contents (<20 ng/g dw) with phenanthrene and its alkylated homologues most abundant were found in deep-water sediments containing organic matter of marine biological origin. In coastal shelf sediments, PAH concentrations varied with the origin of organic matter; perylene and alkylated homologues of phenanthrene and chrysene. In general, studies to date reveal the presence of low amounts of pyrogenic and other PAHs in the sediments of the Laptev Sea and eastern Siberian Sea shelf.

Table 4.37. Concentrations of the Σparent PAH (ng/g dw) and PAH ratios in surface bottom sediments from the Arctic seas (Dahle et al., 2008b).

	[1]Σparent PAH[a]		ANT/(ANT+PHE)		FLT/(FLT+PYR)		IND/(IND+BP)		BAA/(BAA+CHR)	
	n	mean ± SD (min – max)	n	mean ± SD	n	mean ± SD	n	mean ± SD	n	mean ± SD
Arctic Ocean (west)	19	607 ± 267 (120 – 1243)	17	0.02 ± 0.01	19	0.49 ± 0.06	18	0.51 ± 0.22	19	0.10 ± 0.02
Arctic Ocean (east)	7	22 ± 7 (12.9 – 33.4)	0	–	1	0.50	1	0.38	6	0.32 ± 0.05
Canadian Arctic	6	105 ± 53 (43.7 – 157)	4	0.03 ± 0.01	6	0.46 ± 0.07	6	0.31 ± 0.08	5	0.13 ± 0.07
Mackenzie Delta	4	597 ± 102 (511 – 745)	4	0.01 ± 0.00	4	0.35 ± 0.02	4	0.16 ± 0.00	4	0.17 ± 0.02
Beaufort Sea	12	748 ± 376 (228 – 1391)	12	0.03 ± 0.02	12	0.37 ± 0.03	12	0.17 ± 0.06	12	0.17 ± 0.04
Chukchi Sea	3	90 ± 32 (66.3 – 127)	2	0.04 ± 0.02	3	0.47 ± 0.04	3	0.23 ± 0.05	3	0.17 ± 0.07
East-Siberian Sea (east)	7	18 ± 13 (1.9 – 44.2)	0	–	0	–	0	–	7	0.28 ± 0.06
East-Siberian Sea (west)	20	11 ± 11 (1.8 – 40.5)	0	–	0	–	0	–	20	0.24 ± 0.08
Laptev Sea deep water	24	129 ± 150 (3.0 – 640)	13	0.03 ± 0.01	16	0.57 ± 0.09	15	0.60 ± 0.16	23	0.19 ± 0.11
Laptev Sea shelf (<100 m depth)	24	18 ± 11 (3.7 – 44.0)	0	–	0	–	0	–	24	0.24 ± 0.08
Kara Sea (northeast)	13	66 ± 49 (6.8 – 187)	11	0.06 ± 0.03	13	0.65 ± 0.08	13	0.33 ± 0.23	13	0.15 ± 0.03
Kara Sea (southwest)	13	79 ± 93 (2.6 – 267)	13	0.06 ± 0.03	13	0.64 ± 0.16	13	0.42 ± 0.19	13	0.21± 0.07
Yenisey Estuary	4	129 ± 26 (97.3 – 155)	4	0.08 ± 0.01	4	0.57 ± 0.03	4	0.34 ± 0.17	4	0.20 ± 0.01
Ob Estuary	5	69 ± 45 (22.4 – 139)	5	0.15 ± 0.09	5	0.62 ± 0.06	5	0.22 ± 0.20	5	0.19 ± 0.11
Kara Sea (Novaya Zemlya bays)	7	104 ± 55 (61.8 – 205)	7	0.05 ± 0.04	7	0.76 ± 0.06	7	0.50 ± 0.14	7	0.16 ± 0.02
Barents Sea (northeast)	34	203 ± 114 (31.8 – 490)	29	0.05 ± 0.05	34	0.59 ± 0.14	31	0.45 ± 0.13	34	0.21 ± 0.09
Barents Sea (FJL nearshore)	19	191 ± 70 (61.6 – 374)	16	0.14 ± 0.20	19	0.53 ± 0.10	15	0.45 ± 0.20	19	0.23 ± 0.07
Barents Sea (northwest)	86	664 ± 518 (53.5 – 3138)	78	0.02 ± 0.02	86	0.48 ± 0.09	74	0.31 ± 0.14	86	0.17 ± 0.09
Barents Sea (Svalbard inshore)	20	2144 ±965 (600 – 3632)	20	0.03 ± 0.01	20	0.48 ± 0.05	19	0.20 ± 0.04	20	0.21 ± 0.06
Barents Sea (southeast)	47	281 ± 327 (7.5 – 1698)	34	0.21 ± 0.20	46	0.64 ± 0.13	34	0.56 ± 0.15	47	0.30 ± 0.16
Pechora Sea	59	156 ± 347 (3.9 – 2289)	50	0.11 ± 0.08	59	0.56 ± 0.20	47	0.51 ± 0.11	59	0.38 ± 0.17
Barents Sea (southwest)	131	206 ± 167 (13.5 – 968)	119	0.09 ± 0.09	131	0.58 ± 0.09	123	0.51 ± 0.11	131	0.24 ± 0.09
Barents Sea (Kola and Pechenga bays)	12	1988 ± 1781 (251 – 6187)	12	0.37 ± 0.14	12	0.58 ± 0.05	7	0.41 ± 0.05	12	0.42 ± 0.06
White Sea	11	54 ± 34 (8.4 – 113)	11	0.22 ± 0.17	11	0.61 ± 0.03	11	0.23 ± 0.03	11	0.26 ± 0.02
Greenland Sea	1	142	1	0.02	1	0.55	1	0.59	1	0.11
Baffin Bay, WS Greenland coast	17	118 ± 132	17	0.06 ± 0.04	17	0.48 ± 0.08	17	0.49 ± 0.10	17	0.28 ± 0.12

ANT=anthracene, PHE=phenanthrene, FLT=fluoranthene, PYR=pyrene, IND=indeno[1,2,3-*cd*]pyrene, BP=benzo[*ghi*]perylene, BAA=benz[*a*]anthracene, CHR=chrysene. [a]Σparent PAH: sum of naphthalene, acenaphthylene, acenaphthene, fluorene, phenanthrene, anthracene, pyrene, fluoranthene, chrysene, benz[*a*]anthracene, benzo[*e*]pyrene, benzo[a]pyrene, benzo[*b,j,k,*]fluoranthene, benzo[*ghi*]perylene, indeno[1,2,3-*cd*]pyrene and dibenzo[*a,h*]anthracene.

4.6.3.2.7. Circumpolar summary

As a large number of studies have shown that PAH concentrations in most of the Barents Sea have varied little over the past decade (Dahle et al., 2008a), this assessment has combined data for the 1990s and 2000s to allow comparisons with concentrations measured in other Arctic seas over this period (1991–2005) (Dahle et al., 2008b) (Table 4.37). The regional distribution of total parent PAHs (ΣParentPAH) in the Barents Sea indicates a decrease from Svalbard (average 2144 ng/g dw) to the south-east (Pechora Sea, 156 ng/g dw) (Table 4.37). A decrease in ΣParentPAH concentrations continues through the Kara Sea (66–129 ng/g dw) and eastward into the Laptev Sea, East-Siberian Sea and Chukchi Sea, where ΣParentPAH concentrations are in the range 11–129 ng/g dw. This decrease in ΣParentPAH levels in the eastern Arctic Seas is accompanied by a decrease in PAH isomer pair ratios (Table 4.37) associated with a predominance of petrogenic PAHs (Yunker et al., 1996). In contrast, high PAH ratios are typical of the White Sea and south-eastern part of the Barents Sea due to a prevalence of combustion and/or anthropogenic sources (Table 4.37).

Levels of ΣParent PAH in the Beaufort Sea and Mackenzie River delta (748 ng/g and 597 ng/g dw, respectively) are similar to those measured in the north-western part of the Barents Sea and adjacent western part of the Arctic Ocean (664 ng/g and 607 ng/g dw, respectively) (Table 4.36). Petrogenic PAHs prevail in bottom sediments from these areas.

4.6.3.3. Marine biota

Petroleum hydrocarbons including PAH compounds are distributed in all compartments of the environment and are readily taken up by biota. Bioaccumulation of petroleum hydrocarbons and PAHs has been clearly demonstrated for aquatic plants, plankton, molluscs, crustaceans, echinoderms, annelids and fish (Thomas et al., 1983; GESAMP, 1993; Patin, 1999). The extent of bioaccumulation varies from organism to organism and depends on many factors including the length of exposure, water solubility and the lipophilicity of each individual compound, the mix of compounds present, and environmental conditions such as temperature, salinity and the presence of particulates and dissolved organic matter. Ultimately, any biological effects associated with bioaccumulation will depend on the rates of uptake, depuration, metabolism and excretion. In mammals, PAHs are readily absorbed from the gastrointestinal tract due to their lipophilicity. They are then rapidly distributed among various tissues, but tend to be concentrated in body fat. Metabolism of PAHs occurs via the cytochrome P450-mediated mixed-function oxidase system with oxidation or hydroxylation as the first step. The resultant epoxides or phenols might be detoxified in a reaction to produce glucoronides, sulphates or glutathione conjugates (Samanta et al., 2002). Some epoxides can metabolise into dihydrodiols, which in turn undergo conjugation to form soluble detoxification products. Many PAHs, however, can be metabolically transformed to other, highly reactive epoxides some of which are carcinogenic or more toxic than the parent compounds.

4.6.3.3.1. Alaska, USA

A baseline study of contaminants in biota in the vicinity of the Alaskan Northstar and Liberty offshore oil and gas activity areas was conducted in 2001 (MMS, 2003). Samples of five species of finfish – four-horned sculpin (*Myoxycephalus quadricornis*), Arctic cod (*Boreogadus saida*), broad whitefish

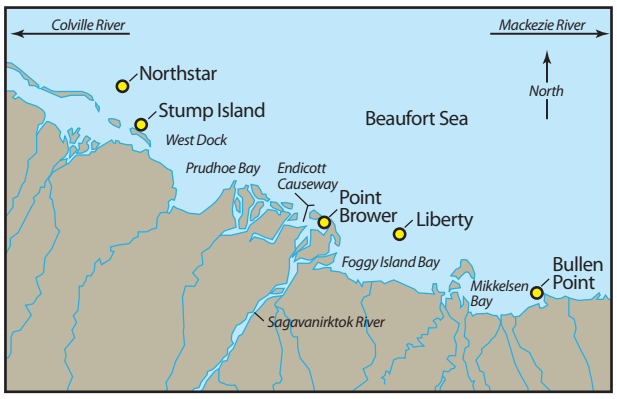


Figure 4.25. Locations where fish were sampled during the 2001 baseline study in Alaska (MMS, 2003).

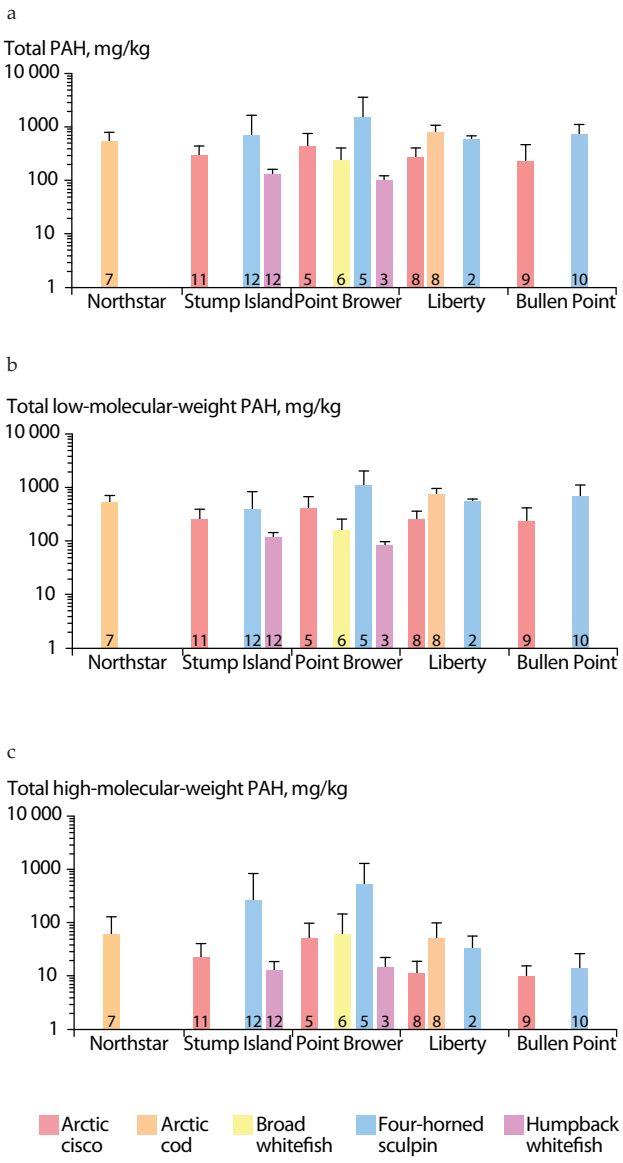


Figure 4.26. Lipid-normalized wet weight concentrations of (a) total PAH, (b) total low-molecular-weight PAH, and (c) total high-molecular-weight PAH for the various fish species and locations sampled in the 2001 baseline study in Alaska (MMS, 2003). Error bars depict one standard deviation and the numbers of individual fish making up the samples are indicated at the bottom of each bar.

(*Coregonus nasus*), humpback whitefish (*C. pidschian*), and Arctic cisco – were collected from five locations in the Beaufort Sea between the Colville and Mackenzie Rivers (Figure 4.25). Tissues from these fish were analyzed for a suite of trace metals and organic contaminants (petroleum-related as well as various organochlorine compounds).

Mean whole-body concentrations for total PAHs ranged from <100 to >1000 ng/g (lipid-normalized wet weight). There was no significant effect of sampling location on the concentrations of total PAHs in any of the fish species. For sampling locations where more than one species of fish was analyzed, the four-horned sculpin had the highest concentrations of total PAHs (Figure 4.26a).

For low-molecular-weight PAHs (2- and 3-ring compounds), there were no significant differences among sampling locations for any of the species of fish (Figure 4.26b). For the high-molecular-weight PAHs (4- and 5-ring compounds), the concentrations in two species were related to sampling location: Arctic cisco (p= 0.0168) and four-horned sculpin (p=0.0009) (Figure 4.26c). For Arctic cisco, Point Brower had higher values than the other sampling locations. For the four-horned sculpin, both Point Brower and Stump Island had higher concentrations than Bullen Point.

Bile hydrocarbons and hydrocarbon metabolites were determined for two PAH compounds – phenanthrene and benzo[*a*]pyrene – which represent compounds with three and five aromatic rings, respectively. These PAHs also respectively represent the low-molecular-weight and high-molecular-weight PAH compound classes. In general, phenanthrene is more representative of fresh petroleum and benzo[*a*]pyrene is representative of weathered petroleum and, especially, pyrogenic sources (e.g., forest fires, internal combustion engines). A statistical analysis applied to determine whether there was a significant relationship between the concentrations of bile hydrocarbons/hydrocarbon metabolites and sampling location showed that the variation of phenanthrene equivalents and sampling location was highly significant for four-horned sculpin; Bullen Point, Stump Island and Liberty were very similar and all higher than Point Brower. One individual Arctic cisco at Bullen Point had a value of 10 000 ng/g phenanthrene equivalents. The variation of benzo[*a*]pyrene equivalents with sampling location was highly significant for Arctic cisco; Bullen Point was higher than Stump Island and Point Brower. In four-horned sculpin, benzo[*a*]pyrene equivalents were significantly higher at Stump Island than at Point Brower. Many of the fish assayed had values in excess of 5000 ng/g benzo[*a*]pyrene equivalents.

4.6.3.3.2. Canada

In 2002, concentrations of alkanes and PAHs were measured in seven fish species (inconnu, *Stenodus leucichthys*; saffron cod, *Eleginus navaga*; Pacific herring, *Clupea harengus pallasi*; burbot, *Lota lota*; Arctic flounder, *Liopsetta glacialis*; lake whitefish, *Coregonus clupeaformis*; least cisco, *Coregonus sardinella*) and in invertebrates (isopods and amphipods) at seven potential offshore drilling locations in the southern Beaufort Sea (Kavik-AXYS, 2004). For whole fish analyses, alkanes, parent PAH and alkylated PAH occurred in the ranges 0.2–32, 0.4–114, and <0.1–700 ng/g ww, respectively. Among the different species, the highest concentrations were found in least cisco. The concentrations of alkanes in the invertebrates (1.9–5 ng/g ww) were at the lower end of the range seen for fish. Excluding least cisco, the PAH concentrations (6–20 ng/g ww for parent PAH, 4–23 ng/g ww for alkylated PAH) in invertebrates were higher than in fish. This may be related to the superior ability of vertebrates over invertebrates to metabolize PAH compounds.

Stern (2003) reported concentrations of parent and alkyl PAH concentrations in the muscle tissue of five species (lake trout, *Salvelinus namaycush*; lake whitefish, Pacific herring, northern pike, *Esox lucius*; inconnu) caught in the Husky Lakes area in 2002. Parent PAH was in the range 1–10 ng/g ww; the range for alkylated PAH was 12–66 ng/g ww.

4.6.3.3.3. Greenland

PAH concentrations were measured recently in macroalgae (bladder wrack; *Fucus vesiculosus*), blue mussels (*Mytilus edulis*), lugworm (*Arenicola marina*), and shorthorn sculpin (*Myoxycephalus scorpius*) livers from three sampling locations (with different populations) in southern Greenland, with samples collected in 1999 and 2000 (Riget et al., 2003). Concentrations of total PAHs varied from 41 ng/g ww in a bladder wrack sample to 370 ng/g ww in a lugworm sample. The data show that the invertebrates, especially the lugworm, have a significantly higher level of PAHs than the macroalgae. This is probably due to biomagnification and to the macroalgae living only in the water phase, whereas the invertebrates live also in the sediment, where concentrations of PAHs are usually higher. Concentrations of PAHs in blue mussels, collected from 14 different sites in southern Greenland, averaged about 90 ng/g ww (range 32–210 ng/g ww) and varied with age and size. These concentrations are similar to those observed for blue mussels in the Baltic Sea. No obvious differences in PAH concentrations were observed between different size groups, but year-to-year variations seem to be more dominant. Concentrations were higher for the samples collected at Qaqortoq, and this may indicate that pollution, probably related to boat traffic, is more pronounced there. Nineteen liver samples from shorthorn sculpins were analysed. Although fish of different age and sex were collected, no obvious differences were observed between young–old and male–female groups. Average values of 425 ng/g ww were observed (range 270–600 ng/g ww). The slightly higher concentrations of PAHs found in blue mussels from areas of higher human population were not found in sculpins. Liver from shorthorn sculpins collected near the dump sites in 2002 during the dumpsite survey in Greenland (Asmund, 2007) had significantly lower concentrations of PAHs (about 257 ng/g ww) than sculpins collected at Kronprinsens Island (about 897 ng/g ww).

On the Nuussuaq Peninsula at Marraat, the presence of oil-stained stones scattered along the coast may indicate oil-bearing strata and the existence of oil seeps. Blue mussels and fish from this area have been analysed for hydrocarbon content (Mosbech et al., 2007). Total hydrocarbon concentrations ranged from 130 µg/g ww in blue mussels to 540 µg/g ww in sculpin liver. The pattern of 2- and 3-ring aromatics (alkylated homologues of naphthalene, phenanthrene and dibenzothiophene) and petroleum biomarkers (hopanes) found by gas chromatography-mass spectroscopy showed the regular pattern of light fuel oil. It was concluded that observed hydrocarbons in the biota samples probably indicated inputs from both immature petrogenic hydrocarbons of possible lacustrine origin and local pollution by fuel oil. Average PAH concentrations of

0.12 μg/g ww (range 0.10–0.14 μg/g ww) were measured in blue mussels, and 0.39 μg/g ww (range 0.29–0.49 μg/g ww) in sculpin liver. These levels are similar to those reported by Riget et al. (2003) for samples collected in the Qaqortoq area.

4.6.3.3.4. Faroe Islands

PAH concentrations in molluscs were recently measured in the coastal zone of the Faroe Islands (Høydal, 2004). All invertebrate samples were analysed for 23 different PAHs. For all species a seasonal pattern was observed: the highest PAH content found in winter (December to February), lower concentrations in spring (March to May) and lowest concentrations in summer (June to August). The seasonal differences could not be explained by differences in the lipid or water content of the invertebrates. The highest accumulation of PAHs was found in blue mussels; horse mussels (*Modiolus modiolus*) and snails had lower concentrations. The accumulated fraction of high molecular weight PAHs in blue mussels was also higher than the corresponding fraction in limpets and periwinkles (Høydal, 2004).

4.6.3.3.5. Norway

Concentrations of PAH in the soft-body tissue of blue mussels were measured at various locations along the Norwegian coast during 1995–2002 (Green et al., 2004). In the Arctic regions (Mo i Rana) the concentrations exceeded 1 μg/g ww in 1995 and 2000, were lower in the other years, and as low as 0.17 μg/g ww in 2002.

The measurement of PAHs in fish muscle is included in the routine environmental monitoring program for offshore petroleum installations in Norway. The objective is to document the extent to which discharges from the oil and gas installations cause a deterioration in fish quality. In these studies, PAHs were measured in the muscle of cod and haddock caught in the North Sea, Norwegian Sea, and Barents Sea during the period 1995–2005 (Klungsøyr and Johnsen, 1997; Klungsøyr et al., 2001; Grøsvik et al., 2007). PAH concentrations from all sea areas around Norway were at very low (background) concentrations, and there was no indication that the fish caught in the open sea are contaminated with PAHs. This may be due to a combination of low exposure to hydrocarbons in most areas (except perhaps at locations very close to oil platforms) and the effective metabolism and excretion of PAHs by fish.

4.6.3.3.6. Russia

Concentrations of total hydrocarbons and PAHs in marine mammals, fish and molluscs collected throughout the seas of the Russian Arctic during 1997 to 2003 are presented in the Appendix to this chapter.

Total PAH concentrations (16 compounds) of the soft tissue of blue mussels from the coastal Barents Sea collected in 2002–2004 were in the range 14–237 ng/g ww (Savinova et al., 2006b). In blue mussels collected in 2006 from the White Sea, total PAH concentrations in the soft tissues were significantly lower at 1–13 ng/g ww (Savinova et al., 2007).

PAHs (sum of nine compounds) and concentrations of nC_{11} to nC_{31} n-alkanes were measured in liver and muscle of Atlantic cod (*Gadus morhua morhua*), haddock (*Melanogrammus aeglefinus*), sand dab (*Hippoglossoides platessoides limandoides*) and thorny skate (*Raja radiata*) from the Barents Sea collected in 2001; haddock liver had both the highest PAH concentrations (84 ng/g ww) and highest n-alkane concentrations (46 μg/g ww) (Kireeva et al., 2002). Similar measurements made in 1998 on a wider range of marine biota sampled in the same area also indicated that haddock liver contained the highest concentrations of PAHs (88 ng/g ww) and n-alkanes (95 μg/g ww) (Matishov et al., 1998). Average PAH concentrations in liver of Atlantic cod from the Barents, Kara and Pechora seas were all similar ranging from 41 to 49 ng/g ww (RCMA, 2005); the corresponding concentration in polar cod (*Boreogadus saida*) from the Pechora Sea was 42 ng/g ww. Savinova et al. (2007), reported total PAH concentrations (16 compounds) in the liver of White Sea cod (*Gadus morhua marisalbi*) and navaga (*Eleginus navaga*) both collected in the White Sea of 9 to 15 ng/g ww.

Concentrations of PAHs (eight compounds) in samples of harp seal (*Phoca groenlandica*) muscle, liver and blubber collected in the White Sea in 1997 were within the range 5 to 111 ng/g dw with the highest concentrations in adult female blubber (Plotitsyna, 1998). The same study found concentrations of n-alkanes (nC_{12} to nC_{29}) of 3–14 μg/g ww; the highest value occurred in whitecoat liver. Total PAH concentrations (16 compounds) in the blubber of ringed seals (*Phoca hispida*) from the Yugorsky Peninsula and Chukchi Peninsula were very similar (163 and 175 ng/g ww, respectively). The average concentrations (ng/g ww) of PAHs (16 compounds) in walruses (*Odobenus rosmarus*) from the Chukchi Peninsula area decreased in the order: blubber, 137 > liver, 31 > muscle, 12 > kidney, 9. Naphthalene and phenanthrene were the most abundant PAH compounds in all marine mammal samples analysed (RCMA, 2005).

4.7. Conclusions

The completeness of the assessment has been limited by several factors. Most importantly, the published scientific literature on sources and inputs of petroleum hydrocarbons and PAHs to the Arctic environment from petroleum industry activities is inadequate for a comprehensive assessment. As a result, it was necessary to acquire as much additional up-to-date information as possible from national authorities and the petroleum industry in the Arctic countries on oil spills, discharges of the various types of wastes to the terrestrial and marine environments, emissions to the atmosphere, and environmental baseline information. In general, acquiring the required information proved a slow and incomplete process. Also, information on discharge permits, emissions to the atmosphere linked to metres drilled, drill waste formulations for each type of well, total production and disposal volumes, and treatment of produced water was not provided by all Arctic countries. Finally, compiling information from multiple sources using very different reporting formats makes a comparison of data difficult and prone to 'double counting'.

The previous AMAP assessment (AMAP, 1998) stressed that the intercomparability of concentrations of petroleum hydrocarbons and PAHs was severely limited by a lack of the necessary information required to judge data quality and representativeness. This led to a strong recommendation that action be taken by the circumpolar countries to harmonize methods and introduce appropriate data quality and reporting protocols so as to increase the intercomparability of data sets. Although there has been some improvement in this respect, much progress is still required to achieve the level of data intercomparability desired.

Oil and gas exploration and extraction have taken place in several Arctic countries for many years and new projects for development of oil and gas resources are presently underway,

particularly in Russia. Most of the activity takes place in the Arctic itself; some activity also occurs in adjacent areas in such a way that discharges and emissions, including those associated with transportation of oil and gas, may influence Arctic seas and land. During the early years of exploration and production activities in the Arctic, environmental impacts were typically more severe than those associated with corresponding activities today. This change is due to a combination of factors, including improved technology, stricter regulations, improved environmental management plans, and a better understanding of the vulnerability and susceptibility of Arctic ecosystems to various sources of disturbance.

Monitoring programs for petroleum hydrocarbons and PAHs on an operational basis in the oil and gas industry and in the receiving environment are variable both in extent and in terms of how they are carried out among the circumpolar countries. Lack of detailed information on methods of sample collection, sample analysis, and reporting and quality control procedures make it difficult and sometimes impossible to compare monitoring datasets. Nonetheless, some general statements can be made.

1. Petroleum hydrocarbons and PAHs have natural and anthropogenic sources in the Arctic. Natural sources include petroleum seeps and forest fires. Anthropogenic sources include oil and gas industry activities, chronic and episodic releases from shipping and transportation, combustion of fossil fuels for energy, heat and transportation (primarily in areas of human settlement), and long-range transport from temperate and sub-Arctic locations.

2. The concentrations of petroleum hydrocarbons and PAHs in all environmental compartments in the Arctic are generally low compared with other areas of the world. Elevated concentrations exist on local scales. Examples are areas of oil seeps, the Mackenzie River delta, areas around Svalbard, and areas of industrial point source contamination. In water, the highest concentrations occur in areas such as the coastal Kara Sea, southern Barents Sea and southern Laptev Sea where the legislated MPC (Maximum Permissible Concentration) value of 50 μg/L is often exceeded. It can be expected that the quality of nearshore marine waters of Arctic Russia will further decrease in the future as further industrialization of the watersheds of north-flowing Russian rivers and as the huge oil and gas developments on the Russian shelves proceed. In biota, concentrations of PAHs are generally low because most organisms such as fish, birds and mammals can metabolise and excrete PAHs. Although in most cases the concentrations of PAHs are below the thresholds required to cause observable effects in biota, there remains concern around areas of chronic hydrocarbon pollution because PAHs can persist for extended periods under Arctic conditions, and because some PAHs and some metabolic by-products of PAHs are known carcinogens. There are large regional variations in the concentrations of PAHs in sediments. In most areas background concentrations of total PAHs are less than 500 ng/g dw. For example, total PAH concentrations in central and southern Barents Sea sediments are in the range 200–400 ng/g dw and analysis of sediment cores indicates that concentrations have not increased during the past 15 years. Areas of naturally-enriched PAH concentrations also occur. Examples include Svalbard, where weathering and eroding of coal-rich geological structures release hydrocarbons into coastal waters (resulting in total PAH levels of 3000–6000 ng/g dw), and the Mackenzie River delta (total PAH 4000–15 000 ng/g dw), a receiving environment for hydrocarbons from the vast petroleum-rich Mackenzie River basin which contains a number of natural oil seeps. Generally, the PAH profile for offshore sediments suggests a predominantly pyrogenic source whereas that for nearshore sediments, particularly those from harbours, suggests a significant petrogenic source.

A first attempt to produce a petroleum hydrocarbon budget for the Arctic, conducted as part of this assessment, provides some insight into the relative importance of different sources of petroleum hydrocarbons and PAHs, and relates this to the concentrations observed in the different compartments of the Arctic environment. The current version of the budget is considered more qualitative than quantitative due to the lack of data for many sources on a circumpolar basis. Nonetheless, and based on data currently available, the budget shows that natural sources and inputs are very important for the concentrations of petroleum hydrocarbons found in different compartments of the Arctic environment. For example, on the basis of the budget constructed, exploration and production activities of the petroleum industry currently account for less than 1% of the total average annual input of petroleum hydrocarbons and PAHs to the circumpolar Arctic. Oil spills are the largest anthropogenic contributor of petroleum hydrocarbons in the Arctic environment accounting for 4–8% of average annual inputs. Other important anthropogenic inputs include industrial activities and the use of petroleum products. It is estimated, based on many simplifying assumptions, that oil and gas activities including spills could account for as much as 58% (up from the current 6%) of the annual natural input of petroleum hydrocarbons and PAHs by 2040, the projected year of peak oil production in the Arctic.

The biggest concern related to oil and gas activities in the Arctic remains a catastrophic oil spill in the marine environment. The Arctic is characterized by low temperatures, ice, and permafrost, and experiences large seasonal variations with extensive periods of darkness and cold. These environmental factors affect the rates of weathering and biological degradation processes that take place following a spill. It is the general view that these processes, which act to remove oil from the environment following a spill, are slowed down in the Arctic. The consequence is that oil and contaminants may persist for longer periods causing longer chronic exposures to biota and extending the recovery time for both Arctic biota and the Arctic environment. Furthermore, oil spill response in the Arctic Ocean will almost always be a challenge due to its remoteness, the severe environmental conditions, a limited logistical infrastructure, and inadequate technology for effective oil spill clean-up in Arctic conditions, particularly for oil spilled in sea ice or under sea ice. Oil spilled on land is also a concern because the thousands of kilometres of aging pipelines currently in place suffer from maintenance deficiencies; climate change effects such as permafrost degradation are likely to further negatively impact this situation. Land spills, however, are of lower concern than marine spills because spreading of the spilled oil can be more readily contained and clean-up is much easier.

At present, hydrocarbon contamination in the Arctic is not a widespread problem, apart from areas of intense human activity. The most highly contaminated areas in the Arctic are close to human settlements, near industrial or military areas, and in areas where oil spills have occurred. Harbours and certain rivers and estuaries also show elevated concentrations of petroleum hydrocarbons and PAHs. The contaminant status in some areas where accidental and operational oil spills have occurred is not well known, and further studies are needed (e.g., in the area of the Komi Republic / Pechora River basin affected by the Usinsk pipeline rupture).

There is also a need for improved dissemination of detailed information for studies of chemical contamination conducted around oil and gas installations. This information should be presented in a format that will readily allow evaluation of the quality and intercomparability of the data.

Appendix 4.1.
Concentrations of petroleum hydrocarbons in terrestrial biota

Location	Year	Species	Tissue	n	Mean concentration	Units	Remarks	Source
Russia								
Kola Peninsula	2001	Lichens	whole	–	411	ng/g dw	Σ 14 PAH[a]	AMAP, 2004
Pechora Basin				–	128			
West Taymir				–	295			
East Taymir				–	838			
Chukotka Inland				–	194			
Chukotka Coast				–	802			
Kola Peninsula	2001	Bryophytes (mosses)	whole	–	522	ng/g dw	Σ 14 PAH[a]	AMAP, 2004
Pechora Basin				–	640			
West Taymir				–	965			
East Taymir				–	766			
Chukotka inland				–	286			
Chukotka Coast				–	404			
Kola Peninsula	2001	*Rangifer tarandus* (reindeer)	muscle	–	48	ng/g ww	Σ 8 PAH[b]	AMAP, 2004
Pechora basin				–	38			
West Taymir				–	55			
East Taymir				–	37			
Chukotka				–	61			
Kola Peninsula	2001	browser birds	muscle	–	66	ng/g ww	Σ 6 PAH[c]	AMAP, 2004
Pechora basin				–	76			
West Taymir				–	66			
East Taymir				–	58			
Chukotka inland				–	86			
Chukotka coast				–	58			
West Taymir	2001	grazer birds	muscle	–	32	ng/g ww	Σ 6 PAH[c]	AMAP, 2004
East Taymir				–	43			
Chukotka				–	78			
Pechora basin	2001	omnivore birds	muscle	–	53	ng/g ww	Σ 6 PAH[c]	AMAP, 2004
West Taymir				–	44			
East Taymir				–	75			
Chukotka inland				–	43			
Kola Peninsula	2001	molluscivore birds	muscle	–	44	ng/g ww	Σ 6 PAH[c]	AMAP, 2004
East Taymir				–	61			
Chukotka inland				–	30			
Chukotka coast				–	27			
Kola Peninsula	2001	piscivore birds	muscle	–	53	ng/g ww	Σ 6 PAH[c]	AMAP, 2004
East Taymir				–	55			
Chukotka inland				–	81			
Greenland								
Usuk	1999	Lichens	whole	–	31	µg/kg ww	Σ 26 PAH[d]	Riget et al., 2003
Near Igaliko				–	33			
Igaliko				–	39			
Usuk	2000	Lichens	whole	–	79	µg/kg ww	Σ 26 PAH[d]	Riget et al., 2003
Near Igaliko				–	93			

[a] Naphthalene, acenaphtylene, biphenyl, 2-methylnaphthalene, fluorene, acenaphthene, phenanthrene, anthracene, fluoranthene, pyrene, benz[*a*]anthracene, chrysene, benzo[*b*]fluoranthene, benzo[*k*]fluoranthene; [b] Naphthalene, 2-methylnaphthalene, fluorene, phenanthrene, anthracene, fluoranthene, pyrene, chrysene; [c] Naphthalene, 2-methylnaphthalene, fluorene, phenanthrene, fluoranthene, pyrene; [d] 14 lighter PAH (2–3 rings) and 12 higher PAH (4–7 rings).

Appendix 4.2.
Concentrations of petroleum hydrocarbons in freshwater biota.

Location	Year	Species	Tissue	n	Mean concentration	Units	Remarks	Source
Russia								
Kola Peninsula	2001	*Lota lota* (burbot)	muscle	–	154	ng/g ww	Σ 5 PAH[a]	AMAP, 2004
West Taymir				–	133			
East Taymir				–	144			
Kola Peninsula	2001	*Esox lucius* (pike)	muscle	–	21	ng/g ww	Σ 5 PAH[a]	AMAP, 2004
Chukotka inland				–	94			
Pechora basin	2001	*Perca fluviatus* (perch)	muscle	–	74	ng/g ww	Σ 5 PAH[a]	AMAP, 2004
Pechora basin	2001	*Leuciscus idus* (ide)	muscle	–	49	ng/g ww	Σ 5 PAH[a]	AMAP, 2004
Greenland								
Lake near Igaliko	2000	*Salvelinus alpinus (L.)* (Arctic char) males	liver	–	290	µg/kg ww	Σ 26 PAH[b]	Riget et al., 2003
Itilleq				–	250			
Lake near Igaliko	2000	*Salvelinus alpinus (L.)* (Arctic char) females	liver	–	120	µg/kg ww	Σ 12 PAH[c]	Riget et al., 2003
Itilleq				–	190			

[a] Naphthalene, 2-methylnaphthalene, fluorene, phenanthrene, fluoranthene; [b] 14 lighter PAH (2–3 rings) and 12 higher PAH (4–7 rings), [c] 12 higher PAH (4-7 rings).

Appendix 4.3.
Concentrations of petroleum hydrocarbons in marine biota.

Location	Year	Species	Tissue	n	Mean concentration	Units	Remarks	Source
Russia								
Kola Peninsula	2001	*Coregonus* spp. (whitefish)	muscle	–	69	ng/g ww	Σ 5 PAH[a]	AMAP, 2004
Pechora basin	2001	*Coregonus* spp. (whitefish)	muscle	–	79	ng/g ww	Σ 5 PAH[a]	AMAP, 2004
West Taymir	2001	*Coregonus* spp. (whitefish)	muscle	–	112	ng/g ww	Σ 5 PAH[a]	AMAP, 2004
East Taymir	2001	*Coregonus* spp. (whitefish)	muscle	–	106	ng/g ww	Σ 5 PAH[a]	AMAP, 2004
West Taymir	2001	*Coregonus autumnalis* (Arctic cisco)	muscle	–	74	ng/g ww	Σ 5 PAH[a]	AMAP, 2004
East Taymir	2001	*Coregonus nasus* (broad whitefish)	muscle	–	51	ng/g ww	Σ 5 PAH[a]	AMAP, 2004
Chukotka inland	2001	*Coregonus nasus* (broad whitefish)	muscle	–	54	ng/g ww	Σ 5 PAH[a]	AMAP, 2004
Chukotka coast	2001	*Thymallus arcticus* (Arctic grayling)	muscle	–	78	ng/g ww	Σ 5 PAH[a]	AMAP, 2004
Barents Sea	2001	*Gadus morhua morhua* (Atlantic cod)	muscle	25	2	ng/g ww	Σ 9 PAH	Kireeva et al, 2002
Barents Sea	2001	*Gadus morhua morhua* (Atlantic cod)	liver	25	47	ng/g ww	Σ 9 PAH	Kireeva et al, 2002
Barents Sea	2001	*Melanogramus aeglefinus* (haddock)	muscle	25	5	ng/g ww	Σ 9 PAH	Kireeva et al, 2002
Barents Sea	2001	*Melanogramus aeglefinus* (haddock)	liver	25	84	ng/g ww	Σ 9 PAH	Kireeva et al, 2002
Barents Sea	2001	*Hippoglassoides platessoides limandoides* (sand dab)	muscle	25	4	ng/g ww	Σ 9 PAH	Kireeva et al, 2002
Barents Sea	2001	*Hippoglassoides platessoides limandoides* (sand dab)	liver	25	69	ng/g ww	Σ 9 PAH	Kireeva et al, 2002
Barents Sea	2001	*Raja radiate* (thorny skate)	muscle	10	5	ng/g ww	Σ 9 PAH	Kireeva et al, 2002
Barents Sea	2001	*Raja radiate* (thorny skate)	liver	10	31	ng/g ww	Σ 9 PAH	Kireeva et al, 2002
Barents Sea	2001	*Gadus morhua morhua* (Atlantic cod)	muscle	25	2	μg/g ww	Σ nC_{11}–nC_{31}	Kireeva et al, 2002
Barents Sea	2001	*Gadus morhua morhua* (Atlantic cod)	liver	25	8	μg/g ww	Σ nC_{11}–nC_{31}	Kireeva et al, 2002
Barents Sea	2001	*Melanogramus aeglefinus* (haddock)	muscle	25	1	μg/g ww	Σ nC_{11}–nC_{31}	Kireeva et al, 2002
Barents Sea	2001	*Melanogramus aeglefinus* (haddock)	liver	25	46	μg/g ww	Σ nC_{11}–nC_{31}	Kireeva et al, 2002
Barents Sea	2001	*Hippoglassoides platessoides limandoides* (sand dab)	muscle	25	4	μg/g ww	Σ nC_{11}–nC_{31}	Kireeva et al, 2002
Barents Sea	2001	*Hippoglassoides platessoides limandoides* (sand dab)	liver	25	9	μg/g ww	Σ nC_{11}–nC_{31}	Kireeva et al, 2002
Barents Sea	2001	*Raja radiate* (thorny skate)	muscle	10	2	μg/g ww	Σ nC_{11}–nC_{31}	Kireeva et al, 2002
Barents Sea	2001	*Raja radiate* (thorny skate)	liver	10	14	μg/g ww	Σ nC_{11}–nC_{31}	Kireeva et al, 2002
Barents Sea	1998	*Gadus morhua morhua* (Atlantic cod)	muscle	75	6	ng/g ww	Σ 9 PAH	Matishov et al, 1998
Barents Sea	1998	*Gadus morhua morhua* (Atlantic cod)	liver	75	72	ng/g ww	Σ 9 PAH	Matishov et al, 1998
Barents Sea	1998	*Melanogramus aeglefinus* (haddock)	muscle	75	5	ng/g ww	Σ 9 PAH	Matishov et al, 1998
Barents Sea	1998	*Melanogramus aeglefinus* (haddock)	liver	75	88	ng/g ww	Σ 9 PAH	Matishov et al, 1998
Barents Sea	1998	*Hippoglassoides platessoides limandoides* (sand dab)	muscle	40	5	ng/g ww	Σ 9 PAH	Matishov et al, 1998
Barents Sea	1998	*Hippoglassoides platessoides limandoides* (sand dab)	liver	40	74	ng/g ww	Σ 9 PAH	Matishov et al, 1998
Barents Sea	1998	*Clupea harengus harengus* (Atlantic herring)	muscle	75	11	ng/g ww	Σ 9 PAH	Matishov et al, 1998

Appendix 4.3. Cont.

Location	Year	Species	Tissue	n	Mean concentration	Units	Remarks	Source
Barents Sea	1998	*Sebastes mentella* (deepwater redfish)	muscle	40	7	ng/g ww	Σ 9 PAH	Matishov et al, 1998
Barents Sea	1998	*Sebastes mentella* (deepwater redfish)	liver	40	45	ng/g ww	Σ 9 PAH	Matishov et al, 1998
Barents Sea	1998	*Anarhichas minor* (spotted catfish)	muscle	11	55	ng/g ww	Σ 9 PAH	Matishov et al, 1998
Barents Sea	1998	*Anarhichas minor* (spotted catfish)	liver	11	46	ng/g ww	Σ 9 PAH	Matishov et al, 1998
Barents Sea	1998	*Gadus morhua morhua* (Atlantic cod)	muscle	75	2	µg/g ww	Σ nC_{12}–nC_{29}	Matishov et al, 1998
Barents Sea	1998	*Gadus morhua morhua* (Atlantic cod)	liver	75	41	µg/g ww	Σ nC_{12}–nC_{29}	Matishov et al, 1998
Barents Sea	1998	*Melanogramus aeglefinus* (haddock)	muscle	75	4	µg/g ww	Σ nC_{12}–nC_{29}	Matishov et al, 1998
Barents Sea	1998	*Melanogramus aeglefinus* (haddock)	liver	75	9	µg/g ww	Σ nC_{12}–nC_{29}	Matishov et al, 1998
Barents Sea	1998	*Clupea harengus harengus* (Atlantic herring)	muscle	75	5	µg/g ww	Σ nC_{12}–nC_{29}	Matishov et al, 1998
Barents Sea	1998	*Sebastes mentella* (deepwater redfish)	muscle	40	8	µg/g ww	Σ nC_{12}–nC_{29}	Matishov et al, 1998
Barents Sea	1998	*Sebastes mentella* (deepwater redfish)	liver	40	9	µg/g ww	Σ nC_{12}–nC_{29}	Matishov et al, 1998
Barents Sea	1998	*Anarhichas minor* (spotted catfish)	muscle	11	6	µg/g ww	Σ nC_{12}–nC_{29}	Matishov et al, 1998
Barents Sea	1998	*Anarhichas minor* (spotted catfish)	liver	11	9	µg/g ww	Σ nC_{12}–nC_{29}	Matishov et al, 1998
Barents Sea, Kola Bay	2003	*Gadus morhua morhua* (Atlantic cod)	liver	–	36	ng/g ww	Σ 18 PAH	Alexeeva et al., 2005
SE Barents Sea	1998	*Chlamys islandica* (scallop)	muscle	150	5	ng/g ww	Σ 9 PAH	Matishov et al, 1998
Barents Sea, Yarnyshanaya Bay	2003	*Mytilus edulis* (blue mussel)	soft tissue	–	15	ng/g ww	Σ 18 PAH	Alexeeva et al., 2005
SW Barents Sea	1978	*Paralithodes camschatica* (red king crab)	muscle	9	1	ng/g ww	Σ 9 PAH	Matishov et al, 1998
White Sea	1997	*Phoca groenlandica* (harp seal) adult female	muscle	8	5	ng/g ww	Σ 8 PAH	Plotitsyna, 1998
White Sea	1997	*Phoca groenlandica* (harp seal) whitecoat	muscle	8	6	ng/g ww	Σ 8 PAH	Plotitsyna, 1998
White Sea	1997	*Phoca groenlandica* (harp seal) adult female	liver	8	40	ng/g ww	Σ 8 PAH	Plotitsyna, 1998
White Sea	1997	*Phoca groenlandica* (harp seal) whitecoat	liver	8	38	ng/g ww	Σ 8 PAH	Plotitsyna, 1998
White Sea	1997	*Phoca groenlandica* (harp seal) adult female	lubber	8	112	ng/g ww	Σ 8 PAH	Plotitsyna, 1998
White Sea	1997	*Phoca groenlandica* (harp seal) whitecoat	lubber	8	101	ng/g ww	Σ 8 PAH	Plotitsyna, 1998
White Sea	1997	*Phoca groenlandica* (harp seal) adult female	muscle	8	6	µg/g ww	nC_{12}–nC_{29}	Plotitsyna, 1998
White Sea	1997	*Phoca groenlandica* (harp seal) whitecoat	muscle	8	3	µg/g ww	nC_{12}–nC_{29}	Plotitsyna, 1998
White Sea		*Phoca groenlandica* (harp seal) adult female	liver	8	11	µg/g ww	nC_{12}–nC_{29}	Plotitsyna, 1998
White Sea	1997	*Phoca groenlandica* (harp seal) whitecoat	liver	8	14	µg/g ww	nC_{12}–nC_{29}	Plotitsyna, 1998
White Sea	1997	*Phoca groenlandica* (harp seal) adult female	lubber	8	5	µg/g ww	nC_{12}–nC_{29}	Plotitsyna, 1998
White Sea	1997	*Phoca groenlandica* (harp seal) whitecoat	lubber	8	4	µg/g ww	nC_{12}–nC_{29}	Plotitsyna, 1998
Norway								
Mo i Rana st. 1	1998	*Mytilus edulis* (blue mussel)	soft body	_	1850	ppb dw	Sum of up to 41 PAH	Green et al., 2004
Mo i Rana st. 2	1998	*Mytilus edulis* (blue mussel)	soft body	_	629	ppb dw	Sum of up to 41 PAH	Green et al., 2004
Mo i Rana st. 2	1999	*Mytilus edulis* (blue mussel)	soft body	_	917	ppb dw	Sum of up to 41 PAH	Green et al., 2004
Mo i Rana st. 2	2000	*Mytilus edulis* (blue mussel)	soft body	_	1160	ppb dw	Sum of up to 41 PAH	Green et al., 2004

Appendix 4.3. Cont.

Location	Year	Species	Tissue	n	Mean concentration	Units	Remarks	Source
Mo i Rana st. 2	2001	*Mytilus edulis* (blue mussel)	soft body	_	824	ppb dw	Sum of up to 41 PAH	Green et al., 2004
Mo i Rana st. 2	2002	*Mytilus edulis* (blue mussel)	soft body	_	170	ppb dw	Sum of up to 41 PAH	Green et al., 2004
Faroe Islands								
Hvannasund	2002	*Littorina obtusata* (invertebrate)		–	48	µg/kg ww	Σ 23 PAH	Høydal , 2004
Hvannasund	2002	*Littorina obtusata* (invertebrate)		–	18	µg/kg ww	Σ 23 PAH	Høydal , 2004
Hvannasund	2002	*Littorina obtusata* (invertebrate)		–	5	µg/kg ww	Σ 23 PAH	Høydal , 2004
Svínáir	2002	*Littorina obtusata* (invertebrate)		–	11	µg/kg ww	Σ 23 PAH	Høydal , 2004
Hvannasund	2002	*Nucella lapillus* (invertebrate)		–	37	µg/kg ww	Σ 23 PAH	Høydal , 2004
Hvannasund	2002	*Nucella lapillus* (invertebrate)		–	12	µg/kg ww	Σ 23 PAH	Høydal , 2004
Hvannasund	2002	*Nucella lapillus* (invertebrate)		–	10	µg/kg ww	Σ 23 PAH	Høydal , 2004
Svínáir	2002	*Nucella lapillus* (invertebrate)		–	3	µg/kg ww	Σ 23 PAH	Høydal , 2004
Hvannasund	2002	*Patella vulgata* (invertebrate)		–	19	µg/kg ww	Σ 23 PAH	Høydal , 2004
Hvannasund	2002	*Patella vulgata* (invertebrate)		–	6	µg/kg ww	Σ 23 PAH	Høydal , 2004
Hvannasund	2002	*Patella vulgata* (invertebrate)		–	5	µg/kg ww	Σ 23 PAH	Høydal , 2004
Svínáir	2002	*Patella vulgata* (invertebrate)		–	9	µg/kg ww	Σ 23 PAH	Høydal , 2004
Svínáir	2002	*Patella vulgata* (invertebrate)		–	3	µg/kg ww	Σ 23 PAH	Høydal , 2004
Svínáir	2002	*Patella vulgata* (invertebrate)		–	0	µg/kg ww	Σ 23 PAH	Høydal , 2004
Velbastaður	2002	*Patella vulgata* (invertebrate)		–	2	µg/kg ww	Σ 23 PAH	Høydal , 2004
Velbastaður	2002	*Patella vulgata* (invertebrate)		–	0	µg/kg ww	Σ 23 PAH	Høydal , 2004
Velbastaður	2002	*Patella vulgata* (invertebrate)		–	0	µg/kg ww	Σ 23 PAH	Høydal , 2004
Trongisvágur	2002	*Patella vulgata* (invertebrate)		–	26	µg/kg ww	Σ 23 PAH	Høydal , 2004
Trongisvágur	2002	*Patella vulgata* (invertebrate)		–	131	µg/kg ww	Σ 23 PAH	Høydal , 2004
Trongisvágur	2002	*Patella vulgata* (invertebrate)		–	49	µg/kg ww	Σ 23 PAH	Høydal , 2004
Hvannasund	2002	*Mytilus edulis* (blue mussel)		–	365	µg/kg ww	Σ 23 PAH	Høydal , 2004
Hvannasund	2002	*Mytilus edulis* (blue mussel)		–	175	µg/kg ww	Σ 23 PAH	Høydal , 2004
Svínáir	2002	*Mytilus edulis* (blue mussel)		–	117	µg/kg ww	Σ 23 PAH	Høydal , 2004
Svínáir	2002	*Mytilus edulis* (blue mussel)		–	40	µg/kg ww	Σ 23 PAH	Høydal , 2004
Svínáir	2002	*Mytilus edulis* (blue mussel)		–	7	µg/kg ww	Σ 23 PAH	Høydal , 2004
Kaldbak	2002	*Mytilus edulis* (blue mussel)		–	107	µg/kg ww	Σ 23 PAH	Høydal , 2004
Kaldbak	2002	*Mytilus edulis* (blue mussel)		–	117	µg/kg ww	Σ 23 PAH	Høydal , 2004
Kaldbak	2002	*Mytilus edulis* (blue mussel)		–	77	µg/kg ww	Σ 23 PAH	Høydal , 2004
Trongisvágur	2002	*Mytilus edulis* (blue mussel)		–	537	µg/kg ww	Σ 23 PAH	Høydal , 2004
Trongisvágur	2002	*Mytilus edulis* (blue mussel)		–	1375	µg/kg ww	Σ 23 PAH	Høydal , 2004
Trongisvágur	2002	*Mytilus edulis* (blue mussel)		–	215	µg/kg ww	Σ 23 PAH	Høydal , 2004

Appendix 4.3. Cont.

Location	Year	Species	Tissue	n	Mean concentration	Units	Remarks	Source
Kirkjubø	2002	*Modiolus modiolus* (invertebrate)		–	24	µg/kg ww	Σ 23 PAH	Høydal , 2004
Kirkjubø	May 2002	*Modiolus modiolus* (invertebrate)		–	3	µg/kg ww	Σ 23 PAH	Høydal , 2004
Kirkjubø	August 2002	*Modiolus modiolus* (invertebrate)		–	5	µg/kg ww	Σ 23 PAH	Høydal , 2004
Greenland								
Usuk	1999	Sand hopper (invertebrate)		_	98	µg/kg ww	Σ 26 PAH[b]	Riget et al., 2003
Igaliko	1999	*Arenicola marina* (lugworm)		_	370	µg/kg ww	Σ 26 PAH[b]	Riget et al., 2003
Igaliko	1999	Periwinkle (invertebrate)		_	45	µg/kg ww	Σ 26 PAH[b]	Riget et al., 2003
Usuk	2000	*Fucus vesiculosus* (bladder wrack) (macroalgae)		_	66	µg/kg ww	Σ 26 PAH[b]	Riget et al., 2003
Igaliko	2000	*Fucus vesiculosus* (bladder wrack) (macroalgae)		_	66	µg/kg ww	Σ 26 PAH[b]	Riget et al., 2003
Qaqortoq	2000	*Fucus vesiculosus* (bladder wrack) (macroalgae)		_	41	µg/kg ww	Σ 26 PAH[b]	Riget et al., 2003
Usuk	2000	Arctic wrack (macroalgae)		_	43	µg/kg ww	Σ 26 PAH[b]	Riget et al., 2003
Usuk	2000	Sweet tangle (macroalgae)		_	43	µg/kg ww	Σ 26 PAH[b]	Riget et al., 2003
Usuk	2000	Sea lettuce (macroalgae)		_	52	µg/kg ww	Σ 26 PAH[b]	Riget et al., 2003
Usuk	2000	Enteromorpha		_	24	µg/kg ww	Σ 26 PAH[b]	Riget et al., 2003
Usuk	1999	*Mytilus edulis* (blue mussel)		_	40	µg/kg ww	Σ 26 PAH[b]	Riget et al., 2003
Igaliko harbour	1999	*Mytilus edulis* (blue mussel)		_	80	µg/kg ww	Σ 26 PAH[b]	Riget et al., 2003
Igaliko outside harbour	1999	*Mytilus edulis* (blue mussel)		_	45	µg/kg ww	Σ 26 PAH[b]	Riget et al., 2003
Usuk	2000	*Mytilus edulis* (blue mussel)		_	115	µg/kg ww	Σ 26 PAH[b]	Riget et al., 2003
Igaliko	2000	*Mytilus edulis* (blue mussel)		_	130	µg/kg ww	Σ 26 PAH[b]	Riget et al., 2003
Qaqortoq	2000	*Mytilus edulis* (blue mussel)		_	210	µg/kg ww	Σ 26 PAH[b]	Riget et al., 2003
Usuk	1999	*Myoxycephalus scorpius* (shorthorn sculpin) females	liver	_	510	ng/g ww	Σ 26 PAH[b]	Riget et al., 2003
Igaliko	1999	*Myoxycephalus scorpius* (shorthorn sculpin) females	liver	_	340	ng/g ww	Σ 26 PAH[b]	Riget et al., 2003
Usuk	2000	*Myoxycephalus scorpius* (shorthorn sculpin) females	liver	_	450	ng/g ww	Σ 26 PAH[b]	Riget et al., 2003
Igaliko	2000	*Myoxycephalus scorpius* (shorthorn sculpin) females	liver	_	370	ng/g ww	Σ 26 PAH[b]	Riget et al., 2003
Qaqortoq	2000	*Myoxycephalus scorpius* (shorthorn sculpin) females	liver	_	440	ng/g ww	Σ 26 PAH[b]	Riget et al., 2003
Usuk	1999	*Myoxycephalus scorpius* (shorthorn sculpin) males	liver	_	380	ng/g ww	Σ 26 PAH[b]	Riget et al., 2003
Igaliko	1999	*Myoxycephalus scorpius* (shorthorn sculpin) males	liver	_	430	ng/g ww	Σ 26 PAH[b]	Riget et al., 2003
Usuk	2000	*Myoxycephalus scorpius* (shorthorn sculpin) males		_	380	ng/g ww	Σ 26 PAH[b]	Riget et al., 2003
Igaliko	2000	*Myoxycephalus scorpius* (shorthorn sculpin) males		_	520	ng/g ww	Σ 26 PAH[b]	Riget et al., 2003
Qaqortoq	2000	*Myoxycephalus scorpius* (shorthorn sculpin) males		_	370	ng/g ww	Σ 26 PAH[b]	Riget et al., 2003
Station TAB1 near Thule base	2001	Sculpin		_	337	ng/g ww	Σ 25 PAH	Glahder et al., 2003
Station NSB near Thule base	2001	Sculpin		_	364	ng/g ww	Σ 25 PAH	Glahder et al., 2003
Station TAB31 near Thule base	2001	Sculpin		_	463	ng/g ww	Σ 25 PAH	Glahder et al., 2003

Appendix 4.3. Cont.

Location	Year	Species	Tissue	n	Mean concentration	Units	Remarks	Source
Station TAB37 near Thule base	2001	Sculpin		_	543	ng/g ww	Σ 25 PAH	Glahder et al., 2003
Station BS near Thule base	2001	Sculpin		_	353	ng/g ww	Σ 25 PAH	Glahder et al., 2003
Qaanaaq	2001	Sculpin		_	47	ng/g ww	Σ 25 PAH	Glahder et al., 2003
Station TAB1 near Thule base	2001	*Pelecypoda* spp. (clams)		_	160	ng/g ww	Σ 25 PAH	Glahder et al., 2003
Station NSB near Thule base	2001	*Pelecypoda* spp. (clams)		_	108	ng/g ww	Σ 25 PAH	Glahder et al., 2003
Station TAB31 near Thule base	2001	*Pelecypoda* spp. (clams)		_	283	ng/g ww	Σ 25 PAH	Glahder et al., 2003
Station TAB37 near Thule base	2001	*Pelecypoda* spp. (clams)		_	210	ng/g ww	Σ 25 PAH	Glahder et al., 2003
Station TAB1 near Thule base	2001	bay scallop		_	108	ng/g ww	Σ 25 PAH	Glahder et al., 2003
Station NSB near Thule base	2001	bay scallop		_	143	ng/g ww	Σ 25 PAH	Glahder et al., 2003
Station TAB31 near Thule base	2001	bay scallop		_	157	ng/g ww	Σ 25 PAH	Glahder et al., 2003
Station TAB37 near Thule base	2001	bay scallop		_	166	ng/g ww	Σ 25 PAH	Glahder et al., 2003
Station BS near Thule base	2001	bay scallop		_	88	ng/g ww	Σ 25 PAH	Glahder et al., 2004
Qaanaaq	2001	bay scallop		–	131	ng/g ww	Σ 25 PAH	Glahder et al., 2003
Canada								
Coastal Beaufort Sea (69–70°N, 134–137°W)	2002	*Stenodus leucichthys* (inconnu)	whole	1	613	ng/g ww	Σ nC_{12}–nC_{36}, pristine, phytane	KAVIK-AXYS, 2004
Coastal Beaufort Sea (69–70°N, 134–137°W)	2002	*Stenodus leucichthys* (inconnu)	whole	1	613	ng/g ww	Σ nC_{12}–nC_{36}, pristine, phytane	KAVIK-AXYS, 2004
Coastal Beaufort Sea (69–70°N, 134–137°W)	2002	*Eleginus navaga* (saffron cod),	whole	4	1640	ng/g ww	Σ nC_{12}–nC_{36}, pristine, phytane	KAVIK-AXYS, 2004
Coastal Beaufort Sea (69–70°N, 134–137°W)	2002	*Clupea harengus pallasi* (Pacific herring)	whole	2	23350	ng/g ww	Σ nC_{12}–nC_{36}, pristine, phytane	KAVIK-AXYS, 2004
Coastal Beaufort Sea (69–70°N, 134–137°W)	2002	*Lota lota* (burbot),	whole	5	513	ng/g ww	Σ nC_{12}–nC_{36}, pristine, phytane	KAVIK-AXYS, 2004
Coastal Beaufort Sea (69–70°N, 134–137°W)	2002	*Liopsetta glacialis* (Arctic flounder)	whole	2	2055	ng/g ww	Σ nC_{12}–nC_{36}, pristine, phytane	KAVIK-AXYS, 2004
Coastal Beaufort Sea (69–70°N, 134–137°W)	2002	*Coregonus clupeaformis* (lake whitefish)	whole	3	840	ng/g ww	Σ nC_{12}–nC_{36}, pristine, phytane	KAVIK-AXYS, 2004
Coastal Beaufort Sea (69–70°N, 134–137°W)	2002	*Coregonus sardinella* (least cisco)	whole	2	14580	ng/g ww	Σ nC_{12}–nC_{36}, pristine, phytane	KAVIK-AXYS, 2004
Coastal Beaufort Sea (69–70°N, 134–137°W)	2002	Amphipods	whole	3	1690	ng/g ww	Σ nC_{12}–nC_{36}, pristine, phytane	KAVIK-AXYS, 2004
Coastal Beaufort Sea (69–70°N, 134–137°W)	2002	*Saduria entomona* (isopods)	whole	2	4190	ng/g ww	Σ nC_{12}–nC_{36}, pristine, phytane	KAVIK-AXYS, 2004
Coastal Beaufort Sea (69–70°N, 134–137°W)	1987	Phytoplankton		–	14800	ng/L	Σ nC_{12}–nC_{36}, pristine, phytane	Yunker et al., 1991
Coastal Beaufort Sea (69–70°N, 134–137°W)	1987	Zooplankton		–	146–1760	mg/L	Σ nC_{12}–nC_{36}, pristine, phytane	Yunker et al., 1991
Coastal Beaufort Sea (69–70°N, 134–137°W)	2002	*Stenodus leucichthys* (inconnu)	whole	1	6	ng/g ww	Σ 17 PAH	KAVIK-AXYS, 2004
Coastal Beaufort Sea (69–70°N, 134–137°W)	2002	*Stenodus leucichthys* (inconnu)	whole	1	nd	ng/g ww	Σ 25 alkylated PAH	KAVIK-AXYS, 2004
Coastal Beaufort Sea (69–70°N, 134–137°W)	2002	*Eleginus navaga* (Saffron Cod)	whole	4	4	ng/g ww	Σ 17 PAH	KAVIK-AXYS, 2004

Appendix 4.3. Cont.

Location	Year	Species	Tissue	n	Mean concentration	Units	Remarks	Source
Coastal Beaufort Sea (69–70°N, 134–137°W)	2002	*Eleginus navaga* (Saffron Cod)	whole	4	2	ng/g ww	Σ 25 alkylated PAH	KAVIK-AXYS, 2004
Coastal Beaufort Sea (69–70°N, 134–137°W)	2002	*Clupea harengus pallasi* (Pacific herring)	whole	2	6	ng/g ww	Σ 17 PAH	KAVIK-AXYS, 2004
Coastal Beaufort Sea (69–70°N, 134–137°W)	2002	*Clupea harengus pallasi* (Pacific herring)	whole	2	1	ng/g ww	Σ 25 alkylated PAH	KAVIK-AXYS, 2004
Coastal Beaufort Sea (69–70°N, 134–137°W)	2002	*Lota lota* (burbot)	whole	5	3	ng/g ww	Σ 17 PAH	KAVIK-AXYS, 2004
Coastal Beaufort Sea (69–70°N, 134–137°W)	2002	*Lota lota* (burbot)	whole	5	1	ng/g ww	Σ 25 alkylated PAH	KAVIK-AXYS, 2004
Coastal Beaufort Sea (69–70°N, 134–137°W)	2002	*Liopsetta glacialis* (Arctic flounder)	whole	2	5	ng/g ww	Σ 17 PAH	KAVIK-AXYS, 2004
Coastal Beaufort Sea (69–70°N, 134–137°W)	2002	*Liopsetta glacialis* (Arctic flounder)	whole	2	5	ng/g ww	Σ 25 alkylated PAH	KAVIK-AXYS, 2004
Coastal Beaufort Sea (69–70°N, 134–137°W)	2002	*Coregonus clupeaformis* (lake whitefish)	whole	4	1	ng/g ww	Σ 17 PAH	KAVIK-AXYS, 2004
Coastal Beaufort Sea (69–70°N, 134–137°W)	2002	*Coregonus clupeaformis* (lake whitefish)	whole	4	2	ng/g ww	Σ 25 alkylated PAH	KAVIK-AXYS, 2004
Coastal Beaufort Sea (69–70°N, 134–137°W)	2002	*Coregonus sardinella* (least cisco)	whole	2	72	ng/g ww	Σ 17 PAH	KAVIK-AXYS, 2004
Coastal Beaufort Sea (69–70°N, 134–137°W)	2002	*Coregonus sardinella* (least cisco)	whole	2	425	ng/g ww	Σ 25 alkylated PAH	KAVIK-AXYS, 2004
Coastal Beaufort Sea (69–70°N, 134–137°W)	2002	Amphipods	whole	3	11	ng/g ww	Σ 17 PAH	KAVIK-AXYS, 2004
Coastal Beaufort Sea (69–70°N, 134–137°W)	2002	Amphipods	whole	3	13	ng/g ww	Σ 25 alkylated PAH	KAVIK–AXYS, 2004
Coastal Beaufort Sea (69–70°N, 134–137°W)	2002	*Saduria entomona* (isopod)	whole	2	18	ng/g ww	Σ 17 PAH	KAVIK–AXYS, 2004
Coastal Beaufort Sea (69–70°N, 134–137°W)	2002	*Saduria entomona* (isopod)	whole	2	20	ng/g ww	Σ 25 alkylated PAH	KAVIK-AXYS, 2004
Husky lakes	2002	*Salvelinus namayacush* (lake trout)	whole	10	6	ng/g ww	Σ 17 PAH	Stern, 2003
Husky lakes	2002	*Salvelinus namayacush* (lake trout)	whole	10	36	ng/g ww	Σ 25 alkylated PAH	Stern, 2003
Husky lakes	2002	*Coregonus clupeaformis* (lake whitefish)	whole	5	6	ng/g ww	Σ 17 PAH	Stern, 2003
Husky lakes	2002	*Coregonus clupeaformis* (lake whitefish)	whole	5	15	ng/g ww	Σ 25 alkylated PAH	Stern, 2003
Husky lakes	2002	*Clupea harengus pallasi* (Pacific herring)	whole	5	2	ng/g ww	Σ 17 PAH	Stern, 2003
Husky lakes	2002	*Clupea harengus pallasi* (Pacific herring)	whole	5	26	ng/g ww	Σ 25 alkylated PAH	Stern, 2003
Husky lakes	2002	*Esox lucius* (northern pike)	whole	1	10	ng/g ww	Σ 17 PAH	Stern, 2003
Husky lakes	2002	*Esox lucius* (northern pike)	whole	1	34	ng/g ww	Σ 25 alkylated PAH	Stern, 2003
Husky lakes	2002	*Stenodus leucichthys* (inconnu)	whole	1	2	ng/g ww	Σ 17 PAH	Stern, 2003
Husky lakes	2002	*Stenodus leucichthys* (inconnu)	whole	1	14	ng/g ww	Σ 25 alkylated PAH	Stern, 2003

[a] Naphthalene, 2-methylnaphthalene, fluorene, phenanthrene, fluoranthene; [b] 14 lighter PAH (2–3 rings) and 12 higher PAH (4–7 rings).

Appendix 4.4.
Concentrations of petroleum hydrocarbons in marine and estuarine sediments.

Location	Year	n	Depth	Mean concentration	Units	Remarks	Source
Russia							
Pechora Sea	1998	–	Surface bottom sediment	95	ng/g dw	Σ >18 PAH	Dahle et al., 2006a
White Sea				74			
Kara Shelf Sea				133			
S-E Barents Sea				260			
Novaya Zemlya inshore				222			
S-W Barents Sea				462			
Ob and Yenisey estuaries				367			
N-E Barents Sea				447			
Franz Josef Land inshore				520			
N-W Barents Sea				2109			
Kola and Pechenga Bays				3635			
Guba Pechenga	1997	–	Bottom sediments	1481	ng/g dw	Σ 27 PAH	Savinov et al., 2003
Areas adjacent to Guba Pechenga				255			
Barents Sea, Bear Island Bank	2001–2002	17	Marine surface sediments	969	ng/g dw	Σ 13 PAH	Zhilin et al., 2004
Central Plateau		3		135			
Barents Sea, littoral of inner Kola Bay	2003	–		12562		Σ 18 PAH	Zhilin et al., 2003
Barents Sea, inner Kola Bay	2003			2419			
Pechora River delta, 2000–2003	2001–2002	–	Surface sediments	358	ng/g dw	Σ 21 PAH	Dauvalter, 2004
Barents Sea, outer Motovsky Bay	2002	–	Marine surface sediments	154	ng/g dw	Σ 18 PAHs	Alexeeva et al., 2005
Barents Sea, outer Motovsky Bay				153			
Barents Sea, outer Kola Bay				1084			
Barents Sea, outer Kola Bay	2003	–	Marine surface sediments	823	ng/g dw	Σ 18 PAHs	Alexeeva et al., 2005
Kara Sea	1993	–	Particulate phase	138	ng/g	Σ >12 PAH	Fernandes and Sicre, 1999
Ob mixing zone				46			
Ob riverine zone				9			
Yenisey mixing zone				299			
Yenisey riverine zone				236			
Kara Sea	1993	–	Surface sediment	37	ng/g dw	Σ >12 PAH	Fernandes and Sicre, 1999
Ob mixing zone				54			
Ob riverine zone				33			
Yenisey mixing zone				111			
Yenisey riverine zone				88			
Tsivolki Bay, Kara Sea	2004	–	Bottom sediments	291	ng/g dw	Σ >27 PAH	Savinov et al., 2006
Stepovogo Bay, Kara Sea				253			
Abrosimova Bay, Kara Sea				294			
Novaya Zemlya Trough, Kara Sea				632			
South-Western Kara Sea				323			
Norway (Svalbard)							
Svalbard inshore	1998	–	Surface bottom sediment	8697	ng/g dw	Σ >18 PAH	Dahle et al., 2006a
Aldger Island	1992	–	0–5 cm depth	150	ng/g	Perylene only	Petrova et al., 2000
			90–100 cm depth	580			
			190–200 cm depth	160			

Appendix 4.4. Cont.

Location	Year	n	Depth	Mean concentration	Units	Remarks	Source
Faroe Islands							
Referensu statión	2002	–	Bottom sediments, coastal zone	83	ng/g dw	Σ 18 PAH	Dam and Danielsen, 2003
Sandagerð				3975			
Út fyri havnará				8900			
Kongabrúgvin				2840			
Bursatangi				4572			
Millum Tinganes og Bacalao				9025			
Vágsbotn				15954			
Landingarplássi í Vágsbotni				62220			
Útfyri skipasmiðuna				713			
Álakeri				83			
60º21.41′N, 05º22.94′W		–	Bottom sediments from Faroe-Shetland channel	27	ng/g dw	Σ PAH[a]	Hoydal and Dam, 2004
60º23.20′N, 05º13.89′W				31			
60º20.85′N, 05º09.92′W				45			
60º23.53′N, 05º07.50′W				32			
60º35.43′N, 05º18.58′W				23			
60º32.30′N, 05º12.34′W				28			
60º26.50′N, 05º00.03′W				24			
60º22.07′N, 04º56.91′W				19			
60º28.97′N, 05º55.92′W				21			
60º24.88′N, 04º45.79′W				16			
60º33.21′N, 04º56.18′W				36			
60º28.39′N, 04º45.70′W				23			
60º34.79′N, 04º49.20′W				20			
60º31.92′N, 04º44.64′W				33			
60º32.00′N, 04º38.00′W				29			
60º38.06′N, 04º37.96′W				23			
60º21.41′N, 05º22.94′W				22		Σ 16 PAH	
60º23.20′N, 05º13.89′W				18			
60º20.85′N, 05º09.92′W				22			
60º23.53′N, 05º07.50′W				19			
60º35.43′N, 05º18.58′W				26			
60º32.30′N, 05º12.34′W				34			
60º26.50′N, 05º00.03′W				20			
60º22.07′N, 04º56.91′W				18			
60º28.97′N, 05º55.92′W				21			
60º24.88′N, 04º45.79′W				16			
60º33.21′N, 04º56.18′W				27			
60º28.39′N, 04º45.70′W				22			
60º34.79′N, 04º49.20′W				25			
60º31.92′N, 04º44.64′W				32			
60º32.00′N, 04º38.00′W				26			
60º38.06′N, 04º37.96′W				34			
Greenland							
Station TAB1 near Thule base	2001	–	Sediment	367	ng/g dw	Σ 25 PAH	Glahder et al., 2003
Station NSB near Thule base				469			
Station TAB31 near Thule base				291			
Station TAB37 near Thule base				115			
Qaanaaq				135			

Appendix 4.4. Cont.

Location	Year	n	Depth	Mean concentration	Units	Remarks	Source
Eqalugarsuit	2002	–	Waste dump sediments	138	ng/g dw	Σ 17PAH	Greenland AMAP studies (waste dump sediments), 2002.
Nanortalik				281			
Nanortalik (Fine, organic rich)				1373			
Attu				132			
Aasiaat				85 and 200			
Scoresbysund				644			
Thule				156			
Disko				357			
South-West Greenland	2004	–	Deep sea sediments	80	ng/g dw	Σ 17PAH	Greenland AMAP studies (deep sea sediments), 2004.
Atammik				75			
Seep F, South-West Greenland				57			
Seep E, South-West Greenland				76			
Canyon A, South-West Greenland				57			
Lysefjord				48			
Young Sound				59			
North Greenland Sea				921			
Canada							
Southern Beaufort Sea (69.5–71.5°N, 127–137°W)	2002	–	Marine surface sediments	13	µg/g dw	Σ nC_{12}–nC_{36}, pristane, phytane	KAVIK-AXYS, 2004
Southern Beaufort Sea (69–70°N, 134–137°W)				10	µg/g dw	Σ nC_{12}–nC_{36}	Yunker et al., 2002
Southern Beaufort Sea (69–70°N, 134–137°W)				912	ng/g dw	Σ 17 PAH	KAVIK-AXYS, 2004
Southern Beaufort Sea (69–70°N, 134–137°W)				7	ng/g dw	Σ 17 PAH	Yunker et al., 2002
Southern Beaufort Sea (69–70°N, 134–137°W)				2	ng/g dw	C_1–C_4 Naphthalenes + Phenanthrenes, Anthracenes – Retene	Yunker et al., 2002
Beaufort Sea (69.6–71.5°N, 127–139°W)				1946	ng/g dw	25 alkylated PAH	KAVIK-AXYS, 2004
Baffin region				24	ng/g dw	Σ 17 PAH	DND, 1994
Cambridge Bay				27	ng/g dw	Σ 17 PAH	DND, 1994
Cambridge Bay				94	ng/g dw	Alkylated PAH	DND, 1994
Mackenzie Delta Lakes (68.7°N, 134.7°W)	2002	–	Freshwater surface sediments	13	ng/g dw	Σ nC_{12}–nC_{36}	Yunker et al., 2002
				12	ng/g dw	Σ 17 PAH	Yunker et al., 2002
				3	ng/g dw	Naphthalenes, Phehanthrenes, Anthracenes, Retene	Yunker et al., 2002

[a] Naphthalene, phenantrene, dibenzothiophene and their C_1–C_3 alkylated homologues.

Appendix 4.5.
Concentrations of petroleum hydrocarbons in seawater and ice.

Location	Year	Matrix	n	Mean concentration	Units	Remarks	Source
Canada							
Southern Beaufort Sea	1987	Seawater	–	17	µg/L	Σ nC_{11}–nC_{36}	Yunker et al., 1991
Coastal Beaufort Sea (69–70°N, 134–137°W)	2002	Seawater	–	10	µg/L	Σ nC_{12}–nC_{36}, pristane, phytane	KAVIK-AXYS, 2004
Southern Beaufort Sea (69–70°N. 134–137°W)		Marine particulates	–	5	µg/g dw	Σ 17 PAH	KAVIK-AXYS, 2004
Southern Beaufort Sea (69–70°N. 134–137°W)		Marine particulates	–	8	µg/g dw	Σ 17 PAH	KAVIK-AXYS, 2004
Coastal Beaufort Sea (69–70°N, 134–137°W)		Marine particulates	–	4	µg/L	Σ nC_{12}–nC_{36}, pristane, phytane	KAVIK-AXYS, 2004
Russia							
Mendeleev Rise, Arctic Ocean	2000	Snow (dissolved)	–	2	ng/L	Σ 8 PAH	Nemirovskaya and Novigatskii, 2003
Franz-Victoria Trough, N. Barents Sea		Snow (dissolved)	–	30	ng/L	Σ 8 PAH	Nemirovskaya and Novigatskii, 2003
Mendeleev Rise, Arctic Ocean		Snow (particulate)	–	3	ng/L	Σ 8 PAH	Nemirovskaya and Novigatskii, 2003
Franz-Victoria Trough, N. Barents Sea		Snow (particulate)	–	50	ng/L	Σ 8 PAH	Nemirovskaya and Novigatskii, 2003
Mendeleev Rise, Arctic Ocean		Upper ice core (dissolved)	–	6	ng/L	Σ 8 PAH	Nemirovskaya and Novigatskii, 2003
Franz-Victoria Trough, N. Barents Sea		Upper ice core (dissolved)	–	110	ng/L	Σ 8 PAH	Nemirovskaya and Novigatskii, 2003
Mendeleev Rise, Arctic Ocean		Lower ice core (dissolved)	–	4	ng/L	Σ 8 PAH	Nemirovskaya and Novigatskii, 2003
Franz-Victoria Trough, N. Barents Sea		Lower ice core (dissolved)	–	86	ng/L	Σ 8 PAH	Nemirovskaya and Novigatskii, 2003
Mendeleev Rise, Arctic Ocean		Upper ice core (particulate)	–	21	ng/L	Σ 8 PAH	Nemirovskaya and Novigatskii, 2003
Franz-Victoria Trough, N. Barents Sea		Upper ice core (particulate)	–	47	ng/L	Σ 8 PAH	Nemirovskaya and Novigatskii, 2003
Mendeleev Rise, Arctic Ocean		Lower ice core (particulate)	–	54	ng/L	Σ 8 PAH	Nemirovskaya and Novigatskii, 2003
Franz-Victoria Trough, N. Barents Sea		Lower ice core (particulate)	–	35	ng/L	Σ 8 PAH	Nemirovskaya and Novigatskii, 2003
Mendeleev Rise, Arctic Ocean		Water under ice (dissolved)	–	2	ng/L	Σ 8 PAH	Nemirovskaya and Novigatskii, 2003
Franz-Victoria Trough, N. Barents Sea		Water under ice (dissolved)	–	26	ng/L	Σ 8 PAH	Nemirovskaya and Novigatskii, 2003
Mendeleev Rise, Arctic Ocean		Water under ice (particulate)	–	2	ng/L	Σ 8 PAH	Nemirovskaya and Novigatskii, 2003

Appendix 4.5. Cont.

Location	Year	Matrix	n	Mean concentration	Units	Remarks	Source
Barents Sea: Spitsbergen Bank	2001–2002	Surface seawater	3	25	ng/L	Σ 13 PAH	Zhilin et al., 2004
Barents Sea: Bear Island Bank		Surface seawater	21	39	ng/L	Σ 13 PAH	Zhilin et al., 2004
Barents Sea: Hope Island area		Surface seawater	9	35	ng/L	Σ 13 PAH	Zhilin et al., 2004
Barents Sea: Hopen Trehcn		Surface seawater	3	99	ng/L	Σ 13 PAH	Zhilin et al., 2004
Barents Sea: Central Bank		Surface seawater	4	80	ng/L	Σ 13 PAH	Zhilin et al., 2004
Barents Sea: Bear Island Bank	2001–2002	Bottom layer water	1	142	ng/L	Σ 13 PAH	Zhilin et al., 2004
Barents Sea: Hopen Trench			3	110	ng/L	Σ 13 PAH	Zhilin et al., 2004
Barents Sea: Central Bank			1	170	ng/L	Σ 13 PAH	Zhilin et al., 2004

Appendix 4.6.
Concentrations of petroleum hydrocarbons in soils.

Location	Year	Mean concentration	Units	Remarks	Source
Greenland					
Usuk	2000	106	ng/g ww	Σ 26 PAH[a]	Riget et al., 2003
Igaliko	2000	44	ng/g ww	Σ 26 PAH[a]	Riget et al., 2003
Norway (Svalbard)					
Central Spitsbergen D1	2001	142	ng/g	Σ 15 PAH	Gulinska et al., 2003
Central Spitsbergen S3	2001	450	ng/g	Σ 15 PAH	Gulinska et al., 2003
Central Spitsbergen P3	2001	158	ng/g	Σ 15 PAH	Gulinska et al., 2003

[a] 14 lighter PAH (2–3 rings) and 12 higher PAH (4–7 rings).

Appendix 4.7.
Concentrations of petroleum hydrocarbons in air.

Location	Year	Mean concentration	Units	Remarks	Source
Canada					
Alert	1992–1996	0.015 to 0.632	ng/m^3	Σ 12 PAH	Prevedouros et al., 2004

Appendix 4.8.
Concentrations of petroleum hydrocarbons in the freshwater environment.

Location	Year	Phase	Mean concentration	Units	Remarks	Source
Greenland						
Lake near Igaliko	2000	Lake sediments	120	ng/g	Σ 26 PAH[a]	Riget et al., 2003
Norway (Svalbard)						
Ossian	1995	Lake surface sediments	20	ng/g	Σ 11 PAH	Rose et al., 2004
Bjørnvatnet			23			
Ytertjørna			51			
Daltjørna			37			
Tenndammen			191			
Bjørnvatnet	1995	Lake pre-industrial sediments	25	ng/g	Σ 11 PAH	Rose et al., 2004
Ytertjørna			640			
Daltjørna			80			
Tenndammen			379			
Canada						
Mackenzie River at Inuvik	1993	River water suspended particulate	1	mg/L	Σ nC_{12}–nC_{36}	Yunker et al., 2002
Mackenzie River at Aklavik			5			
Mackenzie River Tsiigehtchie			2			
Coppermine River			1.2			
Burnside River			19			
Ellice River			5.7			
Dubwant River			33			
Thelon River			22			
Back River			50			
Kazan River			11			
Hayes River			2			
Quoick River			28			
Lorillard River			24			
Mackenzie River at Inuvik	1994	River water suspended particulate	27	mg/L	Σ nC_{12}–nC_{36}	Yunker et al., 2002
Mackenzie River at Aklavik			69			
Mackenzie River Tsiigehtchie			13			
Mackenzie River at Inuvik	1993	Freshwater particulates	2	μg/g dw	Σ 17 PAH	Yunker et al., 2002
Mackenzie River at Aklavik			1			
Mackenzie River Tsiigehtchie			0			
Coppermine River			0			
Burnside River			0			
Ellice River			0			
Dubwant River			0			
Thelon River			0			
Back River			0			
Kazan River			1			
Hayes River			0			
Quoick River			1			
Lorillard River			0			
Mackenzie River at Inuvik	1994	Freshwater particulates	1	μg/g dw	Σ 17 PAH	Yunker et al., 2002
Mackenzie River at Aklavik			1			
Mackenzie River Tsiigehtchie			1			
Mackenzie River (69.3°N, 133.9°W)			1			

[a] 14 lighter PAH (2–3 rings) and 12 higher PAH (4–7 rings).

References

Aas, W., S. Solberg, T. Berg, S. Manø and K.E. Yttri, 2006. Overvåking av langtransportert forurenset luft og nedbør. Atmosfærisk tilførsel, 2005. (Monitoring of contaminated air and precipitation from long range transport. Atmospheric inputs 2005.) Report No. OR 36/2006, 162p. (In Norwegian)

ACIA, 2005. Arctic Climate Impact Assessment. Cambridge University Press. 1042p.

AEUB, 2001. Alberta's reserves 2000 and supply/demand outlook 2001-2010. Alberta Energy and Utilities Board Statistical Series 2001-98. 123p.

AKDEC, 2003. Statewide Summary of Oil and Hazardous Substance Spill Data (Fiscal Years 1996-2002). Provisional report prepared by the Alaska Department of Environmental Conservation, Division of Spill Prevention and Response. November 2003.

AKDEC, 2005. Situation report #105. Alaska Department of Environmental Conservation, Division of Spill Prevention and Response, Prevention and Emergency Response Program.

AKDEC, 2006. Situation report #22. Alaska Department of Environmental Conservation, Division of Spill Prevention and Response, Prevention and Emergency Response Program.

AKDEC, 2007. 10-Year Statewide Summary of Oil and Hazardous Substance Spill Data (July 1, 1995 - June 30, 2005). Alaska Department of Environmental Conservation, Division of Spill Prevention and Response, June 2007.

Alberty, R.A. and A.K. Reif, 1988. Standard chemical thermodynamic properties of polycyclic aromatic hydrocarbons and their isomer groups. I. Benzene series. Journal of Physical and Chemical Reference Data, 17:241-253.

Alexeeva, L., D. Samsonov, G. Chernik, D. Matishov, O. Mokrotovarova, N. Plotitsyna, A. Zhilin, V. Savinov and T. Savinova, 2005. Monitoring of hazardous substances in the coastal areas of the Barents Sea: harmonisation with OSPAR's Joint Monitoring and Assessment Programme (JAMP) – Kola Bay. Report APN-414.2678, 108p.

Alyeska Pipeline Service Company, 2007. The Facts. Trans Alaska Pipeline System 2007. Annual report of the Alyeska Pipeline Service Company, http://www.alyeska-pipe.com

AMAP, 1998. AMAP Assessment Report: Arctic Pollution Issues. Arctic Monitoring and Assessment Programme. Oslo, Norway. xii + 859p.

AMAP, 2004. Persistent Toxic Substances, Food Security and Indigenous Peoples of the Russian North. Final report. Arctic Monitoring and Assessment Programme (AMAP), Oslo. AMAP Report 2004:2. 192p.

Amiro, B.D., J.B. Todd, B.M. Wotton, K.A. Logan, M.D. Flannigan, B.J. Stooks, J.A. Masons, D.L. Martell and K.G. Hirsch, 2001. Direct carbon emissions from Canadian forest fires, 1959-1999. Canadian Journal of Forest Research, 31:512-515.

Anderson, C.M. and R.P. LaBelle, 2000. Update of comparative occurrence rates for offshore oil spills. Spill Science and Technology Bulletin, 6(5):303-321.

Asmund, G., 2007. Recepientundersøgelser ved grønlandske lossepladser. Faglig rapport fra DMU nr. 612, 116 pp. Available at http://www.dmu.dk/Pub/FR612.pdf.

Bambulyak, A. and B. Frantzen, 2005. Oil transport from the Russian part of the Barents Region. Status per January 2005. Svanhovd Environmental Centre, Norway. 91p.

Bambulyak, A. and B. Frantzen, 2007. Oil transport from the Russian part of the Barents Region. The Norwegian Barents Secretariat and Akvaplan-niva, Norway. 87p.

Banet, A., 2005. Review of Arctic Alaska Oil Seeps. Unpublished data. Bureau of Land Management, Anchorage, Alaska.

Banet, A., 2006. Legacy wells in Alaska National Petroleum Reserve. Unpublished data. Bureau of Land Management, Anchorage, Alaska.

Becker, P.R. and C.A. Manen, 1988. Natural oil seeps in the Alaskan Marine Environment. Final Report, Outer Continental Shelf Environmental Assessment Program (OCSEAP), U.S. Department of Commerce, Technical Information Service, PB88-235965 p.114p.

Bennett, R., S. Blasco, V. Kostylev, B. MacLean, A. Rochon, J. Hughes-Clarke, J. Beaudoin and J. Bartlett, 2005. Seabed processes and geologic structure of the Northwest Passage. Poster presentation at the 2nd ArcticNet scientific Meeting, Banff, Alberta.

Blasco, S.; Bennett, R.; Kostylev, V.; Campbell, P.; Shearer, J.; and Conlan, K. 2006. Seabed topography, geological structure and evolution of the sea-ice regime on the Beaufort Shelf at the western entrance to the Northwest Passage: CCGS Nahidik Research. The Canadian Geomorphology Research Group, 3rd Annual ArcticNet Scientific Meeting. December 12-15, 2006. Victoria, British Columbia.

BLM, 2003. Integrated Activity Plan/Environmental Impact Statement. NE National Petroleum Reserve, Alaska. Bureau of Land Management, U.S. Department of the Interior.

BLM, 2005. Supplemental for the 1998 Environmental Impact Assessment/ Integrated Activity Plan for the Northeast National Petroleum Reserve in Alaska. Bureau of Land Management, U.S. Department of the Interior.

Bobrovitskaya, N.N., A.V. Kokorev and N.A. Lemeshko, 2003. Regional patterns in recent trends in sediment yields of Eurasian and Siberian Rivers. Global and Planetary Change, 39:127-146.

Boehm, P.D., 1987. Transport and transformation processes regarding hydrocarbon and metal pollutants in offshore sedimentary environments. In: D.F. Boesch and N.N. Rabalais (eds.). Long-Term Environmental Effects of Offshore Oil and Gas Development, pp. 233-286. Elsevier Applied Science.

Boehm, P.D. and J.W. Farrington, 1984. Aspects of the polycyclic aromatic hydrocarbon geochemistry of recent sediments in the Georges Bank region. Environmental Science and Technology, 18:840-845.

Boehm, P.D., J. Brown, H. Camp, L. Cook, J. Trefry, R.P. Trocine, R.D. Rember, G. Shepard, C. Malme, P. Krumhansl and M.L. Knack, 2001. Arctic Nearshore Impact Monitoring in the Development Area (ANIMIDA) Phase I. Draft Final Report. Contract No. 143501-99-CT -30998. U.S. Department of the Interior, Minerals Management Service.

Boitsov, S., J. Klungsøyr and H. Jensen, 2007. Concentrations of petroleum hydrocarbons in sediments and seawater from the Barents and Norwegian Seas 2003-2005. Fisken og Havet No. 3, 52p.

Bojesen-Koefoed, J.A., F.G. Christiansen, H.P. Nytoft and A.K. Pedersen, 1999. Oil seepage onshore West Greenland: evidence of multiple source rocks and oil mixing. In: A.J. Fleet and S.A.R. Boldy (eds.). Petroleum Geology of Northwest Europe. Proc. 5th Conference, pp. 305–314. Geological Society of London.

Borovinskih, A.P., 2004. Regulative and legal support of ecological rehabilitation of oil polluted territories in Republic of Komi. Proc. 3rd Scientific Practical Conference on Ecological Work in Oil and Gas Fields of the Timano-Pechora Province. Conditions and Prospects, pp. 15-17. Ukhta, September 6-9, 2004.

Box, J.E. and D.H. Bromwich, 2004. Greenland ice sheet surface mass balance 1991-2000. Application of Polar MM5 mesoscale model and in-situ data. Journal of Geophysical Research, 109, D16105.

BP, 2005. Russian resolution. BP Frontiers, Issue 13: August 2005.

Brandvik, P.J., T. Strøm Kristiansen and P.S. Daling, 1996a. Weathering properties of the *Frøy* crude, Lillefrigg condensate and the 75:25 blend of these two products – a handbook for Elf Norge. SINTEF report no. 41.5135/01/96.

Brandvik, P.J., T. Strøm Kristiansen, A. Lewis, P.S. Daling, M. Reed, H. Rye and H. Jensen, 1996b. The Norwegian Sea Trial 1995: offshore testing of two dispersant application systems and simulation of an underwater pipeline leakage – a summary paper, pp. 1395 1405. Proc. 19th Arctic Marine Oilspill Technical Seminar, Environment Canada, Ottawa, Ontario.

Brandvik, P.J., I. Singsaas and P.S. Daling, 2004. Oil spill R&D in Norwegian Arctic waters with special focus on large-scale oil weathering experiments. Proc. Inter Spill Conference, Trondheim, Norway, 14-17 June, 2004.

Brumbaugh, R. and S. Porhola, 2004. Alaska Legacy Wells Summary Report: National Petroleum Reserve, Alaska. Bureau of Land Management 66p.

Budzinski, H., I. Jones, J. Bellocq, C. Piérard and P. Garrigues, 1997. Evaluation of sediment contamination by polycyclic aromatic hydrocarbons in the Gironde Estuary. Marine Chemistry, 58:85-97.

Burns, W.A, P.J. Mankiewicz, A.E. Bence, D.S. Page and K.R. Parker, 1997. A principal-component and least-squares method for allocating polycyclic aromatic hydrocarbons in sediment to multiple sources. Environmental Toxicology and Chemistry, 16:1119-1131.

CAFF, 2005. Status and trends in species and populations. In: Arctic Flora and Fauna. Conservation of Arctic Flora and Fauna. Akureyri, Iceland.

Carey, J.H., E.D. Ongley and E. Nagy, 1990. Hydrocarbon transport in the Mackenzie River, Canada. Science of the Total Environment, 97/98:69-88.

Carroll, J.L., V. Savinov, T. Savinova, S. Dahle and D.C.G. Muir, 2007. Fluvial transport of dissolved PAHs to the Arctic Ocean from the Russian rivers Ob and Yenisei. 21st International Symposium for Polycyclic Aromatic Compounds, Trondheim, 5-10 August, p.120.

CCME, 1998. Canadian (interim) sediment quality guidelines for the protection of aquatic life. Canadian Council of Ministers of the Environment. Environment Canada, Ottawa, Ontario.

Chapman, R.M. and E.G. Sable, 1960. Stratigraphy and structure of the Okpikruak and Kiruktagiak Rivers area, Alaska. U.S. Geological Survey Open File Report.

Chernova, T.G., P.S. Rao and Y.I. Pikovskii, 2001. The composition and source of hydrocarbons in sediments taken from the tectonically active Andaman Backarc Basin, Indian Ocean. Marine Chemistry, 75:1-15.

Christensen, G.N., V. Savinov, T. Savinova, L. Alexeeva, A. Kochetkov, A. Konoplev, A. Pasynkova, D. Samsonov, V.A. Dauvalter, N.A. Kashulin and S.S. Sandimirov, 2007a. Screening studies of POP levels in bottom sediments from selected lakes in the Paz watercourse. Akvaplan-niva report 3665.01. 64p.

Christensen, G.N., V. Savinov, T. Savinova, L. Alexeeva, A. Kochetkov, A. Konoplev, A. Pasynkova, D. Samsonov, N.A. Kashulin, I. Koroleva, S.S. Sandimirov and D. Morozov, 2007b. Screening studies of POP levels in fish from selected lakes in the Paz watercourse. Akvaplan-niva report 3665.02. 44p.

Conly, F.M., R.W. Crosley and J.V. Headley, 2002. Characterizing sediment sources and natural hydrocarbon inputs in the lower Athabasca River. Journal of Environmental Engineering and Science, 1:187-199.

Connell, D.W. and G. J. Miller, 1981. Petroleum hydrocarbons in aquatic ecosystems – behavior and effects of sublethal concentrations. 1. CRC Critical Reviews in Environmental Control, 11(1):37-104.

Cushing, D.H. and J.J. Walsh, 1976. The Ecology of the Seas. W.B. Saunders Co. 467p.

Dahle, S., G.G. Matishov, T.N. Savinova, B. Killie and V.M. Savinov, 1998. Polyaromatic hydrocarbons in bottom sediments from the White Sea. Proc. VII International Conference on The Study, Rational Exploitation and Conservation of Natural Resources of the White Sea, pp.51-53, Arkhangelsk, September 1998. (In Russian)

Dahle, S., V. Savinov, G. Matishov, A.Evenset and K. Næs, 2003. Polycyclic aromatic hydrocarbons (PAHs) in bottom sediments of the Kara Sea shelf, Gulf of Ob and Yenisei Bay. Science of the Total Environment, 306:57-71.

Dahle, S., V. Savinov, V. Petrova, J. Klungsøyr , T. Savinova, G. Batova and A. Kursheva, 2006a. Polycyclic aromatic hydrocarbons (PAHs) in Norwegian and Russian Arctic marine sediments: concentrations, geographical distribution and sources. Norwegian Journal of Geology, 86:41-50.

Dahle, S., G. Matishov and V. Savinov, 2006b. Sources of PAH contamination of bottom sediments from selected Barents Sea harbours. Arctic Shelf Conference, Murmansk 15-17 November 2006.

Dahle, S., V. Savinov, J. Klungsøyr, S. Boitsov, N. Plotitsyna, A. Zhilin, T. Savinova and V. Petrova, 2008a. Polyaromatic hydrocarbons (PAHs) in the Barents Sea sediments: Small changes over the recent 10 years. Marine Biology Research (accepted).

Dahle, S., V. Savinov, V. Petrova, J. Klungsøyr, D. Thomas, N. Plotitsyna, S. Boitsov and T. Savinova, 2008b. Circumpolar distribution of PAHs in marine bottom sediments. Proc. 2nd Annual Arctic Frontiers Conference, Tromsø, 20-25 January 2008.

Daling, P.S. and P.J. Brandvik, 2002. Weathering of oils at sea: comparisons between field data and model predictions. In: C.A. Brebbia (ed.). Oil Spill - Modelling and Processes, WIT Press.

Daling, P.S. and T. Strøm, 1999. Weathering of oils at sea: model/field data comparisons. Spill Science and Technology Bulletin, 5:63-74.

Dam, M. and J. Danielsen, 2003. Havnarvág- ein kanning av dálkingarstøðuni á Havnarvág og Yviri við Strond á sumri 2002. Technical part of a report for Tórshavn Kommune, Faroe Islands. (In Faroese)

Dauvalter, V., 2004. Geochemistry of the Pechora River delta, northeast European Russia. NGF Abstracts and Proceedings. Arctic Geology, Hydrocarbon Resources and Environmental Challenges, pp. 31-34. Tromsø, 23-26 May 2004.

Devon Canada Corporation, 2003. Beaufort Sea Exploration Drilling Program Application. Section 10: Chemical Oceanography. Calgary, Alberta. 116p.

DND, 1994. Historical Ocean Disposal in the Canadian Arctic: Survey of materials disposed in Cambridge Bay and the state of the marine environment. Department of National Defence, Ottawa. 76p.

DND, 1995. Baffin Region Ocean Disposal Investigation: Seabed debris and contaminant inputs near Iqualuit, Resolution Island, Cape Dyer and Kivitoo. Department of National Defence, Ottawa. 127p.

DNV, 2007. Environmental and baseline surveys in Region VI – Haltenbanken 2006. Summary Report. Det Norske Veritas Report No. 2007-0530, 47p.

Dome Petroleum Limited, Esso Resources Canada Limited and Gulf Canada Resources Inc. 1982. Beaufort Sea–Mackenzie Delta Environmental Impact Statement. Volume 3A.

Dunbar, M.J., 1977. Polar Oceans. Arctic Institute of North America. 682p.

Eberlein, G.D., R.M. Chapman and C.D. Reynolds, 1950. The stratigraphy and structure of the Aupuk anticline: U.S. Geological Survey, Open File Report.

Eisler, R., 1987. Polyaromatic hydrocarbon hazards to fish, wildlife and invertebrates: A synoptic review. US fish and Wildlife Service. Biological Report 85 (1.11).

Elvidge, C.D., K.E. Baugh, B.T. Tuttle, A.T. Howard, D.W. Pack and C. Melesi, 2007. A twelve year record of national and global gas flaring volumes estimated using satellite data. Final report to the World Bank-May 30, 2007. http://www.ngdc.gov/dmsp/interest/gas_flares.html

Environment Canada, 2006. Hazardous Materials Spill Database. http://www.e-engine.ca/eps-Spillreportonnes per annumsp

Eriksson, M., E. Sodersten, Z. Yu, G. Dalhammar and W.W. Mohn, 2003. Degradation of polycyclic aromatic hydrocarbons at low temperature at aerobic and nitrate-reducing conditions in enrichment cultures from northern soils. Applied and Environmental Microbiology, 69:275-284.

ESMAP, 2003. Russia Pipeline Oil Spill Study. Joint UNDP/World Bank Energy Sector Management Assistance Program (ESMAP). 87p.

FEMA, 2004. Taken from the Federal Emergency Management Agency (FEMA) website. http://www.fema.gov/hazards/earthquakes/nehrp/sty_oil.shtm

Fernandes, M.B. and M.-A. Sicre, 1999. Polycyclic aromatic hydrocarbons in the Arctic: Ob and Yenisei Estuaries and Kara Sea Shelf. Estuarine, Coastal and Shelf Science, 48:725-737.

Filler, D.M. and D.L. Barnes, 2003. Technical procedures for recovery and evaluation of chemical spills on tundra. Cold Regions Science and Technology, 37:121-135.

Fingas, M.F. and B. Hollebone, 2001. The fate and behaviour of oil in freezing environments. In: Marine oil spills in ice and cold conditions. National Board of Waters and Environment. 14p.

Frantzen, B. and A. Bambulyak, 2003. Oil transport from the Russian part of the Barents Region. July 1, 2003. Svanhovd Environmental Centre, Norway. 27p.

French, H.M., 1980. Terrain, land use and waste drilling fluid disposal problems, Arctic Canada. Arctic, 33(4):794-806.

French, H.M., 1985. Surface disposal of waste drilling fluids, Ellef Ringnes Island, NWT: Short-term observations. Arctic, 38(4):292-302.

Gazprom, 2006. 2005 Environmental Report Gazprom.16 Nametkina St., 117997, Moscow, V-420, GSP-7 http://www.gazprom.ru/eng/articles/article8931.shtml

Gazprom, 2007. 2006 Environmental Report Gazprom. 16 Nametkina St., 117997, Moscow, V-420, GSP-7 http://www.gazprom.ru/eng/articles/article8931.shtml

GESAMP, 1993. Impact of oil and related chemicals and wastes on the marine environment. Joint Group of Experts on the Scientific Aspects of Marine Pollution. Report No. 50: 180p.

Glahder, C.M., G. Asmund, P. Mayer, P. Lassen, J. Strand and F. Riget, 2003. Marine Recipient Investigation near Thule Airbase 2002. Danish Ministry of Environment, Report No. 449. (In Danish)

Glotov, A.S., 2005. Status assessment based on environmental monitoring of Kumzha-9 emergency well at Nenetsky State Nature Reserve territory. Naryan-Mar.

GNWT, 2003. History of oil and gas in the Northwest Territories. Government of the Northwest Territories. www.iti.gov.nt.ca/mog/oil_gas/history

Golubeva, N.I., 2002. The results of studies on heavy metal contamination of air in open areas of the Barents Sea. Proceedings of the International conference on Present-day Issues of Oceanology of Shelf Seas of Russia, pp. 39-41. Rostov-na-Donu, June 13-15 2002. (In Russian)

Gordeev, V.V., 2002. Pollution of the Arctic. Regional Environmental Change, 3:88-98.

Green, N.W., K. Hylland, A. Ruus and M. Walday, 2004. Joint Assessment and Monitoring Programme (JAMP). National Comments Regarding the Norwegian Data for 2002. Niva Report OR-4778.

Grøsvik, B.E., S. Meier, K. Westrheim, H. Skarphéðinsdóttir, B. Liewenborg, L. Balk and J. Klungsøyr, 2007. Condition monitoring in the water column 2005: Oil hydrocarbons in fish from Norwegian waters. Institute of Marine Research, Bergen. Norway. 34p.

Gubinova, M.V., 2004. Problem of restoration of superficial water objects as habitat of hydrobionts in Usinsk area. Proc. 3rd Scientific Practical Conference on Ecological Work in Oil and Gas Fields of the Timano-Pechora Province. Conditions and Prospects, pp. 135-138. Ukhta, September 6-9 2004.

Gulinska, J., G. Rachlewicz, W. Szczucinski, D. Baralkiewicz, M. Kozka, E. Bulska and M. Burzyk, 2003. Soil contamination in high Arctic areas of human impact, central Spitsbergen, Svalbard. Polish Journal of Environmental Studies, 12(6):701-707.

Hallet, B., L. Hunter and J. Bogen, 1996. Rates of erosion and sediment evacuation by glaciers: A review of field data and their implications. Global and Planetary Change, 12:213-235.

Hanna, E., P. Huybrechts, I. Janssens, J. Cappelin, L. Steffen and A. Stephens, 2005. Runoff and mass balance of the Greenland Ice Sheet: 1958-2003. Journal of Geophysical Research, 110:D13.

Hansen, N., V.B. Jensen and P. Johansen, 1978. Hydrocarbons in marine organisms and sediments in Færingehavn. Report for the Ministry of Greenland. (In Danish)

Hasholt, B., 1996. Sediment transport in Greenland. In: D.E. Walling, B.W. Webb (eds). Erosion and Sediment Yield: Global and Regional Perspectives, pp. 105-114. International Association of Hydrological Sciences, Report No. 236.

Hasholt, B., N. Bobrovitskaya, J. Bogen, J. McNamara, S.H. Mernild, D. Milburn and D.S. Walling, 2005. Sediment transport to the Arctic Ocean and adjoining cold oceans. Fifteenth International Northern Research Basins Symposium and Workshop, pp. 41-67. Lulea to Kvikkjokk, Sweden. 29 Aug – 2 Sept, 2005.

Headley, J.V., P. Marsh and K.M. Peru, 2002. Origin of polycyclic aromatic hydrocarbons in lake sediment cores of the Mackenzie Delta. Journal of Environmental Science and Health, 37:1159-1180.

Hellou, J., C. Upshall, I.H. Ni, J.F. Payne and Y.S. Huang, 1991. Polycyclic aromatic hydrocarbons in harp seals (*Phoca groenlandica*) from the Northwest Atlantic. Archives of Environmental Contamination and Toxicity, 21:135-140.

Hites, R.A., R.E. Laflamme and G.G. Windsor, 1980. Polycyclic aromatic hydrocarbons in marine/aquatic sediments: Their ubiquity. In: L. Petrakis, F.T. Weiss (eds.). Petroleum in the Marine Environment, pp. 289-311. American Chemical Society Advances in Chemistry Series, 185.

Høydal, K., 2004. Background levels of oil-derived pollution in fish and invertebrates from the coastal zone around the Faroe Islands – Biomarker analyses in fish and analyses of PAH and metals in invertebrates. Masters Thesis, University of Copenhagen.

Høydal, K. and M. Dam, 2004. Føroya Umhvørvi í Tølum 2003, Heilsufrøðiliga Starvstovan, report 2. (In Faroese)

Hrudey, S.E., 1979. Sources and characteristics of liquid process wastes from Arctic offshore hydrocarbon exploration. Arctic, 32(1):3-21.

Ilyin, G.V., 2005. Regularities of accumulation and structure of petroleum hydrocarbons in sediments of the Kara Sea. Proc. International Conference on Theory and Practice of Complex Marine Studies in the Interests of Economy and Safety of Russian North, pp. 60-62. Murmansk, March 15-17 2005.

Ilyin, G.V., D.G. Matishov and N.E. Kasatkina, 2004. Chemical contamination and accumulation of radionuclides in the elements of the Barents and White Seas Ecosystem. In: Complex Investigations of Processes, Characteristics and Resources of Russian Seas of North European Basin, pp. 436-459. Apatity. Kola Science Center RAS. (In Russian)

Ivanov, G.I., 2002. Methodology and results of ecological and geochemical studies in the Barents Sea. St.Petersburg, VNIIOkeangeologia, 153p. (In Russian)

Janicki, E., 2001. A brief history of hydrocarbon exploration in the Northwest Territories. A publication of the C.S. Lord Northern Geoscience Centre, Northwest Territories Geoscience Office.

Jensen, V.B., N. Hansen and P. Johansen, 1977. Hydrocarbons in marine organisms and sediments off West Greenland. Report for the Ministry of Greenland. (In Danish)

Johansen, P., V.B. Jensen and A. Büchert, 1977. Hydrocarbons in marine organisms and sediments off West Greenland. International Council for the Exploration of the Sea. Fisheries Improvement Committee. E:21. 23p.

Johnsen, A.R., L.Y. Wick and H. Harms, 2005. Principles of microbial PAH-degradation in soil. Environmental Pollution, 133(1):71-84.

Jonkers, G., F.A. Hartog, W.A.I. Knaepen and P.F.J. Lancee, 1997. Characterization of NORM in the oil and gas production (E&P) industry. I. Proc. International Symposium on Radiological Problems with Natural Radioactivity in the Non-nuclear Industry, pp. 23-47. Amsterdam, September 8-10, 1997.

Jørgensen, E., R.Velvin and B. Killie, 2000. Miljøgifter i marine sediment og organismer i havneområdene ved Harstad, Tromsø, Hammerfest og Honningsvåg 1997-98. (Contaminants in marine sediments and organisms in harbor areas at Harstad, Tromsø, Hammerfest and Honningsvår 1997-98). Akvaplan-NIVA Report no 412.99.988. 123p.

Kaplan, I.R., 1974. Natural Gases in Marine Sediments. Plenum Press.

KAVIK-AXYS, 2004. Devon Canada Corporation Beaufort Sea Exploration Drilling Program Application. Chemical Oceanography Baseline Study. 116p.

Killie, B., D. Matishov, S. Dahle and J. dos Santos, 1997. The contamination state of sediments in the Kola Fjord, Russia. Akvaplan-niva report no.414.96.894. 16p.

Kireeva, L.I., T.N. Shalyapina and N.F. Plotitsyna, 2002. Aliphatic and polycyclic aromatic hydrocarbons in commercial fishes of the Barents Sea. In: Ecology of Northern Territories of Russia: Problems, prognosis of situations, routes of development, solutions pp. 404-408. (In Russian)

Klungsøyr, J. and S. Johnsen, 1997. Oil hydrocarbons in fish from Norwegian waters 1993-95. Fisken og Havet No. 17. 52p.

Klungsøyr, J., G. Tveit and K. Westrheim, 2001. Condition monitoring in the water column 2000-2001: Oil hydrocarbons in haddock (*Melanogrammus aeglefinus*). Technical Report, Institute of Marine Research, Bergen, Norway. 45p.

Klyujkov, E., 1997. Report of Russian State Hydrometerological Institute (RSHMI) on the studies in Guba Palkina and Kandalaksha Bay of the White Sea in 1997. RSHMI, St. Petersburg. 118p. (In Russian)

Korshenko, A., I. Matveichuk, T. Plotnikova and V. Luchkov, 2003. Marine waters quality on hydrochemical parameters. Annual Report 2002. St. Petersburg, Hydrometeoizdat. (In Russian)

Korshenko, A., I. Matveichuk, T. Plotnikova and V.Luchkov, 2005. Marine water pollution. Annual Report 2003. St. Petersburg, Hydrometeoizdat. (In Russian)

Kostiuk. L.W., M.R. Johnson and G. P Thomas, 2004. University of Alberta Flare Research Project. Final Report November 1996-September 2004. University of Alberta Department of Mechanical Engineering, 254p.

Kotkin, N.Ya., A.P. Kotsepalov and P.V. Kozlov, 1997. Ecological situation and main work trend on environmental protection in the Nenets Autonomous Okrug. Report to Second International Conference: Arctic Town and Environment. Naryan-Mar. 22p.

Leffingwell, E. de K., 1919. The Canning River region, northern Alaska. U.S. Geological Survey Profession Paper 109, 251p.

Leirvik, F., 2005. Weathering properties of the *Linerle* crude. SINTEF report 2005 No: STF80MK F05209, Trondheim, Norway.

Levy, E.M., 1978. Visual and chemical evidence for a natural seep at Scott Inlet, Baffin Island, District of Franklin. Geological Survey of Canada, pp. 21-26. Current Research Paper: 78-1B.

Levy, E.M. and M. Ehrhardt, 1981. Natural seepage of petroleum at Buchan Gulf, Baffin Island. Marine Chemistry, 10:355-364.

Lindgren, C. and E. Lindblom, 2004. Short-term effects of accidental oil pollution in waters of the Nordic countries. A report for the Nordic Council of Ministers by IVL Swedish Environmental Research Institute Ltd., Stockholm.

Lindquist, S.J., 1999. The Timan-Pechora Basin Province of Northwest Arctic Russia: Domanik – Paleozoic Total Petroleum System. U.S. Department of the Interior, U.S. Geological Survey Open-File Report 99-50-G On-Line Edition.

Lobchenko, E.E., 1997. Annual report on quality of surface water in the Russian Federation in 1996. St. Petersburg, Hydrometeoizdat. 737p. (In Russian)

Lobscheid, A.B. and T.E. McKone, 2004. Constraining uncertainties about the sources and magnitude of polycyclic aromatic hydrocarbon (PAH) levels in ambient air: the State of Minnesota as a case study. Atmospheric Environment, 33:5501-5515.

Lukoil, 2004. Lukoil Analyst Databook 2004. Moscow, Russia. 64p. http://www.lukoil.com/materials/doc/LUKOIL-DataBook-Eng%20(2004).pdf

Macdonald, R.W., L.A. Barrie, T.F. Bidleman, M.L. Diamond, D.J. Gregor, R.G. Semkin, W.M.J. Strachan, Y.F. Li, F. Wania, M. Alaee, L.B. Alexeeva, S.M. Backus, R. Bailey, J.M. Bewers, C. Gobeil, C.J. Halsall, T. Harner, J.T. Hoff, L.M.M. Jantunen, W.L. Lockhart, D. Mackay, D.C.G. Muir, J. Pudykiewicz, K.J. Reimer, J.N. Smith, G.A. Stern, W.H Schroeder, R. Wagemann and M.B.Yunker, 2000. Contaminants in the Canadian Arctic: Five years of progress in understanding sources, occurrence and pathways. Science of the Total Environment, 254:93-234.

Mackay, D., 1978. The fate and behaviour of oil in cold climates. Unpublished report to Environment Canada. 124p.

Mackay, D., W. Stiver and P.A. Tebeau, 1983. Testing of crude oils and petroleum products for environmental purposes. In: Proc. 1983 Oil Spill Conference pp. 331-337. Washington, D.C.

Makarova, M.Yu., 2004. Application of biological product "Universal" in rehabilitation of oil polluted areas. Proc. 3rd Scientific Practical Conference on Ecological Work in Oil and Gas Fields of the Timano-Pechora Province. Conditions and Prospects, pp. 229-232. Ukhta September, 6-9, 2004.

Margesin, R. and F. Schinner, 1999. Biological decontamination of oil spills in cold environments. Journal of Chemical Technology and Biotechnology, 74:381-389.

Marsili, L., A. Caruso, M.C. Fossi, M. Zanardelli, E. Polikti and S. Focardi, 2001. Polycyclic aromatic hydrocarbons (PAHs) in subcutaneous biopsies of Mediterranean cetaceans. Chemosphere, 44:147-154.

Matishov, G.G. and B.A. Nikitin, 1997. Nauchno-metodicheskie podhody k ocenke vozdejstvija gazoneftedobychi na jekosistemy morej Arktiki (na primere Shtokmanovskogo proekta). (Scientific and methodological approaches to an evaluation of the influence of oil and gas exploration on Arctic seas ecosystems [Shtokmann project case study]). Apatity, 393p. (In Russian)

Matishov, D.G., G.V. Iljin, N.F. Plotitsyna, A.F. Kochkurkin and G.V. Stepakhno, 1998. Report of Murmansk Marine Biological Institute. Estimation of pollution of commercial objects of the Barents Sea, Murmansk. (In Russian)

Mech 435 Applied Combustion Home Page. Course Notes 7. IC Engine Exhaust Emissions.

Miller, D.J., T.G. Payne and G. Gryc, 1959. Geology of Possible Petroleum Provinces in Alaska. 131p., 4 sheets, scale 1:2,500,000.

Milliman, J.D. and R.H. Meade, 1983. World-wide delivery of river sediment to the oceans. Journal of Geology, 91:1-21.

Ministry of Natural Resources of the Russian Federation, 2005. A report on the condition of the environment in the federal okrugs of Russia. (In Russian). Urals Federal Okrug, http://www.mnr.gov.ru/part/?act=more&id=98&pid=222; North-Western Federal Okrug, http://www.mnr.gov.ru/part/?act=more&id=101&pid=222

Miskevich, I.V., 1996. Hydrochemical regime of the Pechora Sea. Proc. Conference Dedicated to 300th Anniversary of the Russian fleet, Arkhangelsk, pp. 26-29. (In Russian)

MMS, 2003. Task 8 of the Arctic Nearshore Impact Monitoring in the Development Area (AMIMIDA) Project. Baseline characterization of anthropogenic contaminants in biota associated with the Alaska OCS Liberty and Northstar oil and gas production unit in the nearshore Beaufort Sea. Final Report prepared by Applied Marine Sciences, Inc., under Contract No. 01-99-CT-30998, to 10908 for the Minerals Management Service, Alaska OCS Office, Anchorage.

MMS, 2005. Task 2 of the Arctic Nearshore Impact Monitoring in the Development Area (ANIMIDA) Project. Hydrocarbon and metal characterization of sediments, bivalves and amphipods in the ANIMIDA Study Area. Final Report submitted to the Minerals Management Service Alaska Outer Continental Shelf Region by EXPONENT Ltd.

MMS, 2006. Draft Environmental Impact Statement for 5-year leasing plan (2007–2012) for the U.S. Outer Continental Shelf. U.S. Department of the Interior, Minerals Management Service.

Mohn, W.W. and G.R. Stewart, 2000. Limiting factors for hydrocarbon biodegradation at low temperatures in Arctic soils. Soil Biology and Biochemistry, 32:1161-1172.

Mohn, W.W., C.Z. Radziminski, M.-C. Fortin and K.J. Reimer, 2001. On site bioremediation of hydrocarbon-contaminated Arctic tundra soils in inoculated biopiles. Applied Microbiology and Biotechnology, 57:242-247.

Moldestad, Ø.M. and A. Lewis, 2006. Oil weathering at sea. Safety at sea – Demo A, subtask A4. SINTEF report nr. STF80MK A06095, SINTEF, Trondheim.

Monsen, B., G.N. Christensen and S. Rognerud, 2007. Undersøkelse av vann og sedimentkjemi i 32 kystnære innsjøer i Nordland, Troms og Finnmark. (A study of water and sediment chemistry in 32 near-shore lakes in Nordland, Troms and Finnmark). Norwegian Institute for Water Research. (In Norwegian)

Mosbech, A., K.L. Anthonsen, A. Blyth, D. Boertmann, E. Buch, D. Cake, L. Grøndahl, K.Q. Hansen, H. Kapel, S. Nielsen, N. Nielsen, F. Von Platen, S. Potter and M. Rasch, 2000. Environmental Oil Spill Sensitivity Atlas for the West Greenland Coastal Zone. Ministry of Environment and Energy, The Danish Energy Agency and Bureau of Minerals and Petroleum, Government of Greenland.

Mosbech, A., A.B. Hansen, G. Asmund, I. Dahllöf, D.G. Petersen and J. Strand, 2007. A chemical and biological study of the impact of a suspected oil seep at the coast of Marraat, Nuussuaq, Greenland. With a summary of other environmental studies of hydrocarbons in Greenland. NERI Technical Report No. 629, 61p.

Mudroch, A., R.J. Allan and S.R. Joshi, 1992. Geochemistry and organic contaminants in the sediments of Great Slave Lake, Northwest territories Canada. Arctic, 45:10-19.

Muljak, V.V. and V.G. Ivanov, 2004. Complex decision making in the zone of ecological disaster in Usinsk area of Republic of Komi. Proc. 3rd Scientific Practical Conference on Ecological Work in Oil and Gas Fields of the Timano-Pechora Province. Conditions and Prospects, p. 24-29. Ukhta, September, 6-9 2004.

Naidu, A.S., J.J. Kelley, J.J. Goering and M.I. Venkatesan, 2001. Historical Changes in Trace Metals and Hydrocarbons in the Inner Shelf Sediments, Beaufort Sea: Prior and Subsequent to Petroleum-Related Industrial Developments. OCS Study, MMS 2001-061. Anchorage, Alaska.

NAS, 1985. Oil in the Sea. Inputs, Fates and Effects. National Academies of Sciences Press.

NAS, 2003. Oil in the Sea III: Inputs, Fates and Effects. National Academies of Sciences Press.

Nation, L., 2004. Greenland potential to be gauged - Elephants could be hiding in Arctic. American Association of Petroleum Geologists, AAPG Explorer, November 2004 Issue.

Neff, J.M., 1979. Polycyclic Aromatic Hydrocarbons in the Aquatic Environment. Applied Science Publishers. 262p.

Nemirovskaya, I., 2004. Hydrocarbons in the Ocean. (Snow–Ice–Particulate matter–Bottom sediments). Nauchnyi Mir. 328p. (In Russian)

Nemirovskaya, I. A. and A.N. Novigatskii, 2003. Hydrocarbons in the snow and ice cover and waters of the Arctic Ocean. Geochemistry International, 41:585-594.

NRPA, 2005. Natural Radioactivity in Produced Water from the Norwegian Oil and Gas Industry in 2003. Norwegian Radiation Protection Authority. Strålevern Rapport 2005:2, 51p.

OLF, 2006. RKU – North Sea: Updated report on the regional consequences of petroleum activities in the North Sea. Oil Industry Association, 288p. (In Norwegian)

Page, D.S., P.D. Boehm, G.S. Douglas, A.E. Bence, W.A. Burns and P.J. Mankiewicz, 1999. Pyrogenic polycyclic aromatic hydrocarbons in sediments record past human activity: a case study in Prince William Sound, Alaska. Marine Pollution Bulletin, 38:247-260.

Page, D.S., P.D. Boehm, W.A. Stubblefield, K.R. Parker, E.S. Gilfillan, J.M. Neff and A.W. Maki, 2002. Hydrocarbon composition and toxicity of sediments following the *Exxon Valdez* oil spill in Prince William Sound, Alaska, USA. Environmental Toxicology and Chemistry, 21:1438-1450.

Panday, S. and M.Y. Corapcioglu, 1994. Theory of phase-separate multicomponent contaminant transport in frozen soils. Journal of Contaminant Hydrology, 16:235-269.

Patin, S. 1999. Environmental Impact of the Offshore Oil and Gas Industry. EcoMonitor Publishing. 425p.

Paull, C.K., W.U. Ussler III, S.R. Dallimore, S.M. Blasco, T.D. Lorenson, H. Melling, B. Medioli, F.M. Nixon and F.A McLaughlin, 2007. Origin of pingo-like features on the Beaufort Sea shelf and their possible relationship to decomposing methane gas hydrates. Geophysical Research Letters, 34, L01603, doi:101029/2006GL027977.

Pelyushenskii, V., 2005. A flare over the Pechora. State resource management magazine (http://www.gosresurs.ru), Issue 1, July 2005. (In Russian)

Petrova, V., 1990. Geochemistry of PAH of the World Ocean polar zones sediments. In: Organic matter of bottom sediments of polar zones of the World Ocean, pp.70-129. (In Russian)

Petrova, V., 2001. Geochemistry of polycyclic arenes along river-sea section. Geochemistry International, 39:1015-1022.

Petrova, V.I., G.I. Batova and M.A. Galishev, 2000. Correlation diagnostics of hydrocarbon anomalies in bottom sediments of the Arctic shelf. Geochemistry International, 38(3):265-273.

Plotitsyna, N.F., 1998. Content of pollutants in harp seals (*Phoca groenlandica*) of the White Sea population. International Council for the Exploration of the Sea. CM 1998/V:2. Theme Session (V) on Recovery and Protection of Marine Habitats and Ecosystems from natural and Anthropogenic Impacts, pp. 1-12.

Plotitsyna, N.F., L.I. Kireeva, T.N. Shaljapina, T.A. Zimovejskova and T.A. Golubeva, 2002. Environmental contamination of the Barents Sea. In: Proc. International conference on Oil and Gas of the Arctic Shelf. Murmansk 13-15 November 2002. CD (In Russian)

Popov, A.N., G.N. Ertsev, P.I. Konkin and M.J.U. Markarova, 2004. Ecological efficiency of regenerative work in oil-polluted zones in Usinsk area of Republic of Komi. Proc. 3rd Scientific Practical Conference on Ecological Work in Oil and Gas Fields of the Timano-Pechora Province. Conditions and Prospects, pp. 30-42. Ukhta 6-9 September 2004.

Pozzoli, L., S. Gilardoni, M.G. Perrone, G. De Gennaro, M. De Rienzo and D. Vione, 2004. Polycyclic aromatic hydrocarbons in the atmosphere: monitoring, sources, sinks and fate. I: Monitoring and sources. Journal of Analytical and Environmental Chemistry - Annali di Chimica, 94:17-32.

Prevedouros, K., E. Brorström-Lundén, C.J. Halsall, K.C. Jones, R.G.M. Lee and A.J. Sweetman, 2004. Seasonal and long-term trends in atmospheric PAH concentrations: evidence and implications. Environmental Pollution, 128:17-27.

RCMA, 1995. Data on petroleum hydrocarbons. Unpublished manuscripts. Regional Centre, Monitoring of the Arctic (RCMA) of Roshydromet.

RCMA, 2005. Materials to Chapter 5 "Concentrations and fate of petroleum hydrocarbons and polycyclic aromatic hydrocarbons and other relevant contaminants in the Arctic" of the AMAP report on the influence of activities in the area of oil and gas production in the Arctic. (Generalized data on the concentrations of total petroleum hydrocarbons and polycyclic aromatic hydrocarbons in the Arctic environmental components for the period 1999 to 2004. Regional Centre for Monitoring of the Arctic (RCMA) of Roshydromet.

Reeh, N., C. Mayer, H. Miller, H.H. Thomsen and A. Weidick, 1999. Present and past climate controls on fjord glaciations in Greenland: Implications for IRD-deposition in the sea. Geophysical Research Letters, 26(8):1039-1042.

Rice, S.D., M.G. Carls, R.A. Heintz and J.W. Short, 2003. Hydrocarbon composition and toxicity of sediments following the *Exxon Valdez* oil spill in Prince William Sound, Alaska, USA. Comment. Environmental Toxicology and Chemistry, 22:2539-2540.

Riget, F., J. Christensen and P. Johansen, 2003. AMAP Greenland and the Faroe Islands 1997-2001. Vol. 2: The Environment of Greenland. Ministry of the Environment.

Robertson, A., 1998. Petroleum hydrocarbons. In: AMAP Assessment Report: Arctic Pollution Issues, pp. 661-716. Arctic Monitoring and Assessment Programme (AMAP), Oslo, Norway.

Rose, N.L., C.L. Rose, J.F. Boyle and P.G. Appleby, 2004. Lake-sediment evidence for local and remote sources of atmospherically deposited pollutants on Svalbard. Journal of Paleolimnology, 31:499-513.

Samanta, S.K., O.V. Singh and R.K. Jain, 2002. Polycyclic aromatic hydrocarbons: environmental pollution and bioremediation. Trends in Biotechnology, 20:243-248.

Sanin, P.I., 1976. Uglevodorody nefti. (Oil hydrocarbons). Uspekhi Khimii, 55(8):1361-1394. (In Russian)

Savinov, V.M., T.N. Savinova, J. Carroll, G.G. Matishov, S. Dahle and K. Næs, 2000. Polycyclic aromatic hydrocarbons (PAHs) in sediments of the White Sea, Russia. Marine Pollution Bulletin, 40:807-818.

Savinov, V.M., T.N. Savinova, G.G. Matishov, S. Dahle and K. Næs, 2003. Polycyclic aromatic hydrocarbons (PAHs) and organochlorines (OCs) in bottom sediments of the Guba Pechenga, Barents Sea, Russia. Science of the Total Environment, 306:39-56.

Savinov, V., S. Dahle, N. Plotitsyna, G. Chernik and S. Kalmykov, 2006. Analyses of oil contamination in the Kara Sea and Barents Sea sediments, 2006. Akvaplan-niva report APN-433.3158. 44p.

Savinova, T., V. Savinov, G. Christensen, N. Kashulin, I. Koroleva, D. Morozov, A. Konoplev, L. Alexeeva and D. Samsonov, 2006a. POP levels in freshwater environment in the joint Russian-Norwegian border area and effect on fish. Proceedings from SETAC, North America 27th Annual Meeting, 5-9 November 2006. Montreal, Canada.

Savinova, T., V. Savinov, L. Alexeeva, G. Chernik, D. Matishov, O. Mokrotovarova, N. Plotitsnya, D. Samsonov, A. Zhilin and M. Zueva, 2006b. Monitoring of hazardous substances in the coastal areas of the Barents Sea: harmonisation with OSPAR's Joint Monitoring and Assessment Programme (JAMP). Akvaplan-niva report, APN-414.3723:1-32.

Savinova, T., V. Savinov, N. Green, H. Hovind, O. Korneev, S. Marasaev, E. Kaminsky, L. Alexeeva, A. Kochetkov, A. Konoplev, E. Pasynkova, D. Samsonov, K. Galaktionov and V. Markov, 2007. Monitoring of hazardous substrances in the coastal areas of the White Sea: harmonisation with OSPAR's Joint Monitoring and Assessment Programme (JAMP). Akvaplan-niva report, APN-414.3723.2:1-74.

Schindler, J.F., 1983. The Second Exploration, 1975 – 1982 National Petroleum reserve in Alaska (Formerly Naval Petroleum Reserve No. 4).

Schrader, F.C., 1904. A reconnaissance in northern Alaska across the Rocky Mountains, along Koyukuk, John, Anaktuvuk, and Colville Rivers and the Arctic coast to Cape Lisburne, in 1901. U.S. Geological Survey Professional Paper 20, 139p.

Shaw, D.G., D. J. McIntosh and E.R. Smith, 1979. Arene and alkane hydrocarbons in nearshore Beaufort Sea sediments. Estuarine and Coastal Marine Science, 9:435-449.

Skotvold, T., E. Wartena and S. Rognerud, 1997. Heavy metals and persistant organic pollutants in sediments and fish from lakes in Northern and Arctic regions of Norway. Statlig program for forurensningsovervåking SFT. Rapport 688/97. APN-514.660.1

Sørstrøm, S.E., 1989. Full-scale experimental oil spill at Haltenbanken, Norway. Oceanographic Company of Norway (OCEANOR). Report OCN89054. Trondheim, Norway.

Speight, J.G., 1999. The Chemistry and Technology of Petroleum. Marcel Dekker.

Stein, R. and R.W. Macdonald, 2004. The Organic Carbon Cycle in the Arctic Ocean. Springer-Verlag. 263p.

Steinhauer, S.S. and P.D. Boehm, 1992. The composition and distribution of saturated and aromatic hydrocarbons in nearshore sediments, river sediments and coastal peat of the Alaskan Beaufort Sea: Implications for detecting anthropogenic hydrocarbon inputs. Marine Environmental Research, 33:223-253.

Stern, 2003. Parent and alkylated PAH in fish from the Husky Lakes, NWT. Unpublished data courtesy of G. Stern, Department of Fisheries and Oceans, Canada (Freshwater Institute), Winnipeg, Manitoba.

Surgutneftegas, 2006. Annual Report 2005. Open Joint Stock Company *Surgutneftegas* ul. Kukuyevitskogo, 1, Surgut, Khanty_Mansiysky Autonomous Okrug – Yugra, Tyumenskaya Oblast, Russian Federation, 628415.
http://www.surgutneftegas.ru/pictures/year_2005en.pdf

Taskaev, A.I. and M. Ju. Makarova, 2002. Expediency of biological parameters in estimating condition of oil-polluted and rehabilitated lands. Proc. 2nd Scientific Practical Conference on Ecological Work in Oil and Gas Fields of the Timano-Pechora Province. Conditions and Prospects, pp. 71-76. Usinsk, September 9-13 2002.

Taskaev, A.I., J.P. Shubin, A.N. Popov, O.A. Loskutova, E.B. Fefilova and A.B. Zakharov, 2004. Changes in communities of hydrobionts in the basin of the river Kolva after the oil spill disaster on the oil pipeline. Proc. 3rd Scientific Practical Conference on Ecological Work in Oil and Gas Fields of the Timano-Pechora Province. Conditions and Prospects, pp. 215-228. Ukhta, September 6-9 2004.

Thomas, D.J., 1979. Seabed releases of gas around the Tingmiark K-91 wellsite in the southern Beaufort Sea. Dome Petroleum Ltd. Calgary, Alberta.

Thomas, D.J. and R.W. Macdonald, 2005. A petroleum hydrocarbon budget for the Arctic. Presentation at the AMAP International Symposium on Oil and Gas Activities in the Arctic, pp. xxxiv. St. Petersburg, 13-15 September 2005.

Thomas, D.J., R.W. Macdonald and A.B. Cornford, 1983. A comparison of natural and petroleum resource development-related fluxes of metals, hydrocarbons, and nitrogen, phosphorus and silicon to the Beaufort Sea. Canadian Technical Report of Hydrography and Ocean Sciences 30: 48p.

Thomasson, H., 1991. Glaciofluvial sediment transport and erosion. Arctic hydrology present and future tasks. Norwegian National Committee for Hydrology. Report No. 23, 27-36.

Thorpe, S.A., 1995. Vertical dispersion of oil droplets in strong winds – the *Braer* Spill. Marine Pollution Bulletin, 30(1):756-758.

Tolkachev, V.F., 2000. Road to Oil. Arkhangelsk. 608p.

Tsyban, A., 1999. Ecological investigations in the Russian Arctic seas: results and perspectives. Aquatic Conservation: Marine and Freshwater Ecosystems, 9:503-508.

US Department of Health, 1999. Fish tissue exposure investigation. Ashland, Wisconsin.

USDOD, 1981. Installation restoration program records search for Alaska DEW line stations dated 10/1981 by CH2M Hill. Report to U.S. Air Force Engineering and Services Center Directorate of Environmental Planning, Tyndall Air Force base, Florida, October, 1981 U.S. Department of Defense 181p.

USEPA, 1983. Flare Efficiency Study. EPA-600/2-83-052. U.S. Environmental Protection Agency.

USEPA, 1998. Locating and estimating air emissions from sources of polycyclic aromatic hydrocarbons. EPA – 454/R-98-014. Office of Air Quality Planning and Standards, U.S. Environmental Protection Agency, 450p.

USEPA, 2003. Authorization to Discharge under the National Pollutant Discharge Elimination System (NPDES) for Facilities Related to Oil and Gas Extraction. Permit No. AKG-33-0000. U.S. Environmental Protection Agency.

USEPA, 2005. Final Ocean Discharge Criteria Evaluation of the Arctic. NPDES General Permit for Oil and Gas Exploration. Permit No. AKG280000. U.S. Environmental Protection Agency.

USEPA, 2006. Authorization to Discharge under the National Pollutant Discharge Elimination System (NPDES) for oil and gas exploration facilities on the outer continental shelf and contiguous state waters. Permit No. AKG280000. U.S. Environmental Protection Agency

USGS, 1991. Environmental status of 28 oil and gas exploration areas of operation in the National Petroleum Reserve-Alaska. USGS Open File Report 91-458. U.S. Geological Survey.

USGS, 2003. Water and sediment quality in the Yukon River Basin, Alaska during Water Year 2001. Open-File Report 03-427, Department of the Interior, U.S. Geological Survey. 126p.

Venkatesan, M.I., 1988. Occurrence and possible sources of perylene in marine sediments – a review. Marine Chemistry, 25:1-17.

Vorobyev, Yu., V. Akimov and Yu. Sokolov, 2005. Prevention and liquidation of emergency spills of oil and oil products. Octavo publishing, 54p. (In Russian)

Walker, M.I. and A. Lewis, 1995. Emulsification processes at sea – Forties crude oil. Proc. 18th Arctic Marine Oilspill Technical Seminar, pp. 471-493. Environment Canada, Ottawa, Ontario.

Walsh, F.B., 2006. The Kopanoar mud volcano on the Mackenzie Shelf, Beaufort Sea: implications for methane release on arctic shelves. Atlantic Geology, 42:120-121.

Wania, F., R. Semkin, J.T. Hoff and D. Mackay, 1999. Modelling the fate of non-polar organic chemicals during the melting of an Arctic snowpack. Hydrological Processes, 13:2245-2256.

Webber, E.J. and Detterman, R.L., 1947. Preliminary report of the stratigraphy and structure of the area of the Nanushuk River, Alaska. U.S. Geological Survey Open File Report.

Westerholm, R.N., T.E. Alsberg, A.B. Frommelin, M.E. Strandell, U. Rannug, L. Winquist, V. Grigoriadis and K-E. Egeback, 1988. Effect of fuel polyaromatic content on the emissions of polycyclic aromatic hydrocarbons and other mutagenic substances from a gasoline-fueled automobile. Environmental Science and Technology, 22:925-930.

Whittington, C.L. and J.M. Stevens, 1951. Stratigraphy and structure of part of the Carbon Creek and Ketik anticlines, Alaska. U.S. Geological Survey Open File Report.

Whyte, L.G., A. Schultz, J.B. van Beilen, A.P. Luz, V. Pellizari, D. Labbé and C.W. Greer, 2002. Prevalence of alkane monooxygenase genes in Arctic and Antarctic hydrocarbon-contaminated and pristine soils. FEMS Microbiology Ecology, 41:141-150.

Wilcke, W., 2000. Polycyclic aromatic hydrocarbons (PAHs) in soil – a review. Journal of Plant Nutrition and Soil Science, 163:229-248.

Woodward, D.F., E. Snyder-Conn, R.G. Riley and T.R. Garland, 1988. Drilling fluids and the Arctic tundra of Alaska: Assessing contamination of wetlands habitat and the toxicity to aquatic invertebrates and fish. Archives of Environmental Contamination and Toxicity, 17:683-697.

Yertsev, G.N., 2004. Scientific and engineering support of reclamation work of oil-polluted land. Proc. 3rd Scientific Practical Conference on Ecological Work in Oil and Gas Fields of the Timano-Pechora Province. Conditions and Prospects, pp. 75-84. Ukhta, September 6-9, 2004.

Yertsev, G.N, G.M. Barenboim, A.I. Taskaev, 2000. Experience of elimination of emergency oil spills in Usinsk District of the Republic of Komi (Project materials). Syktyvkar. 183p. (In Russian)

Yunker, M.B. and R.W. Macdonald, 1995. Composition and origins of polycyclic aromatic hydrocarbons in the Mackenzie River and on the Beaufort Sea shelf. Arctic, 48:118-129.

Yunker, M.B., R.W. Macdonald, B.R. Fowler, W.J. Cretney, S.R. Dallimore and F.A. McLaughlin, 1991. Geochemistry and fluxes of hydrocarbons to the Beaufort Sea Shelf: A multivariate comparison of fluvial inputs and coastal erosion of peat using principal components analysis. Geochimica et Cosmochimica Acta, 55:255-273.

Yunker, M.B., R.W. Macdonald, W.J. Cretney, B.R. Fowler and F.A. McLaughlin, 1993. Alkane, terpene and polycyclic aromatic hydrocarbon geochemistry of the Mackenzie River and Mackenzie shelf: Riverine contributions to Beaufort Sea coastal sediment. Geochimica and cosmochimica Acta, 57:3041-3061.

Yunker, M.B., L.R. Snowdon, R.W. Macdonald, J.N. Smith, M.G. Fowler, D.N. Skibo, F.A. McLaughlin, A.I. Danyushevskaya, V.I. Petrova and G.I. Ivanov, 1996. Polycyclic aromatic hydrocarbon composition and potential sources for sediment samples from the Beaufort and Barents Seas. Environmental Science and Technology, 30:1310-1320.

Yunker, M.B., S.M. Backus, E. Graf Pannatier, D.S. Jeffries and R.W. Macdonald, 2002. Sources and significance of alkane and PAH hydrocarbons in Canadian Arctic rivers. Estuarine, Coastal and Shelf Science, 55:1-31.

Zagvozdkin, V.K., 2004. OAO Lukoil policies in the environmental protection in XXI century. Proc. 3rd Scientific Practical Conference on Ecological Work in Oil and Gas Fields of the Timano-Pechora Province. Conditions and Prospects, pp. 22-27. Ukhta, September 6-9 2004.

Zakharov, A.B., 2002. Conditions and management of fish resources in conditions of economic development of northern regions. Proc. 2nd Scientific Practical Conference on Ecological Work in Oil and Gas Fields of the Timano-Pechora Province. Conditions and Prospects, pp. 107-108.

Zhilin, A.Yu., N.F. Plotitsyna and L.I. Kireeva, 2003. Polycyclic aromatic hydrocarbons in bottom sediments of the Kola Bay, Barents Sea. In: Sixth International Conference and Exhibition Aquaterra, pp.188-192. St. Petersburg, November 11-12 2003. (In Russian)

Zhilin, A.Yu., M.A. Hovikov and N.F. Plotitsyna, 2004. Polycyclic aromatic hydrocarbons in water and bottom sediments. In: M.S. Shevelev (ed.). Investigations by PINRO in the Spitsbergen Archipelago area pp. 68-75. PINRO Press. (In Russian)

Zhuravel, V.I. and M.I. Mansurov, 2005. Risk vozniknoveniya i organizaciya likvidacii razlivov nefti pri tankernyh perevozkah v Barencevom more. Trudy 7-y mezhdunarodnoy konferencii i vystavki po osvoeniyu resursov nefti i gaza rossiyskoy Arktiki i kontinental'nogo shel'fa SNG RAO/CIS OFFSHORE 2005, pp. 449-454. SPb.13-15 sentyabrya 2005.

Zlotnikova, T.V., V.V. Tetelmin, Yu.V. Utkin, G.I. Berdov, V.M. Kostyukin, A.N. Greshnevikov and G.M. Sadchikov, 1999. Recommendations of Parliamentary Hearings on Environmental Problems Related to Construction and Operation of Trunk Oil and Gas Pipelines. State Duma of the Federal Assembly of the Russian Federation (1999/06/08, hall #830, 10.00-18.00). State Duma Press Release, 10 September 1999.

Chapter 5
Effects of Oil and Gas Activity on the Environment and Human Health

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5.1. Introduction

The exploration, development, and production of oil and gas fields are capable of causing significant changes in the Arctic environment; many examples show that they have already done so. From the initial exploration and seismic phase to the extraction and field operations and the transportation of the product, oil and gas activities involve the construction of dedicated infrastructure that can change the quality of the surrounding environment and modify habitat for terrestrial wildlife and marine life. In the Arctic, these activities generally take place in areas where there is little other development because of the remoteness of the sites onshore and offshore. Changes to the environment due to accidents (spills, blowouts) can be significant locally and can last for long periods (decades). Changes to the environment can be direct or indirect, and can range from physical and chemical changes to biological changes. The reliance of indigenous peoples in the Arctic on fish and wildlife as part of a traditional diet and lifestyle places a responsibility on industry and regulatory agencies to ensure that there are no significant changes to the abundance or quality of plants and animals used as human food. In some areas of the Arctic, commercial fisheries and aquaculture may be seriously impacted by oil and gas operations when chemicals associated with the industry are released. The potential impacts of oil and gas exploration, extraction, and transportation activities on the health of northern people are also a concern.

It is clear from the magnitude of the activities associated with oil and gas development that the environmental impacts can be considerable. New methods of high resolution three-dimensional mapping of oil and gas fields require a higher density of track lines on tundra and taiga where the soils and vegetation are sensitive to scarring. Resource developers have made significant improvements to seismic field programs to reduce damage to the ecosystem. For example, in recognition of known effects on the Arctic environment, operators in some areas restrict activity to the winter season in order to reduce damage to sensitive tundra, they use drill platforms with smaller footprints and less gravel to reduce habitat loss, they run seismic tracks by helicopter, and they avoid seismic exploration during times of the year when local wildlife populations may be undergoing critical processes like nesting, moulting, migration, or calving. Changes to reduce the effects of field programs occur as the results of research projects are incorporated into field activities or as regulators require new methods. However, the productivity, and hence recovery, of Arctic terrestrial systems is low, and scarring from earlier practices is still evident in areas of the Arctic, and the geographic extent of the application of improved practices is not well documented.

This chapter considers the impacts of oil and gas development according to three broad disciplines: aquatic impacts (marine and freshwater), terrestrial impacts, and human health impacts. For the aquatic environment, the chapter considers the chemical compounds involved (mostly hydrocarbons), their biological uptake and clearance, acute and chronic toxicity (largely based on laboratory studies), effects of exposure on the quality of fishery products, mechanisms of toxic actions, sub-lethal effects, and effects on natural populations of experimental and accidental spills. Attention is also given to issues associated with noise from ships and seismic exploration at sea. For the terrestrial environment, consideration is given to chemical exposure and the effects of spills on tundra and taiga and also to the effects of physical disturbance and recovery. The section on human health examines the sources and routes of exposure of non-occupationally exposed populations in the Arctic to chemicals associated with oil and gas activities in the Arctic. It also addresses, for human populations, the uptake, metabolism and toxic effects of the major substances (and their transformation products) associated with oil and gas activities.

This chapter extends the assessment conducted by AMAP in 1998 (AMAP, 1998) by including recent research and aspects not covered in the 1998 assessment and by better characterizing the short- and long-term effects of oil and gas development in the Arctic. All aspects of oil and gas activities including exploration, production, storage, and transportation of oil and gas products are examined. The assessment is based on scientific information developed largely over the past 40 years from laboratory experiments, ecosystem experiments, and accidental and experimental spills of petroleum products. The chapter considers some effects of physical disturbance of the environment and effects on populations near oil developments and oil spills. Although the intent for this review was to assess the individual and cumulative effects from oil and gas activities in the Arctic, some of these exposures and effects associated with minor components of oil and gas have been addressed by other AMAP assessments, for example, radioactivity (AMAP, 2004a), heavy metals (AMAP, 2005), and persistent organic pollutants (AMAP, 1998, 2004b). Hence this assessment will address in detail only those impacts on the Arctic not addressed by other AMAP assessments.

As in other AMAP assessments, consideration of the unique physical, chemical, and biological features of the Arctic plays a central role in assessing the potential impacts from oil and gas activities. The most obvious differences between Arctic and temperate conditions include low temperature, ice cover and terrestrial permafrost, extreme seasonal variation in sunlight, and relatively low levels of

nutrients for microorganisms. These physical properties shape the biological and ecological characteristics of the Arctic biological community. For example, the seasonal variation in temperature and light results in short, intense summers during which large numbers of resident and migratory Arctic animals must complete critical stages of their life cycles (e.g., nesting, birthing, rearing of young, moulting). Other species undergo hibernation or torpor for several months to overwinter, or undertake long migrations to nesting or calving areas. Oil and gas activities that interfere with, or interrupt these activities can impact on these populations and can affect the availability and quality of subsistence foods consumed by indigenous Arctic populations. Low ambient temperatures affect the physical, chemical, and toxicological characteristics of oil. However, surprisingly little effort appears to have been given to exploring the effects of temperature on the toxicity of hydrocarbons, or other compounds used in the industry.

The literature on the effects of oil and natural gas, their products and many of their individual chemical components on plants, animals and humans throughout the world is extensive. A relatively small proportion of this literature deals with the Arctic, and what is available is often associated with baseline data for a specific initiative, such as the development of an oil or gas field, the approval of a pipeline, or the effects of a spill. This results in a flurry of research of uneven quality by operators and regulators in an effort to understand the environmental consequences of the project. For example, much of the initial research on the toxicology of crude oils and petroleum in aquatic systems was conducted during the 1970s and 1980s because of large oil spills in the marine environment and the diffuse releases of petroleum-based products from cities and industries to rivers and lakes. At the same time, interest in the development of oil and gas reserves in the North American Arctic led to studies of oil on land (e.g., Jenkins et al., 1978) and in the sea (Sergy and Blackall, 1987). Relatively little basic toxicological research on the effects of oil and gas components, using modern analytical methods and more sensitive dose-response protocols, has been conducted for aquatic, avian, and wildlife species since that time. In addition, this assessment frequently relies on experimental work that has been done with temperate organisms or with samples of oils in temperate waters. While the toxicological principles established in temperate settings should largely apply to Arctic organisms, the rates of processes and sometimes the species exposed may differ.

For non-occupational human populations in the Arctic there are few health effects studies. As a result, this chapter relies almost exclusively on potential impacts in populations, should exposures occur, and makes use of population impacts of spills and other contamination events reported in other cold sub-Arctic areas.

The resources devoted to research within sub-areas of the Arctic are uneven, driven by public, regulatory, and scientific concerns. For example, in assessing the ecological information available to understand the potential impacts of drilling in the Arctic National Wildlife Refuge, Berger et al. (2001) determined that 44% of the peer-reviewed studies and grey literature on plants and animals on the Alaskan North Slope up to 1998 involved caribou, while only 2% related to moose, and 4% to vegetation. Caribou studies outnumbered the total number of the next five most-studied species (Berger et al., 2001). While the emphasis on caribou is understandable in the northern context, given its role as a keystone species in the terrestrial environment and its cultural, social, and nutritional importance to indigenous peoples, the lack of research on other parts of the biological system makes the integration of knowledge needed to predict the potential impacts of expanding oil and gas activities very difficult. In only one instance, the Alaskan North Slope, has the overall environmental impact of a large, mature oil and gas project in the Arctic been documented and assessed by an independent panel of experts (NAS, 2004).

For the present assessment, effort has been made to review a reasonable proportion of studies within most relevant areas of toxicological, ecological, and human health research. Studies have been selected for review when they offer insights into possible effects within the Arctic, even though those studies may have been conducted outside the Arctic, or with species not native to the Arctic. The environmental issues posed by oil and gas activities are common to many Arctic countries and the application of studies across political and geographic boundaries helps to fill gaps in northern research in general. There are many national programs for conducting research and regulating oil and gas activities, but there is no overall program in place for the circumpolar Arctic, which leads to inconsistencies in research and regulation within the various regions of the Arctic. The desirability of any such international program may be an appropriate subject for further discussion. Possibly, this assessment will be an early step in the development of coordinated circumpolar research to address major gaps in understanding the effects of large-scale oil and gas development.

Production from Arctic wells releases small amounts of oil and production chemicals, but these generally represent known amounts at known locations. The effects of such releases can be monitored even in the presence of confounding factors and reduced through adaptive management. Non-industry sources of hydrocarbons can cause ambiguity in the interpretation of hydrocarbons originating from the oil and gas industry. There are areas of natural seepage such as the Mackenzie River in the western Canadian Arctic (Wilson et al., 1973), the eastern Canadian Arctic (Levy and Erhardt, 1981), and the Pechora River in Arctic Russia (Chapter 4). Chemical profiles of hydrocarbons from petroleum sources can sometimes be discriminated from similar profiles of hydrocarbons originating from other sources, for example, natural plant products and combustion/atmospheric deposition (Yunker and Macdonald, 1995). The Ob and Yenisey rivers contain polycyclic aromatic hydrocarbons (PAHs) from eroded soils and pyrogenic sources in Siberia, providing a significant source of these compounds to the Kara Sea (Fernandes and Sicre, 1999). The impacts of seeps on biota and local hydrocarbon concentrations are unclear. Research and monitoring studies in Prince William Sound, Alaska, have failed to conclusively identify the magnitude and chemical nature of natural sources of hydrocarbons more than a decade after the *Exxon Valdez* oil spill. A scientific debate centres on the question of whether the background PAH signal is from natural oil 'seeps', and whether it has exposed biota in the region to natural hydrocarbons to which the organisms have become adapted. Alternatively, the hydrocarbons may be derived from coal deposits, from which the hydrocarbons are not biologically available (Boehm et al., 2000; Short et al., 1999, 2000).

Partitioning PAHs at a site into likely source components will require ongoing improvements in the analytical description of the compounds and their isomers. To this uncertainty is added the recognition that the Arctic environment is undergoing unprecedented changes due to climate warming (ACIA, 2005). Macdonald et al. (2005) summarised a number of changes in the physical environment of the Arctic that have occurred over the last decade (e.g., changes

in wind fields, length of melt season, changes in precipitation patterns) many of which may influence the transport of persistent organic chemical contaminants, such as PAHs. Changes to the physical environment will result in major changes to the biotic community on land and in water as habitats undergo restructuring and these will also complicate the assessment of future spills or discharges.

5.1.1. General introduction to the sources of oil and petroleum hydrocarbons and related substances relevant to toxicology

5.1.1.1. Types and sources of oil and gas-related compounds

5.1.1.1.1. Petroleum hydrocarbons

As described in Chapter 4, oil is composed of hundreds or thousands of structurally similar compounds, including principally normal and branched alkanes, cycloalkanes, unsaturated aliphatic compounds, aromatic compounds with one to six aromatic ring structures, some heterocyclic compounds containing atoms of nitrogen, oxygen or sulphur, small amounts of other elements such as nickel and vanadium and occasionally small amounts of naturally-occurring radionuclides.

From a toxicological perspective, the PAHs are of particular interest. The association between petroleum and PAH is often ambiguous; PAHs are present in petroleum products but they originate from other sources as well, notably from the incomplete combustion of almost any carbon-based fuel (e.g., forest fires, burning wood or coal). Combustion-related PAHs are often discharged into the air and have become widely dispersed. Flaring practices at the well head are a source of PAHs that may be of concern with respect to human health.

Most of the acute toxicity of oils to aquatic organisms has been attributed to the low-boiling aromatic fraction composed mainly of substituted benzenes and naphthalenes with very small amounts of larger ring structures (see section 5.3.3). These compounds are the most soluble in water and hence are found partially in the water phase after oil and water are mixed. Alkylated phenols with small alkyl substituents are relatively soluble and these too can contribute to acute toxicity. The higher-boiling PAHs are subjects of growing environmental interest because of their persistence in sediment, and their associations with enzyme induction, cellular proliferative diseases, genetic alterations and chronic effects on diverse organisms. The alkylated phenols have attracted interest because some with larger alkyl substituents are responsible for endocrine disruption in many vertebrates. The alkanes are major components of oil but are thought to contribute little to its acute toxicity although a few studies have suggested otherwise.

Anderson et al. (1974) and Winters et al. (1977) have listed the major compounds in the 'water-soluble fraction' (WSF) of several oils: substituted benzene, naphthalenes, phenols and anilines. Studies of the content of the WSF coming from preparations with fresh oils have suggested that the compounds responsible for most of the acute toxicity in the aquatic ecosystem are the low-boiling aromatic compounds typically with one or two aromatic rings. During weathering of oil, most of these volatile compounds are lost and the acute toxicity of the WSF prepared from weathered oils contains proportionately more higher PAH and proportionately less volatile compounds (Carls et al., 1999; Neff et al., 2000). In some oils the toxicity of the WSF is higher than predicted from the combined concentrations of monocyclic aromatic hydrocarbons, PAHs, and phenols, indicating that other components of the WSF or complex interactions may contribute to the toxicity (Neff et al., 2000).

Volatile organic compounds such as benzene, toluene and xylene are also of potential concern to human populations because of their rapid entry into air following a spill and their effects on the nervous system.

5.1.1.1.2. Produced water

Produced water is the largest stream of waste from the production phase of conventional oil and gas wells (see Chapter 4; section 4.2.4). The produced water volume increases over the economic lifetime of a producing field and may be up to 95% of the total volume produced by the end of a field's production history. If disposed of at the surface (to rivers, lakes and onto land areas) produced waters may be of toxicological concern, primarily to aquatic systems. Produced water contains formation water, injection water, and other chemical additives such as hydrate inhibitors, emulsion breakers, flocculants, coagulants, defoaming agents, scale and corrosion inhibitors, bactericides and other substances (Lee et al., 2005). The composition of produced water was described by Tibbetts et al. (1992) and Røe Utvik (1999) and is discussed in Chapter 4, section 4.2.4.

Within the Arctic region, Russia is the only country that still discharges significant volumes of produced water, partly following treatment, to the surface environment onshore (see Chapter 4, section 4.2.4). Norway discharges some treated produced water into the Norwegian Sea from offshore activities. Discharge of produced water is likely to decline as regulatory changes restrict its discharge; in many areas produced waters are re-injected. The maximum permissible oil concentration in treated waters for discharge purposes varies from country to country, and is based on toxicity considerations rather than on the volume discharged. For example, the regulatory restriction for Norwegian waters is a hydrocarbon content of 30 mg/L or less.

5.1.1.1.2.1. Aquatic toxicity of alkylphenols

Alkylphenols are natural components of crude oil (Ioppolo-Armanios et al., 1992; Taylor et al., 1997; Rolfes and Andersson, 2001). As a result of their solubility in water they are found in the aqueous phase after water/oil separation and therefore have been discharged into the sea with the produced water. The alkylphenols are typically found in concentrations of 0.6–10.0 mg/L in produced water. About 80% of the total amount consists of the most water-soluble phenols (phenol and cresol). Of the remaining components, the higher alkylphenols from butyl- to heptylphenols occur in low concentrations of 0.07–237 μg/L (Grahl-Nielsen, 1987; Brendehaug et al., 1992; Røe and Johnsen, 1996; Boitsov et al., 2004; Faksness et al., 2004).

5.1.1.1.3. Drilling fluids and drill cuttings

Drilling creates substantial quantities of drilling wastes composed of rock cuttings, drilling fluid, and a wide range of chemical additives. As described in Chapter 4 (section 4.2.3.1), drilling fluids are referred to as either oil-based drilling muds, synthetic-based drilling muds, or water-based drilling muds, depending on the base fluid used. Due to their toxic properties, oil-based drilling muds have been largely phased-out in favour of water- or synthetic-based

drilling muds and are only used where necessary for geotechnical or drilling-safety reasons. An OSPAR regulation has prohibited discharge of wastes from oil-based types of drilling fluids since 2000 (OSPAR, 2000). Drilling fluids and drill cuttings contaminated with drilling fluids are either discharged (typically to the sea-floor near the drilling platform in the case of offshore drilling operations), injected into a well, or sent to special landfill sites for storage or disposal. Oil-based drilling muds and cuttings with a high hydrocarbon content are treated to 'decontaminate' the material prior to landfill.

5.1.1.1.4. Methanol, gas condensate and natural gas

Russian researchers have devoted much attention to toxicity studies with methanol, 'gas condensate' and 'natural gas' (Kosheleva et al., 1997). During production and transport, methanol is used to prevent the formation of hydrates which are especially dangerous at low temperatures, both in wells and during transportation of gas through pipelines. Areas affected by any spills of this gas mixture or of gas condensate would be expected to be relatively small under open-water or broken-ice conditions because of the high volatility of these components.

5.1.1.2. Environmental weathering of oil and petroleum hydrocarbons

The impact of a spill of petroleum hydrocarbons in either the terrestrial or aquatic environment is critically dependent on the composition and properties of the spilled oil and how these properties change over time. For example, an oil slick in the aquatic environment spreads under the influences of winds and currents and its composition changes (Chapter 4, section 4.4.3.2; Mackay, 1985). For large spills (of over 1000 barrels), persistence of the slick has been shown to be a function of the size of the original spill and temperature. Persistence is greater at lower temperatures.

In addition to changes in the size of a slick, there are changes in the chemical composition of the oil comprising it. These changes, which are described as 'weathering', are difficult to describe quantitatively (e.g., Payne and McNabb, 1984), and occur by several mechanisms, as discussed in Chapter 4 (section 4.4.3). Weathering is important toxicologically because acute toxicity to aquatic organisms is reduced as volatile components are lost to the atmosphere by evaporation (section 4.4.3.3; Harrison et al., 1975).

The receiving environment determines to a large degree which weathering pathways predominate and the rates at which they occur. Mechanisms driven by heat or light or microbial metabolism will be relatively unimportant in the Arctic in winter. The greater persistence of toxic components of oil in cold or ice-covered water is the major reason why an oil spill in the Arctic represents a greater risk than that posed by the same amount of the same oil in temperate waters. The main early weathering pathway is evaporation under open-water conditions but that pathway is slowed by cold temperature and virtually stopped by ice cover. Metabolism of hydrocarbons by indigenous microorganisms from the Beaufort Sea was slower at lower incubation temperatures (Bunch and Harland, 1976). Oil spilled under ice is still subject to dissolution, adsorption to particulate matter, biological uptake and to advection by tides and currents (see Chapter 4, sections 4.4.3.9 and 4.4.3.10).

Following the *Exxon Valdez* oil spill in Prince William Sound, the most abundant aromatic compounds reported were naphthalene and alkylated naphthalenes (Neff and Stubblefield, 1995). These compounds became less dominant over the sampling period of several months. More soluble (and volatile) components (benzene, alkylated benzenes) are removed from the oil by dissolution or by emulsification, the latter depending on weather and sea conditions. Many components of oil are changed by physical means (e.g., light), chemical means (e.g., oxidation) or biological means (e.g., microbial metabolism). Some adsorb to particulate matter, especially to small particles in the 'silt' and 'clay' size ranges (Brown, 2005), and may settle to the bottom or accumulate in living organisms. The result is a changing mixture to which aquatic organisms are exposed for varying periods.

Organisms may be exposed to hydrocarbons floating on the water, dissolved in the water, dispersed as fine or colloidal droplets in the water or associated with particulate matter including sediment and single-celled organisms. They may be exposed through consumption of contaminated food, including, apparently, small droplets of oil (Sturdevant et al., 1996) and by contact. Even after volatile components have evaporated, air-breathing marine mammals may be exposed to them briefly by inhalation of contaminated air near the air-water interface.

5.1.1.3. Solubility

Dissolution of petroleum hydrocarbons in Arctic marine waters is discussed in Chapter 4 (section 4.4.3.8). The solubility of PAHs generally increases slightly with temperature and decreases slightly with salinity (Whitehouse, 1984), with the effect of temperature greater than the effect of salinity. Cold Arctic water should dissolve less hydrocarbon than warmer water and so help to limit exposure of organisms to a dissolved phase. Notwithstanding the solubilities of individual hydrocarbons, Percy and Mullin (1975) noted higher initial concentrations of oil in seawater when mixtures were prepared at 0 °C rather than at 8 °C. It is likely that the mechanics of mixing oil and water control the exposure as much as the true solubilities of component hydrocarbons. The WSF would be expected to contain both dissolved hydrocarbons and fine droplets not truly dissolved. These results suggest that spills in cold Arctic waters, especially seawater, may result in exposure of aquatic organisms to 'concentrations' of oil as great as they might experience in temperate settings in spite of the slightly greater solubility of hydrocarbons in warmer water. The reduced microbiological degradation of oil in Arctic waters extends the potential period of exposure. Oil originating from the *Exxon Valdez* was found in beach sediment 12 years after the initial oiling in 1989 (Short et al., 2004). When oil penetrates below the oxygenated surface layers of sediment it degrades very slowly. Experience following the spill from the *Arrow* in Chedabucto Bay, Nova Scotia, Canada, showed that bunker C oil trapped in sediment persisted for a very long period; Vandermeulen and Gordon (1976) suggested the period could be as long as 150 years.

5.1.1.4. Solar radiation

As discussed in Chapter 4 (section 4.4.3.7), the extreme seasonal variation in solar radiation and temperature at Arctic latitudes predicts seasonal differences in weathering. Some photo-oxidation products are more soluble than the original oil components which may enhance the toxicity of the water phase, for example to the diatom *Skeletonema costatum* and to sea urchin eggs (*Strongylocentrotus droebachiensis* and *S. pallidus*). Opportunities for photochemical processes are obviously present during summer but limited or non-existent in winter (Sydnes et al., 1985).

Solar UV radiation can also cause more extensive 'burns' in human populations when skin sensitization resulting from exposure to fresh oil has occurred.

5.1.1.5. Biological metabolism

Many bacteria not only survive exposure to hydrocarbons but use hydrocarbons as a source of energy. Russian studies have shown that 63% of cultures of 50 genera of *Pseudomonas, Mycobacterium, Bacillus, Arthrobacter, Flarobacterium,* and *Micrococcus* can transform benzo[*a*]pyrene by as much as 20–50% from an initial concentration (0.001 mg/L) during seven days. However, it was noted that accumulation of benzo[*a*]pyrene in the marine environment has been observed despite the proliferation of microflora that can oxidize it (Pfeiphere and Platipira, 1986). Microbial metabolism slows with cold temperature even when adequate resources of oxygen and the nutrients nitrogen and phosphorus are available (Bunch and Harland, 1976; Atlas, 1977; Westlake, 1982). Hydrocarbon-metabolizing bacteria appear to be present in Arctic marine waters but in low abundance (Bunch and Harland, 1976; Arhelger et al., 1977; Atlas et al., 1978).

5.2. General introduction to the toxicology of oil and petroleum hydrocarbons

Research into the toxicology of hydrocarbons has been conducted for over two centuries, since the association between exposure to soot and cancer of the scrotum in chimney sweeps was discovered in mid-seventeenth century England (IARC, 1985). Since about 1960, research from field and laboratory experiments has better defined the toxicity of crude oil, natural gas, and many of their individual components and other industry-related compounds, like alkylated phenols, to a number of plant and animal species and to humans. Advancements in toxicology and the estimation of exposure in natural populations have led to the incorporation of risk assessment into decommissioning and remediation assessments and into the evaluation of the potential environmental and health effects of new projects.

5.2.1. Exposure in non-human species

The pathways of exposure of plants and animals to oil and its components have been delineated by extensive research and generic schemes for the exposure of organisms are shown in Figure 5.1. Mammals are exposed primarily through ingestion of contaminated food and water, inhalation, and by dermal absorption of hydrocarbons through the skin or by preening oiled fur. The liver is a major site of metabolism in mammals, reducing the concentration of PAHs and other hydrocarbons to very low levels in organs and tissues (Albers, 2003), even though elevated levels of hydrocarbons may be present in the diet. By-products of this metabolism are often metabolically active and may be more toxic to the animal than the original compounds. Birds have metabolic ability similar to mammals, but have the additional exposure pathway of transferring oil from the external surface to incubating eggs. Waterborne hydrocarbons are absorbed into fish and aquatic invertebrates through the gills during respiratory exchanges, through the gut by eating droplets of oil or oil-contaminated food, by physical contact and sometimes by inhalation in marine mammals.

Albers (2003) documented the range of biological effects of petroleum or individual PAHs reported in the literature

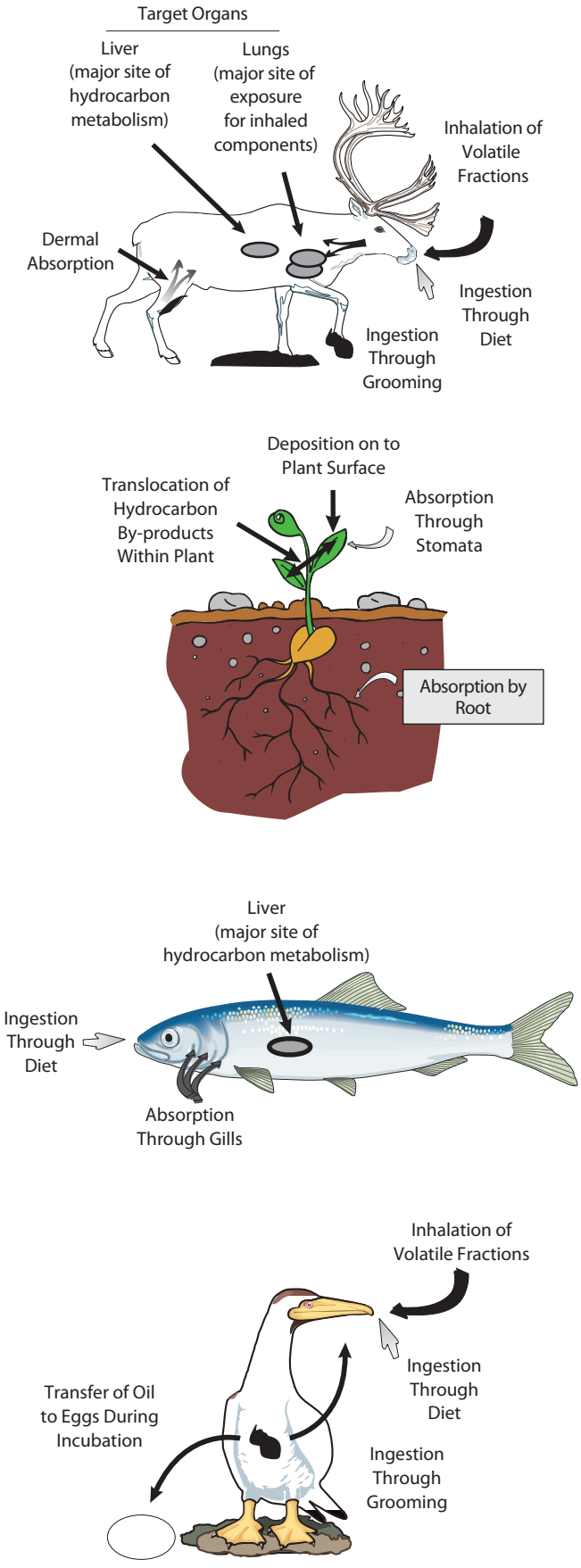

Figure 5.1. Pathways of hydrocarbon exposure and metabolism in mammals, plants, fish and birds. These pathways were established through laboratory and field studies, mostly in temperate regions, and their significance in Arctic regions is largely unknown.

Table 5.1. Summary of effects on non-human species from petroleum or individual PAHs (Albers, 2003).

Effect	Plant or Microbe	Invertebrate	Fish	Reptile or Amphibian	Bird	Mammal
Individual organisms						
Death	X	X	X	X	X	X
Impaired reproduction	X	X	X	X	X	X
Reduced growth and development	X	X	X	X	X	
Altered rate of photosynthesis	X					
Altered DNA	X	X	X	X	X	X
Malformations			X		X	
Tumours and lesions			X	X		X
Cancer			X	X		X
Impaired immune system			X		X	X
Altered endocrine function		X	X		X	
Altered behaviour		X	X	X	X	X
Blood disorders			X	X	X	X
Liver and kidney disorders			X		X	X
Hypothermia					X	X
Inflammation of epithelial tissue				X	X	
Altered respiration or heart rate			X	X		X
Impaired salt gland function				X	X	
Gill hyperplasia			X			
Fin erosion			X			
Groups of organisms						
Local population change	X	X	X		X	X
Altered community structure	X	X	X		X	X
Biomass change	X	X	X			

Columns indicated by 'X' for microalgae, microbes and soil invertebrates under groups of organisms could indicate either a positive (growth) or negative (population reduction) response after hydrocarbon exposure. In general, columns marked under groups for plants, invertebrates, and populations of vertebrates indicate studies showing a decrease in population size.

for individual non-human species and for groups of species, or biological communities (Table 5.1). Death, or diverse sub-lethal effects (e.g., impaired reproduction, altered DNA) have been reported in all non-human species tested. Reduced growth and development have been reported in all groups except mammals. Sub-lethal responses include histological pathologies and abnormalities, altered behaviours, enzymatic activities, membrane functions, and blood disorders, such as red blood cell damage. Under some conditions, some populations of microbes and tolerant benthic fauna have been shown to increase upon exposure to petroleum. Table 5.1 provides a summary of the types of effects that have been noted in field and laboratory exposure studies, but it does not report the type of oil or petroleum product used in the study, or the exposure conditions. In addition to the direct effects of exposure there is the possibility of indirect effects such as destruction of prey or predators but these are extraordinarily difficult to prove in statistically convincing ways.

5.2.2. Exposure in humans

Oil and gas activities can contaminate the air through spills and direct site emissions, can contaminate soil and habitat, can contaminate water used for drinking and bathing and can contaminate food consumed by local populations. Human exposure to oil and gas components and related production substances is primarily through skin contact (oil and contaminated water and soil), by breathing in volatile organic compounds and occasionally fine oil droplets, and through ingestion of contaminated food and water.

Individuals and subgroups in the population may respond differently to similar exposures. Age, health status, reproductive state and length of exposure can all affect responses.

5.2.3. Ecological and health risk assessment

Basic methods for conducting risk assessments in aquatic and terrestrial systems and for humans have been published by virtually all the Arctic countries. The methods follow the general risk assessment methodology of defining the amount of a chemical to which an organism is exposed, the exposure levels at which biological effects are expected to occur, and an estimate of the likelihood that effects might occur in the population in the area of interest. Usually the exposure level is compared to a dose which is considered to be a 'lowest observable effect level' (LOEL) or a 'no observed effect level' (NOEL) from laboratory or field studies and a conclusion is reached whether a biological effect might occur under those conditions. Sometimes it is the 'lowest observed adverse effect level' (LOAEL) or its equivalent NOAEL which is used. Uncertainty, or safety factors are included to account for sensitive species, cross species extrapolation, severity of the finding, length of the animal exposure and observation period and related experimental uncertainties.

For non-human species, the 'risk' calculations are usually conducted with 'valued' species that are defined by consultation with the public, or from a scientific understanding of the ecosystem involved. In the Arctic, valued ecosystem components are often keystone or umbrella species such as caribou/reindeer, moose, seabirds, raptors, whales, commercial fish or invertebrates, or food species consumed in subsistence fishing, and rare, sensitive or threatened species.

For human populations 'risk' calculations may also take specific account of the developing fetus (pregnant women), infants, children, the aged or specific occupational groups. The potential for cancer to result from exposure to a substance is usually treated differently to the approach described above (i.e., NOAEL, LOAELs and uncertainty/safety factors). For substances which are shown to be (or have the potential to be) animal or human carcinogens, a risk calculation is used to establish a benchmark exposure which would lead to less than

one in ten thousand or less than one in one million additional cancers in humans. Jurisdictions then target to keep exposures well below the bench mark exposure.

Ecological risk assessments for oil and gas facilities in the Arctic are rarely published in the primary literature. Any ecological risk assessment requires considerable data and generally these are more readily available for the aquatic and marine systems than for the terrestrial system (see Chapter 4).

5.3. Impacts on aquatic ecosystems

The impact of hydrocarbons on aquatic ecosystems is of great interest immediately after large spills although small but more frequent spills may collectively represent as great a risk. Large spills will probably always defy complete description regardless of how much effort is given to them. The best characterized spill to date is the *Exxon Valdez* spill in 1989 in Prince William Sound, Alaska. Much effort has been spent on describing lethal and sub-lethal toxicological effects on marine organisms exposed to the oil at varying levels and durations in laboratory experiments, and on extended ecological observations of local flora and fauna. The greatest difficulty lies with extrapolating beyond relatively short-term effects on individuals to longer-term effects on populations and communities. Peterson and Holland-Bartels (2002) listed several issues which remain problematic: delayed impacts of initially sub-lethal effects, populations that arise following losses of key individuals critical to social structures, chronic impacts of toxic components that have persisted for a long period, and delayed impacts of habitat degradation and other indirect effects.

Exposure of aquatic animals to spilled hydrocarbons varies with climatic and weather conditions. Pathways by which hydrocarbons are lost include evaporation, microbial metabolism and photolysis. Low temperature, darkness and ice cover all retard the loss or degradation of hydrocarbons and so exposure duration for aquatic animals increases under Arctic winter conditions. Partitioning of PAHs to particulate matter results in contamination of sediments and the PAHs persist there, apparently for decades or longer, resulting in the chronic exposure of benthic organisms and the potential release of hydrocarbons back to the water column. Advection by tides and currents distributes hydrocarbons more widely but also dilutes them. Exposure to other industry-related compounds has been changing due to regulatory actions. For example, oil-based drilling muds have largely been replaced by less toxic alternatives and restrictions on the toxicity of produced water discharges are leading to reduced risk of exposure to these materials.

5.3.1. Uptake, metabolism, and excretion

Aquatic animals take up hydrocarbons from water and some, especially vertebrates, metabolize them to more polar compounds and then excrete them, often through the bile. Laboratory experiments have described the uptake, metabolism, and depuration of individual hydrocarbons by many species and chemical residue studies have described hydrocarbons found in animals and plants exposed following spills or other exposures. The presence of a suite of hydrocarbons or their metabolic by-products in animals is helpful in establishing that exposure has taken place. However, the next step in establishing a linkage between hydrocarbon concentrations in organisms and biological effects on those organisms is not straightforward. Hydrocarbons are metabolized, especially by vertebrates, to a series of oxidized products and hence the measurement of parent hydrocarbons by themselves does not describe the exposure adequately. Measurements of metabolic by-products, in addition to the original hydrocarbons, have been helpful (e.g., Krahn et al., 1986).

Following a spill into water where fishing of any type takes place, urgent questions arise concerning the suitability of fishery products for human consumption. Analysis of hydrocarbons in aquatic species has been used as one method for determining the quality of fishery products after a spill and the risks they may pose to human consumers (e.g., Webster et al., 1997; Gilroy, 2000).

Studies have identified low levels of hydrocarbons in marine organisms from the Arctic. For example, a recent report on PAHs in liver of fish from an isolated lake (Ferguson) in Greenland showed levels comparable to those in high-altitude lakes in Europe (Vives et al., 2004). The authors argued that the source of hydrocarbons to these isolated lakes would have been atmospheric deposition. The levels in the fish were low, in the range of 9–44 ng/g for the sum of 13 individual PAH compounds. Given the location of Ferguson Lake, these low levels of hydrocarbon residues are obviously possible without implicating the oil and gas industry. Zobell (1971) reviewed earlier studies showing levels of benzo[*a*]pyrene in the range of 0–230 ng/g (dw) in Arctic crustaceans, although analytical methods available at that time were less specific than those using more recent technologies.

5.3.1.1. Invertebrates and microorganisms

James (1989) reviewed studies of uptake, metabolism, and depuration of hydrocarbons by marine invertebrates – animals that generally have reduced ability to metabolize hydrocarbons relative to vertebrates. Uptake and depuration experiments with hydrocarbons have frequently used compounds with tritium or carbon labels. For example, Figure 5.2 shows the uptake and depuration of ^{14}C during exposure of spot shrimp (*Pandalus platyceros*) to ^{14}C-naphthalene either dissolved in the water or complexed with bovine serum albumin. Naphthalene was accumulated within hours to levels about 100 times those in the water. Uptake was slower when naphthalene was complexed with protein prior to exposure. When transferred to clean water, most of the radioactivity administered as naphthalene was cleared from the shrimp within 36 hours but a small proportion (~5–20%) was refractory and was not cleared after several days. Most of the radioactivity in the shrimp was present as metabolic products rather than as the original naphthalene, indicating that these organisms had some ability to metabolize naphthalene. Studies with other taxa have shown that a small fraction of radioactivity remains in the organisms for at least several weeks (e.g., zooplankton; Harris et al., 1977). More recent work, using greatly improved analytical methods, indicated little metabolic alteration or elimination of a suite of hydrocarbons by the mussel *Mytilus edulis* (Baussant et al., 2001a).

Majewski and Scherer (1985) measured uptake of radioactively labelled benzo[*a*]pyrene and fluorene by the mussel *Mytilus edulis* and the diatom *Phaeodactylum tricornutum* in laboratory cultures of synthetic seawater at 10 °C. Both species accumulated the hydrocarbons from the water quickly with the mussels more efficient than the algae. When the algae were allowed to accumulate benzo[*a*]pyrene and then fed to the mussels, the mussels accumulated very little benzo[*a*]pyrene from them. Direct uptake from water was more important than uptake from

food. Hydrocarbons taken up from the water reach deep, internal organs readily. Palmork and Solbakken (1981) added radiolabelled phenanthrene to seawater and monitored its uptake into several organs of horse mussels (*Modiolus modiolus*). The greatest accumulation was in hepatopancreas followed by gills, genital tissue, and mantle. About half the radioactivity was cleared from all four organs within the first few days but a significant proportion (10–20%) remained in the organisms after 28 days. Hydrocarbons can be taken up by ingestion. Leppanen and Kukkonen (2000a) found that feeding oligochaetes (*Lumbriculus variegates*) accumulated more pyrene and benzo[*a*]pyrene from lake sediment than non-feeding oligochaetes, showing that benzo[*a*]pyrene ingested with sediment was a component of accumulation. The period of contact between lake sediment and hydrocarbons affected the uptake by the worms. Experiments with sediment stored longer after treatment resulted in less efficient uptake by the worms. Movement of the hydrocarbons into sediment aggregates from which they were less readily extracted probably caused this. The animals accumulated the hydrocarbons approximately to a steady state within about a week (Leppanen and Kukkonen, 2000b). The oligochaetes cleared some of the pyrene as intact pyrene, but, curiously, more polar metabolic by-products were retained longer. Palmork and Solbakken (1980) administered ^{14}C-phenanthrene intragastricly to Norway lobsters (*Nephrops norvegicus*) and found that radioactivity was taken rapidly into heart, green gland, hepatopancreas, and gonads as well as being found in stomach and intestine. The highest amounts were found in hepatopancreas, muscle, and gonads. These animals clearly had the capacity to absorb phenanthrene from the gut and to distribute it to internal organs. However, they appeared to excrete phenanthrene unchanged (Solbakken and Palmork, 1981).

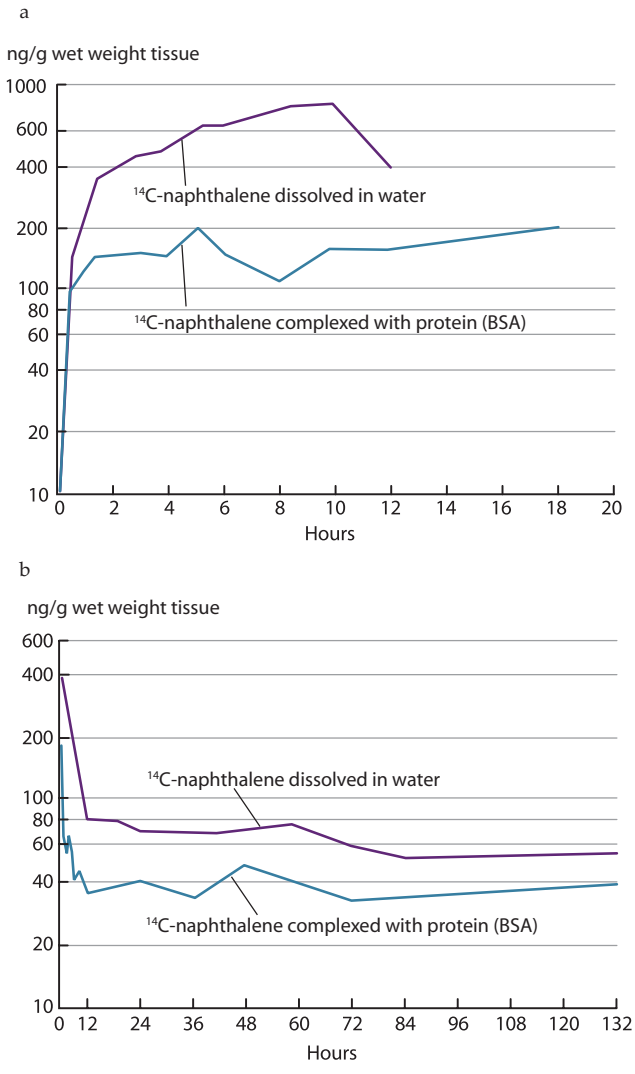

Figure 5.2. (a) Accumulation of naphthalene from water by larval spot shrimp (*Pandalus platyceros*) and (b) depuration after transfer to clean water (redrawn from the original by Sanborn and Malins, 1977).

McElroy et al. (1989) derived regression equations describing bioconcentration factors of polychlorinated biphenyls (PCBs) and PAHs by *Mytilus edulis*. The equations were:

$$\text{For PCBs, Log BCF} = 1.73 + 0.594 \log K_{ow}$$

$$\text{For PAHs, Log BCF} = -1.40 + 0.965 \log K_{ow}$$

where K_{ow} is the octanol/water partition coefficient.

Solving these equations for two hypothetical compounds with log K_{ow} = 5, the PAH would have a log BCF of 3.4 as compared with 4.7 for the PCB, indicating that the PAH would be taken up less than one tenth as effectively as the PCB. Experiments with PAHs bioconcentrated from dispersed oil droplets (Blended Arabian Light) have confirmed that the uptake of a variety of PAHs by *Mytilus edulis* was directly related to the K_{ow} but that uptake reached a maximum for PAHs with K_{ow} about 5 and increased little with PAHs more hydrophobic than that (Baussant et al., 2001b).

The biological availability of PAHs from water is affected by the presence of some types of dissolved or suspended organic matter. Experiments with *Pandalus platyceros* showed that the presence of bovine serum albumin reduced the uptake of naphthalene. Several experiments have been reported showing that other materials in exposure experiments can modify the biological uptake by test animals. For example, Akkanen and Kukkonen (2003a) conducted exposures of *Daphnia magna* to tritiated benzo[*a*]pyrene in water with variable levels of natural dissolved organic matter (DOM) from Lake Kontiolampi, Finland. Uptake was reduced in the presence of DOM but the effect was nonlinear with a large effect exerted by low concentrations of DOM and decreasing increments of effect with increasing amounts of DOM. The results were explained by adsorption of the benzo[*a*]pyrene to the DOM in a way that reduced its availability to the daphnids.

Hydrocarbons can be accumulated from the sediment, although less efficiently than from water. Sediment and soils seem to be the major environmental reservoirs for PAHs (e.g., Hites et al., 1977; Gschwend and Hites, 1981; Jones et al., 1989). It has been difficult to establish clearly the extent to which accumulation from sediment is a one-step transfer from sediment to organisms or a two-step process of transfer from sediment to water to organisms. Early experiments by Roesijadi et al. (1978) exposed clams (*Macoma inquinata*) to sand treated with Prudhoe Bay crude oil and found very little accumulation of 2-methylnaphthalene. When clams were suspended over the oiled sand and not allowed to come into contact with it, they accumulated almost the same amount of 2-methylnaphthalene as those in contact with sediment, suggesting that accumulation directly from sediment was negligible. Similar experiments on the uptake of anthracene, 3-methylcholanthrene, dibenz[*a,h*]anthracene, pyrene, benzo[*a*]pyrene and benzo[*b*]fluoranthene either directly from water or from sediment by short-necked clams (*Tapes japonica*) were reported by Obana et al. (1983; redrawn by McElroy et al., 1989). Uptake was rapid from water but slow

from sediment. In these experiments, clams held in contact with the sediment accumulated the PAHs more rapidly than clams held above the sediment surface. Probably both one-step and two-step mechanisms apply but differ in relative importance in different settings and with different organisms. In some instances, the properties of the organism affect the availability of PAHs from sediment or other adsorptive material. For example, Crocker et al. (1995) noted the ability of a naphthalene-degrading bacterial strain of *Pseudomonas putida* (strain 17484), to metabolize naphthalene sorbed to chemically modified clay. However, *Alicaligenes* sp. (isolated from petroleum-contaminated soil) was able to metabolize naphthalene but only after it had desorbed from the clay. In this instance, the availability of the sorbed naphthalene was different for the two organisms. Regardless of the efficiency and mechanisms of uptake from sediment, the fact that it occurs at all is a concern because hydrocarbons persist there for long periods.

In a field setting, Fukuyama et al. (2000) conducted a reciprocal transplant experiment in which bivalves (*Protothaca staminea*) were moved between oiled and unoiled intertidal habitat in the vicinity of the *Exxon Valdez* spill. The clams were transplanted in 1994, five years after the initial spill, and hydrocarbons still present in sediment at the oiled site were accumulated by the clams transplanted from the unoiled site. Hydrocarbon levels fell in the clams transplanted from the oiled site to the unoiled site. These experiments indicate that hydrocarbons were exchanged between the clams and their habitat, with levels in clams tracking, after a lag time, those in the habitat, consistent with a partitioning mechanism governing exchanges of hydrocarbons between clams and habitat.

Several factors influence the clearance of hydrocarbons. Harris et al. (1977) studied the retention of naphthalene by the barnacle *Calanus helgolandicus* over a range of 5–16 °C. Using a 24-hour test period, they found that almost 80% of naphthalene was retained at 5 °C but only about 25% at 16 °C. In addition, factors like body size, dry weight, ash-free dry weight, and lipid content correlated with naphthalene retention. Of these, lipid content correlated most strongly with retention. James (1989) examined the effect of temperature on the clearance of ^{14}C-benzo[*a*]pyrene by the spiny lobster (*Panulirus argus*). Whether the radioactivity derived from benzo[*a*]pyrene was measured in the whole animal, the hepatopancreas or the tail muscle, the effect was the same: it was cleared more rapidly in summer temperatures (26.5–29 °C) than in winter temperatures (13.5–16.5 °C). Over this temperature range, the amount of radioactivity remaining six weeks after exposure was about an order of magnitude lower in the summer than in the winter. After three days, very little of the remaining residue in the lobsters was present as intact benzo[*a*]pyrene indicating high ability to metabolize this compound. Among the metabolic by-products tentatively identified (by high performance liquid chromatography, HPLC) were: 1-, 3-, 7-, and 9-hydroxybenzo[*a*]pyrenes, 1,6-, 3,6-, and 6,12-quinones and 7,8-, 4,5-, and 9,10-dihydrodiols. The production of metabolites was about the same in winter as in summer indicating that the slow clearance in winter was due not to slow biotransformation but rather to slow elimination of biotransformed products. These results have ominous implications for clearance of hydrocarbons and their metabolites by invertebrate animals in the Arctic, where temperatures are below those used in these studies for most of the year. These results suggest a need for similar studies conducted at low temperatures with cold-adapted Arctic species.

Uptake from water has been treated primarily as a function of the partitioning properties of the hydrocarbons. However, the properties of the organisms can determine uptake and/or retention of hydrocarbons. For example, Skadsheim et al. (2005) suggested that body size affected accumulation of hydrocarbons from a water-accommodated fraction from crude oil. The smallest organisms (algae, *Isochrysis galbana*) were found to contain proportionately more of the PAHs with higher molecular weights than the larger organisms (crustacean, *Artemia salina* and fish, *Scophthalmus maximus*), after exposure to the 'water accommodated fraction' for 24 hours. The algae contained about 53% (of total PAHs) as dibenzothiophenes and alkylated dibenzothiophenes while the crustaceans contained only about 27% of these compounds and the fish about 6%. The authors pointed out that these organisms differed taxonomically and that differences in hydrocarbon residues might be due partially to greater capacity of the larger organisms to metabolize larger PAHs. These experiments are difficult conceptually because organisms of different sizes are likely to differ in a number of physiological variables in addition to size. Nonetheless the effect of body size or other expressions of body surface or mass on net accumulation of hydrocarbons merit further attention. These same authors raised the interesting question of the influence of water pressure. That is, do toxicology results obtained with organisms in shallow water apply to organisms from deep-sea areas?

Regardless of the sources, field studies have confirmed the uptake of PAHs by diverse taxa. Contaminated invertebrate animals have been recovered after oil spills (e.g., Blumer et al., 1970). Hellou et al. (1994) identified several PAHs in muscle and hepatopancreas of spider crab (*Hyas coarctatus*) and snow crab (*Chionoecetes opilio*) from locations off Newfoundland. Several residue studies reported levels of hydrocarbons in invertebrate animals after the *Exxon Valdez* and *Braer* spills.

5.3.1.2. Fish

Hydrocarbons in water are taken up rapidly by fish. Lee et al. (1972) exposed several species of temperate fish to tritiated naphthalene and benzo[*a*]pyrene and found that radioactivity was present in deep tissues after only a few minutes. Naphthalene was accumulated more readily than benzo[*a*]pyrene in spite of its lower K_{ow}. The pattern of radioactivity suggested uptake by gills followed by accumulation of both original hydrocarbon and several metabolic by-products in liver, gut and muscle. The main pathway of depuration was through the bile. The disagreement between the observed bioconcentration of naphthalene and benzo[*a*]pyrene and that predicted by partitioning based on hydrophobicity (K_{ow}) has been confirmed by subsequent experiments. For example, Baussant et al. (2001b) described the uptake of PAHs from oil in the water by young turbot (*Scophthalamus maximus*). They observed that uptake rate constants increased slightly with increasing K_{ow} up to about 4 and then declined for PAHs with K_{ow} greater than 4. The elimination rates were essentially independent of K_{ow} in these young turbot with the result that bioconcentration factors declined for PAHs with K_{ow} above 4.

Rapid uptake and depuration of ^{14}C-benzene by northern anchovy (*Engraulis mordax*) and striped bass (*Morone saxatilis*) was reported by Korn et al. (1976). Experimental exposures of larvae of several species of fish (zebrafish, *Danio rerio*; Atlantic cod, *Gadus morhua*; herring, *Clupea harengus*; and turbot; *Psetta maximus*) showed that all species accumulated PAHs readily from the water (Petersen and Kristensen,

1998). The authors compared the uptake and depuration by eggs and larvae of zebrafish; uptake and depuration rate constants were lower for eggs than for larvae, but the ratio of these constants, a measure of the bioconcentration factor, was about the same in both life stages. The uptake of benzo[*a*]pyrene from seawater through the gills by polar cod (*Boreogadus saida*) was measured at a temperature of -1 °C (Ingebrigtsen et al., 2000); depuration via bile was confirmed in these experiments. Zitko et al. (1984) noted low contamination of liver of Atlantic cod and haddock (*Melanogrammus aeglefinus*) following a blowout of gas condensate from a well near Sable Island (Nova Scotia) in 1984. Experimental laboratory exposures of Atlantic salmon (*Salmo salar*) to the same condensate indicated rapid uptake of hydrocarbons and slower clearance. Lockhart et al. (1989) reported an experiment with xylene suggesting that temperature influenced the uptake rate more than the clearance rate. Young Arctic char (*Salvelinus alpinus*) were exposed to ^{14}C-methyl-labeled *o*-xylene and more label was accumulated at 15 °C than at 5 °C. However, the clearance half-life was about the same (6 hours) at both temperatures.

Fish can accumulate hydrocarbons from their food. Roubal et al. (1977) incorporated radioactively labelled naphthalene and anthracene into food pellets given to coho salmon (*Oncorhynchus kisutch*) fingerlings. Sampling at intervals from 24 to 336 hours after feeding revealed radioactivity in liver, brain and flesh on all occasions, with the highest amounts in the samples taken at 24 hours. Varying proportions of tissue radioactivity were found as metabolites rather than original hydrocarbons. Several species of fish and Norway lobsters were fed phenanthrene and body tissues were analyzed for parent phenanthrene and metabolic by-products. The major metabolite formed was 1,2-dihydro-1,2-dihydroxy-phenanthrene, especially in liver (Solbakken and Palmork, 1981). Studies of hydrocarbons taken up by young pink salmon (*Oncorhynchus gorbuscha*) and chum salmon (*O. keta*) following the *Exxon Valdez* spill suggested consumption of small droplets of oil (Sturdevant et al., 1996; Carls et al., 1996). After consumption of a droplet, probably a two-step uptake process similar to that hypothesized for sediment may apply in the intestine (i.e., from droplet to water and then from water to organism).

Different life stages of the same species may accumulate hydrocarbons differently. Thomas et al. (1989) exposed coho salmon smolts and jacks to the WSF of Cook Inlet crude oil in seawater and measured the aromatic hydrocarbons in the liver, gut and muscle after intervals of up to 20 days. Smolts accumulated 5 to 30 times more aromatic hydrocarbons than jacks. The levels of fat in liver, gut and muscle from smolts were higher than in those from jacks and the authors hypothesized that differences in the accumulation of hydrocarbons may have been the result of different levels of fat in the two life stages. The levels of fat in organisms have been invoked repeatedly to help explain the uptake and release of fat-soluble toxicants. It may be hypothesized that lipid levels would ameliorate toxicity of hydrocarbons by sequestering them, at least temporarily, with possible release later in the annual cycle. Such a mechanism was hypothesized early in investigations of the ecotoxicology of DDT; toxicity became evident in lake trout (*Salvelinus namaycush*) larvae as reserves of yolk containing DDT were mobilized (Burdick et al., 1964).

Fish can obtain hydrocarbons from sediment. McCain et al. (1978) exposed English sole (*Parophrys vetulus*) to sediment treated experimentally with Alaskan North Slope crude oil and found that the fish accumulated hydrocarbons into skin, muscle and liver. Several low-boiling aromatic compounds were identified in the fish, notably tetramethylbenzene and 1- and 2-methylnaphthalene. Hellou et al. (1995) described accumulation of several PAHs and alkylated PAHs in muscle of winter flounder (*Pseudopleuronectes americanus*) from sediment treated experimentally with Hibernia crude oil. The highest concentrations found in muscle were the alkylated naphthalenes for which levels up to about 100 ng/g were found. The lowest levels found were for the alkylated tricyclic phenanthrenes and anthracenes. The 'biota-sediment accumulation factors' correlated with the water solubility of the PAHs. This was explained by a two-step accumulation with movement of the hydrocarbons from sediment to water and then from water to fish. The low bioaccumulation of the larger, less soluble PAHs was because such small quantities partition into the water phase. Consequently, in order to build up its tissue concentrations to equilibrium, a fish must respire a larger volume of water and that requires a longer exposure time. For example, the theoretical time required to reach equilibrium for naphthalene was 5 days, for phenanthrene 85 days, for pyrene 348 days, and for benzo[*a*]pyrene nearly 7 years. Hellou (1996) tabulated levels of PAHs reported in wild fish from several locations. There is some uncertainty in comparing levels reported by studies in which different analytical methods were applied, but levels in muscle ranged from below detection to 689 ppm (μg/g dw) with the very high values reported from the Arabian Gulf. Levels in liver were generally higher than in muscle with a range from below detection to 885 μg/g dw. PAHs in gonads ranged from below detection to 57 μg/g dw.

Measurable levels of hydrocarbons have been reported in Arctic fish and other marine organisms irrespective of the mechanisms of uptake (see Chapter 4). Levels of various hydrocarbons in Arctic freshwater and marine biota have been tabulated in Chapter 4. These levels range up to 290 μg/kg (ww) in freshwater fish (sum of 26 PAHs in liver of Arctic char from near Igaliko, Greenland). For marine animals (invertebrates, fish, mammals) most tissue samples had levels under 100 ng/g ww but a few exceeded 1000 ng/g. Brown et al. (1996) examined eggs of herring with oil visibly adhering to them following the *Exxon Valdez* spill and observed the same profile of hydrocarbons in eggs as in the source oil.

Some of the components of produced water have the potential to accumulate in biota. Among the most pharmacologically active are the derivatives of phenol. Sundt and Baussant (2003) investigated the uptake of several tritiated alkylated phenols (C_4–C_7) by juvenile Atlantic cod maintained in seawater at 11 °C. The fish accumulated the tritium rapidly when the labelled phenols were presented in seawater with maximum concentrations reached within about two days. The bioconcentration factors for the four phenols ranged from about 100 to 500 and half-lives in the fish ranged from two days for 4-*tert*-butylphenol to eight days for 4-*n*-heptylphenol. Uptake of the same phenols from food was less effective with only 8–14% absorbed. Generally, the highest tissue radioactivity levels were found in bile and intestinal contents, consistent with the short half-lives.

5.3.1.3. Marine mammals

The routes of uptake by marine mammals include direct contact, consumption of contaminated foods, and possibly inhalation of volatile compounds. The low efficiency of clearance by invertebrate animals makes it likely that animals that feed on them (e.g., walrus, *Odobenus rosmarus rosmarus* and sea otters, *Enhydra lutris*) will be exposed

through their diet. Bodkin et al. (2002) noted that otters dig large amounts of sediment to obtain food and hence are exposed to both contaminated sediments and food organisms (often sea urchins). Whatever the means of uptake, marine mammals do accumulate hydrocarbons, although levels are usually very low because they metabolize them so quickly. Hellou et al. (1990) summarized levels of PAHs that have been found in several species of seals and whales; all were below 3 μg/g when expressed as Venezuelan crude oil equivalents. Holsbeek et al. (1999) identified low levels (up to 107 ng/g dw) of PAHs with two to four aromatic rings in blubber of sperm whales (*Physeter macrocephalus*) stranded in Belgium in the winter of 1994/95. High concentrations of hydrocarbons were found in some marine mammal tissues following the *Exxon Valdez* spill. Sea otter carcasses (n = 878) were recovered following the spill. By virtue of their habits and lack of blubber for insulation, these animals are especially vulnerable to oil spills. Mulcahy and Ballachey (1994) reported hydrocarbon concentrations in ten oiled carcasses recovered from western Prince William Sound, Alaska. Chemical residue analyses for aliphatic and aromatic hydrocarbons were carried out on liver, kidney, muscle, brain, and intestine and the patterns for different hydrocarbons determined. The residue levels were all low but fell into three groups hypothesized to have been derived from otters with different exposure and survival histories:

- Group 1, aliphatic hydrocarbons 6 μg/g ww, aromatic hydrocarbons 0.7 μg/g ww, consistent with animals that had died shortly after exposure to the oil;
- Group 2, aliphatic hydrocarbons 10 μg/g ww, aromatic hydrocarbons 2 μg/g ww, consistent with animals that had survived long enough following exposure to move hydrocarbons into tissues and to begin metabolizing them; and
- Group 3, aliphatic hydrocarbons 11 μg/g ww and aromatic hydrocarbons 3 μg/g ww, consistent with animals that had survived long enough following exposure to move hydrocarbons into tissues but died before sufficient metabolism had taken place to reduce tissue concentrations (Mulcahy and Ballachey, 1994).

Hydrocarbon levels were reported from twelve sea otters from areas of Alaska where there had been no spills. These otters contained mean levels of aliphatic hydrocarbons in kidney, liver and muscle of 2.8–3.8 μg/g as compared to values of 6.1–7.7 μg/g in otters recovered after the *Exxon Valdez* spill. The levels of aromatic hydrocarbons in the same organs from the unexposed otters were 0.16–0.17 μg/g and those for the oiled otters from Prince William Sound were 0.60–1.50 μg/g.

Harbour seals (*Phoca vitulina richardii*) were sampled in 1989 and 1990 from oiled and clean areas of Prince William Sound and from reference areas. Harbour seals had been observed to swim in the oil, to breathe at the oil/water interface and to contact oil onshore. "Many seals were heavily coated with oil from the time of the spill in March (1989) until their annual moult in August" (Frost et al., 1994). Samples of liver, blubber, muscle, brain, kidney, heart, and lung were analysed for low- and high-molecular weight hydrocarbons but only very low levels were found in all organs except blubber. Hydrocarbons measured in blubber of oiled seals corresponded to the gross observation of the severity of oiling. Considering the seals collected in 1989, those with oiling described as 'very heavy' had blubber mean low-molecular weight aromatic compounds (LAC) over 200 ng/g. Seals with oil described as 'heavy' had LAC about 100 ng/g; one with 'moderate' oiling had LAC of 43 ng/g; two with 'light' oiling had LAC of only 10 ng/g. These variously oiled seals may be compared with seals with no oiling for which the blubber levels averaged ~10 ng/g. The bile from some of these seals was analysed by HPLC for fluorescent compounds using wavelengths to determine naphthalene equivalents and phenanthrene equivalents. The amounts of these fluorescent materials were high, ranging up to a maximum of 365 μg/g in the bile of one heavily oiled pup. Naphthalene equivalents were consistently higher than phenanthrene equivalents and seals with no known exposure had positive readings. Nonetheless, all but the lightly oiled seals were readily distinguished from unoiled seals and the amounts of naphthalene equivalents and phenanthrene equivalents in bile correlated with the physical description of oiling. These observations indicate clearly that the seals accumulated hydrocarbons and metabolized them so efficiently that blubber and bile were about the only organs in which residues were found for long.

There have been a few experimental exposures of marine mammals to oils. Engelhardt et al. (1977) exposed ringed seals (*Phoca hispida*) by immersion in a 1-cm thick layer of Norman Wells crude oil for 24 hr in seawater. The seals were then sacrificed on days two, six and seven following the initial exposure. Oil in body tissues was measured fluorometrically and was found at highest concentrations (up to ~60 μg/g) in urine and bile two days after treatment. Lower levels of oil were found in blood (~10 μg/g), plasma (~5 μg/g), blubber (~4 μg/g), brain (~2 μg/g), liver (2-3 μg/g after two days but up to ~9 μg/g by seven days), kidney (~5 μg/g), and muscle (~13 μg/g), with only a trace amount found in lung (~1 μg/g). When the oil was fortified with tritiated benzene and incorporated in gelatin capsules in fish fed to seals daily for five days, radioactivity appeared rapidly in the blood and then declined rapidly over a period of 28 days. The low concentrations found in deep organs of oiled marine mammals confirm the need for caution in the interpretation of chemical residues of hydrocarbons in these organs as indicators of exposure history.

Some information is available on the effects of ingested crude oil on polar bears (*Ursus maritimus*) (Engelhardt, 1981; Øritsland et al., 1981). Three bears were externally dosed with Midale crude oil and monitored for physiological responses. Given an opportunity to avoid oil on the surface of an experimental pool, the bears did so, indicating that they could detect the oil and chose to avoid it when possible. Bears are clean animals and spend much time grooming their fur to remove any foreign substance (Stirling, 1990). The bears ingested the oil by licking it from their exposed fur. Much was eliminated by vomiting and in the faeces, but some was absorbed and later found in body fluids and tissues (Engelhardt, 1981). In October 1989, a sub-adult polar bear drank four litres of hydraulic oil in Churchill, Manitoba (Derocher and Stirling, 1991). The bear's fate is unknown, but it is assumed to have died (Øritsland et al., 1981). There is other anecdotal information that wild polar bears occasionally seek out and consume motor oil and grease (Engelhardt, 1983).

Marine mammals would inhale hydrocarbons whenever they surfaced to breathe in the vicinity of a recent oil spill. Øritsland et al. (1981) reported on the effects of polar bears inhaling hydrocarbon vapours. Their report indicated that inhalation of hydrocarbons from unweathered crude oil in a confined space may have contributed to the death of two of three polar bears exposed to oil in their experiments. Given the effects of diffusion, dispersion and winds on an open

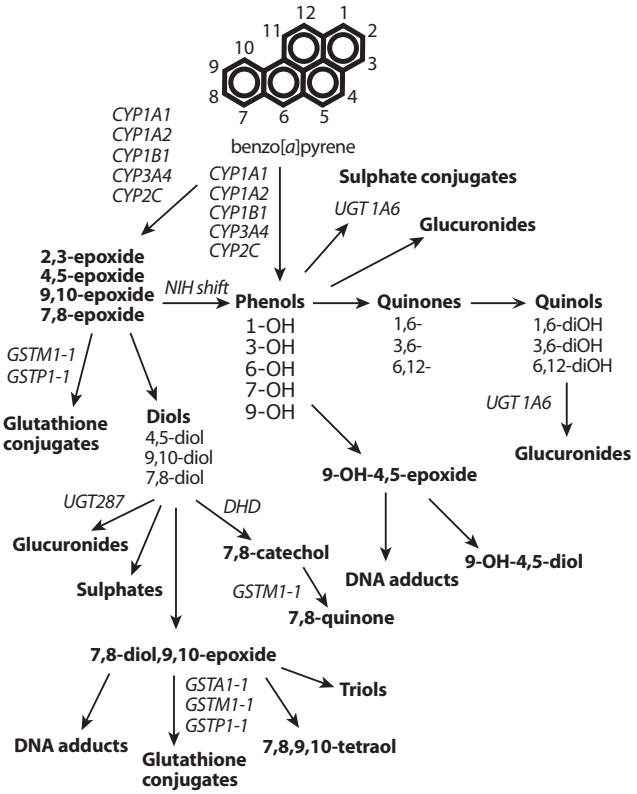

Figure 5.3. Metabolism of benzo[*a*]pyrene including activation by phase I enzymes (CYP) and deactivation by phase II enzymes (GSTA, GSTM, GSTP, UGT) (Binderup et al., 2004).

ocean spill, it is likely that harmful concentrations of vapours would rapidly dissipate and be short-lived. Following an oil spill, most light hydrocarbons would evaporate within a few days to a week and inhalation would not be likely to threaten individual polar bears or the population. Geraci (1990) noted that volatile hydrocarbons would pose an inhalation hazard to cetaceans in the area for a short time after a spill.

Wolfe et al. (1994) estimated that the maximum concentration of volatile hydrocarbons to be about 9 ppmv in the air over the centre of a slick of *Exxon Valdez* oil. The slick covered about 300 km^2 (although not in a single slick) and the half-life of these hydrocarbons in air ranges from about one to seven days with the monoaromatic compounds degraded most rapidly. A marine mammal in this slick would have no choice but to inhale any hydrocarbons in the air immediately over its location and so that could be as high as 9 ppmv. Furthermore, this would continue, presumably at a reduced rate, until the slick became depleted in volatile hydrocarbons or until the animal moved elsewhere. Once in the air, the hydrocarbons would be mixed vertically and advected to other locations by air movements, both of which would reduce the concentration in air just above the surface. The half-lives of low-boiling hydrocarbons in air are brief, within a day or two. The *Exxon Valdez* slick situation approximates the worst case that might be expected following an oil spill into open water. For comparison, the most stringent U.S. Environmental Protection Agency acute exposure guidelines for humans exposed for eight hours are: benzene, 9 ppm; toluene 200 ppm; and xylenes 130 ppm (http://www.epa.gov/oppt/aegl/). Hence, if the sensitivity of marine mammals is comparable to that of humans, and if benzene is a major component of the mixture of hydrocarbons, then it is possible that marine mammals could be affected by inhalation for a few days following a large spill.

The key to understanding the low levels of hydrocarbons found in vertebrates appears to lie in the clearance of these compounds following their biotransformation into by-products that are easily excreted (e.g., Krahn et al., 1984). Biotransformation is the sum of several processes by which hydrocarbons (or other compounds) are subjected to chemical change by living organisms. Lockhart, (W.L., Canada Dept. Fisheries and Oceans (retired), unpublished data) injected Norman Wells crude oil into harp seal (*Phoca groenlandica*) pups very shortly after birth and found strong induction of liver aryl hydrocarbon hydroxylase (AHH) activities indicating that the capacity to metabolize hydrocarbons was present in these animals very early in life, possibly before birth. Biotransformation processes are enzymatically mediated and result in either alteration of the parent compound (Phase I enzymes) or formation of products involving combinations of normally occurring substances and the parent biotransformed hydrocarbon (Phase II enzymes). The phase I enzymes introduce reactive, polar functional groups into lipophilic molecules; and these products are usually more water-soluble than the parent structures. The phase II enzymatic reactions produce a conjugation product that is amenable to excretion from the body. A metabolic chart showing products that can be formed from benzo[*a*]pyrene is shown in Figure 5.3.

5.3.1.4. Seabirds

Most studies relating to the metabolism of hydrocarbons in birds focus on the induction of enzymes and the presence of sub-lethal effects caused by the toxic components of the oil, or the toxic metabolic by-products. Hydrocarbons are metabolised in birds through the CYP4501A family of enzymes, forming oxidative metabolites that may be more toxic than the parent compounds (Troisi and Borjesson, 2005). The enzyme system is efficient enough that the levels of PAHs from birds feeding in oiled areas remain low in muscle and liver. Because of the rapid metabolism of hydrocarbons, the presence of elevated levels of the P450 enzymes in birds in areas of an oil spill provides the strongest evidence of exposure to the oil (Trust et al., 2000). Troisi and Borjesson (2005) reported the development of an immunoassay which measures PAH levels (as benzo[*a*]pyrene equivalents) in blood plasma, although not metabolic by-products. A survey of plasma PAH levels in oiled guillemots in the U.K. reported a mean concentration of 1.05 μg/g benzo[*a*]pyrene equivalents. The non-destructive assay may be a useful monitoring tool for measuring exposure levels to PAHs in seabirds in background areas, to establish a baseline, and after spills (Troisi and Borjesson, 2005).

The study of the uptake and metabolism of hydrocarbons and PAHs by seabirds and waterfowl is needed to be able to predict the long-term effects in seabirds. Peterson et al. (2003) noted that effects from the *Exxon Valdez* oil spill had continued for a decade after the spill, and that ecological relationships between compartments of the marine ecosystem have shifted because of declines in some food items and uneven rates of recovery of seabird species from the oiling. Several seabird species are demonstrating prolonged biological effects, due in part to changes in their food items and forage areas, but also from continuing exposure to residual oil from the spill. For example, adult pigeon guillemots (*Cepphus columba*) feeding in oiled sites showed elevated levels of hepatic cytochrome P450 enzyme activities, which are indicative of continuing exposure to hydrocarbons, presumably from remnants of the

oil spill (Golet et al., 2002). Clearly, more research is needed into the uptake, metabolism and effects of hydrocarbons in seabirds under these exposure conditions to be able to predict long-term effects from both large and small spills, particularly in the Arctic where low temperatures will slow the rate of oil degradation.

5.3.1.5. Concluding comments

Much of the literature describing uptake, metabolism and excretion of hydrocarbons has been developed under conditions and with species typical of temperate waters. The general pathways undoubtedly apply but rates of processes in poikilotherms are generally slower at lower temperatures. In general, aquatic organisms take up hydrocarbons, especially the low-boiling aromatic hydrocarbons, readily from water reaching equilibrium within a few days or less. The same applies to alkylated phenols. Hydrocarbons are taken up from sediment and food, although not as readily as from water. The bioconcentration factor from water to animals correlates statistically with the octanol/water partition coefficient up to a log K_{ow} value of 4 to 5. The major route of uptake by fish and invertebrates seems to be via gills. Aquatic mammals can inhale volatile hydrocarbons. For fur-bearing mammals like otters and polar bears, the principal mechanism of oil uptake is physical contact with fur followed by grooming as the animals attempt to remove the oil. Uptake from sediment and suspended particulate matter occurs and probably involves a two-step process, firstly of partitioning from sediment to water, including interstitial water, and secondly of partitioning from water to organisms. Animals that consume contaminated sediment or food or even droplets of oil can accumulate the hydrocarbons from the gut. Cetaceans appear more resistant to physical oiling. Geraci (1990) tabulated thirteen instances of observed encounters between whales and oil but no instances in which the oil was established to have caused injury to the whales.

Hydrocarbons are excreted rapidly by vertebrates, especially mammals and birds. Vertebrates metabolize and excrete hydrocarbons so efficiently that they do not build up high body burdens as they do with the structurally similar chlorinated hydrocarbons. Hydrocarbons found as body residues are at about an order of magnitude lower than chlorinated hydrocarbons of comparable physical properties. Excretion is often via the bile. The sequence in vertebrates seems to be enzymatic oxidation, often by epoxidation and hydroxylation in liver, followed by conjugation, often with sugars or amino acids or other small molecules, and then excretion largely via the bile and gut. Hydrocarbons are cleared by some, probably most, invertebrates, but not as efficiently as by vertebrates, due to the low levels of the required enzymatic activities in these animals. In some instances the clearance by invertebrate animals appears to be largely as the intact PAH but in others the more polar metabolites are formed. Clearance from poikilotherms usually depends on the temperature, generally with slower clearance at cooler temperatures.

The role of lipids in both the accumulation and retention of hydrocarbons bears further investigation. Potentially, stored lipids can buffer an organism from toxic effects by sequestering lipid-soluble toxicants. Arctic organisms often have enormous seasonal variation in lipid content. This in turn predicts different uptake and depuration dynamics at different seasons and possibly different risks of toxicity as well.

5.3.2. Tainting or chemical contamination of fishery resources and implications for human consumption

The most obvious immediate impact of oil on a fishery, in addition to mortality, is through tainting or chemical contamination of fishery products as a result of the accumulation of hydrocarbons. Tainting and contamination or even fear of them has resulted in closures of commercial fishing with obvious economic implications. The implications of oil tainting and contamination for consumption of subsistence foods are particularly relevant for the indigenous peoples of the Arctic who harvest and consume substantial quantities of fish and wildlife for community consumption.

5.3.2.1. Tainting

Tainting refers to undesirable tastes and odours imparted to fishery products (e.g., Blumer et al., 1970; Wilder, 1970; Connell, 1971). Many instances of tainting have been reported, but only a few have been from Arctic waters. The oily taste and odour of fishery products is readily detected by taste panels composed of people who smell and/or taste samples with no knowledge of the prior history of the samples. Several techniques for conducting taste assessments are well established in food and beverage industries. Usually these consist of assigning a numerical rank to the intensity of any taint detected or of discriminating a tainted sample from two or more untainted controls. The compounds most likely to be responsible for tainting are the low-boiling aromatic compounds (Ogata and Miyake, 1973) although the relatively long persistence of taint in some instances suggests an involvement of more persistent components or possibly of tainting by metabolic by-products. Tainting and contamination or even fear of them has resulted in closures of commercial fishing with obvious economic implications. For example, Thomas et al. (1989) reported that an oil spill in 1987 in Cook Inlet, Alaska, resulted in the harvest of oil-contaminated salmon and that forced closure of the fishery there. Effects on subsistence fishing are less well known but fears of contamination were sufficient to stop subsistence harvests in some communities following the *Exxon Valdez* spill (Walker and Field, 1991). Fall and Field (1996) reported that oil from the *Exxon Valdez* spill fouled water and beaches used for subsistence fishing, hunting and gathering. Following that, subsistence harvests fell by 77% in the year after the spill, largely because of uncertainty about the safety of subsistence foods. With subsistence foods, there is no marketing step as a point for regulatory action and the issue is one of risks to subsistence consumers. The implications of oil tainting and contamination for consumption of subsistence foods are particularly relevant for the indigenous peoples of the Arctic who harvest and consume substantial quantities of fish and wildlife for community consumption (e.g., Kinloch et al., 1992; Fall and Field, 1996; Nuttall, 2005).

Brandal et al. (1976) investigated tainting of Atlantic salmon and saithe (*Gadus virens*) experimentally by exposing them to an aged slick of *Ekofisk* crude oil for 68 days. The responses by the two species were quite different with the salmon having become tainted at the time of the first observations, four days after the beginning of exposure. In contrast, the saithe barely became tainted at all throughout the whole experiment. The salmon had lost the taint when samples were taken on day 22. Extraction of hydrocarbons from the salmon revealed a relationship between tainting and the level of naphthalenes, although the authors took

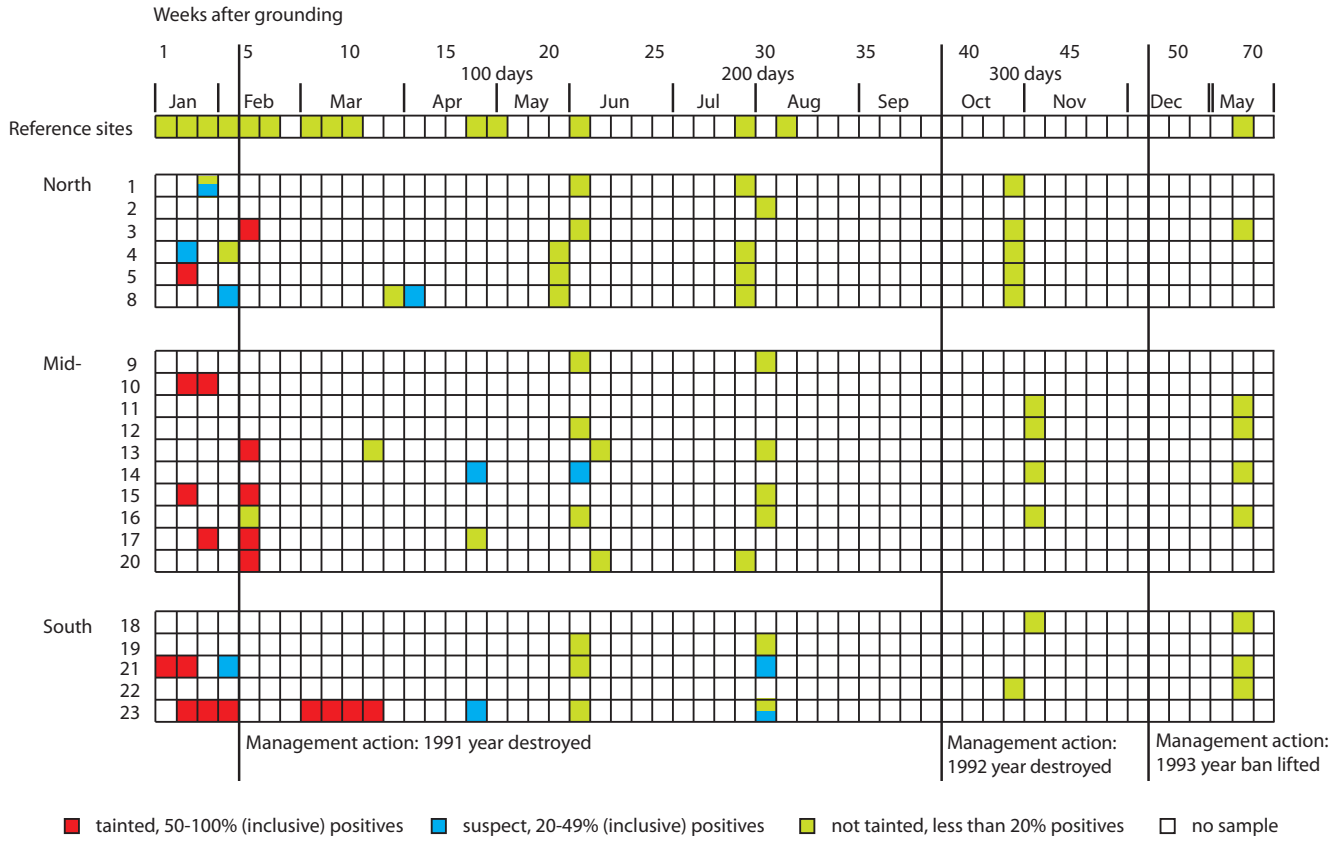

Figure 5.4. Incidence of tainting in Atlantic Salmon muscle, 1993–1994. The fish were sampled from aquaculture pens in the southern Shetland Islands following the *Braer* spill in January 1993 (Whittle et al., 1997).

care to point out that this did not establish naphthalenes as the cause of the taint. In keeping with these observations on saithe, Grahl-Nielsen et al. (1976) recovered saithe from nets oiled with Iranian crude oil following the grounding of the *Drupa* near Stavanger, Norway; in spite of oil on the nets, the saithe were not tainted.

Atlantic salmon from aquaculture pens in the southern Shetland Islands were found to be tainted after the spill from the *Braer* in January 1993, and some remained tainted for up to 30 weeks after the spill (Figure 5.4). Wild fish of several species became tainted while others did not. Dab (*Limanda limanda*), haddock, plaice (*Pleuronectes platessa*), lobster (*Homarus* spp.), queen scallop (*Chlamys opercularis*) and great scallop (*Pecten maximus*) became tainted but Atlantic cod, lemon sole (*Microstomus kitt*), ling (*Molva molva*) and edible crab (*Cancer pagurus*) did not. The species that remained tainted longest were the shellfish, queen scallop and great scallop, which were still judged to be tainted or 'suspect taint' when sampled over a year later in March 1993 (Topping et al., 1997). The authors reported that mussels (*Mytilus* spp.) were not assessed for tainting until February 1994, and were not tainted at that time; however, mussels examined in June 1994 did have a petroleum-associated taint.

A very small spill of diesel at Arnold's Cove, Newfoundland, resulted in the possible exposure of penned lobsters (*Homarus americanus*) for up to 10 hr (Williams et al., 1985). Lobsters from the pens and unexposed controls from a reference area were examined by taste panels with ambiguous results. The authors concluded that the exposure at Arnold's Cove had caused the lobsters there to taste differently from those taken from the control area but that they were not tainted sufficiently to make them unacceptable for market. This appears to be the only instance in which a spill-induced change in sensory evaluation was judged to be insufficient to suspend the marketing of a fishery product.

Tainting has been used to monitor the status of commercial fish species taken in the northern North Sea at points further than 3 km from production platforms. Samples of Atlantic cod, plaice, and dab were assessed for tainting but none was found to be tainted (McIntosh et al., 1990). The sampling locations greater than 3 km from production sites were outside the normal zone of contamination of sediments from the disposal of production by-products. Shepherd and Moffat (1998) described the use of sensory assessment as a monitoring tool in instances of contamination by hydrocarbons.

Two cases of tainting have been reported from the Canadian North. Lockhart et al. (2002a,b) reported petroleum hydrocarbon tainting of fish following spills into the Athabasca River, Alberta, in 1982 and into the Cameron River, Northwest Territories, in 1983. In the former case, the winter whitefish fishery was closed for the season following a spill into the Athabasca River of effluent from a refinery for extracting oil from oilsand. In the latter case, diesel was spilled when a truck overturned near the Cameron River on 1 March 1983 when the river was still ice-covered. There was no local winter fishery but a fisherman complained of a tainted whitefish caught downstream from the spill site several weeks after ice had broken up. The spilled fuel penetrated the snow (about 60 cm deep) and made its way to the river under the snow. A sample recovered from the river on 9 March retained its gas chromatographic profile, its toxicity to fish, and its ability to taint fish. Laboratory experimental studies were undertaken by exposing whitefish (*Coregonus clupeaformis*) to the diesel and tainting was produced after an exposure of only 2.5 hr.

Howgate et al. (1977) conducted experimental exposures of plaice, Norway lobster and brown shrimp (*Crangon cran-*

gon). Oil was mixed with sand and the mixture was spread on the bottom of exposure tanks before test animals were added. After one to two days (the first sampling period) all three organisms were rated tainted in comparison with controls. Tainting remained strong until the end of the exposures (seven to eight days with plaice, nine to ten days with Norway lobster, and four days with shrimp). Ernst et al. (1989) exposed Atlantic cod to WSF preparations of three crude oils (Hibernia, Brent, Amauligak) and a drilling mud base oil. Fish were tainted by the highest concentrations (1–4 ppm) of each of the WSFs. The Amauligak oil was one produced on test in the Canadian Arctic and its WSF resulted in tainting at the two highest concentrations (0.3 and 2.7 ppm). Following exposures, fish were transferred to clean water and depuration of tainting was rapid. Fish tainted initially were no longer tainted after only four days. The difference between the results for cod and those for salmonids is similar to that reported by Brandal et al. (1976) for saithe and salmon. The level and distribution of fat in edible parts of the fish may offer an explanation for the difference or at least a hypothesis to test. Rainbow trout (*Oncorhynchus mykiss*) and Arctic char were exposed to Norman Wells crude oil in the laboratory (Lockhart et al., 2002b) with the results shown in Figure 5.5. Tainting was produced rapidly; it was apparent at all exposure concentrations after only 4 hr. Fish were transferred to clean water after 72 hr of exposure but taint remained detectable, although diminished in intensity for the duration of the experiment (600 hr in Arctic char and 840 hr in rainbow trout). Very similar results were obtained by Davis (1995) with trout tainted experimentally with diesel oil; tainting persisted for as long as ten weeks after exposure was ended.

Davis et al. (1997) described experiments in which four species were exposed to several different oils, with and without chemical dispersants, and then examined by sensory panels. Tainting was observed consistently following exposure concentrations below 1 mg/L. Tainting threshold values for Forties crude oil mixed with dispersant (Noble Superdispersant NS25) for 24 hr at temperatures in the range of 10.8–13.8 °C were: Atlantic salmon 98 µg/L, rainbow trout 109 µg/L, edible crab >7700 µg/L, and blue mussel 32.3 µg/L. The presence of dispersant may have had some small effect on tainting but any such effect was too small to be of statistical significance. Experiments with different exposure times revealed that tainting could be induced in as little as 30 min. As exposures were maintained for longer periods up to 24 hr, the intensity of tainting usually became greater but much of the tainting effect was achieved during the first few hours. After exposure was completed, fish were maintained in clean water in order to describe the loss of taint. The intensity of tainting fell to below detection levels over a period of 120 days with loss of taint generally slower after exposure to diesel than exposure to either Forties crude or Medium Fuel Oil. Depuration of taint was examined following exposures

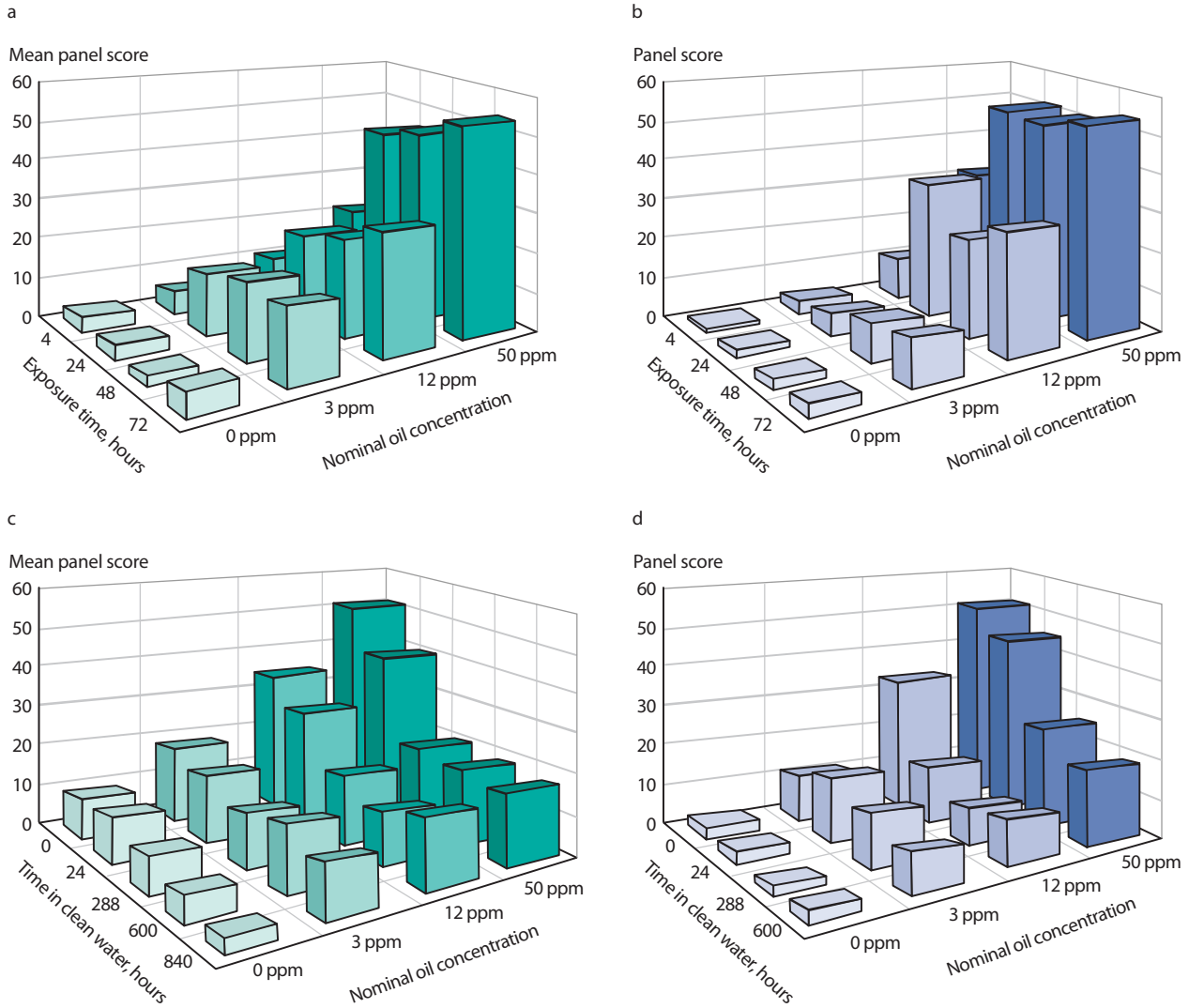

Figure 5.5. Uptake and clearance of oily taint in rainbow trout (a: uptake; b: clearance) and Arctic char (b: uptake; d: clearance) in laboratory exposures to *Norman Wells* crude oil (Lockhart et al., 2002b).

to four different levels of tainting and the depuration was related to the intensity of exposure with longer retention of taint typical of higher exposures. Comparing loss of taint (induced by Forties crude oil) among species, salmon and trout both lost their oily flavour at approximately the same rate but the mussels retained the taint for longer, in keeping with field observations following the *Braer* spill.

5.3.2.2. Chemical contamination

The companion issue of contamination of fishery products with identifiable levels of hydrocarbons has also been used as a basis for closures of fishing. An example of the use of chemical contamination to regulate a fishery following an oil spill was described by Gilroy (2000) following a spill from the M/V *New Carissa* at Coos Bay, Oregon on 4 February 1999. Following the spill, harvesting of shellfish was closed (on 11 February). The decision to reopen the shellfish fishery (on 4 March) after the spill was based on a level of benzo[*a*] pyrene equivalents falling below 10 μg/kg, a level calculated from risks of cancer to human consumers of the shellfish. This incident and several similar incidents used PAHs as the compounds on which to base fishery decisions because of their association with cancer and the ability to calculate risks based on consumption scenarios. The study did not report any tests for tainting and it is unlikely that the PAH residues would have been the cause of any tainting that may have occurred. The use of benzo[*a*]pyrene residues as a surrogate for PAHs in general has been suggested as a means of assessing the risks associated with oil spills (Binderup et al., 2004).

5.3.2.3. Concluding comments

Fish and aquatic invertebrates can acquire oily taints and odours after exposures of less than an hour at concentrations ranging down into the parts-per-billion range. The presence of tainting and/or measurable chemical contamination has forced the closure of commercial fisheries following oil spills with consequent economic losses. Similarly, rejection of tainted or contaminated fishery products or fear of such tainting or contamination has reduced consumption by subsistence fishers. The persistence of tainting after exposure has ceased varies among species. In lean fish like cod or saithe, tainting can be cleared in a few days; however, in fatty fish like salmonids tainting can last for weeks or months. Experience with the *Braer* spill suggests that tainting of invertebrate species can last longer than tainting of fish, which is consistent with the more limited capacity of invertebrate animals to metabolize hydrocarbons. Benthic invertebrates are probably more sensitive than pelagic organisms to contamination of sediments, with consequent long-term exposure. Laboratory experiments describing uptake and depuration of taint are probably oversimplifications but do suggest retention of taint for periods of a few months. It is not clear exactly which components of oil are responsible for tainting, but the most likely candidates are the low-boiling aromatic hydrocarbons. While there have been some tainting experiments with species found in the Arctic (whitefish, Arctic char), there do not appear to be any that have been conducted under Arctic conditions. This is especially the case for tainting experiments with invertebrate species. The loss of sales of fishery products is one of the easiest financial losses to calculate and hence offers a basis for legal claims for damages. In the instance of a subsistence fishery, however, the damage may not be estimated in purely economic terms. Experience following the *Exxon Valdez* spill has shown that the harvest of subsistence foods can be reduced by as much as 77%. Further sociological effort needs to be given to derive a satisfactory measure of damage to subsistence fisheries.

5.3.3. Toxicology of oils and hydrocarbons

The toxicology of oils, hydrocarbons, and related compounds was not discussed in the previous AMAP assessment (AMAP, 1998). Moore and Dwyer (1974) identified five responses over thirty years ago: direct lethal toxicity, sub-lethal disruption of physiological or behavioural activities, effects of direct coating by oil, incorporation of hydrocarbons in organisms which may cause tainting and/or accumulation of hydrocarbons in food chains, and changes in biological habitats, especially alteration of substrate characteristics. Some additional effects receiving growing attention include genetic effects, interactions between exposures to some aromatic hydrocarbons and sunlight, and social effects on species with strong social interactions.

5.3.3.1. Mechanisms of toxicity

Early work by Wiebe (1935) suggested that one mechanism by which oil exerted its toxicity on fish was by coating gas-exchange surfaces thereby limiting the uptake of oxygen and clearance of carbon dioxide. Experiments by Tagatz (1961) with American shad (*Alosa sapidissima*) indicated that effects of petroleum products were greater at low oxygen levels, implying that exposed fish required additional oxygen. Increased opercular rates were observed in pink salmon exposed to the WSF of Prudhoe Bay crude oil (Thomas and Rice, 1975). If diffusion of oxygen were impaired, then the fish might be expected to compensate by increasing opercular rates. In addition to possible impeded oxygen exchange, several authors have noted that oxidation of hydrocarbons by fish would require additional oxygen. Some compounds used in the oil and gas industry may compound this effect. For example, methanol at a concentration of 50 mg/L was reported to reduce the oxygen content of water (Kosheleva et al., 1997), possibly by supporting growth of saprophyte microflora (Sokolov et al., 1968).

The clinical picture of acute poisoning in fish has been described as a progression through several phases: excitation, suppression and loss of coordination of movements, narcosis, and death. The duration of each phase depends on the nature of the toxicant (Kosheleva et al., 1997). The latent period between the initiation of exposure and the observation of symptoms can be very brief – seconds or minutes, testimony to the rapid penetration of hydrocarbons into the fish, probably through gills. When young cod were exposed to gas condensate, the excitation phase lasted for 2–3 hr. The fish became localized at the surface of the aquarium, sometimes taking vertical positions, or jumping out of the water and swallowing air bubbles. Breathing rate was intensified. Then the fish became passive and fell to the bottom of the aquarium; reaction to external stimulation was suppressed. Faulty coordination of movement and balance were observed after 5–6 hr and the fish took lateral or inverted positions. Sensitivity to external stimulation was sharply decreased and a comatose state ensued. Breathing rate accelerated with increased amplitude of fluctuation of branchiate covers. Young cod died with signs of suffocation – with widely open branchiate covers and mouth (Kosheleva et al., 1997). Symptoms such as these imply effects on central nervous system control of critical functions such as respiration. Death by suffocation in oil-treated fish is similar, at least at a cellular level, to death induced by low levels of oxygen in the water.

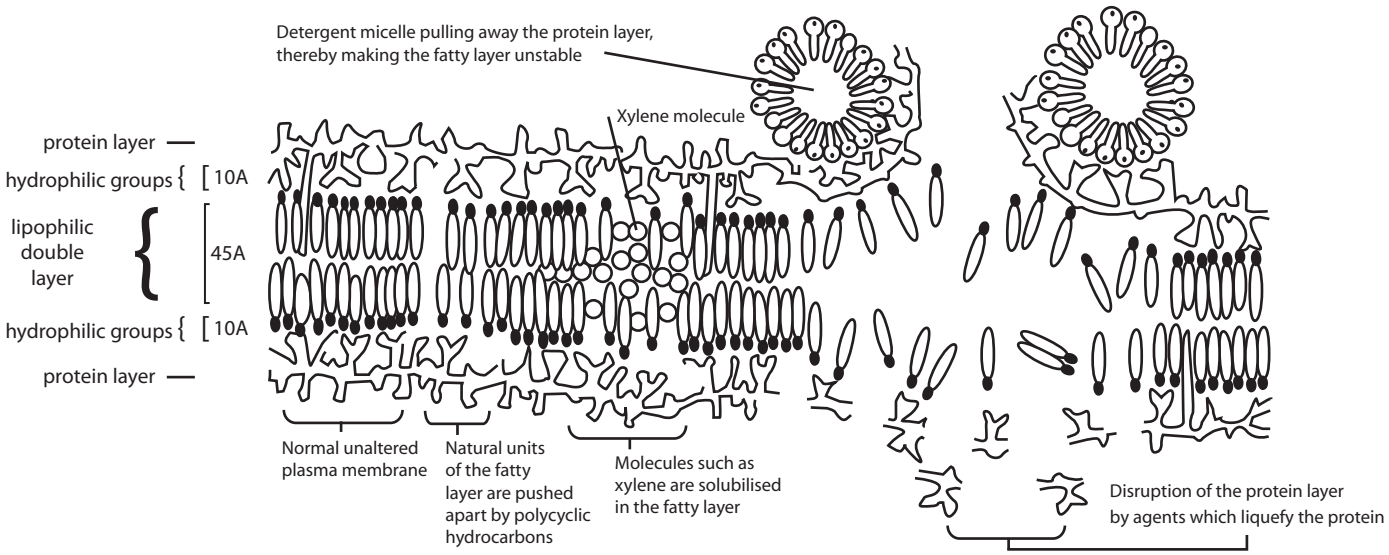

Figure 5.6. Early (1954) drawing of a mechanism by which several molecules including hydrocarbons may disrupt the physical integrity of the lipid layers of a membrane and result in passage of materials through the membrane (van Overbeek and Blondeau, 1954).

Toxic action can be explained, at least partially, by effects on processes associated with biological membranes. For example, van Overbeek and Blondeau (1954) offered a schematic model (Figure 5.6) for the toxicity of hydrocarbons to cell membranes and that model is still instructive, although the action seems more complicated than the Figure implies. The schematic model shows the penetration of hydrocarbon molecules into the lipid bilayer of a cell membrane with resulting physical disruption and loss of semi-permeability. A number of effects on algal cells were described by Kosheleva et al. (1997) (disintegration of chlorophyll, destruction of membranes, accumulation of waste products or blocking of gas exchange) all of which can be explained by interference with cytoplasmic membranes and loss of their functional integrity (Tkachenko and Zvylev 1985; Konstantinov 1986).

Sabo and Stegeman (1977) suggested that membranes in the estuarine fish *Fundulus heteroclitus* were probably the sites of action of oil. DiMichele and Taylor (1978) suggested that acutely lethal exposure to naphthalene acted primarily on the blood, probably on red cell membranes. They suggested that cell membranes are made more rigid by naphthalene and that more rigid cells are more subject to haemolysis or to physical blocking of capillaries. Failure of kidneys of young Atlantic cod noted in short-term Russian studies of exposures to gas condensate at concentrations of 5.2–7.8 mg/L was interpreted as evidence of the effects on permeability functions of membranes in osmoregulatory and secretion processes (Kosheleva et al., 1997). Studies of the effects of naphthalene on ion fluxes in midge larvae (*Chironomus attenuatus*) have suggested that the effect is not only on the structural integrity of membranes but also on the inhibition of enzymes systems associated with membranes, notably ATPase (Darville et al., 1983). Cellular respiratory functions dependent on mitochondrial membranes are sensitive to Prudhoe Bay crude oil, especially the aromatic components (Payne et al., 1987).

Several authors have reported a period of narcosis following exposure to hydrocarbons. For example, Brodersen (1987) noted rapid narcosis (within 20 min) when larval king crabs (*Paralithodes camtschatica*) or kelp shrimp (*Eualus suckleyi*) were exposed to the WSF of Cook Inlet crude oil. Nunes and Benville (1978) reported that clams (*Tapes semidecussata*) exposed to benzene displayed an initial period of activity followed by increasing time to respond to tactile stimulation and then by lack of response. Presumably this effect follows from impaired functions of the central nervous system. The narcosis effect is reversible with cessation of exposure, but it would be likely to have effects on wild animals where even short periods of narcosis would probably render animals susceptible to predators (unless the predators were equally affected).

A potentially important new mechanism has been suggested based on observations following the *Exxon Valdez* spill in Alaska, namely the disruption of social behaviour in some marine mammals and perhaps for other animals that maintain a well-defined social order (e.g., Dahlheim and Matkin 1994; Peterson et al., 2003).

5.3.3.2. Physical oiling

Physical oiling of body surfaces of some marine mammals (otters, polar bears) and birds, while not 'uptake' in the same sense as uptake into deep organs does represent adherence of oil to the animals often with toxic consequences. The usual mode of action is destruction of thermal insulating properties of fur with resulting hypothermia. Measures of physical oiling can be as simple as noting the proportion of body surface covered with oil. It is probably the most important mechanism of acute toxicity in these animals. Kooyman et al. (1977) examined the thermal conductance of pelts of several marine mammals and found that heat conductance was generally increased by oiling, meaning that heat is lost more readily from an oiled pelt than from a clean one. Pelts of sea otters and fur seals were affected most. Williams et al. (1988) confirmed the increased thermal conductance in oiled sea otter pelts and noted that it depended on whether or not a layer of air was retained in the fur. Davis et al. (1988) examined live Alaskan otters after coating 20% of the body surface with oil and found that whole body thermal conductance of the animals increased; they used vigorous muscular activity to help maintain body temperature. The otters spent more time swimming and shivering. Experiments on live polar bears and pelts showed thermal conductance of their fur increased significantly after oiling, and that oiled bears showed increased metabolic rates and elevated skin temperatures (Irving, 1972; Frisch et al., 1974).

Heavy oiling can also interfere with swimming ability in young seals. After oil drifted ashore in west Wales in 1974, Davis and Anderson (1976) noted that some grey seal (*Halichoerus grypus*) pups were so heavily coated with oil that they could not swim and they drowned when washed from the beach into the sea. Large numbers of harp seal pups were oiled and an unknown number killed when oiled in the Gulf of St. Lawrence (Warner, 1969).

Seals and whales with a layer of blubber do not depend on the pelage for thermal insulation and hence physical oiling is less acutely harmful to them. With cetaceans, skin contact has been observed but it appears to affect the animals very little if at all (Geraci, 1990). Fouling of baleen with oil has been investigated as a potential means of injury but oiling appears to have little effect on the functional properties of baleen (Geraci, 1990; Bratton et al., 1993).

Anderson et al. (1974) suggested mechanical clogging of respiratory surfaces as a mode of physical toxicity. It can be difficult to differentiate between toxicity caused by physical oiling and that caused by chemical toxicity, especially in small animals such as most invertebrates.

5.3.3.3. Photo-enhancement of hydrocarbon toxicity

Two phenomena have been described linking toxicology of PAHs and exposure to light. The condensed ring structure of PAHs causes them to absorb ultraviolet (UV) radiation strongly and natural sunlight is a source of UV radiation. Firstly, PAHs are subject to photochemical modification in water so that organisms can be exposed to the original PAHs plus any photochemically produced by-products. Secondly, PAHs can be taken up by organisms without prior modification and the enhancement in toxicity is observed when the exposed organisms are subjected to sunlight. The latter is often referred to as photo-enhancement of toxicity and was reviewed by Arfsten et al. (1996).

Generally, PAHs containing three or more aromatic rings have aqueous solubilities so low that toxic concentrations are not reached in truly dissolved form. However, Landrum et al. (1987) reported that anthracene was some 400 times more toxic to juvenile bluegill sunfish (*Lepomis macrochirus*) when exposures took place in the presence of sunlight. This effect has been observed in many taxa (frog tadpoles, *Rana pipiens*; mosquito larvae, *Aedes aegypti*; water flea, *Daphnia pulex* (Landrum et al., 1987); mysids, *Mysidopsis bahia* (Cleveland et al., 2000); copepods, *Calanus marshallae* and *Metridia okhotensis* (Duesterloh et al., 2002); oligochaetes, *Lumbriculus variegates* (Ankley et al., 1995); aquatic plants (Huang et al., 1993); and others (species tabulated in McDonald and Chapman, 2002).

Most of the early work on photo-enhancement of toxicity was done with freshwater organisms. The first observations of this phenomenon in marine organisms were those of Pelletier et al. (1997) working with the bivalve *Mulinia lateralis* and the mysid shrimp *Mysidopsis bahia*. These animals were exposed either to individual PAHs (anthracene, fluoranthene, pyrene) or to water-accommodated fractions of several oils (No. 2 fuel oil, Arabian Light Crude, Prudhoe Bay Crude and No. 6 fuel oil). The tests were conducted under fluorescent lights or UVA 340 bulbs that simulated natural sunlight. The combination of exposure to individual PAHs and illumination with UVA resulted in enhanced toxicity relative to that of PAH exposure under fluorescent lights. For example, the LC_{50} (concentration lethal to 50% of exposed animals) of fluoranthene to juvenile *Mulinia lateralis* bivalves was 3310 μg/L when tested in the presence of fluorescent light but 1.8 μg/L when tested under UVA light. With the exception of No. 2 fuel oil, the toxicities of the extracts of oils were increased when the exposures were conducted under UV radiation. For example, the extract from Prudhoe Bay crude oil killed none of the mysids when exposed at 0.1 g/L (mixing ratio, not measured concentration) under fluorescent light but killed all of them when exposed under UVA light.

Duesterloh et al. (2002) tested the phototoxicity of PAHs from weathered oil to two copepod species, *Calanus marshallae* and *Metridia okhotensis*, in natural daylight in Juneau, Alaska. These two copepods are important in the sub-Arctic food chain; both are translucent and present in surface waters. The authors used an artificially weathered North Slope Crude (20% volume reduction), then coated glass beads with the oil, drained the coated beads, and dried them under a hood for four days at 25 °C. Organisms were exposed to the oil by bubbling fresh seawater through columns containing the beads (or a control with clean beads). The animals were exposed in Erlenmeyer flasks in sunlight or covered to avoid sunlight. The experiments were done on a clear day and an overcast day in Juneau. Both species accumulated PAHs with a bioaccumulation factor of about 8000. They noted that the high surface to volume ratio of plankton contributes to bioaccumulation in small animals. The animals were examined following exposure, and 17 hr later. Ultraviolet radiation or oil alone affected 0–10% of the organisms. Combining the oil and UV radiation affected 80–100% of the organisms.

Several investigators have used similar experimental designs in which test organisms are exposed to PAHs, then moved to clean water and exposed to sunlight or some other source of UV radiation. By itself, neither the PAH nor the light is lethal. Bowling et al. (1983) used this type of experiment to demonstrate clearly that exposure to sunlight induced toxicity of anthracene in bluegill. Photo-enhancement of toxicity is controlled by the intensity of the light and by the amount of PAH in the organism. Ankley et al. (1995) showed that toxicity of fluoranthene to oligochaetes (*Lumbriculus variegates*) was described by the product of anthracene in tissues and the intensity of the light. The phenomenon applies to oil as well as to individual PAHs. Barron et al. (2003) exposed eggs and larvae of Pacific herring (*Clupea harengus pallasi*) to weathered Alaska North Slope crude oil with and without sunlight and UVA and found that toxicity increased dramatically when fish exposed to oil were subsequently exposed to sunlight or sunlight plus UVA. The effect of light on toxicity occurred when PAH residues in the fish were in the range 1–10 μg/g ww.

The mechanism(s) responsible for photo-enhancement of toxicity are of interest. Landrum et al. (1987) reviewed several possible mechanisms, mostly involving the formation of free radicals, and noted that the end result is generally damage to biological membranes. McCloskey and Oris (1993) observed several responses in bluegills exposed to anthracene and UV radiation, all consistent with the disruption of cell membrane integrity. Weinstein et al. (1997) described ultrastructure damage to gill membranes of fathead minnows (*Pimephales promelas*) following exposure to fluoranthene and UV radiation. The damage was of two types: the formation of lipid droplets, which was consistent with rapid lipid peroxidation, and an inflammatory reaction characterized by edema, leukocyte infiltration and vasoconstriction. The combined result of these changes was an increase in the water-blood diffusion distance (by almost 300%) in the gills that resulted in respiratory stress. Choi and Oris (2000) showed that the combination of exposure to anthracene and solar UV radiation resulted in oxidative stress in the form of superoxide anion production and lipid peroxidation in liver microsomes of bluegills.

McDonald and Chapman (2002) and Chapman and Loehr (2003) have argued that the photo-enhancement of PAH toxicity may be a laboratory artefact largely irrelevant in real settings. However, the Arctic has been receiving relatively high inputs of UV radiation in summer months due to depletion of stratospheric ozone (Kerr and McElroy, 1993) and this seems likely to continue for some time (Weatherhead et al., 2005). During this period the Arctic will probably receive greater inputs of PAHs as the oil industry becomes more extensive. The experiments of Duesterloh et al. (2002) and others showing the photo-enhanced toxicity of PAHs seem persuasive. Any interaction between these two inputs would be of concern in areas of shallow, clear water where young fish or other relatively translucent organisms might be present. Consequently, the Arctic, at least during summer, appears to represent an area of special concern for this interaction.

5.3.4. Lethal aquatic toxicology

The most common measure of toxicity is the acute lethal effect of exposure on organisms exposed under controlled laboratory conditions. With aquatic organisms, the exposure is usually measured as the amount of toxicant in the water of the test vessels. Typically, the organisms are exposed to a graded series of concentrations of the toxicant and mortality is recorded after the desired time intervals. Exposure conditions vary widely in studies of the lethal toxicology of oils to aquatic organisms (e.g., species of test organisms, exposure water [pH, dissolved oxygen, salinity etc.], duration of exposure, and external conditions such as light and temperature). Efforts have been made to standardize as many variables as possible, especially when the results are to be used for regulatory purposes. An important variable that has received surprisingly little study is temperature. The effects of temperature are difficult to predict because it acts both on the behaviour of hydrocarbons in water and on metabolic rates in aquatic poikilotherms.

5.3.4.1. Testing for acute lethal toxicity with aquatic organisms

Oils in general and most individual hydrocarbons are almost insoluble in water and consequently toxicologists have had to find ways to produce experimental exposure media reproducibly. Following the early work of Wiebe (1935) and later work by Anderson et al. (1974), investigators have usually mixed oil and water at some known mixing ratio, allowed the oil and water layers to separate for some time, and then tested organisms in dilutions of the aqueous layer. Variations of this procedure have been used to prepare what has been called the 'water-soluble fraction' (WSF) or 'water-accommodated fraction' (WAF) or 'oil-water dispersion' (OWD). This mixture consists mostly of dissolved low-boiling aromatic hydrocarbons (substituted benzenes and naphthalenes) and small, suspended droplets. If sufficient time is given for layers to separate, the presence of fine droplets is reduced. A criticism of this approach has been that the mixing ratio of oil to water in these preparations is much higher than might arise in the case of a real spill. Ongoing effort has been given to finding improved ways of preparing exposure media (e.g., Blenkinsopp et al., 1996). After dilution of the WSF with variable amounts of water to achieve the desired range of concentrations, the test animals are introduced and observations of mortality are taken at intervals, usually over about 96 hr. Many authors have reported losses of hydrocarbons from these preparations and that has prompted alternative approaches in efforts to maintain the exposure concentrations. The simplest of these is to enclose or cap the test solutions and prevent volatilization losses physically (e.g., Rogerson et al., 1983). Another design has been the preparation of fresh dilutions so that the exposure medium can be fully or partially replaced at intervals. The most elegant solution has been the development of various types of continuous-mixing apparatus to produce dilutions of the oil/water mixture continuously (e.g., Benville et al., 1981; Sanni et al., 1998).

The statistic usually reported to describe acute, lethal toxicity is the LC_{50} or the numerically equivalent median tolerance limit (TLm). Since the response typically varies with exposure time, the expression of toxicity usually includes an exposure time. For example, the 96-hr LC_{50} is the concentration lethal to 50% of the animals over an exposure of 96 hr.

The contribution of low-boiling aromatic compounds with one to two or even three aromatic rings to the toxicity of WSF preparations is not in doubt. Whether other compounds may contribute to the acute toxicology of whole oil is a matter for further research. Certainly several PAHs contribute to sublethal effects and chronic effects but these compounds are not usually identified in WSF preparations and probably contribute little to acute toxicity of WSF preparations. Aliphatic hydrocarbons are usually taken to have little toxic effect (e.g., Payne et al., 1995) although there have been occasional indications that they may be toxic to some organisms (e.g., Busdosh and Atlas, 1977). Anderson et al. (1987) distilled Prudhoe Bay crude oil to produce a fraction deficient in monoaromatic compounds and found that the toxicity to shrimp (*Pandalus danae*) was reduced greatly but, surprisingly, toxicity to sand lance (*Ammodytes hexapterus*) was unaffected. Additional distillation reduced diaromatic compounds further and again reduced the toxicity to shrimp, but had little effect on the toxicity to sand lance. Apparently, toxicity to shrimp was controlled by the low-boiling aromatic fraction whereas toxicity to sand lance was related to higher-boiling compounds. The most lipophilic compounds in oil are the PAHs and long-chain aliphatic hydrocarbons. Barron et al. (1999) tried to evaluate the contribution of PAHs to acute toxicity. They exposed mysid shrimp (*Mysidopsis bahia*) to WAFs of three previously weathered oils – 5x, B12, and C8. Oil 5x with the highest PAH content (175.8 μg/L) had the lowest toxicity to the shrimp (LC_{50} 1.5 mg/L). Oil B12 with a lower PAH content (132.5 μg/L) had a considerably lower LC_{50} (0.88 mg/L), and the remaining oil C8 with the lowest PAH content (25.6 μg/L) had an LC_{50} of 0.91 mg/L, almost the same as oil B12 in spite of the lower PAH content. Evidently the PAH content was not the primary determinant of acute toxicity in these experiments.

In addition to the problem of preparing reproducible mixtures of oil and water, toxicologists lack a fully satisfactory way to express exposure concentrations. The simplest method is to state the total amount of oil per volume of water (the mixing ratio, often called the nominal concentration). However, this overstates the exposure dramatically because only a small proportion of the oil (often less than 1%) dissolves and the dissolved fraction excludes most of the compounds present originally in the oil. Optical measurements such as the infrared absorption or ultraviolet absorption or fluorescence have been applied to extracts of exposure mixtures. These readings can be compared with those obtained from pure samples of the source oil or specific individual hydrocarbons, often naphthalene, and the concentration of oil in the exposure mixture can be estimated. Optical methods of analysis offer an improvement over the mixing ratio but they fail to identify the hydrocarbons present in the sample. Infrared methods sense particularly the C-H bonds that are abundant in alkanes, which comprise much of the oil, but

which seem to contribute little to its toxicity. Ultraviolet methods sense particularly aromatic nuclei which are abundant in some oils and some of which contribute significantly to both the acute toxicity and to chronic effects. Both methods are vulnerable to false positive results because materials other than hydrocarbons may have overlapping optical properties. The ultraviolet methods, either absorption or fluorescence, represent improvement over the infrared methods for toxicological interpretations. Increasingly, studies have examined extracts of exposure mixtures by gas chromatography or gas chromatography/mass spectrometry to determine separately the quantities of many compounds that are summed to provide the estimate of concentration. Gas chromatography or gas chromatography/mass spectrometry are the preferred methods derived to date. The general improvement in analytical methods often compromises comparisons among studies in which different analytical methods have been used.

5.3.4.2. Effects of temperature on toxicity

There appears to be no general study of the effects of temperature on the toxicology of hydrocarbons to cold-adapted aquatic organisms although a few specific studies exist. Cairns et al. (1975) reviewed the effects of temperature on the toxicity of a variety of substances to aquatic animals and microorganisms and noted that toxicity usually increases with increasing temperature; however several exceptions were noted and their review did not include hydrocarbons. Mayer and Ellersieck (1986) tabulated LC_{50} data for 410 chemicals to 66 species of freshwater animals. Their primary interests were pesticides but their data include a few hydrocarbons (benzene, toluene, xylene). They reviewed earlier studies of the effect of temperature on toxicology with freshwater organisms and noted that toxicity usually increases as temperature increases. The bases for different toxicity at different temperatures have been suggested to include differing respiration rates, differing rates of uptake and elimination, and differing rates of metabolic detoxification. However, the toxicity of a few compounds is greater at lower temperatures. Mayer and Ellersieck (1986) tested the relationship between LC_{50} and temperature using covariance methods for 48 chemicals in 90 temperature tests and found that slopes differed significantly from zero in 26 tests with 19 chemicals. Slopes were negative in 23 tests, meaning that the LC_{50} declined as the temperature increased. That is, the chemicals were more toxic at higher temperatures. Three compounds with significant positive slopes were the pesticides DDT, methoxychlor and dimethrin. The three hydrocarbons tested, benzene, toluene and xylene had no consistent relationship between temperature and toxicity with the species and test conditions used.

Other studies (e.g., Korn et al., 1979) have indicated that the effects of oil pollution would be more severe in the Arctic than in other regions due to the effects of Arctic climatic conditions on the physical state of the oil, on actions to combat oil spills, and on the biological characteristics of many Arctic species that determine the rate of recovery of the populations. Korn et al. (1979) determined the toxicity of toluene, naphthalene and the WSF of Cook Inlet crude oil to pink salmon fry at 4, 8, and 12 °C. The differences in toxicity between temperatures were small and generally not significant. When the tests used shrimp (*Eualus* spp. and *Pandalus goniurus*) toxicity was either unchanged or slightly greater in the warmer water. Larvae of the cold-water species, Tanner crab (*Chionoecetes bairdi*) were slightly less sensitive to WSF of Alaska North Slope crude oil than larvae of two standard warm-water test species, the saltwater mysid (*Mysidopsis bahia*) and the fish *Menidia beryllina*, tested at 7 and 25 °C respectively (Perkins et al., 2005).

Maher (1982) prepared WSFs from a crude oil (Arabian) and a lubricating oil (Macomba 82) and reported higher levels of aromatic hydrocarbons in the WSFs prepared at the lower temperature. Similarly, Rice et al. (1975) prepared WSFs of Prudhoe Bay crude oil in freshwater and in seawater and found higher levels of hydrocarbons in seawater. WSFs of Alaska North Slope crude oil showed higher levels of BTEX (benzene, toluene, ethylbenzene and xylene) when prepared at 7 °C rather than at 25 °C (Perkins et al., 2005).

There is a serious lack of toxicology data for hydrocarbons under cold conditions using cold-adapted, Arctic animals.

5.3.4.3. Chemicals tested for toxicity

5.3.4.3.1. Oils and hydrocarbons

Many reviews of environmental issues associated with the oil and gas industry and with the toxicology of hydrocarbons have been published (Malins, 1977; Whipple et al., 1981; Percy and Wells, 1984; Teal and Howarth, 1984; U.S. National Research Council, 1985, 2003; Engelhardt, 1985; Samiullah, 1985; Engelhardt et al., 1989; Patin, 1999; Neff, 2002; Albers, 2003; Armsworthy et al., 2005a; Ikävalko, 2005). Whole symposia have been devoted to the effects of individual, large, northern spills (e.g., Wells et al., 1995; Rice et al., 1996 [*Exxon Valdez*]; Davies and Topping, 1997 [*Braer*]). It is often difficult to compare different toxicology studies with oils and hydrocarbons because there are so many different pure hydrocarbons, so many different crude and refined products, and so many test species, life stages, exposure conditions, and analytical procedures. For example, the PAH group contains many compounds and variously substituted alkyl homologues of them. Scannell et al. (2005) noted that the alkylated homologues of PAHs are usually more toxic to aquatic organisms than the parent, unsubstituted hydrocarbons. In spite of extensive literature on the toxicology of oils, a number of aspects remain for further investigation. Eggen et al. (2004) outlined several 'challenges' that remain in ecotoxicology in general and that apply to the ecotoxicology of oil: low concentrations of pollutants and long exposure times (chronic effects), multiple effects by single pollutants, complex mixtures of pollutants, multiple stressors, and ecosystem complexity.

Several investigators have worked with individual hydrocarbons rather than the mixtures present in oils. The advantage of using a single, pure compound is that toxicity can be attributed unambiguously to that compound whereas work with mixtures can attribute the toxicity to the mixture only, not to individual components. For example, the toxicities of several individual low-boiling hydrocarbons were reported by Pickering and Henderson (1966) using small fish of temperate habitat (fathead minnow; goldfish, *Carassius auratus*; bluegill; guppy, *Lebistes reticulatus*). The toxicities were in the low mg/L range: benzene 22–37 mg/L; toluene 24–59 mg/L; ethylbenzene 32–97 mg/L; xylenes (mixed) 21–37 mg/L; styrene 25–75 mg/L; and isoprene 43–240 mg/L. The U.S. National Academy review in 1985 (U.S. National Research Council, 1985) provided a Figure summarizing the acute toxicity of a number of aromatic hydrocarbons to marine invertebrates and fish (Figure 5.7). It is apparent from this Figure that the acute toxicity of compounds with a single aromatic ring (benzene and alkylated benzenes) is generally lower (i.e. the LC_{50} values are higher) than that of compounds with two or more aromatic rings

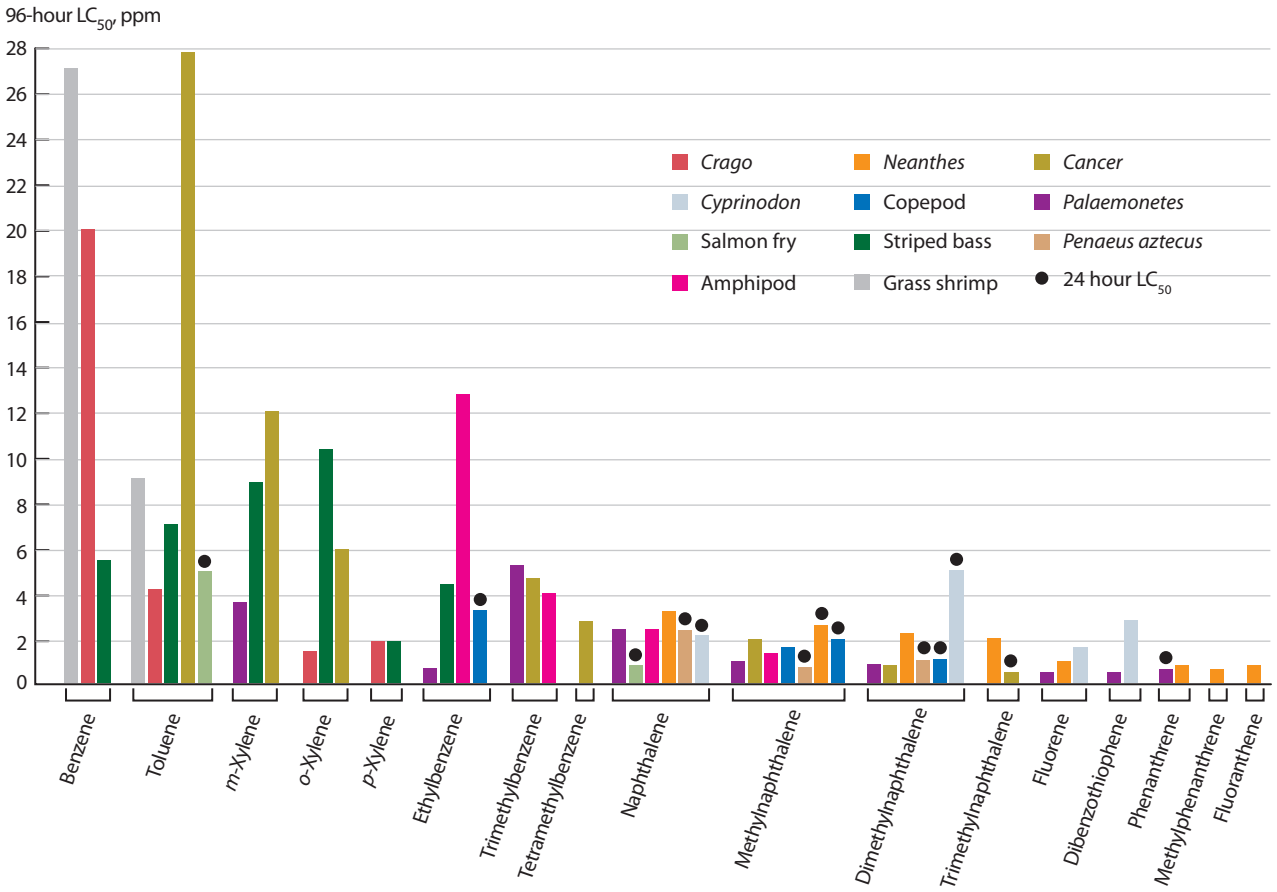

Figure 5.7. Acute toxicity of several groups of low-boiling aromatic hydrocarbons to marine macroinvertebrates and fish (U.S. National Research Council, 1985).

(naphthalenes and substituted naphthalenes, fluorine, etc.). Acute toxicity values for several hydrocarbons to diverse species were tabulated by Frost et al. (2002). In spite of the lower toxicity of the low-boiling hydrocarbons, they are considered to be responsible for most of the acute toxicity of crude and refined oils because of their relatively high concentrations in WSF preparations. The higher-molecular weight PAHs are usually not found in aqueous extracts of oils and are considered to make little contribution to acute toxicity. However, the low concentrations of PAHs cannot be excluded rigorously as contributors to acute toxicity and they are responsible for chronic and sub-lethal toxicity.

Many species of marine organisms have been used to determine the toxicity of different crude and refined petroleum products. Craddock (1977) listed approximately 60 pages of acute toxicity results with coelenterates, echinoderms, molluscs, annelids, arthropods, and chordates (including species not found in the Arctic). Generally, the larvae and eggs of fish were the organisms most sensitive to soluble hydrocarbons; lethal concentrations of these compounds were in the range of 0.1–1.0 ppm (mg/L) (Table 5.2). When the test oil was No. 2 fuel oil, the range for larvae and eggs was slightly wider at 0.1–4 mg/L and for crude oil it was wider still at 0.1–100 mg/L. Older fish and crustacea were usually more tolerant, in the range 5–50 mg/L soluble hydrocarbons. Gastropods and bivalves were the most tolerant with lethal concentrations of soluble hydrocarbons in the range 10–100 mg/L and 5–500 mg/L, respectively. Seasonal differences can be an additional source of variation. Anderson et al. (1987) measured the toxicity of several preparations of Prudhoe

Table 5.2. Ranges of the acute toxicity of petroleum to marine animals (Craddock, 1977).

Organism	Lethal concentration, ppm, mg/L
Soluble hydrocarbons	
Finfish	5 – 50
Larvae and eggs	0.1 – 1.0
Pelagic crustacea	1 – 10
Benthic crustacea	1 – 10
Gastropods	10 – 100
Bivalves	5 – 500
Other benthic invertebrates	1 – 10
No. 2 fuel oil	
Finfish	550
Larvae and eggs	0.1 – 4.0
Pelagic crustacea	5 – 50
Benthic crustacea	5 – 50
Gastropods	50 – 500
Bivalves	30000 – 40000
Other benthic invertebrates	5 – 50
Fresh crude oil	
Finfish	88 – 18000
Larvae and eggs	0.1 – 100
Pelagic crustacea	100 – 40000
Benthic crustacea	56
Gastropods	?
Bivalves	1000 – 100000
Other benthic invertebrates	100 – 6100
Gasoline	
Finfish	91
Diesel fuel	
Finfish	240 – 420
Waste oil	
Finfish	1700
Larvae and eggs	1 – >25
Pelagic crustacea	15 – >50
Residual oils	
Finfish	2000 – 10000

Bay crude oil to shrimp (*Pandalus danae*) in summer and in autumn/winter and found the shrimp more sensitive in summer; the same experiment with sand lance showed no seasonal difference. As expected, soluble hydrocarbons were generally more toxic than refined fuel oils or crude oil.

Vandermeulen (1987) compiled a summary of the acute toxicity of several hydrocarbons and oils to freshwater organisms. Some of the results for northern species are shown in Table 5.3 along with a few additional results. While there are a few unusual values, the LC_{50} values for benzene were in the range 5–25 mg/L. Toluene and ethylbenzene were somewhat less toxic but naphthalene was more toxic with LC_{50} values below 10 mg/L. The samples of crude oil were toxic mostly in the range 1–10 mg/L. Moles et al. (1979) determined the toxicity of benzene and Prudhoe Bay crude oil WSF to several species of fish from Alaska (Table 5.3) and found TLm values for dolly varden (*Salvelinus malma*), chinook salmon (*Oncorhynchus tschawytscha*), coho salmon, pink salmon, sockeye salmon (*Oncorhynchus nerka*), Arctic char, Arctic grayling (*Thymallus arcticus*), three-spine stickleback (*Gasterosteus aculeatus*), and slimy sculpin (*Cottus cognatus*). Excluding the sticklebacks, the 96hr TLm values for Prudhoe Bay crude oil WSF for these species were in the range 2.75–6.44 mg/L (when measured as total aromatics by gas chromatography, or 1.25–3.00 mg/L when measured by IR). The sticklebacks were more tolerant with a TLm of >10.45 mg/L (or >6.89 when measured by IR). The TLm values for benzene for these fish species were in the range 11.73–24.83 μL/L.

Scannell et al. (2005) recently reviewed the published literature on the toxicity of PAHs to aquatic organisms as part of a review of environmental standards for PAHs for the State of Alaska. The authors found that acute toxicity values for freshwater organisms (mostly LC_{50} and EC_{50} [the concentration that produces 50% of the effect]) ranged from 1.6 μg/L (rainbow trout, naphthalene) to 325 000 μg/L (*Daphnia magna*, fluoranthene) with a median acute toxicity value for freshwater organisms of 1865 μg/L (of 126 values found). The lowest chronic toxicity value for freshwater organisms was cited as a range from 0.8 to >30 μg/L (rainbow trout). The median chronic toxicity value for freshwater organisms was 234.5 μg/L (of 30 values found). The lowest acute toxicity value for a marine organism was 0.23 μg/L for the bivalve *Mulinia lateralis* exposed to pyrene and UV light. The median acute toxicity value for marine organisms was 930 μg/L of 214 values found. The lowest chronic toxicity value for a marine organism was 0.4 μg/L for eggs of Pacific herring exposed to Alaska North Slope crude. The median value for chronic toxicity of the 39 values found was 92 μg/L.

There is much consistency among the various reviews assembled over the past three decades. If there is any trend it is probably that growth in the body of research available has revealed toxic effects at lower concentrations than were evident in earlier literature. Several of the values found by Scannell et al. (2005) fall in the low μg/L range, while many early results were in the low mg/L range. This is probably because investigators have worked with sensitive species and have identified toxic endpoints more sensitive than acute death. Methods of analysis for hydrocarbons have improved greatly and more recent work often gives improved characterization of the exposure mixtures. This latter point may help to explain some of the variation among toxicity studies. Differing mixtures of hydrocarbons may yield a similar expression of concentration in spite of differences in composition and toxicity. Alternatively, similar mixtures of hydrocarbons may yield different expressions of concentration when analyzed by different methods or when compared against different analytical standards.

Seemingly extraneous features of the experimental setup can sometimes alter the results. For example, studies have found that the presence of *Laminaria* reduces the toxicity of oils and drilling fluids to other marine organisms. Kosheleva et al. (1997), citing work by Matishov et al. (2003) and Muraveiko et al. (1994), tabulated LC_{50} values for gas condensate, Prirazlomnyi crude oil, and several drilling fluids with and without Laminarians present. This effect was hypothesized to be caused by accelerated oxidation of oil products and by their partial adsorption by the *Laminaria* fronds.

5.3.4.3.1.1. Produced water

Neff (1987) reviewed 108 results from toxicity testing of produced water with 22 species of marine and estuarine animals and found that 11.1% of results fell in the range 1000 to 9999 ppm and that the remaining 88.9% had LC_{50} values exceeding 10 000 ppm. Holdway (2002) tabulated acute toxicity values for produced water from temperate and tropical areas; EC_{50} or LC_{50} values ranged from ~6 to 59% produced water for several marine organisms and chronic toxicity endpoints were in the low ppm range with some less than 1 ppm. Brendehaug et al. (1992) provided toxicity test results with produced water from *Gullfaks* and *Statfjord* fields in the Norwegian sector of the North Sea using the diatom *Skeletonema costatum*, the brine shrimp *Artemia salina*, and *Photobacterium phosphoreum* in the Microtox test. The 24-hour mortality of undiluted Gullfaks produced water was in the range 0–35.7% for *Artemia*; the EC_{50} for *Skeletonema* was in the range 4.5–67.6% produced water and that for *Photobacterium* was 5.7–19.2%. Results with produced water from *Statfjord* were similar. All the samples were relatively non-toxic and would be expected to exert effects only very close to a source where minimal dilution had taken place.

In terms of hydrocarbons in produced water, concentrations have been reported by Berry and Wells (2005) to exceed the NOEC (No Observable Effect Concentration) for benzene, naphthalene and phenanthrene. While there is considerable variability not only among fields but also among wells within a field, the benzene levels in produced water were reported by Berry and Wells (2005) to range from 80 to 4300 μg/L while the NOEC for benzene was reported to be 170 μg/L for the crab *Cancer magister* (Frost et al., 2002). In the worst case therefore, the benzene alone in undiluted produced water could exceed its NOEC. Similarly with naphthalene the range found in produced water is 5.3–1000 μg/L while the NOEC is as low as 21 μg/L (Frost et al., 2002). The phenanthrene concentrations in produced water are lower at 0–30 μg/L but the most sensitive NOEC is also lower at only 1.5 μg/L (Frost et al., 2002). These Figures suggest that dilution by a factor greater than 10-fold would be required to reduce concentrations below those that might result in observable effects in the most sensitive animals. Since these data are available for only a few hydrocarbons, some refinement of the dilutions required can be anticipated as additional data become available. Several mixing models were used and these showed that dilution of 1000-fold or greater would be expected within 150 m of the source. It is not clear whether diluting to render harmless the compound requiring most dilution would have the same effect for the whole mixture. Berry and Wells (2004) applied a joint toxicity model for benzene and naphthalene and calculated that the risk quotient (the ratio of concentration in a mixture to the NOEC) was reduced essentially to zero within 20 m of the source. These observations suggest that produced water from wells at sea constitutes a risk of toxicity to marine organisms,

Table 5.3. Acute toxicity of hydrocarbons and oils to freshwater fish (adapted from Vandermeulen, 1987).

Toxicant	Static or Flow	LC_{50} (96hr LC_{50} unless stated otherwise)	Organisms tested	Source
Benzene (WSF)	F	5.3 mg/L	Rainbow trout juvenile	DeGraeve et al., 1982
	F	15.1 mg/L (LC_{30})	Fathead minnow adult	DeGraeve et al., 1982
	S	11.96 µL/L (96hr TLm)	Dolly varden juvenile	Moles et al., 1979
	S	11.73 µL/L (96hr TLm)	Chinook salmon juvenile	Moles et al., 1979
	S	14.09 µL/L (96hr TLm)	Coho salmon juvenile	Moles et al., 1979
	S	14.71 µL/L (96hr TLm)	Arctic grayling juvenile	Moles et al., 1979
	S	15.41 µL/L (96hr TLm)	Slimy sculpin juvenile	Moles et al., 1979
	S	24.83 µL/L (96hr TLm)	Three-spine stickleback adult	Moles et al., 1979
Toluene (WSF)	F	55–72 mg/L	Fathead minnow embryo	Devlin et al., 1982
	F	25–36 mg/L	Fathead minnow protolarva	Devlin et al., 1982
	F	18–31 mg/L	Fathead minnow 30 day adult	Devlin et al., 1982
	F	6.3 mg/L	Coho salmon fry	Moles et al., 1981
	S	13 mg/L	Bluegill juvenile	Buccafusco et al., 1981
Ethylbenzene	S	150 mg/L	Bluegill juvenile	Buccafusco et al., 1981
Naphthalene	F	1.6 mg/L	Rainbow trout juvenile	DeGraeve et al., 1982
	F	7.9 mg/L	Fathead minnow adult	DeGraeve et al., 1982
	F	6.1 mg/L	Fathead minnow	Holcombe et al., 1984
	F	1.99 mg/L	Fathead minnow juvenile	Rowe et al., 1983
	F	0.12 mg/L (27 day LC_{50})	Rainbow trout eggs/larvae	Rowe et al., 1983
	F	2.1 mg/L	Coho salmon fry	Moles et al., 1981
Acenaphthene	S	1.7 mg/L (48hr LC_{50})	Bluegill juvenile	Buccafusco et al., 1981
Fluoranthene	S	4.0 mg/L	Bluegill juvenile	Buccafusco et al., 1981
Anthracene	S	12 µg/L (48hr LC_{50})	Bluegill	Bowling et al., 1983
Aromatic mixture (65% toluene)	F	15.2 mg/L (1hr LC_{100})	Rainbow trout	Zbanyszek and Smith, 1984
Crude, mixed blend sweet emulsion	F	8.4 µL/L	Fathead minnow	Hedtke and Puglisi, 1982
Crude, Prudhoe Bay WSF	S	1.25 mg/L (96hr TLm)[a]	Dolly varden juvenile	Moles et al., 1979
	S	1.47 mg/L (96hr TLm)[a]	Chinook salmon juvenile	Moles et al., 1979
	S	1.45 mg/L (96hr TLm)[a]	Coho salmon juvenile	Moles et al., 1979
	S	1.79 mg/L (96hr TLm)[a]	Sockeye salmon juvenile	Moles et al., 1979
	S	2.04 mg/L (96hr TLm)[a]	Arctic grayling juvenile	Moles et al., 1979
	S	2.17 mg/L (96hr TLm)[a]	Arctic char juvenile	Moles et al., 1979
	S	3.0 mg/L (96hr TLm)	Slimy sculpin juvenile	Moles et al., 1979
	S	>6.89 mg/L (96hr TLm)	Threespine stickleback adult	Moles et al., 1979
Crude, Norman Wells WSF	S	10.4–11.6 mg/L	Rainbow trout larvae	Lockhart et al., 1987
Crude, Cook Inlet WSF	F	0.73 mg/L	Coho salmon smolts	Thomas et al., 1989
WSF	F	1.08 mg/L	Coho salmon jacks	Thomas et al., 1989
No. 1 fuel oil Emulsion	F	56.7 µL/L	Fathead minnow	Hedtke and Puglisi, 1982
WSF	S	1.6–1.63 mg/L (48hr LC_{50})	Rainbow trout larvae	Lockhart et al., 1987
Gasoline WSF	S	5.4–6.8 mg/L (48hr LC_{50})	Rainbow trout larvae	Lockhart et al., 1987
No. 2 fuel oil (diesel) WSF	S	1.6 mg/L	Lake whitefish juvenile	Lockhart et al., 2002a
WSF	S	2.43–2.52 mg/L (48hr LC_{50})	Rainbow trout larvae	Lockhart et al., 1987
Emulsion	F	38.6 µL/L	Fathead minnow	Hedtke and Puglisi, 1982
Floating layer	S	160 mL/L	Fathead minnow	Hedtke and Puglisi, 1982
Floating layer	S	48.3 mL/L	Fathead minnow	Hedtke and Puglisi, 1982
Used crankcase oil WSF	F	16.6 mL/L	Fathead minnow	Hedtke and Puglisi, 1982
Floating layer	S	12 mL/L	Fathead minnow	Hedtke and Puglisi, 1982

[a]the toxicity tests were performed on out-migrant fish tested in freshwater.

but a risk that is confined to a volume of water very close to the source. Berry and Wells (2004) used several models to illustrate risk calculations. They calculated the dilution of benzene and naphthalene in produced water discharged near Sable Island, Canada, under various sea scenarios. Their calculations showed that the major route of dilution of toxic material would be rapid advection away from a source and that risks to local organisms would be low.

Planned discharges from wells on land or to freshwaters would be of greater concern because volumes of water available for dilution are smaller and influences of currents and tides are negligible. There are some differences in the toxicity of hydrocarbons between fresh and salt waters but the main difference is probably in opportunity for dilution and advection. However, in most Arctic countries produced water is reinjected and so the only discharges would be accidents that occur only rarely and last for short durations or where releases are permitted by regulatory bodies.

Among the many compounds found in produced water are several alkyl phenols. Interest in these compounds derives from their effects as endocrine-disrupting chemicals. They bind to cellular estrogen receptors in the nuclei of target cells and alter the affinity of the receptors for specific sequences of DNA with the result that DNA functions are altered (Gross et al., 2003). Brendehaug et al. (1992) reported that the sum of phenol to C_7 phenol in *Gullfaks* and *Statfjord* produced water samples ranged from 0.6–9.7 mg/L. Generally these compounds are more soluble than the widely studied nonyl- and octyl phenols but they raise similar questions regarding possible endocrine effects. Myhre et al. (2004) conducted a comprehensive risk assessment for these compounds near several Norwegian drilling operations and concluded that they pose no such risk to the fish. They cited unpublished data from the UK with similar conclusions. Frost et al. (2002) tabulated acute toxicity and NOEC values from published literature for several classes of alkyl phenols. For example, three NOEC values given for unsubstituted phenol fell in the range 100–13 000 µg/L. Two NOEC values were given for C_2 and C_3 phenols; 1500 µg/L for fathead minnow and 100 µg/L for *Daphnia magna*, both in freshwater. The C_4 and C_5 alkyl phenols were more toxic; the NOEC for pentylphenol to goldfish (*Cyprinus carpio*) was only 18 µg/L in chronic exposures using reproductive endpoints. The C_6 and C_7 alkyl phenols were intermediate in toxicity with a 7-day NOEC for Atlantic cod of 190 µg/L using mortality as the endpoint. The most sensitive NOEC reported for C_9 phenol was 6.1 µg/L for rainbow trout for 4-*tert*-octylphenol in 21-day exposures with reproductive endpoints. These data suggest that concentrations in the low µg/L levels would be harmful to marine organisms if exposure was maintained over periods of days to weeks. In view of ongoing discharges of produced water at some sites, long-term exposure is possible for organisms that fail to avoid a plume. The volume of seawater in which these conditions might apply would be quite confined and so the alkylated phenols in produced water appear to pose a risk only to highly localized populations.

5.3.4.3.1.2. Drilling fluids and drill cuttings

Holdway (2002) reported a paucity of acute toxicity data for drilling fluids in temperate and tropical sites. Older oil-based fluids are more toxic than newer water-based or synthetic-base fluids. Engelhardt et al. (1989) edited some 42 papers on the subject of 'Drilling Waste', several of which concerned drilling in the Arctic. However, much of the older literature dealing with oil-based muds is now largely academic because Arctic countries have eliminated discharges of oil-based fluids. There are no current officially sanctioned discharges.

Neff (1987) noted that over 400 toxicity tests had been done with 70 different water-based drilling muds using 62 species of marine animals. Almost 90% of the LC_{50} values were over 10 000 ppm and only two (with copepods) were below 100 ppm. Crawford and Gates (1981) described some sub-lethal effects on embryos of killifish (*Fundulus heteroclitus*) and on sand dollars (*Echinarachnius parma*) following exposures to high concentrations of a lignosulphonate mud containing barium sulphate. Responses observed included rates of heart contractions in killifish (no effect at 100 ppm, severe effect at 10 ppt), embryo hatching in killifish (no effect at 10 ppm, some effect at 100 ppm, severe effect at 1 and 10 ppt), embryonic development of sand dollar (no effect at 100 ppm, delayed development at 1 ppt) and fertilization of sand dollar eggs (no effect at 100 ppm, severe effect at 1 ppt). Drilling wastes, for the most part had little effect on the test animals at concentrations below 10 ppm. Given the dilution available for disposal of these muds, concentrations above 10 ppm (mg/L) are found only in close proximity to sources (Neff, 1987). Osborne and Leeder (1989) identified chronic lethal effects of Canadian Beaufort Sea cuttings (Niptek L-19-A and Adgo G-24 wells) on sand dollars in laboratory exposures. Holdway (2002) cited experiments showing that newer fluids based on esters of fish oils have very low toxicity (LC_{50} to algae *Isochrysis* sp. and larval prawn *Penaeus monodon* greater than 10%).

The past practice of disposal of oil-based drilling muds and cuttings at marine drilling sites posed a serious, although localized, risk to benthic organisms; indeed sometimes no benthic organisms survived at all directly under and within a short distance of a platform (e.g., Davies et al., 1984a). Davies et al. (1984a) described four zones around sites of disposal of oil-based drilling mud cuttings. The innermost zone 0–500 m from the discharge was described as an 'Impoverished and highly modified benthic community beneath and very close to the platform the seabed can consist of cuttings with no benthic fauna'. Muds and drill cuttings are released only during the drilling phase although their ecological effects persist for longer periods. Olsgard and Gray (1995) applied sensitive statistical techniques to drill sites on the Norwegian Shelf where oil-based muds were used and found effects on benthic animals extending out as far as 6 km. Their statistical methods were more sensitive than previously applied techniques like diversity indices and areas affected around sites ranged from 10–100 km^2. Furthermore, examination of sites where production had ceased revealed that the area affected did not begin to contract immediately after production ceased as had been expected but rather continued to increase in size for several years after discharges ceased. Olsgard and Gray (1995) reported that areas affected were less extensive where water-based drilling muds were used. The effects of these releases may not be confined to benthic invertebrates; Atlantic cod and haddock captured close to North Sea production sites showed subtle induction of AHH enzymes but whiting (*Merlangius merlangus*) did not (Davies et al., 1984b). This type of sub-lethal effect on the fish may be attributed to the composition of the mud used. Payne et al. (1985) compared several base oils for their ability to induce enzyme activities (such as EROD) in rainbow trout and found that the oils varied considerably from essentially no potency to potency

similar to diesel. The regulatory elimination of discharges of oil-based muds throughout the Arctic should eliminate any concern over discharges of these materials.

In addition to the questions associated with disposal of insoluble materials on the seabed, there is the question of effects of these materials while still in the water column. Finely divided materials may take long periods to settle to the seabed and some evaluation is appropriate while they remain suspended. An example of an approach to this question is the design by Borseth and Tollefsen (2004) who reported a sensitive biomarker-based approach using cod and mussels suspended near the Troll B platform in the North Sea. In this particular instance, there was evidence that the reference sites had been exposed; however, the approach taken has merit for Arctic sites.

5.3.4.3.2. Mixtures of oil and dispersants

Spilled oil is sometimes dispersed using chemical surfactant products that emulsify the oil into small droplets that behave differently from undispersed slicks. Dispersants are typically proprietary and research literature describes them by trade names rather than by chemical composition. Since these products are marketed for use on oil spills, interest in them concerns their joint action with oil, not in toxicity they may have by themselves. Sprague et al. (1982) provided an extensive review of experience with dispersants.

Foy (1979, 1982) determined the toxicity of mixtures of Prudhoe Bay crude oil and dispersant Corexit 9527 to several species of invertebrates from the Canadian Arctic: amphipods *Anonyx laticoxae*, *Anonyx nugax*, *Boeckosimus* sp., *Boeckosimus edwardsi*, *Gammarus oceanicus*, *Gammarus setosus*, *Onisimus litoralis* (juveniles and adults) and a copepod (*Calanus hyperboreus*) and to one species of fish, juvenile fourhorn sculpin (*Myoxocephalus quadricornis*). The 96-hour LC_{50} values for oil alone were reported as oil concentrations measured by fluorescence spectroscopy and the results for the invertebrates were in the range 32–73 mg/L, with five values reported as >51, >55, >53, >47, and >59 mg/L. Parallel exposures for a 10:1 mixture of oil and Corexit 9527 yielded LC_{50} values of 24–196 mg/L. Similar results were obtained with the sculpins. There was little effect of the dispersant on the toxicity of the oil. However, a number of other studies have indicated that oil dispersed chemically is more toxic than undispersed oil. Linden (1975) compared the toxicity of Venezuelan crude oil to larval Baltic herring (*Clupea harengus membras*) with and without the dispersants Finasol OSR 2 or BP 1100 X and found that the toxicity of oil-dispersant mixtures was greater than the toxicity of oil alone. The same phenomenon is evident in Figure 5.8 in which more rainbow trout were killed by Norman Wells oil dispersed with either Corexit 7664 or Corexit 9600 than with oil alone. Penrose (1982) explained this pattern of results not as a change in the inherent toxicity of the oil but rather as a result of the increased biological availability resulting from increased solubilization of the oil by dispersants. Khan and Payne (2005) reported a higher mortality in Atlantic cod and sculpin (*Myoxocephalus octodecemspinosus*) exposed to a dispersant-WSF mixture than for the dispersant (Corexit 9527) or the WSF alone. The dispersant-WSF mixture also resulted in 5 to 11 times more sub-lethal gill lesions in cod, sculpin and cunner (*Tautogolabrus adspersus*) (Khan and Payne, 2005). Dispersion of several oils with Corexit EC9500 (a hydrocarbon-based reformulation of water-based Corexit 9527) enhanced the availability of hydrocarbons as indicated by induction of cytochrome P4501A enzymatic activities in young rainbow trout (Ramachandran et al., 2004). In addition to laboratory experiments, a major field experiment was conducted to assess the use of a dispersant under Arctic conditions, namely the Baffin Island Oil Spill program which is discussed in section 5.3.11.1.2.

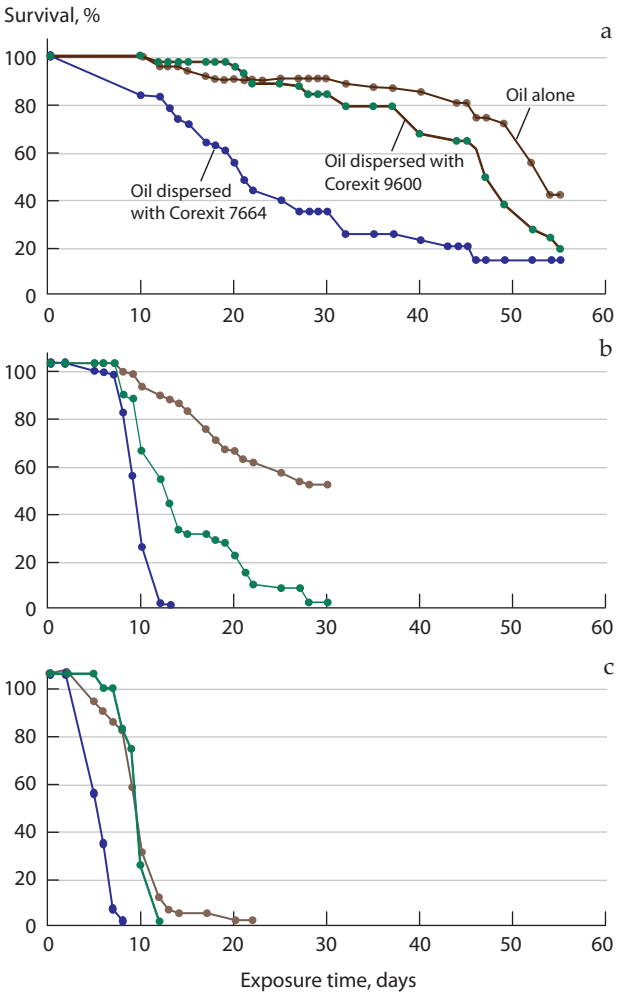

Figure 5.8. Chronic toxicity of *Norman Wells* crude oil alone and *Norman Wells* crude oil dispersed with either Corexit 7664 or Corexit 9600 to young rainbow trout at 10 °C (Lockhart et al., 1996). (a) 30 µL/L mixing ratio or 0.15 mg/L measured; (b) 90 µL/L mixing ratio or 0.39 mg/L measured; and (c) 300 µL/L mixing ratio or 1.51 mg/L measured.

The decision to use dispersants will be difficult in any spill circumstance. The most important objectives will probably drive the decision. If the objective is the protection of organisms affected by an oil slick on the surface, like seabirds, then adding a dispersant may be helpful. However, that benefit is likely to be achieved at the expense of the organisms in the water column since they may experience more serious exposure as a result of dispersion.

5.3.4.4. The Beckman Microtox® bioassay system

The Beckman Microtox® bioassay system has been applied widely as a rapid, sensitive and low-cost method of obtaining toxicity data that usually correlate well with more conventional and more costly toxicity test procedures. The system is based on the emission of light by the bacterium *Photobacterium phosphoreum* in cultures treated with the toxicant of interest. The light emitted by the test cultures is compared with that emitted by untreated controls and generally expressed as the concentration of toxicant required to suppress light emission to half that of controls.

Table 5.4. Toxicity of some hydrocarbons as indicated by the Microtox® test. Each number represents the concentration of compound that results in a 50% reduction in light emission after 5, 10 or 15 minutes relative to untreated controls. The inclusion of *p,p´*-DDT is for comparison with another compound commonly found in environmental samples. Data from the Computox toxicity database, Environment Canada, 1995.

	5 min, mg/L	15 min, mg/L	30 min, mg/L
Acenaphthene		0.74	
Acenaphthylene	0.23	0.24	0.28
Acridine	6.66	6.97	7.47
Aniline	64.4	69	70.6
Anthracene		33.4	
Benzene			103
Benzo[*a*]pyrene			8.11
Chrysene		1.49	
Cyclohexane	199		226
Cyclooctane	1.55	1.78	2.14
p,p'-DDT	7.41	6.75	6.16
Dibenzofuran	0.79	0.86	1.09
Dibenzothiophene	0.1	0.11	0.12
Ethylbenzene	6.55	7.69	9.68
1-Ethylnaphthalene	0.15	0.19	0.24
Fluorene		3.23	
n-Hexane	104	104	104
2-Hydroxyfluorene	0.42	0.5	0.68
1-Methylnaphthalene	0.3	0.4	0.5
2-Methylnaphthalene	0.3	0.3	0.3
2-Methylphenanthrene	0.4	0.3	0.3
Naphthalene	0.81	0.91	0.93
1-Naphthol	2.23	2.34	2.81
2-Naphthol	0.22	0.24	0.27
n-Octane		726	
1-Octanol	5.93	7.93	7.26
Phenanthrene	0.042	0.049	0.073
Toluene	17	19.7	19.7
m-Xylene	2.61	3.36	7.18
Xylenes	16.1		

Table 5.4 lists the Microtox® results for some hydrocarbons and their oxidation products and for *p,p´*-DDT, an organochlorine insecticide, for comparison. The results, not surprisingly, vary over a wide range from 0.042 mg/L for phenanthrene to 726 mg/L for *n*-octane. Over a third of the compounds affected these bacteria at concentrations of <1 mg/L; about a third affected them in the range 1–10 mg/L; and somewhat less than a third affected them at concentrations >10 mg/L. Most of the hydrocarbons were more toxic in the Microtox® test than DDT. The Microtox® results are usually consistent with the results from several taxa shown in Tables 5.2 and 5.3 and confirm the toxicity of relatively low-boiling aromatic hydrocarbons as being generally within the range of ~0.1–15 mg/L. When Microtox® was applied to sediments collected in the vicinity of the *Exxon Valdez* spill it was rejected as a monitoring technique because it failed to show a relationship between toxicity and hydrocarbon concentrations measured in sediments amended with oil (Wolfe et al., 1996). The reason for this apparent insensitivity was suggested to be the presence of 'higher molecular weight constituents of crude oil (perhaps the alphaltene-polar components)'. It was hypothesized that these components, when present at high concentrations in a relatively fresh, unweathered form, may mask the bioavailability and toxicity of PAHs in the oil, at least as measured by Microtox®.

5.3.4.5. Algae and vascular plants

Oils have been used as herbicides to help control terrestrial plants for decades. Early work indicated that the mode of herbicidal action was by physical insertion of hydrocarbons into the lipid structure of cell membranes so that semi-permeability was destroyed and the membranes leaked (van Overbeek and Blondeau, 1954; Baker, 1970). Acute toxicity is difficult to measure with aquatic plants because simple observation does not always reveal whether a given plant is dead or alive. More often, toxicity to plants is measured as the amount of oil or hydrocarbon that reduces some variable function (e.g., growth, photosynthesis) by some stated proportion, often 50%. This is often referred to as an EC_{50} (the concentration that produces 50% of the effect) instead of an LC_{50} (the concentration causing 50% mortality).

Hsiao (1978) determined the effects of several crude oils on laboratory cultures of Arctic marine phytoplankton. Several crude oils (10 mg/L of Atkinson Point, Norman Wells, Pembina, or Venezuela) were added directly to culture flasks and incubated at 0, 5, or 10 °C for ten days. The algae tested were the green flagellate *Chlamydomonas pulsatilla* Wohlenweber, and the diatoms *Chaetoceros septentrionalis* Oestrup, *Navicula bahusiensis* Grunow, and *Nitzschia delicatissima* Cleve, all isolated and cultured from the southern Beaufort Sea. The effects on the survival of different species were quite different (Figure 5.9). Survival of *Chlamydomonas* was reduced somewhat by Day 2 by all the oils but then recovered to near initial values. The diatoms suffered reduced survival with no indication of recovery. *Chlamydomonas, Navicula*, and *Nitzschia* generally suffered less growth inhibition at 0 °C than at the warmer temperatures but *Chaetoceros* showed greater growth inhibition by oils at the low temperature. The authors hypothesized that an oil spill in the Arctic, especially a spill under ice, could result in considerable change in the composition of algal species. In contrast, Hanna et al. (1975) observed little or no effect of Norman Wells oil on freshwater algal populations in experimental tubes installed in a lake near Norman Wells, NWT, Canada.

Kauss and Hutchinson (1975) described the effect of WSFs of several oils on cell numbers of the freshwater alga *Chlorella vulgaris* in laboratory cultures. They tested WSFs from seven Canadian crude oils and from outboard motor oil. The oils differed in their effects on cell density of *Chlorella* with the least toxic reducing cell numbers to 95% of controls and the most toxic reducing them to 59% of controls after two days of exposure. All the WSFs appeared to lose most or all of their toxicity to the algae after eight days and some actually enhanced cell numbers by then. The reduction in toxicity with aging of oil was due to the evaporation of volatile components from the test flasks. The algae were exposed to several pure, low-boiling aromatic components, namely benzene, toluene, *o*-xylene and naphthalene. Benzene reduced cell numbers dramatically at the high concentrations of 1000 and 1744 mg/L. Toluene was more toxic with moderate cell reductions at 250 mg/L and large reductions at 505 mg/L. *o*-Xylene was more toxic still with cell reductions at 100 and 171 mg/L. Naphthalene resulted in smaller but dose-dependent effects on cell numbers at concentrations in the 3–30 mg/L range. The concentrations required to reduce *Chlorella* numbers by 50% were reported to be: benzene, 525 mg/L; toluene, 245 mg/L; *o*-xylene, 55 mg/L, and naphthalene 33 mg/L. Such high concentrations are helpful in establishing the relative potency of different hydrocarbons but it seems unlikely that they would be reached or maintained for very long under environmental conditions.

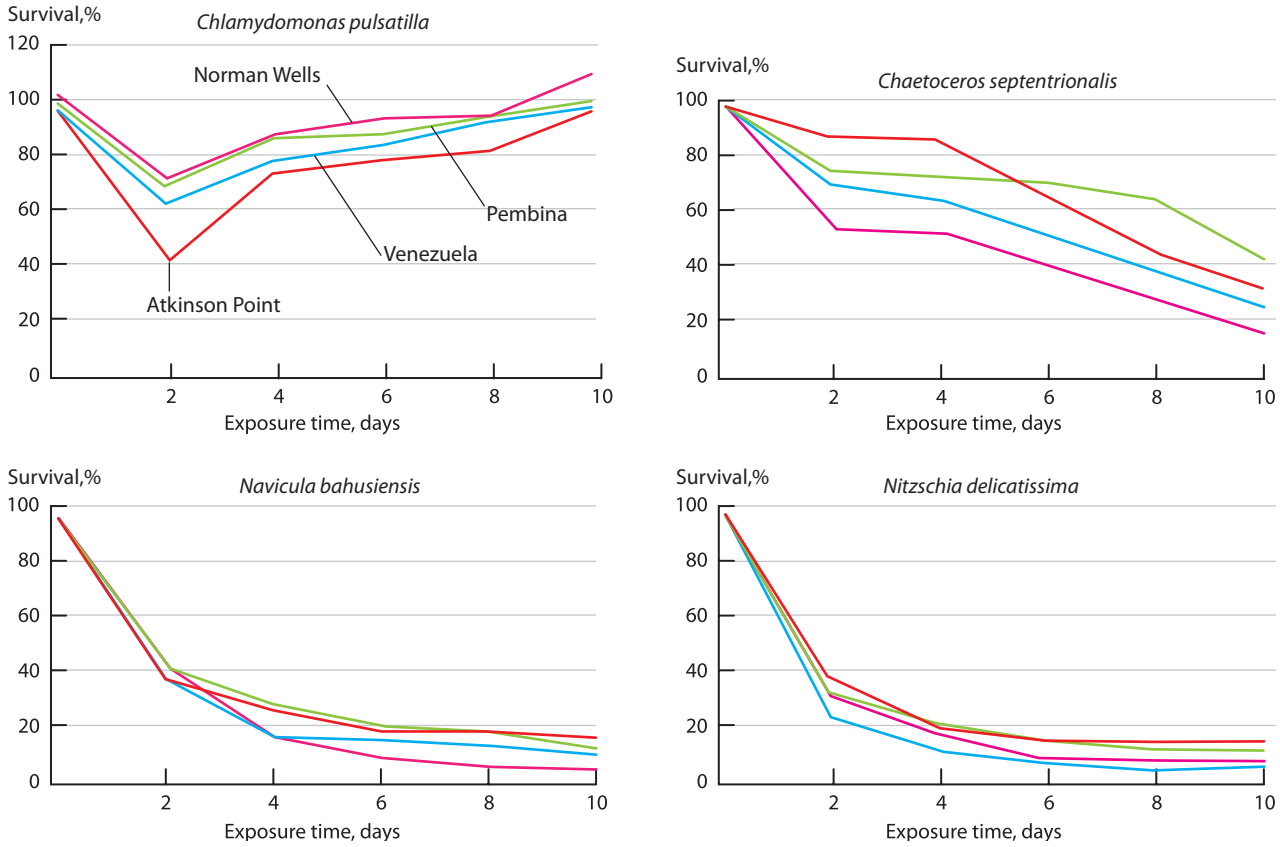

Figure 5.9. Survival of Arctic marine phytoplankton following exposure to crude oils at 10 mg/L (mixing ratio) in the laboratory at 10 °C (Hsiao, 1978).

Continuous exposures of a mixed culture of marine algae *Halosphaera viridis* and *Rhizosolenia* sp. (collected from Ura Bay, Russia) to gas condensate were described by Kosheleva et al. (1997). Condensate concentrations of 0.01, 0.1, 1, 10, and 100 mg/L were used over an exposure period of 31 days. A decrease in photosynthetic activity was noted on the sixth day in concentrations of condensate from 0.1 to 100 mg/L. Thereafter, photosynthesis oscillated and approached control values on the fifteenth day. These fluctuations in intensity of photosynthesis were described as natural and typical of such experiments (Veselovski and Veselova, 1990; Novikov, 1992). However, a toxic effect of gas condensate in concentrations of 0.1–100 mg/L was evident on the twentieth day and remained almost to the end of the experiment. Microscopic analysis of cells in the mixed culture suggested that the diatom *Rhizosolenia* sp. was more sensitive than the green-yellow *Halosphaera viridis*. Effects on numbers of cells in the cultures were similar but this response was less sensitive than the photosynthetic rate. Based on these experiments, gas condensate would not be expected to render an appreciable influence on photosynthesis by these marine phytoplankton at concentrations of 0.01 mg/L or less. Hence 0.01 mg/L might be considered the maximum safe concentration for these organisms (Kosheleva et al., 1997). These results conform well to studies on the effects of crude oil. Patin (1999) noted in long-term experiments (70 days) that primary production decreased by 50% when marine phytoplankton were exposed to oil products in concentrations of 0.05–0.5 mg/L

Similar experiments were conducted with *Rhizosolenia* sp. exposed to a series of concentrations of methanol, which at concentrations of 0.05 and 0.5 mg/L produced no negative impact on growth. These results are similar to results with the freshwater green alga *Scenedesmus obliquus* (Voropayev et al., 1986). According to these data, methanol at concentrations of 40 mg/L and less caused no mortality of algae; but did inhibit algal reproduction at concentrations in the range 1.6–40 mg/L.

Chronic exposures of algae to 'natural gas' were carried out using a mixture comprising approximately 79.4% propane, 12.4% butane and 8.2% ethane. Exposures to this mixture were conducted at concentrations of 0.3, 0.9, 1.9, and 3.6 mg/L for a period of 35 days. Populations of algae in the different concentrations never differed from the control samples by more than 25% (Stroganov, 1971). However, the statistical analysis revealed differences from the control samples in concentrations of gas of 1.9 and 3.6 mg/L (at α = 0.05). When photosynthesis was examined, the only concentration that failed to exert an effect was 0.3 mg/L (Shulyakovski, 1985; Veselovski and Veselova, 1990). In general, natural gas caused effects on the algae in concentrations of 0.9 mg/L and more. This suggests a 'no effect' concentration of natural gas in seawater of about 0.3 mg/L.

Several investigators have noticed a biphasic reaction of marine algae to crude oil and its low-molecular aromatic components. Low concentrations stimulate photosynthesis and high concentrations inhibit it (Nelson-Smith, 1977). The stimulating action of crude oil may be attributed to the presence in it of regulators of growth (Glegg and Koevening, 1974) and of trace quantities of metals (Hsiao et al., 1978), which may act as micro-elements. Dunstan et al. (1975) observed growth enhancement by *Amphidinium carterae* exposed to xylene at concentrations from 100 and 10 000 µg/L with inhibition at higher levels of exposure. The same species showed no enhancement of growth when exposed to benzene or toluene. Another species, *Cricosphaera carterae*, showed enhanced growth when exposed to toluene or xylene but not to benzene. Growth was inhibited in most

species when concentrations reached 100 mg/L. Similarly, Stepanjan (2002) found that low concentrations of crude oil and benzene (<1 mg/L) stimulated the development of sprouts of *Fucus vesiculosus* and gametophytes of *Laminaria saccharina*. The biphasic response of stimulation at low levels and toxicity at higher levels has been observed frequently in toxicology studies and has been termed 'hormesis' (Stebbing, 1982).

Periphyton species seem relatively tolerant of oil. McCart and Denbeste (1987) placed artificial substrates oiled with Prudhoe Bay crude oil in two streams in northern Alaska and examined their colonization by periphyton over periods of 45 and 90 days during the summer. The results indicated little effect of the oiling on the subsequent colonization by several species of periphytic algae.

Miller et al. (1978) monitored populations of algae in small tundra ponds near Barrow, Alaska, following treatments with Prudhoe Bay crude oil. They described a series of potential effects of these treatments on the phytoplankton:

- immediate toxicity by the soluble aromatic fraction;
- prolonged toxicity by persistent products;
- altered physical-chemical conditions below floating oil (e.g., temperature changes);
- enhancement of metabolism due to increased nutrient availability from oil degradation;
- stimulation of nitrogen-fixing algae and bacteria;
- increased leaching of nutrients from vascular macrophyte stands;
- removal of grazing pressure (by toxic effects on zooplankton); and
- possible competitive monopolization of nutrients by oil-degrading heterotrophs.

They found that initial toxic responses, which were usually of short duration, were followed by increases in algal populations. The most predictable response to oil was a change in species composition, particularly the loss of *Rhodomonas* with increasing dominance of Chrysophyta *Uroglena*, Chlorophyta, *Chlamydomonas*, and Pyrrophyta (dinoflagellates). In some oil treatments, a blue-green alga (*Oscillatoria*) emerged as the dominant species. The immediate toxic effect of the oil in the ponds was more evident on the zooplankton and the resulting reduction in grazing pressure offered a plausible explanation for the changes in algal populations. Pond experiments resulted in the elimination of the dominant zooplankton species, *Daphnia middendorffiana*, and the fairy shrimps *Branchinecta paludosa* (O.F. Müller) and *Polyartemiella hazeni* (Murdoch). Neither the zooplankton nor some of the phytoplankton, notably *Rhodomonas* sp., recovered over several years because the ponds were small and isolated from each other such that recolonization was unlikely after a species had been eliminated. A similar experiment was done by treating part of a lake cordoned off with a sea-curtain with different results. The importance of grazing by zooplankton was much reduced in the lake in comparison with the ponds. In bioassay experiments with lake water, photosynthesis was reduced by as much as 50% by oil concentrations as low as 2 ppm. Primary production and chlorophyll were greatly reduced in the oil-treated part of the lake. The effect on algae in the lake could not be explained by changes in zooplankton, and was attributed to other effects of the treatment.

Oil is also toxic to aquatic vascular plants when exposures are high. Bioassays with axenic cultures of duckweed (*Lemna gibba, L. minor, L. perpusilla*) were conducted with WSFs of several oils (South Louisiana, Bunker C, No. 6 fuel oil, raw coal distillate and SRC 11 fuel oil) (King and Coley, 1985). Growth of the duckweeds was measured by counting fronds over a period of eight days. WSFs of Bunker C, Louisiana crude and No. 6 fuel oil had little effect on *L. gibba* or *L. perpusilla*. *Lemna minor* was sensitive to these WSFs but even with this species the maximum growth reduction was only about 30% relative to controls. In contrast, all three species were highly sensitive to WSFs of SRC-11 fuel oil and raw coal distillate. Blundon et al. (1987) treated several species of northern shoreline plants with either Norman Wells oil or oil plus chemical dispersant in laboratory exposures. All plant types were affected by the oil. Longpre et al. (2000) noted that rushes (*Scirpus pungens*) were able to survive, grow, and produce new shoots in sediment oiled experimentally at concentrations typical of spill incidents. Plants grown in sediment treated with low concentrations (controls and oil <4.56 g/kg) survived and grew normally. Those in higher concentrations mostly survived with reduced growth. In the most heavily oiled sediment (72.9 g/kg), most of the plants (87%) were killed.

5.3.4.6. Protozoa

Single-celled protozoans (*Holosticha* sp. and *Euplotes haron*) from the Barents Sea were exposed to gas condensate at 0.1–50 mg/L (emulsified, *Holosticha*) and high concentrations of methanol (25 000–100 000 mg/L, *Euplotes*) over an exposure period of 72 hr in petri cups. Gas condensate at 50 mg/L killed all the animals within 2.5 hr (Kosheleva et al., 1997). Gas condensate at 25 mg/L suppressed cell division, but low concentrations (5 and 10 mg/L) stimulated it, rather like the hormetic response with some algae and low boiling aromatic hydrocarbons (Dunstan et al., 1975). Methanol was much less toxic to these organisms with deaths recorded at concentrations over 25 000 mg/L. No differences from controls were observed at 500 mg/L of methanol. Gas condensate at concentrations below 2.5 mg/L had no observable effects on the animals. The authors noted that this experiment may have been compromised by the development of oil-tolerant microorganisms which may have served as food for the animals. Two Baltic Sea protozoa, infusorians *Euplotes harpa* and *Stylonihia mytilis*, had LC_{50} values to crude oil greater than 30 mg/L (N.B. analytical methodology not clear, probably a mixing ratio).

Rogerson et al. (1983) used gas-tight syringes as exposure chambers to determine the toxicity of 17 hydrocarbons to the ciliate protozoan *Tetrahymena elliotti*. These syringes prevented the commonly observed loss of volatile components during exposure. The authors found that the concentration of hydrocarbon lethal to this species under these conditions was related inversely to the solubility of the hydrocarbons (r = +0.915). That is, the most soluble compounds had the highest lethal concentrations, or the lowest toxicity. Expressed in terms of the octanol-water partition coefficient (K_{ow}), the lethal concentrations were described as:

$$y = -1.145\ x + 6.12\ (r=0.903)$$

where y is the $\log_{10}$ hydrocarbon concentration lethal to 100% of organisms (mmol/m^3) and x is $\log_{10} K_{ow}$. The compounds with highest values for K_{ow} were lethal at the lowest concentrations. The bioconcentration of several hydrocarbons was shown by McElroy et al. (1989) to be related to

the K_{ow} in *Mytilus edulis.* In these examples, both bioconcentration and toxicity were related to the tendency to partition into lipids. The toxicity results are similar to those for green algae reported by Hutchinson et al. (1979).

5.3.4.7. Invertebrates

Oils, both crude and refined are generally toxic to aquatic invertebrates. Percy and Mullin (1975) tested several crude oils for toxicity to the Arctic amphipod *Onisimus affinis* by preparing the oil/water mixtures at either 8 or 22 °C and then exposing the animals at either 0 or 8 °C for 96 hr. Mortality was usually higher when the exposures were carried out at the warmer temperature. O'Brien (1978) exposed three species of zooplankton from Alaska (*Daphnia middendorffiana, Branchionecta paladosa* and *Heterocope septentrionalis*) to Prudhoe Bay crude oil. Oil was added to aquaria in which submerged, screened cylinders contained the test animals. Effectively this procedure exposed the animals to a mixture much like the water-soluble fraction but without the insoluble fraction having been removed from the aquaria. The concentrations were expressed in terms of the original amount of oil added and so appear higher than they would actually have been. *Daphnia middendorffiana* were killed by all the concentrations within five days. The lowest concentration used was nominally 0.06 mL/L. The authors calculated a lower concentration of 7.4–11 mg/L on the assumption that 14–21% of the oil would have dissolved. The experiment was repeated with a lower dosage that proved to be non-toxic. A similar experiment with the fairy shrimp *Branchionecta paladosa* produced similar results. Linden (1976a) plotted survival, swimming behaviour, and morbidity in amphipods (*Gammarus oceanicus*) over periods of up to 49–62 days and found that effects became more severe with increasing exposure time. Koroleva (1977) investigated the influence of diesel oil on zooplankton (*Calanoida, Harpacticoida*) using organisms from Kandalaksha Bay (White Sea). The copepods were killed by diesel concentrations above 1 mg/L and mortality increased with duration of exposure. A concentration of 10 mg/L was lethal after a two-day exposure (Kosheleva et al., 1997).

Within a group of closely related compounds, small differences in chemical structure have significant effects on toxicity. Ott et al. (1978) found that the toxicity of naphthalene and three methylated derivatives to the estuarine copepod *Eurytemora affinis* varied depending on the number of methyl groups. Over the range of zero to three methyl substituents, the LC_{50} declined from 3.8 mg/L for naphthalene to 0.3 mg/L for trimethylnaphthalene.

Busdosh and Atlas (1977) exposed amphipods (*Gammarus saddachi* Sexton and *Boeckosimus* [=*Onisimus*] *affinis* Hansen) to Prudhoe Bay crude oil and fractions of it in an experimental procedure that allowed the test organisms either to contact a slick of oil or to be prevented from doing so. The authors expressed toxicity in terms of the exposure time required to kill half the animals and found that, for example, death occurred within four days for *Boeckosimus affinis* that could contact a slick but required eight days for those animals that could not. All members of both groups were killed after ten days of exposure. Surprisingly, the paraffinic fraction was more toxic to *B. affinis* than the aromatic fraction; the asphaltic fraction was essentially non-toxic. Subsequently, Busdosh (1981) carried out longer-term exposures of the same amphipod (*Boeckosimus* [*Onisimus*] *affinis*, collected from Elson Lagoon, near Barrow, Alaska) to the WSF of Prudhoe Bay crude oil. Exposures were continued for 16 weeks and the lowest exposure used was one mixed as 1 mg/L. The concentration measured at the start of the exposure to this mixture was 0.2–0.23 mg/L and this fell about 10-fold to 0.017–0.019 mg/L after 14 days. This mixture killed none of the amphipods in 1 week and half of them in about 10 weeks; one quarter survived the entire 16 weeks. Higher mixing ratios (up to 1 part oil to 10 parts water) had relatively little incremental effect on either concentrations or on survival of the amphipods. Busdosh (1981) demonstrated both chronic and acute effects of the WSF of Prudhoe Bay crude oil on the amphipods at concentrations as low as, or lower than, 1 ppm, depending on the duration of exposure. The effects of oil on feeding may have contributed to mortality, given sufficient exposure time. Busdosh demonstrated oil effects on movement of the amphipod, and because movement is directly related to food-finding in an Arctic lagoon, speculated that this effect contributed to mortality.

Russian experiments used the calanoid copepod *Acartia longiremis* to determine the toxicity of gas condensate over periods of 10–30 days (Kosheleva et al., 1997). Similarly, the toxicities of methanol and natural gas were determined using the harpacticoid copepod *Idyaea furcata*. All three toxicants exerted a narcotic effect on the crustaceans. During the initial contact, the animals showed increased activity but this was replaced by decreased activity, stupor and death. Animals not killed often showed recovery from narcosis induced by methanol and natural gas. Methanol was the least toxic (LC_{50} 5500 mg/L); natural gas (LC_{50} 0.7 mg/L) was much more toxic and gas condensate was more toxic still (LC_{50} 0.4 mg/L). The physical form of the toxicant affected the rate at which toxicity occurred; emulsified gas condensate at concentrations of 50, 100, and 250 mg/L caused complete mortality of the copepods in 19 hours, but non-emulsified condensate at higher concentrations of 500 and 1000 mg/L required 24 hours to kill all the copepods. In longer-term tests, all concentrations of gas condensate above 0.01 mg/L resulted in reduced survival relative to untreated controls. With natural gas, only the solution at 0.4 mg/L had no effect on the copepods. These crustaceans were more tolerant to methanol; complete mortality was registered by the end of the test only at a concentration of 10 000 mg/L. Methanol above 100 mg/L elicited behavioural changes in the copepods; the copepods became grouped at the bottom where they were less active than usual, apparently showing a weak narcotic effect.

Different life stages of at least some invertebrate animals have very different sensitivities to oil. Linden (1976a) exposed amphipods (*Gammarus oceanicus*) to mixtures of three different oils in seawater taking the concentrations to be the volumetric mixing ratios. In all cases, the juvenile animals were markedly more sensitive than the adults. For example, the 48-hr LC_{50} for adult amphipods to Venezuelan crude oil was 550 μL/L while that for juveniles was only 0.8 μL/L, a difference of over 500-fold. Comparable large differences were found when the tests were done with No. 1 fuel oil and No. 4 fuel oil. Linden reported that mortality caused by exposure to oils continued after the exposure was stopped by transfer of surviving amphipods to clean seawater. For example, after exposure to Venezuelan crude oil at dosages of 100 and 300 μL/L for 48 hr and transfer to clean seawater, most of the survivors from treated groups died over 30 days while untreated amphipods survived throughout. Similarly, young amphipods *Gammarus finmarchicus* collected from Yura Bay of the Motovski Gulf and acclimated to the laboratory were killed by concentrations of natural gas two to

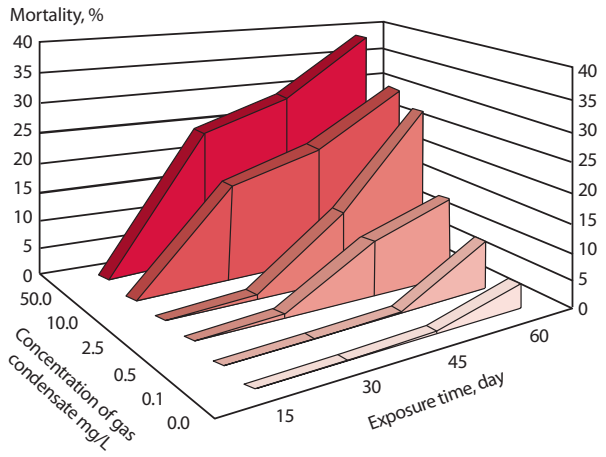

Figure 5.10. Lethality of gas condensate to mature gastropods (*Littorina obtusata*) from Yura Bay, Russia, during laboratory exposures of up to 75 days duration.

three times lower than those in which the adults survived (Kosheleva et al., 1997). These descriptions of differential effects of toxicants on young animals predict that population level effects will depend on the age structure of a community when it is exposed. The first response is likely to be mortality of young amphipods. Other Russian studies have noted that eggs and nauplii of the crustacean *Artemia salina* are relatively insensitive to crude oil (LC_{50} >30 mg/L) but adults are more sensitive (LC_{50} 1 mg/L) (Kosheleva et al., 1997). Effects on amphipods are important ecologically because they comprise one of the largest taxonomic groups of Barents Sea invertebrates and where they dominate the diet of many fish (Bryazgin 1973).

Experimental exposures of mature gastropods *Littorina obtusata* collected from the littoral zone in Yura Bay to gas condensate, natural gas and methanol over periods from 4–75 days were described by Kosheleva et al. (1997). The gastropods appeared relatively resistant to gas condensate and methanol; the LC_{50} for gas condensate was 13 700 mg/L and for methanol 18 800 mg/L. In longer-term exposures, survival of the molluscs during 60-day exposures to methanol and natural gas at concentrations up to 1000 and 7.2 mg/L, respectively, did not differ from the controls. Gas condensate, however, in concentrations exceeding 0.5 mg/L, caused increased mortality with increasing exposure time (Figure 5.10).

5.3.4.8. Fish

Fish are highly sensitive to the presence of oil or aqueous extracts of oil. However, there are differences among species, among life stages within a species, among oils, and among oil preparations. The acute toxicity of low-boiling hydrocarbons or the WSF of different oils is usually in the 0.1–15 mg/L range when the concentrations in the water are maintained for a few days (Tables 5.2 and 5.3; Figure 5.7; Scannell et al., 2005). However, Table 5.2 reveals wide variation in the sensitivity of a single group of organisms to different types of oils and wide variation among the sensitivities of different organisms to a single type of oil. Russian experiments with young Atlantic cod indicated an LC_{50} of 1500 mg/L for gas condensate while that for natural gas was 4 mg/L. Methanol was much less toxic to these fish, with acute toxicity in the range 15 000–20 000 mg/L.

Fish eggs appear to be more resistant to the effects of oil than post-hatch larval stages. Early studies by Rice et al. (1975) found that eggs of pink salmon were several times more resistant to Prudhoe Bay crude oil than alevins or fry. These studies were reported in mixing ratios of oil in water and so the stated exposure values were undoubtedly higher than the animals would have experienced. No eggs were killed by the highest exposure of 3.2 mL/L for 96 hr although hatching success was about half that observed with lower exposures. Longer observation periods often detect responses not evident by the end of conventional exposures. Smith and Cameron (1979) examined the effect of Prudhoe Bay crude oil on Pacific herring and found that the effects sometimes became apparent after exposure had been terminated at the time when eggs began to hatch.

The effects of salinity on toxicity are seldom described because most species occur in either seawater or freshwater but not in both. In cases of species that do occur in both habitats, toxicity appears to be greater in seawater than freshwater. Moles et al. (1979) compared the toxicity of Prudhoe Bay crude oil and benzene to smolts of sockeye salmon, dolly varden char and outmigrants of pink salmon under comparable conditions in both freshwater and seawater. The toxicities of both oil and benzene were consistently greater (TLm levels lower) in seawater by a factor of about two: TLm (LC_{50}) values fell in the range 2.2–8 mg/L in freshwater but 1.1–3.7 mg/L in seawater.

Most of the toxicity statistics reported cover responses over 96 or fewer hours. Many studies have shown, however, that longer exposures result in increased mortality. Moles (1998) compared the 4-day and 28-day toxicities of Cook Inlet crude oil to several species and found that the chronic toxicity was about twice as great as the acute toxicity. Figure 5.8 shows the survival of young rainbow trout over 56 days of exposure to the WSF of Norman Wells oil and it is clear that fish not killed over the conventional 24- to 96-hr exposure times continued to die over several weeks of continuous exposure (Lockhart et al., 1996). Kenaga (1982) considered acute and chronic toxicity results for a range of chemicals including a few hydrocarbons. While some compounds were much more toxic in chronic exposures, the hydrocarbons generally were only slightly more toxic chronically than acutely. For example, fluoranthene was about three times more toxic to the mysid shrimp *Mysidopsis bahia* when exposed over periods described as chronic than over periods described as acute. The issue of long-term exposure to PAHs in sediments has taken on growing interest with long-term studies of the effects of the *Exxon Valdez* spill (e.g., Peterson et al., 2003).

Important experiments were reported by Birtwell et al. (1999) in which marked pink salmon fry were exposed to the WSF of Alaskan North Slope crude oil for ten days. The exposure concentrations were 25–54 µg/L for the low-dosage group or 178–349 µg/L for the high-dosage group. The acute 96-hr LC_{50} for this WSF preparation to these fish was 1–2.8 mg/L. The treated fry and untreated control fish were then released into the Pacific Ocean for subsequent recovery as mature adults two years later. There were no differences in recoveries of treated fish as compared with untreated controls.

5.3.4.9. Marine mammals

There have been a few experimental exposures of marine mammals to oil. Several ringed seals were exposed experimentally to a surface layer of Norman Wells crude oil (Geraci and Smith, 1976). When six seals were exposed to the oil for 24 hr in the Northwest Territories where they had been captured, only transient eye discharge and minor changes

in plasma enzymes were noted. After removal of the seals to clean water they recovered completely within a few days. In contrast to this result, three seals were transported to the University of Guelph in southern Ontario and maintained there for two months prior to a second, similar oil immersion experiment. In this case, all three seals died within 71 minutes of initial exposure. The authors attributed this to a sub-lethal stress syndrome commonly displayed by captive seals. Seals in this stressed state often die with little or no warning and the exposure to oil was sufficient to trigger their deaths. Six harp seal pups in the Gulf of St. Lawrence were coated with Norman Wells crude oil by brushing it onto the hair over their entire bodies. They were coated again the next day with crude oil from Midale, Saskatchewan. No behavioural changes were observed over the 3- or 4-day observation period after oiling and core body temperatures were unchanged. At autopsy, there were no pathological changes that could be related to coating with oil (Geraci and Smith, 1976).

Sea otters were examined extensively following the spill of the *Exxon Valdez*. Ballachey et al. (1994) summarized studies of otters there; 994 otters are known to have been killed because carcasses were recovered. Estes (1991) cited unpublished data from the U.S. Fish and Wildlife Service in Alaska suggesting that only about one carcass is recovered for each five deaths. However, carcass recoveries provide a minimum count of mortality because many are not recovered. They did not offer an estimate of total mortality other than a Figure of 'several thousand'. Garrott et al. (1993) estimated otter mortality to be around 2650 individuals with a very wide confidence interval of 500 to 5000. The total population was estimated to have been 6546 and so the mortality rate must have represented over a third of all otters there. Mulcahy and Ballachey (1994) reported hydrocarbon residues in oiled carcasses of otters and all but one of these otters showed either pulmonary emphysema or gastric ulcerations or both. The potential causes of death included hypothermia due to the loss of insulating capacity of the pelage. Death may also have been due to pulmonary or nervous system dysfunction. In the latter cases, death would have followed quickly after exposure without high accumulation of hydrocarbons in tissues. Aerial survey data from 1993 to 2000 have shown that numbers of otters from most areas have increased; however, otters from a heavily oiled area near Knight Island have not recovered (Bodkin et al., 2002).

When polar bears were oiled experimentally (Engelhardt, 1981; Øritsland et al., 1981) an unexpected response was the reaction of the skin to the oil; it developed an abnormally high temperature. The result of the oiling was significant cold stress in the bears, and a number of the same general responses as have been observed in exposed ruminants (e.g., tremors, vomiting). A number of changes were observed in blood chemistry due to the ingestion of the oil, and urine and bile were identified as the major clearance routes. The authors noted that no effort was made to determine if bears show any tendency to avoid the oil. Two of the three experimentally oiled bears died, and the third would have died if not for extensive veterinary care. Death was caused by failure of the kidneys (acute renal insufficiency) from oil ingestion. However, many other factors probably contributed to death including suppression of lymphoid activity, failing liver function, a disorder in red blood cell formation, and ulcers in the gastrointestinal tract. Polar bears are curious animals and are attracted to new smells and unfamiliar objects; some have died after consuming ethylene glycol, and rhodamine B used as a runway marker (Derocher and Stirling, 1991). The effects of human disturbance on denning behaviour of females in the autumn were investigated by Amstrup (1993) in the Arctic National Wildlife Refuge in Alaska. Female bears were quite tolerant of disturbances by a number of human activities including low-level aircraft, snow vehicles, seismic surveys, oilfield operations and field surveys. When females were disturbed early in denning, they moved elsewhere (as far as 120 km) and made new dens. Disturbance did not affect the production or survival of cubs, their weight, or their skull sizes.

The cetaceans normally found in the Arctic, at least for part of the year, namely beluga (*Delphinapterus leucas*), bowhead (*Balaena mysticetus*) and narwhal (*Monodon monoceros*), use restricted habitat like shallow bays and estuaries for part of the time for feeding and breeding (Würsig, 1990). When in these areas they would be likely to experience prolonged exposure in the event of a spill. The consequences for the whales are largely unknown although observations of whales in temperate locations suggest that these animals are relatively resistant to injury from oil.

5.3.4.10. Seabirds

The most obvious lethal effect of oil on seabirds is from marine oil spills, where hundreds of thousands of seabirds can become coated with oil and die. Seabirds can also be exposed through ingestion of hydrocarbons in food and water, by the preening of feathers and the inhalation of fumes from the spill (Figure 5.1) (Albers, 2003). Very few studies report data for the rates of uptake of hydrocarbons from food into birds, or for the inhalation of volatile components from spilled oil, but it is reasonable to assume that the uptake rates of hydrocarbons from the diet are high.

Much research has been conducted on the effects of oiling on seabirds, much of it with a view to finding methods of removing the oil to allow birds to recover. Oil on the feathers of seabirds removes the insulating layer of air in the plumage, causing the loss of thermal balance (Jenssen, 1994). In essence, the loss of heat from the body leads to hypothermia and death. This heat loss is considerably higher when in water, especially cold water, than on land. In addition, coated birds tend to eat and drink less and have lower levels of plasma sodium and chloride levels (Hughes et al., 1990). Diving birds may not be able to submerge properly to feed. Although cleaning oil from coated feathers has been done on a limited number of birds, it is possible that the combination of oiling and handling birds produces stress-related immunosuppression leading to infections and reproductive and behavioural problems (Briggs et al., 1996). Given the isolation of bird colonies in the Arctic, the possibility of cleaning birds (or other animals) within a reasonable period of time after a spill is remote. The extreme nature of the environment and cold surface waters in the Arctic would undoubtedly cause a very high rate of mortality in birds exposed to oil.

The continuing exposure of seabirds to the residue from old spills and new releases of oil from shipping and spillage, and the linkage between continuing exposure with slow recovery of the populations and known biochemical effects (Peterson et al., 2003), indicates that much more information is required in the area of toxicology of oils to birds. After a number of studies on the acute toxicity of oil to seabirds in the 1970s and 1980s, relatively little research was conducted in this area until recently, when new chemical and biological tests provided more insight into the biological effects of exposure.

Less is known about the ingestion of oil as a factor contributing to death of seabirds after spills. Early studies of

feeding single doses of processed oils to adult mallards produced a range of biological effects similar to those observed in ducks exposed to oil pollution in the wild (Hartung and Hunt, 1966). Major differences in the toxicity of individual crude and refined oils were found, probably as a result of the proportion of aromatic compounds in the oils. Sub-lethal effects included lipid pneumonia, fatty livers, oedema, and adrenal cortical hyperplasia. Many of the changes produced in the adult birds were transitory, and declined when dosing stopped (Hartung and Hunt, 1966). These early studies showed that some oils were very low in toxicity to ducks when they were living under optimal conditions but if transferred to suboptimal conditions at low environmental temperatures, the oils became much more toxic. These observations were supported in later studies which showed that birds dosed with oil at normally non-toxic levels (1–3% of the diet), had very high rates of mortality when placed under cold stress (Holmes et al., 1979). The authors suggested that petroleum ingestion acts as a non-specific stress which lowers the ability of the birds to acclimate to changing environmental conditions. These additional stresses that increase the toxicity of oil may be important factors in estimating the effects of exposure to oil in Arctic waters.

It is evident from several studies that the embryo is the most sensitive stage in avian development and that very small amounts of some oils are highly toxic to developing embryos. Hoffman and Albers (1984) evaluated the toxicity of several crude oils and petroleum products and determined that oil used to control road dust was the most toxic of those tested with mallard eggs, followed by South Louisiana crude oil. Prudhoe Bay crude oil was moderately toxic of the oils and products tested. The toxicity was related to the levels of PAHs in the oil, with the oil with the highest toxicity (lethal dose of 0.3 μL per egg) also having the highest proportion of PAHs (Hoffman and Albers, 1984). Other studies by the authors using synthetic mixtures with individual compounds also showed the high embryotoxicity and teratogenicity of the aromatic fraction of Prudhoe Bay crude oil.

Couillard and Leighton (1991a) tested the toxicity of five crude oils and a fuel oil and reported LD_{25} values (a lethal dose to 25% of the exposed eggs) of 3.2 μL per egg for Prudhoe Bay crude oil and an ED_{25} for liver necrosis (estimated dose to cause liver necrosis in 25% of the surviving embryos) of 4 μL per egg. Doses to eggs were kept low in the study to prevent effects due to smothering and interference with gas exchange through the eggshell. A similar study with No. 2 fuel oil showed significant reduction in hatching success in gulls.

A significant observation in the understanding of the exposure of eggs to oil is the transfer of oil from the adults to the eggs. Studies from as far back as the mid-1950s showed that oil on the breast feathers of adult birds can be transferred to their eggs (Figure 5.1) causing mortality in the developing eggs (Hoffman and Albers, 1984). Although oil was applied directly to the breast feathers of adults in early experiments, Albers (1980) showed that incubating adults feeding on water with oil applied to the surface of a nearby water supply at a rate of 100 mL/m^2 would transfer the oil back to the clutch of eggs. Eggs are turned by the parents during incubation and this ensures complete coverage of the eggs with oil. Quantities as low as 2.5 mL of fuel oil applied to the feathers surrounding the brood pouch of laughing gulls (*Larus atricilla*) resulted in the death of 41% of the incubated eggs, relative to 2% in the control group (King and Lefever 1979). Direct application of 20 μL of the fuel oil to the eggs caused 83% mortality. Another study showed that 10–20 μL of No. 2 fuel oil caused significant mortality in the eggs, but only 4–8 days after laying (Lewis and Malecki, 1984). Quantities as high as 100 μL produced little or no effects after midway through the 28-day incubation period, suggesting that the critical time of exposure is a few days after the eggs are laid.

The small amounts of oil required to cause toxic effects in incubating eggs suggests that this may be a very important exposure pathway in areas where adults may be exposed to small amounts of oil on land or water. Through this mechanism, diffuse oil releases from shipping or small pollution sources may have a direct impact on the reproductive success of local waterfowl and seabirds.

5.3.4.11. Sediment toxicology

Most hydrocarbons in spilled oil are highly insoluble in water and partition readily to sediment particles where some can persist for decades or longer. The toxicity associated with sediment contaminated with oil has become a significant area for research, especially with regard to PAHs. Burton et al. (2003) reviewed the methods available for collecting and testing contaminated sediment. Bioassay methodologies are well developed for several freshwater and marine organisms and these, when combined with sediment chemistry, offer powerful tools for the investigation of events that result in contamination of sediment.

Since the major sink for PAHs in the aquatic environment is the sediment and PAHs can remain there for long periods, their biological availability is of great interest. Payne et al. (1988) examined winter flounders after exposure to oil-treated sand for a period of four months from February to June 1984. The levels of hydrocarbons in the oiled sand averaged ~0.06 μg/g in the controls and ranged up to ~590 μg/g in the highest treatment level, with some differences in the mixtures of different PAHs measured. For example, pyrene was the dominant PAH in the control tanks while fluorene and its analogues were dominant in most experimental tanks. The most consistent and sensitive response by the fish exposed to this sediment was in the liver mixed-function oxygenase activity (EROD) that was elevated even at the lowest dosage, which averaged at ~0.4 μg/g in the sand. The second most sensitive response was liver weight as a proportion of body weight; liver weight was higher in the second lowest dosage of ~27 μg/g in the sand. A number of other responses were noted at the higher dosages. The importance of this study lies in its implication that concentrations of PAHs of <1 μg/g in the oiled sand were taken up by the fish and elicited a response by them. Such concentrations have been observed in very remote Arctic locations (e.g., Cross 1986; see also Chapter 4).

A detailed examination of marine sediment following the *Exxon Valdez* spill was reported by Wolfe et al. (1996). Bioassays of sediment with the amphipod *Ampelisca abdita* and larvae of the Pacific oyster (*Crassostrea gigas*) were conducted using sediment collected from different depths and locations, some in areas oiled after the spill and others not oiled. The authors found considerable variability in the results, partly because of natural oil seepage in the area. However, tests with *Ampelisca* were conclusive in showing toxicity associated with the spill in intertidal sediment collected from heavily oiled sites. Similar tests showed the absence of toxicity in sediment collected from depths of 20 m or more. Sediment from intermediate depths mostly gave ambiguous results. Toxicity was still detected in shallow sediment (0–6 m) from heavily oiled sites in 1991 two years after the spill.

Mikhailova et al. (2001) examined the influence of oil-polluted bottom sediment on embryonic development of Siberian sturgeon (*Acipenser baeri*). The toxicity was expressed as asynchronous development, partial mortality, anatomical deformations, retardation in development of embryos and pre-larvae forms, disorders in hydration and heart rhythm of embryos and pre-larvae forms, delay in hatching, and increase in post-embryonic mortality rate. The critical concentration of crude oil in the bottom sediment for the roe used in the tests immediately after fertilization was 12–30 mg/kg. The maximum admissible concentration of oil components in the bottom sediment for embryos of the Siberian sturgeon was estimated at 20 mg/kg.

Several countries have established guidelines for levels of PAHs that may be used for assessing the quality of marine or freshwater sediments (Table 5.5). These levels vary considerably among different hydrocarbons with some cited as ranges. For example, Canada and the USA use a PEL (Probable Effect Level) of 391 μg/kg for naphthalene while European countries, through the OSPAR Commission use an EAC (Ecological Assessment Criterion) of 50–500 μg/kg. Canada has an additional target of only 34.9 μg/kg, about 10-fold lower than the PEL. These levels in Table 5.5 can be compared with data on individual PAHs in Chapter 4. It is often difficult to apply these guidelines to judge the quality of sediment from contaminated sites because the units of measurement are reported differently. For example, the Russian experience with sturgeon embryos resulted in a 'maximum admissible concentration of oil components' of 20 mg/kg but the guidelines are expressed compound by compound. Without knowledge of the components of the oil, its tolerable concentration cannot be calculated with confidence.

In addition to bioassay studies to determine the acute or chronic toxicity of sediment collected from spill zones, there are questions of effects on endogenous benthic organisms or demersal fish. For example, studies of chronic contamination of sediment with PAHs have demonstrated a number of effects on demersal fish in Puget Sound (e.g., neoplasms and liver lesions in English sole; Krahn et al., 1986) and in brown bullheads (*Ictalurus nebulosus*) from the Black River, Ohio (e.g., Baumann et al., 1987).

Most hydrocarbons partition readily to non-polar substrates (including living tissues of aquatic organisms), suspended particulate matter, and sediments. Long-term contamination of sediments with low concentrations of PAHs has resulted in several instances of cellular proliferative diseases. This implies that the evaluation of the impacts of PAH-contaminated sediments will remain problematic for the Arctic. Part of the problem is that PAHs in sediments may derive from sources unrelated to the oil and gas industry. For example, the U.S. National Research Council estimated that the largest source of hydrocarbons to the Canadian Arctic was through atmospheric transport and that the hydrocarbons in the air are principally derived from combustion of diverse fuels including wood and coal. PAHs have been measured in air samples from Alert on northern Ellesmere Island (Halsall et al., 1998) and in sediments in isolated Arctic lakes (Lockhart et al., 1993). Atmospheric dispersal distributes PAHs throughout the abiotic environment and they have become ubiquitous at low levels, even in remote areas. The concentrations of combustion-related PAHs are roughly ten times higher in air in winter than in summer months due to the transport of combustion products from Eurasia, while PAHs from forest fires and revolatilisation from soils and water surfaces reach maxima in the summer. Some combustion-related PAHs appear to have declined over the last decade as combustion-related pollution from the former Soviet Union countries has declined (Becker et al., 2006).

5.3.4.12. Concluding comments

The acute toxicity data and observations after spills leave no doubt that oil is acutely toxic to a wide range of aquatic organisms including those from the Arctic. Exposures of aquatic organisms to the lower-boiling aromatic compounds with one or two aromatic rings to produce mixtures in the low ppm range kill aquatic organisms consistently over exposure periods of less than a week. Generally, young organisms undergoing rapid growth and development show greater sensitivity than adults of the same species. The influences of temperature and salinity on toxicity have been examined in a few studies only and appear to be relatively

Table 5.5. Guidelines for maximum concentrations (μg/kg) of selected hydrocarbons and PAHs in marine sediments in relation to aquatic life (Buchman, 1999; CCME, 2005; OSPAR, 1997). When relevant, the Faroe Islands consults the OSPAR Ecotoxicological Assessment Criteria for guidance.

	USA	Canada		OSPAR
	PEL[a]	Quality guidelines	PEL	EAC[b]
Acenaphthene	88.90	6.71	88.9	
Acenaphthylene	127.87	5.87	128.0	
Anthracene	245.00	46.90	245.0	50–500 (f)
Benz[*a*]anthracene	692.53	74.80	693.0	100–1000 (p)
Benzo[*a*]pyrene	763.22	88.80	763.0	100–1000 (p)
Chrysene	845.98	108.00	846.0	100–1000 (p)
Dibenz[*a,h*]anthracene	134.61	6.22	135.0	
Fluoranthene	1493.54	113.00	1494.0	50–500 (p)
Fluorene	144.35	21.20	144.0	
2-Methylnaphthalene	201.28	20.20	201.0	
Naphthalene	390.64	34.60	391.0	50–500 (f)
Phenanthrene	543.53	86.70	544.0	100–1000 (f)
Pyrene	1397.60	153.00	1398.0	50–500 (p)

[a] PEL: Probable Effect Level; [b] EAC: Ecotoxicological Assessment Criterion. These criteria have been adopted by the OSPAR Commission for the preliminary assessment of Joint Assessment and Monitoring Programme (JAMP) chemical monitoring data with the aim of identifying potential areas of concern; they have no legal significance. Criteria marked with (f) are firm; those marked with (p) are provisional.

small. The effects of temperature have received surprisingly little attention but appear to be in the direction of slightly greater toxicity at higher temperatures. Temperature acts both on the behaviour of the hydrocarbons in water and also on the organisms and the combination is not always predictable. In those instances where a species occupies both marine and freshwater habitat at different times, the toxicity of hydrocarbons appears to be somewhat greater in seawater. When experimental exposures are carried out for periods of weeks or months, lethal and sub-lethal effects have been observed that were not evident in acute tests. While the usual exposure period of 96 hr is adequate for pelagic species, an extended exposure period of 28 days has been recommended as the minimum to describe the toxicity of oils to demersal fish and invertebrate animals (Moles, 1998). There is considerable overlap among taxa with regard to lethal concentrations with larval fish generally among the most sensitive.

The other relatively well understood risk of oil spills to aquatic animals is that of physical oiling of marine mammals and seabirds. Physical oiling is particularly serious for animals that derive insulation from fur rather than from blubber because oiled fur loses its insulating ability and the animals die from hypothermia. If the oiling is severe enough, seal pups drown. Oiled seals, otters, and polar bears ingest oil during grooming.

A less well understood risk is the long-term contamination of shallow sediment, especially intertidal sediment, with persistent hydrocarbons, namely the PAHs. Once associated with the sediment these compounds can persist for years and can be supplied to organisms for equally long periods. Chronic studies with oiled sediment have confirmed the ability of these compounds to act on the fish at low levels over long periods. They act through sub-lethal toxicology mechanisms like induction of enzymes, tumours, and genetic changes. Studies from a number of sites, particularly Puget Sound and Prince William Sound, have provided strong evidence for the role of PAHs in sediment in several diseases of demersal fish. Short et al. (2003) have argued persuasively that these slow-acting compounds pose a special risk to the eggs of demersally spawning fish. This type of action requires further investigation, especially with Arctic sediment that has been contaminated for long periods.

5.3.5. Sub-lethal aquatic toxicology

In addition to the lethal effects of hydrocarbons on aquatic organisms, a variety of sub-lethal effects have been described. Reviews of sub-lethal effects of oils and hydrocarbons on aquatic organisms were prepared by Johnson (1977) and Patten (1977) covering about 150 references available at that time. Several pages of sub-lethal effects on Arctic organisms were compiled by the U.S. National Academy in 1985 (U.S. National Research Council, 1985). Vandermeulen (1987) tabulated sub-lethal responses to freshwater organisms under five categories: developmental, physiological, pathological, behavioural, and metabolic. The concentrations at which these responses were observed ranged from 0.21 μg/L (benzo[*a*]pyrene) to 20 mg/kg (3-methylcholanthrene) with most of the results in the ppb range. It has been argued that lethal toxicology is a crude indicator of ecological responses (e.g., Percy and Mullin, 1977). Many different sub-lethal responses have been described, sometimes called biomarkers, in efforts to understand the actions of hydrocarbons on animals and to provide indicators more sensitive than mortality. These sub-lethal responses range from molecular events to whole-organism responses (e.g., behaviour) but are frequently biochemical in nature (see McCarthy and Shugart, 1990, and Huggett et al., 1992). Often biomarkers have been interpreted as sensitive indicators of exposure to classes of chemical compounds rather than as measures of effects on organisms. They are both. Ideally these responses are described from experimental exposures in laboratories where confounding variables are controlled and from wild populations exposed to the same or similar toxicants. Some of these responses (e.g., induction of mixed-function oxygenase enzyme activities, tissue histology, bile metabolites of PAHs) have been applied productively to investigations of oil spills. Some have been applied successfully to Arctic species and at Arctic temperatures (e.g. P4501A1 protein and EROD activity in Arctic char; Jørgensen and Wolkers, 1999) and suites of biomarkers are being applied in different Arctic areas to establish background levels or zones of impact (e.g., Borseth and Tollefsen, 2004). The toxicological importance of these sub-lethal studies is high. For example, they help to define and explain pharmacological mechanisms by which oils affect organisms. They help test for the presence or absence of cause/effect relationships. They offer indicators of responses to chemically induced stress earlier and often with less statistical ambiguity than traditional population data. They offer biological tests to determine whether feral organisms are under stress from oil or other environmental variables. The application of biomarkers to Arctic aquatic animals seems likely to grow in importance because few Arctic populations are known well enough to enable detection of subtle effects by any other means. As sub-lethal responses occur in a wide variety of species this section is organized by response rather than by taxonomy.

Two difficulties apply in the use of sub-lethal responses. One is their relationship to biological and environmental variables other than the events of interest. For example, a sub-lethal measure may be sensitive to species, season, gender, reproductive phase, life stage, genetic stock, temperature, and exposure to other substances not under investigation (e.g., Goksøyr and Forlin, 1992). These difficulties can usually be accommodated through sampling designs such as the inclusion of appropriate 'control' samples. The more difficult problem is the generally unknown relationship between the sub-lethal measurements and population or ecological responses (e.g., Lee and Anderson, 2005). Having found a sub-lethal response, the question is then how it should be interpreted in terms of biological risk to whole animals or to populations of animals. In principle, sub-lethal changes in individuals might be expected to be linked to the fitness of those individuals to survive and carry out normal biological functions. If fitness of a significant proportion of individuals in a population is compromised then the effects may become evident in population statistics. However, rigorous linkages have not been established. Whole symposia have been dedicated to studies aimed at understanding relationships among different levels of organization (Sutcliffe, 1994). Lacking clear links among different levels, the dilemma for biologists is to choose some mixture of measures that may be sensitive but of uncertain relevance and measures that may be insensitive but of certain relevance. Population changes in pelagic fish are difficult to relate causally to exposure. At the current level of understanding, one of the most helpful roles of sub-lethal responses is to probe for causality or at least for consistency with causality. The situation often arises in which biomarker responses are demonstrable but population effects are not. The question then is whether the biomarker responses should be taken as evidence of significant or objectionable impacts. The most precautionary approach is to interpret the sub-lethal respons-

es as evidence, if the stimulus persists, of biological effects on the organisms and to treat them the same as lethal effects or population effects in arguments about impacts and damages. The other extreme is to ignore sub-lethal responses and admit damages only after mortality or demonstrable population change are shown. The solution probably lies in expanded multidisciplinary approaches to define relationships among levels of biological organization. Research studies provide a growing list of sub-lethal responses and progress is being made to define relationships both to dosage history and to higher order responses.

The risks posed by higher-boiling PAHs are probably understood best from studies of sub-lethal effects. These compounds persist for long periods in sediment and can exert effects on demersal fish, eggs, and benthic organisms long after a spill. The types of effects are varied but often involve anatomical deformities, genetic damage, and tumours. These effects of PAHs are sometimes caused by the biotransformed products of the PAHs rather than by the PAHs directly. The PAHs are readily taken up by aquatic organisms but fail to become highly concentrated in vertebrates due to biotransformation to more easily excreted metabolites. However, this process imposes a different risk since some of the effects of PAH are caused by the biotransformed products. Chapter 4 lists PAHs found in tissues of a wide variety of Arctic animals. Some sub-lethal effects are considered in the following sections. These are listed separately but should not be taken as independent of each other. For example, an anatomical deformity may be the visible expression of a genetic or biochemical effect.

5.3.5.1. Algal photosynthesis

Hutchinson et al. (1979) provided convincing evidence that the biological effect of a number of common hydrocarbons on two unicellular freshwater green algae, *Chlorella vulgaris* and *Chlamydomonas angulosa*, was related closely to the water solubility of the hydrocarbons. The effect on the algae was expressed as a graph plotting the log of the molar concentration of hydrocarbon (pmol/mL) required to reduce photosynthesis by 50% (EC_{50}) against the log of the solubility (pmol/mL) (Figure 5.11). The three most toxic hydrocarbons were the three aliphatic hydrocarbons: decane, dodecane, and tetradecane. The least toxic hydrocarbons were benzene, toluene, and cyclopentane. The hydrocarbons with the lowest solubility were consistently the most toxic to photosynthesis in these experiments. The relationship appeared to hold for aliphatic and aromatic hydrocarbons and for some chlorinated hydrocarbons.

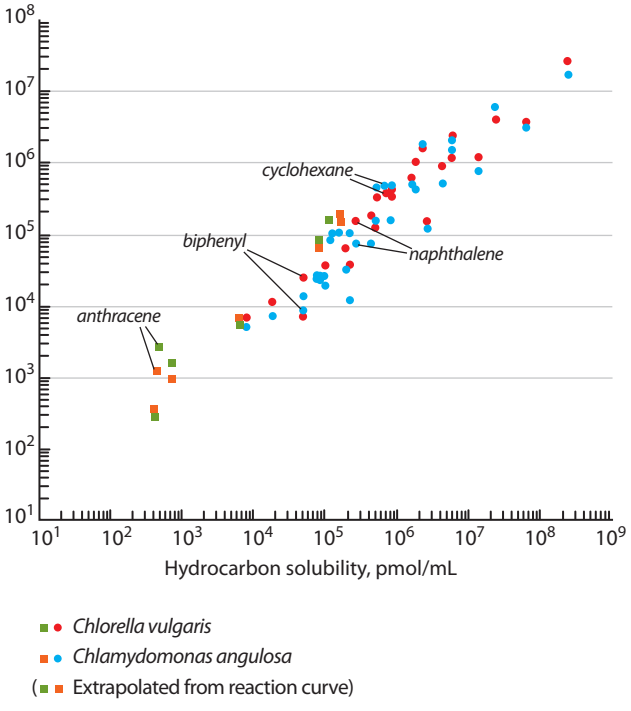

Figure 5.11. Relationship between solubility and toxicity for a wide range of hydrocarbons and two species of unicellular freshwater green algae (Hutchinson et al., 1979).

Russian studies have shown that machine oil and diesel oil affect the photosynthetic activity of the macrophytous algae *Fucus vesiculosus*, *F. serratus, Laminaria saccharina,* and *Rhodymenia palmate*. Diesel oil was more toxic for *Fucus* and machine oil was more toxic for *L. saccharina* and *R. palmate.* Some antagonism was observed in cases of simultaneous influence of machine oil and diesel oil (Chromov and Prokhorova, 1979). The toxicology of oil to *Fucus* species has been investigated extensively as a result of the *Exxon Valdez* spill (Van Tamelen and Stekoll, 1996) and is discussed briefly in section 5.3.9.2.

5.3.5.2. Moulting by invertebrates

Enhanced moulting of hammarides was observed during experimental exposures to natural gas (Kosheleva et al., 1997). At each shedding of the chitin shell the content of water in the bodies of these crustaceans increases sharply. The process of changing the chitinous shell in amphipods is controlled by hormones of the XY-organs and sinus glands (Kirshenblat, 1965). Exposure to natural gas appeared to affect the endocrine control of moulting.

5.3.5.3. Fish

5.3.5.3.1. Mixed-function oxygenase enzymes and activities

The induction of enhanced levels of liver mixed-function oxygenase enzyme activities (MFO) has been observed frequently as a measure of sub-cellular responses to oil, especially in vertebrates. MFO activities have been applied widely to indicate exposure to aromatic compounds with planar configurations and these include several PAHs, coplanar PCBs, and chlorinated dibenzodioxins and dibenzofurans. Enzyme activities increase following exposure to chemicals with the structural properties required to bind to a cellular receptor, the aryl hydrocarbon receptor. The increased enzyme activities are the result of the synthesis of new enzyme protein; antibodies to this protein are now available and have been applied in sensitive assay procedures (e.g., Stegeman et al., 1988; Jørgensen et al., 2001). Since the structural requirements for binding to the aryl hydrocarbon receptor are satisfied not only by several PAHs but also by other compounds ubiquitous in the environment (e.g., compounds in bleached kraft pulp mill effluent and certain PCBs), there can be some ambiguity in efforts to relate induction to exposure to hydrocarbons. Usually this lack of absolute specificity of the MFO response is not a serious disadvantage because exposure to other inducers would be unlikely to change in phase with changes in hydrocarbons following a spill. Notwithstanding the lack of absolute specificity of the MFO induction response, it offers one of the most reliable biological measures available to investigators trying to establish whether fish have experienced exposure

to inducers. The MFO response does not necessarily imply that the organism has been harmed; rather it implies that the organism has experienced exposure to a compound with the required structural properties and is adjusting to that.

The elevation of MFO activities has been observed in many laboratory experiments (e.g., James and Bend, 1980) using different species of fish and different oils or hydrocarbons. These activities have been used to monitor fish exposed to oil spilled or leaked (e.g., George et al., 1995b). Arctic species like Arctic char, polar cod and burbot (*Lota lota*) all display the response when exposed experimentally (Lockhart et al., 1989; Wolkers et al., 1998; George et al., 1995a). MFO activities have been applied to distinguish wild fish from contaminated and clean sites (Payne, 1976; Addison and Edwards, 1988) and to monitor spills (e.g., George et al., 1995b; Collier et al., 1996) and the rates of recovery of contaminated sites (Stegeman, 1978; George et al., 1995b; Huggett et al., 2003). Stegeman (1978) found that fish from a marsh contaminated by an oil spill in 1969 had not recovered to unexposed values when sampled in 1976 and 1977. However, five species of fish from the area influenced by the *Exxon Valdez* oil spill in 1989 showed no induction when sampled in 1999. The MFO response has been used successfully to detect effects of spilled oil on feral or caged fish on several occasions (e.g., George et al., 1995b; Stagg et al., 2000). The response appears to be robust in the sense that it is not affected by short-term stress of up to several hours that might be associated with capture or sampling (Jørgensen et al., 2001). Several instances have been reported in which the MFO enzymes have been used to monitor environmental quality in the vicinity of offshore drilling facilities. For example, Stagg et al. (1995) found evidence of enzyme induction in dab taken from sites within 20 km of North Sea drilling platforms as compared with fish from greater distances. These platforms were reported to have used predominately low-toxicity mineral-oil-based muds. In contrast, Mathieu et al. (2005) found no evidence of induction of EROD activities in American plaice (*Hippoglossoides platessoides*) taken near the *Terra Nova* oil field where muds with aromatic components were not used. The MFO response suggests that there has been greater exposure of the fish at the North Sea sites than at the *Terra Nova* site. The most likely source of inducing compounds near a drilling platform is disposal of used muds and drill cuttings. Some studies have indicated that oils used in oil-based muds induce MFO activities (Payne et al., 1987). Cuttings from a well being drilled using a paraffinic oil containing less than 1% aromatic compounds caused no induction of MFO activities in winter flounder (Addison et al., 1984).

These enzyme activities are important not only as biomarkers of exposure. They play a significant role in the metabolism of hydrocarbons (and other substrates) to more polar metabolites and their subsequent excretion. Sijm and Opperhuizen (1989) provided an extensive list of the reactions catalyzed by these enzymes. Sometimes the introduction of oxygen into substrates results in enhanced toxicity as when PAHs are oxidized to epoxides and diols that are active carcinogens (Stegeman and Lech, 1991). When these enzymes are inhibited, hydrocarbons are metabolized less readily. For example, Akkanen and Kukkonen (2003b) administered the MFO-inhibitor, piperonyl butoxide, along with pyrene to daphnids (*Daphnia magna*) in freshwater and the combined treatment resulted in less efficient conversion of pyrene to polar metabolites. These enzymatic activities or the actual proteins themselves are among the most successful sub-lethal responses available for vertebrates. Their levels are generally much lower in invertebrate animals but the principles seem similar. Surveys to establish normal ranges of catalytic activity or levels of the proteins responsible would be useful with Arctic species.

5.3.5.3.2. Fluorescent compounds in bile

The MFO enzyme system introduces an atom of oxygen into hydrocarbon substrates resulting in compounds more readily excreted. These polar metabolites have proven useful in surveys of PAHs and their potential effects, notably through the analysis for fluorescent aromatic compounds in bile of fish (Krahn et al., 1984, 1986) and marine mammals (Frost et al., 1994). The fluorescent compounds are not really a sublethal effect in the usual sense but they are products of the inducible enzymatic process of hydrocarbon metabolism. The combination of the analysis of bile for fluorescent aromatic compounds with liver enzyme activities was helpful in the assessment of exposure of fish to oil following the *Exxon Valdez* spill (Collier et al., 1996) and in many experimental studies of the metabolism and elimination of PAHs.

5.3.5.3.3. Haematology and blood chemistry

McCain et al. (1978) reported that haematocrit values were slightly higher in English sole exposed to oiled sediment (20.06%) than in unexposed controls (18.08%). Haemoglobin levels followed the same pattern with a value of 5.5 mg/dL in exposed fish as compared with 4.2 mg/dL in controls. These differences were found during the first few weeks of exposure; after 51 days the differences were no longer significant statistically. There were no differences throughout the study period between the groups with respect to red cell or leucocyte counts. DiMichele and Taylor (1978) generally found no differences in haematocrits of mummichogs (*Fundulus heteroclitus*) exposed to naphthalene.

Lethal concentrations of gas condensate were observed to have a major effect on the blood of fish. Pronounced leukocytosis was observed in young Atlantic cod. Erythrocytes characteristically showed several morphological anomalies. The quantity of unripe cells increased from 4.7% in controls to 40% (Kosheleva et al., 1997). Similar clinical signs of intoxication and disorders in peripheral blood were described by other authors in some species of the Black Sea and Caspian fishes in acute crude oil toxicosis (Kotov, 1976; Dokholya et al., 1980; Mazmanidi and Kotov 1981). At concentrations of gas condensate of 0.5–500.0 mg/L the numbers of erythrocytes increased at higher concentrations and greater proportions of the cells were immature, taken as an indication of increased erythropoiesis probably in response to decreased efficiency of oxygen transport (Kosheleva et al., 1997). If the shift to younger red blood cells and higher haematocrits by fish is a response to reduced oxygen carrying capacity, then it represents a serious impairment in fitness likely to have consequences for survival.

Serum glucose and serum protein were both elevated in mummichogs exposed to naphthalene at aqueous concentrations of 0.02 mg/L or more (DiMichele and Taylor, 1978). Thomas et al. (1981) found some elevation in plasma glucose following exposure to fluorene at 750 μg/L for 3 hr. Serum cortisol has been used frequently as a general measure of a response to stress in fish, however, the cortisol response to hydrocarbons in fish is somewhat confusing. DiMichele and Taylor (1978) exposed mummichogs to several concentrations of naphthalene and found a dose-dependent relationship between naphthalene and cortisol in three of the expo-

sures; however, the control and the highest concentration did not fit that pattern. Clear elevations of plasma cortisol were reported in striped mullet (*Mugil cephalus*) exposed to fluorene at 750 μg/L for periods of 1 and 3 hr (Thomas et al., 1981). Russian data on cortisol in blood of Atlantic cod exposed to the WSF of crude oil (at 200 mg/L) showed a gradual increase over 120 hr of exposure from 2.6–5.1 μg/L to 15.2–17.3 μg/L. However, cortisol is not specific to stress induced by oil and diverse sources of stress can induce elevated levels in fish. For example, the cortisol response was shown by sea bass (*Dicentrarchus labrax*) exposed to seismic noise (Santulli et al., 1999).

An experiment to evaluate ingestion of oil by seals was carried out on five captive ringed seals at the University of Guelph. Each was given 5 mL of oil per day for five days by putting capsules of oil in the fish normally fed to the seals. Blood samples were collected at intervals for four weeks following dosage and no effects were found except for higher activities of blood creatine phosphokinase in three treated animals. A second ingestion experiment was conducted with harp seal pups in the Gulf of St. Lawrence. One group received 75 mL of Norman Wells crude oil and the other group 25 mL. One pup from each group was killed at 1, 2, 4, 6, 8, and 10 days after ingestion and again, there was little apparent effect of the oil (Geraci and Smith, 1976).

Blood biochemistry appears to be a good indicator of stress in general but not necessarily a reliable indicator of sub-lethal toxicity of hydrocarbons specifically. Blood values can change very rapidly and are often considered to be among the 'flight' or 'fight' responses. Many variables can stress the animals and elicit these responses with or without the presence of hydrocarbons.

5.3.5.3.4. Respiration

Studies of the effects of hydrocarbons on respiration have often indicated increased consumption of oxygen, but not always. Studies by Brocksen and Bailey (1973) showed that oxygen consumption was increased in young chinook salmon following exposure to 5 or 10 μL/L of benzene. Thomas and Rice (1975) recorded dose-dependent changes on opercular rates of pink salmon fry exposed to the WSF of Prudhoe Bay crude oil. These changes lasted for several hours and were induced by a concentration of oil of 2.83 mg/L, or about 21% of the LC_{50} for these fish. Feeding instars of mosquito larvae (*Aedes aegypti*) exposed to the WSF of gasoline consumed more oxygen than unexposed controls but non-feeding stages did not (Berry et al., 1978). Mean respiration rates of the isopod *Ligia exotica* exposed to the No. 2 fuel oil at about the TLm concentration were always higher than controls although the differences failed to differ statistically (Dillon et al., 1978). Stekoll et al. (1980) exposed clams (*Macoma balthica*) to Prudhoe Bay crude oil in laboratory experiments and found that oxygen consumption increased from 0.406 μL/mg/hr in controls to 0.486 and 0.578 μL/mg/hr in clams exposed to 0.3 and 3 mg/L respectively. Activity levels and oxygen consumption were increased in the shore crab *Carcinus maenas* exposed to dilution of the WSF of Fortes crude oil (Depledge, 1984). Oiling the fur of sea otters caused an increase in their oxygen consumption, probably in an effort to generate heat lost due to the reduced thermal insulating properties of oiled fur (Costa and Kooyman, 1982).

In contrast, spider crabs (*Hyas araneus*) exposed to North Sea oil by injection or via sediment contaminated experimentally showed no change in respiration in spite of the fact that heart rates increased (Camus et al., 2002). Busdosh and Atlas (1977) reported decreased oxygen consumption by amphipods exposed to paraffinic, aromatic, and asphaltic fractions of Prudhoe Bay crude oil or Arctic diesel oil. Similarly, Capuzzo and Lancaster (1981, 1982) generally found reduced oxygen consumption in larval lobsters (*Homarus americanus*) exposed to South Louisiana crude oil.

It appears that the effect of hydrocarbons is generally to increase oxygen consumption unless narcosis is observed and in that event, oxygen consumption is decreased for the duration of the effect. It has been suggested that additional oxygen is needed to support the enzymatic oxidation of the hydrocarbons taken up by the animals. Measures of oxygen consumption can help understand responses to hydrocarbons but these measurements are difficult to make on field-collected specimens without capture artefacts. Taken together with the observations on red blood cell maturity status and haematocrits, these studies suggest the need for additional study of oxygen transport and consumption in animals exposed to oil. In view of the importance of oxygen consumption to sustained muscular activity, additional studies are warranted to define relationships among exposure to hydrocarbons, consumption of oxygen, and animal performance under controlled conditions.

5.3.5.3.5. Growth and feeding

Growth of fish has some potential for relating the condition of fish to exposure to hydrocarbons. It is one of the few measurements made on individual animals that may have acknowledged ecological relevance. Its main disadvantages are that many variables can affect growth and that it is not necessarily very sensitive (e.g., Woltering, 1984). Feeding is difficult to observe in any quantitative way in spill settings but is a component of growth. Several authors have reported reduced feeding by fish exposed to petroleum products but there are few quantitative observations on this. McCain et al. (1978) observed reduced feeding by English sole exposed to sediment oiled with Alaskan North Slope crude oil. Khan and Kiceniuk (1983) exposed winter flounder to sediment contaminated with oil and noted that food consumption decreased by about a third after four weeks of exposure and then progressively less food was consumed for the remainder of the exposure lasting 13 weeks. Emaciation was reported in Atlantic cod following exposure to the WSF of Hibernia or Venezuelan crude oils over 81–91 days (Khan and Kiceniuk, 1984). The concentrations of hydrocarbons experienced by the cod fell in the range 50–100 μg/L in one experiment and 150–300 μg/L in another. Engelhardt et al. (1981) reported anorexia in rainbow trout exposed to emulsions of Norman Wells and Venezuelan crude oils. Hendricks et al. (1985) noted reduced growth and enlargement of liver in rainbow trout fed benzo[*a*]pyrene over 18 months. Ryan (1999) studied feeding by measuring the amount of food consumed by rainbow trout exposed to the WSF of Norman Wells crude oil. Feeding was reduced by exposures to concentrations between 0.55 and 1.92 mg/L for two days and feeding stopped completely by three days. Growth in length of rainbow trout was reduced by exposure to the WAF of Norman Wells crude oil for periods of 30–55 days (Lockhart et al., 1996). Effects on feeding are not confined to fish. Berman and Heinle (1980) noted that copepods (*Acartia clausi* and *A. tonsa*) stopped feeding when exposed to the WAF of No. 2 fuel oil at a concentration of ~250 μg/L. There were no effects at 70 μg/L. The exposure affected both the amount eaten and the selection for sizes of food particles consumed. Busdosh (1981) noted reduced feeding by am-

phipods (*Boeckosimus* [*Onisimus*] *affinis*) exposed to the WSF of Prudhoe Bay crude oil at measured concentrations from 0.23 to 5.2 mg/L for exposure periods from 1 to 16 weeks.

Given the many observations of reduced feeding by animals exposed to oil, effects on growth would be expected. However, measures of growth such as size-at-age and condition are probably not diagnostic tests for oil-related damage but rather are more valuable as tests for consistency with exposure to oil.

5.3.5.3.6. Genetic effects

Investigations into the genetic effects of exposure to PAHs have become more frequent as genetic technologies have advanced (reviewed by Shugart et al., 2003). Relatively straightforward effects occur when exposure results in a change in either somatic or germ cells of an individual leading to somatic or heritable mutations. A mechanism by which petroleum hydrocarbons can cause mutations is through the PAHs which form covalent adducts with DNA. The evidence for production of genetic effects following oil spills has sometimes been equivocal. For example, Collier et al. (1996) examined fish of several species after the *Exxon Valdez* spill in 1989 but found no evidence of DNA adducts in spite of the fact that the fish were exposed to PAHs as judged by fluorescent compounds in bile. However, Hose et al. (1996) and Brown et al. (1996) found dose-dependent increases in the percentage anaphase aberrations in larval Pacific herring from areas oiled by the same spill. Furthermore, Brown et al. (1996) were able to induce this effect in laboratory exposures. Hypothesized genetic effects on larval pink salmon from oiled streams in Alaska (Bue et al., 1996) have been challenged as possible artefacts of the presence of hatchery fish in the samples (Cronin and Maki, 2004). In other unrelated studies the incidence of DNA strand breaks were measured in blue mussels (*Mytilus edulis*) put out experimentally in a contaminated area (Reykjavik harbour, Iceland) and compared with those placed in a clean reference site (Hvalfjorethur, Iceland). The mussels from Reykjavik harbour had consistently higher strand breakage as measured by the electrophoretic 'Comet' assay in gill cells and haemocytes, especially in mussels placed in the intertidal zone (Halldórsson et al., 2004). A similar comparison was reported by Barsiene et al. (2004) in which sites over a gradient in pollution were compared in the Baltic and North Seas. In this instance the frequencies of micronuclei were examined in red blood cells of flounder (*Platichthys flesus*) and wrasse (*Symphodus melops*) and in gill cells of blue mussels with the result that higher frequencies of micronuclei were found at more highly polluted sites. It is not clear yet how specific these responses are but they appear to offer powerful tools for detecting and confirming responses to hydrocarbons.

A different type of genetic effect is a selective effect by hydrocarbons on susceptible organisms resulting in altered gene frequencies in surviving populations. The phenomenon of selection for tolerance to specific chemicals is well documented; Georghiou (1986) listed many examples of species which have become resistant to pesticides applied to control them. There have been suggestions of tolerance to oils by organisms with a history of exposure to oil. Kanter et al. (1971) observed that clams (*Mytilus californianus*) taken from an area of natural oil seepage were more resistant to oil exposure than clams from areas with no history of exposure. It is not clear whether this was a result of genetic selection of organisms living in the seepage zone. Rossi and Anderson (1978), working with the polychaete *Neanthes arenaceodentata*, and Lee and Nicol (1981) working with the isopod, *Sphaeroma quadridentatum*, both described resistance to oil in offspring of parents exposed to oil even though the offspring themselves were never exposed until tested. These experiments suggest a genetic basis for increased tolerance to oil and that would confer obvious advantages on animals subsequently exposed to oil. The identification of genes conferring tolerance to oil, especially if convenient assays for them were developed, would be useful in monitoring for spill effects.

It is more difficult to establish the importance of another type of genetic selection effect because almost any type of stress can act as an agent of selection. The genes involved need not be related to the toxicology of the stress agent; they may simply select with such genes. An individual bearing mutations that result in reduced fitness is less likely to survive or reproduce. However, if an environmental agent produced a highly recessive deleterious mutation, it could persist in a population for many generations (e.g., 'founder mutations'), but it would not be expressed in most individuals carrying it and so would not be available for selection. Some such recessive genes appear to persist in spite of their disadvantages to homozygotes if they offer unexpected advantages to heterozygotes in situations unrelated to their deleterious properties. Cronin and Bickham (1998) simulated the effects of the *Exxon Valdez* oil spill using models of the frequencies of hypothetical genes that regulated egg survival. They concluded that detecting genetic impacts on populations exposed for a short time to an oil spill would be difficult or impossible. Given the difficulty of establishing cause/effect relationships with effects of this type, they seem unlikely to offer methodologies for defining or monitoring effects of Arctic spills, at least until improved molecular techniques become available.

5.3.5.3.7. Behaviour

Observations of the effects of oil on the behaviour of exposed animals have been popular laboratory experiments and have been conducted with diverse taxa and techniques. For example, Busdosh (1981) noted dramatic reductions in movement by amphipods (*Boeckosimus* [*Onisimus*] *affinis*) in measured concentrations of Prudhoe Bay crude oil of 0.2–5.2 mg/L for periods up to 14 weeks. Field observations of behaviour following exposure to oil have been recorded often, frequently with marine mammals. For example, killer whales (*Orcinus orca*) were observed to swim through oil slicks on five occasions in Prince William Sound after the *Exxon Valdez* spill and did not appear to avoid the oil (Matkin et al., 1994).

Exposures to gas condensate, methanol and 'natural gas' has been seen to change the behaviour of hammarides (Kosheleva et al., 1997). The bodies of these crustaceans became slightly extended and unnaturally bent. Some individuals made chaotic throws and rotated about the axis; antennae and body muscles twitched and flexed convulsively. Probably the main target of pharmacological action was the central nervous system (Dokholyan et al., 1979; Mikhailova et al., 1986). However, amphipods recovered from non-lethal intoxication with methanol and gas, which is typical for depressants (Metelev et al., 1971; Pomazovskaya, 1979). The characteristic phases of intoxication were: a state of hyperactivity, retardation of reactions, complete loss of sensitivity, and death. Long-term studies suggested that safe concentrations of gas condensate, natural gas

and methanol for mature hammarides were 1, 2, and 5000 mg/L, respectively. The gas condensate, unlike all other compounds tested, affected reproductive behaviour of mature individuals. For example, the controls had five mated pairs; exposures to 0.5 and 1 mg/L had four mated pairs; exposures to 5 and 20 mg/L had only one mated pair, and exposure to 50 mg/L had none.

Behavioural observations with the Arctic amphipod *Onisimus affinis* and the coelenterate *Halitholus cirratus* exposed to several crude oils (Atkinson Point, Pembina, Venezuela, Norman Wells) were reported by Percy and Mullin (1977). Locomotory activity by amphipods was reduced by exposure to all four crude oils and the severity of effect was related to the dosage of oil. Norman Wells crude oil had the greatest effect. The authors noted that amphipods exposed to Pembina crude oil for 24 hr and then transferred to clean seawater failed to recover locomotory activity. Results were similar although less dramatic with *Halitholus* exposed to Norman Wells and Pembina crude oils. These animals were less sensitive to Venezuelan crude oil and almost insensitive to Atkinson Point crude oil.

Rosenberg and Wiens (1976) placed wire baskets of oiled and unoiled artificial substrates (stones 5 to 8 cm in diameter) in the Trail River, Northwest Territories, Canada, and examined the types and numbers of aquatic insects that colonized them at intervals up to 335 days. A comparison of colonization by chironomids of treated and untreated baskets as a measure of effect showed that ten species were positively affected by the oil, ten were not affected, and nine species were negatively affected. Mayflies were affected more negatively than chironomids with *Heptagenia* and *Stenonema* failing to colonize oiled baskets in spite of high abundance on unoiled ones (Rosenberg et al., 1980).

Macoma balthica clams made little response to Prudhoe Bay crude oil stranded on the mud beds where they were buried in experimental aquaria in Alaska (Taylor and Karinen, 1977). However, exposure to the WSF of Prudhoe Bay crude oil resulted in some of the clams emerging from their buried locations. The WSF inhibited the burial rate of clams that were unburied at the time of exposure. When the exposure was carried out by adding oiled sediment to established aquaria, the results were similar with clams emerging from the sediment. Stekoll et al. (1980) noted similar behaviour with this species in long-term experiments and observed chronic mortality in excess of that in controls. Clams of several species were observed to emerge from their buried locations in the sediment after exposure to artificially aged Lagomedio oil in the Baffin Island experimental spill (Cross and Thomson, 1987).

Rice (1973) conducted avoidance tests with young pink salmon in a trough with inputs of solutions from both ends and an overflow in the middle. When seawater mixed with Prudhoe Bay crude oil was introduced into one end and uncontaminated seawater into the other, pink salmon fry avoided the contaminated seawater at concentrations estimated to be as low as 1.6 mg/L. Under these test conditions, the 96-hr TLm was found to be 88 mg/L indicating that the young salmon could detect and avoid oil at levels well below those acutely toxic to them. Atlantic cod have been found to detect hydrocarbons and change their behaviour even at low μg/L levels (Hellstrom and Doving, 1983) and adult cod avoid water with hydrocarbon concentrations of 50–100 μg/L (Bohle, 1986).

Sand goby (*Pomatoschistus minutus*) changed their swimming behaviour in response to laboratory exposures to concentrations of <1 μg/L of the WSF of North Sea crude oil (Berge et al., 1983). The threshold concentration at which swimming effects were noted was 0.3 μg/L (Reiersen and Berge, 1985). Some fish can detect the presence of oil in sediment and behave in ways to avoid it. The sand lance is a small fish that buries itself in soft marine sediment to avoid predators. Pinto et al. (1984) conducted choice experiments with different types of sediment (gravel, sand, silt) and the sand lance chose to bury themselves in sand over either gravel or silt. Then a third option was introduced by contaminating one or other sediment with Prudhoe Bay crude oil at either a high concentration (~1100–1200 mg/L) or a low concentration (128–161 mg/L) of Prudhoe Bay crude oil. When a choice between clean sand and oiled sand was presented, the fish clearly chose the clean sand. Although clean sand was preferred over clean gravel, when the sand was oiled, most of the fish buried themselves in the gravel. The fish could detect and avoid oiled sand. These experiments suggest that oiled sandy substrates would be avoided with whatever consequence that may imply for the fish. Avoidance of PAH-contaminated sediment was reported for amphipods (*Eohaustorius estuarius*) when offered a choice between contaminated and clean sediment, although the avoidance response was only slightly more sensitive than mortality (Kravitz et al., 1999). In contrast, choice experiments with clean and contaminated sediment showed that juvenile spot (*Leiostomus xanthurus*) did not alter feeding behaviour at moderate to high PAH concentrations (22 mg PAH/ kg dry sediment), which would put them at risk for both sub-lethal and lethal effects. At very high levels of PAH contamination (122 mg PAH/ kg dry sediment) the fish stopped feeding, probably due to a narcotic effect of PAH released into the water column (Hinkle-Conn et al., 1998).

McCain et al. (1978) reported slight but consistent behavioural changes in English sole exposed to oiled sediment. The fish from the oil-exposed group were less active and did not feed as well as fish from the untreated group. Conversely, unusual hyperactivity was reported in cod exposed chronically to the WSF of Hibernia or Venezuelan crude oils (Khan and Kiceniuk, 1984). Striped mullet exposed to dibenzofuran, fluorene, carbazole, and dibenzothiophene all exhibited profound changes in behaviour at exposure concentrations of 1 mg/L or more (Thomas et al., 1981). Fish were reported to swim weakly on the surface and then to lose the ability to turn. Uncontrolled twitching and dystonic movements were observed. Finally they became paralysed, sank to the bottom, and died. The concentrations of brain neurotransmitters were determined in the fish from the experiments with dibenzofuran; epinephrine, dopamine and serotonin were all found to have become depleted relative to untreated controls (Thomas et al., 1981). The effects of oil on central nervous system biochemistry have not been investigated rigorously.

Siniff et al. (1982) carried out two oiling experiments with sea otters in Prince William Sound, Alaska. Four otters were oiled, each with 25 mL of Prudhoe Bay crude oil applied to a small portion of the pelage, and released with radio-transmitters attached. The level of activity displayed by the oiled otters increased dramatically for about the first week after treatment, mostly activity associated with grooming. Feeding by the oiled otters, as measured by dive time, was unchanged for most of the otters and all survived the period of observation of about three weeks. A second experiment with two otters was conducted by oiling one half of a pool with a partition under which the otters could swim. The otters spent little time in the oiled side of the pool (less than 1 min/hr) indicating that they could detect and avoid the oil

but nevertheless they became covered in oil. Then one otter was cleaned with detergent Polycomplex-11 and released back into Sheep Bay where it had been captured originally. This otter was lost because the radio-transmitter failed. The second oiled otter was not cleaned but was placed in clean seawater where it died 10 hr later. Autopsy results indicated hypothermia as the most likely cause of death.

In a laboratory study to determine polar bears' ability to detect and avoid unweathered crude oil, three polar bears were coaxed individually along a passageway leading to a small pool containing 7000 L of seawater covered with a 1-cm surface slick of unweathered crude oil (Øritsland et al., 1981; St. Aubin, 1990b). None of the bears entered the pool voluntarily, but all three investigated its oil-covered surface. As the bears stretched over the pool to reach seal blubber suspended inaccessibly from the top of the cage, the door was closed behind them, forcing them into the water. The bears made deliberate attempts to escape, and did escape to some extent by supporting themselves on the cage bars surrounding the pool; they continued attempting to grab the bait. When the cage door was opened after 15–50 min, the bears left the pool immediately. These brief observations confirm that polar bears can detect oil and at least initially would not voluntarily enter oiled water. Once covered by oil, these bears showed no obvious aversion to it. They actively ingested oil during grooming.

Investigators have studied petroleum avoidance in a variety of cetaceans (Grose and Mattson, 1977; Shane and Schmidly, 1978; Goodale et al., 1981; Gruber, 1981; Evans, 1983; Owen, 1984; Sorensen et al., 1984). The whales in these controlled experiments did not avoid slicks but swam through them, seemingly without ill effect. However, the animals avoided surfacing in the areas of heaviest oil concentration. Studies by Geraci et al. (1983) determined that once a dolphin entered an oil slick, its response after that was to avoid it. There does not appear to be any oil fouling of the skin of free-living whales; oil may not stick to the surface of the skin or contact with oil may be rare because free-living whales avoid oil slicks (Geraci et al., 1986).

5.3.5.3.8. Carbohydrate metabolism

Mazmanidi and Kovoleva (1975) exposed two Black Sea species of fish (*Spicara smaris* and *Solea lascaris*) to 'dissolved petroleum products' at 29 mg/L (for up to 10 hr) and found that glycogen storage in liver, heart, and red muscle was reduced. Glycogen in white muscle, however, was not affected. When the concentration of petroleum was reduced to 0.05 mg/L, but the exposure time extended to as much as 40 days, reductions in glycogen were noted in all four tissues. Riley and Mix (1981) exposed oysters (*Ostrea edulis*) to naphthalene and then examined the uptake and distribution of labelled carbon following a short pulse of treatment with ^{14}C-glucose. They found an increased flow of glucose carbon into all measured pools of carbon, indicating increased catabolism of protein and polar lipids. The authors suggested that chronic exposure would result in decreased growth efficiency unless increased carbon metabolism was offset by increased carbon assimilation. Sabo and Stegeman (1977) presented electron micrographs of liver of *Fundulus heteroclitus* from two marshes, one with a history of contamination by No. 2 fuel oil and the other with no history of contamination. There was relatively little glycogen or neutral fat visible in the livers of the fish from the contaminated habitat. Hawkes (1977) found the virtual elimination of glycogen storage from liver of rainbow trout fed Prudhoe Bay crude oil for 75 days. In contrast, Payne et al. (1988) found enlarged livers in winter flounder exposed to PAHs in the sediment. The liver enlargement was apparently due, at least in part, to increased energy stores of glycogen. (Changes in liver lipids were not significant statistically.)

5.3.5.3.9. Lipid metabolism

Sabo and Stegeman (1977) collected fish (*Fundulus heteroclitus*) from oil-contaminated and clean habitats and examined the synthesis of lipids from ^{14}C-labelled glucose or acetate in isolated liver preparations from both groups. The fish from the clean habitat incorporated more label into phospholipid than fish from the contaminated habitat when the label was presented as glucose. However, when the label was presented as acetate, the fish from the contaminated habitat incorporated more label into phospholipid. There was little difference in incorporation of label into triglycerides when glucose was the source. However, a large percentage decrease was found in the fish from the contaminated habitat. The authors suggested that these results are consistent with an altered structure of either intracellular or cell-surface membranes. Hawkes (1977) noted depletion of liver stores of lipids in rainbow trout fed Prudhoe Bay crude oil for 75 days. However, McCain et al. (1978) found increases in lipid storage in livers of English sole exposed to sediment treated with Alaskan North Slope crude oil as did Payne et al. (1988) with winter flounders maintained over sediment contaminated with a mixture of PAHs.

The measures of stored reserves of energy seem potentially useful in spill settings, although over exposure periods of several days and longer. In the Arctic, spills have the potential for chronic exposure and these tests will be useful in confirming physiological actions of the hydrocarbons. Related to those may be measures of organ size such as liver size as a proportion of body size.

5.3.5.3.10. Deformities

Hannah et al. (1982) reared fertilized eggs of rainbow trout on sand that had been coated with 1– 500 µg/g of ^{14}C-labelled benzo[*a*]pyrene. This produced concentrations of benzo[*a*]pyrene in the water of 0.08–2.99 µg/L. There were no differences in the numbers of eggs that hatched between treated and untreated groups, however there were differences in the time required to complete 50% of hatching. Alevins from eggs treated with all concentrations of benzo[*a*]pyrene were slightly shorter in length than those from untreated eggs. Incidences of anomalies were higher in treated groups and some additional abnormalities not found in the controls were evident (cyclopia, apparent anophthalmia, microphthalmia, albinism, insufficient yolk sacs and lack of retinal pigment). Juvenile herring and goby (*Gobiidae*) showed morphological changes after the spill of the M/S *Eira* in the Baltic in 1984; juveniles were smaller than those taken during a normal summer and the rear parts of their bodies were short and curved with damage observed in skeleton and jaws (Ikävalko, J. 2005, pers. comm. MTK, The Central Union of Agricultural Producers and Forest Owners, Helsinki, Finland).

Exposure to oil has been found to affect the eyes of fish. Hawkes (1977) examined the size and consistency of eye lenses from rainbow trout fed Prudhoe Bay crude oil for a year. The lenses were enlarged (lens volume of controls, 100.9 mm^3; lens volume for treated fish, 226.1 mm^3) and soft. The author reported that relatively mild pressure perma-

nently compressed a lens from a treated fish into an amorphous mass whereas a lens from an untreated fish returned to its normal geometry after the same pressure. Payne et al. (1978) found statistically significant enlargement of eye lenses in cunners exposed to Venezuelan crude oil for six months although the difference was less striking (diameter 0.37 cm in controls, 0.41 cm in exposed). Lense fibres were reported to be affected and retinal oedema was noted in cutthroat trout (*Salmo clarki*) exposed to Wyoming crude oil in a diluter apparatus at 450 and 520 μg/L (Woodward et al., 1981). Hawkes and Stehr (1982) induced necrosis in retinal neurons by exposing embryos of surf smelt (*Hypomesus pretiosus*) to the partially weathered WAF of Cook Inlet crude oil for 3 hr/day at a concentration of only 54 μg/L. They reported necrotic neurons in the forebrains of embryonic surf smelt exposed to the WAF of Cook Inlet crude oil. The exposures took place for only three hours per day over a period from 6 days after fertilization until either 21 or 27 days after fertilization (15 or 21 days of exposure). Exposed embryos showed necrotic forebrain neurons in 66–80% of individuals but none were found in controls. Brains and eyes were the only organs showing structural effects.

Several deformities were described in sand sole (*Psettichthys melanostictus*) treated with benzo[*a*]pyrene by adding an ethanol solution of benzo[*a*]pyrene to the aquarium containing eggs (Hose et al., 1982). The hatching success of the treated eggs was reduced relative to ethanol-only controls, and the eggs that did hatch did so earlier than controls. A number of other anatomical abnormalities were described. Similar experiments were conducted with flathead sole (*Hippoglossoides elassodon*) but the benzo[*a*]pyrene was complexed with bovine serum albumin before addition to the test aquarium. In this instance, the effects on hatching did not materialize but structural effects not found in controls were reported in neural and ocular tissues.

Visible deformities are valuable observations following exposure to oil or other toxicants. With the possible exception of taste and odour of fishery products, there are no other observations that indicate so convincingly that biological damage has occurred.

5.3.5.3.11. Immunocompetence

Several responses of fish to oil or PAHs may be mediated by effects on the immune system (Reynaud and Deschaux, 2006) but the mechanisms responsible remain poorly understood. Palm et al. (2003) fed juvenile chinook salmon a mixture of 14 PAHs for 28 days and then subjected them to challenge with the bacterial pathogen *Listonella anguillarum*. The exposure to PAHs had no effect on growth or on the responses of the fish to the pathogen. In one experiment fish that had been exposed to the PAHs were vaccinated against *L. anguillarum* and subsequently exposed to the pathogen. Prior exposure to the PAHs did not diminish the ability of the fish to acquire immunity to the pathogen. Further research will be required to determine the role of tests of immune functions in spill monitoring and in mechanisms producing several other effects.

5.3.5.3.12. Ionoregulation

Several studies have examined plasma major ions and osmolality in experimental animals. Rainbow trout were acclimated either to seawater or to freshwater and then exposed to emulsions of Norman Wells or Venezuelan crude oils (Engelhardt et al., 1981). When the exposures were in freshwater the concentrations of the major plasma ions chloride, sodium and potassium and osmolality fell but when exposed in seawater they increased. These changes in plasma composition were not accompanied by changes in plasma cortisol but were accompanied by histological changes to gills. Some of these changes would be consistent simply with altered fluxes of water, with diffusion into the fish in freshwater and out of the fish in seawater in response to osmotic gradients. This interpretation would predict that water flow into freshwater animals would increase their water content if the flow exceeded the kidney capacity to clear it. Observations of increased water content, oedema or ascites, have been observed following exposures of fish to several toxicants including oil (Linden, 1976b; Helder, 1980; Lockhart et al., 1996; Marty et al., 1997a). Hutchinson et al. (1979) noted that exposure to various hydrocarbons caused algal cells to leak potassium and manganese, suggesting impaired integrity of plasma membranes. While these tests of ion content and fluxes are unlikely to be diagnostic of oil intoxication when used alone, they do offer useful confirmatory information. Their reliability when applied to specimens that have undergone the trauma of capture and sampling will require further investigation.

5.3.5.3.13. Egg production

Russian studies on exposures of gastropods (*Littorina obtusata*) to natural gas, methanol, and gas concentrate noted reduced egg production by females at concentrations of natural gas in the range 3.3–7.2 mg/L, methanol at 10–1000 mg/L, and gas condensate at 50 mg/L. Death of embryos was enhanced and hatching decreased in methanol at 10 mg/L and higher. At 50–500 mg/L, development stopped at the stage of pigmentation of the shell, and at 1000 mg/L at the stage of its formation (Kosheleva et al., 1997).

5.3.5.3.14. Egg hatching

Johannessen (1976) exposed fertilized eggs of Barents Sea capelin (*Mallotus villosus*) to seawater extracts of Ekofisk oil. Hatching was monitored daily over about six weeks and was consistently lower in exposures to 25 ppb or greater. The results at 10 ppb were equivocal and so the threshold for this effect was between 10 and 25 ppb. The author noted fouling of eggs by bacteria and conceded that the effect on hatching may have been indirect as a result of fouling. Linden (1976b) noted reduced hatching in Baltic herring following exposures of embryos to relatively high levels of Venezuelan crude oil and several dispersants. Hawkes and Stehr (1982) reported large reductions in hatching of surf smelt eggs after exposure to very low concentrations (54 and 128 μg/L) of the WAF of Cook Inlet crude oil. In a control group, 51.6% of eggs hatched but in the group exposed to 54 μg/L only 12.1% hatched while in the group exposed to 128 μg/L hatching was reduced to 7.6%.

Some observers have found that the timing of hatching is altered by exposure to oil. Anderson et al. (1977) found that exposure to the WSF of South Louisiana crude oil resulted in changes in the rate of hatching of *Fundulus similes*. Controls started hatching at 24 days and hatching was complete by 27 days. Eggs exposed to a 10% dilution of the WSF of the oil began hatching on day 17 and continued until day 30. Those exposed to the 30% dilution began hatching on day 20 and continued until day 28. These exposures apparently stimulated early hatching by some but not all of the embryos. The authors removed the chorionic membrane from some embryos of *Cyprinodon variegatus* to see whether this membrane offered any protection to the embryos. They

found that its removal had no effect on the hatching success of the embryos and that it was apparently not an effective barrier against the effects of hydrocarbons. Early hatching of eggs of Pacific herring was reported by Brown et al. (1996) following the *Exxon Valdez* spill.

Egg hatching is clearly important for population analyses and its timing and success rates are both amenable to field observation. Coupled with histological analyses of developing embryos, examination of eggs seems a very valuable component of an investigation of any spill that occurs at times and places where unhatched eggs occur.

5.3.5.3.15. Embryological development

Sea urchins (*Strongylocentrotus droebachiensis*) from Kislaya Bay, Motovsky Gulf, were exposed to natural gas in seawater. A concentration of 7.9 mg/L reduced fertilization success to 57% of that of controls (Kosheleva et al., 1997) although there was no such effect at 6 mg/L or less. Enhanced mortality of embryos was observed starting at concentrations of 3.9 mg/L. In these experiments, methanol was less toxic than natural gas with the most sensitive effects on embryos noted at concentrations of 50 mg/L and above. The hydrocarbons of natural gas were approximately ten times more harmful than methanol. Embryos at the beginning of cleavage and larvae at the pluteus phase were the most sensitive to both toxicants. The most sensitive response to natural gas was at 1.8 mg/L.

5.3.5.3.16. Heart rate

Linden (1976b) recorded consistent increases in heart rates in embryonic Baltic herring during the period from 132 to 324 hr after fertilization, about the time of hatching. The embryos exposed to light Venezuelan crude oil showed about half the increase recorded for controls. Those embryos exposed to a combination of oil with any of three dispersants (Finasol OSR-2, BP 1100X, or Finasol SC) had even lower heart rates. This response was one of several observed in the treated embryos including several deformities. Heart rates declined in embryos of small, temperate, estuarine fish (*Fundulus heteroclitus*, *F. simulus*, *Cyprinodon variegatus*) following exposure to the WSF prepared from No. 2 fuel oil or South Louisiana crude oil (Anderson et al., 1977). Heart rates responded most clearly at relatively high concentrations of 50% or more of the undiluted WSF. Russian studies of the WSF of crude oil in the range 10–100 μg/L led to appreciable changes in the cardiac rhythm of Barents Sea cod, and in electrocardiograms (ECG) (Kosheleva et al., 1997). The irregular heart rhythm persisted for 15–20 days after applying the toxicant. Related changes in electrical activity of smooth muscles were noted. Oil at 50 μg/L suppressed muscle activity within 3–4 min after the beginning of exposure and this persisted until exposure was ended. The amplitude of serial potentials of peristaltic movements of the stomach decreased gradually. Heart rates in crustaceans have been found to increase in experimental studies with shore crabs (*Carcinus maenas*) (Depledge, 1984) and spider crabs (*Hyas araneus*) (Camus et al., 2002). These observations may be difficult to make in field-caught specimens without confounding the observations with artefacts induced by capture.

5.3.5.3.17. Breathing and coughing rates

Opercular breathing rates were measured in pink salmon fry during exposure to the WSF of Cook Inlet crude, Prudhoe Bay crude, and No. 2 fuel oil (Rice et al., 1977). Results were similar for the three oils. Breathing rates increased at the beginning of exposure and peaked after about 3 hr before declining, although rates did not fall to pre-exposure levels over the 72 hr. There was clear dose-dependency with higher exposure concentrations resulting in higher breathing rates. The exposure required to induce the increased breathing rates was about 30% of the 96-hr TLm. The pattern found for the cough response was essentially the same as that for the breathing rate. The authors hypothesized that the opercular breathing rate increased in order to obtain extra oxygen needed to support the energy requirements of metabolism and excretion of the hydrocarbons. Russian studies noted the occurrence of 'respiratory cough' at a concentration of 10 μg/L of the WSF of crude oil. (Kosheleva et al., 1997) and a decrease in the amplitude of movement of branchiate flaps, and faster breathing; these increased with increasing concentrations of oil components.

Heart rate, breathing, and coughing are sensitive to acute exposure to oil. It may be feasible in some settings to make direct observations of these responses in feral animals, but their most successful applications have been in laboratory settings. Capture and handling of wild animals is likely to confound field observations of these responses.

5.3.5.3.18. Histopathology

Histological examination of tissues has been one of the most widely and productively applied examinations following both experimental exposures to oils and following oil spills. Hawkes (1977) described several structural effects of experimental exposure to oil on the histology of gills, skin, liver, eye lens, and other organs. Following exposure to only 100 μg/L of the WSF of Prudhoe Bay crude oil for five days, the gills of coho salmon and starry flounder (*Platichthys stellatus*) showed loss of surface cells or the first two to three layers of cells. Immature sub-surface mucous glands were exposed and their contents were sometimes exuded. This may account for the excess mucous often reported with fish. Skin of English sole exposed to 13% WSF for five days had many mucous glands empty although attempts to repeat these observations were inconclusive. Rainbow trout fed dietary Prudhoe Bay crude oil at a relatively high dosage of about 11 mg oil per day (for fish weighing 90 g) for two weeks showed dramatic reductions in glycogen as compared to unexposed controls (Hawkes, 1977). Proliferation of the endoplasmic reticulum of liver cells was apparent. Fish exposed in longer feeding studies (75 days) lost virtually all their liver glycogen stores and some of their lipid stores. Still longer studies (a year) resulted in invasion of liver sinusoids with collagen fibres. English sole exposed to sediment oiled with Alaskan North Slope crude oil revealed no pathology in most organs (McCain et al., 1978). However, fish from the oil treatment had severe hepatocellular lipid vacuolization with more than 95% of the liver cytoplasm taken up by lipid droplets and lipid vacuoles. They found extensive whorls of rough endoplasmic reticulum in liver of exposed sole. DiMichele and Taylor (1978) reported several histopathologies in mummichogs exposed to naphthalene: gill hyperplasia, gill haemorrhage, taste bud necrosis, nasal and lateral line necrosis, pancreatic ischemia, hepatic ischemia, cerebral ischemia, renal ischemia, hypertrophy of interrenal cells, necrosis of interrenal cells, necrosis, and sloughing of the intestinal mucosa and muscle necrosis.

Khan and Kiceniuk (1984) noted an intense dark pigmentation of the skin in cod exposed to the WSF of two crude oils. Histologically, they observed massive gill hyperplasia of mesenchymal cells in both primary and secondary lamel-

lae, culminating in extensive fusion of secondary lamellae. The numbers of mucous-producing cells were increased relative to controls. Histological alterations were found in liver and testis. Hepatocytes were conspicuously divided into microvesicles in oil-treated cod. Testes of control cod had all spermatogenic stages including spermatids and sperm in the lumina of the seminiferous tubules. Oil-treated cod, however, had testes in either spermatocytic or spermatogonia stages but few spermatids.

Russian studies of acute poisoning of young Atlantic cod with gas condensate reported internal organs overfilled with blood following ruptures of blood vessels with small and massive haemorrhaging. Mass degradation of curved channels was noted; their borders were erased or invisible, and in their place was a shapeless congestion of collapsing epithelial cells, degenerate changes both in the cytoplasm (granulation, vacuolization) and in the nucleus (mainly pycnosis, less often swelling). Gills were affected by gas condensate and natural gas at concentrations exceeding 0.5 and 1.8 mg/L respectively. Observations included thickening of the free ends of lamellae, swelling of lamellae due to an increase in the number and volume of epithelial cells, local expansion of lamellar capillaries, scaling of the epithelium of respiratory plates, rupture of endothelium and blood infusion into the inter-lamellar space. The quantity of mucous cells in an active state on lateral surfaces of lamellae increased as did the numbers of chloride cells with poorly coloured vacuolated contents, reported by Fedorov (1967) to be a sign of physiological exhaustion. The severity of these histological effects was related directly to the degree of exposure to the hydrocarbons.

Following the grounding of the *Amoco Cadiz* off the coast of France in March 1978, plaice were collected over the period 1978 to 1980 and examined for histological evidence of injury (Haensly et al., 1982). Many effects were found, the most frequent being fin and tail necrosis, hyperplasia and hypertrophy of gill lamellar mucous cells, gastric gland degeneration, decreased hepatocellular vacuolation (lipid), increased concentration of hepatic macrophage centres, and lateral trunk muscle degeneration. Field studies reflect the experience of the animals which may include exposure to multiple stressors. For example, Simpson and Hutchinson (1992) obtained samples of dab from several sites along a gradient of contamination in the North Sea. Histological changes along the gradient were found in heart, liver, and kidney. These changes were consistent with exposure to environmental contaminants, but were not specifically linked to hydrocarbons because the sites were also contaminated with organochlorine compounds and metals. This illustrates a general difficulty with biological monitoring, that is, assigning effects in the presence of multiple possible causes.

Oiled sea otters from Prince William Sound after the *Exxon Valdez* spill were recovered with various states of oiling and examined microscopically by Lipscomb et al. (1993). They examined otters directly from the sea and others that had been taken to facilities to clean them. They found that interstitial pulmonary emphysema, centriolobular hepatic necrosis, and hepatic and renal lipidosis were associated with exposure to the oil.

Histopathological analyses are among the most consistent and most readily interpreted of all sub-lethal responses especially as applied to liver and gills of fish. Collecting samples appropriate for histopathology must be given high priority in the event of a spill in the Arctic. Accumulation of images of key organs (liver, gill, eye, etc.) representing the 'normal' state of key species will facilitate interpretation of samples collected after a spill.

5.3.5.3.19. Reproductive steroids

Polycyclic aromatic hydrocarbons or their metabolites have been implicated as endocrine-disrupting chemicals in fish because of the associations between measures of reproductive performance and exposure to PAHs as measured by fluorescent materials in the bile (reviewed by Gross et al., 2003). However, the role of PAHs in field observations of fish reproduction is not established clearly. Truscott et al. (1992) exposed winter flounder to sand mixed with five levels of Venezuelan crude oil in laboratory tanks. The exposures lasted for four months, from February until June. Water temperatures ranged from -1 °C in February to +5 °C in June. Blood samples taken from the males after the four-month exposure revealed a significant dose-dependent decrease in levels of testosterone glucuronide and 11-ketotestosterone glucuronides in blood plasma. Free testosterone and 11-ketotestosterone were not affected. The lowest treatment level was 5 mL oil per 45 kg dried sediment and even that low level of treatment affected both glucuronides. The authors noted that the roles of androgen glucuronides are not fully understood in fish but may function as pheromones during spawning. Testes comprised a much smaller proportion of body weight (0.81%) in cunners exposed chronically to Venezuelan crude oil than unexposed controls (1.33%) (Payne et al., 1978). Decreased plasma 17-β-estradiol levels were found in female flounder after chronic dietary exposure to PAH for 12 weeks but no effect was found on ovarian growth (Monteiro et al., 2000).

Tests for circulating reproductive hormones are highly relevant to reproductive processes and can be applied to field-caught specimens. They have been helpful in understanding stresses caused by other contaminants and seem likely to be helpful in understanding potential effects of hydrocarbons. This is a specialized area of biochemistry that merits more active research and is relevant not only to the effects of hydrocarbons but also to those of alkylated phenolic compounds in produced water.

5.3.5.3.20. Vitellogenesis

Closely related to the reproductive steroids is the control of vitellogenesis, the process of oocyte maturation and yolk incorporation. The egg yolk precursor protein, vitellogenin, is synthesized in the liver of females and is specific to each species. Nicolas (1999) reviewed a number of studies of the effects of PAHs on vitellogenesis in fish. It is clear that PAHs can inhibit vitellogenesis, although the mechanisms responsible are not clear. Johnson et al. (1988) found impaired ovarian development in English sole from contaminated areas of Puget Sound and the effect on maturation was related statistically to the induction of AHH enzyme activity. Since PAHs are potent inducers of AHH activity, along with related structures like PCBs, it appears that exposures to these materials might be expected to inhibit ovarian maturation. The potential for effects of PAHs on vitellogenesis is probably greatest in chronically contaminated sites and perhaps with species like flounders that associate closely with sediment. However, lower-boiling hydrocarbons can also affect sexual maturation. Chronic exposure (8 weeks) of female Atlantic croaker (*Micropogonias undulates*) to WSF (2.5 or 5%) or naphthalene (0.5 and 1 ppm) blocked sexual maturation or reduced gonadal growth and induced oocyte atresia (Thomas and Budiantara, 1995).

A number of other environmental chemicals possess the ability to alter production of vitellogenin in fish, among them the alkylated phenols (reviewed by Gross et al., 2003). Frost et al. (2002) tabulated concentrations of alkylated phenols ac-

tive in affecting the synthesis of vitellogenin. The two NOEC values for effects on vitellogenesis in carp (*Cyprinus carpio*) exposed to pentylphenols were 90 and 320 µg/L; corresponding values for other reproductive measures were lower. Lower values of 1.6 and 10 µg/L were given for rainbow trout exposed to octylphenol. For roach (*Rutilus rutilus*) the value was 10 µg/L. A lower NOEC of 6.4 µg/L was reported for rainbow trout exposed to nonylphenol. In general, the alkylated phenols affect vitellogenesis when exposures in the low µg/L levels are experienced for a few days.

5.3.5.3.21. Oedema or ascites fluid

Several authors have noted oedema or ascites in fish exposed to oil. Pacific herring larvae taken in May 1989, from oiled areas of Prince William Sound after the *Exxon Valdez* spill in March 1989, were compared with larvae from unoiled areas (Marty et al., 1997a). Larvae were exposed to a graded series of dispersions of oil in water in the laboratory. Among several lesions found, one of the most consistent was ascites. In these fish the peritoneal cavity was distended with fluid and sometimes the pericardial space was expanded. Linden (1976b) reported similar results with Baltic herring. Oedema was induced in young rainbow trout by exposing them chronically to the WSF of Norman Wells crude oil (Lockhart et al., 1996). Barron et al. (2003) reported yolk sac oedema in herring larvae exposed to weathered Alaska North Slope oil. This effect, like most others, is not specific to petroleum hydrocarbons since it has also been induced by experimental exposures to chlorinated hydrocarbons (e.g., Helder, 1980). However, it is simple to measure and offers a convenient test in spill investigations.

5.3.5.3.22. Diseases

Several studies have suggested interactions between exposure to petroleum hydrocarbons and diseases. There are good reviews of diseases in fish (e.g., Hodgins et al., 1977; Sindermann, 1979) but the causative linkages between exposure to petroleum and diseases are often complicated by the possibility of other, unrelated causes. Russian studies of young cod exposed to hydrocarbons noted a sharp drop in the number of leukocytes, offering a plausible mechanism for reduced resistance to infectious diseases (Kosheleva et al., 1997).

The examination of PAH metabolites in bile of fish has provided statistical evidence linking exposure to PAHs with diseases. The levels of these metabolites were related statistically to neoplasms and other liver lesions in English sole from Puget Sound (Krahn et al., 1986). The geographical distribution of these pathologies was consistent with contaminants identified in sediment, especially PAHs (Malins et al., 1984). Several studies have shown associations between various types of cancers in fish and exposures to these contaminants (e.g., Baumann, 1984; Black and Baumann, 1991). Tumours in brown bullheads from the Black River, Ohio, a tributary to Lake Erie, have been reported (e.g., Baumann et al., 1987) to be associated with high levels of PAHs in the sediment. (The source of the PAHs was a coking plant.) The tumours were detected in relatively young fish with higher frequencies of liver tumours in three-year-old fish than in two-year-old fish. The incidence varied seasonally with higher frequencies in the autumn than the spring. These studies, although not 'Arctic', serve to illustrate the effects that high levels of PAHs in sediment can have on fish. Some PAHs have proven to be carcinogenic in well-controlled laboratory experiments. Dimethylbenz[*a*]anthracene induced liver neoplasms in two species, *Poeciliopsis lucida* and *P. monacha* (Schultz and Schultz, 1982). Similarly, benzo[*a*]pyrene was shown to be carcinogenic in rainbow trout (Hendricks et al., 1985). Rainbow trout were injected with benzo[*a*]pyrene monthly for 12 months and then held for an additional six months before being killed for examination; 46% of treated fish had liver tumours as compared with only 1 fish (4%) in controls. Tumours in fish are valuable indicators of chronic states of contamination of the habitat with PAHs and possibly other materials and for follow-up studies at intervals after a spill. They are not suitable for monitoring acute damage by spills or for furnishing evidence that immediate damage was induced by a spill.

Sindermann (1979) described fin erosion as the 'best known and least understood disease of fish from polluted waters'. Fin erosion in fish exposed to oil has been reported frequently but it is not always clear whether this should be considered a deformity or a disease induced by the exposure. Fin erosion can be induced by exposure to oil (e.g., Minchew and Yarbrough, 1977; Giles et al., 1978; Woodward et al., 1981, 1983). Linden (1975, 1976b) described eroded fins and deformed bodies with abnormal flexures in larval Baltic herring exposed to crude oil or oil/dispersant mixtures. Caudal fin erosion in all fish surviving chronic exposure to Wyoming crude oil was reported by Woodward et al. (1981). Figure 5.12 illustrates a control and a treated fingerling rainbow trout after 56 days of exposure to Norman Wells crude oil (Lockhart et al., 1996). However, it appears that fin erosion can be caused by several environmental stressors. Sindermann (1979) cited references to studies in which PCBs, crude oil, lead, zinc, cadmium, and low levels of dissolved oxygen have all induced fin erosion in fish. Wellings et al. (1976) reported fin erosion in starry flounder and English sole from the Duwamish River in Washington. This river has been contaminated by PAHs but the fish there have been exposed to other potential causative agents as well and so it is not clear what the cause(s) of the fin erosion were. The authors suggested that the disease is probably caused by actions of several factors including chemical

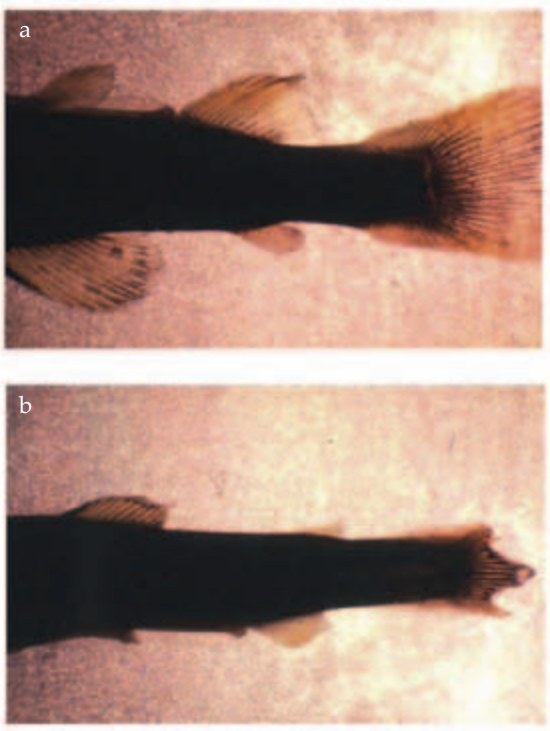

Figure 5.12. Fins of young rainbow trout either (a) untreated or (b) after 56 days of exposure to the water-soluble fraction of *Norman Wells* crude oil (Lockhart et al., 1996).

pollution, physical factors, mechanical injury, and others. Sometimes there is an association with bacterial pathogens but not always. Fin erosion also occurs as a normal part of the life cycle of some fish species, for example, reproductive activity associated with moving substrate for nests.

Fremling (1981) described a spill of No. 6 fuel oil into Lake Winona, Minnesota. Oil was discovered on the lake surface on 15 April 1979, when the lake was still about 75% covered by ice. On 30 May large numbers of dead and dying bluegill sunfish were seen floating on the surface. The cause of death was determined to be infection with *Flexibacter columnaris*. Spawning bluegills dig shallow nests in the sediment at this time, an activity that brought them into contact with oil in the sediment. This type of disease is common in stressed fish and was concluded by biologists at the time that the oil had added an additional critical stress to those of winter, nest building, and crowding, with the result that about 190 000 fish were killed by the infection.

Following the spill of the M/S *Eira* in the shallow Quark area of the Baltic Sea in 1984, perch (*Perca fluviatilis*) were reported to suffer from skin diseases, and the size of their liver and gonads had changed (Ikävalko, 2005).

A brief description was given of 'white eye syndrome' of shrimp (*Penaeus aztecus*) following chronic exposures to Empire Mix, Nigerian, and Saudi Arabian crude oils in experimental ponds (Minchew et al., 1979). No incidences of this disease were observed in unexposed controls. Histopathologically, the disease was described as 'liquefactive necrosis of the crystalline cone, ommatidia, and all associated structures in the diseased areas'. It was not established whether bacterial or viral infections may have been involved but the association with exposure to oil was clear.

5.3.5.3.23. Parasites

Khan and Kiceniuk (1983) exposed winter flounder with a natural infection of the trematode *Steringoporus furciger* to the WSF of Venezuelan crude oil or to sediment contaminated with the same oil for periods of 34 and 160 days. After treatment with either contaminated water or sediment, the incidence of the parasitism was reduced. Similar experiments were conducted with Atlantic cod infected naturally with the acanthocephalan, *Echinorhynchus gadi* using either Hibernia or Venezuelan oils with very similar results. Apparently, in these cases, the oil affected the parasites more than it affected the fish.

5.3.5.4. Seabirds

Although crude oils are low in acute toxicity when ingested by adult birds, a wide range of sub-lethal physiological and histological effects in birds caused by exposure to crude oils have been reported. The effects include reduced growth (Peakall et al., 1985) and increased adrenal gland weight (Peakall et al., 1982), changes in liver and spleen weights (Szaro et al., 1981), renal necrosis and haemolytic anaemia (Fry and Lowenstine, 1985). Many of the pathological and histological changes reported in laboratory dosing experiments duplicated the effects observed in seabirds exposed to crude oils in the wild (Fry and Lowenstine, 1985). Long-term exposure to 4000 ppm of petroleum hydrocarbons caused no visible signs of toxicity in mallards, however liver weight and hepatic blood flow increased substantially (Patton and Dieter, 1980). Increased metabolic rate has been reported in some studies, while others note increases in thyroid hormones and hepatic cytochrome P-450 enzymes but no increase in metabolic rate (Jenssen et al., 1990). Haemolytic anaemia and associated changes in red blood cells were reported in herring gull (*Larus argentatus*) nestlings given daily doses of Prudhoe Bay crude oil (Leighton et al., 1983, 1985), while significant disruption of hormone levels in herring gulls and black guillemots (*Cepphus grylle*) occurred after single oral doses of crude oil (Peakall et al., 1981).

In one study, the reduction in growth rate of black guillemot after a single exposure of crude oil was considered to be significant enough to possibly reduce the survival of chicks that would undertake a long migration soon after fledging (Peakall et al., 1980). Boersma et al. (1988) also noted similar delays in fledging for storm petrel (*Oceanodroma furcata*) chicks but the increase (about two days longer to fledge for chicks exposed to crude oil) for that species was judged to be minor. A detailed assessment of the continuing effects of petroleum hydrocarbon exposure in black oystercatchers (*Haematopus bachmani*) several years after the *Exxon Valdez* spill showed elevated levels of hydrocarbon in the diet of chicks due to feeding in areas with residual contamination, and slightly lower growth rates in those chicks, but no difference in fledging success on heavily oiled shorelines (Andres, 1999). In contrast, pigeon guillemots showed little recovery ten years after the *Exxon Valdez* spill, possibly due to continuing effects in the adults from exposure to oil and reduced availability of key prey species (Golet et al., 2002).

Many of the physiological and histological effects in oil-exposed birds may lead to significant changes in the hepatic enzymes and immune system (Briggs et al., 1997; Fairbrother et al., 2004; Grasman, 2002). Changes to the gut lining, liver function, and anaemia can lead to malnutrition and a reduction in immune function, making the exposed birds far more susceptible to death by bacterial and viral infection. A field study of tree swallows (*Tachycineta bicolor*) feeding on tailings ponds associated with Canada's bitumen tar sands showed elevated levels of hepatic cytochrome P-450 levels in response to elevated PAHs in their diet, but field tests of immune response showed no immunosuppression in the most highly exposed nests (Smits et al., 2000). The authors hypothesized that exposure was too low to cause a prolonged biological effect. Lesser scaup (*Aythya affinis*) feeding in the heavily polluted Indiana Harbor Canal showed elevated levels of three cytochrome P-450 enzymes as a result of PAH exposure, and also showed a direct correlation between PAH concentration in tissues and chromosomal damage in blood (Custer et al., 2000). These results are consistent with other studies showing increased chromosomal damage in common eiders (*Somateria mollissima*) exposed to higher PAH levels in the Baltic Sea relative to the Beaufort Sea population (Matson et al., 2004).

A series of studies also show significant effects of oil exposure on circulating hormone levels in seabirds, including those hormones controlling reproduction. A review of the effects of petroleum on the avian endocrine system summarizes several changes in hormone levels, including reduced levels of hormones such as progesterone and estradiol, which would ultimately reduce the rate of reproduction (Scanes and Mcnabb, 2003). Mallard ducks fed crude oil showed an 84% decline in the rate of egg-laying and a 33% decline in eggshell thicknesses (Holmes et al., 1978). The addition of 1–3 mL of either of two crude oils to 100 g of dried food caused significant declines in the rate of egg laying in exposed birds. A similar decline in reproduction was observed in Cassin's auklet (*Ptychoramphus aleuticus*), a member of the alcid family of birds that are among the most frequently exposed to marine oil pollution, fed 300–1000 mg of Bunker C fuel oil (Ainley et al., 1981).

A range of sub-lethal effects have been reported in embryos exposed to crude oil and petroleum products. The most common effect reported in these studies is liver necrosis and oedema (Couillard and Leighton, 1991a). In a sequential study of the effects in developing chicken embryos dosed with 10 μL of Prudhoe Bay crude oil on Day 9 of incubation, Couillard and Leighton (1991a) reported that effects in the liver were observed two days later and persisted until Day 18. Histological effects were observed in liver, kidney, and spleen. A related study (Couillard and Leighton, 1990) showed that the time of exposure is a major factor in determining toxicity, with a 16-fold decrease in sensitivity between Day 8 and Day 9 of incubation, probably as a result of the development of specific membranes within the egg.

Oil has also been shown to be teratogenic in developing embryos. Hoffman and Albers (1984) reported a range of effects, including malformed bill, fused vertebrae, and stunted growth after application of small amounts of crude oils.

5.3.5.5. Concluding comments

Many sub-lethal responses have been described for aquatic animals exposed to oils and the list of responses grows as new research is reported. The sub-lethal studies are useful for monitoring fish and other mobile animals because population studies are so difficult statistically. Sub-lethal responses are typically observed at concentrations of oil in water in the range of 1 μg/L or higher. Sub-lethal effects may be the only way in which effects on pelagic animals can be demonstrated convincingly. The toxicology of oils is usually expressed as a function of the concentration of low-boiling aromatic compounds in water but these compounds are rarely reported from spill sites. Thus the measurements that would help to link the exposures and toxicology are rarely available. PAHs were measured in seawater of Prince William Sound after the *Exxon Valdez* spill and concentrations reached the low μg/L level (Short and Harris, 1996), below the levels usually associated with acute toxicity even if the entire suite of PAHs present consisted of the low-boiling derivatives of benzene and naphthalene. However, these levels are within the range associated with sub-lethal effects and such effects were found (Peterson et al., 2003) at heavily oiled sites. The levels also exceed some NOEC values for PAHs. NOEC values are typically in the low μg/L level, below (by definition) the lowest levels found to exert sub-lethal or lethal effects.

Despite its low acute toxicity, the ingestion of oil causes a number of sub-lethal effects in seabirds, many of which disappear after exposure ceases. The most vulnerable stage for seabirds is the developing embryo, which can be killed by μL amounts of some crude oils. Exposure to these levels can occur from the adult birds that transport oil from small, diffuse spills to the nest on breast feathers, and contaminate the eggs during incubation. Other than large oil spills that can directly kill large number of adult birds, this mechanism might be capable of significant loss of productivity in colonies and nesting areas close to local sources of oil pollution.

Sub-lethal effects observed in laboratory exposures and in case studies of accidental or planned spills offer the best means of understanding the biological effects of oil and hydrocarbons in the environment. These effects are most readily interpreted at the levels of tissues, organs, and individual animals. Their power is with their sensitivity and their weakness is with their extension to populations. An active area of research in future years will be testing for linkages between sub-lethal effects on individuals and effects on populations.

Table 5.6. Standardized NOEC values for several hydrocarbons to aquatic organisms (Frost et al., 2002). Tests were rated for validity and only those considered valid are reported here. NOECs were often calculated as LOEC/2.

Hydrocarbon	Type of organism	Species	NOEC endpoint	NOEC, μg/L
Benzene	Crustacean	*Cancer magister*	20 days, survival	170
	Fish	*Morone saxalis*	28 days, growth, physiology	1550[a]
Toluene	Algae	*Selanastrum capricornutum*	8 days, growth inhibition	9400
	Crustacean	*Daphnia magna*	21 days, reproduction	1000
		Ceriodaphnia dubia	7 days, reproduction	9000
	Fish	*Oncorhynchus kisutch*	40 days, growth	1400
		Pimephales promelas	32 days, growth	4000
		Cyprinodon variegatus	28 days, mortality	3200
Naphthalene	Crustacean	*Cancer magister*	40 days, duration of larval development	21
	Fish	*Pimephales promelas*	30 days, hatching and growth	450
		Pimephales promelas	30 days, mortality	1800
		Oncorhynchus gorbuscha	5 weeks, growth (body wet weight)	120
		Oncorhynchus gorbuscha	5 weeks, growth (body length)	560
		Oncorhynchus gorbuscha	5 weeks, growth (body weight)	370
Phenanthrene	Algae	*Anabaena flos-aqua*	2 weeks, growth	600
	Crustacean	*Daphnia magna*	21 days, reproduction	21
		Daphnia magna	21 days, reproduction	150
		Daphnia pulex	7–11 weeks, reproduction	110
		Daphnia pulex	7–11 weeks, growth (length)	60
		Daphnia pulex	21 days, reproduction/growth	1.5
	Fish	*Oncorhynchus mykiss* fry	60 days, behaviour	19
		Brachydanio rerio	28 days, growth (length)	28
		Brachydanio rerio	28 days, growth (wet weight)	49
Anthracene	Crustacean	*Daphnia magna*	21 days, reproduction/growth	1.9
		Daphnia magna	21 days, growth	2.2
		Daphnia magna	21 days, reproduction	1.1
		Daphnia magna	21 days, reproduction	0.95
	Fish	*Pimaphales promelas*	42 days, hatching success	6
Chrysene	Crustacean	*Daphnia magna*	21 days, mortality and reproduction	1.4
Benzo[*a*]pyrene	Fish	*Brachydanio rerio*	42 days	6.3

[a] calculated from LOEC=3100/2.

5.3.6. No-effect levels, low-effect levels, and safe levels of hydrocarbons

In principle, it is not possible to prove a universal negative. However, toxicologists have tried to approximate this by using a graded series of exposures and defining the 'lowest observable effect level' (LOEL) or 'lowest observable effect concentration' (LOEC) or 'predicted no effect concentration' (PNEC). The 'no observable effect level' or 'no observable effect concentration' (NOEL or NOEC) is often taken to be the next lower exposure than that at which an effect was observed. Sometimes it is calculated by dividing the LOEL by a factor, often two (Frost et al., 2002). The danger in this approach is that the response being studied may not be the most sensitive effect shown by the test animals. However, NOEC values do provide helpful guidance for concentrations that should be avoided. NOEC values were calculated for a number of hydrocarbons and organisms (Frost et al., 2002; Table 5.6) and the most sensitive of these was about 1 µg/L. Experiments with cutthroat trout exposed to Wyoming crude oil in a continuously mixing dilutor apparatus suggested a LOEL of about 24 µg/L (Woodward et al., 1981) and similar experiments with a refined oil (collected from a seepage site on the North Platte River below a refinery in Casper, Wyoming) suggested a LOEL in the range of 24–39 µg/L (Woodward et al., 1983). The US Environmental Protection Agency has suggested that a value of one one-hundredth (0.01) of the 96-hr flow-through LC_{50} can be used as the LOEL. DeGraeve et al. (1982) suggested a NOEC value for naphthalene of 450 µg/L based on exposures of eggs and larvae of fathead minnows. Devlin et al. (1982) used a flow-through design to estimate a LOEC value of 6000 µg/L for toluene to larval fathead minnows. Statoil scientists tabulated PNEC values from published literature and found that they generally ranged from <1 µg/L (anthracene) to almost 10 mg/L for a diverse group of organisms and exposure conditions (Frost et al., 2002). A considerable body of Russian work on the aquatic toxicology of 'natural gas', gas condensate, and methanol to a variety of Barents Sea taxa suggested the range of NOEC values shown in Figure 5.13. For example, the NOEC for methanol to young cod was high at 1000 mg/L while that for gas condensate was 0.5 mg/L and for natural gas 1.8 mg/L.

Norway uses NOEC/LOEC values as the basis for environmental risk assessment which supports decision-making in the oil and gas industry and in government. The goal is to link biomonitoring procedures to risk assessment using biomarkers as a 'bridge'. LOECs and species sensitivity distributions for biomarkers are important tools in this initiative.

5.3.7. Effects of drilling fluids, drill cuttings, and produced water

In spite of any sub-lethal effects or sediment quality issues, fish of some species seem to be attracted to production platforms (e.g., Atlantic cod; saithe, *Pollachius virens*; ling). Jorgensen et al. (2002) used surgically implanted ultrasonic transmitters to show that about half the cod so tagged near a platform remained there and some moved to neighbouring platforms. Experimental gill nets near platforms yielded bigger catches than nets set further away (Lokkeborg et al., 2002). It is not clear what features of the platforms are attractive to the fish but observations like these give rise to debate about whether or not platform structures might be left in place after production has ended to serve as artificial habitat (e.g., Baine, 2002).

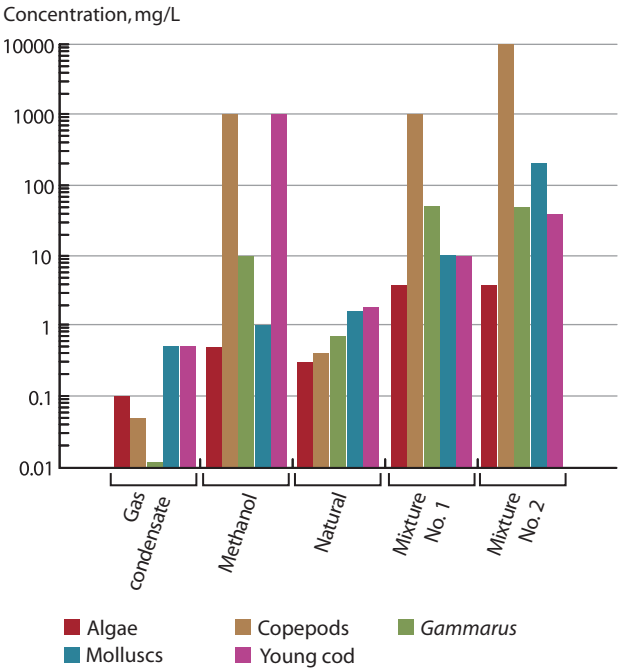

Figure 5.13. Concentrations at which no observable effects are apparent in a range of Barents Sea taxa exposed to natural gas, gas condensate and methanol.

Armsworthy et al. (2005b) tested several used and fresh synthetic oil-based drilling muds for effects on scallops (*Placopecten magellanicus*) in chronic exposures (53–72 days) and found little or no treatment-related mortality. However, sublethal effects on tissue weights were found at concentrations below 1 mg/L for several of the muds. The water clearance rate for scallops was defined as the amount of water filtered per individual per hour. This rate was reduced by exposures to the muds at concentrations as low as 0.1 mg/L. The authors reported that the suspended solid components of the muds had the same impact as the whole muds. When scallops were examined from sites at graded distances from the Hibernia platform during a period when 455 t of cuttings, including 80 t of synthetic-base oil were discharged, the effects were negligible (Cranford et al., 2005). For the most part, concentrations above 10 mg/L are found only in very close proximity to sources (Neff, 1987). The zone within a few kilometres of drilling has been the area at greatest risk and that zone appears to have become smaller as water-based muds replace those used formerly.

Accumulated drill cuttings from near the North West Hutton platform (U.K. section of the North Sea, where oil-based muds were discharged) were toxic to several benthic species. Grant and Briggs (2002) used benthic invertebrates and the Microtox® test to determine the toxicity of the sediment and then compared the results statistically with their chemical composition. Their results indicated that toxicity was associated with the hydrocarbon content.

The North Sea is probably the most studied offshore gas and oil production area in the world. The influence of produced water was initially assessed through modelling contaminant levels (PAHs) in blue mussels and passive samplers in efforts to address dispersion and bioavailability. The modelled levels sometimes agreed well with passive samplers and blue mussels but large discrepancies sometimes occurred (Utvik and Gärtner, 2007). Following an international workshop (Hylland et al., 2002a,b) biological effects methods have increasingly been incorporated along

with chemical measurements to monitor the environmental consequences of produced water discharges in the Norwegian sector of the North Sea. Four locations were identified: 500 m, 2000 m, and 10 000 m from the platform in the projected direction of the effluent plume as well as a reference location. Three complementary strategies were used to investigate biological effects in the water column: field collection of organisms; caging of blue mussels and Atlantic cod; and *in vitro* and *in vivo* testing of extracts from produced water, seawater and passive samplers (Hylland et al., 2007). Effects were found at locations closest to the platform for histological hepatic biomarkers (lysosomal responses, neutral lipid accumulation, peroxisome proliferation, histopathology) in saithe (Bilbao et al., 2007b), but there were no apparent effects on other biomarkers in saithe such as hepatic EROD/CYP1A (McIntosh et al., 2007) or plasma vitellogenin (Scott et al., 2007). Smolders et al. (2007) observed effects on cellular energy allocation (CEA) in blue mussels caged for five weeks at the locations closest to the platform. There were also clear effects on blue mussels at all locations in the plume in histological endpoints (Bilbao et al., 2007a). Results for bile PAH metabolites clearly indicated that the cod had been exposed to PAHs in the plume (Aas et al., 2007), a result which was also reflected in responses observed by Danischewski (2007) for GST (glutathione-S-transferase) and histopathologically by Feist et al. (2007) and Bilbao et al. (2007a).

Long-chain ($>C_4$), *para*-substituted alkylphenols are known xenoestrogens that can induce endocrine disruption in fish even in very low concentrations (Jobling and Sumpter, 1993; Servos, 1999). *Ortho*-substituted, *meta*-substituted and short-chain alkylphenols have little or no estrogen effect (Routledge and Sumpter, 1997). Thomas et al. (2004a,b) used *in vitro* methods (yeast estrogen screen) to detect estrogen receptor (ER) agonists in produced water from North Sea oil installations. They found that produced water contained ER agonists which they identified as C_1 to C_5 and C_9 alkylphenols with a possibility of others. The activity corresponded to estradiol equivalents from the low ng/L to 91 ng/L. This corresponds well with the levels of alkylphenols, considering that alkylphenols are of the order of 1000 times weaker ER agonists (Routledge and Sumpter, 1997). The question arises whether produced water discharges can produce an endocrine-disrupting effect in the marine life around production platforms. Produced water is rapidly diluted after being discharged; computer simulations show 30 and 100 times dilution respectively 10 m and 100 m from the outlet. Further dilution is, however, slower, and the model showed that 1:1000 dilution occurred as far as 1000 m from the outlet (Neff, 2002). There are no empirical data on concentrations of long-chain alkylphenols in the sea around offshore installations. One study from a North Sea offshore installation showed that phenol and light alkylphenols (C_1–C_4) occurred at concentrations of 486 and 140 ng/L, respectively (Riksheim and Johnsen, 1994). The most potent alkylphenol, 4 *tert*-octylphenol, induces endocrine effects in fish in the low µg/L range (Jobling and Sumpter, 1993). The long-chain alkylphenols ($>C_5$) are only present in produced water in the low µg/L concentrations and because of the high dilution factors from offshore discharges it is not likely that endocrine effects occur except in the close vicinity of the discharge point. This is the conclusion of Myhre et al. (2004) in a risk assessment of reproductive effects of alkylphenols in produced water on fish stocks in the North Sea.

Very recent research and monitoring activities in the North Sea have indicated that effects of produced water may extend further than would be expected from dispersion and risk modelling (Hylland et al., 2006). In that study, Atlantic cod, haddock, and saithe were sampled in three areas in the northern North Sea: the Tampen region (large inputs of produced water and a history of discharges of oil-based muds), the Sleipner region (small inputs of produced water) and the Egersund bank (a reference area, no nearby offshore activity). Haddock were found in sufficient numbers in all areas. Higher concentrations of some PAH-metabolites were found in fish collected in the area with the highest input of produced water (Tampen) compared to the other two areas (Hylland et al., 2006). PAH tissue residues were, however, close to the detection limit in all three areas (Klungsøyr and co-workers, Institute of Marine Research, Bergen, Norway, pers. comm.) and there were small differences among areas in the concentrations or patterns of PAHs. There were clearly elevated levels of hepatic DNA adducts in haddock from the areas influenced by produced water and apparent effects on hepatic antioxidant defence mechanisms as well. These adducts may have resulted from exposure to produced water or, perhaps more likely, to older oil-containing cuttings. Haddock collected in the area with highest produced water inputs had a different lipid composition in fillets, but whether this was related to produced water or offshore activity is not clear (Hylland et al., 2006).

5.3.8. Noise in the marine environment

Natural sounds are everywhere in the sea depending on the state of the sea, the depth, ice conditions, and the kinds of animals present. To these natural background sounds are added various sounds attributable to human activities including normal vessel travel and other sounds beyond oil and gas operations which define the ambient noise levels. Ambient noise levels in the Alaskan Arctic range from 68 to almost 100 dB in the Chukchi Sea (Brueggeman et al., 1990), and under sea ice noise levels (fracturing ice) can increase to 124–137 dB (Buck and Greene, 1979). The sounds typical of the oil and gas industry are mainly from seismic airguns, production facilities, and ships. Much research has been done on the effects of sound on fish and marine mammals. Sound is transmitted more efficiently in water than in air and aquatic animals are sensitive to it over a range of frequencies and intensities. Low-frequency sounds are especially well propagated through water. With the excellent transmission of sound in water and the relatively poor transmission of light, sound is often regarded as the most important sense possessed by some aquatic animals. Sound dissipates as it propagates away from its source. Seismic airgun noise is made up of low frequencies of 10–3000 Hz with a loudness of about 250 dB near the source. The sound is emitted as a series of discharges of compressed air from an array towed behind a ship. The noise is directed downwards and the levels emitted to the side of a towed array may be much lower.

5.3.8.1. Effects of seismic activity and noise

There have been regulatory and technological changes in the conduct of seismic surveys. For example, new airguns emit less energy that old ones. Passive acoustic monitoring is used to hear whales and seismic shooting is shut down when whales are nearby.

5.3.8.1.1. Fish and invertebrates

5.3.8.1.1.1. Detection and production of sound

Audiograms have been developed for over 50 species of fish, representing less than 0.2% of fish species (Parker, 1903 and Bigelow, 1904 in Kenyon et al., 1998). Hearing bandwidth is quite variable and may range from 100 Hz to 180 kHz (Dunning et al., 1992; Nestler et al., 1992; Astrup and Møhl, 1993; Mann et al., 2001). Sounds from seismic airguns are normally considerably below 1 kHz.

Fish detect sound in two different ways: sound moves directly from water to hair cells that stimulate the otoliths or indirectly from pressure generated by the swim bladder or other gas bubbles near the ear, which transmits motion to the otoliths (Fay and Popper, 1999). Fish also detect water movements by the lateral lines, which operate in concert with hearing to detect sound (Sand, 1981; Coombs and Montgomery, 1999). The mode of sound detection depends on how well developed the swim bladder is in a given fish species. For instance, elasmobranchs (sharks and skates) lack swim bladders and rely on hair cells (Corwin, 1981).

Invertebrates also lack swim bladders and detect sound differently to fish. Invertebrates are particularly sensitive to substrate displacements but stimulation seems to be poor relative to that in fish. Some crustaceans may be able to detect sound from vibrations by way of specialized organs (Budelmann, 1992; Popper et al., 2001).

Many fish and invertebrates also use sound production to communicate. Acoustic communication is most often associated with territorial behaviour, mate finding, courtship, and aggression. Sound production may also be used for long-distance communication and communication during conditions of poor underwater visibility (Zelick et al., 1999).

5.3.8.1.1.2. Pathological effects

In water, acute injury and death of organisms exposed to seismic energy generally occurs when the received pressure is both high and it quickly rises and decays (Hubbs and Rechnitzer, 1952). These characteristics when applied to today's seismic airgun arrays result in the expected pathological zone for fish and shellfish occurring within a few metres of the seismic sound source (Buchanan et al., 2004). Some cod and plaice (flatfish) die within 48 hr of very close exposure to seismic pulses (Matishov, 1992), which typically involve experiments of caged fish exposed to high cumulative levels of seismic energy. There are, however, many more examples of no mortality of fish exposed to seismic sources (Falk and Lawrence, 1973; Holliday et al., 1987; Sverdrup et al., 1994; La Bella et al., 1996; Santulli et al., 1999; McCauley et al., 2000a,b, 2003; Bjarti, 2002; IMG, 2002; Hassel et al., 2003). Similar outcomes have been reported for fish eggs and larvae where seismic impacts were similar to experimental controls and mortalities were limited to exposures very close (0.3–3 m) to the airguns (Kostyuchenko, 1972; Dalen and Knutsen, 1986; Holliday et al., 1987; Matishov, 1992; Booman et al., 1996; Dalen et al., 1996). Studies have shown that seismic impacts on fish stocks and their eggs and larvae are far less than natural mortality and, therefore, should be considered insignificant (Saetre and Ona, 1996).

Shellfish have been reported to show no effects to seismic airguns except in rare cases. No effects were reported for snow crabs (adult male, egg-carrying females, and fertilized eggs), Dungeness crab larvae, and bivalves exposed to seismic airguns, even when within 1 m of the source (Pearson et al., 1994; La Bella et al., 1996; Christian et al., 2003). In one isolated case, the development rate of an egg mass from a single female snow crab had a higher percentage of less-developed eggs than the unexposed mass; however, this difference could have been attributed to natural variability due to the small sample size (Christian et al., 2003). Taken together, the available data have not demonstrated any chronic mortality to fish or shellfish from exposure to seismic sounds, except in a few isolated instances during exposure to high pressures very close to the sound source in controlled experiments.

5.3.8.1.1.3. Physiological effects

Very limited studies of biochemical responses by marine fish and invertebrates to acoustic stress have been conducted to assess their potential effects in reducing reproductive capacity and adult abundance. Various measures have been used to assess physiological effects of seismic energy exposure on fish and invertebrates. Measures have included changes in primary stress hormones, adrenaline, cortisol, proteins, enzymes, and cell type counts between exposed and unexposed animals to seismic energy. However, increases in physiological stress of various species (Atlantic salmon, European sea bass, squid, cuttlefish, and snow crab) either do not appear to occur at all or are at most temporary (Sverdrup et al., 1994; McCauley et al., 2000a,b; Christian et al., 2003). The times necessary for these biochemical changes to return to normal are variable but typically short term.

5.3.8.1.1.4. Behavioural effects

Studies have been conducted on the effects of seismic energy on fish and invertebrate abundance, distribution, migration patterns, and catchability. Studies have typically shown that animals exposed to seismic energy return to normal behaviour after cessation of seismic sounds. Sand lance responded to seismic sounds by temporarily increasing their swim rate, shifting their vertical distribution, and exhibiting a startle response (Hirst and Rodhouse, 2000; Hassel et al., 2003). Atlantic salmon and haddock temporarily decreased in abundance and catch rate in response to seismic sounds, but their sensitivity declined with increasing distance from the sound source and they resumed normal behaviour within five days of seismic operations ending (Engås et al., 1996). Demersal fish temporarily swam to the bottom during seismic airgun pulses causing a decline in catch per unit effort (Dalen and Raknes, 1985; Dalen and Knutsen, 1986; Løkkeborg, 1991; Skalski et al., 1992). Other species involved in studies indicating temporary behaviour responses to underwater sound include rockfish (*Sebastes* spp., Pearson et al., 1992), Pacific herring (Schwarz and Greer, 1984), and Atlantic herring (Blaxter et al., 1981). In almost all cases, fish returned to normal behaviour within minutes of seismic operations ending (Engås et al., 1996).

Marine fish inhabiting an inshore reef off the coast of Scotland were observed before, during, and after airgun firing (Wardle et al., 2001). Some startle responses were observed, but the airgun sounds had little effect on the daily behaviour of resident fish.

Studies of caged or confined fish, squid, and cuttlefish have also been conducted to assess effects of seismic sounds on behaviour (McCauley et al., 2000a,b). Behaviour of fish exposed to seismic sounds include startle response, faster swimming, movement to the part of the cage furthest from the seismic source (i.e., avoidance), and eventual habituation. Fish behaviour appeared to return to normal 15–30 min after ending seismic shooting. Squid responded to airgun

firing by releasing ink and/or jetting away from the source and moving to the surface, where the sound intensity was less. These experiments indicate that fish and invertebrate behaviour may change in response to seismic sounds, but that the changes seem to be temporary.

Shellfish behaviour in response to seismic sounds is similar to that reported for most fish. Snow crab displayed no drastic decrease in catch rate before or after an area was exposed to seismic shooting (Christian et al., 2003). Caged crab positioned 50 m below a 7-airgun array, showed no obvious startle behaviours (Christian et al., 2003). However, snow crab catch rates from Newfoundland, Canada, significantly decreased immediately after a pass by a seismic survey vessel. Similarly, a school of shrimp observed on a fishing vessel sounder off Newfoundland shifted downwards and away from a nearby seismic source. Effects, however, were temporary in both the snow crab and shrimp observations (Buchanan et al., 2004).

5.3.8.1.2. Marine mammals

The effects of noise on marine mammals were reviewed thoroughly by Richardson et al. (1995). Marine mammals respond to noise but responses have been temporary and have not been demonstrated to have a biologically significant impact on individuals or populations. Extensive studies of marine mammal responses to seismic and other oil and gas operations have been conducted in the western Beaufort Sea (US and Canadian) for decades. These studies have documented responses of bearded seal (*Erignathus barbatus*), ringed seal, walrus, bowhead whale, grey whale (*Eschrichtius robustus*), beluga, and polar bear which are briefly described in the following sections. Over 100 000 km of seismic data have been collected in the Beaufort and Chukchi Seas over the past 30 years and the populations of marine mammals have remained robust indicating that impacts have harmed them very little or not at all.

5.3.8.1.2.1. Baleen whales

Baleen whales generally avoid operating airguns, but avoidance radii are quite variable (Malme et al., 1984, 1985, 1988; Richardson et al., 1986, 1995; Ljungblad et al., 1988; Richardson and Malme, 1993; McCauley et al., 1998, 2000a; Miller et al., 1999; Richardson, 1999; Gordon et al., 2004). Whales often show no overt reactions to pulses from large arrays of airguns at distances beyond a few kilometres, even when airgun pulses remain well above ambient noise levels for longer distances. However, baleen whales exposed to strong noise pulses from airguns often react by temporarily deviating from their normal migration route around the sound source and/or interrupting their feeding and moving a relatively short distance away from the sound source. Richardson (1997) described four types of reaction by bowhead whales: tolerance (no obvious reaction), subtle changes in behaviour (e.g., reduced durations of surfacings or dives), conspicuous changes in activity (e.g., ceasing to feed), and avoidance (swimming away when seismic or other vessel approaches).

Richardson (1997) suggested that subtle changes in behaviour may occur as much as 72 km from a seismic ship (although these changes were not significant statistically). Overt changes were not observed until sources were much closer, about 7.5 km, however, the whales continued feeding until the ship came within 3.7 km at which point they stopped feeding and moved away. The author observed that the greatest number of bowhead sightings was 10–20 km offshore when no seismic vessels were operating, but 20–30 km offshore when there was seismic activity. The observed changes in behaviour of migrating bowhead and grey whales appeared to be of little or no biological consequence to the animals (Richardson et al., 1995; Richardson, 1999). They simply avoided the sound source by adjusting their track within the natural boundaries of the migration corridors. This observation of more seaward migration is consistent with the experience of whaling captains from coastal Alaska (Albert, 1997). Whalers have observed that bowhead whales move further offshore after seismic exploration. The greater distance from shore makes the whales more difficult to locate and hunt and makes hunting more dangerous. Hauling a carcass a longer distance to shore for butchering also makes spoilage and loss more likely. Richardson et al. (1999) confirmed the displacement of bowhead whales by nearshore seismic activity.

Feeding grey whales near St. Lawrence Island, Alaska, were observed when subjected to airgun noise or to recorded drillship noise (Malme et al., 1987). Only subtle changes were observed in whale behaviour when sources were a few kilometres or less from the whales.

Studies of grey and bowhead whales have shown received levels of pulses in the 160–170 dB range seem to cause avoidance behaviour in a substantial fraction of the animals exposed. Seismic pulses from large airgun arrays often diminish to those levels (i.e., 160–170 dB) at distances of 4.5–14.5 km from the source depending on a variety of conditions. Bowhead whales on their summering grounds in the Canadian Beaufort Sea showed no obvious reactions to pulses from seismic vessels at distances of 6–99 km and received sound levels of 107–158 dB on an approximate rms basis (Richardson et al., 1986); their general activities were indistinguishable from those of a control group. However, subtle but statistically significant changes in surfacing–respiration–dive cycles were evident with statistical analysis. However, bowhead whales usually did show strong avoidance responses when seismic vessels approached within a few kilometres (~3–7 km) and when received levels of airgun sounds were 152–178 dB (Richardson et al., 1986, 1995; Ljungblad et al., 1988). In one case, individuals engaged in near-bottom feeding began to turn away from a 30-airgun array with a source level of 248 dB at a distance of 7.5 km and swam away when it came within about 2 km. Some whales continued feeding until the vessel was 3 km away. This work and a more recent study by Miller et al. (2005) showed that feeding bowhead whales tend to tolerate higher sound levels than migrating whales before showing an overt change in behaviour. The feeding whales may be affected by the sounds, but the need to feed may reduce the tendency to move away.

Bowhead whales migrating west across the Alaskan Beaufort Sea in autumn, in particular, showed avoidance out to distances of 20–30 km from a medium-sized airgun source (Miller et al., 1999; Richardson et al., 1999; Richardson, 1999). From 1996 to 1998, a partially-controlled study of the effect of Ocean Bottom Cable (OBC) seismic surveys on westward-migrating bowhead whales was conducted in late summer and autumn in the Alaskan Beaufort Sea (Miller et al., 1999; Richardson et al., 1999; Richardson, 1999). Aerial surveys showed that some westward-migrating whales avoided an active seismic survey boat by 20–30 km, and that few bowheads approached within 20 km. Received sound levels at those distances were only 116–135 dB. Some whales apparently began to temporarily deflect their migration path when as much as 35 km from the airguns. At times when

the airguns were not active, many bowhead whales moved into the area close to the inactive seismic vessel. Avoidance of the area of seismic operations did not persist beyond 12–24 hr after seismic shooting stopped. These and other data suggest that migrating bowhead whales are more responsive to seismic pulses than summering bowheads, but responses were temporary and short term, and there is no evidence of bowhead whales moving outside their normal migration corridor.

Malme et al. (1986, 1988) found that 50% of a group of grey whales ceased feeding at an average received pressure level of 173 dB to pulses from a single 100 in^3 (1639 cm^3) airgun off St. Lawrence Island in the northern Bering Sea. In addition, 10% of whales interrupted feeding at received levels of 163 dB. Malme at al. (1986) estimated that an average pressure level of 173 dB occurred at a range of 2.6–2.8 km from an airgun array with a source level of 250 dB in the northern Bering Sea. There was no indication that western grey whales exposed to seismic noise were displaced from their overall feeding grounds near Sakhalin Island during seismic programs in 1997 (Würsig et al., 1999) and in 2001. However, there were indications of subtle behavioural effects and (in 2001) localized avoidance by some individuals (Johnson, 2002; Weller et al., 2002), but they were not biologically significant to the population. Cott et al. (2003) reported, based on sightings of 262 individuals, that bowhead whales in the southern Beaufort Sea were displaced from the area of seismic operations as indicated by observations from the seismic vessel. This displacement was sufficient to be detected by aerial surveys of the whales. They noted 'swimming' behaviour and 'diving' behaviour but no increases in 'swimming away' behaviour.

Experiments were conducted on larger numbers of grey whales migrating along the Californian coast. Malme and Miles (1985) concluded that, during migration, changes in swimming pattern occurred for received levels of ~160 dB re 1 µPa and higher, on an approximate rms basis. The 50% probability of avoidance was estimated to occur at a closest point of approach distance of 2.5 km from a 4000-in^3 (65548 cm^3) array operating off central California. This would occur at an average received sound level of ~170 dB (rms). Some slight behavioural changes were noted at received sound levels of 140–160 dB (rms).

Data on short-term reactions (or lack of reactions) of cetaceans to impulsive noises do not necessarily provide information about long-term effects. It is not known whether impulsive noises affect reproductive rate or distribution and habitat use in subsequent days or years. However, grey whales continue to migrate annually along the west coast of North America with amazing regularity despite intermittent seismic exploration and ever increasing ship traffic (Malme et al., 1984). Bowhead whales continue to travel to the eastern Beaufort Sea each summer despite seismic exploration in their summer and autumn range for many years (Richardson et al., 1987). Populations of both grey and bowhead whales grew substantially during this time to the point where grey whales are at or near carrying capacity and bowhead whales are approaching carrying capacity (Angliss and Outlaw, 2005).

5.3.8.1.2.2. Toothed whales

Little systematic information is available about reactions of toothed whales (beluga, narwhal, and killer) to noise pulses. Beluga exhibit changes in behaviour when exposed to strong, pulsed sounds similar in duration to those typically used in seismic surveys (Finneran et al., 2000, 2002). However, the animals tolerated high received levels of sound (>200 dB) before exhibiting aversive behaviours. Beluga summering in the eastern Beaufort Sea may have avoided the area of seismic operations (two arrays with 24 airguns per array) by 10–20 km, although beluga occurred as close as 1540 m to the line seismic operations (Miller et al., 2005). Cott et al. (2003) sighted 18 beluga from a seismic vessel in the Canadian Beaufort Sea, all observed to be swimming away from the vessel. The authors suggested that these whales had an avoidance distance of 10–20 km from seismic operations. Observers stationed on seismic vessels operating off the U.K. from 1997 to 2000 have provided data on the occurrence and behaviour of various toothed whales exposed to seismic pulses (Stone, 2003; Gordon et al., 2004). Killer whales were found to be significantly farther from large airgun arrays during periods of shooting compared to periods of no shooting. The displacement of the median distance from the array was ~0.5 km or more. Killer whales also appear to be more tolerant of seismic shooting in deeper water. There are no data on narwhal response to seismic noises.

5.3.8.1.2.3. Pinnipeds

Monitoring studies in the Alaskan and Canadian Beaufort Sea during 1996 to 2002 provided considerable information about behaviour of seals exposed to seismic pulses (Harris et al., 2001; Moulton and Lawson, 2002; Miller et al., 2005). These seismic projects usually involved arrays of 6 to 16 with as many as 24 airguns with total volumes of 9176 to 25563 cm^3. The combined results suggest that some seals avoid the immediate area around seismic vessels. In most survey years, ringed seal sightings tended to be farther away from the seismic vessel when the airguns were operating than when they were not (Moulton and Lawson, 2002). However, these avoidance movements were relatively small, of the order of 100 m to (at most) a few hundred metres, and many seals remained within 100–200 m of the trackline as the operating airgun array passed by. Seal sighting rates at the water surface were lower during airgun array operations than during no-airgun periods in each survey year except 1997. Miller et al. (2005) also reported higher sighting rates during non-seismic than during line seismic operations, but there was no difference for mean sighting distances during the two conditions nor was there evidence that ringed or bearded seals were displaced from the area by the operations. The behavioural data from these studies indicated that some seals were more likely to swim away from the source vessel during periods of airgun operations and more likely to swim toward or parallel to the vessel during non-seismic periods. No consistent relationship was observed between exposure to airgun noise and proportions of seals engaged in other recognizable behaviours (e.g., 'looked' and 'dove'). Such a relationship might have occurred if seals seek to reduce exposure to strong seismic pulses, given the reduced airgun noise levels close to the surface where 'looking' occurs (Moulton and Lawson, 2002; Miller et al., 2005). Four ringed seals were tagged and tracked in the southern Beaufort Sea (Cott et al., 2003) and they followed the same course through an area of seismic activity as they did in the following year when no seismic activity was underway. Consequently, bearded, ringed, and probably spotted seals (*Phoca largha*, least amount of data on reaction to seismic operations) do not show a strong avoidance reaction to seismic operations. Pinnipeds frequently do not avoid the area within a few hundred metres of operating airgun arrays,

even for large airgun arrays (e.g., Harris et al., 2001). Reactions are typically very localized and confined to relatively small distances and durations, with no long-term effects on individuals or populations.

5.3.8.1.2.4. Pacific walrus

There have been no studies specifically testing the hearing abilities of walrus. However, open-water seismic exploration produces underwater sounds from airgun arrays that may be audible tens of kilometres from the source (Richardson et al., 1995). Such exploration activities could potentially disturb walruses summering in the Arctic Ocean at varying ranges. In addition, source levels at the airguns are thought to be high enough to cause hearing damage in pinnipeds. It is likely that walrus hearing and sensitivities are similar to pinnipeds at close range, and therefore, it is possible that walruses within the 190 dB safety radius of seismic activities could suffer temporary threshold shift (TTS), however, the use of acoustic safety radii and monitoring programs are designed to ensure that marine mammals are not exposed to potentially harmful noise levels. Open-water seismic exploration is normally conducted in ice-free areas, considerably south of the pack ice. It is highly unlikely that more than a relatively few walruses would be present in open-water areas, and therefore, it is not expected that seismic exploration would disturb many walruses or have more than a negligible impact on the individual and no impact on the population. Walrus hauled out on land have been approached by aircraft with reactions ranging up to stampede for water with resulting injuries or deaths, especially of young. Richardson et al. (1995) listed several instances of this, one of which resulted in death by crushing of 21 calves and abortion of two foetuses.

5.3.8.1.2.5. Polar bears

Airgun effects from seismic operations on polar bears have not been studied. However, polar bears on pack ice would be unaffected by underwater sound. Polar bears in the water are unlikely to be affected by seismic noise. Sound levels received by polar bears in the water would be attenuated because polar bears generally have their head above or at or near the surface. Received levels of airgun sounds are reduced near the surface because of the pressure release effect at the water's surface (Greene and Richardson, 1988; Richardson et al., 1995). Consequently, underwater noise is undetectable or weak to polar bears while swimming (Richardson et al., 1995). Bears in the Beaufort Sea region have been recovering from over-harvesting prior to quotas set in Canada in 1968 and cessation of all but subsistence hunting in Alaska in 1972 (Stirling, 2002). The fact of this recovery implies little or no impact of seismic operations during the recovery period. The health of these populations is likely to reflect the health of their prey populations which primarily include ringed and to a lesser degree bearded seals but also walruses and other marine mammal species.

5.3.8.2. Effects of noise from sources other than seismic shooting

Marine mammals react to ship noise even when there is no seismic noise. When ice formed a barrier to whale migration in Lancaster Sound (Canada), and ice-breaking ships approached, the observed behaviour of the whales (beluga and narwhal) and recordings of their vocalizations indicated that they were aware of a ship when it was as much as 80 km away; when a ship was 35–50 km away, beluga moved to avoid it. Narwhal, however, moved very little and did not avoid the ship until it was much closer (Finley and Davis, 1986). Detection distances were considerably greater than reaction distances. When ice was present in Lancaster Sound (Canada) background noise was especially evident near the ice edge and consisted largely of vocalizations by marine mammals, notably bearded seal, narwhal and beluga (Finley and Davis, 1984a,b). Noise levels were reduced under fast ice away from edges.

Cosens and Dueck (1987, 1993) listed several studies showing that beluga and other odontocetes are most sensitive to frequencies at or above 10 kHz. Cosens and Dueck (1987) made helicopter surveys of narwhal and beluga in Lancaster Sound in 1986 while several ships passed through. They classified whale behaviour and noted different frequencies of those behaviours when ships were in the vicinity. Cosens and Dueck (1993) recorded sound spectra from three different ships and noted differences among them that helped explain how whales would react to them. At a low frequency of 50.1 Hz, even the noisiest ship would be detectable by beluga at 100 m. However, at a high frequency of 5010 Hz, the same ship would be detectable at 240 km.

Military sonar has been implicated in the stranding of Cuvier's beaked whales (*Ziphius cavirostris*) in the Mediterranean Sea; these strandings are thought to have been caused by loud low frequency active sonar testing by the Greek Navy (Frantzis, 1998). Direct physical injury to several species of cetaceans by formation of gas bubbles in internal organs correlated exactly with military sonar near the Canary Islands (Jepson et al., 2003), however, this type of sound differs from seismic noise and bears little relevance to the oil and gas industry.

The behaviour of beluga in southern Hudson Bay with regard to small boats is of interest. The whales on the east side of the bay have been hunted for decades from small boats fitted with outboard motors. The whales avoid these boats. It is not clear whether hunting has killed the least wary whales leaving a population of wary individuals, or whether the whales have learned to associate the boats with danger and so to avoid them. The whales on the west side of the bay have not been hunted in the Churchill area and whale-watching has become a popular tourist activity there. Beluga appear to be curious about small boats and approach the boats so closely that it has become necessary to fit the boats with robust wire cages around the propellers to avoid injury to whales and damage to motors (Cosens, S. Department of Fisheries and Oceans, Winnipeg, 2006, pers. comm.).

5.3.8.3. Concluding comments

There is no doubt that sounds typical of the oil and gas industry, namely ships and seismic exploration, are detected by marine organisms and that they respond to them. Fish can be killed by explosive charges up to 1000 m away but explosives are used rarely and only under very restricted circumstances (e.g., in shallow water frozen to the bottom). Fish catches have been reduced near seismic shooting at sea; one study measured effects on catches for up to 18 km from airgun seismic operations and observed that catches did not return to normal for several days after seismic operations ceased. Available data suggest that there may be physical impacts on eggs and on larval, juvenile, and adult stages at very close range. For typical source levels associated with commercial seismic arrays, close proximity to the source would result in exposure to high energy levels. However, sound sources used by the oil and gas industry in the Arctic

generate low energy levels. Eggs and larval stages would be unable to escape such exposures, but juveniles and adults would probably avoid them. In the case of eggs and larvae, it is likely that the numbers adversely affected by such exposure would be small in relation to natural mortality. The data available suggest short-term impacts; and those apparent only after exposure at close ranges. Any fish or invertebrate organisms very close to the source would probably represent a small proportion of the population; meaning population effects would probably be undetectable.

Marine mammals seem tolerant of seismic noise although they tend to avoid it when it occurs within a few kilometres of them. They can detect ship noise or seismic noises at greater distances, but may not react to it at great distances. Beluga can detect approaches of ships at distances up to 50 km or perhaps further. However, they do not necessarily move to avoid ships until they are a few kilometres away. Bowhead whales have been observed to migrate at greater distances from shore when near-shore seismic operations are conducted, and that makes subsistence hunting for them much more difficult and dangerous. The behavioural effects of seismic operations are transitory and behaviour returns to normal shortly after the seismic activity ceases. The apparent induction of gas-bubble disease in several cetaceans by military sonar is of interest. This response is not relevant to seismic shooting but may offer a guide for pathological examinations in any future cases in which injury by seismic noise may be hypothesized. Impacts of seismic activity at sea are among the easiest impacts to mitigate. Voluntary observation for marine mammals or fishing activity can guide a seismic ship; if whales or fishing activity are in an area, seismic shooting can be delayed.

5.3.9. Effects of oil and hydrocarbons on natural populations

The effects of the oil and gas industry on animal populations are the effects of greatest interest but are the effects most difficult to establish unequivocally. Biological effects on individual animals and on populations following spills have been described in the open literature and at dedicated symposia (e.g., Wells et al., 1995; Rice et al., 1996; Davies and Topping, 1997). Several factors make population effects difficult to describe: variability in most populations; variable exposure to oil in terms of concentrations, oil composition and time; high mobility of some animals; limitations in pre-exposure population data; commercial or subsistence harvesting; and other confounding factors such as spatial variability in supporting habitats and intensity of sampling. The measures with recognized importance in assessing fish populations include abundance, age structure, size/age relationships, growth rate, condition, age at sexual maturity, longevity, measures of reproduction such as fecundity and frequency of breeding, normal *versus* abnormal appearance of individuals and organs, and others such as migratory habits. It is extraordinarily difficult to establish whether populations of pelagic fish have been affected by an oil spill in the ocean. Hilborn (1996) listed five methodologies for studying the impacts of the *Exxon Valdez* spill on populations: (1) counts of dead animals, (2) pre- and post-spill comparison of abundance, (3) oiled *versus* unoiled comparisons of abundance, (4) oiled *versus* unoiled comparison of vital rates, and (5) direct experimental oiling. Of these, the most desirable is pre- and post-spill abundance comparison, but pre-spill data are rarely available. It seems unlikely that future studies will have greater resources available than those that were available after the *Exxon Valdez* spill. Studies following the *Exxon Valdez* spill will probably provide a benchmark for the best that can be done for many years to come.

The population level of biological organization is not readily manipulated experimentally except on a very small scale and so the information is generally restricted to observations of small, experimental treatments, accidental spills or areas of natural oil seepage. Nevertheless, many efforts have been made to detect effects on populations and sometimes, clear population changes have been demonstrated, especially following spills or near drilling sites or in relatively small, or contained water bodies. Several areas of natural seepage are known in the Arctic; they seem to offer natural laboratories for the study of effects of hydrocarbons on local populations but they have attracted little research attention.

5.3.9.1. Microorganisms

Some bacteria typically increase in abundance after exposure to hydrocarbons because they can use hydrocarbons as a source of energy. For example, Horowitz and Atlas (1977) found increases in numbers of bacteria capable of using gasoline as a substrate in water and sediment for several weeks following contamination of a lake near Barrow, Alaska. A year after the spill, bacterial abundance was still higher at the more highly contaminated sites (Horowitz et al., 1978). In contrast, Jordan et al. (1978) studied bacteria in a lake 240 km south of Prudhoe Bay, Alaska, treated experimentally with a small amount of Prudhoe Bay crude oil. The numbers of bacteria in the water were higher in the oil-treated area but not significantly so, except for one month out of 11 in which counts were made. Numbers of bacteria in sediment were not affected. Similarly, Bunch (1987) followed populations of bacteria in water following the Cape Hatt experimental oil spill and found little effect. It is not known whether changes in the abundance of bacteria are related to changes in other organisms such as zooplankton. Lavrentyev (1994) reported changes in planktonic ciliate communities in a small, shallow lake heavily contaminated with oil in the Russian Arctic tundra (68° N, 57° E). In July 1989, the concentration of oil products in the water was as high as 11 mg/L, a level that would be acutely toxic to many organisms if it consisted of the low-boiling aromatic fraction. The response was a simplification of the protozoan community structure with low species diversity but enhanced abundance of a few tolerant species.

Arhelger et al. (1977) observed much slower rates of metabolism of dodecane by bacteria in water from the Fletcher T3 ice island than in water from Barrow, Alaska, which in turn had slower rates of metabolism than water from Valdez, Alaska. Their observations may suggest a latitudinal gradient in the activity of oil-degrading bacteria with lower activities at higher latitudes. Experimental studies of the persistence of oil in cold-temperature mesocosms confirm that degradation by bacterial action is relatively slow under these conditions (Siron et al., 1993, 1995). In experiments during spring with a water temperature of 4 ± 1.5 °C, the half-lives of several PAHs in the water column ranged from about 1.5 to 7.5 days (Siron et al., 1995). Curiously, the half-lives were generally shorter at the highest exposure concentration, possibly as a result of stimulation of oil-degrading bacteria. The ability of marine sediment bacteria from coastal Beaufort and Chukchi Sea sediments to metabolize hydrocarbons was found to be quite limited indicating that biodegradation by these organisms would be slow in this region (Braddock et al., 2004).

Table 5.7. Descriptions of benthic organisms and hydrocarbon content of sediments near oil production platforms in the North Sea (Davies et al., 1984a).

Zone	Maximum extent within range, m	Biology	Chemistry
1	0–500	Impoverished and highly modified benthic community (beneath and very close to the platform the seabed can consist of cuttings with no benthic fauna)	Hydrocarbon levels high. Sediments largely anaerobic. Hydrocarbon concentrations 1000 times or more higher than background
2	200–2000	Transition zone in benthic diversity and community structure	Hydrocarbon levels above background. Hydrocarbon concentrations 10–700 times background
3	800–4000	No benthic effects	Hydrocarbon levels return to background. Hydrocarbon concentrations 1–10 times background
4		No benthic effects	No elevation of hydrocarbons

5.3.9.2. Marine algae

Studies in Prince William Sound following the *Exxon Valdez* spill revealed effects on local populations of several intertidal plants and animals, notably on the brown alga *Fucus gardneri* (e.g., Stekoll et al., 1996). Reference and oiled sites were chosen in areas described as exposed rocky, coarse textured, estuarine, and sheltered rocky. Sites were selected to extend over a range of tidal depths up to 3 m of vertical drop. The samples were collected from three regions in 1990 and 1991 following the spill in March 1989. Taxa sampled included algae (*Fucus*), limpets, mussels, barnacles, littorine gastropods, and oligochaetes. Pooled results from over ten thousand pair comparisons showed that substantially more oiled sites were biologically impoverished relative to reference sites than *vice versa*. Van Tamelen and Stekoll (1996) carried out detailed studies of this type on *Fucus* in Herring Bay and found reduced algal cover and reduced reproduction at many oiled sites but almost no oiled sites at which these measures exceeded reference sites. Separate studies on the colonization of rocks oiled by the spill confirmed impaired colonization by *Fucus* (Duncan and Hooten, 1996). The results represented a combination of effects of oiling and effects of aggressive physical cleaning by means such as spraying with high pressure hot water. The authors suggested that the physical effects of cleaning may have had greater effects on the intertidal algae than those that might have occurred if the oil had been left in place. High tolerance of *Fucus vesiculosus* to fuel oil was reported by Notini (1978) from studies following the spill of the *Irini* in Gastviken, Sweden, in 1970. Although *Fucus* itself was hardly affected after that spill, invertebrate animals that use *Fucus* as a substrate for attachment were affected.

5.3.9.3. Invertebrates

Populations of benthic invertebrate animals are sensitive to stress in their habitat (Schindler, 1987) and these animals have offered some of the best opportunities to detect population changes in multicellular organisms (see reviews by Rosenberg and Resh, 1993 and Wright et al., 2000). This sensitivity is probably partly due to the relative immobility of many invertebrate animals and to the tendency of many pollutants including PAHs to become associated with sediment. The main sources of population information on benthic invertebrates have been studies of spills and environmental effects monitoring programs conducted near sources. In an extreme case, benthic populations of some species were found to be essentially absent from Jarla Lake in Sweden, a condition thought to have arisen because of a long period of discharges of oils into the lake (Bengtsson and Berggren, 1972). Some combination of oiled habitat and post-spill clean-up operations resulted in impacts on several key invertebrate species following the *Exxon Valdez* spill. Highsmith et al. (1996) reported injury to a wide range of taxa with detailed results for dominant taxa in the intertidal zone. The dominant invertebrate taxa affected included the limpet *Tectura persona*, barnacles (*Chthamalus dalli, Semibalanus balanoides* and *Balanus glandula*), the mussel *Mytilus trossulus*, periwinkles *Littorina sitkana* and *L. scutulata* and oligochaetes. The abundance and biomass of each of these species were reduced with the exceptions of *C. dalli* and the oligochaetes, whose abundance and biomass increased after the spill. The spill occurred in 1989 and Highsmith et al. (1996) found recovery to be incomplete when sampling was concluded in 1991.

Many studies of benthic invertebrate populations around drilling and production sites have been reported. Davies et al. (1984a) summarized the effects determined by many previous studies as a series of overlapping zones around several platforms where oil-based muds were used in the North Sea. The shapes of the zones varied with currents and the intensity of industrial operations. Their biological and chemical descriptions are given in Table 5.7. The closest zone (0 to 500 m) showed high levels of hydrocarbons, largely anaerobic conditions, and highly impoverished benthic communities; the second zone (200 to 2000 m) was described as a transition zone in benthic diversity and community structure; the third zone (800 to 4000 m) was one where no benthic effects were detected although hydrocarbons were still above background; and the fourth, most distant, zone showed no benthic effects and no contamination by hydrocarbons. Generally effects on the animals were statistically correlated with levels of hydrocarbons measured in the sediment. Gray et al. (1990) used a combination of counts of benthic animals and chemical analyses of sediment around the *Ekofisk* and *Eldfisk* oil fields in the North Sea and were able to identify subtle effects 2–3 km from sources.

The most widespread ecological impacts identified around Norwegian drilling vessels during the period of oil-based drilling muds were in the *Draugen* field in Haltenbanken in 1994, extending for about 5 km (disturbed fauna in an area of 14 km^2, Myhrvold et al., 1995). At other fields faunal disturbances were confined to within 2–3 km in the main current direction. A monitoring program has been in place around Norwegian drilling installations since 1985 and has produced a wealth of information on sediment contamination and responses of the benthic macrofauna to discharges. The main Norwegian fields producing oil/gas in the area covered by this assessment are in the Haltenbanken area of the Norwegian Sea. The benthic communities in the Haltenbanken area, and also in the northern deeper region of the North Sea, show sufficiently strong similarities to

corresponding communities in the Barents Sea to suggest that the responses are relevant for assessing the potential responses of Arctic benthic communities to future drilling discharges (Ellingsen and Gray, 2002).

The outermost area experienced subtle effects such as small changes in the abundance of relatively rare species, and sometimes the complete absence of sensitive species. Closer to a discharge site (e.g., less than 1 km) the disturbance is typically seen as increased dominance of species generally found to prosper during moderate eutrophication such as the bristle worms *Chaetozone setosa*, *Pholoe minuta*, *Pseudopolydora paucibranchiata* and others, and the bivalve *Thyasira sarsi*, as well as the more or less complete disappearance of the main taxonomic groups such as the echinoderms. Close to, and at the surface of the cuttings deposits the main and sometimes the only colonizer is the bristle worm *Capitella capitata*. This species was the sole colonizer on *in situ* substrates of oil-based cuttings during five years in a recolonization experiment (Bakke et al., 1988). This response pattern seems universal on the Norwegian shelf irrespective of latitudinal position or type of organic base drilling fluid used, and is believed to be a response to the organic enrichment caused by the drilling fluids rather than to toxicity (Hartley et al., 2003). Monitoring data have suggested that the prime factor causing faunal disturbance was the organic base fluid (Hartley et al., 2003). Cuttings deposits may contain substantial amounts of heavy metals and barium, which are considered to be essentially non-bioavailable (Hartley et al., 2003). The seafloor near drilling sites had elevated levels of metals, primarily barium, but chromium, lead, copper, nickel, and zinc also, mostly correlated with barium, but without clear correlation to faunal patterns. The threshold for effects of total hydrocarbons to the most sensitive species was reported to be in the range of 10–50 mg/kg dw or just above background for shelf sediment (<10 mg/kg dw). A typical NOEC limit has been proposed at 50 mg total hydrocarbons/kg dw, but this has been questioned in light of far higher levels found without disturbance in benthic communities exposed to oil spills; for example, >1000 mg/kg dw after the *Braer* accident in Shetland (Kingston et al., 1995). The overall impression from the Norwegian offshore monitoring is that disturbance in the fauna should be expected when total hydrocarbons exceed 100 mg/kg dw. Signs of reduction in overall community diversity index values (e.g., the Shannon-Wiener index) are normally not seen until hydrocarbons exceed 100 mg/kg dw.

Oil-based muds are being used less and less and replacements have consistently been found to result in reduced impacts. Norwegian experience has shown that the overall disturbance area seldom exceeds 500–1000 m from the discharge location and the size of the area is seldom larger than 0.5 km^2. If oil-based muds are not used in the Arctic, then including results from experience with oil-based muds could be argued as irrelevant. However, studies with these muds provide valuable guidance for methodologies and types of effects that may be detected for comparison with any drilling technologies. Most recent wells have been drilled with olefin-based fluids. For installations where only water-based cuttings have been discharged there is often no detectable faunal disturbance beyond 250 m (the closest stations in regular monitoring). Newer water-based fluids have technical performance qualities almost as good as the organic-phase fluids, and consequently it is anticipated that future drilling in the Arctic region will use only water-based fluids. According to the 'zero harmful discharge' concept adopted in Norway for the Arctic, discharge of cuttings will probably be allowed only when drilling the top section of a well, where the cuttings are disposed directly onto the seafloor around the well. Experimental studies have shown some effects of whole water-based muds and individual mud constituents on marine organisms but impacts have not been convincingly identified from field monitoring surveys. Effects on fauna appear to be due to physical factors (e.g., burial, smothering, and disturbance of feeding efficiency [in molluscs] through ingestion of clay-size barite or bentonite particles and accumulation in digestive vacuoles) rather than to acute toxicity (Cranford et al., 1999; Neff, 2002; Neff and Gnatek, 2004). The pattern of impacts on benthic marine animals in monitoring Norwegian wells is that oil-based cuttings elicit the most widespread impacts and water-based cuttings the least, if any. Ester-based cuttings have been shown to cause severe but short-lived effects due to their rapid degradation and resulting oxygen depletion in the sediment. Olefin-based cuttings are degraded fairly rapidly, but without causing oxygen deficiency and hence have short-lived and modest effects on the fauna.

Monitoring results have demonstrated a fairly rapid recovery of the invertebrate fauna in areas outside the cuttings deposit footprint after cessation of oil-based drilling. For the *Draugen* field the area of faunal disturbance was reduced from 14 km^2 in 1994 to 0.04 km^2 in 2000 (disturbance to less than 250 m except in one direction; DnV, 2001). Today, few Norwegian fields have areas of faunal disturbance exceeding 1 km^2 and none exceeding 2 km^2. For a series of repeatedly monitored stations around the *Statfjord* field in the northern North Sea the Shannon-Wiener fauna diversity index value increased from between 1 and 2 just after drilling to over 4 three years later (Schaanning and Bakke, 2006). Complete community recovery within three to six years after termination of drilling seems realistic outside the 250 m periphery.

Relatively rapid recovery was noted following a small spill of aviation gasoline into a mountain stream in South Dakota, USA, in 1969. The spill resulted in the death of most of the aquatic invertebrates for at least two miles downstream from the spill (Bugbee and Walter, 1973). Stoneflies and mayflies were most sensitive. In this case, in which upstream reaches were unaffected, populations were affected drastically but recovered partially after one year and completely after three years.

Miller et al. (1978) described important experiments in which tundra ponds in Alaska were treated with Prudhoe Bay crude oil (at rates of 0.24–13 mL/L) and the effects monitored in several taxa of pond organisms. The dominant zooplankton *Daphnia middendorffiana* (O.F. Müller) and fairy shrimps (*Branchinecta paludosa* [Murdoch] and *Polyartemiella hazeni*) were eliminated (Miller et al., 1978; O'Brien, 1978). Effects on aquatic insects were highly species specific. Caddis fly (*Asynarchus*) larvae and pupae and stonefly (*Nemoura*) nymphs were apparently eliminated. These tundra ponds offer little opportunity for recolonization, and recovery of some taxa had not occurred several years later. Other species, mainly chironomids, were affected relatively little and some not at all (Mozley and Butler, 1978).

Cross (1986) surveyed benthic animals at a loading facility for the Bent Horn well in the high Canadian Arctic and compared that site with a 'control' site a few kilometres away. The results indicated that the industrial activity had little or no effect on the animals but the author cautioned that the number of samples collected was too few to offer high confidence in the comparison. This study included analyses of animals and sediment for hydrocarbons and the results indicated little

difference in concentrations between the samples from the loading facility and those from the control site. Statistical designs have been improved and have made it possible to detect increasingly subtle effects. Green and Montagna (1996) discussed the use of small-scale spatial variance in contaminant concentrations or benthic animal abundance as measures of impact. Green (2005) offered principles of sampling that can be used to monitor a spill or a point source.

5.3.9.4. Fish

Even in intensively studied cases of large spills in the ocean, like the *Braer* and *Exxon Valdez* spills, it has been difficult to establish the extent of effects on populations of pelagic fish. For example, many studies of larval herring have shown an array of sub-lethal effects including genetic effects both in collections from Prince William Sound, Alaska, and from controlled experiments with the same oil (e.g., Brown et al., 1996; McGurk and Brown, 1996; Norcross et al., 1996; Hose et al., 1996; Kocan et al., 1996; Marty et al., 1997a). Experimental exposures of larval herring to oil-water dispersions have resulted in responses similar to those observed in field collected herring. Adult herring from oiled locations after the spill in 1989 showed hepatic necrosis typical of viral hemorrhagic septicaemia virus; those from reference locations in 1989 and from all locations in 1990 and 1991 showed no liver necrosis (Marty et al., 1999). Pearson et al. (1999) reported dramatic changes in the harvest of herring from Prince William Sound in the years following the spill in 1989. No herring were harvested in 1989 but the catch was above average in 1990 and reached record numbers in both 1991 in 1992. Then in 1993 and 1994 the fishery collapsed to a small fraction of its former numbers. Analyses by Pearson et al. (1999) of potential causes of the collapse indicated that it was not caused by the spill in 1989. Thus, it remains unclear how to relate herring catches to the spill or to the many sub-lethal effects.

Bue et al. (1996) reported significantly increased mortality of pink salmon embryos from oiled streams relative to reference streams for at least two years following the *Exxon Valdez* oil spill. Many sub-lethal effects on larval salmon have been described from experimental exposures in gravel treated with Prudhoe Bay crude oil (Marty et al., 1997b,c). The field results have been challenged as possible artefacts (Cronin and Maki, 2004). Brannon et al. (1995) and Maki et al. (1995) reported several measurements of life stages of pink salmon from Alaska and found them to be similar between oiled and unoiled sites. Subsequently, Brannon et al. (2001) argued that the effects reported on pink salmon eggs in spawning streams were probably artefacts of the timing of fish collections. Regardless of the resolution of differing interpretations, the salmon case illustrates the extraordinary difficulty of establishing cause/effect relationships with wild fish.

However, difficulties often arise when trying to compare one spill with another. Growth of larval Pacific herring (*Clupea pallasi*) was affected in Prince William Sound (Marty et al., 1997a) but Gallego et al. (1995) examined the fine structure of otoliths of herring (*Clupea harengus*) larvae following the *Braer* oil spill and found no indication of growth inhibition. Wright et al. (1997) examined populations of sandeels (Ammodytidae, mostly *Ammodytes marinus*), a species important as forage for marine mammals, seabirds, and larger fish, around the Shetland Islands after the *Braer* oil spill in January 1993. The authors had available data on larval growth from 1990 to 1992 prior to the spill and from 1993 following the spill. It is evident that the young sandeels taken before and after the spill grew similarly (Figure 5.14). This species appears subject to wide variation in abundance and has been exploited commercially since 1974 but landings declined from 1982 until the fishery was closed in 1991. Surveys of abundance indicate great variation with higher numbers in 1993 than in 1992, followed by a return to lower numbers in 1994 and then higher numbers again in 1996 (Goodlad and Napier, 1997).

In general, so many variables acting independently or together can exert effects on populations of fish that it is almost impossible to make an unequivocal statement about effects of exposure to oil on populations of pelagic fish.

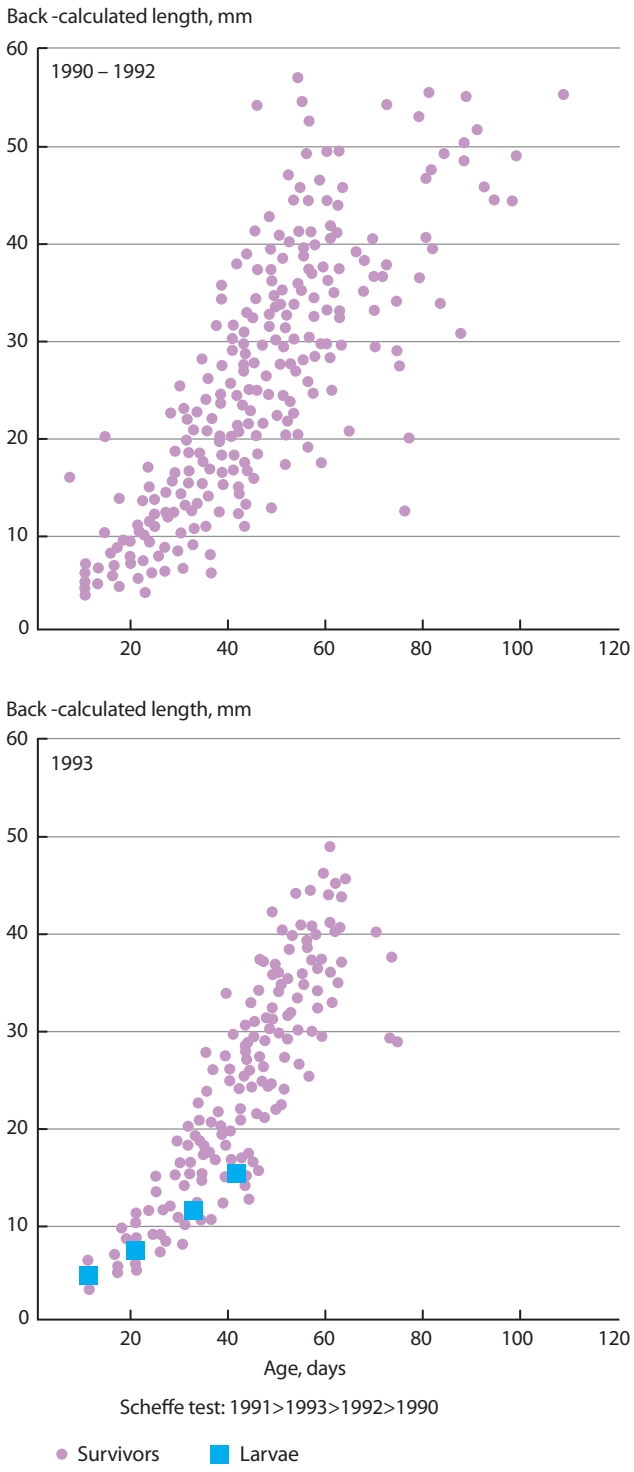

Figure 5.14. Length of sandeels as a function of age before (1990-1992) and after (1993) the *Braer* spill in January 1993 (Wright et al., 1997). The Scheffe test is a statistical method for determining whether the fish are different in length-at-age for different years.

5.3.9.5. Marine mammals

Marine mammals are among the most readily observed animals at sea and often suffer obvious acute physical oiling after spills. Geraci and St. Aubin (1990) reviewed the risks posed by oil to marine mammals. St. Aubin (1990a) catalogued 29 events in which encounters between oil and pinnipeds have been recorded. Oiled seals or sea lions (Otariidae) have been observed frequently but it has been extraordinarily difficult to establish a link between the presence of the oil and any effects on the animals. In some instances oiled seals have been found dead and in others they have apparently recovered from oiling. An early study of elephant seal (*Mirounga* spp.) pups by on-site observation and by returns of tagged animals, suggested little or no impact from oiling (Le Boeuf, 1971).

Most spills have been in temperate waters but a few have been in cold waters. A relatively small spill involved 4000 gallons of bunker C oil spilled into the Gulf of St. Lawrence in March/April 1969. It was reported that several thousand harp seal adults and pups were oiled and an unknown number killed (Warner, 1969). Dead seals were reported from sightings in the water, on the shores of the Strait of Belle Isle, and on the ice. Harp seals bear their pups on the ice in the Gulf of St. Lawrence at this time of year and the spill occurred at their most vulnerable time. In another case with some similarities, Davis and Anderson (1976) investigated the effects of a spill of unidentified oil on the coast of Wales in September 1974, when grey seals (*Halichoerus grypus*) were pupping. They found that the effects of human disturbance during the investigation were probably more harmful to the seals, mothers or pups, than the oil. They noted that two pups became so heavily coated with oil that they could not swim and drowned when washed out to sea. There was a small but significant difference in peak weights attained by pups with oiled pups having a lower peak weight than unoiled pups and this was hypothesized to reduce chances of pup survival after weaning.

Spies et al. (1996) cited studies showing that about 300 harbour seals were lost from a population of about 2200 in Prince William Sound. In the same area, local pods of killer whales were well known from several years of annual surveys before the spill and the disappearance of 15 killer whales from one pod was noted between autumn 1988 and summer 1990 (Dahlheim and Matkin, 1994). This represents a rate of loss unique among killer whales from the North Pacific. The coincidence with the spill suggests a relationship but there is no obvious mechanism for this; in 1994 the pod deteriorated and 5 individuals were observed with another pod – an unprecedented action for this species – suggesting an alteration in social behaviour.

Sea otters from Prince William Sound have probably been studied more than any other population of marine mammals and provide the best evidence to date for an effect of a spill on a population of marine mammals. The acute effect of oiling on sea otters is through loss of thermal insulation and hypothermia. Kooyman et al. (1977) examined the physical effect of oiling on the properties of otter fur and found that oiling reduced thermal insulating properties and could explain the deaths by hypothermia of oiled otters. Garrott et al. (1993) estimated that about 2700 sea otters were lost from a population of about 6500 in the oiled areas of Prince William Sound following the *Exxon Valdez* spill. However, long-term study of otters in Prince William Sound following the spill in 1989 has shown more subtle effects (Monson et al., 2000; Bodkin et al., 2002). Analysis of the age structure of otters washed up on beaches in western Prince William Sound indicated that otters suffered decreased survival for about a decade, particularly those otters alive at the time of the spill. However, even otters not born at the time of the spill in 1989 showed the same effects but less severely (Monson et al., 2000). The otter population at Knight Island, where oiling was heavy after the spill and acute mortality was high, has failed to recover in spite of the fact that otters in less severely oiled areas have recovered. Reproduction was not impaired and food resources were not limiting (Dean et al., 2002); but recruitment into the local population failed due to poor survival and/or immigration of young otters (Bodkin et al., 2002). Cytochrome P-450 protein remained elevated in the Knight Island otters relative to otters from other sites experiencing recovery. Population recovery in this species was not by redistribution of adult otters among locations but rather by reproduction and immigration of young otters. This failure resulted in a protracted recovery period not completed by 2000. Some other otter populations from Alaska in areas not affected by the *Exxon Valdez* spill are experiencing striking reductions, apparently as a result of increased predation by killer whales (Estes et al., 1998). Continuing studies following the *Exxon Valdez* spill of long-term persistence of oil, of subtle biological effects, and of processes of recovery have been described by Peterson et al. (2003).

With regard to bowhead whales, oil and gas-related activity in Alaska, has had no detectable adverse population-level effects in the Chukchi and Beaufort Seas. The Bering-Chukchi-Beaufort (BCB) Seas population of bowhead whales is well-studied and has increased over the time that oil and gas activity has occurred. Indeed these whales may have reached the lower limit of the stock's original size before intensive commercial whaling. The only apparent response by this species has been the use of more seaward migration routes in response to near-shore seismic activity.

In addition to any effects of oiling, polar bears are vulnerable to disturbance in dens. Females in the Beaufort Sea area make dens in snow either on land or on sea ice in November and emerge with their cubs in March/April. Dens on land are potentially vulnerable to disturbance by human activity such as aircraft, motorized equipment, and people on foot (Amstrup, 2000). Dens on ice are probably less vulnerable to most human disturbance; these dens move with the ice and may be as much as 1000 km from the starting point by the time the bears emerge. However, Amstrup (1993) noted that bears were relatively tolerant to these disturbances. The population of bears increased during the 1970s and 1980s indicating that they have been able to tolerate the levels and types of human disturbances during those years.

5.3.9.6. Seabirds

5.3.9.6.1. Oil spills

Seabirds are one of the most obvious victims of marine oil spills, and their vulnerability to oil has often been dramatically illustrated to the public as oiled birds wash ashore after spills. The direct oiling of seabirds during major oil spills has long been recognized for its impact on large numbers of seabirds and nesting areas. For detailed reviews of the effects of oil pollution on seabirds see Bourne (1968), Holmes and Cronshaw (1977), Clark (1984, 1987), Leighton et al. (1985), U.S. National Research Council (1985), Dunnet (1987), Hunt (1987), Anker-Nilssen (1987), Wiens (1995), U.S. National Research Council (2003), and Peterson et al. (2003). In addition, the impacts of smaller spills and the effects on seabird populations of long-term, chronic releases of waste oil have been the subject of considerable research over several decades of laboratory and field research. Monitoring

the long-term effects of spills has led to some important insights into the lingering effects of oil spills (Lance et al., 2001; Peterson et al., 2003) and has shown that long-term ecological effects may be more significant than direct oiling of seabirds (Peterson et al., 2003). Very few studies have been conducted in the Arctic or with Arctic species, although laboratory studies and studies from sub-Arctic and temperate waters can be used to assess the potential impact of oil releases at northern latitudes.

Despite extensive field studies during spill events and near intensive oil and gas activity, the direct and indirect effects on seabirds of oil released to the environment are difficult to quantify (Wiens, 1996). The total number of birds killed by oiling in a spill, one key estimate of the magnitude of effects, is estimated from the carcasses recovered, multiplied by a recovery factor. Experiments have reported a number of recovery rates varying with distance to the coast and to wind speed and direction (Clark, 1984; Stowe and Underwood, 1984). A generic ratio of 10:1 (unrecovered:recovered) is often used to estimate total mortality, although a review of 45 major marine spills up to 1991 suggested that a ratio of 4/5:1 may be more appropriate (Burger, 1993). For example, following a relatively small oil spill in the Skagarak (~600 t) in 1981, around 45 000 oiled birds were found dead, and it was estimated that 100 000 to 400 000 birds had died (Anker-Nilssen and Røstad, 1982). The number of seabirds killed during the *Exxon Valdez* spill (from 100 000 to 300 000) (Piatt et al., 1990) is generally considered to be among the highest numbers yet reported, when over 36 000 seabird carcasses were recovered (Paine et al., 1996; Piatt and Ford, 1996), even though larger spills have occurred (Burger, 1993). This estimate has steadily increased in the non-technical press to over 600 000 (Wiens, 1996). Most authors agree that a single recovery rate is probably not suitable for all spills, and the value tends to vary with the type and location of a spill, its proximity to nesting and feeding areas, and the species of seabird affected. Based on a study of carcasses of king eiders (*Somateria spectabilis*) recovered on beaches after a small spill in Alaska, Fowler and Flint (1997) indicated that carcass persistence and the probability of detection under northern conditions after a spill depend on the beach type, weather conditions, and losses due to scavenging by, for example, Arctic foxes (*Alopex lagopus*). The recoverable carcasses will decline over time (Van Pelt and Piatt, 1995) as the carcasses decompose, making it mandatory to begin surveys shortly after spills. Models estimating the total numbers of seabirds killed during natural or anthropogenic events have been developed (e.g., Van Pelt and Piatt, 1995), but uncertainty in the estimates of the total number of birds affected remains high. Without accurate estimates of the numbers of individual species killed, the full impacts of the spills in northern waters will not be known.

Research into oil spill exposure, population effects, and recovery of seabird populations is continuing to shed light on the lingering spill impacts and the life-history traits of seabirds that influence long-term effects of oil spills (for reviews of the response of seabird communities after the *Exxon Valdez* spill see Wiens et al., 1996, 2001; Day et al., 1997a,b; Lance et al., 2001; and Peterson et al., 2003). An assessment of seabirds and oil spills in the waters of the U.K. reported that over 70% of the ten groups of seabird carcasses recovered were alcids, with the overwhelming majority guillemots and razorbills (Stowe and Underwood, 1984). Similarly, more than 70% of the carcasses retrieved from Prince William Sound soon after the *Exxon Valdez* spill were murres (73.7%), possibly because they were present in large pre-breeding aggregations at the time, followed by sea ducks (5.3%) (Piatt et al. 1990). Later surveys recovered larger numbers of procelariids, gulls, and puffins.

Although seabird communities showed initial impacts from the spill followed by recovery (Wiens et al., 1996), the recovery of individual species through the years following the spill varied markedly (Day et al., 1997a). Species showing continued negative impacts after the spill tended to be resident species that fed in the intertidal zone (Day et al., 1997a). A further assessment of the recovery of seabird populations by Lance et al. (2001) reported that several species showed continuing negative effects nine years after the event, and that the effects may be increasing in some species of ducks. The authors attributed the trends to a combination of continuing spill effects and natural variability. These bird studies provide some of the data used by Peterson et al. (2003) to conclude that effects of the *Exxon Valdez* spill were continuing within the ecosystem a decade after the spill.

In general, seabirds breeding in the Arctic possess a number of characteristics which make some species more susceptible to oil pollution than others. Many species have long-range spring and autumn migrations, are colonial or semi-colonial, and have a slow reproductive capacity with delayed maturity, low fecundity and high adult survival. Some seabird species nest in densely populated colonies and concentrate to feed at ice edges, polynyas, open leads in the ice, or in other areas where prey species congregate (Levy, 1983). The birds are particularly susceptible to direct and indirect effects of oiling at these times. Most Arctic seabird species overwinter in small, confined nesting areas where exposure to oil could significantly affect a large section of the population. A study of common murres (*Uria aalge*) in the European Arctic has shown that major oil spills on the wintering grounds in the English Channel and Bay of Biscay had doubled the mortality of adults of breeding age (Votier et al., 2005), although the impact on the size of the nesting colonies appeared to be slight. These additional factors need to be considered when assessing potential impacts from oil and gas activities.

The extent to which seabird species are exposed to surface releases of oil depends on their method of feeding, time spent resting on the water surface, and diet. Birds that feed at sea throughout the year (e.g., alcids, diving ducks, many terns, and gulls) and for a part of the year (e.g., some ducks, grebes, loons, and phalaropes) can be considered sensitive to oil spills. Species which spend most of the time swimming or diving are most vulnerable to oil while those that spend most of the time airborne, snatching food from the surface, are less vulnerable. Large guillemots (*Uria* spp.) and ducks moult their flight feathers after the breeding season and are unable to fly for two to seven weeks. Large guillemots and most diving ducks spend this flightless period at sea, where they are safe from terrestrial predators. Most ducks gather in flocks during the moulting period, while the large guillemots (*Uria* spp.) undertake a more dispersed swimming migration. Birds that aggregate in small areas on the sea are more vulnerable than those species that disperse, because a single spill has the potential to affect a significant proportion of the population. High seabird concentrations are found in colonies, moulting and feeding areas, and in leads in the ice during winter and spring. If an oil spill kills all the birds in a colony, the recolonization and population recovery will depend on the size and location of neighbouring colonies (Cairns and Elliot, 1987). It will also depend on the extent of movements of seabirds between colonies (meta-populations), on which there is a lack of information (Wooller et

al., 1992). All these factors may contribute to some species being more sensitive to oil spills than others.

The seabird populations that are believed to be most seriously affected by acute oil spills in the long-term are those with a low reproductive capacity and corresponding high average lifespan. This is the strategy especially adopted by alcids and fulmars (Hudson, 1985; Furness and Monaghan, 1987; Croxall and Rothery, 1991). The size of such a seabird population is more sensitive to changes in adult survival than to changes in juvenile survival or breeding success. Long time-series observations have shown that these populations are not as stable as previously anticipated and do undergo natural long-term fluctuations. However, seabirds, like alcids and fulmars, with a long life span may also have delayed maturation. Often pre-breeding and non-breeding individuals ('floaters') in these populations form a pool that acts as a buffer from which individuals may be recruited to replace losses from breeding populations (Dunnet, 1982). The length of the delayed maturation may in part be determined by available breeding sites (Dunnet, 1982). Also, in years with low food availability at the colonies more adult birds increase their survival by skipping breeding and saving their reproductive effort for a more favourable year. These characteristics allow some species to recover from major losses to the population, such as those caused by oil spills, much more rapidly than others.

Seabird populations normally undergo significant fluctuations in numbers due to external natural factors and to density-dependent factors. A larger permanent non-breeding pool is usually seen as an adaptation to natural catastrophes. Prolonged periods of severe storms can make foraging difficult and result in naturally-occurring 'wrecks', in which large numbers of birds are killed. One wreck that occurred in the North Sea in February 1994 was estimated at 25 000 birds, mainly guillemots (Ritchie and O'Sullivan, 1994). A wreck of 100 000 guillemots was reported from the Gulf of Alaska in April 1970 (Bailey and Davenport, 1972; Hudson, 1985). Another wreck of 121 000 guillemots occurred in Alaska in 1993 (Van Pelt and Piatt, 1995). In light of these large natural mortality events, the extent to which the oil spill-related mortality will affect the population depends on whether the population can recover using normal population compensatory mechanisms and whether exposure to oil reduces the reproductive capacity of the population.

Seabirds are generally believed to be subject to density-dependent regulation although currently there is little clear evidence of the mechanism involved (Wooller et al., 1992; Weimerkirsch, 2002). Wiese et al. (2004) showed that the body mass of thick-billed murre (*Uria lomvia*) nestlings at the time that they depart from the colony is lower in larger colonies. Availability of nest sites in seabird colonies can act as a density-dependent factor regulating the breeding populations, especially in a proximate fashion and at a local level. Food availability is considered the factor most likely to limit overall numbers of seabirds (Croxall and Rothery, 1991) and this regulation is believed primarily to take place during breeding, where the feeding areas are confined to areas near the colonies (Alerstam and Høgstedt, 1982). However, many population regulating factors are operating and density-independent environmental effects and parasites can be more important (Croxall and Rothery, 1991; Weimerkirsch, 2002).

Arctic seabirds are particularly susceptible to human-related activities because of the short summer season when the colony nests and rears the young and the adults moult to renew their plumage. Timing is critical for these processes to occur during the short productive, open-water season, and disruption due to either natural or oil and gas activities can result in a major loss of young. A study of the energy budget of chicks of the red knot (*Calidris canutus*), an Arctic shorebird, showed that the growth rate of the knot chicks was much higher than those of similar species in a more temperate climate and they foraged more to make up for the high energy demands caused by cold temperatures after hatching (Schekkerman et al., 2003). The total metabolised energy during the pre-fledging period was 89% higher than the value predicted from allometric relationships, and was among the highest values reported for birds. To meet this high energy demand, the chicks probably have a very high foraging intake rate (Schekkerman et al., 2003). This study demonstrates the type of specialised ecological traits and life history strategies (Sandercock et al., 2005) of some Arctic species that are critical for the maintenance of the population.

Some species may also be susceptible to disturbance during the summer moulting period when human activity is high (Mosbech and Boertmann, 1999). Startle responses in flightless, moulting birds can cause significant changes in behaviour and a high demand on energy reserves. Oil exploration in Greenland that led to increased helicopter traffic caused disturbance reactions in geese at a distance of 10 km with large helicopters and 5 km with small helicopters (Mosbech and Glahder, 1991). In contrast, a study in Alaska found no effects on the number and diving behaviour of long-tailed ducks (*Clangula hyemalis*) moulting in an area of underwater seismic testing (Lacroix et al., 2003). Although there was no apparent difference in site fidelity or diving intensity with control areas, the authors stated that the results should be interpreted with caution because logistical and ecological factors limited their ability to measure subtle effects (Lacroix et al., 2003). Pink-footed geese (*Anser brachyrhynchus*) showed large changes in activity patterns, with grazing time significantly depressed with increased helicopter traffic. Oil and gas activities may mimic natural processes, increasing the hazard to seabirds. For example, although studies have shown that some seabirds avoid feeding in areas with obvious oil slicks (Lorentsen and Anker-Nilssen, 1993), species such as the storm petrel (*Oceanodroma furcata*), are attracted to oil slicks that may indicate whaling activity or large die-offs of fat-rich invertebrates (Boersma et al., 1988).

In addition to large accidental spills and smaller spills that cause oil slicks on the sea surface that constitute the main oiling hazard to seabirds, there is increasing evidence that chronic low-level concentrations of hydrocarbons in the sea can have a significant effect on survival and reproductive performance of seabirds (U.S. National Research Council, 2003; Peterson et al., 2003). Many small illegal discharges of oily waste from ships can cause mortalities of the same order of magnitude as the large spills (Holmes and Cronshaw, 1977; Wiese and Ryan, 2003). Wiese and Robertson (2004) estimated that offshore Newfoundland, Canada, about 300 000 thick-billed murres and little auks (*Alle alle*) are killed each winter by illegal discharges of oily waste. Although discounted by some authors (Leighton et al., 1985), beach surveys (Camphuysen and Huebeck, 2001) show that species such as the thick-billed murre, common eider, and common murre are particularly susceptible to oiling because of their habitat and feeding strategies (Holmes and Cronshaw, 1977). Although assessments specific to the Arctic have not been published, it seems likely that areas of intense shipping in the Arctic would show similar effects on local seabirds due to spillage (see Chapter 4), and the numbers will increase as shipping increases in the north. Data

are needed for Arctic shipping routes, particularly those in northern Europe and Russia to determine the significance of spilled oil on local seabird populations.

The extreme example of oil spills near major seabird colonies raises the question: can populations become extinct in oil spill catastrophes? Historical examples show that bird populations in general can recover from very small populations (Ryan and Siegfried, 1994). However, extinction of a seabird species has occurred due to hunting (e.g., the great auk, *Pinguinus impennis*; Lyngs, 1994) and seabird colonies have been deserted, with oil pollution as a major contributing factor.

Marginal populations such as puffins in Brittany, at the southern border of their distribution, have been affected. In this case a puffin colony crashed due to a combination of natural causes and oil pollution following the wreck of the *Amoco Cadiz* on the Brittany coast (Clark, 1984). This colony was later restocked with puffins from the Faroe Islands (Clark, 1984). In southern California, the guillemot colony on Devil's Slide Rock was extirpated in the 1980s, mainly due to a number of oil spills (Parker et al., 2007). Later this colony was recolonized using social attraction techniques (Parker et al., 2007). The disappearances of puffins and guillemots from the English Channel coast during the Second World War was probably related to oil spills as a result of the enormous pollution from sinking and burning ships (Gaston and Jones, 1998). These studies show that local oil-related effects can be significant on seabird populations, to the point of locally extirpating a population. The extent of disturbances required to cause this level of effects on Arctic colonies is largely unknown.

The potential impacts of oil spills on seabird populations can be evaluated by creating mathematical models using estimates of the mortality caused by an oil spill and empirical population parameters derived from detailed research programs (e.g., Ford et al., 1982). Modelling changes in population size is very complex, and the process involves high levels of uncertainty. Modelling approaches represent some of the best methods used to estimate the effects of oil spills on seabird colonies, but are limited by the fragmentary understanding of the quantitative dynamics of ecosystems. The influence of key intrinsic (genetic, physiological, etc.) and extrinsic (resources, competition, etc.) factors in regulating population size has always been difficult to discern. However, it has become very clear that closed population models that assume equilibrium values for population parameters are not appropriate for most natural populations. The concept of the meta-populations, or the idea that a species might be organised into localised groups of interacting populations, occupying one or several habitats, seems to better explain population dynamics.

Ford et al. (1982) developed simulation and analytical models to estimate the impact of oil spills occurring within feeding areas of colonial seabird populations. The analysis was hampered by the lack of field information on several critical model parameters. First, the work pointed out features of seabird biology which merit closer attention, and gave some general ideas of what may happen in an oil spill. In a given scenario, a spill (approx. 620 m^3) occurs during the middle of the breeding season 24 km from an island (St. George, Pribilof Islands) with very large colonies of guillemots and kittiwakes (*Rissa* sp.). This results in 68% mortality of adult guillemots and 10% mortality of adult kittiwakes. As a crude first-level estimate, the authors simulated that it will take around 40 years before the guillemot population returns to the pre-spill level. However, the model did not account for increased population growth due to decreased competition in the depleted population (density dependence), so the recovery rate could be higher. The most important factor regarding population impact is adult oil spill mortality, as could be expected for a long-lived species. A complete breeding failure in one year may have a lesser effect than a 5% one-time die-off of adults (Ford et al., 1982). Sensitivity analysis showed that the model is extremely sensitive to the foraging distribution of birds around colonies and to variations in the rate at which a population responds to the occurrence of a perturbation by adjusting its foraging distribution (Ford et al., 1982).

Samuels and Ladino (1983) developed a model to determine the effects of hypothetical oil spills on seabird populations in the mid-Atlantic region of the U.S.A.. Their model used density-dependent factors in contrast to the model of Ford et al. (1982). The authors assumed the number of young produced per breeding bird to be inversely related to the total adult population size. Using life-table data for common terns (*Sterna hirundo*) they found that if 25% of all age classes were killed by an oil spill, the tern population (colony) would require nearly 20 years to recover. However, the form of density dependence used by Samuels and Ladino is largely speculative. If a colony experiences a large mortality and no immigration the reproductive outcome per individual may decrease because of larger predation. Although further data are required for making realistic models of seabird population responses to oil spills, these population models can be useful tools in predicting the relative sensitivity of different areas, periods, and seabird species.

The focus of impact assessments and the associated research has been on understanding the possible effects on seabirds at the population level. The protection of populations is a readily understandable idea and it is the most important concept in the management of marine living resources. It is important that strategies designed to protect large, healthy populations also apply to small populations that may lose a significant fraction of the adult breeding population in a spill, or a year's reproductive output from a major disturbance. Small populations may take much longer to recover from major changes in population size and risk the possibility of extirpation if numbers fall below a threshold level.

In conclusion, major oil spills do have the potential to deplete bird populations and seabird colonies may be extirpated. However, experiences from spills indicate a considerable resiliency of seabird populations to single catastrophic events, although changes in the diet and foraging patterns in the areas of the spill may slow recovery considerably (Peterson et al., 2003). The experience from oil spills in sub-Arctic and temperate waters suggests it is unlikely that an oil spill can wipe out a seabird population unless other factors, such as environmental factors, hunting and by-catch in gillnets, hamper the recovery of the population, or the population is small and has a very restricted distribution.

This conclusion stresses the importance of a holistic approach to the management of seabird populations in relation to environmental impact assessments of oil activities in the Arctic. Where the oil activities introduce a new risk to the seabirds, it is inevitable that there will be large uncertainty on impact predictions of large oil spills. It is important that strategies to protect seabird populations take into account the current status of the populations which may be under stress from hunting or by-catch in gillnets. One example to illustrate this point could be Brünnich's guillemot (thick-billed murre) colonies in West Greenland, which have been declining for decades due to hunting, disturbance, and by-catch in gillnets (Kamp et al., 1994; Boertmann et al., 1996). Many large colonies have been abandoned, and have not been recolonized even though by-catch in gillnets and the

detrimental spring and summer hunting has almost ceased. The total population in West Greenland is still rather large, but the interaction between the colonies is not known (i.e., within the meta-populations), and recolonization or restocking of extirpated colonies has not occurred to any extent (Parker et al., 2007). Further research on seabird colonies in the Arctic will help to refine the information needed to predict oil spill effects with more precision.

Because the population dynamics are so complex, small impacts can be important at the population level. Experience from spill events in sub-Arctic waters generally shows the resilience of populations and the ability of the population to recover, however no major spills have occurred in the Arctic. Although populations have developed strategies to handle natural 'catastrophic events' caused by weather or food-shortage, the cumulative stress may affect a population via several pressures over time (e.g., oil spill in spring followed by disruption of nests in summer) or by several pressures over a short time frame (e.g., mortality both of adults and preferred food items during a spill and a loss of nesting habitat from the same spill). Despite uncertainties in the current understanding of Arctic seabird population dynamics, it is clear that an oil spill, regardless of size, could have major effects on local seabird populations in the Arctic if it occurs at the right place and time. It is prudent to protect smaller populations by establishing strict criteria against the release of oils from any facility or ship. This is critical given the expected increase in oil and gas activity over the next decades. Plans should be established to collect information on affected bird species immediately following a spill as part of emergency response measures.

5.3.9.6.2. Physical structures

The physical structures of oilfield operations may also provide hazards to wildlife, in particular migrating seabirds and waterfowl. In temperate regions, seabirds have been shown to gather around drilling platforms due to the presence of strong lighting at night, flaring, and increased numbers of food species in the waters around the structures (see Wiese et al., 2001 for a review). Wastes from the platforms, sediments stirred up during the drilling process, and the presence of the physical structure itself result in greater numbers of food species for the birds. Also, some seabird species use light cues at night for feeding and are attracted to bright lights and flaring on the rigs. Several studies have shown much denser numbers of seabirds near the rigs than in background areas. In one study in the Bering Sea, seabird numbers increased significantly around the ARCO Ocean Ranger after 'spudding in', while there were no changes in seabird numbers in background areas (>10 km from the structure) (Baird, 1990). The species present in the greatest numbers around the rigs included shearwaters (*Puffinus* spp.), northern fulmars (*Fulmarus glacialis*) and glaucous-winged gulls (*Larus glaucescens*).

Several studies suggest that the physical structures, and the activities associated with them, may have significant impacts on seabirds, both in the day and at night. A review of the causes of bird mortalities in the U.S.A (Erickson et al., 2005) reported extreme examples of bird strikes on towers and a natural gas pumping facility of up to 10 000 birds in a single event, with average numbers at some towers of 1000 to 2000 birds of several species per year. In one instance, approximately 3000 passerines (primarily warblers, redstarts and sparrows) died within 75 m of a sour gas flare stack in northern Alberta (Bjorge, 1987), however, examination of the carcasses indicated that the birds died from inhalation of stack emissions and not from striking the tower. An analysis of birds striking power lines has shown that species with high wing loading and low aspect ratio have a greater risk of colliding with power lines (Bevanger, 1998). These species include the puffins, sea ducks, grebes, and murres that fly at high speeds. Although estimates of birds killed in mainland America from impacts with power lines and communication towers may exceed 130 million birds per year, no estimates of the impacts of structures from oil and gas activities in the Arctic have been published.

An assessment in the western U.S.A has shown that open pits or sumps are also a significant source of wildlife mortality. Birds landing on the oily water become coated with oil and die from exposure or starvation (Trail, 2006). Previous surveys have shown that about 80% of the wildlife species caught in the pits comprises birds. One survey reported ducks, great blue herons (*Ardea herodias*), hawks, and crows (*Corvus brachyrhynchus*), in addition to some land mammals (Flickinger, 1981). While previous estimates placed the number of birds killed annually in these pits at about 2 million in the U.S.A, new industry practices and phasing out of the pits has reduced the number of birds killed by half in recent years (Trail, 2006).

Studies conducted on the North Slope of Alaska indicate that many species seem to be unaffected directly by current oil and gas activity. Truett et al. (1997) examined black brant (*Branta bernicula nigricans*) and lesser snow geese (*Chen c. caerulescens*) breeding in and around the oilfields for evidence of effects on reproduction from physical disturbance of habitat and the release of contaminants. The major limits on reproduction were weather and predation of eggs by foxes and gulls. Development had caused some loss of habitat but there was no evidence that it affected the size of the population or rate of reproduction. Similarly, pacific loons (*Gavia pacifica*) nesting in impoundments formed during oilfield development had lower reproductive success in impoundments than in natural ponds, but the difference was not statistically different (Kertell, 1996). Again predation by foxes was the major factor in the loss of eggs, and it was often associated with the drawdown of the impoundment which left the nests open to predation. The higher density of foxes in the oilfield relative to the surrounding area (Burgess, 2000) suggests that predation on nesting waterfowl in the developed area might be greater than in natural habitats. Kertell (1996) indicates that reproduction in the loons might be improved by stabilizing the water level of the impoundments used for nesting.

5.3.9.7. Concluding comments

Major oil spills virtually always result in efforts to describe their biological and economic consequences and sometimes their social consequences. Populations of oil-degrading bacteria often increase in number although this is not always the case. These organisms offer examples of responses to oil pollution at the level of populations. Populations of benthic animals can be analyzed with high sensitivity because of their relatively restricted movements. These studies are well advanced and have provided strong evidence of local but sometimes severe effects near drilling sites and following spills. After a population has been affected by oil, recovery can be expected after the oil has been at least partially degraded but a source of similar animals is required for recolonization. Thus recoveries in isolated lakes or headwaters of streams are slow or may not happen.

The experience with monitoring wells on the Norwegian shelf has shown that today's drilling practices cause only modest and confined environmental impacts on the sea floor. With the present restrictions on discharges from drilling with organic-based fluids, only water-based cuttings are likely to be discharged during drilling throughout the Arctic region. Although solid evidence for describing *in situ* effects of water-based cuttings discharges is somewhat limited on the Norwegian shelf since many of the so-called water-based fields have incidents of organic fluid discharges, the evidence available from other parts of the world suggests that the impact on the benthic fauna will be limited to smothering close to the discharging vessel and to possible changes in grain size distribution and mineralogy of the sediment in a zone around the cuttings deposit footprint. Exploratory drilling with water-based mud has so far shown no elevation in hydrocarbon concentrations in the sediment and no signs of disturbance to the benthic macrofaunal communities. Studies of populations of benthic animals near discharge sites are so sensitive that studies of sub-lethal effects on these organisms appear to offer little advantage as proxies for population effects. However, when population effects are detected, or perhaps in regional surveys, sub-lethal effects and pathologies are helpful in efforts to establish causality or to identify areas of chronic stress.

Populations of large animals are not readily manipulated experimentally and so most of what has been learned is based on studies following spills. Some spills are described in Appendix 5.1. Populations of pelagic fish have been the most difficult to assess. Not even the intense studies of fish in Prince William Sound, Alaska, have resolved unequivocally the question of impacts on fish in that region. The most convincing evidence of impacts on fish has come from studies of eggs or larval fish that were examined for sub-lethal effects and pathologies and from experimental exposures.

With the exception of visible oiling and mortality of oiled animals, evidence of effects on marine mammal populations has been difficult to obtain. Sometimes many individuals have been killed but the significance of that for populations has rarely been established. The number of sea otters was reduced by approximately a third in the area of Prince William Sound after the *Exxon Valdez* spill but populations there have recovered, except for one from the most heavily oiled area, although around a decade was required. The migratory behaviour of bowhead whales on the coast of Alaska has been altered temporarily by seismic activity. A more difficult case to assess is that of a pod of killer whales in Prince William Sound. The data make an intriguing suggestion of some effect on the social structure of these whales but no obvious mechanism is apparent.

The seasonal nature of vulnerabilities of young animals, notably fish and some marine mammals and seabirds, to oil may offer an opportunity to conduct Arctic oil and gas activities including marine transport at times and locations to minimize the chances of exposure of young animals to oil.

5.3.10. Effects of land structures on freshwater systems

The issues with structures like roads, runways, culverts and gravel pads, and other temporary or semi-permanent structures are principally of concern for terrestrial plants and wildlife. However, aquatic habitat can also be disrupted, especially during construction. For example, stream crossings may be made in such a way that erosion and/or fish passage become problems. Truck traffic on ice roads poses questions of noise and pressure changes. Noise is transmitted through ice and fish can generally sense sound in the range from 50 Hz to over 2000 Hz, with some species more sensitive than others. Fish need sound to be at least 10 dB above background levels to discriminate it from background noise. Fish avoid vessels when the noise level exceeds background by 30 dB or more (Mitson, 1995). Stewart (2003) searched published literature on effects of truck-generated noise in freshwater but found little information. However, the author thought it unlikely that the noise would be sufficient to damage fish physically or to elicit startle or alarm responses.

Sumps were used in the past to store process water and drilling fluid in the Arctic. Their use has now been largely replaced by down-hole injection of wastes or by shipping disposed materials to disposal facilities in the south, however sumps might be the preferred disposal option for cost considerations or where re-injection is not possible. Sumps rely on freezing and backfilling to incorporate wastes into the permafrost (Hrudey, 1979) and to seal wastes over the long term, however high salt levels in drilling muds that intrude into the permafrost reduces the integrity of the sump and results in the leakage of material. Several examples of leaking sumps, as indicated by dead vegetation in areas adjacent to the sump, or sumps with collapsed caps, have been reported in northern Canada (Kokelj and GeoNorth, 2002). Uncapped sumps can also become flooded, particularly in low lying areas, which can mobilise the previously contained materials. In one study, almost 50% of the sumps assessed in the Mackenzie Delta region of northern Canada were in the process of collapsing or had completely collapsed (Kokelj and GeoNorth, 2002).

An ecological study of the effects around sumps may be indicative of some of the issues associated with the construction and operation of sumps under Arctic conditions (Smith and James, 1985). A sump associated with Parsons D-20 drill site in northern Canada (drilled from April to December 1976) resulted in 500 t of solids for disposal, 46% of which was potassium chloride (KCl, by weight). Waste leakage started almost immediately, with vegetation killed over a distance of 210 m from the sump to a small lake, probably due to high salinity in the plume. The sump remained unfrozen for a winter due to the pressure of the fill and high salinity. A second sump was constructed and materials moved from the original sump, which resulted in some recovery of the plants around the original spill plume. A sump, built in 1977 at another well, had much lower levels of KCl (1.3% solids), but began to show serious degradation of the terrain as the ground became waterlogged and the supernatant melted ground ice. Plant damage was restricted to a fairly small area but the mud berm was eroded by spring meltwater and this resulted in the movement of mud from the sump. Revegetation of the mud-covered areas by some plant species occurred over the next few years (Smith and James, 1985).

Current estimates place the number of sumps in northern Canada at 300 to 400. Approximately 700 oil and gas production and exploration sites in Alaska have onsite drilling waste disposal pits. Use of the pits in Alaska has been phased out and the unused pits are being dewatered and restored (NAS, 2004). Pits containing oil-based muds are to be excavated, dewatered, and backfilled to restore the sites. Current estimates of the use and number of old pits in Russia were not available for this assessment. An ongoing project in Canada is presently assessing the integrity of the permafrost seal of these pits in light of increasing temperatures in an attempt to determine the long-term environmental consequences of the sumps. Because of the large number of pits, or sumps, present throughout the Arctic, the results

from the permafrost integrity studies should be used to assess the potential environmental consequences of eroding permafrost throughout the circumpolar Arctic.

5.3.11. Experimental oil spill studies

5.3.11.1. Marine experimental spills

There have been several experimental marine oil spills in the Arctic, the most thoroughly studied was the Baffin Island oil spill in northern Canada (Sergy and Blackall, 1987). For aquatic organisms, the most difficult sources of hydrocarbons to quantify are accidental discharges of oil from production or storage sites or from pipelines or ships (Macdonald and Bewers, 1996). Studies of the *Exxon Valdez* spill in 1989 have provided the most comprehensive assessment of a marine spill in cold water to date. Table 5.8 shows a comparison of the ideas in place historically and those developed from the *Exxon Valdez* oil spill experience in Alaska. Several earlier investigations showed long-term effects similar to those described in Alaska and contributed to the growth of the concepts tabulated by Peterson et al. (2003). The earlier knowledge of effects has been neither lost nor supplanted; rather a more comprehensive ecological overview has been added. The most obvious effects are acute mortality or injury to biota and uptake of hydrocarbons that cause tainting of fishery products. These effects are reasonably well understood. Less obvious are chronic and sub-lethal effects caused by long-term exposure to higher-boiling components (largely polycyclic aromatic hydrocarbons; PAHs) that remain in sediment for long periods; these effects are less well understood.

5.3.11.1.1. Balaena Bay, Northwest Territories, Canada

Adams (1975) reported small, experimental spills of 64 m^3 of Norman Wells and Swan Hills crude oils in winter into circular areas skirted under the ice. The intent of these experiments was to examine the effects of oil on the transmission of light through the water. Lower light levels were found below ice containing entrapped oil but primary productivity was slightly enhanced near the oil. In keeping with this, the abundance and taxonomic diversity of phytoplankton was slightly increased in samples contaminated with oil. Spring melt to ice-free status occurred about ten days earlier in the oil-contaminated area than in the rest of the bay.

5.3.11.1.2. Cape Parry, Northwest Territories, Canada

The retention of toxic properties by oil under ice was illustrated by an experiment in the aquatic ecosystem (Percy and Mullin, 1975). A sample of Norman Wells oil was recovered from an experimental spill under the ice in a small bay on Cape Parry, Northwest Territories, Canada. The oil had spent the period from November 1974 to May 1975 under the ice but its toxicity to the amphipod *Onisimus affinis* was essentially the same as that of fresh oil.

5.3.11.1.3. The Baffin Island Oil Spill (BIOS) study

The Baffin Island oil spill experiments, conducted in 1982 and 1983, were designed primarily to determine whether chemical dispersion would increase or decrease the environmental effects of spilled oil and to evaluate other shoreline protective and clean-up techniques (Sergy and Blackall, 1987). A small bay was treated by releasing artificially aged Lagomedio crude oil at high tide during an onshore wind so that the oil was deposited ashore as the tide receded. Another small bay was treated by releasing the same oil mixed with the dispersant Corexit 9527 (10 parts oil to 1 part dispersant). A variety of chemical and biological studies were conducted to define the persistence and movements of the oil and the effects of exposure to it on several taxa of local biota; the study sites contained few birds, fish, or marine mammals and these animals were not included in the observations.

Table 5.8. Changing paradigms in oil ecotoxicology, moving from acute toxicity based on single species toward an ecosystem-based synthesis of short-term direct plus longer-term chronic, delayed, and indirect impacts (Peterson et al., 2003).

Old paradigm	Emerging appreciation
Physical shoreline habitat	
Oil that grounds on shorelines other than marshes dominated by fine sediments will be rapidly dispersed and degraded microbially and photolytically.	Oil degrades at varying rates depending on the environment, with subsurface sediments physically protected from disturbance, oxygenation, and photolysis; retaining contamination by only partially weathered oil for years.
Oil toxicity to fish	
Oil effects occur solely through short-term (~4 day) exposure to water-soluble fraction (1- to 2-ringed aromatics dominate) through acute narcosis mortality at parts per million concentrations.	Long-term exposure of fish embryos to weathered oil (3- to 5-ringed PAHs) at ppb concentrations has population consequences through indirect effects on growth, deformities and behaviour with long-term consequences on mortality and reproduction.
Oil toxicity to seabirds and marine mammals	
Oil effects occur solely through short-term acute exposure of feathers and fur and resulting death from hypothermia, smothering, drowning, or ingestion of toxic substances during preening.	Oil effects are also substantial (independent of means of insulation) over the long term through interactions between natural environmental stressors and compromised health of exposed animals, through chronic toxic exposure from ingesting contaminated prey or during foraging around persistent sedimentary pools of oil, and through disruption of vital social functions (care giving or reproduction) in socially organized species.
Oil impacts on coastal communities	
Acute mortality through short-term toxic exposure to oil deposited on shore and the shallow seafloor through smothering accounts for the only important losses of shoreline plants and invertebrates.	Clean-up attempts can be more damaging than the oil itself, with impacts recurring as long as clean-up (including both chemical and physical methods) continues. Because of the pervasiveness of strong biological interactions in rocky, intertidal and kelp forest communities, cascades of delayed, indirect impacts (especially of trophic cascades and biogenic habitat loss) expand the scope of injury well beyond the initial direct losses and thereby also delay recoveries.

These releases of oil, with or without dispersant, had little or no effect on the numbers of bacteria in the water (Bunch, 1987). There were short-term effects on the benthic fauna. The benthic animals were not killed but divers reported that species that normally burrow into the sediment were found on the sediment surface and that some individuals displayed apparent narcosis or loss of response to mechanical stimulation (Cross and Thomson, 1987). Numbers of the bivalve *Serripes groenlandicus* observed on the sediment surface declined quickly, probably by reburial or possibly by predation. Although neither oil alone nor oil dispersed with Corexit 9527 caused large-scale mortality of benthic infauna, it may have been responsible for some subtle effects. Fifteen of 15 biomass-related measurements were unaffected and 43 of 46 density-related variables were unaffected. The density of the polychaete *Spio* sp. was probably affected by oil in the sediment but the other two density-related changes (total polychaetes and juvenile *Macoma* clams) were minor and transitory. The dry meat weight of clams *Macoma calcarea* tended to increase between August and September in the reference site, but not in the contaminated sites; a possible effect of oil in the sediment.

Histological and biochemical examinations of clams (*Mya truncata* and *Macoma calcarea*) were made immediately before the spills, immediately after the spills, and a year after the spills (Neff et al., 1987). The results provided no clear-cut evidence of effects on the pathology of these clams. However, there were some subtle indications that the oil was not entirely without effects. There were more frequent histological aberrations in clams from the bay treated with oil alone than in those from untreated areas or areas treated with dispersed oil. A few granulocytomas, non-neoplastic cellular conditions, were found in *Mya truncata* from the bays receiving dispersed oil a year after the spills. These structures have been associated with degraded water quality in earlier studies (Lowe and Moore, 1979). The clams were heavily parasitized before the spills but the results suggested somewhat more so afterwards in the bay with undispersed oil. This study reported a number of biochemical measurements on the clams and concluded that neither type of oil release exerted severe stress on these animals. The conclusion from this study was that dispersed oil probably caused greater acute effects on the clams but that undispersed oil probably caused greater stress over the long term.

Effects were similar for the epibenthos (Cross et al., 1987a). Starfish (*Leptasterias polaris*) and sea urchins (*Strongylocentrotus droebachiensis*) were common in the area and were not affected by the surface oil release. These animals, however, made dramatic short-term responses to the dispersed oil; for two days after the oil release they were observed lying on the substrate in unnatural positions (e.g., upside down). Some did not respond or responded slowly when prodded. Behaviour returned to normal within two weeks. Neither type of oil release affected densities of starfish. Failure to induce sustained effects was thought to have been due to the short time that high concentrations occurred in the water and to the relatively low level of oil in the sediment. A similar lack of impact applied to macroalgae (Cross et al., 1987b).

More controlled exposures were conducted at the test site using experimental holding facilities built for that purpose. Mageau et al. (1987) described several behavioural responses of two bivalves (*Mya truncate* and *Serripes groenlandicus*) and the sea urchin *Strongylocentrotus droebachiensis* in the experimental aquarium exposures operated to simulate the actual exposures in the treated bay and an adjacent bay. The results from the treated bay suggested an exposure sequence of 6 hr at 10 mg/kg (as oil), 6 hr at 100 mg/kg and 6 hr at 5 mg/kg for a total exposure time of 18 hr. Several behavioural changes were observed in all three species over periods from <1 hr to 15 hr. Most changes were found to be reversible over periods of up to two weeks after exposure had ceased. Tissue hydrocarbon residues persisted in the animals for periods longer than the behavioural changes suggesting that the changes in behaviour were more in response to hydrocarbons in the water than to hydrocarbons assimilated by the animals.

The BIOS project included an examination of the effects of oil on under-ice organisms, namely the meiofauna (Cross and Martin, 1987) and the algae (Cross, 1987), using buoyant enclosures to confine the oil under the ice surface. Dispersed oil caused dramatic decreases in the densities of copepods and polychaetes in the enclosures. The authors were unsure whether these animals were killed or displaced from their habitat under the ice. Effects on the ice algae were not observed, in contrast to the expectation based on many other studies cited. This lack of effects on the ice algae was thought to have been due to the short period of exposure to high concentrations of oil and the two-day period between application of the oil and sampling for algae.

The overall conclusions arising from the BIOS experiments were that there is no compelling reason to prohibit the use of chemical dispersants on spills in the Arctic. Furthermore, that there is no ecological reason for clean-up of oil should it encounter a setting like that used in the experiments. In the event that spilled oil should approach shoreline of high biological value or human use, dispersants would be warranted.

5.3.11.1.4. Svalbard

Ikävalko et al. (2004) studied the effects of crude oil bioremediation on Arctic sea-ice biota. The field experiment was carried out in Van Mijenfjorden, Svalbard, from 22 February to 25 April 2004. During the 63-day experiment, Statfjord crude oil with and without Inipol EAP22 (oil bioremediation product) and fish meal was spread onto a snow-free surface of sea ice to determine effects on sea-ice biota. In February, the sea-ice biota consisted solely of several species of diatoms at all depths of ice. At the end of the experiment the natural sea-ice community consisted of both diatoms and euglenid flagellates (abundance ≤3400 cells per 100 mL of melted sea ice). The treatment with oil only, oil+Inipol, and oil+fish meal lead to a general decrease in protist diversity and abundance. The oil-only treatment caused the most dramatic decrease in the abundance of protists throughout the ice cover, while the addition of oil+Inipol, and oil+fish meal lead to the disappearance of all protist groups except diatoms. The negative effects of Inipol and fish meal were most severe at the ice surface, while the interior and bottom parts of the ice cover were less impacted by the treatments. Heterotrophic flagellates seemed to be able to migrate downward from the ice surface when oil only was added onto the ice. The use of oil only, and oil+Inipol induced the formation of diatom resting spores. In control samples, the addition of fish meal only caused a notable increase in heterotrophic flagellates (≤25 000 cells per 100 mL of melted sea ice).

5.3.11.1.5. Enclosure studies

Two plastic bags 1 m in diameter and 13.5 m deep were placed in Lindaspollene, a land-locked fjord in western Norway in March 1980 (Skjoldal et al., 1982). One served as a control and the other was treated with 0.5 L of unweathered Ekofisk crude oil. Polar photo-oxidation products from the oil appeared quickly and reached concentrations as high as 5 mg/L. This slowed the development of the normal spring bloom of phytoplankton. The oil addition stimulated the bacterial population in the treated bag and resulted in increased mineralization of hydrocarbons and increased consumption of inorganic nutrients.

5.3.11.2. Freshwater experimental spills

There have been many small experimental spills of oil in Arctic freshwaters of Canada and Alaska, often with the intent of describing the behaviour of the oil itself (e.g., Kershaw, 1990). Snow and Scott (1975) spilled two types of crude oil (Norman Wells and Pembina) on two small Arctic lakes. Both oils rapidly lost lower boiling components by evaporation; 45% of Pembina oil and 49% of Norman Wells oil were lost in the first 48 hr. Heavy mortality of organisms, mostly aquatic insects, was noted in samples of the slicks on the first two days after the spill, which was attributed to chemical toxicity of the oils. Lower numbers of organisms continued to be taken in samples of the slicks and this was thought to be due to entrapment of organisms by physical oiling. The organisms most affected were those that make use of the water surface film: gerrids, corixids and beetles of the genus *Haliplus*. There was a conspicuous peak of seston, particulate carbon and particulate nitrogen, some three weeks after the oil was added, apparently the result of microbial and phytoplankton blooms. No such blooms were observed in control sites. The authors noted a 'massive increase of blue green algae' on samplers suspended in oiled areas. Surprisingly, the addition of the oils resulted in a eutrophying effect implying perhaps that phytoplankton growth had been limited by carbon.

Miller et al. (1986) introduced Prudhoe Bay crude oil into a small stream in Alaska (68°39′ N, 149°21′ W) just as ice was forming in autumn 1981. One of the experiments was conducted by coating baskets of rocks with known quantities of oil in order to study the effect of oil on colonization of the rocks by invertebrate animals. In one stream they found that there was virtually no colonization over the period September to June, and they recovered 90.7% of the oil initially used in the baskets. The stream apparently dried up for some, perhaps most, of the experimental period but the amount of oil recovered was surprisingly near 100% over nine months. That is, the sum of all processes together removed less than 10% of the hydrocarbons over the winter.

Shindler et al. (1975) treated two frozen ponds near Ottawa, Canada, one by injecting Norman Wells crude oil under the ice and one by spilling the oil on top of the ice. These treatments resulted in increased numbers of bacteria especially during the following summer. The oil apparently caused selection for bacteria well adapted to rich sources of organic nutrients. Scott and Shindler (1978) treated small, freshwater ponds in winter with Norman Wells or Pembina crude oil and sampled populations of phytoplankton and zooplankton a year after treatment. Algal populations were affected relatively little but zooplankton abundance was reduced in oiled ponds.

5.3.12. Potential for cumulative toxicological effects

Toxicologically there is a possibility of cumulative effects whenever organisms are exposed to two or more toxicants simultaneously. Oil, produced water, and drill cuttings are all complex mixtures that include many different hydrocar-

Table 5.9.Guidelines for maximum concentrations (μg/L) of selected hydrocarbons and PAHs in freshwater in relation to aquatic life and particularly fish (Buchman, 1999; CCME, 2005; PINRO, 2006). The EU values are working proposals being developed in relation to the Water Framework Directive.

	USA	Canada	Russia	EU	
	Aquatic life	Aquatic life	Water in commercial fishing reservoirs	AA–QS[a] Inland waters	MAC–QS[b] Inland waters and Transitional
Acenaphthene	1700	5.8			
Acridine		4.4			
Anthracene		0.012		0.1	0.36
Benzene	5300	370		1.7	49
Benz[*a*]anthracene		0.018			
Benzo[*k*]fluoranthene				0.03[c]	[d]
Benzo[*b*]fluoranthene				0.03[c]	[d]
Benzo[*ghi*]perylene				0.0016[c]	[d]
Benzo[*a*]pyrene		0.015		0.05	[d]
Ethylbenzene	32000	90			
Fluoranthene	3980	0.04		0.09	0.9
Fluorene		3			
Indeno(1,2,3-*cd*)pyrene				0.0016[c]	[d]
Naphthalene	2300	1.1	4	2.4	80
Phenanthrene	30	0.4			
Pyrene		0.025			
Methanol			100		
Quinoline		3.4			
Toluene	17500	2			

[a] AA–QS: proposed quality standard referring to the annual average concentration; [b] MAC–QS: proposed quality standard referring to short-term concentration peak maximum admissible concentration; [c] Sum of 2; [d] sum of all: 0.1 μg/L

bons, including phenolic compounds, several metals and radionuclides; thus, there are obvious possibilities for toxicological interactions. This type of interaction may enhance or ameliorate the toxicity of the individual toxicants presented alone. For example, planar chlorinated hydrocarbons like PCBs induce the mixed function oxygenase system induced by PAHs. These chlorinated compounds are ubiquitous in Arctic organisms (AMAP, 2004b) and some toxicological interaction might be expected. Areas of natural seepage of oil may offer opportunities to test for this type of interaction. Another toxicological interaction is that of exposure to both PAHs and sunlight. Exposures of test animals to sunlight or UV light after they have been exposed to a number of PAHs have resulted in lethal responses when neither the light nor the PAHs would be lethal individually (e.g., Duesterloh et al., 2002). This type of interaction has the potential for more frequent occurrence in the Arctic with depletion of stratospheric ozone and with the likely expansion of the oil and gas industry. The organisms at greatest risk for this interaction are translucent ones like larval fish in shallow-water habitat where light can penetrate.

Cumulative effects are often considered in a broader context of interactions among different types of activities. For example, toxicological effects might occur in habitat affected by some independent action such as dredging and the combined effects might differ from those of either alone. This type of interaction is relatively easy to predict for fixed installations like roads, bridges, pipelines, and terrestrial seismic lines, and would be evaluated as part of the environmental impact assessment process. The presence of significant numbers of people working on oil projects can in itself result in pressure on local resource species of fish or wildlife if harvesting is permitted. Again, this type of interaction between the industry and local resources is easily prevented by limiting local resource harvesting by industry personnel. The most difficult interactions to predict are those that involve accidental marine spills that occur at unanticipated locations and times. Models and scenarios can be helpful to estimate potential impacts of these interactions but experience from past spills and experiments offers the best guidance.

5.3.13. Guidelines and regulatory standards for water and sediments

Most Arctic countries have adopted guidelines describing concentrations of hydrocarbons and other substances they regard as acceptable in water and often in sediment as well (see Chapter 2). The intent of these guidelines is to protect environmental components from inputs that would exceed the guideline concentrations. These may be individual country standards (Canada, U.S.A, Russia) or standards agreed jointly among several countries (European Union). A number of these standards are listed in a series of tables for freshwater (Table 5.9), seawater (Table 5.10) and sediments (Tables 5.5 and 5.11). Maximum acceptable concentrations are usually established for a particular use. For example, standards for drinking water (Table 5.12) may differ from standards for freshwater for the protection of aquatic life (Table 5.9). Standards are helpful in the interpretation of environmental measurements since it is possible to tell at a glance whether a concentration would be considered excessive for a given use at a given site. There are numerous differences among the various sets of standards and possibilities for harmonizing these standards across AMAP

Table 5.10. Guidelines for maximum concentrations (μg/L) of selected hydrocarbons and PAHs in seawater in relation to aquatic life (Buchman, 1999; CCME, 2005; OSPAR, 1997). The EU values are working proposals being developed in relation to the Water Framework Directive. When relevant, the Faroe Islands consults the OSPAR Ecotoxicological Assessment Criteria for guidance.

	USA	Canada	EU		OSPAR
	Aquatic life	Aquatic life	AA–QS[a] Transitional, Coastal and Territorial waters	MAC–QS[b] Inland and Transitional waters	EAC[c]
Acenaphthene	970				
Acenaphthylene	300				
Anthracene	300		0.01	0.36	0.001–0.01 (p)
Benzene	5100	110	1.7	49	
Benz[*a*]anthracene	300				
Benzo[*k*]fluoranthene			0.03[d]	[e]	
Benzo[*b*]fluoranthene	300		0.03[d]	[e]	
Benzo[*ghi*]perylene	300		0.0016[d]	[e]	
Benzo[*a*]pyrene	300		0.005	[e]	0.01–0.1 (p)
Chrysene	300				
Dibenz[*a,h*]anthracene	300				
Ethylbenzene	430	25			
Fluoranthene	40		0.09	0.9	0.01–0.1 (p)
Fluorene	300				
Indeno[1,2,3-*cd*]pyrene	300		0.0016[d]	[e]	
2-Methylnaphthalene	300				
Naphthalene	2350	1.4	1.2	80	5–50 (f)
Phenanthrene	7.7				0.05–0.5 (p)
Pyrene	300				0.05–0.5 (p)
Toluene	6300	215			

[a] AA–QS: proposed quality standard referring to the annual average concentration; [b] MAC–QS: proposed quality standard referring to short-term concentration peak maximum admissible concentration; [c] EAC: Ecotoxicological Assessment Criteria. These criteria have been agreed by OSPAR for the preliminary assessment of Joint Assessment and Monitoring Programme (JAMP) chemical monitoring data with the aim of identifying potential areas of concern; they have no legal significance. Criteria marked with (f) are firm, those marked with (p) are provisional; [d] Sum of 2; [e] sum of all = 0.1 μg/L.

Table 5.11. Guidelines for maximum concentrations (µg/kg) of selected hydrocarbons and PAHs in freshwater sediments in relation to aquatic life (Buchman, 1999; CCME, 2005).

	USA	Canada	
	Probable effect level	Quality guideline	Probable effect level
Acenaphthene		6.71	88.9
Acenaphthylene		5.87	128
Anthracene		46.9	245
Benz[*a*]anthracene	385	31.7	385
Benzo[*a*]pyrene	782	31.9	782
Chrysene	862	57.1	862
Dibenz[*a,h*]anthracene		6.22	135
Fluoranthene	2355	111	2355
Fluorene		21.2	144
2-Methylnaphthalene		20.2	201
Naphthalene		34.6	391
Phenanthrene	515	41.9	515
Pyrene	875	53	875

Table 5.12. Maximum allowable concentrations (µg/L) in drinking water for some hydrocarbons, PAHs, sulphide, some metals and naturally occurring radionuclides (Health Canada, 2007; USEPA, 2007; PINRO, 2006; EU, 1998; Norwegian Ministry of Health and Care Services, 2001). EU values are used as a guide by the Faroe Islands.

	Canada (1.5 L water/day)	USA (2L water/day)	Russia	EU	Norway
Benzene	5	5		1	1
Benzo[a]pyrene	0.01	0.2	0.005	0.01	0.01
Ethylbenzene	<2.4 [a]	700			
Methanol			3000		
Naphthalene			10		
PAHs [b]				0.1	0.1
Toluene	<24 [a]	1000			
Xylenes (total)	<300 [a]	10000			
Barium	1000	2000			
Cadmium	5	5		5	5
Chromium	50	100		50	50
Lead	10	15		10	10
Mercury	1	2		1	1
Uranium	20	30			
Sulphide (as H_2S)	<50 [b]				
Sulphate	<500 000 [b]				
^{210}Lead	0.1 [c]				
$^{224/226/228}$Radium	2/0.6/0.5 [c]	5 [d]			
$^{228/230/232/234}$Thorium	2/0.4/0.1/20 [c]				
$^{234/236/238}$Uranium	4/4/4 [d]				

[a] aesthetic objectives; [b] sum of benzo[*b*]fluoranthene, benzo[*k*]fluoranthene, benz[*ghi*]perylene, indeno[1,2,3-*cd*]pyrene; [c] maximum acceptable concentration in Bq/L; [d] ^{226}Radium and ^{228}Radium combined, in pCi/L.

countries need to be explored. The main problems with standards lie in their implementation, enforcement, and updating as new knowledge becomes available.

The OSPAR Commission has placed restrictions on the discharge of produced water at sea with the intent of reducing the toxicity of these discharges. The area subject to the OSPAR Convention includes the Arctic part of the North-East Atlantic extending from 42° W to 51° E. The Contracting Parties to the Convention include Denmark (also for the Faroe Islands and Greenland), Iceland, and Norway. The eastern part of this area is off the coast of Russia, but Russia is not a Contracting Party to the Convention. Contracting Parties agree jointly on Recommendations which are then implemented via national laws and/or regulations in each country. These set minimum requirements only; individual countries may enact more stringent standards. There are no sanctions against countries that do not apply the recommendations. Furthermore, OSPAR Recommendations apply to offshore installations only. A summary of relevant OSPAR Recommendations is included in Chapter 2, Appendix 2.2. OSPAR Recommendation 2006/4 sets a requirement for a maximum concentration of dispersed oil in produced water of 30 mg/L, applicable from 1 January 2007, without specific limits for the amount of produced water actually discharged. Section 3.2 requests that the overall amount of produced water discharged in 2006 be reduced by at least 15% relative to the amount produced in 2000 for all installations under a Contracting Party's jurisdiction. In addition, for installations constructed after 1 January 2002 or installations substantially modified after that date, Contracting Parties are requested to minimize such discharges with the ultimate aim of zero discharges. Other OSPAR recommendations specify the sampling and analytical procedures that should be used to implement the produced water recommendation.

Several countries have initiated environmental effects monitoring programs giving increased attention to identifying and measuring biological responses to industry activity

(e.g., Gordon et al., 2000; Armsworthy et al., 2005a). These programs have begun to use sensitive measures of biological change induced in individual animals of diverse taxa and to document changes in individuals and in community assemblages. Environmental effects monitoring programs are able to assess the effects of 'normal' operating procedures at a site. As a result, the risk that those normal activities will result in unacceptable or unknown environmental damage has become less. The greatest risks remaining are unplanned, accidental releases of large volumes of oil at unpredictable locations. These risks can be addressed only by comprehensive emergency preparedness plans, by securing comprehensive baseline data, and by concerted efforts to minimize human errors.

5.3.14. Concluding comments and recommendations on the impacts of oil and gas activity on aquatic ecosystems

The major issue in assessing the impacts of oils on aquatic biota is the occurrence of spills and blowouts that result in severe biological effects. Experimental studies and case histories of spills leave no doubt that oil is toxic to aquatic organisms. Hydrocarbons are taken up from the water by many aquatic animals, principally via the gills, and by contact, by consumption of contaminated food or sediment, and even by consumption of oil droplets. Air-breathing aquatic mammals and birds may also take hydrocarbons in by inhalation if they are caught in a spill area. The ability to metabolize and excrete hydrocarbons is most highly developed in vertebrates and, consequently, concentrations of hydrocarbons in these organisms do not usually build up to levels typical of other compounds with similar partitioning properties like PCBs. The pathway of metabolism by fish and mammals is oxidative with the production of various oxygen-containing by-products that are more polar than the original hydrocarbons and that are more readily conjugated and excreted, often via the bile. In the case of some PAHs, metabolism can convert the original hydrocarbon to metabolites that are more active in the initiation of biological injury than the original hydrocarbons.

Literature on the experimental toxicology of oils to fish and other aquatic organisms is extensive and shows clearly that oil is acutely toxic when exposures ranging from about 100 ppb to the low ppm range are maintained for periods of a few days. NOECs are typically in the low ppb range, usually determined in exposures ranging from a few days to a few weeks. The acutely toxic components of oils are principally the low-boiling aromatic compounds, typically the one- to three-ring aromatic compounds that are more soluble in water than most other components of oil. Aquatic animals from north temperate zones or from the Arctic, in cases for which data are available, appear to have about the same sensitivity to the toxic effects of oils as similar animals from temperate regions. Hence the literature on the toxicology of oils to temperate aquatic animals is instructive for Arctic animals although it is not a substitute for further work with Arctic species.

The current literature is limited in terms of studies that assess quantitatively the effects of temperature on the uptake, elimination, and toxicity of hydrocarbons. The few studies available suggest that the overall effect of temperature is relatively small in terms of acute toxicity but significant in terms of retention of hydrocarbons within body tissues. With the exceptions of several studies driven by the development of oil reserves in Prudhoe Bay and the Norwegian Sea, there has been surprisingly little experimental work on the toxicology of oils to high Arctic species. Impact studies of sub-Arctic spills like the *Exxon Valdez* and the *Braer* are instructive for the Arctic, but studies within the Arctic are needed. More attention must be given to experimental toxicological studies designed to answer questions arising from three areas: chronic contamination of sediments with PAHs; long-term contamination of seawater with low levels of hydrocarbons; and interactions between exposures to PAHs and sunlight for shallow-water organisms.

There are taxonomic and developmental differences in sensitivity to oils; typically larval organisms are more sensitive than other life stages and fish are more sensitive than molluscs, for example. Most experimental exposures have been conducted over periods of a few days but longer-term exposures have revealed effects not evident over short periods. Exposures for longer periods are relevant for the Arctic because oils can persist for longer there, especially under conditions of cold, dark, and ice cover.

The most consistent mechanism to explain the acute toxicity of oils is through disruption of the functions of biological membranes. This mode of action has been indicated for over half a century and is suggested by visible histological effects on fish gills, by effects on membrane-associated enzymes like ATPase, by narcosis, and by movements of water across membranes. There is a dramatic effect of physical oiling, especially on fur-bearing animals like sea otters and seabirds. Oiling affects the thermal properties of the pelage and the animals lose body heat and suffer hypothermia, sometimes lethally. This type of effect appears to be unimportant for those marine mammals that rely on blubber for insulation (e.g., whales and seals). Physical oiling may affect the respiratory apparatus of aquatic invertebrate animals and cause suffocation. Heavy oiling also affects the swimming ability of seal pups.

There are several indications that body fats play an important role in the accumulation and depuration of both hydrocarbons and the oily taints and odours that arise from them. This requires further investigation with Arctic species that may accumulate most of their calories over a short feeding season and store those calories as fat for a long, non-feeding season. The implications of exposure to oil can be expected to vary with stages in the fat cycle and this requires further investigation in Arctic animals. A number of observations, notably of behaviour of several species, necrotic brain cells and depleted neurotransmitters, suggest an involvement of oil with the central nervous system. Very little effort has been given to investigating the effects of oil on neurochemistry and that seems an important issue. Hydrocarbons have often been reported to affect egg hatching; either its timing or its completeness. Mechanisms responsible for effects on hatching or timing of hatching have not been established and merit further research.

One immediate effect of biological uptake into fish and other animals consumed by humans is the tainting of fishery products with oily tastes and odours. This effect can be produced by short-term exposures and can persist for periods up to several months and even longer in animals that have limited ability to metabolize and excrete hydrocarbons. Different species become tainted to different intensities; the levels of fats in edible organs offer the most consistent hypothesis to explain these differences. Fatty species like salmonids are particularly susceptible to tainting. Similarly, different oils taint to different intensities and produce off-flavour with different tendencies to persist. For example,

in one study the taint from diesel fuel persisted longer that that from Forties crude oil. These effects on the quality of fishery products are the major causes of immediate post-spill economic losses to the fishing industry. The effects on subsistence fisheries are particularly relevant to the Arctic; tainting or contamination, or the fear of them, caused huge disruption in harvest and use of traditional marine products following the *Exxon Valdez* spill. The actual components of oils responsible for tainting are not known definitively although the aromatic hydrocarbons with one to three rings are the most likely candidates.

Both experimental exposures and contaminated sites have shown that PAHs are capable of inducing effects not revealed by acute toxicity studies. PAHs in sediment have been associated with enzymatic effects, tumours, and genetic and hormonal effects. At least some of these appear to be caused by metabolites formed from PAHs, rather than by the PAHs themselves. A growing body of literature has demonstrated an additional type of toxicity of several PAHs, namely enhanced toxicity under conditions of exposure to UV light or sunlight. This enhancement of toxicity can be large, up to several hundred-fold. UV light has become more intense at high latitudes with the growth of the Arctic ozone 'hole' and so this interactive effect seems potentially important in the Arctic during spring and summer, especially for organisms that are translucent (e.g., larval fish) and that inhabit shallow zones where light can penetrate. Hydrocarbons are already present in the Arctic and the growth of the oil and gas industry seems likely to add to existing levels. The severity of this effect depends both on the body concentration of PAH and on the intensity of incident UV light. These interactions need to be described more fully under suitable experimental protocols designed with Arctic animals in mind.

Studies of pelagic fish populations are usually performed after spills have occurred. However, for several reasons, these studies are seldom definitive in establishing whether impacts have occurred or in establishing cause/effect relationships when impacts are detected. While the relevance of these efforts is high, and their costs are very high, their sensitivity has been low. Sub-lethal measurements often have the opposite problem, high sensitivity but unknown relevance. There is hope that ways will be developed to link the two approaches but that is unlikely to be simple or timely. Lacking such linkages, the best chance of detecting the effects of petroleum on Arctic fish probably lies with sub-lethal measurements such as enzyme induction, gill histology and deformities of eyes and other organs in larval fish and possibly chromosome morphology and others. Several agencies are examining the many sub-lethal effects that have been described in efforts to select those that seem most instructive (e.g., ICES, 2007). Population descriptors of less mobile organisms, notably benthic invertebrates, are well advanced and sub-lethal effects on these taxa appear to offer little advantage over population studies. Population responses by benthic animals range from localized extinction to little or no detected effect. While many efforts have been made to predict population effects from various sub-lethal responses, these have not generally been definitive. The best opportunities to define such linkages probably arise from spills (accidental and experimental) in relatively contained settings and such spills should be seen as opportunities to advance knowledge of potential linkages. Areas of known natural seepages appear to offer opportunities to learn how organisms adapt to long-term exposures but studies of these areas have been restricted to chemical studies of hydrocarbons.

Sub-lethal effects of many types have been observed in laboratory experiments and in samples collected following spills and from discharges of produced water. Sub-lethal effects have often been described at exposures lower than those causing mortality. Among the most consistent of these has been the induction of liver enzymes in fish and histological damage to gills. The definition of no-effect levels is sometimes taken as the next lower exposure than the one that produces an observable biological effect or as an arbitrary fraction of an exposure known to cause an effect. Estimates available suggest no-observable-effect levels in the low ppb concentration range. Sub-lethal effects are less well defined than acute toxicity effects and will undoubtedly be the subject of further research. A type of sub-lethal effect not previously recognized has emerged from long-term study of the *Exxon Valdez* spill in Alaska, namely disruption of social organization of animals that have a strongly developed social structure. This type of effect was hypothesized in killer whales for which pod structures were altered in ways not seen previously in spite of many years of detailed behavioural observations.

The Arctic Council (2002) has offered general guidelines for the operation of offshore oil and gas facilities but more specific guidance would sometimes be useful. For example, it would be useful to agree on some standardized protocols for measurements of selected sub-lethal responses. Given a series of standard measurements, international surveys of baseline conditions in key Arctic animals could then be done in order to put in place data from the early 2000s. This type of activity is already well advanced in Norway and can profitably be extended throughout the Arctic. This would help interpret future measurements in instances of spills in the Arctic. These protocols need to be incorporated by Arctic countries into environmental effects monitoring programs. Similarly, international protocols for the quantitative description of petroleum tainting of fishery products need to be applied to a few species across their Arctic ranges. Measurements undertaken in accordance with these protocols could serve as a rational basis for settling future liability and damage claims.

An emerging area of research is that of genetic toxicology. Hydrocarbons can interfere with DNA replication in several ways and cause replication errors. It is difficult to know how important these changes are for populations of Arctic animals. Genetic changes that decrease the fitness of animals would be expected to reduce their chances of survival and reproduction. Consequently, a deleterious mutation should be eliminated by natural selection. However, if the mutation occurs in germ cells and is recessive, then it will be expressed only by individuals homozygous for it and it may persist and even multiply for many generations. Mussels contaminated by the *Erika* spill were fed to rats and genetic damage was identified in liver of the rats (Lemiere et al., 2005). Further research with improved molecular technologies can be expected to help determine the significance of genetic effects for animals in general or for consumers of contaminated animals.

Discharges of produced water may be problematic in the Arctic if they are released to the environment. The composition of produced water is variable with some components known to be toxic. The volumes of produced water involved can be as much as ten times the total hydrocarbon production from a well. The trend is towards disposal of produced

water by re-injection and the issue will become academic if re-injection is practised throughout the Arctic.

Noise by seismic exploration and by ships can disrupt the behaviour of local animals, notably marine mammals and moulting or nesting seabirds. The effects of noise can be mitigated relatively easily for marine mammals because the animals are visible and seismic activity can be stopped when the animals are nearby. Oil companies have responded to many of the concerns raised by adapting field programs to avoid critical periods in wildlife life cycles (denning, nesting, or moulting) where possible. Effects of noise on fish are less readily observed but they appear to be temporary except for cases where explosives are used in seismic exploration very near the fish.

Dramatic local effects on populations of benthic animals have been associated with releases of oil-based drilling muds. These discharges differ from spills in the sense that they are planned and regulated for each location. The study of benthic populations is well validated statistically and is very sensitive. Future drilling at Arctic sites is unlikely to use oil-based muds and so the practice of discharging oily waste is unlikely to occur in the future in the Arctic.

In contrast to studies of the benthos, the study of pelagic fish populations following oil spills or planned discharges has been difficult partly because of the limitations of local pre-exposure population data, statistical difficulties, and high mobility of these animals. Attempts to distinguish effects on fish populations of even the most intensely studied spill to date, the *Exxon Valdez* spill, remain controversial. Attempts to attribute causality to spills on the basis of population statistics are often confounded by other changes resulting from independent potential causes. While the relevance of these efforts is high, and their costs are very high, their sensitivity has been low. For this reason, the use of less ambiguous techniques like biomarkers seems likely to become more prevalent with fish. A small suite of carefully chosen biomarkers, perhaps including biochemical and histological measures and taste and odour descriptors, is capable not only of providing evidence of exposure to oil but also of measuring at least some biological responses and of offering strong tests of cause/effect relationships. Population responses by benthic animals range from localized extinction to little or no detected effect. National environmental effects monitoring programs in some countries apply these responses increasingly. Several biomarkers have been proposed for biomonitoring of produced water and oil spill responses in Arctic waters on the basis of laboratory experiments and field background level measurements. Among the best opportunities to define such linkages are spills (accidental and experimental) in relatively contained settings and such spills should be seen as opportunities to advance knowledge. Areas of known natural seepages appear to offer opportunities to learn how organisms adapt to long-term exposures but studies of these areas have been restricted to chemical studies of hydrocarbons.

In terms of risks to animal populations, the geographic areas of greatest economic and public health concerns are those where commercial fishing occurs or where there has been traditional harvest of subsistence foods. If fisheries are closed after a spill event, then financial loss to the fishing industry can be severe. Compensation is already in place with some Inuit. If the animals should be killed or become unfit for consumption, then the harvest is lost and subsistence communities must either suffer shortages of food or change to commercial foods, a change that imposes financial and cultural burdens (e.g., Walker and Field, 1991). Biologically, the greatest risks are nesting and nursery areas and migration corridors where young animals are found since they are generally more susceptible to oil than older individuals of the same species. Shallow waters where light can penetrate are areas of special interest in view of the interaction between PAHs and sunlight.

Much of the literature concerning toxicology of oils to seabirds is now several decades old but indicates that crude oil is generally low in toxicity to adult birds, and sub-lethal effects from chronic exposures generally decline when the exposure ceases. The egg remains the most sensitive life stage, with quantities as low as several microlitres able to kill the developing embryo at critical stages of development. Valuable research after the *Exxon Valdez* spill has also shown that effects on the population might linger for decades after a major spill due to lingering contamination and changes in the biological community. The potential effects of smaller, diffuse spills are still unclear but research in sub-Arctic zones shows that seabirds congregate around offshore facilities and can become contaminated by local spills. Given the higher sensitivity of the egg stage and the ability of the adults to transport oil back to the nest on breast feathers during incubation, the potential exists for effects on local seabird populations without the requirement for large-scale spills. This mechanism of exposure and effect might become more significant as oil and gas activities, and associated activities like shipping, become more prevalent in the Arctic.

5.4. Impacts on terrestrial ecosystems

The terrestrial system in the Arctic consists largely of tundra, with regions of taiga to the south where trees form a larger part of the plant community. A key defining feature of the terrestrial system is the harsh climate, with extremely low temperatures, low or no light for much of the year, low precipitation, and low nutrient availability (see Chapter 6). The soil structure consists of an active layer (i.e., the layer that is subject to annual thawing and freezing) above continuous or semi-continuous permafrost, which is capped by a layer of organic material. This unique structure is critical for maintaining thermal and hydrological balance of the soil. Disturbance of the soil structure, whether by the loss of the organic layer, erosion of the permafrost, or loss of the plant cover can have a severe long-term impact on the terrestrial system.

5.4.1. Uptake, metabolism, and excretion

5.4.1.1. Soils

Hydrocarbons are distributed throughout the Arctic from a combination of natural sources within the Arctic, such as seeps and biogenic sources (AMAP, 1998). Natural and industrial sources continually add hydrocarbons, including PAHs, to the Arctic by atmospheric (Becker et al., 2006), riverine (Fernandes and Sicre, 1999, 2000), and oceanic routes (see Chapter 4). Natural sources, such as volcanic activity, forest and prairie fires, and the decomposition of plant material, and anthropogenic sources, principally oil and gas production and the combustion of fossil fuels, together contribute to the load of hydrocarbons deposited in the north (AMAP, 2004b). Local transport mechanisms, such as dust transport, surface water and ground water seepage, and direct deposition from the atmosphere result in a heterogeneous distribution of hydrocarbons throughout the terres-

trial environment. Key questions are to what extent oil and gas activity changes the background level of hydrocarbons in the Arctic, what biological effects result from increased concentrations of oil and gas-related compounds near facilities and field operations, and what are the impacts of brine and oil spills. Unique hydrocarbon compounds, or groups of compounds, released from anthropogenic sources help to establish the levels related to a specific release (Kaplan et al., 1996). For example, the carcinogen benzo[*a*]pyrene is higher near highways, industrial areas, and developed areas, but has a global background level in soils of about 5–10 μg/kg dw (Sims and Overcash, 1983).

The effects of crude oil and petroleum spilled on soils, and the factors that influence recovery and oil degradation, have been studied in the Arctic for several decades. Atlas (1981, 1985) provided excellent reviews on the weathering of spilled oils and the processes of volatilization, microbial degradation, and photolysis of the individual fractions of the oils. Controlled experiments have been conducted in many Arctic countries on the response of soils to oil exposure, although most were conducted during the 1970s (Collins et al., 1994) and there are very few that apply new findings from recent chemical and biological research, or improved analytical techniques. Studies related to specific sites of contamination, such as flare pits, have shown that hydrocarbons, metals, and salts may leach from contaminated soils (Cook et al., 2002), but the significance of this in the Arctic is probably limited due to the presence of permafrost. Investigations of fuel spill migration in soils at Alert on northern Ellesmere Island, Canada, showed that hydrocarbons did migrate through permafrost at several sites, possibly due to unfrozen water in the soil (Biggar et al., 1998). As permafrost thaws under the influence of climate change, this has the potential to become a significant issue at older sites of oil and gas activity in the Arctic.

Petroleum is a complex mixture of organic and inorganic materials of varying chemical and physical properties, nevertheless some general principles are evident. In general, the n-alkanes are the most readily degraded fraction, with many species of bacteria and fungi able to use hydrocarbons as the sole carbon source (Atlas, 1981). Any factors which promote microbial growth (i.e., increased temperature, moisture, and nutrients) appear to increase the degradation rates of oils (Westlake and Cook, 1975; Sexstone et al., 1978; Mitchell et al., 1979). Measurements of respiration rates and AHH activity in oiled soils show increased microbial activity in the presence of hydrocarbons (Linkins et al., 1978). These processes are much slower under Arctic conditions and oil spills can remain virtually unchanged over several decades if left alone (Atlas, 1985). Collins et al. (1994) reported little weathering of crude oil, and the presence of volatile components in subsurface samples fifteen years after a test spill at Barrow, Alaska. The authors attributed the slow rate of change to a low exchange rate with the atmosphere and to little aerobic microbial activity. In general, the rapid loss of the volatile fraction, which is one of the more toxic fractions in crude oil, is generally considered to be one of the mechanisms whereby the toxicity of spills diminishes rapidly after a spill. Braddock et al. (2003) revisited the Caribou-Poker Creeks Research Watershed 25 yr after experimental spills of 7570 L were released into a black spruce (*Picea mariana*) forest in 1976. They found that the hydrocarbons in areas heavily contaminated with Prudhoe Bay oil were virtually unchanged since the spill, while others had lost over 80% of the original mixture. Similar rates of recovery were observed in a sub-Arctic spill in northern Alberta, Canada, where 9.5 x 10^6 L were spilled from a pipeline over a 10 ha site (Wang et al., 1998). After 25 years, between 15 and 43% of the oil in surface (0–4 cm) soils was lost while subsurface samples (10–40 cm) showed very little loss.

Some of the research in the 1970s related to changes to soil chemistry and structure caused by the application of oils, specifically to address questions of toxicity and soil reclamation under Arctic conditions. Everett (1978) reported that crude oil applied to soil at Barrow, Alaska, reduced the water filtration rates, probably due to increased hydrophobicity of soil particles, and an increase in the seasonal thaw depth. These effects declined as the site was revegetated and the oil decomposed. Everett (1978) also reported changes in the levels of exchangeable cations in the soil, as the hydrocarbons blocked the exchange sites on soil particles. A related study reported that the penetration of the crude oil into the soil depended on soil moisture and drainage characteristics (Sexstone et al., 1978). Degradation of the oil by soil microbes was slower at the drier sites, despite abundant hydrocarbon-utilizing microbes. Again, light hydrocarbons were still observed one year after the spills. Analysis of a diesel fuel spill at Barrow Alaska showed slow movement of the fuel because of shallow permafrost and a short thaw season (Braddock and McCarthy, 1996). Several indicators, including higher bacterial counts, indicated that the fuel was being degraded but that the short thaw season limited the rate of decomposition. A re-analysis of the transport of the fuel (McCarthy et al., 2004) showed that the product probably penetrated the permafrost to a depth of at least a metre, and that free-phase hydrocarbons were still present 20 years after the spill. Some of the fuel was recovered using a trench which intersected the plume, but pockets of the fuel remain and are probably separated by permafrost (McCarthy et al., 2004). These and many other studies laid the groundwork for the understanding that oil degradation and detoxification is much slower in the Arctic because of the physical restrictions of the cold climate, low nutrient availability and dry conditions. These factors significantly extend the length of time that the crude oil retains its most toxic components.

The concentrations of the more toxic hydrocarbons, such as PAHs, are a major concern for soils contaminated with crude oil and petroleum products. PAHs are a major component of crude oils and some, such as benzo[*a*]pyrene, are known carcinogens. Estimated average half-lives of PAHs in soils under temperate conditions range from about six months for low molecular weight PAHs up to several decades for benzo[*a*]pyrene, the PAH of primary toxicological concern. Significantly longer half-lives for hydrocarbons are expected in the Arctic. PAHs are generally removed from soils by volatilization and microbial degradation (Environment Canada, 1994), two processes that are significantly reduced under Arctic conditions. A study by Eriksson et al. (2001) showed an average *in vitro* removal rate of 1–2 μg Total Petroleum Hydrocarbon (TPH) /g soil/day using Arctic soils over the 48-day experiment after an initial rate of 30–50 μg TPH/g soil/day at low temperature (7 °C). The higher rate of decline was considered to be typical for soils contaminated with 2500 μg TPH/g soil. In a subsequent study of PAH metabolism in Arctic soils, Eriksson et al. (2003) reported that 100% of naphthalene and 2-methylnaphthalene were removed at 7°C under aerobic and anaerobic conditions after 48 days while other PAHs, such as pyrene and 9,10-dimethylanthracene, were more recalcitrant and remained virtually unchanged during the study, even at a higher incubation temperature (20 °C). Metabolic compounds were similar under aerobic and anaerobic conditions although the rela-

tive concentrations of the products differed and some, such as 9-fluorenone, were stable and increased in concentration to the point of becoming inhibitory to some strains of denitrifying species of microbes (Eriksson et al., 2003).

A similar effect was noted by Coulon et al. (2004) in which the alkane fraction of fertilized, sub-Antarctic petroleum-contaminated soils declined as the PAH fraction remained relatively constant, however the toxicity of the soil (as indicated by Microtox bacterial assay) increased during the study. Four-ring PAHs, such as pyrene and chrysene, were the most recalcitrant compounds. The authors attributed the increased toxicity to the incomplete degradation of heavy fractions and the formation of toxic metabolites. Although the biochemical process of microbial decomposition appears to be similar to that in mammals (Edwards, 1983; Eisler, 1987), the ultimate fate of the decomposition products is still unclear. Studies of this type add important details to the mechanisms and conditions under which petroleum, and PAHs, are metabolised in Arctic soils. The accumulation of stable compounds to levels that may be toxic to some species of soil microbe indicates an important area of research.

Another mechanism which has drawn much attention is photolysis, which occurs as the ring structure of the PAHs absorbs light of the visible and UV wavelengths (Arfsten et al., 1996). Although it is probably a more important process in the atmosphere and aquatic systems, photolysis also occurs in the upper, exposed layer of soils. Comparative studies with individual PAH compounds, with and without UV-B irradiation, showed inhibited root growth in *Brassica napus* plants only when the compounds were irradiated (Ren et al., 1996). PAH breakdown products from microbial degradation and photolysis appear to be much more toxic than the parent compounds (Arfsten et al., 1996).

Specialised soil screening tests of mutagenic activity in plants and bacteria show some promise, although their applicability under Arctic conditions is unclear. Earthworms grown on PAH-contaminated soils were shown to have increasing numbers of DNA adducts with increasing exposure to the soils (van Schooten et al., 1995). A review of mutagenic activity in soils (White and Claxton, 2004) compiled a number of studies reporting genotoxic and mutagenic endpoints in plants and bacterial mutagenicity tests to measure the genetic toxicity of soils, including those contaminated with petrochemical wastes. Genetic endpoints in plants included sister chromatid exchange, micronuclei, and anaphase aberrations in the root tips. Of the 1312 assessments from 118 studies, White and Claxton (2004) found very few mutagenicity studies using Arctic soil organisms. The ecological significance of these genetic changes is largely unknown but they do indicate the presence of compounds with a known biological activity. The accumulation of mutagenic compounds at petroleum-contaminated sites in the Arctic, should be more closely examined particularly given their slow metabolism under low ambient temperatures.

In response to the known toxicity and mutagenicity of hydrocarbons and PAHs in soil, several countries have developed soil quality criteria, or guidelines, to assist in remediating petroleum-contaminated soils. The Total Petroleum Hydrocarbon Working Group was formed to develop scientifically defensible information for soil clean-up levels that are protective of human health at sites contaminated by petroleum (Twerdok, 1999). The group recommended that the spilled petroleum mixture be analysed to individual fractions, and that the concentration of those fractions be compared to reference doses to derive hazard estimates. Compounds of concern include carcinogenic and non-carcinogenic indicator compounds (Weisman, 1998). Additional factors such as the physical and chemical properties, and the fate and transport of specific chemical components, are also considered.

EPT (2003) reviewed the methods used in several countries to develop ecological soil screening levels for contaminated soils for the U.S. Environmental Protection Agency (OSWER, 2005). Several methods are used to estimate the levels of hydrocarbons in soil that protect humans and the environment, generally for remediation purposes. The methods usually range from use of the lowest reported toxicity value from a relevant study divided by uncertainty and safety factors (Jensen and Sverdrup, 2003) to the selection of a percentile from a distribution of toxicity values (e.g., protection of 10 or 20 percentile). The methods may also include modelling the distribution of components between soil particles, interstitial moisture, and the rhizosphere.

The Canadian Council of Ministers of the Environment (CCME, 2000) uses a weight-of-evidence approach to derive Threshold Effects Levels that may include effects on soil microbes and wildlife, depending on the major compound involved. For hydrocarbons found in petroleum, the Canadian soil quality guidelines provide several levels of protection (Table 5.13), depending on the use of the land and the pathways of exposure to humans and wildlife. For example, more protective values are given for land to be used for residential purposes (because of increased exposure to humans) and parkland (because of increased exposure to wildlife), than for industrial uses. In recognition of the increased toxicity of lower boiling point fractions of complex petroleum, recently proposed Canada-wide standards for the remediation of petroleum hydrocarbon-contaminated sites include lower concentrations for shorter straight-chain hydrocarbons (nC_6 to nC_{10}) than for longer chain hydrocarbons (CCME, 2000).

Soil quality guidelines for hydrocarbons and PAHs from four countries are summarized in Table 5.13. Most countries recommend different levels of protection for different land uses, such as residential or agricultural. The most protective guidelines are generally for agricultural uses. The Canadian guidelines also consider the texture of the soil, with more restrictive levels in fine soils for the more volatile components. The greatest difference in guideline levels is for toluene, with similar levels recommended by Canada, Norway, and Russia (~0.3 mg/kg), while the U.S. value is considerably higher (300 mg/kg).

5.4.1.2. Plants

The uptake and metabolism of petroleum hydrocarbons by plants has been an area of active research owing to the potential for their movement through the food web into wildlife and human consumers. This is particularly the case for PAHs that may be broken down into carcinogenic or mutagenic by-products. Other studies concern the use of plants in the phytoremediation of petroleum-contaminated sites. Several northern species were tested for their ability to remove diesel fuels from soil in a sub-Arctic environment (Palmroth et al., 2002) and some, primarily legumes, were able to accelerate the removal of diesel fuel from soil, although complex components of the diesel fuel were still present after 180 days and a year. Phytoaccumulation was not observed in any of the pine, poplar, legumes, or grass species tested by Palmroth et al. (2002). Owing to the potential for human exposure most of the literature on this subject concerns uptake and metabolism in grains and vegetables, with very little on Arctic plant species or controlled experiments conducted under Arctic conditions.

Table 5.13. Criteria for the protection of the environment and human health for selected hydrocarbons, PAH compounds, and metals in soils (mg/kg dw soil) (PINARO, 2006; Buchman, 1999; CCME, 2004b; SFT, 1999).

	Canada				USA		Norway	Russia
	Agricultural use	Residential use	Commercial use	Industrial use	Urban park/ residential	Agricultural		MPC
	All soils	All soils	All soils	All soils	All soils	All soils	All soils	All soils
Benz[*a*]anthracene	0.1	1	10	10	1	0.1		
Benzo[*a*]pyrene	0.1	0.7	0.7	0.7	1	0.1	0.1	0.02
Benzo[*b*]fluoranthene	0.1	1	10	10	1	0.1		
Benzo[*k*]fluoranthene	0.1	1	10	10	1	0.1		
Dibenz[*a,h*]anthracene	0.1	1	10	10	1	0.1		
Fluorene							0.6	
Fluoranthene							0.1	
Indeno[1,2,3-*cd*]pyrene	0.1	1	10	10	1	0.1		
Naphthalene	0.1	0.6	22	22	1	0.1	0.8	
PAH (sum of 10)							2	
Phenanthrene	0.1	5	50	50	1	0.1		
Phenol	3.8	3.8	3.8	3.8				
Pyrene	0.1	10	100	100	1	0.1	0.1	
Nonchlorinated aliphatics	0.3							
Cadmium		10						
Chromium (total)		64						
Chromium (Cr VI)		0.4						
Lead		140						
Mercury (inorganic)		6.6						
Cadmium		10						
Chromium (total)		64						
Chromium (Cr VI)		0.4						
	Coarse soil	Fine soil	Coarse soil	Fine soil	Coarse soil	Fine soil	Coarse soil	Fine soil				
Benzene									0.008	0.008	0.005	0.3
10^{-5} incremental risk (surface)	0.03	0.0068	0.03	0.0068	0.03	0.0068	0.03	0.0068				
10^{-5} incremental risk (subsurface)	0.03	0.0068	0.03	0.0068	0.03	0.0068	0.03	0.0068				
10^{-6} incremental risk (surface)	0.0095	0.0068	0.0095	0.0068	0.03	0.0068	0.03	0.0068				
10^{-6} incremental risk (subsurface)	0.011	0.0068	0.011	0.0068	0.03	0.0068	0.03	0.0068				
Ethylbenzene	0.082	0.018	0.082	0.018	0.082	0.018	0.082	0.018			0.5	
Toluene	0.37	0.08	0.37	0.08	0.37	0.08	0.37	0.08	300	300	0.5	0.3
Xylene	11	2.4	11	2.4	11	2.4	11	2.4	5	0.1	0.5	

Many plants have the ability to metabolise hydrocarbons. Depending on their physical properties, hydrocarbons enter plants through the leaves, through the stomata, or by absorption into the root (Figure 5.1). Chemicals deposited onto the leaf surface accumulate on the cuticle and gradually penetrate the leaf or enter through stomata (Ugrekhelidze et al., 1997). Plants are also known to scavenge semi-volatile organic compounds, such as PAHs, from the air, and to provide a surface for microbial degradation and photodegradation (Smith et al., 2001).

Uptake by absorption into the root depends on the concentration and molecular weight of the chemical, soil temperature, and the species of plant (Korte et al., 2000). In contrast to these and other studies, corn grown in soils contaminated with 1% fuel oils (low K_{ow}) did not accumulate any of the hydrocarbons in the soil mixture (Chaîneau et al., 1997), whereas plants grown in clean media and then placed in fuel-oil contaminated soil, rapidly accumulated hydrocarbons. This raises the possibility that plants in clean environments dosed with hydrocarbons from spills may accumulate these compounds in stems and leaves. Depending on the chemical involved, the compounds taken up may be released back to the atmosphere or through the roots to the soil. A detailed study of factors involved in the scavenging of PAHs from the air showed that, at the field scale, there were few differences between species in the uptake and retention of PAHs but large differences in the accumulation of some PAHs between seasons. Concentrations of naphthalene, for example, doubled in all five species in winter (Smith et al., 2001), indicating that temperature has a significant effect on air-plant exchange for some PAHs. In many cases, PAHs reported in vegetation are from aerial deposition onto the plant leaves or fruiting body. Factors influencing the air-to-plant partitioning of PAHs include plant architecture and ecology, the physical structure of the plant leaf surface, the gas-particulate distribution of PAHs in the air, and environmental conditions (Smith et al., 2001).

Plants are known to metabolize hydrocarbons, which is the process underlying the phytoremediation of contaminated sites although its effectiveness under Arctic conditions is probably limited. The major enzymatic process metabolising xenobiotic compounds appears to be hydroxylation by the inducible cytochrome P-450 monoxygenases. Plants may also accumulate compounds in a non-toxic form, following their conjugation with endogenous compounds such as sugars and organic acids (Korte et al., 2000). Alkanes and cyclohexanes may be metabolized and the carbon chains used in cellular respiration. Radiolabelled benzene and toluene have been shown to enter plants through leaf stomata and then to be hydroxylated or cleaved and included in cellular processes (Ugrekhelidze et al., 1997). The effects of these compounds and their by-products on the internal structure of the plants range from almost nothing to complete cell destruction and plant death (Korte et al., 2000). Vascular plants take up and then metabolise or conjugate hydrocarbons into chemical forms that may be useable by the plants. There do not appear to be any studies examining these mechanisms in Arctic species, particularly in the non-vascular lichens and mosses, or any studies determining whether the mechanisms and rates of uptake and metabolism are similar in Arctic species and species from temperate regions.

5.4.1.3. Mammals and birds

The literature on the uptake and metabolism of oil and petroleum hydrocarbons in mammals is extensive (Hellou, 1996; Albers, 2003). Petroleum can affect organisms through physical action and smothering, by alteration of habitat and by toxicity (Albers, 2003). Exposure can occur after dermal exposure, inhalation, and the ingestion of contaminated food and water (Figure 5.1). While earlier studies focused on crude oils and refined products, more recent studies have focused on individual components of the complex mixtures, in particular individual PAHs. Many of the studies relating to the toxicity of hydrocarbons have used small mammal models, resulting in an enormous amount of literature on this topic. The presence of PAHs in cigarettes has also resulted in a vast literature on the effects of inhalation of hydrocarbons in small mammal models. Other than oiling studies, very few studies have been conducted on Arctic bird or mammal species, or have attempted to study the effects of petroleum toxicity on wildlife under severe Arctic conditions.

The chemical and physical properties of the various compounds in crude oil and petroleum control exposure pathways and rate of uptake, and the mechanisms by which organisms can metabolise or excrete these compounds. The major difference between birds/mammals and invertebrates is in the activity of mixed function oxygenase enzymes that biotransform PAHs to more polar, water-soluble metabolites that can be further metabolized or excreted (Hellou, 1996). A full review of the toxicity and metabolism of PAHs has been published by the Agency for Toxic Substances and Disease Registry (ATSDR, 1995). Although the review addressed PAH toxicity to humans, most of the studies used mammalian models and so are directly relevant to this assessment. Owing to the efficient mechanisms by which birds and mammals metabolise hydrocarbons, including PAHs, measurement of the compounds in tissues is generally a poor indicator of how much the animal has been exposed to, or whether a biological effect has occurred.

The enzyme systems responsible for these transformations are broadly classified as phase I and phase II enzymes. Phase I enzymes introduce reactive, polar functional groups onto lipophilic toxicant molecules; with the products usually more water soluble than the parent xenobiotic species. Phase II enzymatic reactions produce a conjugation product that is amenable to excretion from the body. Recently, studies of the genetic basis of polymorphic metabolizing enzyme activities involved in drug activation and detoxification have been elucidated. These studies gathered information on the association of genetically determined metabolic variants with the risk of environmentally induced cancer. This type of gene-environment interaction may alter the general view of dose-response patterns and how to characterize risk in an exposed population (Kuljukka-Rabb et al., 2002). It has become increasingly clear that most enzymes involved in hydrocarbon metabolism occur as several isozymes (Hahn, 2002) and that these coexist within the same individual and frequently, within the same subcellular organelle. Cytochrome P450 1B1 (CYP1B1), is overexpressed in many tumours and catalyzes the metabolic activation of procarcinogens such as PAHs. In humans, some of the most toxic components of the by-products of benzo[*a*] pyrene metabolism bind with DNA to form BaP-DNA adducts that may lead to cancer. There is evidence of high variability in the production of BaP-DNA adducts between individuals, with induction in normal human mammary epithelial cells showing a 100-fold variation (Keshava et al., 2005).

Little is known about the PAH transformation potency of human intestinal microbiota. However, using a gastrointestinal simulator it has been demonstrated that the human intestinal microbiota can also bioactivate PAHs, more in particular to estrogenic metabolites. Colon-digested PAH compounds displayed estrogenicity, equivalent up to 1.48

nmol of 17 α-ethynylestradiol (EE2). Inactivating the colon microbiota eliminated these estrogenic effects. Spectroscopic analysis confirmed the microbial PAH transformation by the detection of PAH metabolites 1-hydroxypyrene and 7-hydroxybenzo[*a*]pyrene in colon digests of pyrene and benzo[*a*]pyrene. Because PAH toxicity is also linked to estrogenicity of the compounds, the PAH bioactivation potency of colon microbiota suggests that current risk assessment may underestimate the risk from ingested PAHs (Van de Wiele et al., 2005). In addition, the ingestion of soil-bound PAH resulted in the induction of cytochrome enzymes in the lung, kidney, and spleen in pigs, however the pattern was very different from the induced pattern in rats (Roos et al., 2004), indicating that the uptake, transport, and biological response in each species might vary considerably.

There is some evidence that these biochemical reactions require a threshold level of petroleum compound before the appropriate enzymes are produced (Hellou, 1996). The concentrations of PAHs reported for some wildlife species may also be the result of some PAH configurations that are resistant to mixed function oxygenase enzymes because of their chemical structure. Additional factors that probably influence the rate of breakdown of the PAHs are the species, age, sex, and reproductive status of the test species, and the type and amount of hydrocarbons present (Hellou, 1996). It is clear from these studies that hydrocarbon uptake, metabolism, and excretion is very complex, but in terms of the present assessment, little information is available to test whether special consideration should be made to understanding hydrocarbon exposure in Arctic species.

5.4.1.4. Concluding comments

Most research in the area of petroleum- or oil-contaminated soils has focused on the decline of oil-related compounds through time by microbial and physical processes, however little research has been conducted on the presence of stable, toxic metabolic by-products that might accumulate in soil. This issue will continue to increase in importance as more sites are developed, decommissioned, and abandoned in the Arctic, particularly over the next decades. Newer techniques of chemical analysis, such as 2-D gas chromatography combined with mass spectrometry, should be used to resolve and identify compounds, particularly in the unresolved complex mixtures (UCM) (Gough and Rowland, 1990; Wraige, 1997). Using these techniques it may be possible to quantify compounds with known physical, chemical, and biological properties of concern. Soil quality guidelines are designed to be protective of humans and the environment, but their suitability in the Arctic is unknown.

The physical and chemical character of hydrocarbons in Arctic soils has been well studied although much of the work was conducted about thirty years ago in a few well-studied experiments. Continued follow-up studies on the sites are vital to understanding the long-term recovery of spills and contaminated sites in the north. Improving the clearance of oil from soils by enhancing microbial activity and volatilization appears to hold considerable promise in cleaning historic spills, however the suggestion that stable metabolites that are toxic to normal soil processes (e.g., denitrifying bacteria) needs to be examined more closely. The significance of the accumulation of stable, toxic compounds produced from the metabolism and photolysis of hydrocarbons at Arctic sites is still unknown, but the expected increase in oil and gas activity in the near future will probably result in an increase in sites to be remediated and decommissioned over the next few decades. The current methods used to reclaim these sites may not be suitable for the long-term protection of the Arctic. Also, efforts should be made to explicitly test the suitability of the guidelines for the protection of the Arctic using, for example, cold ambient conditions for the transfer of compounds and estimating the uptake into a receiving environment using data from the north.

The literature relating to hydrocarbon metabolism in birds and mammals is extensive because of the use of small mammal models to understand hydrocarbon toxicity, particularly PAHs, in humans. Mammals and birds in the Arctic are expected to absorb, metabolise, and excrete hydrocarbons and PAHs in the same manner as in southern latitudes, although there are genetic differences in the aryl hydrocarbon receptors that explain differences between species in terms of their ability to metabolise specific compounds (Hahn, 2002). No information was found on the possible interaction between contaminant metabolism and unique characteristics in terrestrial northern species, such as long migrations, specialised physiology and extended hibernation. There is a general lack of knowledge of bird and mammal toxicology in northern species and this does not only apply to oil- and gas-related compounds. The carcinogenicity of the various PAHs that drive much of the human research is not considered to be a significant stress in the maintenance of wildlife populations, relative to other stresses such as disease, habitat loss, and ecological factors. Currently, exposure to PAH compounds and other hydrocarbons appears to be low in terrestrial systems at northern latitudes (Gamberg et al., 2005), although this conclusion needs to be kept under review in terms of local contamination near oil and gas facilities and near brine and oil spills.

5.4.2. Lethal effects

5.4.2.1. Plants

The toxicity of short-chain alkanes and aromatics to plants is well known. Diesel oil has been used as a herbicide because of its efficiency at killing weeds while crops such as carrots remain unaffected (McKendrick, 1999). The short-chain hydrocarbons penetrate and destroy cell membranes, causing the plant to die if a large enough fraction of the plant is affected (McKendrick, 1999). These observations are consistent with those in several oil spill studies showing continuing toxicity to trees and shrubs exposed to spilled crude oil through roots.

Although spills of crude oil and petroleum products are not generally considered to be major threats to the terrestrial environment from well maintained pipelines, leakage does occasionally occur and spills are much more frequent on poorly maintained pipelines (Vilchek and Tishkov, 1997) and during field operations (see Chapter 2). In order to understand the impact of crude oil spills on the terrestrial environment, oil spill experiments were conducted during the 1970s and 1980s in Alaska, Canada, Greenland, and Russia. In most countries, the rationale for the studies was to understand the impact of spills in the northern environment, but also to assess the most appropriate remediation measures when spills occurred. In some of the experiments the impacts of the spills were studied several years after the initial spill to determine long-term impacts and recovery at the sites (Collins et al., 1994; McKendrick, 1999). These considerations become particularly important in the tundra and taiga of the Arctic, as the extremity of factors such as hummocky landscape physiography, temperature, and dryness have major impacts on oil spill behaviour and recovery rates in the affected area.

A major question in determining the extent of oil spill impacts is the means by which the oil is spread. The majority of oil spill studies have examined the movement of crude oil or petroleum products pumped from a single point source, and observed the rate of dispersal on the surface and penetration through the underlying active layer (the seasonally thawed layer overlying permafrost). Rates of oil coverage in these studies were generally from 5–12 L/m^2. The area and depth covered by the oil depends on the depth of the surface layer, the presence of water (which deflects the oil from entering the surface layer), and the roughness of the terrain. Other studies have shown that oil can be dispersed under pressure, such as from a pressurized pipeline, and spread over a large area by wind. Under these circumstances, the amount of oil spread per unit area tends to be lower than in the point source studies, but the areas affected are larger. The third type of dispersal occurs with buried pipelines, where the released oil moves from the pipeline trench and possibly through the active layer above the permafrost (Kershaw, 1990). This form of spill occurred at the large Komi spill in Russia where leaks from a buried pipeline saturated local soils and moved to local wetlands and waterways (see Chapter 4) (Zoltai and Kershaw, 1995). In this scenario, oil was released at a relatively slow rate but little was absorbed, so a much larger land area was covered than was predicted from previous studies. Once released from the pipeline, the oil moved through the active layer and then to the soil surface when the active layer became saturated. From these studies it is apparent that the area of land covered depends on the absorptive properties of the surrounding tundra, moisture levels, the amount and type of oil, and the season in which the oil was released (Zoltai and Kershaw, 1995).

Early experiments conducted in taiga near Norman Wells and tundra near Tuktoyaktuk, Canada between 1972 and 1975 (Hutchinson and Hellebust, 1978) provided some initial results on the impact of spilled crude oil in these biomes. Additional studies on the effects of diesel fuel and the rates of recovery after fertilization to promote growth were also undertaken. Approximately 8000 L of Prudhoe Bay crude oil were applied to the study plots at a rate of 9.1 L/m^2. Most exposed plants died rapidly after treatment, with black spruce continuing to die over the next few years (Hutchinson and Hellebust, 1978). The foliage of black spruce which was exposed directly to the sprayed oil died quickly after contact, while unsprayed foliage remained active until the tree succumbed to the oil treatment some time later. All the vegetation of the sprayed larch (*Larix* sp.) died, regardless of whether it was sprayed or not, and the trees did not regrow the following season (Hutchison, 1984).

These results correspond with root respiration studies (Linkins et al., 1978) that showed declining root biomass in taiga plants over a period of several years following exposure to a spill of crude oil. Hutchinson and Hellebust (1978) reported that several (six to nine) years after the spills, extensive effects on the plant community were still evident, with poor recovery in mosses and lichens. Most of the recovery was observed in dwarf shrubs such as Labrador tea (*Ledum decumbens*), birch (*Betula* sp.), and willow (*Salix* sp.). In general, summer spills of crude oil showed greater reduction in species and slower recovery than winter spills.

Walker et al. (1978) spilled crude oil on six plots of differing plant communities at Prudhoe Bay, Alaska in order to develop species sensitivity maps which would aid the placement of pipelines and clean-up after spills. The six plant communities covered a range of topographies, moisture gradients, and dominant plant types. Prudhoe Bay crude oil was spilled under dry summer conditions, at a rate of 12 L/m^2. Most plants were killed upon exposure to the oil, but after one year various species showed signs of recovery. Sedges (*Carex* sp.) showed moderate to good growth, as did some shrubs. The study supported the observations from previous studies that certain plant groups were probably killed by the oil because of their particular growth form. Sedges and shrubs tend to be more resilient because they develop new plant shoots from rhizomes that are not exposed to the oil. Grasses and lichens were quickly killed and did not recover. The study also showed that crude oil tended to be less toxic under wet conditions, as the oil remained on top of the water and did not saturate the soil and root systems of the plants. The study reported that the extent of damage was determined by the species and growth forms present in the exposed area, the wetness of the community and the level of exposure to oil.

In one series of experiments (Jenkins et al., 1978), Prudhoe Bay crude oil was applied to an area of taiga with black spruce and an understory of Labrador tea, birch (*Betula glandulosa*) and blueberry (*Vaccinium uliginosum*). Various mosses and lichens covered about 50% of the ground. Approximately 7570 L of oil were applied on top of snow cover in the winter at -5 °C and, in a separate experiment, during the summer at an air temperature of 25 °C. Oil released in the winter, penetrated the snow in several places, continued to move downslope for the first 24 hr, and remained stationary until warmer weather, when it moved a further 17 m. The total area covered was 185 m^2, giving a dose rate to the plant cover of 41 L/m^2. Oil released in the summer penetrated the soil cover rapidly, and continued to move downslope for 24 hr. The total area covered in the summer (303 m^2) was considerably larger than in winter, giving a coverage rate of 25 L/m^2, although only about 10% of the oil was visible at the surface, with most moving by subsurface flow.

This experiment is notable for several reasons. As in other studies (Hutchinson and Freedman, 1978), virtually all aboveground foliage that came into contact with oil was quickly killed in both series of experiments. This included virtually all plants in depressions and hollows where oil pooled. In the second growing season after the spills, foliage of black spruce, Labrador tea and cranberry (*Vaccinium* sp.) turned brown and died. In general, damage was more extensive in the summer spill, possibly because the heated oil caused more heat damage and covered a larger area. Importantly, in contrast to other studies, no regrowth was observed on any species that lost its foliage as a result of oil damage.

The sites from this experiment were revisited in 1991 (Collins et al., 1994) to determine the amount of vegetation recovery and to record physical changes in the soil and remaining oil. Both spill sites had increased by about 1–2% in area in the 15 yr since the spill. The depth of thaw of the permafrost, which was thought to be insignificant in the original study was found to have increased significantly under the winter spill, ranging to a maximum depth of 60 cm. The effect was attributed to increased heat retention on the oil-blackened surface which thawed the permafrost below the spill. Although the oil had weathered significantly during the 15 yr, subsurface samples of the oil still retained the more toxic volatile hydrocarbons showing that the degassing of the volatile fractions may not occur under some conditions. Some samples also showed significant microbial decomposition of the alkane fraction, while other samples, usually those from subsurface sites, showed few compositional changes.

After 15 years, the spill areas showed continuing effects on plants and few signs of recovery. Most mosses, lichens, and shrubs were killed soon after the spill, with the only surviv-

ing plants being tussocks of *Eriophorum vaginatum* that were raised above the oil. These areas continued to grow well after 15 years. Several trees died one year after both the winter and summer spills, however by 1991, only 18% of the 151 black spruce trees remained healthy on the winter spill site while 33% of 155 trees were healthy on the summer spill site. No trees had died in the surrounding reference areas. In addition, there were no signs of black spruce seedlings in the oiled areas (Collins et al., 1994).

The oil spill studies were significant because they showed the physical behaviour of crude oil and diesel fuel on the Arctic landscape, and the biological effects over both the short- and long-term. Contact with the oil generally resulted in the death of the plant, and little re-growth or recovery for several years after the experiment. Lichens and mosses, major species in the tundra, were rapidly killed and did not recover for several years. The plant species which recovered tended to spread by rhizomes that were protected in the initial spill. Studies should continue at these sites, using recent improvements in analytical methods, to determine the long-term rates of recovery. If possible, the re-analysis of specimens collected in the early stages of the studies may also provide critical information on changes in oil residues at the site.

A major objective of many oil spill studies has been to determine the best approach to reclaiming sites that have been exposed to accidental spills. Several methods have been proposed, including mechanically cleaning up the spilled oil with heavy equipment, burning, fertilizing the area to promote new vegetation growth, and allowing natural microbial action to break down the oil. Each method has advantages and disadvantages.

The effectiveness of burning spilled crude oil was investigated in five terrestrial habitats in Alaska in winter, summer, and autumn during freeze-up (McKendrick and Mitchell, 1978). In general, burning the spilled oil during the summer growing season was much more detrimental to plants than in winter. There was less damage to the plants if the burning occurred in the autumn when the soil was frozen to a depth of at least 4 cm. Burning during the growing season increased the damage caused by the crude oil in all experimental situations.

Although not in the Arctic, a recent study (Lin et al., 2005a,b) of the burning of an oil spill in a marsh found similar conclusions, with the depth of water over the soil/sediment substrate being a major factor in the ability of plant species to recover. A key finding in the study was that burning removed most of the short-chain aliphatic hydrocarbons and more volatile PAHs. The concentrations of some fractions of short-chain alkanes declined by over 90%, but the degree of removal decreased with chain length, and the concentrations of long-chain alkanes and PAHs with more than four rings increased in the remaining oil (Lin et al., 2005b). Hence, although alkanes declined by 77.5% in diesel fuel and 88.2% in crude oil, the concentration of benzo[*a*]pyrene increased five-fold in the post-burn residue. The authors concluded that burning effectively removes oil from wetlands, but the focus of the study was entirely on the efficiency of the burning process for removing oils and not on the long-term ecological consequences of burning. Other than noting the increase in higher molecular weight fraction in the burn residue, the study did not test for the accumulation of toxic by-products produced during the burn.

A detailed study of the toxicity of crude oil, as the original mixture and as four individual fractions, tested four plant species to help develop Canadian soil remediation guidelines for total petroleum hydrocarbons (ESG International Inc., 2003). The study is notable due to the controlled conditions under which the tests were conducted and the number of test endpoints used to measure biological effects in plants and soil invertebrates. One of the test species was northern wheatgrass (*Agropyron dasystachyum*) which is distributed across the western Canadian Arctic and Alaska. The first fraction (F1) of the crude oil contained mostly low-molecular weight volatile compounds, including benzene, toluene, xylene, and alkylbenzene. The second fraction (F2) comprised about 25% of the crude oil and contained n-alkanes from C_8 to C_{22}, with the most abundant being in the C_{12} to C_{14} range, and about 51% of the alkylated PAHs. The third fraction (F3) contained longer chain n-alkanes and about 47% of the alkylated PAHs.

The study showed significant differences in the toxicity of the four fractions to the plants. Of the four plant species tested, northern wheatgrass was the most sensitive according to a number of seedling emergence and growth criteria. For seedling emergence, the most toxic fraction was F2, with an IC_{50} (inhibitory concentration for 50% of the population) value of 8–12 mg F2/g soil dw, followed by F3 at 60–80 mg F3/g soil dw. For seedling growth, acute exposures with crude oil gave an IC_{50} of 1.05–150 mg oil/g soil dw while F2 showed IC_{50} values of 2.77–8.24 g F2/g soil dw. Under chronic exposures, the relative toxicity of the fractions changed, with F3 much more toxic than other fractions at an IC_{50} of 0.61–54.1 mg F3/g soil dw. The study showed a significant dose-response relationship for many of the biological endpoints (i.e., root and shoot growth, germination, seedling growth) and the development of soil quality guidelines based on rigorous tests of petroleum toxicity under controlled experimental conditions.

5.4.2.2. Mammals

There are very few studies reporting the toxicity of crude oil and its products to terrestrial mammals in the Arctic. Acute tests of weathered *Exxon Valdez* oil up to 5000 mg/kg in the diet of ferrets (*Mustela putorius*) showed no mortality, but small changes in serum albumin levels and decreased spleen weights in the most highly exposed group (Stubblefield et al., 1995). Similar results were obtained with captive mink (*Mustela vison*) which showed no clinical signs of toxicity with exposure to 100 or 1000 μg/kg of crude oil in the diet (Beckett et al., 2002). Studies on sub-lethal effects on the exposed animals reported a number of observations that were consistent with earlier studies. Low toxicity of weathered oil is consistent with studies which show that, in some cases, the most toxic lighter fractions volatilize from the oil soon after the spill.

Chemicals used in the exploration, drilling, extraction, and transportation of oil and gas can be released to the land, water, or air either by accident or as a by-product of routine operations (Edwards et al., 1979). The types of compounds used in the phases of an oil field, their uses, and the relative amounts of each compound used have been summarised in several papers (Coppock, 1997; Edwards et al., 1979; Edwards and Gregory, 1991; see also Chapter 4). The effects of these releases on terrestrial species in the Arctic, and on the northern biological community in general, have not been studied in detail. As a result, prediction of potential impacts of oil- and gasfield chemical exposure to terrestrial Arctic wildlife must rely on studies conducted on wild and domestic species in temperate areas, with extrapolation to Arctic conditions.

There is a lack of studies predicting the possible effects of exposure to oilfield chemicals and releases from oil- and gas fields on large-bodied mammals such as caribou/reindeer and moose, which are clearly of societal and scientific concern. The only comparable studies are those with domestic ruminants that can provide some insight into the effects of oilfield chemicals on ruminants in temperate regions. The impetus

for the research is clinical studies reporting acute exposures to oilfield chemicals causing a high incidence of death in cattle and sheep and a wide range of sub-lethal effects in many of the surviving animals. Exposure generally occurs through direct contact with the spilled chemical or by the ingestion of contaminated food or water (Edwards and Gregory, 1991; Coppock et al., 1995). Coppock et al. (1995) listed several examples of cattle and sheep voluntarily ingesting petroleum, road oil, diesel oil, and water contaminated with aviation fuel. Proposed reasons for the ingestions include thirst, hunger, dietary compensation for chemical imbalances (Coppock et al., 1996), and the result of grazing on poor quality pasture (Edwards et al., 1979). A review of the potential for the Central Arctic caribou herd in Alaska to exhibit similar licking behaviour at sumps and drill sites on the North Slope in Alaska listed several examples of caribou from other herds visiting natural licks, but also licking salt, oil spills, grease fittings, and seismic shotholes (Lawhead et al., 1992). The authors noted that the behaviour has not been observed in the Central Arctic herd, but it is clear that exposure to spilled chemicals via this route can be significant.

The effects of crude oil and petroleum exposure on the animals depend largely on the chemicals present in the mixtures. A review of pathology related to oilfield toxicology (Coppock et al., 1996) reported that the lung is a main target organ where exposure involves petroleum with a high volatile fraction, followed by liver and kidney. Inhalation can occur directly as oil droplets or during emesis or eructation after ingestion of crude oil-contaminated food. In the latter case, the volatile fraction of the crude oil in the gut is passed to the lungs, where it can cause pneumonia and lesions. Lesions from acute crude oil and petroleum exposure were also reported in the gastrointestinal tract, liver, kidney, nervous system, and heart (Coppock et al., 1996). Effects were also noted due to exposure to chromate, arsenic, lead, and glycols and phosphate esters. In general these effects have been observed in animals exposed at very high acute doses. Coppock et al. (1995) cited published work reporting threshold doses of 8 mL/kg body weight for weathered oil, 2.5–5.0 mL/kg body weight for unweathered oil, 4.0 mL/kg body weight for Venezuela crude oil and >1.25 mL/kg body weight for Bunker C oil. These studies clearly show that exposure of wildlife to accidental releases of crude oil or petroleum can be significant and can cause death or major sub-lethal effects in a significant number of the exposed animals.

A more rigorous study on the exposure of ruminants to chronic levels of Pembina Cardium crude oil reported several biochemical changes in individual tissues following exposure (Khan et al., 1996). Doses were significantly lower than those reported from clinical studies of oilfield exposure, however clinical effects at the higher doses (67.4 g/kg body weight) included tremors, vomiting, and pulmonary distress. Cytochrome P-450 enzymes and AHH activities increased markedly in kidney cortex, liver, and lung but returned to normal levels 30 days post treatment. Similar responses were observed in rats exposed by gavage to Alberta crude oil at 0.5–1.25 mL/kg (Khan et al., 2002).

5.4.2.3. Concluding comments

Extensive research has been conducted on the effects of oiling on plants, birds, and mammals for around 50 years, with several good quality studies on terrestrial oil spills conducted within the Arctic. Oil spill studies have shown the long-term impact of crude oil on tundra and taiga biomes. It is apparent from these studies that most plant species are killed upon contact with the oil, and that effects on the plant community may last for several decades. These sites should be revisited, the residual hydrocarbons tested, and the continuing effects on the plants documented with present-day indicators of toxicity. These studies would help define potential effects on the plant communities after spills and would help define clean-up criteria for contaminated sites in the north. Some plant species may recover quickly after a spill due to growth from unaffected rhizomes, however the plant community would not return to the pre-spill composition for a very long time.

Although oiling appears to be the dominant threat to mammals and birds, spills like that from the *Exxon Valdez* show that long-term exposures might be significant, however the frequency and current levels of exposure to oilfield chemicals in wildlife through inhalation, ingestion, and dermal absorption are currently unknown. Localised areas around contaminated sites have not been adequately characterised to allow exposure and risk estimates for terrestrial bird and mammal populations. No studies were found in the primary literature which document exposure scenarios, or the levels and types of chemicals to which wildlife might be exposed from oilfield operations (i.e., from spills to flaring gas). Given the general concern for northern species of mammals and birds as traditional foods for indigenous peoples and for conservation purposes, this is surprising and should be a priority for all areas of production in the Arctic. Although exposure to ground oil spills is unlikely to have major impacts on populations of large terrestrial mammals (e.g., large herds of caribou or reindeer), the potential for exposure in more solitary species and local populations of small mammals and birds is not well known. It seems reasonable to assume that all small mammals and ground dwelling birds near the spill would be killed, or would avoid the spilled area, however the impact of smaller spills and the ecological effects are unknown.

Surprisingly, very little toxicological research has been conducted on the exposure of large terrestrial mammals such as caribou/reindeer to crude oil and brine spills, and the response of the animals to oil spills (e.g., avoidance or attraction). The considerable research conducted on cattle and sheep may serve as a reasonable model of exposure and response in caribou/reindeer, however caribou/reindeer physiology may be markedly different from that of domestic ruminants.

5.4.3. Sub-lethal effects

5.4.3.1. Mammals

Field studies on small mammal populations near petrochemical plants and oil well blowouts at southern latitudes highlight the exposure of mammals to oil- and gas-related contaminants and report a range of toxicological and genetic effects. A series of studies have been conducted on cotton rats (*Sigmodon hispidus*) in areas treated with petrochemical wastes containing petroleum hydrocarbons, PAHs, and a number of inorganic compounds such as metals (e.g., aluminium, arsenic, cadmium, chromium, lead, vanadium) and fluorine (Schroder et al., 1999, 2003; Carlson et al., 2003; Wilson et al., 2003). Soils in the treatment lands contained elevated levels of some PAHs and total petroleum hydrocarbons, which exceeded an action level of 50 mg TPH/kg dw soil for Oklahoma State. Rats living on the sites with elevated PAH and TPH levels had high lead and fluorine levels in bone, and dental lesions associated with elevated level of fluorine (Schroder et al., 2003). A more detailed analysis of dental lesions showed that 46% of the cotton rats from areas with high fluorine levels had dental lesions due to the high exposure (Kim et al., 2001).

The rats from the treated land units also showed significant induction of CYP isozymes, as determined by EROD (ethoxyresorufin *O*-deethylase) and MROD (methoxyresorufin *O*-deethylase) activity, possibly due to exposure to PAHs in soil (Carlson et al., 2003). An associated study also showed changes in several indicators of immune function that were related to the land treatment units. The unit with the greatest effect on the immune system also had the highest concentrations of lead, chromium, cadmium, fluorine, and PAHs (Rafferty et al., 2001; Wilson et al., 2003). Laboratory and Norway rats (*Rattus norvegicus*) exposed to soil contaminated with petroleum waste, but fed a commercial diet, showed a significant induction of liver and lung cytochrome P450-dependent monoxygenases, probably due to PAH in the soil (Fouchécourt and Riviére, 1996; Fouchécourt et al., 1999). The uptake of petroleum hydrocarbons has also been shown in the case of an oil well blowout which spread crude oil up to 13.6 km from the well site. Heermann's kangaroo rats (*Dipodomys heermanni*) collected at two sites (0.5 and 1.9 km from the well) had accumulated several hydrocarbons in the liver and had a PAH profile similar to that in oil from the blowout (Kaplan et al., 1996).

Other studies have hypothesized that gastrointestinal parasites in cotton rats at oil refinery sites would be affected by the levels of the complex mixture of petrochemicals in the soils there (Faulkner and Lochmiller, 2000). A comparison of gastrointestinal parasite communities in rats showed that the abundance of one parasite species was two-fold higher in rats from reference sites, and the number of parasite species was consistently higher at reference sites relative to contaminated sites (Faulkner and Lochmiller, 2000). Together, these studies provide a consistent picture of exposure and sub-lethal effects in mammals in a habitat contaminated with oil and gas-related contaminants. Pathways of exposure are probably through inhalation and the ingestion of contaminated soil and food.

Studies have also shown significant changes in the immune system of exposed mammals, although the significance of the responses is unclear. Measurements of immunotoxicity in cotton rats collected from an abandoned refinery showed that some indicators of immunocompetence were significantly higher at sites contaminated with petroleum wastes relative to reference sites. Other indicators, such as the mass of immune organs and macrophage function, were not significantly different between the levels of exposure (McMurray et al., 1999). The authors suggested that the complexity of the immune system and the chemicals at the refinery mean that immune suppression could occur in response to some contaminants or stimulation of immune function in response to others.

In a study of immune function in beef calves exposed to hydrogen sulphide (H_2S), sulphur dioxide (SO_2) and volatile organic compounds near oil and gas fields in Canada, the number of CD-4 T-lymphocytes was depressed in calves exposed to the highest quartile of benzene and toluene, compared to the lowest quartile exposure, indicating reduced immune function. Respiratory lesions were also more common in the calves exposed to the higher levels of benzene and toluene (Wickstrom et al., 2006b). There was no association between exposure to particulate matter, PAHs, or metals and immune function in beef cattle (Wickstrom et al., 2006a). A similar study in western Canada recorded calving in seven large cattle herds over seven years and correlated several health outcomes (twinning, abortion, stillbirths, neonatal mortality) with proximity to sour-gas flaring, battery flaring, active gas wells, and larger field facilities (Waldner et al., 2001a). No clear relationships were found between exposure to an oilfield source and a birth outcome. Sour-gas flaring was associated with increased calf mortality risk, but only for one year. Volume of flared sour gas was associated with increased risk of stillbirth however and several relationships were also found between exposure and increased productivity in the herds (Waldner et al., 2001b).

Studies have also focused on the genetic implications of exposure to the complex crude oil mixtures because of the mutagenic and genotoxic properties of some PAHs and some metals found in crude oils. In one study, laboratory rats were fed blue mussels contaminated by oil, with PAH concentrations of 100 and 500 μg total PAH/kg dw (Lemiere et al., 2004). DNA damage was observed in the liver and bone marrow of the exposed rats, probably due to the levels of non-substituted PAHs and other oil-related compounds in the mussels (Lemiere et al., 2004). Examination of the genetic diversity of a single gene locus in cotton rats at a petroleum-contaminated site showed no changes in gene diversity which could be attributed to exposure to petroleum-related contaminants (Pfau et al., 2001). On a larger scale, the loss of a large fraction of an exposed population due to a large spill could result in a decrease in the genetic diversity of the population which may make the population more susceptible to future environmental changes (Bickham et al., 1998).

Similar studies were conducted on mink and sea otters to investigate the effects of crude oil following the *Exxon Valdez* spill (Bickham et al., 1998). Females and kits fed weathered crude oil did not show chromosomal damage, but the genome size of the exposed animals was larger than in controls, possibly due to genotoxic compounds in the weathered crude oil. It is not known if the increased genome size was a somatic or a heritable response (Bickham et al., 1998). About a third of the sea otters captured at the *Exxon Valdez* site one and a half years after the spill showed genetic damage which could be attributed to exposure to the crude oil. Similarly, DNA breaks were observed in the leucocytes of Richardson's ground squirrels (*Spermophilus richardsonii*) captured near sources of SO_2, H_2S, and other volatile organic compounds in the oil and gas industry in western Canada (Wikstrom et al., 2006a).

Studies following the *Exxon Valdez* oil spill provide critical information on the potential effects on small mammals near spills but also demonstrate the difficulty in predicting the short- and long-term changes in complex biological communities after catastrophic events. The synthesis of information from these toxicological and ecological studies has led to major advances in understanding the population- and community-level effects that occur during and after large spill events (Peterson et al., 2003). In the case of the *Exxon Valdez* spill, the effects of the spill on sea otter (*Enhydra lutris* spp.) and river otter (*Lontra canadensis*) populations has drawn considerable scientific debate and produced a number of field and laboratory studies that extend the understanding of sub-lethal effects of exposures of oil-based contaminants on wild mammal populations. Some of the uncertainty about the extent of the effects is due to incomplete information about the populations prior to the spill and differences in methodology of the respective studies (Garshelis and Estes, 1997b; Peterson et al., 2001). The studies also show the importance of accurate baseline information prior to catastrophic events and appropriate statistical designs for post-spill studies (Peterson et al., 2001).

There have been several studies on the toxicological effects of Prudhoe Bay oil on the river otter (Bowyer et al., 2003), a semi-aquatic mammal that spends much of its time in the terrestrial environment, but feeds on marine fish and invertebrates in the intertidal and sub-tidal zones (Bowyer et al., 1995). Direct effects include changes in blood enzyme levels (Duffy et al., 1994; Ben-David et al., 2001) and faecal porphyrin profiles and smaller body size in river otters from oiled

Table 5.14. Summary of major environmental impacts predicted in the final environmental impact statement for Prudhoe Bay and the Trans-Alaska Pipeline System, 20 March 1972 (Maki, 1992).

Impact categories
Unavoidable impacts from construction and operation.
Threatened impacts from oil spills.
Impacts resulting from increased human populations in isolated areas.
Vegetation resources
Covering of tundra vegetation with gravel for roads, pads and buildings.
Rising of permafrost table under the gravel base pads, creating damming effects on lateral movement of water during summer.
Additional risks to tundra plants from oil spills.
Vegetation impacts due to exhaust emissions of sulphur dioxide, estimated to average approximately 0.032 ppm.
Freshwater fishery resources
Additional erosion and siltation from construction activities and ongoing maintenance resulting in reduced light penetration, productivity, and mortality of fish eggs.
Additional threats due to oil spills and resulting toxic effects of stream and tundra pond aquatic life.
Large terrestrial mammals
Disturbances associated with oil field construction, maintenance, and operations, including ground and air traffic along roads and pipeline rights-of-way. Species include grizzly bear, caribou, wolf, musk-ox, moose.
Physical disruption of habitat by construction and operation of the oil field will result in displacement of these mammals. Displaced animals would be expected to succumb to mortality or to displace other resident mammals.
Effects of above-ground sections of pipelines on large mammal movements cannot be conclusively predicted. Elevated sections of pipelines, even those with frequent provisions for animal crossing, still act as partial barriers to movement. Potential odours and sounds from the pipeline could further inhibit movement.
Marine mammals in the Beaufort Sea
Primary threats to marine mammals are from potential oil spills. Maximum volume of oil that could drain from a Trans-Alaska Pipeline System (TAPS) pipeline rupture would be approximately 64 000 bbls.
Birds
Construction and operation could contribute to an ever increasing attrition of bird populations through removal of habitat.
Disturbances due to noise, traffic, and operations would probably drive away, at least temporarily, all birds from construction sites.
Geese, swans, loons, cranes, and raptors are less tolerant of disturbanceOil spills could result in widespread bird mortality due to oiling of important waterfowl production areas.

areas (Duffy et al., 1993). Indirect changes included changes in the home range and diet, as populations of food items in the intertidal habitat were affected by the oil (Bowyer et al., 1994). A number of physiological changes related to oil exposure included an increase in the energetic costs of feeding and a reduced diving ability (Ben-David et al., 2000). Although the number of sea otters directly affected during the spill is under debate (Garrott et al., 1993; Garshelis, 1997; Garshelis and Estes, 1997a), post-spill studies indicate that the population in the most heavily oiled areas has not increased, possibly due to poor survival or low rates of emigration and continuing affects of the spill (Bodkin et al., 2002).

5.4.3.2. Concluding comments

It has been apparent since early studies of oil and petroleum toxicity that although low in acute toxicity, crude oil and refined products can cause a number of sub-lethal effects when ingested, inhaled, or absorbed. In general, this toxicity is greatly reduced after the oil becomes weathered. Various hydrocarbon mixtures have been used as herbicides and insecticides because of the highly toxic nature of some hydrocarbon fractions when applied directly. The key in understanding the potential for lethal and sub-lethal effects in the Arctic is in determining the level of exposure to these compounds at present and in the future. As oil and gas activity increases, the likelihood of exposure through spills and contaminated sites increases, leading to increased chances for effects in exposed populations.

Except for seabird toxicology and research on otters and mink at the *Exxon Valdez* spill, very little research has been conducted on the potential impact of petroleum-contaminated sites and releases from oil and gas facilities in the Arctic. Many studies on small mammals around contaminated sites in temperate regions show sub-lethal responses to petroleum contamination in exposed populations. These responses include changes in chromosomal material, histology, and the immune system, and are similar to those in otters and mink after the *Exxon Valdez* spill. Comparable studies should be conducted with small mammals in Arctic settings to determine the importance of these responses in northern populations. There do not appear to be any studies on the responses of large mammals to petroleum exposure which could be used to predict the sub-lethal effects from oil and gas releases. A number of studies demonstrate effects in beef cattle exposed to emissions from the oil and gas industry in western Canada, and these animals might provide good models of exposure for northern ruminants. This information is vital to understanding the potential effects of oil and brine spills to the terrestrial environment.

5.4.4. Physical disturbance and recovery of the terrestrial environment

5.4.4.1. Infrastructure

One of the marked changes owing to oil and gas activity in the Arctic is the placement of industrial infrastructure in an isolated and often pristine environment. The location, extraction, and transportation of oil and gas and their products results in many changes to the land, with the construction of roads, airstrips, gravel pads, pipelines, drill rigs, and, especially in Russia, the expansion of communities to accommodate additional workers. Past practices involved greater surface occupancy by such structures, whereas the use of modern exploration and development practices has greatly reduced many such disturbances in areas such as Alaska (see Chapter 2.5). Nonetheless, potential impacts caused by pipelines, summarised in Table 5.14 for the Trans-Alaska Pipeline system, include a wide range of

changes to the physical and biological environment (Maki, 1992; Englehardt, 1994). The short- and long-term effects of these changes have been studied in small, focused studies although recent assessments have taken a more comprehensive approach to understanding potential effects on the terrestrial system (NAS, 2004). Some of these studies have also conducted a retrospective analysis of the impacts on the environment which have occurred through the life of an oil field (Walker et al., 1987a,b; Walker and Walker, 1991; BLM, 2002).

Several recent reviews have documented the types of physical impacts on the terrestrial environment in the Arctic from oil and gas activities in Alaska (NAS, 2004), Canada (Forbes, 2004) and Russia (Khitun, 1997; Forbes and Sumina, 1999; Forbes et al., 2001). Development for the extraction of oil and gas in the Arctic may require extensive support systems of roads, airstrips, and camps and communities to house workers, in addition to the gravel pads, sumps, pipelines, and production facilities and ancillary structures needed to extract and transport oil and gas. Each of these structures causes changes to the landscape from direct disturbance and possibly also to the surrounding area due to factors such as dust, noise or leakage of materials. A clear example of the expanding nature of developing oil and gas fields is in northern Alaska which has shown continuous growth since the resource was first developed in the 1960s (Figure 5.15). The Figure shows the increase in drilling activities as the oil and gas fields are developed as well as major features such as the Trans-Alaska Pipeline and supporting infrastructure. New technologies and practices are less damaging to the environment than older technologies (NAS, 2004; see also Chapter 2.5), but the slow recovery of the terrestrial ecosystem from the cumulative impacts of early and recent developments means that the 'footprint' of oil and gas development in the terrestrial system is growing, albeit more slowly than in the past, and will continue to expand in the future. Oilfield practices have improved in all countries across the Arctic, but whether they meet the same standards or are applied and enforced uniformly is unknown in some areas.

Northern terrestrial ecosystems have many biological and physical characteristics which make them susceptible to impacts from oil and gas development, as well as from other human activities. The extreme climate in the Arctic, a short growing season, and low levels of precipitation and nutrient availability have resulted in a unique combination of species and growth forms that can survive under the harsh conditions. Several northern mammal and bird species migrate over long distances and spend a relatively short, but critical, time in the Arctic for breeding. These features require special consideration when assessing changes caused by oil and gas activities in the north.

The extraction and transportation of oil and gas in the terrestrial system require construction on the Arctic landscape; previously, this was extensive. The oil and gas fields are located and defined using exploration drilling and a number of seismic techniques. Older techniques and practices caused considerable damage to the terrestrial environment, but modern techniques and practices are more effective and cause less damage to the tundra. Cumulative impact studies, such as those in Alaska (NAS, 2004) and Russia (Forbes, 1995, 1997; Khitun, 1997), have shown that the major impacts on the environment during oil and gas exploration and operations tend to be from construction projects such as production facilities, pipelines, roads, and gravel pads. New technologies, operating procedures and

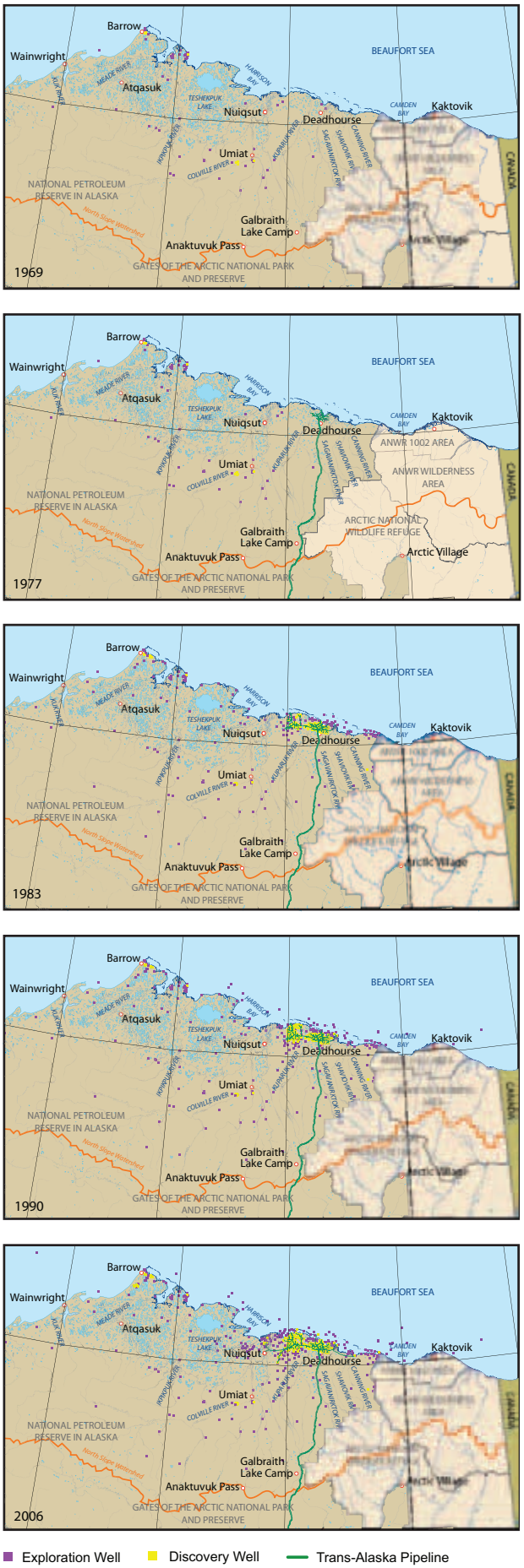

Figure 5.15. North Slope of Alaska showing the expansion of oil and gas operations between 1969 and 2006.

environmental regulations cause less damage in modern projects, but there have been varying degrees of recovery from earlier projects, often taking decades. The effects of these new technologies need to be validated with ongoing research into the effects of current practices.

Khitun (1997) provided a detailed account of effects on the natural landscape caused by the rapid development of the *Yamburg*, *Bovanenkovo*, and *Kharasavei* gas fields and the *Novy-Port* gas-oil field in the tundra regions of the Yamal and Tazovsky Peninsulas in western Siberia. Vegetation in the region comprises moss-lichen tundra species and dwarf shrubs, with wetlands covered by sedges and mosses. In addition to activities associated with the recent oil and gas exploration and development, the area serves as the homeland for several thousand indigenous Nenets, who hunt, fish, and herd reindeer (see Chapter 3). Oil and gas development resulted in degradation of the soil-vegetation cover to 100% in some areas (Khitun, 1997). Disturbance associated with exploratory drilling around wells extended 200 m in diameter. The areas were covered with drilling mud, drums, wood, pipes, and other debris. Surface soils in the area also had higher levels of metals such as barium, strontium, lead, zinc, and tin. Alkaline clay in the drilling muds changed the acidity of the upper soil layer from a pH of 4.5 (acidic) to a pH of 7.0–8.0 (neutral).

Changes to the terrestrial system have also been reported in Alaska and Canada (Walker et al., 1987b; Forbes et al., 2001). Documented effects on Alaska's wetlands include flooding, caused by the disruption of normal flow patterns of surface waters during the freeze-thaw cycle, and thermokarst, a localized thawing of the frozen ground (Walker et al., 1987b). Similar patterns were documented by Forbes (1993) in Canada. Although vehicle transport over unfrozen tundra is no longer practiced in areas such as Alaska, even a single pass by a tracked vehicle when the ground is not frozen can have direct and cumulative impacts that last for decades. In addition to effects on vegetation and permafrost soils, vehicle tracks can substantially affect populations of soil fauna (Kevan et al., 1995). Raised road and gravel-bed construction help to prevent permafrost erosion but impede the natural flow of surface waters. The presence of roads often results in local damage due to road dust (Walker and Everett, 1987; Forbes, 1995), flooding, and thermokarst due to warming effects.

Studies on the development of the North Slope report the construction of hundreds of kilometres of roads and areas covered by gravel or flooded between 1968 and 1983 (Walker and Walker, 1991) (see Chapter 2). Disturbance from flooding, roads, gravel pads, and thermokarst increased over the 15 years prior to the analysis. The authors concluded that the total area covered by the direct and indirect impacts of the development greatly exceeded the physical footprint of the development. Walker and Everett (1987) also reviewed the impact of road dust from two major roads in northern Alaska and reported significant ecological effects near the roads from dustfall. Subsequent studies have shown changes to soil chemistry and the plant communities adjacent to the roads due to road dust (Auerbach et al., 1997; Myers-Smith et al., 2006). The estimates of disturbed areas were updated by the National Academy of Sciences (NAS, 2004) to show that, by 2001, several hundred km of roads, peat roads, and pipelines had been constructed (see Chapter 2). In total over 7000 ha were covered by infrastructure in 2001, compared with 6700 ha in 1988 and 142 ha in 1968 (Figure 5.15). The number of gravel mines remained almost constant over the last 20 years, although the amount of gravel extracted declined considerably (NAS, 2004).

Impacts from these structures range from loss of habitat caused by physical disturbance to more subtle changes in the physical and chemical nature of the environment. Camp construction and drilling that started in the 1940s resulted in the trampling and killing of the vegetative cover, removal of the vegetative or organic mat, and removal of the vegetation and soil at several sites (Lawson, 1986). These impacts resulted in changes to the active layer and permafrost, the extent of which depended on the ground ice volume and material properties during thaws. In areas where meltwater could drain erosion occurred, and some areas required 30 years to attain suitably stable conditions for regrowth of vegetation (Lawson, 1986). Impacts from roads include changes in soil pH, changes in the concentrations of some nutrients, lower levels of soil moisture, and deeper active-layer thaw due to the spreading of road dust and changes in surface albedo (Walker and Everett, 1987; Forbes, 1995; Auerbach et al., 1997; Myers-Smith et al., 2006; Walker et al., 2006). Changes to the physical and chemical environment near the road resulted in significant changes in the plant community, with much higher proportions of gramminoid species and fewer mosses after a number of years (Myers-Smith et al., 2006).

Concentrations of certain elements may also increase in surface soils and lichens due to deposition from gas flaring and emissions from oil and gas facilities (Walker et al., 2006). The extent of changes from the construction of a single structure were reported by Hinkel and Hurd (2006), who measured the changes produced by the construction of a 2.2 km-long, 4 m high snow fence by the local community near Barrow Alaska. The accumulation of snow behind the fence resulted in soil temperatures 2–14 ºC higher in winter than at control sites due to the insulating properties of the snow. The drift thawed four to eight weeks after the thaw at the control site, leading to cooler summer soil temperatures. The net effect was higher soil temperatures behind the fence after six years, erosion of the upper permafrost, thaw subsidence, and widespread thermokarst where the meltwater ponded (Hinkel and Hurd, 2006). Clearly, the presence of infrastructure associated with the oil and gas industry can have impacts much greater than the physical footprint of the structure.

A review of the extensive changes from oilfield development over 25 years in northern Alaska (Walker et al., 1987a) summarised the physical disturbance to the terrain, and the beginning of recovery in some areas over the ten years prior to the study. Disturbance was defined as a factor that displaces the natural system beyond its normal limits of variation. The three major sources of anthropogenic disturbance are off-road transportation, permanent and semi-permanent facilities, and oil and brine spills. For many types of disturbance there are natural analogues for which natural recovery is possible; disturbances range from those having little long-term effect (such as slight alteration to vegetation and then complete recovery) to severe scarring (where recovery is not possible without major intervention) (Figure 5.16). In areas with significant disturbance, revegetation of native plants with fertilization aids recovery, but may require continued treatment to maintain ground cover. The authors did not report how much of the system had recovered over the 25-year period, or estimate how long it would take for the area to recover to its natural state.

The impacts reported by Khitun (1997) from oil and gas construction activities in Russia, and the rates and levels of

recovery of the plant communities, were typical of long-term damage to the active layer and plant communities from physical disturbance. Recovery of affected areas of tundra was very slow and variable, depending on soil moisture, the extent of original damage, and plant species present. The dynamics of these regeneration processes across the circumpolar Arctic were reviewed by Forbes et al. (2001). The most extensive damage was caused by off-road movement near camps and settlements that damaged the uppermost active layer and increased thawing. In dry soils, one pass of a vehicle did not necessarily cause major damage other than loss of some moss and shrubs, but several passes over the same track did cause more extensive damage to the plant community and soil compaction. In moist, saturated soils, single passes of a vehicle could cause major long-term damage to soils and plants. However, these moist areas showed relatively quick recovery with new vegetation after several years, although the original willow and grass species were replaced by sedges. Plant communities dominated by dwarf birch, willow, and sedge were eliminated after six to seven passes of a vehicle, and only 10% of the track had recovered after five years. Revegetation occurred from seeds and spores from original plant species, and undamaged rhizomes from the margins of the impacted area. Although damage is reduced during winter travel, changes in the plant community are evident several years after seismic exploration in winter, particularly in moist sedge-shrub tundra (Felix et al., 1989; Forbes et al., 2001).

The results of the Russian study have also been supported by studies from other regions of the Arctic. The movement of vehicles across the tundra during the early years of oil and gas development has caused significant damage to the tundra that is still evident several years later. Rickard and Brown (1974) and Forbes (1992, 1993) and Forbes and Jeffries (1999) outlined several examples in Alaska and northern Canada where trails over 20 years old were devoid of vegetation and still showed significant erosion of surface soils. Although the practice is now abandoned by most petroleum companies and banned by regulations, impacts on the tundra from off-road vehicles range from the crushing and shearing of woody vegetation during winter to complete destruction of the plant layer followed by subsidence and erosion of the soil surface (Forbes et al., 2001).

Jorgenson and Joyce (1994) reviewed the types of land disturbance caused by oil development in Alaska and suggested several strategies to rehabilitate land for wildlife use. The strategies included: flooding of gravel mine sites for fish habitat; the creation of wetlands in ponds; revegetation of gravel fill and overburden; removal of gravel fill to help restore wet tundra habitats; restoration of tundra

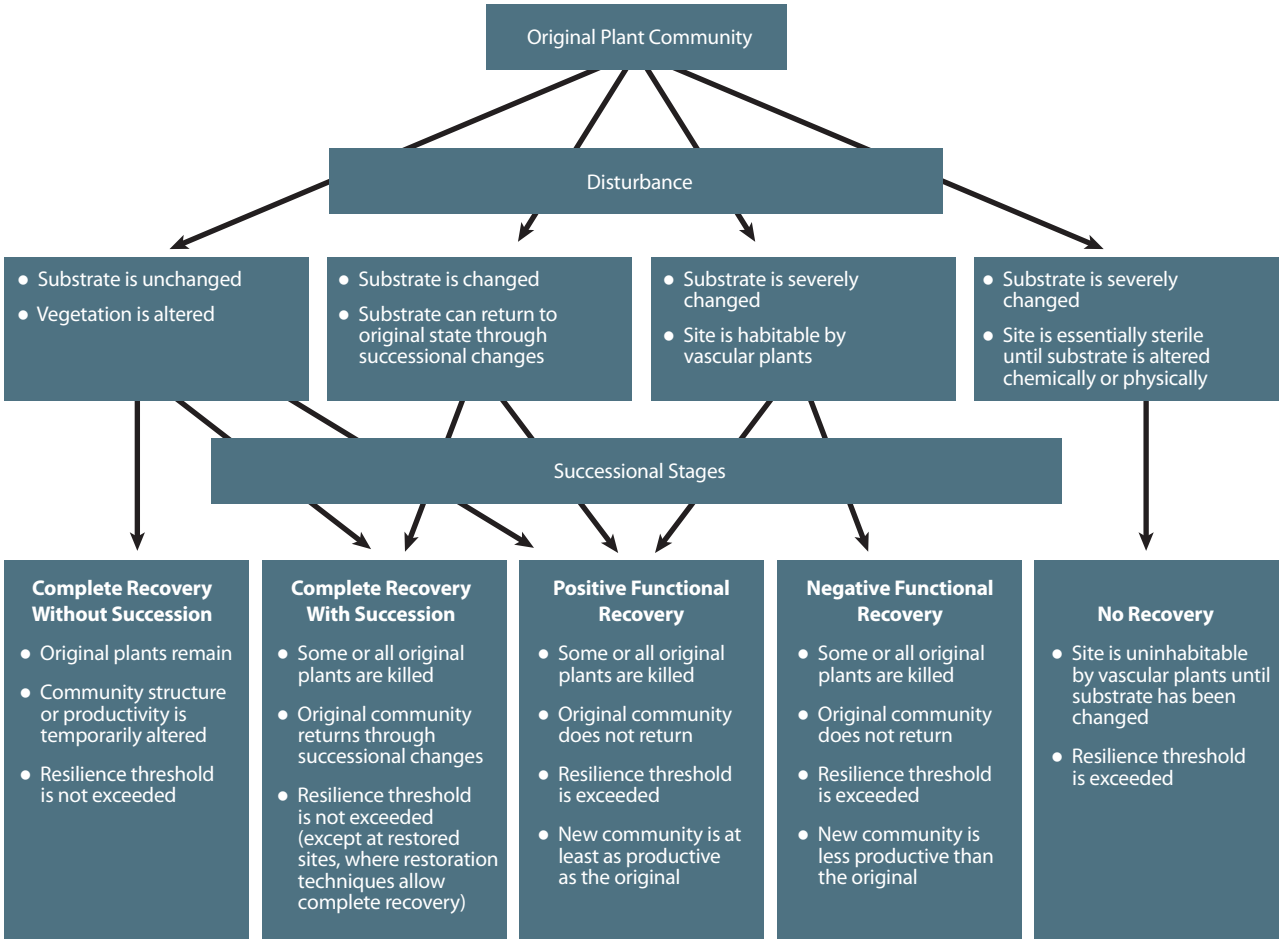

Figure 5.16. Flow chart describing the recovery of an Arctic vegetative community after natural or anthropogenic disturbance causes total or partial destruction of the plant community. Resilience is defined as the ability of the community to return to its natural state. The levels of recovery include one in which the site is uninhabitable by any plant species, as might occur from severe scarring, or scraping of the surface organic layer (Walker et al., 1987a).

on less severely modified habitats; and remediation of oil spills, seawater spills, and drilling muds.

The recovery of disturbed areas is generally very slow. The objective of revegetation studies has been to provide insulation, to slow or arrest erosion in disturbed areas, and to enhance wildlife habitat. Except in extremely wet habitats with few species (Forbes and Jeffries, 1999), actual restoration of the original vegetation has proven difficult to achieve. Findings from around the circumpolar North have recently been reviewed (Forbes and Jeffries, 1999). The most common method used for large totally or partially denuded areas is to seed agronomic varieties of grasses supplemented initially by organic matter and nutrients (nitrogen, phosphorus, potassium). Sites with little or no organic matter, such as abandoned gravel pads and sand and gravel quarries, are among the most difficult to revegetate. These sites tend to be low in nutrients and have a limited capacity to hold moisture. In Alaska, a relatively recent practice is to remove most of the gravel from abandoned pads so that seeded and naturally-invading plants can root more easily in the soils beneath. Flooded areas are also challenging since they cannot be seeded and must be planted with vegetative cuttings of wetland graminoids, which is extremely laborious.

In western Siberia, the prevailing sandy substrates of disturbed areas are not only low in nutrients and moisture-retaining capacities compared to undisturbed tundra, but are also prone to wind and water erosion. Without significant and sustained inputs of nutrients and moisture, most seeded grasses often die within one or two years. In addition to grasses, bundles of cut willow branches are inserted into sandy slopes. These reproduce vegetatively but few branches can survive beyond three years. Compared to Alaska and Canada, the amount of terrain damaged during oil and gas exploration and production is significantly greater in northern Russia (Forbes, 2004). Throughout the Arctic, the often labour-intensive methods coupled with difficulties in obtaining seeds for suitable agronomic species means that assisted recovery and revegetation are generally expensive, especially for large areas. Lax regulation and enforcement also mean that rehabilitation efforts in Russia generally lag far behind those in Canada and Alaska.

5.4.4.2. Subsidence

Subsidence of land surface due to the extraction of subsurface fluids has been cited as a major environmental consequence of the extraction of oil, gas, and formation water around some producing fields, particularly those with comparatively shallow reservoirs. Subsidence occurs as the pressure within the formation decreases as fluids or gas are pumped out and compaction of the reservoir occurs. Examples of compaction have been reported from the North Sea, Venezuela, Japan, and the Netherlands, where subsidence of the *Groningen* gas field is a major concern owing to the dependence of the country on dykes and subsidence of the ground surface by several centimetres could have major consequences (for reviews see Nagel, 2001; Humphries, 2001; Holzer and Galloway, 2005). Subsidence of operating oil and gas fields is also a major concern for the industry because sinking of the land surface might affect structures such as drill rigs and pipelines where tolerances are critical. Subsidence of up to 8 m has occurred in the *Ecofisk* field in the North Sea, which cost several billion dollars to address (Nagel, 2001).

The ecological implications of subsidence and flooding for wetland communities in the Arctic are considerable. Land subsidence, hydrological changes, and oil spills have resulted in significant losses to wetlands in the Mississippi Delta (Ko and Day, 2004), where subsidence rates may be as high as 23 mm/yr. Erwin et al. (2006) documented the effects of rising sea levels on wetland plant and bird communities on the eastern U.S. coast. Submergence of tidal feeding areas and onshore nesting areas will reduce the habitat of shorebirds and waterfowl, although inundation will provide productive feeding areas for some species of open-water feeding ducks and geese (Erwin et al., 2006). Over the long term, the net effects are expected to be negative for breeding waterfowl (Galbraith et al., 2002).

In northern Canada, extraction of gas from the *Taglu* and *Niglintgak* gas fields in the Mackenzie Delta is predicted to cause the flooding of approximately 250 ha over 30 years due to natural and induced subsidence in the Kendall Island Bird Sanctuary (Natural Resources Canada, 2006), which is one of the most important nesting and staging areas for waterfowl in North America. There is a concern that subsidence will lead to flooded nesting habitat and that the total flooded area will have a substantial impact on the sanctuary (Environment Canada, 2006). The impact of the subsidence is difficult to assess because of uncertainty in estimates of the extent of subsidence or effects on nesting waterfowl. This is particularly important in the Arctic where sea levels are projected to change significantly due to climate change over the next few decades.

5.4.4.3. Oil spills

Although rare, large oil spills on the terrestrial landscape have significant impacts on the plant community and soils. Small spills, which generally consist of several litres of crude oil or petroleum product, are far more common than large spills, although there is a general perception that large spills are frequent. Data on oil spills in the Arctic are summarised in Chapter 2 of this assessment.

Very small spills of crude oil and brine occur on a regular basis during field operations and from pipelines. An analysis of spill data in Alaska by Maxim and Niebo (2001) reported a total of 10 600 spills over the 25-year operation of the Trans-Alaska Pipeline, at an average rate of 480 spills/yr, for a total of 327 100 bbl (5.2×10^7 L) of crude oil or products. This total value is dominated by two events: the *Exxon Valdez* oil spill, which accounted for 257 100 bbl (4.1×10^7 L) and the Steel Creek sabotage spill of approximately 16 000 bbl (2.5×10^6 L). By comparison, the most recent large spill in Prudhoe Bay, Alaska occurred in March 2006, when 4800 bbl (7.6×10^5 L) of crude oil leaked from a corroded transit line over 0.8 ha of tundra near a caribou crossing. Excluding the *Exxon Valdez* spill, approximately 5.48 bbl was spilled per million bbl transported through the pipeline. For the Trans-Alaska Pipeline itself, approximately 88% of the crude spills, and 96.3% of product spills were less than 2 bbl (380 L). Conn et al. (2001) analysed a subset of these data and reported that most of the spills (82%) occurred during periods of snow cover, with only 18% occurring during the summer when movement of the oil was expected to be greatest. Large spills (>56 700 L) only comprised 4% of the total number of spills, and accounted for 70% of the volume spilled. Spills of petroleum products occurred from leaking tanks and drums, motor-oil leakage, vehicle accidents and ruptures in hoses, while crude oil was released from well blowouts

and leaks in the pipeline. About 53% of the spills involved crude oil, 40% refined petroleum, and 7% saline water or brine used to add pressure to the pipeline. Most of the spills (75%) were to wet tundra, while 9.7 and 3.7% were to moist or dry tundra, respectively. These statistics show that spills are on ongoing risk with pipelines, regardless of how well they are maintained.

The largest known spill associated with terrestrial pipelines is probably the Komi, or Usinsk, crude oil spill in eastern Russia in 1994 (see Chapter 4). Few scientific studies have been published on the ecological impact of the spill but descriptions of the spill indicate that the land area and inland waterways covered with oil were extensive. The size of the terrestrial system affected ranged initially from 68 to 2110 ha of streams, bog, and marshland as the oil dispersed with the spring melt (Devenis, 1995). Zoltai and Kershaw (1995) estimated that the spill consisted of several small spills over several months. Although there has been concern about the terrestrial community in the area, more concern was expressed for the fishery resources in the local rivers and the Pechora River, which is an important breeding ground for birds and fish. The spill is considered to be among the largest in history and the total volume was several times larger than the *Exxon Valdez* spill. No published information on the ecological effects of this spill, any recovery of the system or monitoring studies on changes in the toxicity of the remaining oil residues were available for this assessment.

5.4.4.4. Concluding comments

Studies to date suggest that physical impacts appear to cause the major changes to the Arctic terrestrial environment from oil and gas activity (see section 5.4.6 for cumulative effects). Early exploration and development resulted in damage to the tundra which is still evident in places. Revegetation strategies appear to help recovery, and have been successful in some areas, but may require much effort to maintain the areas and may result in the introduction of non-native species.

In some areas of the Arctic, oil companies have developed new methods to reduce the impact of oil- and gas-field practices. Conducting seismic, exploration drilling, and construction work in the winter, reducing the size of production pads for wells, eliminating mudpits, reducing or eliminating sumps, and restricting travel of personnel have reduced the amount of change caused by each project. However, improved practices are not consistent across the Arctic and damage from development continues in some areas. Also, as oil and gas activity increases in many Arctic areas, the cumulative impacts of the physical changes will continue to increase, albeit at a slower rate, with new oilfield practices (see section 5.4.6).

More research is needed on issues such as the effects of subsidence throughout the Arctic before their significance can be assessed. There is no question that phenomena such as subsidence and physical disturbance can affect large areas of the Arctic that are important habitats for avian and mammalian wildlife, but there are large uncertainties as to the amount of change occurring due to factors such as natural variation, climate change, the expansion of villages, and those factors associated with a specific type of activity, such as oil and gas development. These issues may become increasingly important as Arctic oil and gas activity expands over the next decade.

5.4.5. Effects on vertebrate populations

5.4.5.1. Mammals

Research into the effects of oil and gas development on terrestrial birds and mammals has been active since environmental concerns began to surface in the 1960s. In some areas, oil and gas activities are the only major developmental activity, and changes in the health, distribution, and behaviour of individual species, or the biodiversity of the terrestrial community, are usually interpreted in terms of the effects of the ongoing development. However, the baseline research that is required to interpret the significance of changes observed after the onset of oil and gas activity is generally limited. The lack of baseline information has led to great uncertainty in the interpretation of potential oil and gas impacts and debate concerning the significance of the changes observed.

In some cases, an expression of interest to develop an oil or gas field has spawned a series of environmental studies over the proposed pipeline route. For example, the proposal for the Arctic Islands Pipeline Project in northern Canada in the 1970s resulted in significant research on marine mammals, seabirds and the Qaminirjuaq caribou herd (Parker, 1972; Dauphine, 1976) even though the project did not ultimately proceed. The proposed Mackenzie Gas Project in northern Canada is also generating a number of surveys and some new research. Accurate evaluation of the potential effects of development is vital for regulators and the public to understand the potential consequences of approving or expanding northern oil and gas projects. The expected increase in the development of Arctic oil and gas reserves, and the size of the resource to be developed, means ongoing research will be vital to that understanding.

5.4.5.1.1. Caribou/reindeer

Caribou/reindeer (*Rangifer tarandus* ssp.) remain one of the most studied wildlife species in the north. This is largely due to their keystone status in the terrestrial ecosystem but also because of their social and cultural importance to indigenous peoples. The species is distributed across the landmasses of the Arctic, with over 60 discreet herds recognized (see Chapter 6). In North America, calving grounds for barren-ground caribou are distributed across the northernmost areas of the mainland and overlie oil and gas interests in western Canada and Alaska (Russell et al., 2002). In Russia, several wild reindeer herds of several hundred thousand animals exist in oil-rich regions, such as the taiga region of Siberia, and showed major declines during the 1990s (Baskin, unpubl. MS). Understanding of the composition and geographical distribution of individual herds is continually evolving (Bergerud, 1978; Williams and Heard, 1986) even though the basic biology and ecology of caribou/reindeer has been studied in much detail (Bergerud, 1978; Russell et al., 1993). Detailed studies of individual herds have provided fundamental data (e.g., Parker, 1972; Dauphine, 1976) on the biology and ecology of the species.

There is also a wealth of local and traditional knowledge from indigenous peoples throughout the circumpolar north that adds to this base of knowledge (Parlee et al., 2005; Lyver and Lutsel K'e First Nation, 2005). Information from western science and traditional knowledge has been combined to help understand the natural history of the species, including the migratory routes, diet, reproductive cycle, and calving sites for individual herds. Understanding the natural variation in the ecology and biology of individual herds is critical

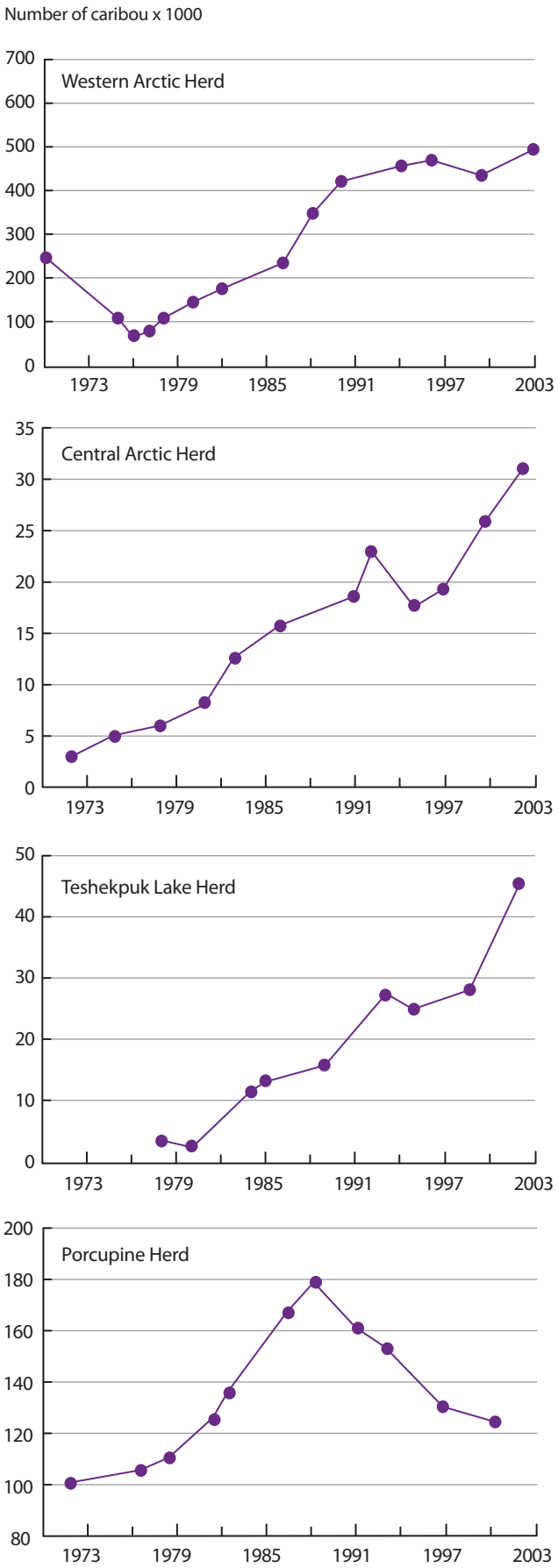

Figure 5.17. Population estimates for the four Alaskan barren-ground caribou herds whose migration route, wintering grounds or calving grounds are in the vicinity of the oil and gas fields on the North Slope of Alaska. Population sizes generally represent minimum herd sizes at the time of the survey (adapted from Murphy and Lawhead, 2000).

to predicting changes to the population that could occur from the development of oil and gas fields. Satellite tracking of radio-collared female caribou has added to real-time monitoring of herds and has provided more accurate census methods (Patterson et al., 2004).

Part of the difficulty in predicting large-scale changes to caribou/reindeer populations from local development is because the species undergoes significant natural population cycles (Syroechkovski, 1990). One of the more extreme examples occurred with the Forty Mile caribou herd in central Alaska, where the population fell from 500 000 animals in the 1930s to about 5000 animals in the mid-1970s (Valkenburg et al., 1994) and then recovered to about 45 000 in 2002 (Alaska Department of Fish and Game, 2003). The Western Arctic caribou herd, located in the Alaskan National Petroleum Reserve, increased from a seriously depleted condition in 1900 to 243 000 by 1970 and then declined to 75 000 by 1976. Similar population swings have been reported in the major wild and tame reindeer herds in Russia (Baskin unpubl. MS). Although these cycles can occur due to the natural causes of overpopulation and overuse of food sources (Crête and Huot, 1993; Murphy and Lawhead, 2000), predators and climatic factors (Griffith et al., 2002; Miller and Gunn, 2003), human-related causes, such as overhunting (Valkenburg et al., 1994; Baskin unpubl. MS) can also have a major effect on herd decline and recovery.

The greatest concentration of research in this area has been conducted on the barren-ground caribou herds on the North Slope of Alaska. Data from these studies show that the interactions between oilfield activities and infrastructure and barren-ground caribou range from beneficial (the use of gravel pads and roads to reduce insect harassment) to detrimental (avoidance of oil and gas activities), possibly to the point where a herd could be displaced from optimal calving habitat. The region provides habitat for four major barren-ground caribou herds; the Porcupine, Teshekpuk Lake, Western Arctic, and Central Arctic herds, three of which have increased in size since the mid-1970s (Figure 5.17). The fourth, the Porcupine herd, has not been directly exposed to oil industry development to date but has declined in numbers since the 1970s and is the focus of international scientific and public debate. The Porcupine herd is one of the most intensively studied caribou herds in the world (Roseneau and Stern, 1974; Russell et al., 1993; Fancy et al., 1994; Griffith et al., 2002) yet the reasons for the decline are still unclear. Research on the effects of current oil and gas activities on both individual animals and the health of the herd is being used to predict the effects of development on the Porcupine herd.

The expansion and decline of caribou/reindeer herds, whether in oil-rich regions or more remote areas, is a balance between biological and environmental factors. Caribou herds expand when recruitment, or the number of calves successfully entering the breeding population, exceeds the rate of mortality in the adult population. Recruitment of calves, a process which involves the birth of calves in late spring and sufficient growth of the calves over the summer-autumn to successfully overwinter, is vital to herd maintenance. For this reason, caribou show a high level of fidelity to specific calving grounds and return to the same region each year to provide optimal diet and favourable environmental conditions to maximize calf survival. In recent years, the use of radio-collared female caribou has shown the dispersal of the Porcupine herd over large tracts of land in Alaska, and northern Yukon and Northwest Territories over most of the year, with the return of the herd to the calving grounds in the Arctic National Wildlife Refuge in spring (Figure 5.18) (Roseneau and Stern, 1974; Griffith et al., 2002).

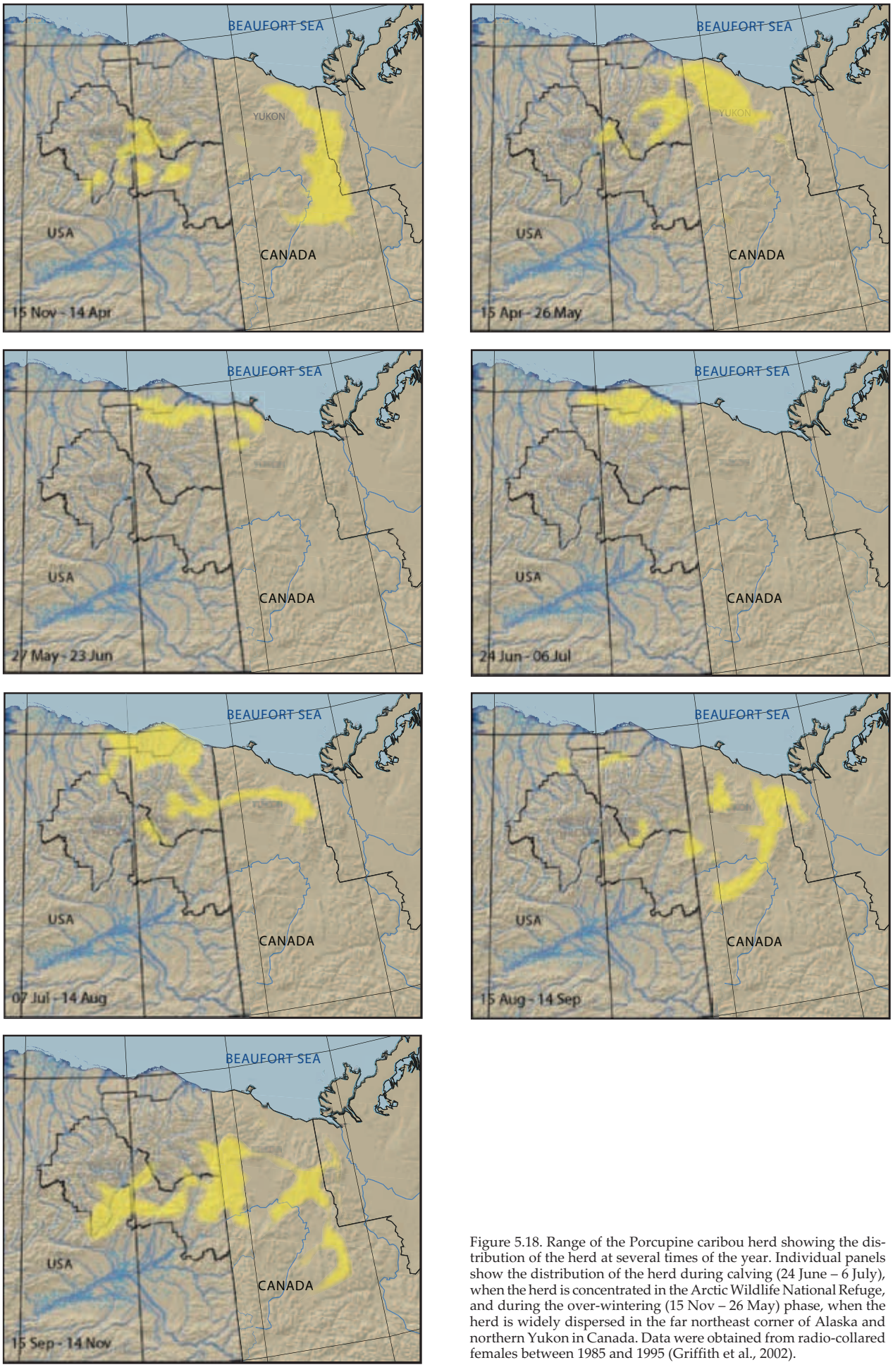

Figure 5.18. Range of the Porcupine caribou herd showing the distribution of the herd at several times of the year. Individual panels show the distribution of the herd during calving (24 June – 6 July), when the herd is concentrated in the Arctic Wildlife National Refuge, and during the over-wintering (15 Nov – 26 May) phase, when the herd is widely dispersed in the far northeast corner of Alaska and northern Yukon in Canada. Data were obtained from radio-collared females between 1985 and 1995 (Griffith et al., 2002).

Adult caribou generally die from disease, starvation, predation, and social interaction, while mortality in calves occurs primarily by predation (Miller and Broughton, 1974; Griffith et al., 2002). Typical predatory species are bears, wolves, and golden eagles. Other causes of mortality in calves include extreme wind chill, social interactions and abandonment (Miller et al., 1988), and birth defects. Predation on calves can result in losses of up to 80% of calves born in a year, while severe weather after birth can cause mortality in 50% of the calves due to high wind chill. In the herds near the oil field in northern Alaska, approximately 80% of females give birth each year and 60–90% of the calves survive the first month (Griffith et al., 2002). However, despite these high birth rates, the Porcupine herd has shown a significant decline (Figure 5.17). Within the four Alaskan herds, the Porcupine herd appears to have a long-term capacity for growth (~5%/yr) which is half that of the other three herds (Griffith et al., 2002).

Research on Alaska's Central Arctic herd suggests a direct link between body condition and nutritional status of the female and the condition of the birthing calves (McEwan and Whitehead, 1972; Cameron et al., 1993). A successful pregnancy is determined by the condition of the female in the previous autumn, with high fat stores and good body condition leading to calves with normal weight at birth. Calf survival in the first month is determined by the condition of the female during late pregnancy; poor female condition at the time of birth might result in the death of the calf the following winter. Events which place additional stress on calving females, such as poor quality diet (Crête and Huot, 1993), extreme weather (Murphy and Lawhead, 2000; Miller and Gunn, 2003), insect harassment, or harassment of the herd during calving, have a direct effect on the survival and success of the calves and the size of the herd (Cameron et al., 1993).

Caribou also require high quality food during lactation so that the females do not deplete their body reserves of fat and protein and the calves can grow quickly (Griffith et al., 2002). Low weight at birth and insufficient nutrition from the mother during lactation or from its diet before the winter due to a suboptimal diet can result in a high loss rate of calves. In addition, extremes in any of these factors can place a heavy biological burden on the female, who may not recover sufficiently for calving in the following year. Studies of caribou energy budgets suggest that females require from 70% (Fancy, 1983) to 90% (Murphy and Curatolo, 1987) of their time in activities related to feeding and digestion (lying, chewing) to maintain body condition. Significant deviation from this level of activity due to insect harassment, extreme weather events, or human-related activity can cause poor condition in the calving females and hence poor survival of the calves. Human-related activities that can affect the energy budget include harassment from vehicles or aircraft or the placement of roads and infrastructure in critical calving areas which cause females and calves to avoid optimal habitat (Vistnes and Nelleman, 2001).

Unpredictable, stochastic events can also affect herd condition. For example, heavy snows prevented the Porcupine herd from returning to its calving grounds in 2000-2001 and caused very high mortality in calves (Russell and McNeil, 2002). A warming climate is projected to increase the amount of snow in the Arctic, with increasing probability of deep snow affecting herd reproduction (McCarthy et al., 2001; ACIA, 2005). Miller and Broughton (1974) suggested that harassment of any herd by aircraft during the peak of calving would probably add significantly to the mortality of calves. Oil and gas activities which displace the herd from prime calving grounds or harass calving females will probably have similar effects on herd recruitment, however evidence for this type of deflection is very difficult to separate from natural variability and the influence of natural factors, such as annual variation in snowmelt (Haskell et al., 2006).

There is considerable debate regarding the degree to which development affects barren-ground caribou in the oil-producing areas of northern Alaska. Some reviews of the status of caribou herds in relation to the Trans-Alaska Pipeline and North Slope oil fields argue that the direct and indirect effects of development on the local caribou herds must be slight because three of the major herds in the area have increased markedly since the 1970s (Figure 5.17); the period of most intense oil development (Maki, 1992; Cronin et al., 1998). These authors suggest that, although individual caribou may be affected, the expanding herds are evidence that oil development and associated activities do not impact on the herds to a significant extent. Maki (1992) suggested that although 2% of the land surface has been altered in northern Alaska, the caribou herds are not currently restricted by suitable habitat and can accommodate the relatively small loss. Researchers studying the same caribou herd and using essentially the same long-term data set may reach very different conclusions on the distribution of the herd (Noel et al., 2004; Joly et al., 2006; Noel et al., 2006). Other research shows that the effects of oil and gas development extend beyond the immediate footprint of the infrastructure (Nellemann and Cameron, 1998). As stated by Cameron et al. (2002), "net growth of the Central Arctic herd is no better evidence of compatibility with development than a net decline would be evidence of a conflict". Also, industry photographs of caribou apparently co-habiting with oil facilities should not be taken as evidence that large-scale effects cannot occur. As noted by Forbes (1999), photographs showing reindeer herders near facilities in Russia probably do not reflect the true nature of the interaction as herders tend to avoid oil facilities owing to the amount of scrap metal, drums, glass, chemicals, and other wastes around the sites that can harm reindeer.

The key to understanding the extent to which caribou/reindeer may be affected by extensive development is in determining the specific mechanisms and pathways through which the effects may occur. Research has shown that the presence of human-related activity can cause changes in the behaviour, distribution, and activity patterns of caribou, particularly females with calves. Behavioural responses to oil and gas activity have been shown to be natural avoidance behaviours, such as those that may occur in the presence of predators or hunters. Initially, reductions in herd size with the development of roads was attributed to increased hunting pressure, however later studies showed that changes in behaviour may be more sensitive than previously thought. For example, in Europe, the distribution of reindeer was altered by the simple presence of humans (Aastrup, 2000), the presence of roads, power lines (Vistnes et al., 2004) and ski trails (Vistnes et al., 2001), and tourist resorts (Nellemann et al., 2000) and production-related facilities for oil and gas (Nellemann and Cameron, 1996). Measurements of responses of Peary caribou (*Rangifer tarandus pearyi*) to harassment by helicopter overflights on Banks Island in Canada showed that the number of extreme behavioural responses depended on several factors, including the number of calves

in the group, the previous activity of the animals, and terrain (Miller and Gunn, 1979).

Research conducted in the 1980s also showed that caribou were more active, and so had more energetic stress, within 600 m of a pipeline and road traffic running in parallel, than undisturbed caribou (Curatolo and Murphy, 1986; Murphy and Curatolo, 1987). Aastrup (2000) reported that groups of caribou in Greenland were more vigilant and avoided humans approaching on foot by fleeing sooner and running farther than individual caribou when calves were present in the herds. In Norway, Vistnes et al., (2004) used grazing intensity patterns in reindeer to show that power lines and roads reduced migration and caused a reduction in available range. This fragmentation occurred 30 years after the power lines were constructed suggesting that the reindeer did not habituate to the presence of the structures. Threatened woodland caribou (*Rangifer tarandus caribou*) were found to be largely unaffected by seismic lines, whereas roads with intermediate levels of use were considered to be semi-permeable barriers to caribou movements, as caribou crossed them about six times less frequently than little used roads (Dyer et al., 2002).

Evidence is accumulating that changes in the distribution and behaviour of caribou in Alaska are consistent with the outcome of European studies, albeit on a much larger scale. Avoidance of roads associated with the pipeline and Prudhoe Bay development area by females of the Central Arctic herd has been reported since the mid-1970s (Cameron et al., 1979; Whitten and Cameron, 1983). Nellemann and Cameron (1996) refined the avoidance estimates and suggested that maternal females were displaced by 4–10 km from surface developments. Nellemann and Cameron (1998) extended these studies and reported the following conclusions: females and calves are far more sensitive to surface development than adult males and yearlings; the greatest incremental impacts are attributable to initial construction of roads and related facilities; and the extent of avoidance greatly exceeds the physical 'footprint' of an oilfield complex. The result of this displacement is that calving females may be pushed into sub-optimal habitat which may affect the nutritional intake of females and calves and reduce the productivity of the herd (Nellemann and Cameron, 1996). Cameron et al., (2005) reported that the calving grounds of the Central Arctic herd have changed significantly near the Prudhoe Bay complex, possibly in response to oil and gas activity (Figure 5.19).

Griffith et al. (2002) noted that the Porcupine herd could be particularly susceptible to oil and gas development because of the lower capacity of the herd for growth relative to other herds on the North Slope. There has been a clear shift by females away from petroleum development infrastructure during the first few weeks after the birth of calves. The herd may also be affected by the lack of high quality alternative calving habitat when displaced from the Arctic National Wildlife Refuge region, and because of the strong link between calf survival, the quality of forage for females, and the increasing risk to calves from predators in the sub-optimal habitat (as reported for the Central Arctic herd; Cameron et al., 2002). The herd may also be displaced into areas with increased predation from grizzly bears (McCabe, 1994).

In contrast to studies reporting avoidance of oilfield infrastructure, studies have also shown that caribou use the open areas of developments, such as gravel pads and roads, to avoid insect harassment. Harassment by insects such as mosquitoes and oestrid warble and bot flies causes

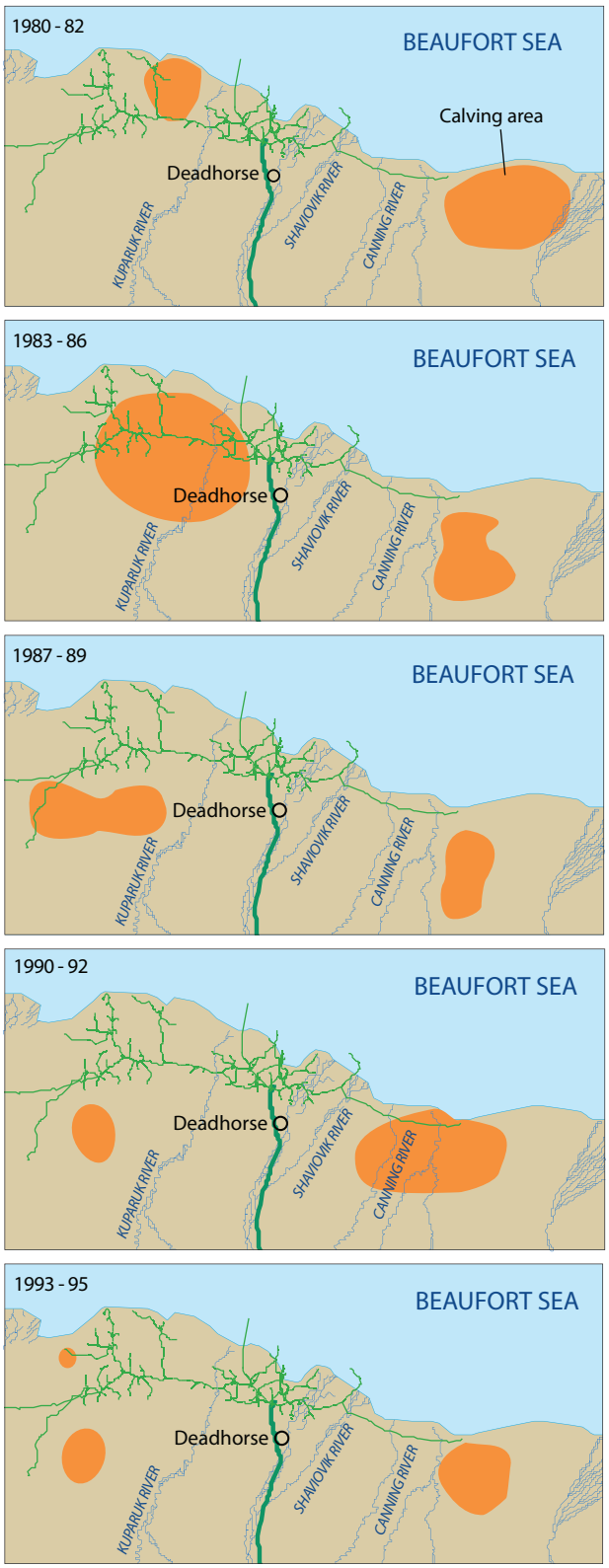

Figure 5.19. Changes in the calving areas of the Central Arctic herd between 1980 and 1995, a time of significant development of the Prudhoe Bay oil and gas infrastructure. Oilfield infrastructure is indicated as the bold green lines (Cameron et al., 2005, after Wolfe, 2000).

significant stress in caribou during the post-calving period (Pollard et al., 1996a). Caribou also move to wind-swept areas near the sea to reduce insect harassment. Lower ambient temperatures and higher wind velocities on the open pads and roads reduce the levels of insects harassing the caribou (Pollard et al., 1996b). Surveys of caribou have shown that members of the Central Arctic herd frequently

used the oil field gravel pads and roads for relief from insects and used the shade from oil field structures when oestrid flies were abundant (Pollard et al., 1996a). Time activity budgets from the active drilling sites show that caribou that are harassed by insects tend to be moving faster and lying down less than those not harassed suggesting that relief from the insects aids in maintaining animal condition (Fancy, 1983; Noel et al., 1998). At times, the need to escape the insects may override avoidance behaviour by the caribou (Noel et al., 1998). Another benefit of oil and gas-related activity is the presence of cutlines through forested areas that may assist in caribou migration (McCourt et al., 1974).

5.4.5.1.2. Grizzly or brown bear

Research on the effects of oil-related activities on other wildlife species is scarce, except in terms of endangered species or species that may interact with caribou/reindeer. For example, predators that could be attracted to the oil facilities also prey on local caribou and much of the research on the impacts of oil and gas development on grizzly bears (*Ursus arctos*) concerns predation on barren-ground caribou, particularly when a major herd becomes concentrated in sub-optimal calving areas.

Because of its need for large, contiguous undisturbed areas for home ranges, particularly in the far north where productivity is low and for its tendency to be attracted to waste sites around northern developments, the grizzly bear, or brown bear (*Ursus arctos*), is one of the species most susceptible to changes caused by oil development. Although the bears may live for up to 25 to 30 years, females do not reproduce until they are about five years old, and breed every second or third year. Hence females only have about two cubs in their first ten years. Servheen et al. (1999) estimated that a female living to 25 years will only add an average of 3.5 females to the population, giving a very low natural rate of increase. The barren-ground grizzly bears which inhabit much of the Northwest Territories and the North Slope of Alaska, are particularly susceptible to disturbance because of the late age of maturity (Servheen et al., 1999), small litter size, and the very large home ranges.

The species is distributed across the holarctic, with populations in Canada, Alaska, and 42 Eurasian countries, although many of the Eurasian populations are remnant populations and considered endangered (Ross, 2002). In Canada, recent estimates place the population at about 29 000 grizzly bears, with about 5000 in the Northwest Territories, a region with expanding interest in oil and gas. The population in Alaska is a similar size with the majority in the southernmost regions of the state. The central and northern regions of the state that contain the North Slope oil fields and the Trans-Alaska Pipeline have a relatively low density of bears (Servheen et al., 1999) and comprise about 40% of the state's bear population. In Alaska, around 1000 of a population of 30 000 bears are legally hunted each year (Servheen et al., 1999). In Canada, 22 bears were killed annually in the Northwest Territories from a population of about 5000 bears, while 92 of 6000 bears were killed in the Yukon.

Research into grizzly bear biology and distribution in relation to oil and gas activities has been reviewed for northern Alaska by Shideler and Hechtel (2000), although the study focused on the role of the bear as a primary predator of caribou on the Alaskan North Slope. The large range of some bears may result from them following migrating caribou herds (Young et al., 2002; McLoughlin et al., 2003). The review reported that the home range of oilfield grizzly bears are about five times greater than those in the south and about twice as large as those of bears in southern Alaska. Sixty to 70 bears inhabit the oil field region, with around 21% feeding on waste and dump sites at the work sites. New policies by the oil companies may lead to dismissal for those who feed bears, which may reduce the presence of bears at these sites. Follman and Hechtel (1990) documented the interactions between black bears and grizzly bears during construction of the Trans-Alaska Pipeline. During the nine years of pipeline construction 192 official bear problems were reported, most relating to bears in camps or dumps. No bear-related injuries were reported, even though the authors documented several examples of feeding of bears by workers, some by hand. The current levels of bear and oil and gas interactions (Shideler and Hechtel, 2000) indicate that bears are present in the vicinity of oil and gas infrastructure. As oil and gas development increases in the north, conflicts with bears around camps and developments will continue to increase and the number of 'nuisance' bears that will be killed to maintain worker safety will increase.

5.4.5.1.3. Arctic fox

The Arctic fox is one of the Arctic species that appears to benefit from the presence of oil and gas developments. The species is highly specialized for life in the extreme Arctic climate with a physiology and natural history that allow it to reduce energy demands during winter and to live with little food for extended periods when food is scarce (Burgess, 2000). Their ubiquitous distribution through the northern tundra ensures that they are a common species around Arctic oil facilities. Because they find food at dumps and landfills and may den in the oil facilities, the Arctic fox is considered to be a 'subsidized' predator. This term is applied to those species that benefit from the presence of the developments but which will continue to hunt to the extent that local bird and mammal populations may become suppressed.

Arctic foxes were studied on the Alaskan North Slope because of concerns about attacks on oil field personnel. Eberhardt et al. (1983) used radio collars to show that Arctic foxes on the North Slope dispersed during the year in a manner similar to other populations reported in the literature, and probably in relation to food availability. A 1973 study by Urquhart (cited by Eberhardt et al., 1983) on the presence of foxes near mobile seismic camps in northern Canada indicated that adults were present around the camps throughout the year but tended to avoid humans during the February-March breeding period, which reduced the number of foxes observed around facilities. Eberhardt et al. (1983) also reported a large number of foxes living on or near the developed sites at Prudhoe Bay. Foxes living at the Prudhoe Bay site were in much better physical condition than reference populations because of the availability of food at the dump. The study also reported that there were several attacks on oil field personnel who were subsequently treated for rabies. The rabies concern was significant enough that 99 foxes were culled from around the sites in 1994 to reduce the attacks (Ballard et al., 2000). Body size, condition, and age distribution of the harvested foxes were similar to other populations reported in the literature (Ballard et al., 2000). The studies suggested that Arctic foxes benefit from the presence of the camps, communities, and oil infrastruc-

ture (Burgess, 2000), however effects on local bird and small mammal populations are unclear.

5.4.5.1.4. Muskrat

The effects of seismic testing on muskrat (*Ondatra zibethicus*) populations were studied in the Mackenzie Delta in Canada during the 1970s (Westworth, 1977). The studies were conducted to determine if seismic tests caused direct injury and stress on individual muskrats, had significant effects on the muskrat population, or caused significant changes in their behaviour and abandonment of 'push-ups'; domes of vegetation covering a plunge-hole in the ice in which the muskrat feeds in winter (Banfield, 1974). Damage to the middle ear and lungs were observed in animals in earthen burrows within 6 m of the detonation and within 15 m in water. No effects on numbers or reproductive capability of the population in the test area were apparent, however the number of active push-ups was significantly higher at distances more than 180 m from the seismic lines, and there was a significant decline in visits to push-ups in the test region over a 24 hr period. The study concluded that major effects on the population were unlikely even though movement of the seismic equipment destroyed active burrows during seismic testing and that there were not enough data to test whether the behaviour of muskrats was affected by the blasting.

5.4.5.2. Concluding comments

These studies show clear evidence of changes to the distribution and composition of terrestrial species in the presence of human activity in the Arctic. These changes are expected to continue, and possibly accelerate, as oil and gas development increases. These effects in Arctic terrestrial populations and biological communities hold the greatest potential for significant long-term impacts. Many wildlife species have critical times during the year when the disruption of normal biological patterns may affect the overall status of the population. The evidence of changes to the distribution of caribou on the calving grounds in Alaska due to human activity is a significant observation which is consistent with other studies showing large-scale changes in distribution in response to human activity (e.g., snowmobiles, trucks, power lines). In recognition of these types of impacts, oil companies in some areas have reduced activity during times of the year when wildlife are most sensitive (i.e., during moulting, calving, nesting), but these practices are probably not universal across the Arctic and conflicts will continue to increase as oil and gas activity increases. It is also important to note that oil and gas activities are occurring as wildlife populations are responding to a changing climate and thus new types of conflict may arise. Further research and monitoring is needed in this area to improve the protection of species.

Oil spills, whether single large events or cumulative small events, have the potential to affect large areas of the terrestrial environment. The loss or fragmentation of habitat by oil or brine spills would affect a relatively small number of wildlife, but would add to the cumulative impacts of the activity. A repeat of a spill the size of the Komi spill would be catastrophic for a large area of the terrestrial system, and the damage would remain for a very long time. The performance of pipelines in the Arctic should continue to be monitored to ensure minimal impacts from spills.

5.4.6. Potential for cumulative effects

Environmental assessment methods have improved significantly since oil and gas reserves were first developed in the Arctic and much research has been conducted on the changes caused by the combined effects from all facets of the oil/gas extraction process. Most studies have examined small-scale changes to the physical and biological environment; however more effort is being put toward understanding the cumulative effects of all changes caused by development. These cumulative effects are the result of many small changes over time or space that add together in a linear or synergistic way to cause larger, unanticipated effects. For example, the effects from a single project are expected to be minor, however when several of the projects are added together the effects become significant. As summarised by the National Academy of Sciences, cumulative effects can be divided into several processes (Table, 5.15).

The clearest evidence for the cumulative effects from oil and gas activity comes from Alaska, where detailed retrospective studies have documented damage to the biotic and abiotic environment, primarily from the early years of development (Walker et al., 1987b; NAS, 2004). While oil and gas developments have occurred in other areas of the Arctic, for example Russia, published information documenting changes was not available for this assessment.

Changes have been made in oilfield practices to reduce ongoing damage and assist recovery at some sites. This is possible because of the relative maturity of the development in Alaska and the confined area of development. For example, in a series of studies, Walker and co-workers have documented some of the cumulative impacts of development in northern Alaska that could act as a model for the type of effects that oil and gas development may have in terrestrial systems in other areas of the Arctic. Early work by Walker et al. (1978) on the effects of oil spills on vegetation led to the development of oil sensitivity maps to estimate the impact of oil spills in areas of the Alaskan North Slope. Data were collected on the response of several species of native plants to oil and diesel exposure in wet and dry habitats. A sensitivity index was estimated from species distribution maps in potentially affected areas and the rate of recovery estimated.

Table 5.15. Summary of processes combining to cause cumulative effects (NAS, 2004).

Process	Description
Time crowding	Frequent and repeated effects on a single medium. For example releases of oil in an area that had not recovered from a previous spill
Space crowding	High density of effects on a single medium. For example, adding the number of roads or number of gravel pads in an area
Compounding effects	Synergistic effects from multiple sources, where the total effect is greater than the sum of the individual effects
Thresholds	Effects become quantitatively different once a threshold is reached
Nibbling	Progressive loss of habitat resulting from a sequence of activities, each of which has slight effects individually

Subsequent work documented the changes resulting from the development on the landscape of the Alaskan North Slope and some of the cumulative effects (Walker et al., 1987a,b). Cumulative effects in the wetlands include flooding caused by road and gravel pad construction. Since the review by Walker and co-workers, the size of the area affected by roads, gravel pads, and dams for example, has increased proportionate to the amount of development, although at a much slower rate (see Chapter 2). In one of the wetter areas of study, indirect effects (e.g., flooding, thermokarst, dust) were greater than direct effects (e.g., roads, gravel pads, excavations) while for other areas the reverse was the case. Major changes are also expected to increase at different rates with development. For example, as new access roads are constructed to service satellite fields, the area affected by roads may increase slowly, and the amount of dust generated by truck activity may continue to increase as truck traffic increases. The project was completed in 1984 and so summarised changes to the geobotanical landscape up until that point. Newer assessments show infrastructure growing more slowly as development of the oilfield matures.

This research has led to the organization of anthropogenic disturbances into spatial and temporal classes and the application of GIS (geographical information system) techniques and mapping of the landscape in the Arctic using high-resolution techniques. Walker and Walker (1991) categorized the types of disturbance caused by oil and gas development on a spatial scale and compared the size affected to natural disturbances (Table 5.16, Figure 5.20).

Disturbances to the landscape associated with oil and gas activity occurred primarily at the microscale level (Table 5.16), or at a site size level (10^{-1} to 10^{6} m^2), whereas the cumulative impact of the smaller effects occurred at the macroscale, or regional, level (10^{5} to 10^{8} m^2). Effects on the geobotanical landscape, from oil and gas activity, included those from small stressors such as waste and debris to dust and large gravel pads. The number of years for recovery were also assigned for each stressor and ranged up to 10^{4} years in many cases. Large gravel pads, gravel mines, buildings and structures, and transportation corridors are among the longest lasting changes (with the exception of climate change due to increasing levels of greenhouse gases). Effects of the same magnitude and time span result from natural processes (Figure 5.20, upper), but the major difference is that one set of changes is controllable by society while the other is not. Subsequent reviews of the effects of development largely support the type and extent of impact of this analysis and show the difficulty in reclaiming the disturbed areas (Forbes et al., 2001). An analysis of the impacts of road dust from the Dalton highway in Alaska has shown a marked change in plant species composition and soil chemistry within the road dust shadow which extends at least 200 m outside the physical imprint of the road (Myers-Smith et al., 2006). Similar findings were reported for Russia (Forbes, 1995).

A recent study (Johnson et al., 2005) used quantitative mathematical methods to predict the cumulative effects of development on four major wildlife species in the Slave geological province in the Canadian Northwest Territories. Although diamond mines and abandoned mineral mines are the major types of development, oil and gas activity is increasing. One of the species studied was the wolverine (*Gulo gulo*) which is listed under the category of 'Special Concern' by the Canadian Committee on the Status of Endangered Wildlife in Canada (COSEWIC, 2003) because of its sensitivity to human activities, particularly habitat loss. Data on vegetation composition, human disturbance, and interspecific interactions were included in a model of resource selection functions. The authors interpreted the response of the models as evidence that human presence and the associated infrastructure can disrupt wildlife movements and reduce the availability of high quality habitat for each of the studied species. Wolves and grizzly bears were predicted to be impacted throughout the year, while impacts to caribou were more seasonal. The authors emphasized that avoidance of high quality habitat does not necessarily translate into population, or even reproductive, effects, because of the ability of species to adapt to changing condi-

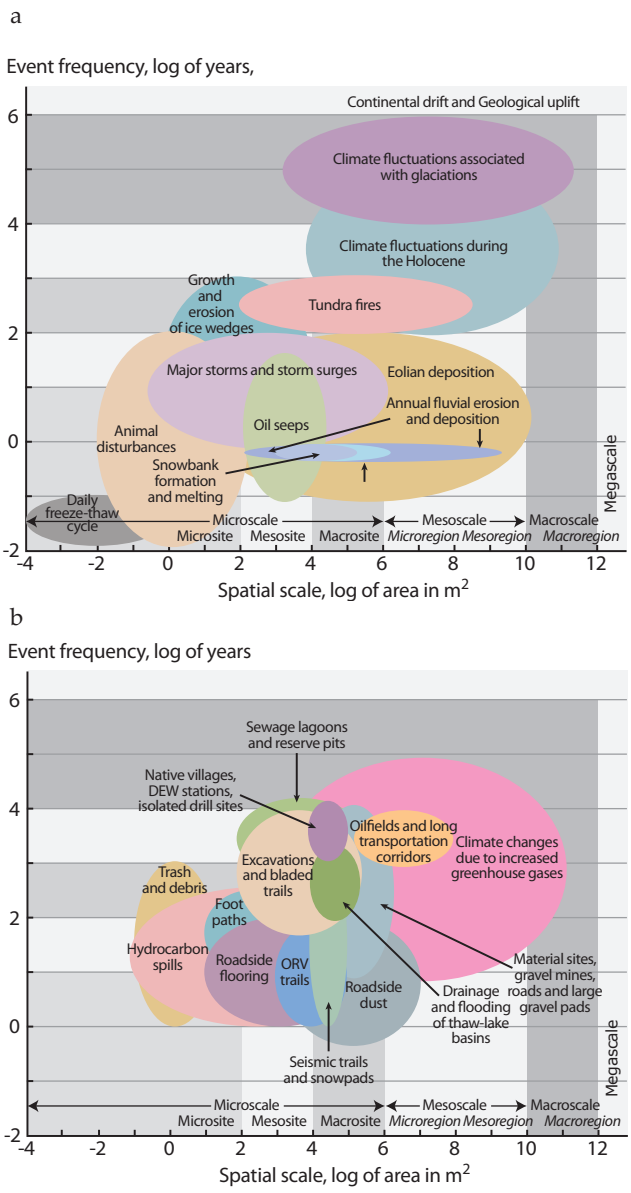

Figure 5.20. Plots showing (a) natural and (b) anthropogenic disturbances in terms of the frequency of the event (years) and the spatial scale, or size of the area affected, for a mature oilfield like the North Slope of Alaska. Large natural events include climate fluctuations, tundra fires and deposition from the atmosphere. The largest anthropogenic disturbance related to oil and gas activity is climate change due to greenhouse gas releases from product combustion. Smaller scale disturbances from oil and gas activities include road dust and infrastructure such as roads and gravel pits. In this analysis, terrestrial oil spills are relatively infrequent and have covered a small area in relation to other disturbances. (Walker and Walker, 1991).

Table 5.16. Anthropogenic disturbances associated with developments in the Arctic, but in particular oil and gas activity (Walker and Walker, 1991). Hierarchical domain terminology follows that of Delcourt and Delcourt (1988). Disturbances are shown graphically in Figure 5.20.

Hierarchical domain	Disturbance	Geomorphic effect	Vegetation effect	Spatial scale, m^2	Recovery time, yr
Macroscale (regional)	Climate change due to increased greenhouse gases	Alteration of permafrost table, colluviation of steep slopes, alteration of periglacial features, thaw lake cycle and coastal erosion	Species migration, ecotone displacement, changes in landscape mosaic, movement of treeline	10^2-10^{11}	10^3-10^5
	Air pollution, NO_x and sulphur emissions	None	Possible damage to lichens	?	?
Mesoscale (regional)	Oil fields, and long transportation corridors	Cumulative effects of many smaller disturbances (see below)	Introduction of weedy species, cumulative effects of many smaller disturbances (see below)	10^5-10^8	10^3-10^4
Microscale (site)					
Macrosite	Roadside dust	Thermokarst in very heavy dust areas	Smothering of vegetation, development of alkaline plant communities, toxic effects to acidophiles, addition of nutrients	10^5-10^7	10^0-10^2
	Roads and large gravel pads	Changed substrate and microclimate, numerous secondary effects due to dust, flooding, and snow adjacent to pads	Burial of natural vegetation, numerous secondary effects due to runoff from road, dust and flooding, burial from gravel due to snow removal, barren dry surface for vegetation	10^4-10^5	10^4-10^6
	Seismic trails and snow pads	Altered microtopography	Compaction or removal of vegetation	10^4-10^6	10^0-10^3
	Material sites and gravel mines	Creation of lakes, removal of near-surface sediments, creation of overburden piles	Removal of vegetation, creation of barren surface for primary succession, burial of tundra adjacent to mines by gravel and overburden piles	10^4-10^6	10^1-10^4
	Native villages, DEW stations, and isolated drill sites	Cumulative effects of many smaller disturbances (see below)	Cumulative effects of many smaller disturbances (see below)	10^4-10^5	10^3-10^4
	Excavations and bladed trails	Thermokarst, erosion, removal of near-surface sediment and vegetative mat, changed microtopography, creation of berms along trails	Compression, removal and burial of vegetation, changed endpoint for succession due to new microclimate and soil character	10^3-10^5	10^2-10^4
Mesosite	Sewage lagoons and reserve pits	Formation of ponds and mud flats	Killing of vegetation	10^3-10^5	10^3-10^4
	Hydrocarbon and brine spills	Changed microtopography due to cleanup activities, thermokarst due to changed thermal regime	Total or partial killing of original vegetation, coating of vegetation in spray spills	10^3-10^5	10^1-10^3
	Roadside flooding	Thermokarst, formation of anaerobic soils, formation of lakes	Drainage of flooding of wetlands, changed temperature and nutrient regimes	10^2-10^5	10^0-10^3
	Off-road vehicle trails	Changed microtopography due to compression	Compression, displacement, and removal of vegetation	10^3-10^5	10^0-10^2
	Structures, sod houses, buildings	Changed microtopography, thermokarst in surrounding areas	Removal of vegetation, eutrophication of vegetation	10^2-10^4	10^3-10^4
Microsite	Footpaths	Elimination of hummocks, erosion, thermokarst	Trampling, compaction and removal of vegetation	10^1-10^3	10^0-10^2
	Trash and debris	Local changes in microtopography	Killing of vegetation	10^1-10^2	10^0-10^3

tions (Johnson et al., 2005). However, it seems reasonable to assume that species that move into sub-optimal habitat will demonstrate changes to the population metrics, over the short- or long-term.

The U.S. National Academy of Sciences (NAS, 2004) assessed the cumulative effects of oil and gas development on the North Slope of Alaska in 2004 in order to provide an integrated, comprehensive assessment of the environmental research conducted to that point. The objective was to assess the known and probable cumulative impacts of oil and gas activities on the physical, biotic, and human environments of the North Slope and adjacent marine areas. Experts from physical and biological sciences, anthropology and economics reviewed the research and discussed ongoing issues with the Alaskan communities in the area.

The major conclusions from the assessment (Table 5.17) range from disturbance of soil and plant communities from the construction of roads, pipelines, and other infrastructure to changes in the biological community with the addition or diminution of species. The major contributors of changes to vegetation were roads, which are a source of dust, and which cause flooding, thermokarst, and roadside snow accumulation. Higher soil temperatures in summer caused permafrost erosion. A number of potential effects on caribou were noted (see also section 5.4.5.1.1). Seismic testing has not generally affected caribou of the Central Arctic herd because testing was done in the winter when the caribou were dispersed, however recent methods of seismic testing have increased the density of the lines. A number of potential effects on caribou are also noted, such as displacement of the herds from optimal habitat and loss of preferred habitat, and the possibility of decreasing productivity in herds most closely associated with oilfield complexes. No effects were observed in muskoxen, while the distribution of grizzly bears and other predators has been changed by the availability of food at dumps and landfills.

The United Nations Environment Programme has developed a program termed GLOBIO (Global Methodology for Mapping Human Impacts on the Biosphere) to estimate the cumulative effects of development in most regions of the world, including the polar regions (UNEP, 2001). The objective is to develop scientifically-sound methodology for mapping the risk of human impacts on the biosphere. The program used the results of published scientific studies to predict the effects of development on plants, birds, and mammals. The size of the area affected by the development is correlated to the size of the infrastructure.

After an extensive review of the literature concerning the effects of development on animal species (UNEP, 2001), the assessment used a model to estimate zones around development in which animal species would be affected by new development. The report cited the literature on the avoidance of caribou within 3–10 km of roads and power lines and the impact on large carnivores up to 0.6 km from roads, when roads reach a certain density. The data were used to estimate zones of influence on species around structures such as pipelines and roads. The zones of influence were areas which resulted in reduced abundance of birds (0–1 km) and large mammals (0–3 km) and cumulative effects on the biotic community shifts up to 20 km. The zones are summarised in Table 5.18. Detailed circumpolar maps were used to predict

Table 5.17. Summary of major changes caused by oil and gas activities in Alaska and the effects to the terrestrial system. (NAS, 2004).

Activity	Effects on the terrestrial system
Growth of industrial activity	• Single oil field expanded to industrial complex of developed field, roads, pipelines and power lines • Effects extend up to several kilometres beyond project footprint
Interactions of climate change and oil development	• Changes due to oil and gas activity difficult to isolate because of changes due to changing climate • Warming climate is expected to change the distribution of plants and animals and affect permafrost, similar to the possible changes from oil and gas activity
Damage to tundra from off-road travel	• Networks of seismic exploration have altered large areas of tundra • Effects include damaged vegetation and erosion • Newer improved seismic techniques cause less damage, but some damage still evident from camps, the large number of vehicles used and the higher spatial density of trails required for 3-D mapping of oil fields
Roads	• Far reaching effects of roads include dust, roadside flooding, thermokarst and roadside snow accumulation • Altered animal habitat and increased accessibility for hunters and tourists
Effects on animal populations	• Increase in predatory species such as grizzly bears, Arctic foxes, ravens and glaucous gulls near development • Reduction in some bird species near industrial development • Decline in reproduction in some caribou herds due to disturbance from oil and gas activity • No evidence of large-scale reduction in caribou herds to date, but continuing spread of industrial activity may ultimately impact on herds
Oil spills	• No major spills in the North Slope oil field operations although there have been three major spills in the pipeline • Many small spills in oil field operations but not frequent or large enough for effects to have accumulated
Expansion into new areas	• Seismic exploration is expanding into new areas with amount of damage reduced, but still evident • Long-term effects in new terrain unknown, particularly in light of climate change
Abandoned infrastructure and unrestored landscapes	• The gravel requirements of new facilities are reduced but less attention paid to previously disturbed sites • Only about 1% of the area affected by gravel fill has been restored • Unclear who is responsible for clean-up

Table 5.18. Potential zones of impact on fauna caused by infrastructure in the Arctic. The types of impact include reduced habitat availability, abundance and survival of some species (UNEP, 2001).

Type of disturbance	Zone of impact, km		
	Reduced abundance	Cumulative impacts on ecosystems	Low disturbance
Plants/vegetation			
Power lines/pipelines	0–0.5	0.5–2	>2
Roads	0–1	1–10	>10
Settlements	0–10	10–30	>30
Fauna			
Power lines/pipelines	0–4	4–16	>16
Roads	0–5	5–20	>20
Settlements	0–10	10–40	>40

impacts into the future with scenarios of current, reduced, and enhanced rates of development.

The results of the analysis indicate that, using the zones of impact estimated by the authors (UNEP, 2001), about 5% of the Arctic was affected by infrastructure in the first part of the twentieth century but that this Figure has now increased to 20–25% of the Arctic land mass, largely because of oil and gas development in Alaska and Russia. Using the recorded growth rate of all infrastructure in the Arctic from 1940–1990 as the basis, by 2050 about 50–80% of the Arctic will show some disturbance from the development of infrastructure. Parts of Russia and Scandinavia will reach those levels within 20–30 yr, while northern Canada and Greenland will develop more slowly. The major impacts will be from fragmentation of Arctic habitat with effects on the biological community from changes in species distribution as some species are more affected than others. These effects may be particularly significant in the Arctic because of the simple, compressed food chains of Arctic terrestrial wildlife which rely on slow-growing forage resources (UNEP, 2001).

5.4.7. Concluding comments and recommendations on the impacts of oil and gas activity on terrestrial ecosystems

Until now, the major impact of Arctic oil and gas activity on the terrestrial system appears to have been from changes to the environment caused by increasing infrastructure such as roads, airstrips, expanded communities, and a general increase in human activity in the form of truck travel, airplanes and helicopters, and shipping. As stated in the review of oil and gas activity in Alaska by the National Academy of Sciences (NAS, 2004), changes to caribou distribution, species distribution (increases in some species, declines in others), and physical changes to the habitat are currently the major effects of oil and gas activity. Dust, noise, and impacts on permafrost and surface hydrology extend the influence of the structures beyond their physical footprint. Expanding oil and gas activity over the next ten years will cause even greater changes in these and other areas of the Arctic. Regional maps of developing oil fields, such as the North Slope of Alaska, show the extent of changes over the last forty years. Despite changes in field practices by oil and gas companies and increased awareness by regulators, an increased volume of activity may cause changes within the biological community. Caribou has been a central focus of ecological research in this area, which is reasonable given its status in the biological community and with indigenous peoples in the North, but the current level of impacts is unclear. Further research is needed on the fundamental ecological interactions between species as development expands and the ecosystem changes, particularly in response to a warming climate. In this regard, research in such areas as the effects on nesting waterfowl and shorebirds and changes in predatory species in the light of oil and gas activity are providing basic information on changes, or the lack of them, on key ecological processes. Possible changes caused by phenomena such as subsidence show the potential for changes on a geological scale caused by the removal of oil, gas, and formation water, and the implications for shorebirds and waterfowl. High quality, peer-reviewed research in these areas should be expanded to include more species and funded through independent agencies to remove questions of credibility of the conclusions.

Oil spill experimental studies throughout the Arctic, primarily during the 1970s and 1980s, provide valuable information on the spill characteristics and weathering of crude oil under Arctic conditions and the recovery of the terrestrial biological community. It is clear from these studies that contact with oil generally results in the death of plants, and that the oil may remain unmodified in the active layer for decades. These study sites should be revisited on a regular basis and monitored using modern chemical techniques to determine the fate and persistence of toxicologically significant components. Soils and plant species should also be monitored for evidence of toxic metabolic by-products from the degradation of hydrocarbons. Some of this information might also be available from natural seeps, which should be studied more intensively for hydrocarbon fate, transport and degradation. Information is needed in this area to reclaim hydrocarbon-contaminated sites in the future. Sites requiring remediation will increase in number as oil and gas activity expands over the next decade, potentially leaving a large number of sites. Monitoring the recovery of these sites should include data on small mammals, birds and other species present.

Most Arctic countries have developed soil quality criteria to protect the environment and human health; however the suitability of soil quality guidelines for the Arctic needs to be tested, particularly in the models used to estimate exposure in receptor species. Many of the guidelines have been developed using physical conditions from temperate regions, and ecological and human health exposure pathways that may not be appropriate in the north. These guidelines provide a valuable basis for the remediation of sites to allow protection of the environment.

A continuing issue with the development and transportation of oil and gas products is the possibility of major oil and brine spills on land. The impacts of these spills on local mammalian and avian species has been largely ignored, although significant exposure to ground oil spills is unlikely for large terrestrial mammals. It seems likely that all small mammals and ground dwelling birds directly affected by

the spill would either be killed or displaced. Repopulation from surrounding areas would probably occur but it would take a long time, probably decades, for the habitat to return to a state where these species could be supported. The cumulative impact of smaller, more frequent spills has not been studied.

Clinical studies from domesticated species indicates that a number of sub-lethal effects, including several directly related to reproductive success, occur as result of the inhalation or ingestion of releases from oil or gas fields. Except for seabird toxicology and research on otters and mink after the *Exxon Valdez* spill, very little toxicological research has been conducted on the potential impact of petroleum-contaminated sites and releases from oil and gas facilities in the Arctic. Tentative relationships have been reported between exposures to some oilfield chemicals, such as benzene, and the frequency of stillbirths in domestic species. There are apparently no studies conducted on the behavioural, physiological or biochemical responses of large Arctic mammals to petroleum exposure which could be used to predict sub-lethal effects from oil and gas releases. Short of dosing caribou/reindeer in controlled experiments, it may be possible to use surrogate species, modelling methods and sub-lethal studies to extract information on uptake, metabolism and the biological consequences of exposure to oil and gas through ingestion and inhalation pathways. This information is fundamental to understanding the potential toxicological effects of oil and brine spills on the terrestrial environment. The significance of the inhalation pathway for exposure to volatile compounds and vented gases should also be studied, given the possibility of flaring gas wells and the presence of sour gas in some fields. At the time of this assessment, there appear to be no estimates of safe inhalation or ingestion limits for oils and other oilfield chemicals in large Arctic mammals. Sub-lethal effects, particularly genetic effects, have also been reported in small mammals exposed on petrochemical sites or downwind from blowouts and emissions from sour gas wells. These studies report valuable linkages between emissions and effects in receptor species although the ecological significance at this time is unclear. More studies on biochemical and genetic markers in small mammals and birds near producing oil and gas fields are needed.

In some areas of the Arctic, oil companies have developed new methods to reduce the impact of oil- and gasfield practices. Conducting seismic and construction work in the winter, reducing the size of pads for wells, reduction or elimination of sumps, and restricting the travel of personnel have reduced the amount of change caused by each project. However, improved practices are not consistent across the Arctic and damage from development continues in some areas. These improvements need to be recognized, further improvements encouraged, and the new practices implemented across the Arctic to reduce changes as much as possible.

Changes in the distribution of some species due to human activity are particularly worrying for threatened and rare species. It is conceivable that oil and gas reserves may be developed in areas of the Arctic that are the only remaining habitat of a threatened species. One example of this type of concern is the development of a field near a small caribou herd that is declining and whose reproduction may be altered by local human activity. Basic research is needed on the best management strategy to ensure the protection of the herd, and the benefits and hazards to the herd from continuing development. These are scientific questions that require further study.

Given the information available for this review, the current cumulative effects on the terrestrial ecosystem in the Arctic that can be attributed to oil and gas activities appear to be comparatively minor, with the displacement of some herds of caribou and reindeer, changes in the distribution of other species, and local effects due to spills. A major qualifier for this conclusion however is the lack of information from Russia which has the majority of land-based oil and gas activity. Several anecdotal stories indicate that effects from drilling and pipeline leakage are significant and long-lasting, however published studies documenting these changes were not available for this assessment. While the initial stages of cumulative effects are noted here (section 5.4.6), expanding oil and gas activities in the next ten to twenty years may cause changes in some areas that may take decades to recover. The Arctic Council should establish rigorous indicators of changing conditions in the Arctic that can be used to monitor the terrestrial system in relation to expanding oil and gas activity.

5.5. Petroleum hydrocarbons and human health

Evaluating the impacts and potential risks associated with exposure to oil and gas exploration, drilling, storage, and transportation for any population is a daunting task whether the exposed population is human, domestic or wild animal, avian, or aquatic. For all these populations, exposures vary by species, location, season, dietary habit, and often gender and age. Within a population, effects can vary by individual size, duration and route of exposure, exposure concentration, concomitant exposure to other substances, and general health of the individual.

The complexity of 'petroleum hydrocarbons' also presents a huge challenge. Chapters 2 and 4 describe in detail the array of processes and methods for oil and gas discovery, production and transportation, and the constituent compounds (volatile and non-volatile, organic and inorganic). Spills of various types of crude oil or refined products (e.g., gasoline, diesel, kerosene) contaminate the environment with different kinds of petroleum hydrocarbons in different proportions depending on their source and level of refining. The components (aliphatic and aromatic compounds, polycyclic aromatic hydrocarbons and branched alkanes) range from highly volatile and biodegradable compounds to hydrophobic and persistent compounds. The transformation of oil as a result of weathering and anaerobic, aerobic, physical and photolytic degradation was described in section 5.1.1.

Tanker spills receive a great deal of media and scientific attention because of the worldwide spill reporting system and the visibility of these spills. Epstein and Selber (2002) stated that globally … *"the quantity of oil released from the combination of smaller accidents, operational leaks and pipeline bursts actually greatly surpasses the amount released from the large spills from super-tankers"*. In the Arctic, total releases of petroleum hydrocarbons from natural seeps and small spills are far greater than from large spills (see Chapter 4). While large catastrophic spills have the potential to cause the most harm to health and the environment (large tonnage, potential for widespread distribution, acute and chronic exposure period, loss of natural resources used for food, etc.), these appear to be very rare in the Arctic region itself (see Appendix 5.1). Considering the amount of petroleum hydrocarbon based production in the Arctic and the extent of pipeline use, losses from pipelines,

pumping stations and other land operations, spills are likely to occur and so are considered in this evaluation of the health impacts of oil and gas activities.

Human exposure to petroleum hydrocarbons also occurs during the handling and use of fuels at home and work (e.g., using gasoline or kerosene as a home solvent and degreaser, vehicle refuelling and home heating fuel spills, use of contaminated containers for cooking or storage, and addictive uses such as sniffing). Human exposures to PAHs can also be very significant from personal, social and cultural practices relating to how food is cooked or preserved (cooking over an open flame and smoking of food increases exposure to several PAHs), how well homes are ventilated from open fires used for heat or cooking, and the use of tobacco products (smoking and chewing lead to significant exposure to PAHs).

Oil and gas exploration and extraction can lead to well site discharges which can contain toxic substances, combustion related gases and particulates, to the introduction of workers from within and outside the Arctic who may be carrying communicable diseases, and to social pressures that affect community structure and tradition. These stressors associated with oil and gas exploration and extraction have complex interactions when coupled with existing stressors on the health of the Arctic population, for example, the availability of health care, climate change, changing diets, and the generalized contamination of some traditional foods.

5.5.1. Scope of the human health evaluation

There is a thin line between physiological, psychological, and sociological health impacts of most toxic substances; most research finds these three types of impacts linked and each capable of influencing the other. The World Health Organization defines health as "*….complete social, mental and physical well-being, not just the absence of disease…*". This assessment will focus primarily on physiological and psychological impacts and to a lesser degree on sociological impacts as these were addressed in Chapter 3. Because the previous AMAP assessment on the impacts of oil and gas activity in the Arctic (AMAP, 1998) did not assess impacts on human populations, this section provides the first assessment of potential human health impacts for any populations in the circumpolar Arctic.

This section first examines sources of human exposure and the types and routes of exposures associated with oil and gas activities in the Arctic. It then examines the uptake, metabolism, excretion, and toxic effects of the major components of oil and gas and their transformation products on laboratory animal species and groups of exposed humans. It then reviews published peer-reviewed studies of impacts on human populations in areas of the Arctic and sub-Arctic where oil and gas activities exist and/or spills have occurred. Finally, this section looks at cumulative impacts and risks for human populations.

This assessment does not provide detailed toxicology information for the subcomponents of oil and gas and their associated products and drilling or processing components because extensive summaries already exist (several US Agency for Toxic Substances and Disease Registry 'ToxProfiles' [ATSDR, 2007]; IARC, 1982, 1987, 1989; API, 2001; Park and Holliday, 1998). Data from the occupational health literature have been considered because they can inform evaluations of non-occupationally exposed populations, however occupational health impacts are not, *per se*, a focus of this assessment.

5.5.2. Human exposure to toxic substances from oil and gas activities in the Arctic

5.5.2.1. Sources

Chapters 2 and 4 described the key oil and gas exploration, extraction/production, transportation, and storage scenarios for the next decade in the Arctic. A short description of oil and gas exploration, drilling, and extraction processes was provided by Verma et al. (2000). This section summarizes key sources of human exposure to substances associated with oil and gas activity. Depending on their composition and volume, some effluents or products have the potential to reach human receptors and may be of relevance to human health.

5.5.2.1.1. Exploration/drilling sites

Gardiner (2003) and USEPA (2005) provided useful overviews of occupational and environmental exposures at offshore oil and gas installations. These overviews were generally relevant for onshore sites as well. The Ocean Discharge Criteria Evaluation prepared by the USEPA (2005) described the discharges associated with oil and gas exploration drilling as a result of the drilling process, equipment maintenance, and personnel housing as follows: *"The major discharges are drilling muds (fluids) and drilling cuttings. Other discharges may include sanitary and domestic wastes, desalination unit wastes, boiler blowdown, test fluids, deck drainage, blowout preventer, uncontaminated ballast and bilge water, excess cement slurry, non-contact cooling water, fire control system test water, and excess cement slurry at the sea floor"*. Composition and amounts of discharges in exploration drilling in Alaskan waters (primarily the Beaufort Sea and Chukchi Sea) are presented in a series of tables (USEPA, 2005). In offshore operations, these materials are commonly discharged directly from the drilling rig or platform into the receiving water, although greater effort is now being made to capture drilling muds and cuttings. On land, these materials are usually captured in holding ponds (sumps), re-injected down the drill hole, or transported to a disposal site.

Drilling muds can contain metals, salts, conditioning chemicals, and crude and refined oil. The most common 'weighting agent' in drilling muds, whether water-based or synthetic, is barite which is composed of barium sulphate and trace contaminants such as mercury, cadmium, arsenic, chromium, copper, lead, nickel, and zinc (USEPA, 2005). Although the metals in barite itself are not thought to be bioavailable, some are of toxicological significance for humans and in 1993 the USEPA adopted a regulation to control the concentrations of heavy metals in drilling fluids, for example, the maximum permissible amounts of cadmium and mercury are 3 mg/kg and 1 mg/kg, respectively. Most oil producing countries have guidance criteria to protect the marine environment. Indirectly, this will also benefit health by reducing the availability of mercury and other metals for uptake by marine species consumed by coastal Arctic peoples.

Drill pipe 'dope' (15% copper, 7% lead) and drill collar 'dope' (35% zinc, 20% lead, 7% copper) can also contribute small amounts of several metals to the drillings muds and cuttings (USEPA, 2005). While lead is a metal of concern for human health, monitoring data for Arctic residents indicate that blood lead levels are very low (AMAP, 1998, 2003).

Some data are available on the characteristics of drilling cuttings from offshore Alaskan exploration sites (USEPA, 2005). The most abundant metals in drill cuttings from this region include iron, aluminium, and zinc which are found (largely undissolved) in the g/kg (ww) of slurry range. Metals

of toxicological significance to humans (largely dissolved and bioavailable) are in the mg/kg range (e.g., lead, 21–298 mg/kg; arsenic, 7–10 mg/kg; cadmium, 0.4–16 mg/kg; mercury, 0.1–0.5 mg/kg). Several organic compounds such as phenanthrene, pyrene and dibenzodioxin (specific chemical identities unspecified) have been reported in the tens and hundreds of mg/kg of slurry range. Approximately 0.2–2 barrels (8.4–84 US gallons) of total drilling waste are produced for each vertical foot drilled at these offshore Alaskan sites (USEPA, 2005). Drilling muds and produced water may also contain low concentrations of naturally occurring radionuclides (Rajaretnam and Spitz, 2000); but these have little potential to impact health as the radiation characteristics (levels and type of decay particle emissions) are weak.

Overall estimates of discharged drilling fluids (muds and cuttings) for the Beaufort/Chukchi Sea areas of exploration (approx. 37 wells) are about 68 000 barrels (i.e., approximately 2.8×10^6 US gallons) (USEPA, 2005).

Discharges of sanitary and domestic waste from drilling sites are highly variable depending on the period of operation, occupancy, platform characteristics, and operational situation. An Alaskan permitting report (USEPA, 2005) estimated that combined sewage volume can vary from around 900 to 75000 gallons per day in the Beaufort/Chukchi Sea region. Most sewage and domestic waste is stored and treated.

5.5.2.1.2. Production sites

Produced water, which comes from the subterranean oil-bearing rock formation during production, may contain metals, salts, naturally occurring radionuclides, a full range of dissolved hydrocarbons (e.g., benzene, xylene, toluene, ethylbenzene), and dispersed oil. It is usually the largest volume waste by-product in the oil extraction process. In wells nearing the end of their productive lives, produced water can make up 98% of the fluids drawn to the surface (Weideman, 1996). In general, onshore produced water is re-injected into the ground and is not discharged onto the land, whereas past practice offshore was to discharge treated produced water directly to the ocean (T. Baker, Indian and Northern Affairs, Canada, pers. comm.). Some jurisdictions such as OSPAR are now moving from maximum allowable concentrations of dispersed oil in produced water to a zero discharge policy. Produced water not re-injected may be discharged into surface waters (O'Rourke and Connolly, 2003) although most produced water is treated with settling or skim tanks and cartridge filtration or gas flotation systems prior to disposal.

Well gas contains several components (methane, ethane, propane, butane, carbon dioxide, and sulphur-containing gases such as hydrogen sulphide (H_2S) and methyl mercaptan). From a health perspective H_2S is the most important gas (see section 5.5.4.3.3). Gas containing more than 1% (10000 ppm) H_2S is generally referred to as 'sour gas'. While wells in sub-Arctic regions such as Alberta, Canada, have average H_2S levels of 10% (and ranging from trace amounts to 80%), levels of H_2S in Arctic crude oil and gas tend to be far lower. Most Arctic gas is referred to as 'sweet gas' because levels are well below 10000 ppm (PCF, 2000).

Flaring can produce particulate matter, PAHs, and sulphur-containing gases. Typical releases from a sweet gas flare are provided in Tables 5.19 and 5.20. In the first example from the Beaufort Sea in the Canadian Arctic, maximum concentrations of contaminants in air (in μg/m^3) are estimated at locations 1.4 km from three sites related to exploratory drilling activity. In the second example, from the southern part of Canada, levels (in mg/m^3) are actually measured, but at 5 m above the flare. According to these data, it is likely that volatile organic chemicals (VOCs) and PAHs disperse rapidly and that levels decline from mg/m^3 to sub-μg/m^3 over 1 km away. Nitrogen dioxide, sulphur dioxide (SO_2) and particulates (measured at 2.5 μm or less) are also predicted to occur at μg/m^3 levels beyond 1 km from the flare. Under a local temperature inversion however, some of these gases could become trapped in the ground level atmosphere for several days and lead to some inhalation exposure in a population living in the immediate area of the inversion.

Table 5.19. Maximum predicted contaminant concentrations (1.4 km from sites shown) due to sweet-gas flaring at a Beaufort Sea exploration site (Devon, 2004).

	Maximum hourly concentration, μg/m^3 [a]			Ambient air criterion, μg/m^3
	SDC [b]	LTC [c]	Ice island	
NO_2	3.5100	3.5300	3.5200	400
SO_2	0.8600	0.8600	0.8500	450
$PM_{2.5}$	1.9300	1.9400	1.9400	30 [d]
Benzene	0.0081	0.0081	0.0081	30 [e]
Formaldehyde	0.0595	0.0598	0.0596	65 [e]
Total PAHs	0.0002	0.0002	0.0002	36 [f]
Naphthalene	0.0006	0.0006	0.0006	500 [f]
Acetaldehyde	0.0022	0.0022	0.0022	90 [e]
Acrolein	0.0005	0.0005	0.0005	28 [f]
Ethyl benzene	0.0735	0.0739	0.0736	-
Hexane	0.0015	0.0015	0.0015	35,000 [f]
Toluene	0.0030	0.0030	0.0030	2000 [f]
Xylene	0.0015	0.0015	0.0015	2300 [f]

No criteria; [a]distance from flare to point of maximum hourly concentrations is 1.4 km; [b]steel drilling caisson platform; [c]landfast tender-assist drilling platform; [d]as a 24-hr average; [e]Alberta criteria; [f]Ontario criteria.

Table 5.20. Predominant VOCs and PAH compounds measured 5 m above a sweet gas flare in Alberta, Canada (Strosher, 1996).

	Concentration, mg/m^3	PAH	Concentration, mg/m^3
Acetylene	53.7	Acenaphthene	2.9
Benzene	144.5	Acenaphthylene	23.2
Ethylene	29.0	Anthracene	42.1
Hexane	8.5	Benz[*a*]anthracene	17.3
Naphthalene	99.4	Benzo[*g,h,i*]fluoranthene	10.2
Styrene	75.5	Chrysene	2.2
Toluene	18.2	Fluoranthene	51.4
Xylene	29.8	Pyrene	32.4

Minor leaks and small spills of crude oil may occur during offshore and onshore drilling as a result of equipment failure, improper seals, human error, and maintenance activities. Blowouts (e.g., *Ekofisk* and *Bravo* in the North Sea, see Appendix 5.1) occur when formation fluids are ejected from the well by subsurface pressure and the well flows uncontrolled. Although the size of these spills is typically small (less than 50 barrels), some can be very large (e.g., the *Bravo* drilling platform blowout in the Norwegian North Sea in 1977 spilled 12 700 m^3 of oil into the sea), especially if there is a delay or difficulty in drilling a relief well or bringing the blowout under control for other reasons (e.g., damaged infrastructure or bad weather). Oil entering water, sediments, or soil can act as a source from which petroleum hydrocarbons continuously re-enter the food chain for months to several years.

5.5.2.1.3. Oil/gas transfer routes and storage sites

Pipelines, pumping stations, transfer terminals, and storage tanks are vulnerable to natural disasters, accidental ruptures, equipment failures, sabotage, and illegal diversions (see Appendix 5.1 for examples in the Arctic and sub-Arctic). While most modern transmission pipelines are well maintained,

some older and poorly maintained flow lines and gathering lines are designed for the life of the field and, according to Epstein and Selber (2002), can have a life span of only 15 yr. Recent spills from gathering lines in an Alaskan production facility (Doyle, 2006) and information on pipeline accidents in Russia (see Chapter 2) point to the significance of aging and poorly maintained pipeline infrastructure. Product losses and spills can lead to: occupational, bystander and local population exposure; contamination of drinking water reservoirs, lakes and rivers (e.g., the Komi pipeline leak in Russia); long-term contamination of soil; increased levels of volatile hydrocarbons; and tainting and chemical contamination of traditional food consumed by local populations. Poland et al. (2003) stated that contamination from routine operations is a significant source of hydrocarbon pollution in the Arctic; for example, leaks from poorly maintained pipelines in Russia account for losses of 5–15% of the total annual oil production.

5.5.2.1.4. Tanker transport

Today, oil makes up over half the annual tonnage of all sea cargoes (O'Rourke and Connolly, 2003). In general, following tanker oil spills in the ocean, most light fractions of crude oil evaporate into the atmosphere or dissolve in the water column. Most aromatic compounds that remain are quickly metabolized and excreted by fish or readily degraded by bacteria (see section 5.3.1); however, some are not as easily metabolized. Of all components of crude oil, PAHs are most likely to enter the food chain and be stored in sediments where they may continue to contribute to food-chain uptake over time.

Volatile organics evaporate rapidly from crude oil spilled on water surfaces and on land; temperature and wind conditions are important variables with respect to the rate of volatilization. Modelling predictions after the *Exxon Valdez* spill estimated that benzene levels in air would drop from 5–0.02 ppm within 12 hr of the spill (Hanna and Drivas, 1993). Similarly, Zhou and Wong (1997), based on model predictions, suggested that *n*-hexane, toluene and *n*-pentane would be completely lost from a spill in one to a few hours. This rapid volatilization and subsequent air shed dilution has consequences for populations living downwind of a spill, namely lower exposure.

High winds following an oil spill together with high waves and rocky shorelines (e.g., the M.V. *Braer* spill, see Appendix 5.1) and the use of high pressure hoses to clean beaches can lead to aerosolized oil droplets which are respirable by both spill workers and local populations (Campbell et al., 1993).

Combustion of oil floating on water has been used as a technique to reduce the impact of spilt oil in actual spills (*Palva*, see Appendix 5.1) and in controlled experimental burns (Fingas, 1995a,b). This technique creates a wide range of airborne PAHs attached to particulates and gases such as SO_2 (a combustion product of H_2S) which have the potential to affect health.

Casal Lareo and Medina (2002) provided guidance on conditions for clean-up crews and commented on safety measures that could be applied to populations living near spill sites.

5.5.2.1.5. Processing and distribution of oil and gas and use of refined products

Although there are very few refineries or gas treatment plants in the Arctic there are some in Arctic Russia and Alaska. These plants are sources of fugitive emissions. Processing of gas and unstable condensate occurs at the Sosnogorsk gas processing works which is one of the oldest enterprises of the Komi Republic (founded in 1941 on the base of the *Sedelsk* gas field). Severgazprom plans to construct a gas chemical complex at the works after 2007. It will incorporate a gas fractioning plant designed to separate the gas mix into components for industrial use both in Russia and abroad (Severgazprom, 2007). There are two refineries in the Russian Arctic (the Ukhta Refinery and the Strezhevoy Oil Refinery) which produce about 250 000 tonnes of diesel products per year. Just south of the Arctic Circle there are two larger refineries (the Achinsk Oil Refinery and the Perm Refinery) which produce about three million tonnes of gasoline, diesel, kerosene, and other fuel products per year. In Alaska, there are two small infield refineries ('topping plants') near the *Kuparuk* and *Prudhoe Bay* fields. There are no human exposure data available for these sites, however some emissions data for these sites can be found in Chapter 2.

Table 5.21. Measured air pollutant concentrations (μg/m3) at Prudhoe Bay, Alaska, 1986–1996 (MMS, 2003; BLM, 2005).

Pollutant [a]	Monitoring sites				National Standards [f]
	CCP [b]	Pad A [c]	CPF-1 [d]	DS-1F [e]	
O_3					
Annual max. 1 hr	115.8	180.3	115.6	100.0	235
NO_2					
Annual	26.3	11.9	16.0	4.9	100
PM_{10}					
Annual	-	-	13.6 [g]	11.2 [g]	50
Annual max. 24 hr	29.3	-	108 [g]	63 [g]	150
SO_2					
Annual	2.6		5.2	2.6 [h]	80
Annual max. 24 hr	10.5		26.2 [i]	13.1	365
Annual max. 3 hr	13.1		44.5	55.0	1300
CO					
Annual max. 8 hr	-	-	920 [g]	575 [g]	10000
Annual max. 1 hr	-	-	1265 [g]	1035 [g]	40000

[a] Lead was not monitored; [b] Site CCP (Central Compressor Plant) Prudhoe Bay monitoring program, selected for maximum pollutant concentrations. All data are for years 1992–1996; [c] Site Pad A (Drill Pad A), Prudhoe Bay monitoring program, site of previous monitoring, selected to be more representative of the general area or neighbourhood. All data are for years 1992–1996; [d] Site CPF-1 (Central Processing Facility), Kuparuk monitoring program, selected for maximum pollutant concentrations measured during November 1990 to October 1992; [e] Site DS-1F, Kuparuk monitoring program site selected to be representative of the general area or neighbourhood. All data are for years 1990–1992; [f] Applicable national ambient air quality standards; [g] Data are from BLM (2005); [h] Minimum instrument detection level; [i] Second highest observed value (in accordance with approved procedures for determining ambient-air quality).

Table 5.22. Comparison of National Ambient Air Quality Standards and measured levels of pollutants from an oil and gas activity centre in Alaska and ambient levels near the site (MMS, 2002). All measured concentrations and NAAQS values are in μg/m^3.

Pollutant	Averaging period	Maximum concentration [a]	Background concentration [b]	Total concentration	NAAQS [c]	% of NAAQS
Initial drilling/commissioning period						
NO_2	Annual	22.1	7.8	29.9	100	29.9
	3-hr	168.7	6.8	175.5	1300	13.5
	24-hr	81.4	4.8	86.2	365	23.6
SO_2	Annual	5.1	0.1	5.2	80	6.5
	24-hr	21.4	7.0	28.4	150	18.9
PM_{10}	Annual	1.3	0.1	1.4	50	2.8
	1-hr	804.0	--	804.0	40000	2.0
CO	8-hr	245.6	--	245.6	10000	2.5
Long-term operations						
NO_2	Annual	19.2	7.8	27.0	100	27.0
	3-hr	183.0	6.8	189.8	1300	14.6
	24-hr	88.2	4.8	93.0	365	25.5
SO_2	Annual	2.7	0.1	2.8	80	3.5
	24-hr	22.0	7.0	29.0	150	19.3
PM_{10}	Annual	1.8	0.1	1.9	50	3.8
	1-hr	804.0	--	804.0	40000	2.0
CO	8-hr	270.4	--	270.4	10000	2.7

[a] All maximum concentrations occur within 200 m of facility boundary; [b] background concentrations include global background and contributions from existing emission sources; [c] National Ambient Air Quality Standards.

Concentration data for volatile compounds associated with upstream operations (producing facilities) from five companies in sub-Arctic Canada were reported by Verma et al. (2000) and discussed in section 5.5.2.3 . Volatile compounds such as toluene, benzene, and xylene can contribute to significant airborne concentrations within the vicinity of the plant although the percentage of individual worker samples which exceeded occupational limits was low (<1% of the samples taken from individual worker monitors exceeded the occupational limit for benzene). Direct losses of non-volatile oil or sludges can contaminate the local environment upon which local populations rely (i.e., soil and ground water contamination, or pollution of nearby water-bodies by contaminated run-off). Flaring at refinery sites can produce particulate matter containing high levels of VOCs, PAHs, and sulphur compounds.

Refined products such as gasoline can also be spilled as a result of leaking storage tanks, during distribution by road and rail tankers, and from refuelling activities at exploration and production sites and in the local community. Gasoline contains very high concentrations of volatile compounds which can enter the ambient air following a spill. A greater risk is exposure to an explosion and particulates from fires which may ensue, not just from the burning fuel, but also from materials set afire during the event (e.g., local structures and their contents). Gasoline sometimes contains additives such as tetraethyl lead; this compound is used far less now than in the past as many countries have banned or severely restricted its use. The significance to biotic receptors is that lead is rapidly taken up into the food chain as alkyl lead and in this form can be a serious health threat if exposure is sustained.

Well gas and refined oil and gas products are used extensively at exploration, production, and transfer/transport sites to run generators, pumps, and other equipment. These emit exhaust containing a range of PAHs and sulphur-containing gases. During times when use of gas- or oil-based combustion equipment is extensive and climatic conditions promote temperature inversions or slow dissipation of combustion products, human populations at or within the near proximity of these facilities have the potential to be exposed to respirable airborne concentrations of several substances. However, levels are likely to be very low.

Combined measurements of ambient and emitted gases, hydrocarbons, and particulates obtained within a few hun-

Table 5.23. Physical properties of oil, gas and their constituents and routes of primary human exposure.

	Vapour pressure at 20 °C, mmHg	Primary route of human population exposure
Crude oil	-	Dermal
Gasoline	38–300	Dermal/Pulmonary
Other fuels (kerosene, fuel oil, JP-8)	0.48 and higher	Dermal/Pulmonary
PAHs	negligible	Dermal/Oral (food)
Naphthalene	0.08	Pulmonary
Benzene	76	Dermal/Pulmonary/Oral (water)
Toluene	22	Dermal/Pulmonary/Oral (water)
Xylene	-	Dermal/Pulmonary/Oral (water)
Trimethylbenzene	1–2	Pulmonary
n-Hexane	-	Pulmonary/Dermal
H_2S	13376	Pulmonary
SO_2	-	Pulmonary
Metals	none	Oral
Radionuclides	none	Oral

dred metres of gas and oil operations in Alaska in the mid-1990s indicate that levels of most substances were in the µg/m^3 range (Tables 5.21 and 5.22). Measurements at these sites tended not to exceed US air quality standards.

Moen et al. (1995) measured exposure levels of volatile petroleum-based substances on oil tankers. Some seamen have reported central nervous system symptoms. Exposure concentrations onboard ship varied by type of cargo, care taken when loading fuel products, and whether hatch covers were open or closed. Most measured levels were below 2 ppm. Levels of PAHs and tetraethyl lead were low.

5.5.2.2. Exposure routes

Exposure routes are largely dependant on physical properties of oil and its individual constituents, and human activity (Table 5.23). Factors influencing the predominant exposure routes for an individual include age, gender, occupation and leisure-based activities, type of pollutant, and proximity to contamination. Human exposure to crude oil, gas and their components and products occurs through:

- direct skin contact (dermal route) with contaminated soil or through handling products or contaminated surfaces;
- inhalation (pulmonary route) of droplets, particles contaminated with petroleum hydrocarbons or metals, or volatile gases found in outdoor air or volatile gases found in indoor air as a result of de-gassing from contaminated water during bathing/showering; and
- consumption (oral route) of contaminated or 'tainted' food, water or soil.

Dermal exposure is most likely to occur where individuals are directly exposed to oil in an occupational setting or temporarily during spill clean-up activities, when handling oil-contaminated clothing or equipment, and occasionally when oil is aerosolized due to wind and temperature. Dermal exposure to lighter fractions of oil such as gasoline or kerosene may occur unintentionally as a result of personal use of these products for degreasing hands and equipment. Occupational dermal exposures can be reduced through education and the use of protective clothing. Local population dermal exposure can be reduced through education by stressing the use of measures to avoid direct contact with oil and to wash clothing and skin surfaces effectively.

Pulmonary exposures can be very variable and significant. High H_2S levels can be sustained for significant periods during well blowouts, for example, the Lodgepole, Alberta (Amoco) sour gas blowout in 1982 lasted 67 days and caused two worker deaths and a range of symptoms such as headaches, nosebleeds, eye irritation, stomach ailments, and breathing problems (ERCB, 1984). Volatile hydrocarbons such as toluene, xylene and benzene can evaporate quickly into ambient air immediately after a spill and can lead to exposures of populations close to the spill site. Oil droplets may enter and be deposited directly into the lungs under specific environmental conditions and clean-up procedures. VOCs and particulates (containing sulphur and PAHs) associated with fires and intentional flaring and droplets in the $PM_{2.5}$ to PM_{10} range can be inhaled and remain in the lung permanently; larger particles are likely to be removed from the lung through turbulence and muco-ciliatory mechanisms. Both retained and transient particulates and droplets can damage lung tissue. Pulmonary exposure is the hardest non-occupational exposure to avoid; human populations must breathe and respirators are seldom practical or available in exposed communities. Levels of volatile aromatics and aliphatics and PAHs above a sweet gas flare are shown in Tables 5.19 and 5.20.

Oral exposure is the most common route of exposure to oil-based PAHs. PAHs are most likely to be ingested as a result of consuming contaminated marine or freshwater foods. Among marine organisms, the bioaccumulation of PAHs occurs to the greatest extent in molluscs and shellfish due to their immobility and direct contact with sediment. Avoidance of tainted food can lead to dietary changes and forced migrations to other hunting areas unaffected by the contamination. Drinking water contamination can also lead to exposure to PAHs and dissolved organic chemicals. Exposure to naturally-occurring radionuclides which are found in drill cuttings and produced water at well sites, would only occur if these wastes contaminated local water supplies. Avoidance of contaminated water and tainted food is very difficult in many Arctic areas as there are few alternative sources of clean food and water nearby for some small communities.

5.5.2.3. Concluding comments

The largest volume of discharges from offshore exploration wells in the Arctic comes from drilling muds and drilling cuttings. Overall, the combined mixture of muds and cuttings can be discharged in the thousands of US gallons annually per well (based on a calculated average of 75 000 US gallons per year from 37 wells in the Beaufort and Chukchi Seas) (USEPA, 2005). The USEPA has calculated hypothetical metal discharges associated with this average annual discharge to be approximately 300–4500 kg lead, 125 kg of arsenic, 5–250 kg cadmium, and 5 kg mercury. In relation to the deposition of mercury and lead via long-range transport, additions to the marine environment from wells are relatively small. Furthermore, the discharges are for a short period of drilling (<1 yr for an exploratory well and 2–3 yr for exploration of an entire area) and are rapidly diluted by seawater. Most of these metals are not dissolved or bioavailable. Fish species in the vicinity of the well sediments may become contaminated but may also not be part of the food supply for human populations.

Organic chemicals such as phenanthrene, pyrene, and dibenzodioxin have been reported quantitatively for wells in Alaska (USEPA, 2005). Using similar calculations, the USEPA calculated that total average annual discharges for these organics may be in the hundreds and thousands of kilograms. The chemical identity of these chemicals is unspecified; but all have the potential to enter the food chain upon which local populations may depend.

Current approaches to the capture and reuse of drilling fluids and a move to zero discharge policies among jurisdictions which apply to offshore and onshore sites, suggest that releases of metals and organics will decline significantly in the future. Typically, land-based operations store fluids in sumps designed for this purpose or are re-injected down the drill hole. Land mammal food chains are much shorter than marine mammal food chains, so the likelihood of human exposure to ground-based contaminants through food chain biomagnification of metals such as mercury is orders of magnitude lower.

Produced water is a high volume waste by-product in the oil extraction process. Onshore produced water has normally been reinjected into the ground and not discharged onto the land, whereas past practice offshore has been to discharge produced water into the ocean. Some jurisdictions are now moving from maximum allowable concentrations of dispersed oil in produced water to a zero discharge policy. Produced water to be discharged is most often treated with settling/skimming

tanks and filtration/flotation systems (O'Rourke and Connolly, 2003). These approaches, offshore and onshore, severely limit human exposure to hazardous substances which may occur in produced water.

Sanitary and domestic wastes are generally high volume and are usually reinjected or, for land-based wells, stored in holding ponds or tanks. Most Arctic countries report that releases to water and land are very unlikely (no information is available for Russia). In the event of a wastewater spill at an offshore facility, ocean dilution is likely to prevent any significant health threats as organic waste breakdown is relatively rapid even in an Arctic Ocean environment. Human exposure to chemicals, bacteria, or viruses in spilled domestic waste is also unlikely because the water surrounding an offshore well site is not used for human consumption or for recreational purposes and fish are rarely caught for consumption at the well site. At onshore well sites, containment areas and routine sewage treatment in settling ponds are likely to be sufficient to prevent any significant human exposure. Currently, permafrost prevents seepage of sewage into nearby water bodies. It is important to prevent accidental discharges of sewage into lakes or rivers. Overall, human exposure to biological and domestic waste associated with oil and gas activities is negligible and impacts on local populations and workers are highly unlikely if basic national guidelines for waste containment and treatment are followed.

Most Arctic gas is sweet gas, namely, natural gas containing <1% (<10 000 ppm) H_2S by weight. Gas releases at the well head may ignite (accidentally or intentionally). Flaring at the well head can lead to particulate matter, VOCs, PAHs, and sulphur-containing gases. Flaring appears to be extensive in the Russian NAO/Komi oil and gas enterprises and has been identified as the largest contributor to airborne pollution in the Nenetz autonomous area (see Chapter 2). Estimates of airborne emissions vary from approximately 30 000 t/yr (hydrocarbons, CO_2, NO_2, SO_2) in the NAO to 10 000 t/yr (based on combustion of nearly one billion m^3 gas) in the YNAO (see Chapter 4).

Sour gas contains more than 10 000 ppm by weight H_2S and can come from wells in Arctic Russia and in sub-Arctic regions of Canada, i.e., Alberta. The Canadian Petroleum Communication Foundation (PCF, 2000) considers an H_2S release resulting in an exposure concentration of ≤10 ppm over the course of 8 hr a minor release; a major release is one that results in human exposure to H_2S concentrations >10 ppm but <500 ppm over an 8 hr period; a catastrophic release is one resulting in human exposure to concentrations ≥500 ppm over an 8 hr period. Exposure to H_2S can be a health concern to humans and animals (section 5.5.4.3.3). Releases of sour gas containing significant amounts of H_2S may occur in Russia, although there are no data available to evaluate the likelihood of such releases, their concentrations, or their impacts on human health.

Minor leaks and small to large spills of crude oil may occur during offshore and onshore drilling as a result of equipment failure, improper seals, human error, maintenance activities, and blowouts. Oil entering water, sediments, or soil can act as a source from which petroleum hydrocarbons such as PAHs continuously re-enter the food chain upon which local populations may depend.

Oil spills from tankers continue to occur worldwide (Appendix 5.1) and some have occurred in the Arctic, but are rare. Spill volumes related to tanker accidents have varied greatly and depend on a variety of conditions and circumstances (weather, damage to the ship, containment capacity, etc.). Following an oil spill in the ocean, most light fractions of crude oil evaporate into the atmosphere or dissolve in the water column; most aromatics that remain (except PAHs) are quickly metabolized and excreted by fish or readily degraded by bacteria.

Volatile organic chemicals such as benzene evaporate rapidly from crude oil spills (Hanna and Drivas, 1993; Zhou and Wong, 1997). These rapid reductions in airborne levels of VOCs (99–100%) reduce the risks to clean-up workers and local populations. The predicted level of 0.02 ppm benzene (64 μg/m^3) in air at one spill site after 12 hr did not exceed 8 hr time-weighted-average occupational levels (Table 5.24) but may exceed some ambient air guidance values by 2- to 30-fold (see Tables 5.19, 5.25).

While high winds following an oil spill may rapidly disperse VOCs, they can cause aerosolization of oil droplets which are respirable by spill workers and local populations (Campbell et al., 1993). Clean-up methods using high pressure hoses and on-site burning may also cause health impacts by exposing clean-up crews to aerosolized oil droplets and combustion products which may be inhaled.

Table 5.24. Occupational exposure limits (ppm) for volatile substances associated with oil and gas activities (NIOSH, 2005).

Volatile substance	OSHA PEL (ST) [a]	OSHA PEL (TWA) [a]	NIOSH REL (ST) [b]	NIOSH REL (TWA)
Acetylene			C 2500	
Benzene	5	1	1	0.1
n-Hexane	-	500	-	500
H_2S	C 20 50[c]	10	10[c]	-
Naphthalene	-	10	15	10
SO_2	-	5	5	2
Styrene	C 200	100	100	50
Toluene	300	200	150	100
Trimethylbenzenes	-	-	-	25
Xylene	-	100	150	100

[a] OSHA PELs (Permissible Exposure Limit), 'TWA' (time-weighted average) concentrations for OSHA PELs must not be exceeded during any 8-hr workshift of a 40-hr workweek. A STEL is designated by 'ST' preceding the value and is measured over a 15-min period unless noted otherwise. OSHA ceiling concentrations (designated by 'C' preceding the value) must not be exceeded during any part of the workday; [b] NIOSH RELs (Recommended Exposure Limit), 'TWA' indicates a time-weighted average concentration for up to a 10-hr workday during a 40-hr workweek. A short-term exposure limit (STEL) is designated by 'ST' preceding the value; unless noted otherwise, the STEL is a 15-min TWA exposure that should not be exceeded at any time during a workday. A ceiling REL is designated by 'C' preceding the value; unless noted otherwise, the ceiling value should not be exceeded at any time; [c] 10 min maximum.

Table 5.25. Ambient air quality guidelines for the protection of human health from long-term exposure to benzene and benzo[*a*]pyrene (PINRO, 2006; EU, 2000, 2004; Norwegian Ministry of the Environment, 2005).

	Russia	EU	Norway
	Maximum permissible concentration	Limit/Target Value (average over a calendar year)	Target Value
Benzene	-	2000 ng/m^3 (6 ppb) [a]	2000 ng/m^3 (6 ppb)
Benzo[*a*]pyrene	1000 ng/m^3	1 ng/m^3 in PM_{10} fraction	-

[a] The value for benzene was 5000 (EU, 2000) but must be reduced by 1000 each year starting in 2006 so the current value is 2000.

As crude oils and well gas from different regions have different compositions, and different components of oil and gas have different relative toxicity, the chemical and physical properties of oil and the chemical content of gas are of considerable relevance to this assessment. Park and Holliday (1998) tabulated how the one European organization of oil companies characterizes crude oils based on 'spreading behaviour' in a spill. This characterization took account of 'pour' points within a given temperature range and the amount of evaporative losses, from 0 to 50%, 24 hr after a spill. The weathering of oil, which takes into account evaporative loss and dissolution of volatile oil components in water, affects health assessments directly for human populations. Weathering may lead to an initial increase in airborne levels of VOCs such as benzene over the short term as the volatile components evaporate, and reduce exposure concentrations over the long term for spill clean-up workers and nearby communities during the post-evaporative stage.

Benzene is a common constituent of crude oil and is of concern owing to its carcinogenicity. It has been reported to be present at median levels approximating 0.3% by weight in western Canadian crude and 0.7% by weight in condensate from gas plants in western Canada (Verma et al., 2000). Amounts vary with different crude oil and gas sources. Workers in the 'upstream' oil and gas sector (i.e., those involved in producing oil and gas as opposed to 'downstream' workers involved in refining and distributing activities) were generally exposed to benzene concentrations over 8 hr below the 1 ppm (8-hr time weighted average) occupational exposure limit for Alberta, Canada, although a few were higher (Verma et al., 2000). Exceedances occurred primarily among gas plant workers where levels reached nearly 58 ppm partially due to additions of benzene, toluene, and xylene to the gas stream. The authors reported no exceedances related to benzene among workers in the heavy oil processing sector, drilling operations, or pipeline activities over an 8-hr period. Owing to the dilution effect associated with movement of these VOCs to community sites several kilometres from exploration or extraction sites, it is unlikely that day-to-day operations at a well (other than a large spill) would contribute significantly to local population exposures to benzene. Estimated exposure concentrations for benzene 1.4 km from a flare site were also very low and below the ambient air criterion of 30 μg/m^3 (Table 5.19) and the Norwegian target value of 2 μg/m^3 (Table 5.25).

There are currently only a few gas plants and refineries in the Arctic (and these occur in Russia and Alaska). Emissions from these downstream sources can be significant. Tsunina (2002) indicated that in Novokuibyshevsk, the health status of the population had declined due to exposure to high ambient air pollution associated with the oil processing activities. The author also undertook an ecological assessment of the area to better understand causes and effects (Tsunina, 2003). No evaluations of population or worker effects have been published for the two small topping plants in Alaska.

Local spills of refined fuels were identified as potential sources of human exposure to VOCs and PAHs. Local crude oil spills are also of concern; pipelines, pumping stations, transfer terminals, and storage tanks are vulnerable to natural disasters, accidental ruptures, equipment failures, sabotage, and illegal diversions. The American Petroleum Institute evaluated nine components of petroleum products for their 'relative potential health concern' (see Table 5.26) which may vary in significance depending on the spill situation (API, 2001). Crude oil spills can result in exposures to benzene, H_2S, and PAHs which may be of potential concern. Gasoline spills may also lead to exposures to benzene and total hydrocarbons of potential concern to health. Heavy fuel oil spills may be of potential concern due to exposure to PAHs.

Other chemicals or metals of possible concern include dispersants (pulmonary irritants) used for clean-up operations; barium, arsenic, cadmium, chromium, and mercury; some persistent organic chemicals such as polychlorinated biphenyls (PCBs) which may appear as contaminants in cutting fluids or oils or leak from electrical equipment; and naturally occurring radionuclides (non-radiation related toxic effects). Migration and clean-up of minor contamination by PCBs in

Table 5.26. Relative potential health concern of spilt petroleum hydrocarbons (adapted from API, 2001).

	Spilled substance						
	Crude oil	Gasoline	Kerosene	Jet fuel	Diesel/ heating oil	Heavy fuel oil	Asphalt
Benzene	2	2	0	0	0	0	0
n-Hexane	1	1	0	0	0	0	0
H_2S	2	0	0	0	0	1	1
Naphthalene	0	0	1	1	1	0	0
PAHs	2	0	0	0	0	2	1
Tetraethyl or tetramethyl lead (TEL/TML)	0	0/1 [a]	0	0	0	0	0
Toluene	1	1	0	1	0	0	0
Total hydrocarbons	1	2	1	1	1	0	0
Trimethyl benzene	1	1	1	1	1	0	0

2: Component of concern, 1: component of low concern, 0: component of negligible or no concern; [a] depends on whether or not the gasoline contains TEL or TML.

gas pipelines was discussed by Eltohami et al. (2004). As it is the dose which determines the risk to health, the amount, route, and duration of exposure to these and other chemicals determine their relative importance for local populations. It seems that exposures (through air, water, and the food chain) of local populations to these 'other chemicals' and metals associated with oil and gas activity and in the vicinity of upstream operations are very small and unlikely to lead to a health concern.

There are few data available concerning on-site and near-site contamination of land mammals as a result of onshore exploration and extraction; data on contaminant levels in marine sediments, fish, and marine mammals related to offshore sites is more extensive. In general, concentrations in country foods of organic chemicals, metals, and radionuclides resulting directly from oil and gas activities are either very low or difficult to apportion directly to oil and gas activity sources.

Guidelines and standards for components of gas and oil are available for air quality (Tables 5.19, 5.21, 5.22, and 5.25), drinking water quality (Table 5.12) and soil quality (Table 5.13). Air concentration data (measured and estimated) from flare sites (Tables 5.19 and 5.20) and at sites without flaring (Tables 5.21 and 5.22) suggest that emissions from oil and gas operations are likely to lead to very low concentrations in air at locations that are only a few kilometres from these sites. Populations living 5 km or more away from oil and gas exploration and extraction sites would have negligible exposure. Periods of temperature inversion could increase these levels moderately. Estimated concentrations are below most published ambient air quality guidelines among Arctic jurisdictions.

The everyday use of fuels at well sites also contributes to airborne levels of PAHs. Based on the current paucity of measured values, emissions from a typical diesel fuel-fired power plant at a well site must be estimated using a series of assumptions (D. Thomas, AXYS, Victoria BC, Canada, pers. comm.). The estimates assume that PAH is the sum of parent and alkylated compounds, as found in average petroleum oil. The calculation assumptions are as follows:

- Annual usage of diesel oil at an 'average' power plant is 25×10^6 L
- An emission factor for diesel-fired power plants of 3.3 ppm
- That PAHs comprise 5% of 200 ppm unburned fuel
- That PAHs will be largely bound to fine particulates
- The residence time of emissions around the power plant point source is 1 hr
- The discharge point is 10 m above ground
- Average wind conditions of 5 km/hr

These assumptions result in an estimated hourly input of PAHs to the atmosphere around the plant of 0.05 kg (0.008 kg combustion derived; 0.042 kg derived from unburned fuel). The corresponding average ambient PAH concentration in air in the immediate vicinity of the plant would be around 20 $\mu g/m^3$ (below the ambient air quality value for total PAHs shown in Table 5.19). Concentrations would then decrease with distance from the discharge point as the emissions mixed with air and moved away from the site.

Natural petroleum seeps have been identified as significant sources of oil and gas in the Arctic (Chapter 4), even though they are not the result of oil and gas activity *per se*. D. Thomas (pers. comm.) estimated the concentration of VOCs in air at the seawater/air or land/air interface at the site of a typical natural petroleum seep. The estimate focused on benzene which is likely to be the worst case emission (i.e., relatively high vapour pressure and water solubility) and is a known carcinogen. Toluene and xylene concentrations in air are likely to be lower than for benzene.

For the calculation, it was assumed that the average seabed seep rate is 100 t/yr. This number is based on an estimated 40 000 t annual petroleum seepage to the Arctic Ocean and a total of about 400 seepage zones. One hundred tonnes per year per seep zone is probably an overestimate. Crude oil contains approximately 5% benzene. So the amount of benzene entering the marine environment per minute in each seep zone would be 10 g. As the oil seeps up from the seabed, it will rise through the water column to the surface in colloidal and sediment-sorbed forms; benzene will dissolve into the water. If the average water depth is 10 m for the seep zone and the aerial extent of a 'typical' seep is 10 m^2 (based on anecdotal evidence), then the upper limit of benzene concentrations would be about 6 mg/L water.

At the sea surface, benzene will evaporate quickly. Experimental measurements of the evaporative half-life for benzene under conditions of a 25 km/hr wind, is about 5 hr. If the surface coverage of the surfacing benzene-enriched water is 20 m^2 (i.e., a very conservative assumption of a two-fold natural dispersion in the water column), then the concentration of benzene in air at the water surface would be approximately 1 mg/m^3 (0.3 ppm by volume). This concentration, which falls between the time-weighted average (TWA) values for worker exposure (Table 5.24) set by the National Institute for Occupational Safety and Health (NIOSH) and the U.S. Occupational Safety and Health Administration (OSHA), will decrease rapidly as the benzene 'cloud' mixes turbulently as it moves downwind and is unlikely to lead to exposures of significant concern to human populations.

A seep on land will differ from the seabed seep scenario in that it presents a constant surface area to the atmosphere (no transfer to water) and evaporation will be the primary dispersion process. Using the same assumptions as for seeps from the seabed, then the concentration of benzene at the land seep surface would be approximately 2 mg/m^3 (0.6 ppm by volume). Again, rapid dilution will take place downwind minimizing the potential for harm to human health.

There are few data available for the Arctic region for comparisons between estimated and measured levels of contaminants in soil or in water related to oil and gas activities. Local spills on land may lead to high levels of PAHs in the permafrost, however soil ingestion by local populations from these sites is highly unlikely and few forage crops are likely to grow or be harvested for human consumption from spill sites in the Arctic. VOCs will not remain in or on the soil and exposure of human populations via soil will be nonexistent. Water in rivers and lakes near a local spill site may contain significant levels of PAHs and, in the short term, dissolved VOCs such as benzene, toluene, and xylene. Consumption of this water by local populations could lead to intakes of PAHs and occasionally some VOCs; however, in most cases where residues are high (close to the spill site), taste and odour will alert local populations to the contamination and consumption of this water will not occur or will be minimal.

5.5.3. Uptake, metabolism, and excretion

Uptake, metabolism, and excretion of oil and gas components and products are dependant on the substance (chemi-

cal or metal), the route of exposure, the length and concentration of exposure, and the general health status of the individual being exposed. Park and Holliday (1998) and the American Petroleum Institute (API, 2001) provided useful summary data for several products considered of concern to human health (see Table 5.26). In general, crude oil contains alkanes, aromatics compounds, sulphur-containing compounds, metals, and radionuclides (see section 5.1.1.1).

5.5.3.1. Crude oil components and some exploration and extraction products

Crude oil exposure is nearly always dermal and occasionally pulmonary (aerosolized oil). Exposure to fresh crude oil can lead to uptake of both the volatile and non-volatile fractions; exposure to weathered crude oil (from 6–8 hr after a spill) normally leads to the uptake of non-volatile fractions only as the volatile fractions have largely evaporated. Boogaard and van Sittert (1995) reported that PAHs are absorbed dermally by petrochemical workers as determined by the presence of a PAH metabolite (1-hydroxypyrene) in urine. The Canadian Council of Ministers of the Environment has undertaken extensive evaluation of petroleum hydrocarbons in soil and provided summary data for dermal uptake of 20 aliphatic and aromatic compounds, and modelling data to assess transfer risks (CCME, 2000, 2001a,b). These technical data are valuable although uptake of petroleum hydrocarbons from Arctic soils is likely to be very low due to limited human exposure (few land-based spills, snow cover, low population density, and little potential for intake through consumption of vegetation).

Exposure to volatile aromatics, sulphurous compounds and particulates (from combustion) is almost exclusively via the pulmonary route. Uptake of gases across the lung is rapid. Uptake of substances such as *n*-nonane, trimethylbenzene and trimethylcyclohexane by rats was similar and rapid regardless of whether inhaled in a mixture or individually (Eide and Zahlsen, 1996). Clearance rates for these volatile compounds from tissue were rapid once exposure ceased, i.e., after a few hours. Human studies tend to confirm these findings (Park and Holliday, 1998). Their review of published uptake and clearance rates for a variety of volatile substances (e.g., toluene, trimethylbenzenes, hexane) indicated that uptake was very rapid and clearance often multi-phasic, taking from a few minutes, to a few hours, to a few days for complete elimination. The multiple phases may be related to de-gassing through the lung (very rapid), metabolism of the substances (moderately rapid), and possible slow release of the substances from fat stores.

Oral exposure to crude oil can lead to uptake of some PAHs. Most are metabolized slowly, conjugated, and excreted in urine or faeces. Very little volatile fraction is normally absorbed following oral exposure of crude oil for two reasons. First, volatile substances only remain in crude oil for short periods after a spill, so exposure via the oral route is rare. Second, ingestion of fresh oil or volatile oil products are likely to lead to vomiting, reducing the potential for uptake by reducing the likelihood of their entry into the intestine.

Metals are usually absorbed rapidly by the intestine. Mercury binds to protein and cadmium is stored in the bone and kidney. Metals are not metabolized *per se*, but can be slowly mobilized and eliminated in urine and faeces. Persistent chemicals like PCBs are only likely to be found in food, are rapidly absorbed by the intestine, and have a strong affinity for lipids. They are metabolized very slowly and have a long half-life (AMAP, 1998, 2003). Radionuclides are also absorbed via the intestine when ingested in water or food. Decay rates are chemical- and isotope-specific as are elimination rates in urine and faeces.

5.5.3.2. Concluding comments

Uptake by skin of PAHs found in crude and refined oil is relatively rapid; however, this does not lead to extensive systemic levels. Short duration exposures of skin can be sufficient to create irritation (see section 5.5.4). Intestinal absorption of PAHs from water and food is partial and metabolism and excretion are moderate such that PAHs are not retained in the body for long. Uptake by skin and pulmonary epithelium of volatile hydrocarbons is very rapid leading to direct internal exposure of the major organs, blood, and the brain. Metabolism and elimination are also rapid. Small particles from combustion sources are inhaled and will often remain in the lung (size range $PM_{2.5}$ to PM_{10}). PAHs and metals adsorbed to the surface of particles are capable of crossing the pulmonary epithelium. Metals are usually stored in bone or bound to protein; they are excreted at a moderate rate.

5.5.4. Toxicity of oil and gas components for human populations

Several review articles describe the toxicology and carcinogenicity of oil and gas, their products and constituents, and the products related to exploration and extraction. The constituents usually covered include individual PAHs, volatile short chain aliphatics (carbon number C_4 to C_{10}) and aromatics, individual gases and metals (Park and Holliday, 1998; IARC, 1982, 1987, 1989; API, 2001; ATSDR, 2007). The findings in these reviews are based on laboratory animal studies, human trials, and occupational data.

This text is not intended to provide an exhaustive review of the effects of all the various components of oil and gas. However, the major components to which local human populations may be exposed as a result of proximity to oil and gas activity are addressed. Reference is made to gasoline and other volatile refined fuels which are not considered major 'upstream' components of oil and gas activities, because they can be released as a result of accidents while handling storage barrels or fuelling vehicles, snowmobiles, helicopters, planes, and generators with these products.

5.5.4.1. Crude oil

Crude oil and especially lighter fractions of crude oil appear to cause dermatitis which can range from mild redness of the skin to kerototic skin lesions. Responses of this type have been seen in clean-up workers following the *Exxon Valdez* spill and others (Appendix 5.1) who may or may not have used or had adequate access to protective equipment. Additional local population exposure data are presented later in this chapter. Park and Holliday (1998) presented the case for PAHs (or their oxidized products formed during metabolism) as the likely agents responsible for skin irritation. Sunlight may increase the effect through the capacity of hydrocarbons to photosensitize skin; a phenomenon observed in humans and experimentally demonstrated in animals (Gomer and Smith, 1980). It may be that crude oil and some of its component PAHs (i.e., anthracene) can inhibit the immune system of the skin in the presence of UV light, but not unless both exposures occur simultaneously (Burnham and Rahman, 1992).

Crude oil can be inhaled if it is aerosolized due to high wind and wave action or through high pressure hose shore-

Table 5.27. International Agency for Research on Cancer evaluations of some oil and gas activities and products (IARC, 1987, 1989).

Evaluation	Classification	Evidence
Benzene	1	*Sufficient* evidence of leukaemia in humans *Limited* evidence of carcinogenicity in animals
Petroleum refining	2A	*Limited* evidence of skin cancer and leukaemia in humans Mix of *sufficient* and *limited* evidence of carcinogenicity in animals to various petroleum products
Gasoline	2B	*Inadequate* evidence of carcinogenicity in humans *Limited* evidence of carcinogenicity in animals exposed to unleaded automotive gasoline
Marine diesel fuel	2B	*Limited* evidence of carcinogenicity in animals
Residual (heavy) fuel oils	2B	*Sufficient* evidence of carcinogenicity in animals
Crude oil	3	*Inadequate* evidence of carcinogenicity in humans *Limited* evidence of carcinogenicity in animals
Jet fuel	3	*Inadequate* evidence of carcinogenicity in humans *Inadequate* evidence of carcinogenicity in animals
Light diesel fuels	3	*Inadequate* evidence of carcinogenicity in humans
Distillate (light) fuel oils	3	*Inadequate* evidence of carcinogenicity in humans (fuel oils) *Limited* evidence of carcinogenicity in animals (fuel oil No. 2).

IARC Classifications: 1: carcinogenic to humans, 2A: probably carcinogenic to humans, 2B: possibly carcinogenic to humans, 3: not classifiable as to its carcinogenicity to humans.

line clean-up operations. If appreciable amounts of aerosolized oil enter the lungs this inhaled oil can cause severe pneumonitis and death. These outcomes are relatively rare.

Various animal studies involving skin exposure to crude oil or distillates reviewed by the International Agency for Research on Cancer (IARC, 1989) led it to conclude that there was 'limited evidence' of carcinogenicity of crude oil in experimental animals and 'inadequate evidence' for the carcinogenicity of crude oil in humans (Table 5.27).

5.5.4.2. Refined fuel products

Several refined fuels (e.g., gasoline, jet fuel, kerosene, diesel) are used routinely to support activities at well and production sites and can be spilled.

Gasoline is made up primarily of volatile hydrocarbons of carbon number C_4 to C_{10}. While gasoline has a number of additives (e.g., dyes, anti-knock agents, detergents) these tend to occur in low concentrations (<0.1%). Gasoline can cause skin dermatitis, photosensitivity, and defatting. Vapour levels >500 ppm can cause eye, nose and throat irritation, headaches, dizziness, and mental confusion. Very high concentration vapours (20 000 ppm) are fatal after a few minutes (API, 2001). IARC (1989) concluded that there is 'limited evidence' that unleaded gasoline causes cancer in animals; however, because of the presence of a known human carcinogen (benzene) in gasoline, IARC classified unleaded gasoline as a 'possible human carcinogen' (Table 5.27).

Occupational exposure to gasoline has been associated with many signs of neurotoxicity including significant effects on intellectual capacity, modifications of psychomotor and visual-motor functions, and immediate and delayed memory. Other gasoline-induced neurotoxic effects (ataxia, tremor, acute or sub-acute encephalopathy syndrome) are ascribable to intentional use (gasoline sniffing) and not to occupational exposure (Caprino and Togna, 1998). Overall, the effects of inhaled gasoline fumes on non-occupationally exposed and non-intentionally exposed individuals are difficult to predict because of variations in gasolines and their additives, individual exposures, duration of exposures, confounding variables, and pre-existing health status.

Jet petroleum, kerosene, and heating oil are considered middle distillates; they are less volatile and explosive than gasoline even though some contain many of the same additives as gasoline (API, 2001). In general, short-term exposure to these fuels causes similar effects to exposure to gasoline, that is, vapours cause irritation of the mucous membranes of the respiratory system, nausea, headaches, dizziness, confusion and, at high concentrations, unconsciousness and death. Mists can cause eye irritation and conjunctivitis and skin irritation.

Ritchie et al. (2002) provided an extensive overview of the effects of the jet fuel JP-8 and concluded that "*there is little or no evidence that acute or long-term JP-8 exposures result directly in cancer, serious organic disease or death in humans*". However, from their review of occupational data and animal studies, they concluded that subtle effects are likely to occur and be persistent over prolonged periods of low-dose exposure. The effects of most concern include persisting changes in brainstem/cerebellar systems, altered neurobehavioral performance capacity and persisting immuno-suppression. There is no consistent evidence that JP-8 itself is carcinogenic to humans or laboratory animals (Table 5.27).

5.5.4.3. Oil and gas components

5.5.4.3.1. Polycyclic aromatic hydrocarbons

Apart from the potential for anthracene and potentially other hydrocarbons to photosensitize skin (section 5.5.4.1) the major health concern over PAHs is their potential to act as tumour promoters and carcinogens (CEPA, 1994). Exposure to airborne PAHs is thought to represent a significant human cancer risk (NTP, 2005) particularly for the oral cavity tissues, lungs, skin, and possibly kidneys. Even though airborne PAH emissions from oil and gas combustion (a flare or use of fuels for power generation) may be different in nature and far lower in concentration, IARC (1987) classified PAH emissions from coke-ovens (known to contain a wide array of PAHs) as carcinogenic to humans. A recent report has also indicated that coal gasification workers have an increased risk of respiratory and urinary system cancers as a result of airborne PAHs (Bosetti et al., 2006). Table 5.28 identifies non-carcinogenic and carcinogenic properties of eleven PAHs. Some health based reference values for the most common PAHs are provided in Table 5.29.

Table 5.28. Potential health effects of selected PAHs (adapted from USAID, 2000).

Chemical	Non-carcinogenic effects (target organs/critical effects)	Potential carcinogenic effects in humans
Benzo[*a*]pyrene	*Oral Exposure (in animals)*: Bone: bone marrow changes and death; Reproductive system: reduced fertility, sterility, foetal death /malformations; Immune system: suppression	Cancer through several routes of exposure: Stomach, throat and mammary tumours. USEPA B2: 'probable human carcinogen'
Fluoranthrene	*Oral Exposure (in animals)*: Blood: effects on red blood cells; Kidney: degeneration; Liver: lesions	No human data [a]
Phenanthrene	*Oral Exposure*: No data for human/animal exposure *Other routes of exposure(animals)*: Skin: sensitivity to light Liver: liver changes	No human data [a]
Pyrene	*Oral Exposure (in animals)*: Kidney, liver: degenerative changes Blood: slight haematological effects	No human data [a] USEPA B2: 'probable human carcinogen'
Dibenz[*a,h*]anthracene	*Oral Exposure*: No data available. *Other routes of exposure (animals)*: Lymph system: tissue changes. Liver, kidneys, and heart: degenerative changes; Immune system: suppression	No human data specifically linking exposure to human cancers, but it is a component of mixtures that have been associated with human cancer. *In animals*: oral exposure linked to mammary/ stomach cancer USEPA B2: 'probable human carcinogen'
Indeno[1,2,3*cd*]pyrene	No data for humans. Immune system: suppression in animals	No human data specifically linking exposure to human cancers, but it is a component of mixtures that have been associated with human cancer. *In animals*: lung cancer USEPA B2: 'probable human carcinogen'
Benzo[*k*]fluoranthene	No data for human/animal exposure	No human data [a] USEPA B2: 'probable human carcinogen'
Benzo[*b*]fluoranthene	No data for human/animal exposure	No human data [a]
Benzo[*g,h,i*]perylene	No data for human/animal exposure	No human data [a]
Chrysene	Immune system: carcinogenic PAHs induce immuno-suppression in laboratory animals	No data specifically linking exposure to human cancers, but it is a component of mixtures that have been associated with human cancer. *In animals*: liver cancer USEPA B2: 'probable human carcinogen'
Benz[*a*]anthracene	*Oral Exposure:* Circulatory system: anaemia; Lymph and intestinal system: degenerative changes; Reproductive system: damage to/loss of fertility, foetal mortality	No human data [a] USEPA B2: 'probable human carcinogen'

[a] No human data specifically linking exposure to human cancers, but substance is a component of mixtures that have been associated with human cancer.

5.5.4.3.2. Volatile compounds

A variety of volatile compounds are rapidly released from crude oil, refined oil products, and well gas when they come into direct contact with air mostly through spills. Some volatile compounds are added to refined products to improve their combustion or storage characteristics, increasing potential exposure concentrations following spills. Occupational exposure limits for several volatile hydrocarbons are provided in Table 5.24.

Benzene, toluene, xylene and trimethylbenzenes

Benzene, a significant constituent of crude oil and most petroleum fuels is the most toxic of all the volatile aromatic compounds. Like most volatile hydrocarbons, low dose benzene exposure (100–250 ppm) results in fatigue, headaches, vomiting, poor coordination, and muscle weakening (API, 2001). At high doses, humans show mental confusion, exhilaration, and lack of self-control and report lasting problems with fatigue, general confusion, headaches, insomnia and skin paresthesia. Baelum et al. (1985) reported neurobehavioral effects related to impaired visual perception, colour discrimination, and the ability to perform multiplication calculations.

Benzene exposure has lead to cancer in animal studies (IARC, 1989) and has been associated with increased risk of leukaemias among industrial workers (Rushton and Romaniuk, 1997; Verma et al., 2000). IARC (1982, 1989) classified benzene as carcinogenic to humans. The mechanism of action appears be associated with direct interaction with blood cell formation. In their review, Park and Holliday

Table 5.29. Health-based values in Canada for substances associated with oil and gas activities.

Component	Non-cancer outcome, acceptable daily exposures [a] (Health Canada, 2004b)	Cancer outcome [a,b], reference values (Health Canada, 1996)	Carcinogenicity class [c] (Health Canada, 1996)
Benzene		TC_{05}: 15 mg/m^3	I
Benzo[*a*]pyrene		TC_{05}: 1.6 mg/m^3	II
Benzo[*b*]fluoranthene		TC_{05}:26.7 mg/m^3	II
Benzo[*j*]fluoranthene		TC_{05}: 32.0 mg/m^3	II
Benzo[*k*]fluoranthene		TC_{05}: 40.0 mg/m^3	II
Indeno[1,2,3]pyrene		TC_{05}: 13.3 mg/m^3	II
Toluene	TDI: 0.22 mg/kg bw/d TDC: 3.8 mg/m^3		IV
Xylene (mixed isomers)	TDI: 1.5 mg/kg bw/d Provisional TDC: 0.18 mg/m^3		IV
Barium	TDI: 16 ug/kg bw/d		
Cadmium	TDI: 0.8 ug/kg bw/d	TC_{05}: 5.1 μg/m^3	II
Chromium (total)	TDI: 1 ug/kg bw/d	TC_{05}: 4.6 μg/m^3	-
Chromium (IV)	TDI: 1 ug/kg bw/d	TC_{05}: 0.66 μg/m^3	I
Lead	TDI: 3.6 ug/kg bw/d		
Mercury	TDI: 0.3 ug/kg bw/d		
Uranium (non-radiological)	TDI: 0.6 ug/kg bw/d		

[a] TDI: Tolerable Daily Intake (total daily intake of a substance on a body weight basis which is not expected to have a harmful effect over a lifetime of exposure), TDC: Tolerable Daily Concentration (exposure concentration in air which is not expected to have a harmful effect over a lifetime of exposure), TC_{05}: Tumourigenic Concentration 05 (concentration in air or water associated with a 5% increase in the incidence of, or deaths due to, tumours considered to be associated with exposure); [b] these values are 'effect levels' and cannot be compared directly with TDI or TDC values for non-carcinogens unless an uncertainty factor is applied (such as 5000 or 50 000) to place them in a risk range of 10^{-5} or 10^{-6}; [c] CEPA Carcinogenicity Classification: I: carcinogenic to humans, II: probably carcinogenic to humans; IV: unlikely to be carcinogenic to humans.

(1998) indicated that low-dose exposure appears to result in direct and reversible haematotoxic effects and that sustained higher dose exposure leads to irreversible bone marrow damage, aplastic anaemia and acute myelogenous leukaemia as a result of the conversion of benzene to toxic phenolic and quinone metabolites. It is not known if there are differences in risk to health associated with multiple lower-dose exposures over a long period of time, compared to fewer, higher dose exposures over a shorter period of time. The concentration in air associated with a 5% increase in incidence of, or mortality due to, tumours (TC_{05}) for human populations has been listed as 15 mg/m^3 (Health Canada, 1996). A summary of the carcinogenic effects of benzene in animals and human populations is available in a Canadian assessment report (CEPA, 1993).

Toluene causes headaches, eye and throat irritation, lacrimation and nasal discharge, headaches, drowsiness, dizziness and weakness at 100 ppm and above (API, 2001). At 800 ppm the onset of the effects is more rapid and severe. Confusion, loss of coordination, and unconsciousness have also been reported after exposure to very high concentrations (>15 000 ppm); sustained exposures of 200–500 ppm for up to three weeks have caused poor coordination, memory loss, and slow reaction times. Effects appear to be reversible following a no-exposure recovery period.

Xylene, like benzene and toluene, causes eye, nose and throat irritation.

Table 5.30. Potential effects on humans of hydrogen sulphide exposure (USAID, 2000).

Concentration, ppm [a]	Resulting condition/effect on humans/exposure guidelines
0.05–0.013	Odour threshold.
0.3–1.0	Detectable by most persons by sense of taste rather than by odour.
3–5	Easily detectable, moderate odour.
10	Beginning eye irritation, maximum concentration.*Permissible exposure level, 8 hrs OSHA, Cal-OSHA, ACGIH.*
15	*OSHA 15 minute permissible exposure level.*
20–30	Strong unpleasant odour, but not intolerable, minimum concentration causing coughing and immediate irritation to the eyes. *Maximum concentration allowable for short exposure CCR Title 8 sec-5155 20ppm Excursion 10 min/8 hrs.*
50	Pronounced irritation of eyes, throat, and lungs. *Maximum Concentration allowable for ceiling limit (CCR Title 8 sec-5155).*
100	Coughing, eye irritation, loss of sense of smell after 2 to 5 minutes, may sting eyes and throat.
200	Marked conjunctivitis (eye inflammation) and respiratory tract irritation after one hour of exposure.
500–700	Loss of consciousness and possibly death in 30 minutes to an hour. Immediate artificial resuscitation will be necessary.
700–1000	Rapid unconsciousness, cessation (stopping or pausing) of respiration and death. Permanent brain damage may result. Artificial resuscitation and oxygen must be given.
1000–2000	Unconsciousness at once, with early cessation of respiration and death in a few minutes. Death may occur even if individual is removed to fresh air at once. Brain damage will occur.

[a] Values from: American National Standards Institute (ANSI), Standard No. Z37.21972; California Occupational Safety and Health Association (C-OSHA) GISO sec-5155; and American Conference of Governmental Industrial Hygienists (ACGIH).

Trimethylbenzenes (1,3,5-TMB; 1,2,4-TMB, 1,2,3-TMB) are skin and eye irritants, and cause headaches, drowsiness, and anaemia. Animal studies indicate that trimethylbenzenes affect the central nervous system with excessive exposure causing neurological and behavioural effects (API, 2001).

5.5.4.3.2.1. n-Hexane

n-Hexane is the most toxic of the alkanes. If ingested, it causes nausea, vertigo, and bronchial and intestinal irritation. Numbness of limbs, confusion, nausea, headaches, throat and eye irritation, and dizziness have been reported when inhalation exposure is at levels greater than 1000 ppm; below 500 ppm no effects have been reported. More significant is the effect of *n*-hexane on peripheral nerves. Repeated exposures of workers in a non-oil related industry at 400–600 ppm lead to muscle weakness, sensory disturbances, and limb pain, although these signs and symptoms appear to be slowly reversible following cessation of exposure (Park and Holliday, 1998). The causal agent appears to be 2,5-hexanedione, a metabolite of *n*-hexane.

5.5.4.3.2.2. Hydrogen sulphide and its combustion products

Hydrogen sulphide is immediately lethal in concentrations greater than 2000 ppm (USAID, 2000). Lethality appears to be due to anoxia in brain and heart tissue. H_2S interferes with oxidative metabolism, which is the primary energy source for cells. A summary of effect levels for H_2S are shown in Table 5.30. Occupational exposure limits for two sulphur-containing gases and for some volatile components of well gas are shown in Table 5.24.

Sustained low-level exposure to H_2S results in irritation of the mucous membranes, particularly those of the respiratory tract and the eyes. Pulmonary oedema occurs at sub-lethal concentrations (250–500 ppm) if there is sufficient exposure before consciousness is lost. Pulmonary oedema has been reported after long-term exposure to levels as low as 50 ppm.

Concentrations >50 ppm can cause changes in visual acuity and perception of blue or rainbow colours around lights, followed by very painful inflammation and even ulceration in severe cases. Sub-chronic (less than a life-time) exposure studies with mice have shown that exposure to 80 ppm causes nasal lesions. Olfactory sensation is lost at 150–200 ppm; hence, the characteristic odour of rotten eggs is not sufficient to warn of lethal exposures. At concentrations ≤150 ppm, symptoms such as the inability to think logically and incoherence have been reported. Victims who have recovered from exposure to H_2S report neurologic symptoms such as headache, fatigue, irritability, vertigo, and loss of sexual desire.

Symptoms of chronic exposures at low levels are conjunctivitis (eye infections), headache, attacks of dizziness, diarrhoea, chronic bronchitis, a grey-green line on the gums, and loss of weight. Hessel et al. (1997) reported that there were no lasting pulmonary function or chronic symptoms found in workers who had been exposed to high enough concentrations to cause reportable symptoms; however workers who had experienced a 'knockdown' exposure (severe enough to cause a loss of consciousness) did have lower spirometric values, shortness of breath during moderate exercise, wheezing, and chest tightness. Nervous system disorders, including paralysis, meningitis, and neurological problems, have been reported but not confirmed. There are insufficient data to unequivocally state that mutagenic, carcinogenic, teratogenic, or reproductive effects do not occur.

Cancer has not been associated with short-term or long-term exposure to H_2S. Schechter et al. (1989) did not identify any excess cancers in a community-based study in Alberta, Canada, downwind of a sulphur-releasing gas plant. However, Lewis et al. (2003) did report a statistically significant increase in mortality among employees in a large Canadian cohort study of a petroleum company associated with long-term exposure to H_2S.

If well fluids containing H_2S ignite, particulates high in sulphur and PAHs can contaminate the air. SO_2 can result from the combustion of H_2S, can be emitted from the well site, and can come from fresh oil spills. It can be detected by smell at concentrations of 780 μg/m^3 (0.3 ppm) or more. Short-term exposure to 2600 μg/m^3 (1 ppm) affects health. Long-term exposure of 300–400 μg/m^3 (0.12–0.15 ppm) for more than 24 hr has been associated with increased hospital admissions and emergency room visits and excess mortality, particularly among the elderly and those with pre-existing cardio-pulmonary conditions. Other studies have indicated an increased prevalence of acute and chronic pulmonary impacts (respiratory distress and reduced lung function) in adults and children exposed to a mean level of 100 μg/m^3 (38 ppb) (Health Canada, 2004a). Health Canada (2004a) issued acceptable exposure ranges for residential indoor air of 50 μg/m^3 for long-term exposure and 1000 μg/m^3 for short-term (5-min average) exposures to SO_2. The Northwest Territories of Canada (Health Department) and the Alberta Department of Health have set ambient air quality standards or guidelines for population health for several well head gases (Table 5.31).

5.5.4.3.3. Drilling fluids

Drilling fluids can contain metals, radionuclides, and volatile hydrocarbons due to the drilling muds used and the drill cuttings and oil brought up in the ejected fluids (sections 5.5.2.1 and 5.5.2.3).

5.5.4.3.3.1. Metals

The toxicity of mercury and cadmium have already been extensively evaluated by AMAP (AMAP, 1998, 2003, 2004b, 2005; Dewailly and Weihe, 2003). Mercury, in its inorganic form and as organic methyl mercury, causes significant irreversible neurological damage and developmental damage in children exposed *in utero* and possibly in early childhood. It can also act as a precursor for cardiovascular damage at low concentrations. Cadmium can affect kidney function and is of concern as a result of exposure through consump-

Table 5.31. Ambient air quality guidelines for some well head gases (μg/m^3) for three exposure periods: an hour, a day and a year (CCME, 2004a). The table shows the lowest value used by Provinces or Territories provided for each gas. Other Provinces or Territories may have higher values.

Gas	1 hr	24 hr	Annual	Source
SO_2	450 (170 ppb)	150 (60 ppb)	30 (10 ppb)	NWT ambient air quality standards
NO_2	400 (160 ppb)	200 (80 ppb)	60 (2 ppb)	Alberta ambient air quality guidelines
H_2S	14 (10 ppb)	4 (3 ppb)		Alberta ambient air quality guidelines
Carbon disulphide (CS_2)	30 (10 ppb)	-		Alberta ambient air quality guidelines

tion of some animal organ meats (liver, kidney) and through the use of tobacco. While mercury has not been shown to increase cancer risk in animal or human studies (AMAP, 1998), cadmium has been assessed as a type II carcinogen ('probably carcinogenic to humans') in Canada based on animal studies (Health Canada, 1996) (Table 5.29).

Barium can damage the heart and cardiovascular system and is associated with high blood pressure in rats exposed to high levels during their lifetimes. The U.S. reference dose for barium is 0.07 mg/kg body weight per day. It is non-classifiable in terms of human carcinogenicity.

5.5.4.3.3.2. Radionuclides

The effects of several radionuclides have also been evaluated by AMAP (AMAP, 1998, 2004a). Levels of naturally-occurring radioactive materials are sufficiently low in biota, vegetation, air, and water that they do not pose a significant radiation risk or chemical risk to community health. While drilling muds and produced water may increase local concentrations marginally, the levels are still low and of negligible human health concern (Rajaretnam and Spitz, 2000).

5.5.4.3.3.3. Oil-based drilling muds

Eide (1990) examined exposure scenarios and effects of inhalation of low aromatic oil-based drilling muds found as vapours or aerosols and concluded that effects on the central nervous system are present at high concentration and that current efforts to reduce worker exposure to <100 mg/m^3 (time-weighted average) are of significant value. There is insufficient information to evaluate long-term exposure to low aromatic oil-based muds.

5.5.4.4. Concluding comments

Overall exposures of local populations to emissions from oil and gas activities in most areas of the Arctic are likely to be very low. Since risk is a function of both the amount of the exposure and the toxicity of the substance under study, it is relevant for this assessment to examine the toxic effects of the components of, and products related to, oil and gas produced in the Arctic.

Several components and products of oil and gas have significant toxicity and the potential to affect health dependent on exposure. Crude oil and various components of crude oil can cause acute and chronic health effects. The impacts of short-term (acute) and low-concentration exposures to crude oil and oil and gas components and products tend to be transient and reversible (acute effects). Longer-term (sub-chronic and chronic) exposures to low and moderate concentrations can lead to sustained damage (chronic effects) most of which is sub-lethal. Acute, high-concentration exposures to the volatile components of oil (especially benzene, toluene, xylene) and gases (primarily H_2S) can be fatal. Sustained exposures to PAHs and benzene also have the potential to cause cancer after a latency period of 10–20 yr. Sustained exposures of local populations in the Arctic to relevant concentrations and via relevant routes for inducing cancer are very unlikely.

Crude oil itself has been shown to produce toxic effects in both humans and animals under specific exposure scenarios. Dermatitis is a concern but tends to be limited to direct contact with oil (on drill sites, as a result of a spill, during clean-up operations) and the effects are transitory. Once exposure ceases, skin inflammation dissipates in a week to ten days. Photosensitization of skin which has been exposed to oil will cause UV-related skin burns to be more severe during periods of direct sunlight and high UV-radiation; however, these conditions are still reversible. This problem is primarily an issue for oil clean-up crews, which sometimes include members of local populations.

There is ample experimental animal and human occupational evidence showing that several days of moderate-level or several hours of high-level inhalation exposure to some individual petroleum components and gases can produce both acute and sub-chronic toxic effects. Most prevalent acute outcomes are eye, nose and throat irritations, headaches, and lassitude. These are common outcomes for volatile hydrocarbons and tend to be reversible once exposure ends. Although spills in the Arctic only occur rarely and there are few refineries or gas plants which may generate emissions, human exposure to these sources of volatile compounds could lead to toxic effects under some circumstances. Inhalation of oil droplets only occurs under rare conditions (oil spray or high wind in a spill or blowout zone) and has not been shown to have any lasting effects in the spill related studies reviewed.

Inhalation exposure resulting from flaring at the well head and open burning of spilled oil can increase exposures to particulate matter which can be readily inhaled and retained in the lungs. Particulates can contain high concentrations of bioavailable contaminants (PAH, sulphurous compounds, dioxins, furans, metals) and acids. Elevated exposures to particulates can cause adverse health effects, based on studies of urban populations, workers, and indigenous people in some tropical regions. Although sustained flaring is not common in most of the Arctic, efforts should continue to reduce and avoid flaring whenever possible. Open burning of spilled oil is also an unhealthy practice, especially if it occurs near communities and should also be avoided whenever possible.

Refined oil products used as fuel, such as gasoline and various diesel fractions, contain higher percentages of VOCs such as benzene, xylene, toluene, and *n*-hexane than are found in the parent crude oil. Exposures to these product fuels, which are used during routine well site operations and for running facilities and vehicles, are probably common. Sustained exposures cause the same effects as listed in section 5.5.4.3.2 because these refined products contain the constituent volatile organics evaluated above. All fuels should be handled with great care and inhalation of fuel vapours should be avoided whenever possible.

Despite their relative toxicity, radionuclides and metals coming specifically from oil and gas activities are unlikely to have measurable, if any, effects on the health of human populations in the Arctic. Amounts of these metals and radionuclides, which may surface in drilling waste, are very small and contribute little to overall environmental contamination and human exposure. Exposure of some Arctic populations to mercury is moderately high and is a health concern; however, these exposures come primarily from natural and anthropogenic sources (via long-range airborne transport) which are unrelated to oil and gas activity in the Arctic.

5.5.5. Effects of oil and gas activities on human populations

5.5.5.1. Cancer among oil and gas workers

Crude oil and some oil components and products (e.g., benzene) have been reviewed by the International Agency for Research on Cancer, the U.S. Environmental Protection Agency and Health Canada based on epidemiology findings

Table 5.32. Recent occupational health studies related to oil and gas activities and cancer.

Study group	Cancer outcome	SMR [a]	Source
Upstream workers (exploration, production, pipeline and other oil/gas transportation)			
Oil production workers – US producing sites. Cohort: part of 11 098 employees (1946–1980)	Thyroid	4	Divine and Barron, 1987
Pipeline workers – US pipeline locations. Cohort: part of 11 098 employees (1946–1980)	No excesses		Divine and Barron, 1987
Exploration, drilling, production and pipeline workers – a Canadian petroleum company. Cohort (updated): 4432 male employees (1964–1994)	No excesses		Lewis et al., 2000
Oil production workers – US producing sites. Cohort (updated): part of 22 124 employees (1946–1994)	Benign and unspecified neoplasms	1.52	Divine and Hartman, 2000
	Acute myelogenous leukaemia	1.92	Divine and Hartman, 2000
Downstream workers (gas and oil refineries, shipment and distribution of fuels)			
Refinery workers – two US refineries. Cohort: 14 149 employees (1950–1986)	Lymphatic and haematopoietic	1.5	Dagg et al., 1992
Refinery workers – three US refineries. Cohort: 25 321 employees (1970–1992)	Kidney	1.9	Shallenberger et al., 1992
Refinery workers – eight UK refineries. Cohort: 34 569 employees (1950–1989)	Malignant melanoma (all workers)	1.8	Rushton, 1993a
	Stomach/lung (labourers only)	1.5/1.1	Rushton, 1993a
Refinery workers – several US refineries. Cohort: 4585 employees (1973–1989)	No excesses		Tsai et al., 1993
Refinery workers – a US refinery. Cohort: 7720 employees (1948–1989)	Lymphatic and haematological	2.3	Tsai et al., 1996
	Lymphatic and reticulosarcoma	6.7	Tsai et al., 1996
Refinery workers – a US refinery. Cohort: 17 844 employees (1937–1987)	Bone cancer	2.08	Satin et al., 1996
	Acute lymphocytic leukaemia	2.60	Satin et al., 1996
	Benign/unspecified neoplasms	1.95	Satin et al., 1996
Refinery workers – an Italian oil refinery. Cohort (updated): 1583 employees (1949–1991)	Lymphoma	1.9	Consonni et al., 1999
	Leukaemia	2.25	Consonni et al., 1999
Refinery and petrochemical workers – a Canadian petroleum company. Cohort (updated): 9266 employees (1964–1994)	Mesothelioma	8.68	Lewis et al., 2000
Oil distributors – multiple locations in Cohort: 23 306	No excesses		Rushton, 1993b
Gasoline distribution/marketing – ships, shipping centres and land terminals in the US. Cohort: 18 135 (1946–1989)	No excesses		Wong et al., 1993
Marketing and distribution – including tank truck drivers and employees in petroleum marketing and distribution centres of a Canadian petroleum company. Cohort (updated): 6800 male employees (1964–1994)	Multiple myeloma	2.8	Lewis et al., 2000

[a] SMR: Standardized Mortality Ratio of 'observed deaths' over 'expected deaths'.

and animal toxicology studies and their findings are summarized in Tables 5.27, 5.28 and 5.29 respectively. A Canadian assessment of benzene (CEPA, 1993) gave an evaluation of cancer outcomes in workers in petrochemical industries. Ward et al. (1997) reviewed cohort mortality studies of petroleum industry workers published between 1987 and 1996. Table 5.32 provides a brief summary of studies relevant to this assessment with a focus on more recent findings. Studies of occupational cohorts working at upstream and downstream sites are of relevance to local population assessments in the Arctic because there is the possibility of exposure to the same chemical mixtures, albeit at lower concentrations, in population groups close to upstream oil and gas activities.

Most studies in Table 5.32 indicate that overall mortality (including overall cancer mortality) expressed as a standardized mortality ratio (SMR) of 'observed deaths' over 'expected deaths', was below that of referent groups, confirming that, in general, mortality in the oil and gas sector is below population norms. While there is a 'healthy worker effect'

to consider, this is encouraging for evaluations of exposed local populations.

While there are fewer studies of upstream oil and gas workers, there appear to be fewer reports of significant elevations of those cancers (kidney cancer, leukaemias) which have been reported over the past 25 years for the industry overall. Divine and Hartman (2000) updated their original assessment of production workers and found mortality and overall cancer mortality among production workers to be significantly less than the general population. Their finding of an increase in acute myelogenous leukaemia in white male workers (SMR 1.92, see Table 5.32) was restricted to those employed before 1940 (SMR 3.74) and employed in production and pipeline jobs for at least 20 years (SMR 2.65). A sub-analysis indicated that the highest SMRs were for three job categories: 'roustabouts' or unskilled labourers (SMR 2.76), pumpers (SMR 2.80), and pipeline personnel (SMR 2.47). Overall leukaemia among white male workers was not significantly different from that of the general population (SMR 0.97). These findings suggest that job activity, length of exposure, and early date of commencement of exposure (before 1940) are important factors and may be associated with increased exposure to benzene which has been associated with increased acute myelogenous leukaemia in refinery workers.

Raabe and Wong (1996) performed a meta-analysis of cohort studies from 43 refineries and several land-based terminals, ships, and distribution centres involved in petroleum distribution in the U.S.A and U.K. This meta-analysis provides a great deal of power (over 208 000 workers and 4.6 million person years) to detect cancer outcomes which might be related to exposure. The total leukaemia meta-SMR from this combined cohort was 1.02, indicating that there is no detectable difference in total leukaemia mortality compared to the referent population for these combined petroleum operations in the U.S.A and U.K.

Several large cohort studies have been updated and have been able to increase the observation period for the cohort. In several studies cited by Raabe and Wong (1996), the SMR values for total and individual leukaemias (acute and chronic myeloid leukaemias, and acute and chronic lymphocytic leukaemias) have decreased as updates have been undertaken (i.e., more deaths added to the review). The authors pointed out that there is sufficient evidence to indicate that benzene exposure can increase acute myeloid leukaemia in workers with at least 200 ppm-years of exposure (i.e., the mean daily exposure multiplied by the number of days at that exposure and divided by 365 days). They suggested that, based on studies with better exposure characterization, the risk of an increase in acute myeloid leukaemia is more likely to be associated with exposures in the 300–530 ppm-years range. The meta-analysis seems to indicate that there is no significant risk among the petroleum workers in this study (both upstream and downstream) of any of the other four leukaemia types.

A Finnish cohort study which examined more recently employed refinery workers (1971–1994) reported cancer incidence rates (not mortality) to be 12% below the referent populations (Pukkala, 1998). The study indicated a significant increase in the incidence of kidney cancer, primarily among male workers expressed as a standardized incidence ratio (SIR 2.8) and a non-significant doubling of the incidence of non-Hodgkin's lymphoma and non-melanocytic skin cancer among male 'blue collar' workers.

Poole et al. (1993) reported the results of a kidney cancer case control study. They examined 102 cases of a combined refinery cohort of more than 100 000 male employees. Each case was matched to four controls by decade of birth and refinery location and industrial hygienists were able to assign semi-quantitative ratings of exposures based on several hydrocarbon streams. None of the exposure categories had a positive association with kidney cancer. However an analysis of longest job held indicated that kidney cancer relative risks (RR) were higher for three groups of workers: labourers (RR 1.9), unit cleaners (RR 2.3), and receipt/storage/movements workers (RR 2.5).

Schnatter et al. (1996) examined benzene exposure and leukaemia in a case-control study. Each of the 14 cases had four matched controls (by year of birth and time at risk). Average benzene exposure concentrations were estimated to be between 0.01 and 6.2 ppm. The authors cautioned that the power of the study was low, however, and no significant association between cumulative exposure or duration of exposure was found.

5.5.5.2. Studies of bystander population effects

Epidemiology studies that examine community or population level effects in spill or exposure zones sometimes have sufficient power to identify health effects not routinely visible in individual animal or human clinical examinations. In remote regions however, populations are often small and sample sizes insufficient for robust analyses of the veracity of effect. This section examines what is known from investigations of accidents that resulted in oil spills or gas leaks and some gas and oil operational sites which might provide insight on population health effects in the Arctic. It also examines the combined and cumulative effects of Arctic stressors which can impinge on the health of Arctic populations within an oil and gas activity area. Application of these findings must take into account a common deficiency of ecological epidemiology studies, that is, the lack of direct exposure monitoring of individuals and the biological plausibility of the outcomes based on animal studies. Corroborating case-control or cohort studies are also important in drawing overall conclusions of effects.

5.5.5.2.1. Populations exposed to oil refineries or petrochemical industries

A study of the local population in the vicinity of petrochemical industries found that there was a significant elevation of liver cancer incidence in men in 16 areas adjacent to the industry compared to 16 matched areas without exposure to petrochemical industries (Yang et al., 1997). Other authors have reported a variety of cancers deaths (bladder, brain, bone, liver, lung) in young Taiwanese adults (Pan et al., 1994; Yang et al., 1997, 1999) and leukaemia and other cancers in children in the U.K. (Knox and Gilman, 1997, 1998), all living near to petrochemical effluents. On the other hand Wilkinson et al. (1999) reported, based on an extensive study of regions in the U.K. in the vicinity of oil refineries, that there were no statistical associations between proximity to an oil refinery and either leukaemia or non-Hodgkin's lymphoma in children.

Like the epidemiology studies carried out with oil and gas industry cohorts, most non-occupational population studies have SMRs or RRs that vary between studies depending on the characteristics of the population considered (e.g., age, size, location, length of exposure). Non-occupational populations are often more mobile, have poorly characterized exposures, and may be in poorer general health

compared to occupational (worker-based) study populations, making interpretation of results more difficult.

There is only limited evidence of effects in populations living in the Arctic. Population health status is reported to have declined near a sub-Arctic refinery in Novokuibyshevsk due to exposure to high ambient air pollution (Tsunina, 2002).

5.5.5.2.2. Populations exposed to natural gas refineries and sour gas emissions

Arctic populations are not likely to be exposed to significant amounts of sour gas as most natural gas from the Arctic is sweet gas (i.e., low sulphur, section 5.5.2.1.2). Where H_2S is encountered, best available technology and good oil field practice for producing and processing fluids containing H_2S should be applied. Discussion of the effects of sulphur-containing gases on local populations is, nevertheless, relevant for this assessment because some areas of the Arctic (e.g., parts of Arctic Russia and the southern parts of the Northwest Territories, Canada) do have sour gas.

Dales and colleagues conducted several analyses of a large cohort of individuals living in the vicinity of a natural gas plant near Pincher Creek, Alberta (Dales et al., 1989a,b; Spitzer et al., 1989). Public complaints about health impacts were reported from 1958 until the plant was decommissioned in 1983. Emissions were primarily SO_2 but H_2S was also present. Strong response rates among the cohorts selected supported the following conclusions. Lung function measures were similar among the control and exposed groups. More people in the exposed area reported respiratory symptoms than in the unexposed area (50% more in the 5–13 years of age group, and 50% more in the group that had never smoked and were 14 years of age or older). After dividing the exposed group into high and low exposure subgroups, 50% more of the younger people reported symptoms, and 10% fewer of the older people reported symptoms in the high exposure subgroup (Dales et al., 1989a).

Assessment of other health outcomes indicated that there were no significant differences between exposed and unexposed groups with respect to mortality rate, cancer incidence, reproductive problems, major ailments, levels of arsenic or other metals in hair, or respiratory function (Spitzer et al., 1989; Schechter et al., 1989). Nor was there apparent selective migration for health reasons among those who had moved out of the exposure area.

Among healthy subjects, partially taken from the same study areas, Dales et al. (1989b) compared reporting of respiratory symptoms against the results of the Ilfeld Psychiatric Symptom Index (PSI). This index measures anxiety, anger, depression, and cognitive disturbance. In this study there was an association between the adjusted odds ratio for respiratory symptoms (ranging from 1.1–2.2) and every 10% increase in the PSI scores. The authors concluded that psychological status was an important factor in reporting of respiratory symptoms and should be taken into account when interpreting questionnaire-based epidemiological studies.

Kewen et al. (1985) reported that there have been significant decommissioning issues related to a natural gas plant site in Pincher Creek, Alberta. Groundwater near the sulphur plant and storage area had sulphate concentrations of up to 2450 mg/L whereas the maximum acceptable concentration in the Guidelines for Canadian Drinking Water Quality (Table 5.12) is 500 mg sulphate/L and less than 50 µg sulphide (as H_2S)/L based on taste and odour. Various heavy metals and organic compounds had accumulated in soils and sediments within some of the former processing and waste collection areas. Soils in the sulphur storage and runoff drainage areas had been affected by acidic runoff. Hydrocarbon residues including paraffins, phenols, and trace organics such as PAHs, benzenes, nitrogen-substituted compounds, and phthalates occur in groundwaters beneath these areas. Modelling of the contaminants in the worst area did not indicate that there would be much movement into groundwater or other surface waters.

Unrelated to the Pincher Creek studies, but relevant to the effects in the field of exposure of mammals to sulphur gas, Waldner et al. (1998, 2001a,b) examined the effects on cattle health of proximity to gas flares (both sweet and sour gas flares), inactive and active gas and oil sites, pumping/compressing stations, and processing plants. No association was found between cattle productivity and exposure of cattle to a sour gas pipeline leak (Waldner et al., 1998). There were occasional increased risks of non-pregnancy associated with proximity to flaring batteries (combined sweet and sour gas and sour gas alone), active gas wells, and large field facilities, but these varied from year-to-year (Waldner et al., 2001a). Abortion risk was not associated with exposure measures; however increased stillbirth and calf mortality (for only two of the four years of the study) were associated with sour gas flaring. Waldner et al. (2001b) did not find any association between sulphur deposition and cattle productivity, except for cumulative exposure to total sulphuration which was associated with decreased productivity. A large study has just been completed by the Western Interprovincial Science Studies Association and no associations were found between the measured exposures and most of the health outcomes (WISSA, 2006). However, at the highest exposure level, there may have been a slight effect on calf mortality (1–2% above background). Although these findings all concern cattle, the data are not strongly suggestive of significant adverse health outcomes which might be of relevance to human populations living downwind of sour gas sources in the Arctic.

5.5.5.3. Biological effects of oil spills on local populations

Two spills on water in the sub-Arctic (tanker accidents involving the M.V. *Braer* and the *Nakhodka*) have provided useful data on the health impacts of these accidents on local populations. Other spills, including a large spill on land in the Arctic (a leaking pipeline in the Komi Republic, Russia) provide information on sustained effects. The key events and findings are summarized in Appendix 5.1. These reports are additional to those already provided on psycho-social effects of the *Exxon Valdez* spill in Alaska, adverse health effects of spills in Ecuador, and the non-peer reviewed information related to contamination of the Tyumen Oblast and the Nizhnevartovsk District in the Khanty-Mansi Autonomous Okrug (see Appendix 5.1 and AMAP, 1998).

Health studies on the local communities in the Shetland Islands following the spill from the M.V. *Braer* are unusual because they focus on exposure of the general population to aerosolized oil and VOCs caused by high winds and stormy seas. Volatile hydrocarbons, oil droplets, and water-oil emulsions were blown inland and appeared on soil, crops, buildings, and animals and led to exposure of town residents who were not part of the clean-up crews.

Campbell and colleagues (Campbell et al., 1993, 1994; Crum, 1993) conducted thorough investigations immediately after the M.V. *Braer* spill (two days after the biggest release of oil when the ship broke apart) and six months later. Hydrocarbon levels on the day the *Braer* broke up were estimated to

be 6.3 ppm and the benzene levels 0.07 ppm; however, the day immediately before and the day immediately after the break-up, hydrocarbons and benzene levels were 0.3 and 0.02 ppm (day before), and 0.07 and 0.02 ppm (day after), respectively (Campbell et al., 1993). These benzene levels are 3-fold higher than the Norwegian air quality guideline (Table 5.25) and less than 10-fold lower than the 8-hr time-weighted occupational exposure value set by NIOSH (Table 5.24). Immediately after the main exposure (i.e., when the ship broke up) exposed persons reported significant increases in headaches, throat irritation, skin irritation, itchy eyes, and mood change. Increases of a lower order were reported for tiredness, diarrhoea, nausea, wheezing, coughing, and chest ache. Symptoms were highest during the three days after the main exposure but persisted until day nine. A range of blood chemistry and urinalysis tests indicated that there were no effects on blood elements or kidney or liver function among those exposed indoors or outdoors compared to controls which were not explained by age, weight, or level of fitness.

Crum (1993) reported that lung function tests on exposed children following the M.V. *Braer* spill were within the 'normal' range (including those with pre-existing asthma) and that there were no differences in lung function at day 3 compared to days 9–12 of the main spill. The author commented that these results were as expected because volatile hydrocarbon levels were not greater than commonly measured urban roadside levels and that most oil droplets were large enough to be filtered out by the nose and throat, preventing lung entry.

Six months after the main exposure related to the M.V. *Braer* spill, Campbell et al. (1994) reported that 7% of exposed people versus 0% of controls reported that their health was poor, and 16% of exposed people reported a deterioration in health compared to 3% of controls. Exposed people reported more tiredness and fever six months after the exposure than during the exposure, but fewer throat/skin/eye irritations and headaches than during the exposure period. The control group also reported changes six months after the spill, that is, more tiredness and headaches. Health questionnaire scores were more than 3-fold higher among exposed people than controls six months after the spill with greater overall scores for skin symptoms, anxiety, and insomnia and lower overall scores for personal dysfunction and severe depression compared to controls.

A health study undertaken by Morita et al. (1999) following the grounding of the *Nakhodka* off the northeast coast of Japan in winter 1997 focused on local residents exposed to the oil because they participated in the clean-up and reported ill health. In general, levels of airborne hydrocarbons were 1–0.5 ppm. The subcomponents of the volatile hydrocarbons measured (total butane, *n*-pentane, benzene, *n*-octane, ethylbenzene, xylene) were all well below occupational limit values. Suspended particulates and H_2S levels were very low (0.9 mg/m^3 and <0.001 ppm, respectively). Major symptoms included headaches, eye and throat irritations, and leg and back pain (probably from the manual clean-up work). Women were more affected than men and, for both men and women, headaches and skin, eye, and throat irritation peaked during the first six days after the spill. The authors did not study any community psycho-social effects but recommended that this work be done.

Zock et al. (2007) reported that following the oil spill from the *Prestige* off the coast of Spain in 2002, exposed groups of fishermen and women who participated in clean-up work had prolonged lower respiratory tract symptoms (self-reported) for up to two years after the spill. The symptoms increased with length of exposure (days and hours per day), and number of clean-up activities undertaken.

The *Sea Empress* oil spill in 1996 involved light crude oil and heavy fuel oil which spread along the west coast of the U.K. A health study of exposed residents found that in general the greatest amount of self-reported illness came after the two major releases of oil (about six days apart) and as the oil spread (Lyons et al., 1999). Self-reported headaches, sore eyes, and sore throats were significantly elevated compared to controls as were symptoms of anxiety and depression.

Poland et al. (2003) in their review of contaminants in the Arctic from all sources indicated that the Komi region pipeline spill, reported to be eight times the volume of the *Exxon Valdez* spill, spread over extensive areas of the region. The spill resulted in contamination of water and fish in the rivers draining the area. While there are reports of tainted foods and poor health, there is no literature available that addresses specific and measured human health impacts associated with this very large spill. Non-peer reviewed information of effects in Russian populations living near the Arctic in the Tyumen Oblast and the Nizhnevartovsk District in the Khanty-Mansi Autonomous Okrug can be found in Appendix 5.1.

5.5.5.4. Sub-Arctic ethnic population impact studies

San Sebastián and Hurtig (2004) summarised the effects of oil and gas development in the Amazon River basin of Ecuador (referred to as the *el Oriente*). The authors described large volume discharges of drilling wastes, regular flaring of waste gases, leaking wells and pumping stations, and extensive pollution of the region. Although their discussion concerned the effects of oil and gas activities on the indigenous population and non-indigenous peasants in the Amazon River basin it may be of relevance for Arctic indigenous populations despite obvious differences between equatorial and Arctic environments and lifestyles, and significant efforts to vigorously manage and control the impacts of oil and gas activities in the Arctic.

Keeping in mind these differences, San Sebastián et al. (2002) and San Sebastián and Hurtig (2004) gave a 'worst case' example of unsustainable oil and gas activities and described adverse health outcomes including higher morbidity and mortality rates, spontaneous abortion, adverse pregnancy outcomes, dermatitis, skin mycosis, and malnutrition among exposed populations compared to populations in areas far from oil extraction. Other studies of the same areas have reported similar effects and include eye, nose and throat irritation, headaches, ear pain, diarrhoea, and gastritis. Cancer studies by these authors (San Sebastián et al., 2001; Hurtig and San Sebastián, 2002) concluded that rates of cancer (total and site-specific) are significantly elevated among men and women in the 'cantones' where oil extraction has been underway for at least 20 years. Significant elevated risks were observed for cancers of the stomach, rectum, skin (melanoma), soft tissue, and kidney in men and for cancers of the cervix and lymph nodes in women. An increase in haematopoietic cancers was observed in the population under ten years of age in the exposed areas in both males and females. This research underlines the importance of implementing regulations and controls among Arctic states to prevent such unsustainable practices and adverse exposures and health outcomes.

O'Rourke and Connolly (2003) examined the distribution of environmental and social impacts of oil exploration, drilling, and extraction among populations and regions. They concluded that current oil and gas activities have a disproportionate impact on indigenous populations and sensitive,

remote ecosystems. In terms of the Arctic, they mention the Eastern Khanty peoples of Western Siberia and the Gwich'in of Alaska but do not provide any criteria for having selected these groups. The authors make extensive use of internal reports and popular literature on websites rather than peer-reviewed scientific journals.

O'Rourke and Connolly (2003) also mentioned the effects of worker migration into oil and gas areas as a vector for disease entry which can affect the health of indigenous people. Gratzfeld (2003) stated that unwanted pregnancies and sexually transmitted diseases often increase among local people when a migrant workforce enters an area where previously local or indigenous communities have been relatively isolated. In addition, oil workers from local indigenous communities who live in camp during two-week periods come into contact with one another and workers from outside the area and often live in crowded conditions, creating a high potential for disease transmission. Indigenous workers may carry the diseases back to their respective communities when they return to their families (Epstein and Selber, 2002). The impacts of disease transmission are far reaching. The indigenous community may not have developed significant immunity to some of the introduced pathogens, early detection and treatment of the disease may be more difficult in isolated Arctic communities, and the spread of infectious disease agents are likely to be more rapid in the crowded indoor living conditions in the cold Arctic environment.

Subsistence communities are often most severely harmed by oil spills. Ecosystem damage can affect access to and use of traditional and subsistence foods. Lost income in local communities unable to fish and sell their products due to a spill or chronic contamination can have far reaching consequences for those with low incomes. For example, O'Rourke and Connolly (2003) stated that … *"The* Exxon Valdez *oil slick covered shorelines used by the Chugach people of Alaska for subsistence hunting, fishing, and gathering. Fifteen Aleut communities in Prince William Sound and the Gulf of Alaska were affected by the oil spill. Subsistence harvests came to a virtual halt after the oil spill. Communities decreased their harvests between 14% and 77% depending on whether they had access to oil-free upland species. One community on Chenaga Bay on the Prince William Sound reduced its harvest from 342 to 148 pounds per person per year. Another community in English Bay on the Kenai Peninsula reduced consumption from 289 to 141 pounds per person per year. The variety of species harvested also declined from 23 to 12".*

5.5.5.5. Psycho-social effects of oil and gas activities

Several authors have reported consistent and well quantified psycho-social impacts associated with oil and gas pollution. Palinkas et al. (1992, 1993, 2004) undertook thoughtfully constructed and evaluated studies of the effects of the *Exxon Valdez* oil spill in Alaska on the psychological health of ethnic groups within and outside the spill area. Their key findings are provided in Appendix 5.1. In evaluating post traumatic stress disorder (PTSD), Palinkas et al. (2004) described the revised definitions provided by the American Psychiatric Association in 1994 to include objective as well as subjective criteria and that an event need not be "*outside the range of normal human experience*". These criteria take into account that "*different people can have profoundly different conceptions as to what constitutes a realistic threat*". The authors argued that PTSD can occur in the absence of death or serious physical injury, if it threatens one's physical integrity or the physical integrity of others and that a diagnosis of PTSD may be associated with a threat to one's social integrity and the physical environment on which the individual or their family depend. PTSD was defined as occurring if a person experienced:

1. One or more of the following symptoms lasting a month or more: persistent unpleasant or disturbing memories; repeated bad dreams or nightmares; feeling worse when in a situation that reminds them of a past event; or flashbacks.
2. Three or more of the following: loss of interest in, or stopped caring about, previously important activities; trying hard not to think of something that happened to them or avoidance of feelings about past events; avoidance of places or activities that remind them of something that had happened; avoidance of other people; loss of feeling or reduction in emotion; change in future plans; or inability to remember part of the past.
3. Two of the following: difficulty falling or staying asleep, irritability or outbursts of anger, or difficulty concentrating; hyper-vigilance, startled by noise, and physiological reactivity upon exposure to events that symbolize or resemble aspects of the traumatic event, manifested either by feeling panicky, fearful or anxious, or by autonomic hyperactivity (i.e., sweating, breathing heavily, heart pounding).

Generalized Anxiety Disorder (GAD) was determined through questions in which respondents had to acknowledge anxiety based on a positive response to questions which related to feeling worried, anxious, or afraid. They also had to have felt as least three additional symptoms of anxiety: motor tension, i.e., one or more symptoms of muscle tension, restlessness, and easy fatigability; symptoms of autonomic hyperactivity; and symptoms of vigilance and scanning, i.e., one or more symptoms of feeling keyed up or on edge, difficulty concentrating, trouble falling or staying asleep and irritability (Palinkas et al., 1993).

Palinkas et al. (1993) concluded in their studies conducted one year after the *Exxon Valdez* spill that Alaska Natives reported more oil exposure and participation in clean-up activities than Euro-Americans and that these factors were associated with a higher prevalence of PTSD in Alaska Natives, but not among Euro-Americans. Palinkas et al. (2004) noted that Alaska Natives reported more disruption of traditional subsistence activities and less family support than Euro-Americans. Alaska Native women were more likely to have a diagnosis of PTSD than Alaska Native men and Euro-Americans of either gender. The authors stated that … *"The destruction of the natural resources is believed to have posed a greater threat to the social resources of the Alaskan Natives than it did to the social resources of other ethnic groups because social relations of Alaskan Natives are grounded in activities of subsistence production and distribution".* Russell et al. (1996) working with Palinkas and others reported that exposure and participation in clean-up showed a significant relationship not only with GAD and PTSD scores, but also with depression, substance abuse, and domestic violence in communities affected by the spill. It is not known if the depression and PTSD findings will persist in this population or disappear over time.

Another research team (Picou and Gill, 1996) gave a summary of chronic stress outcomes resulting from technological disasters involving social uncertainty over biophysical contamination. Using different methods of chronic stress and applying a 'renewable resource community' model (a population of individuals who live within a bounded area and whose primary cultural, social, and economic existences are based

on the harvest and use of renewable natural resources) the authors concluded that: residents from communities within the spill area had higher levels of stress compared to those from communities outside the spill area; residents living in communities based on use of renewable resources showed more stress than those from non-renewable resource-based communities; and that individuals working in the commercial fishing occupations had higher stress levels than those working in other occupations. Stress levels remained high in the most affected communities for three and a half years. The authors concluded that …*"..true restoration [of a community] must also include the reestablishment of a social equilibrium between the biophysical environment and the human community… [which would be] directly facilitated by mitigation strategies designed to resolve community conflict and uncertainty and reduce chronic psychological stress"*.

A follow-up report by the same research team (Arata et al., 2000) gave an even more detailed analysis of resource loss variables, coping measures, and psychological symptoms in communities affected by the *Exxon Valdez* spill. Between 20% and 30% of fishers showed clinically significant anxiety, depression or PTSD. One interpretation is that chronic disruption of the affected communities occurred as a result of the spill.

A health study of exposed residents following the foundering of the *Sea Empress* off the west coast of the U.K. (Lyons et al., 1999) found that in general the exposed population had significantly more anxiety and depression based on two psychological health measures (the Hospital Anxiety and Depression Scale and the SF-36 mental health scale). The authors were unable to determine whether the psychological effects were a generalized response to the oil components or due to concern over the environmental disaster and its potential impacts on employment, health, income, and community.

Campbell et al. (1994) reported that six months after the M.V. *Braer* spill off the Shetland Islands, health questionnaire scores were more than 3-fold higher among exposed people than controls with greater overall scores for skin symptoms, anxiety, and insomnia and lower overall scores for personal dysfunction and severe depression compared to controls. Of note was a general finding that 24% of exposed people and 3% of controls had high enough psychiatric scores in the questionnaire for them to be considered as 'cases', which led the authors to conclude that the data may indicate a 'response to strains on the fabric of this community'.

These studies are consistent internally and use established and innovative tools to determine the nature and extent of psycho-social health impacts in local populations beside spill zones.

5.5.5.6. Health and economic impacts of food tainting on local populations

An obvious immediate impact of an oil spill is the tainting or chemical contamination of fishery products, public fear of tainting or contamination, and public concern over the effects of tainting on health and the local fishing economy. Fish and other aquatic animals can acquire oily taints and odours after periods of only a few hours of exposure to low-boiling hydrocarbons in the low ppm range (see section 5.3.2). The presence of tainting or fear of tainting or contamination often forces the closure of fisheries with consequent economic losses to commercial or subsistence fishers and can affect both jobs and access to high quality, local foods valued by the community. These resource losses can result in psycho-social health effects but rarely lead to significant physical outcomes in individuals or populations unless they are unable to find an adequate alternative food source.

5.5.5.7. Health impacts of pipeline construction

The Honourable J. Berger conducted extensive consultations in the Canadian Arctic and sub-Arctic as a result of the proposed Mackenzie Valley pipeline in the 1970s (Berger, 1983). While the social and political aspects of the consultations and ensuing conclusions were discussed in Chapter 3, those aspects concerning health are discussed here.

Medical witnesses at the Berger Inquiry suggested that a sudden influx of people would bring about changes in health: an increase in the rate of venereal disease (levels twenty times the Canadian average were reported in 1975 in Inuvik, the centre of Canadian Arctic oil and gas activity), a range of childhood diseases associated with changes in diet and abuse of alcohol. Indigenous people in the smaller communities feared an increase in domestic violence, child neglect, community violence, and other abuses they associated with 'drinking' communities. The health professionals who gave evidence predicted that the pipeline project would aggravate social and psycho-social problems existing in communities in the area of the proposed project. The Inquiry report discussed the 'reactive depression' (passivity, lack of interest, lack of motivation and ambition, and a feeling of helplessness) felt by many indigenous people in the valley's communities.

Judge Berger recommended that pipeline construction be postponed for ten years in order to strengthen indigenous society, the indigenous economy – indeed the whole renewable resource sector – and to enable indigenous claims to be settled. He emphasized the importance of promoting existing social and economic strengths and the value of diversification: strengthening of the traditional hunting and trapping economy; the development of local logging and sawmilling operations where feasible; the development of the fishing industry; the development of recreation and conservation; and an orderly program of petroleum exploration in the Mackenzie Delta and the western Arctic which could lead, in due course, to a pipeline along the Mackenzie Valley. In his view, only balanced development would ameliorate or avoid pipeline related effects such as shifts in employment, disruption of family and community life, and a plethora of health and social problems in urban centres.

Since the 1970s there have been many changes in the Canadian Arctic and especially with respect to socio-economic conditions and health services. Some of the changes and perspectives associated with oil and gas activities and pipeline construction are discussed in Chapter 3. In the early 1980s,

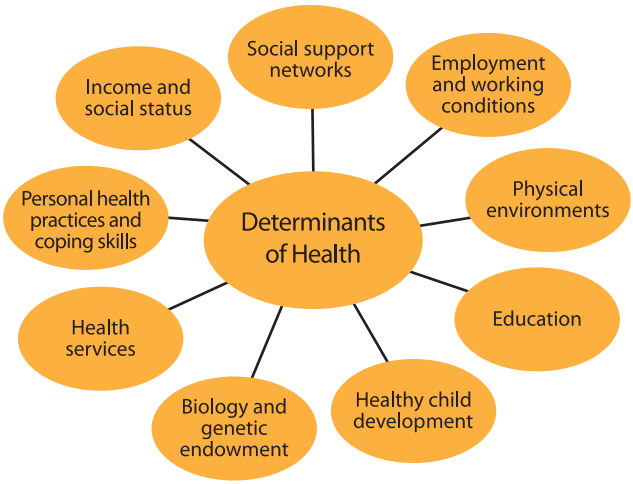

Figure 5.21. Determinants of health (Health Canada, 2004c).

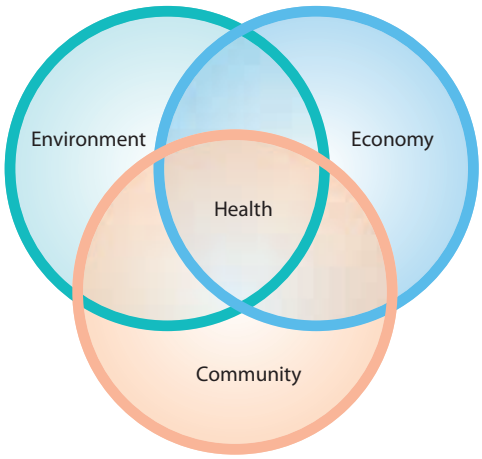

Figure 5.22. Holistic approach to health (Health Canada, 2004c).

a small-diameter oil pipeline was constructed in the Mackenzie Valley without major short-term or ongoing social, economic, or health effects. More recently, in 2004, industry filed an application with the Canadian Federal Government for permission to construct and operate a natural gas pipeline along the Mackenzie Valley as part of the proposed Mackenzie Gas Project. This pipeline would connect natural gas fields in the Mackenzie Delta with existing pipeline infrastructure in southern Canada. If the project gains regulatory approval and environmental clearance and proceeds, the Government of Canada has committed to a CAD 500 million Mackenzie Gas Project Impact Fund to address existing and project-related health and social services needs. Indigenous organisations are playing a key role in planning and priority setting for the use of the funds.

5.5.5.8. Cumulative and combined effects on population health of other contaminants and pressures

In an assessment of the health impacts of oil and gas activities (or any other human enterprise), it is necessary to take into account all factors which could combine to exacerbate or moderate the outcome. These factors are often referred to as the determinants of health (Health Canada, 2004c). Community health is affected by at least nine determinants (Figure 5.21). Expressed a slightly different way, these determinants affect, and are affected by, each other and the economy, the environment, and the community. How they intersect and interact will define 'health' (Figure 5.22).

Education, income status, and social status are known to influence health and employment. Poor, undereducated groups, marginalized in society are often in the poorest health. They have the weakest coping skills and personal health hygiene practices, are often under-nourished and work in the most dangerous, poorly regulated jobs and often without proper equipment or training. The adequacy and availability of health services, social support services, and programs for the support and nurture of children are often lowest for these groups. This situation could contribute to the late detection and spread of disease and could also contribute to these children becoming adults who are in poor mental and physical health, with untreated addictions, unaddressed abuse, or sustained poverty.

Indigenous populations are often among the poorest groups in society as they have often been forced from their ancestral homes and subsistence based way of life on land, water, and ocean by development, governmental policy, environmental factors, or pollution (Bjerregaard, 2003). Their disease rates tend to be higher per capita as seen, for example, in recent evaluations of children's health and adult health in the Arctic (Berner, 2003; SDWG, 2005; Young and Bjerregaard, 2008) and in the Amazon Basin (San Sebastián and Hurtig, 2004).

Environmental contaminant exposure through long-range airborne and waterborne transport adds to disease pressure and social change among Arctic people. These environmental exposures are far higher than any contribution from oil and gas activities. Concentrations of PCBs and mercury in the food supply and in human tissues are of considerable concern in Arctic populations consuming marine mammals and fish from the top of the food chain (especially in Alaska, the eastern Canadian Arctic, Greenland, and parts of the central and eastern Russian Arctic). These exposures, reported by van Oostdam and Tremblay (2003) and AMAP (2004c), have been shown to cause developmental problems and immune system depression in children in some areas and are also suspected of causing other toxic outcomes (Dewailly and Weihe, 2003). The Arctic dilemma, described by Hansen (1998), juxtaposes the nutritional, social, and spiritual importance of traditional foods for Arctic people against the exposure traditional foods provide to toxic contaminants, that is, consumption of some traditional foods provides benefits and risks to health. While oil and gas activities may make minor contributions to local levels of several metals like mercury (sometimes found in oil) and some persistent chemicals like PCBs (occasionally found in cutting oils and machinery), the majority of Arctic-wide exposure occurs as a result of long-range transport of these substances from industrialized regions outside the Arctic (Gilman, 2003). However, in contrast, PAHs found in freshly caught traditional foods in the Arctic are more likely to originate from local oil and gas exploration, natural oil and gas seeps, and local burning (wood smoke, refuse burning, fires) than from long-range transport from populated southern regions.

Another environmental factor of concern for the health of Arctic people is climate change (National Assessment Synthesis Team, 2000; Nelson et al., 2002; ACIA, 2004). A warmer Arctic may enable or cause: more shipping in the Arctic Ocean, more open ports for industrial expansion and commercial fishing operations; changes in fish species composition due to fishing and subsequent effects on marine mammals which feed on these species; less availability of, or access of Arctic subsistence hunters to, marine mammals due to ice conditions, changing ranges of prey populations or lack of availability of fish; arrival of insects or animal species capable of carrying and spreading disease; greater chemical pollution as populations and industrialization grow and consume fossil fuels and goods; loss of communities as a result of thermokarst and shoreline erosion; and others. These factors will add further stress to local communities already affected by oil and gas, industrialization and resource extraction activities, long range pollution, and existing social and health deficits. Climate change effects on duration of ice cover necessary for lake roads and on permafrost may also make shipments of fuel and supplies more hazardous. Pipeline integrity may also be compromised by altered permafrost support infrastructure, leading to additional oil spills and gas line ruptures and human exposure.

Poland et al. (2003) summarised some of the combined pressures, cumulative effects, and remediation options for an Arctic region undergoing rapid economic, environmental, and social change. Detailed evaluation of the combined effects of stressors on health is beyond the scope of

the present assessment; however Dowlatabadi et al. (2004) emphasized the importance of the task and a method for estimating cumulative effects. The authors highlighted governmental attention to cumulative effects as demonstrated by Section 117(2) of the Mackenzie Valley Resource Management Act (covering a geographical region spanning parts of the Northwest Territories and Nunavut, Canada) which requires consideration of *"…any cumulative impact that is likely to result from the development in combination with other developments…"*.

5.5.5.9. Concluding comments

Occupational health studies support the notion that oil and gas activities in the Arctic are unlikely to lead to significant disease in local populations caused by emissions if the work site and operations are managed in a sustainable fashion and oil companies implement all regulatory measures. Benzene, a known carcinogen, tends to occur at low concentrations in the vicinity of oil and gas operations and after spills. Owing to its vapour pressure, benzene volatilizes after a spill with airborne levels declining rapidly within 24 hours to concentrations below a level of concern based on occupational health standards and some ambient air guidelines. Acute myelogenous leukaemia, which is the most consistently reported cancer outcome in refinery-based occupational cohorts, is not likely to occur until exposure reaches more than 200 ppm-years (Raabe and Wong, 1996). Most well sites and transfer facilities have been trying to improve their containment of fugitive emissions and, according to various authors, have been able to reduce occupational levels of benzene. Low-level exposure and short duration exposure to benzene and other VOCs make the risk of cancer in local Arctic populations from this volatile hydrocarbon very unlikely now and in the future.

Although studies of indigenous and non-indigenous peasants in Ecuador living near poorly managed oil and gas activities have reported elevated cancer rates among the exposed population, these results have not been strongly supported by population studies in urban regions near petrochemical facilities. There is also some anecdotal information (not peer reviewed) of excess cancer and other disease among populations living near contaminated oil fields in northern Russia. In general, it is unlikely that there is any excess cancer in most of the Arctic as a result of oil and gas activity because exposures are likely to be very low and those who are exposed are likely to be few in number. A statistically significant identification of excess cancer under these conditions (low exposure and low numbers of exposed people) would be methodologically difficult, if not impossible.

Communities living near significant spill sites do self-report ill health for several months to two years after exposure to airborne hydrocarbons. While measurements of lung function and other medical assays do not indicate specific and statistically significant effects, poor health, deteriorated health and lower respiratory system disorders have been reported. These results depend on the length of exposures sustained by the study population.

Flaring at the well head and open burning of spilled oil can increase inhalation exposures to particulate matter in air. These particles have been shown to cause respiratory health problems in non-Arctic population studies. Although sustained flaring is not common in most of the Arctic, efforts should continue to reduce and avoid flaring whenever possible. Open burning of spilled oil is also an unhealthy practice, especially if it occurs near communities and should also be avoided whenever possible.

Oral exposures to contaminants related to oil and gas via food and water intake are not likely to cause adverse effects in populations living on or near contamination sites or spill zones. Uptake by fish and mammals of metals, persistent compounds like PCBs, and radionuclides found in drilling fluids (muds and cuttings) will be small because they are discharged in relatively small amounts. The metals do not bioconcentrate extensively (except mercury and methylmercury). Lead from additives in fuels is also unlikely to affect health because its contribution to overall lead deposition in the Arctic is very small. Arctic human monitoring data that exist indicate that blood lead concentrations are below a level of concern (AMAP, 1998, 2003). Aliphatic and aromatic compounds are only taken up slowly, most degrade readily and some volatilize. Contamination zones around well sites are relatively small. Human populations tend to avoid eating contaminated food which is tainted (objectionable taste and odour). PAHs, some of which are known carcinogens when inhaled or applied to skin, are not thought to contribute significantly to ill health when ingested. As a result, it is very unlikely that local populations will be exposed via their food to toxic levels of substances released during exploration and production. Water may be contaminated with VOCs, metals, and oil, however, concentrations are usually very low and effects, if any, tend to be transitory.

Access to traditional food is an important determinant of health for most indigenous Arctic populations. An Alaskan permitting document (USEPA, 2005) defined 'significance threshold' pertaining to food security as… *"the level of effect that equals or exceeds the adverse changes in subsistence harvest patterns such that one or more important subsistence resources would become unavailable, undesirable for use, or available only in greatly reduced numbers for a period of one to two years."* In its evaluation it concluded that… *"Exploratory operations within nearshore waters have a higher likelihood of adverse impacts to fisheries, although overall the impact is expected to be minimal. Mammal subsistence harvesting may be affected to the extent that discharge sites may alter the distribution of the animals; however, effects should be insignificant if discharge locations are not in close proximity to each other."* These views on food security and access underscore the importance of good practice and the strict implementation of regulations to prevent impacts on human populations at oil and gas activity sites.

Psychological damage appears to be a consistent impact of oil spill situations (and other major environmental events) and is of concern to health and wellbeing because it can lead to domestic violence, lost income, and loss of community structure. Psychological damage which may be manifest as diagnosed PTSD or other measures of effect (e.g., depression) can be sustained for up to, and beyond, two years in some studies of affected populations. This is a real health effect and is of importance to Arctic communities living near production and transfer facilities where spills have occurred and are continuing to occur.

It is very difficult to predict if any significant substance-based effects will occur in the future in the Arctic without a clear picture of current and future exposures (e.g., who and how many will be exposed, for how long, to which components, at what concentrations, at what locations). It is difficult to extrapolate from one exposure scenario and population effect to another, without knowledge of specific aspects of each event (specific type, quantity, and duration of spill, type of oil or spilled product, presence or absence of gas, weather conditions, population water and food use patterns, and pre-existing population health status). It is difficult to undertake meaningful population studies with

such a variable and complex substance as crude oil and well gas and small local population cohorts.

Cumulative impacts of exposure to a range of substances and conditions are important. Local populations in the vicinity of oil and gas activity have the potential to be affected by spills, fires, gases, and drilling wastes if exposure levels are significant. They may also be affected by persistent contaminants and metals in traditional foods resulting from long-range environmental transport and largely unrelated to oil and gas activities in the Arctic. Food stress, which can affect nutritional status, may come from pressure to change diets due to contamination, lost access to prey, climate change, or social pressure. Substitute foods may be less healthy, especially high carbohydrate-based foods, and predispose the population to disease (e.g., obesity, diabetes). Reduced intake of fish may also significantly reduce intake of Vitamin D and other micronutrients for subsistence populations; Vitamin D deficiencies can lead to a number of debilitating diseases. Local employment and migrant workers can spread disease into communities where there may be little natural immunity and predisposing conditions to infectious disease transfer. Poor early detection and health service response will compound and confound the combined effects of health stressors (determinants of health). Pan-Arctic monitoring (health status, contaminant levels, and community assessments) would be very helpful in assessing both current population health and changes in health status which might result from increased oil and gas activity in the future. Health measures could include lung function, blood parameter measurement, liver and kidney function assays, personal health assessment scales, psychological evaluation, and other situation specific tests. Contaminant monitoring for mercury, lead and cadmium, persistent organic chemicals, and PAHs and their adducts would also be useful. Although occupational health issues are not within the scope of this assessment, worker monitoring and ambient air monitoring at upstream oil and gas sites is essential and serves as a useful surrogate for evaluating the health of populations living near sites of oil and gas activity.

5.5.6. Pan-Arctic risk assessment of the impacts of oil and gas activities on human populations

5.5.6.1. Adequacy and availability of data.

There are adequate data available on uptake, metabolism, excretion, and toxic effects of oil and gas and most of their component parts and products for animals. There are reasonably good data available related to the toxic effects of these exposures in humans, both occupational and non-occupational, although few concern exposures in the Arctic of Arctic people. Some studies report ambient exposure levels, but in general quantitation of levels in the ambient environment to which human populations in the Arctic are exposed is poor and assessment of exposure (concentration and length of exposure) is weak and often non-existent. As a result, hazard assessment (the type of adverse outcome) is possible but risk assessment is very difficult.

Very few information sources reviewed for this assessment were from non-peer reviewed scientific sources. Although some have been included to provide some balance, they have been given little weight in terms of the assessment. While peer review does not ensure the accuracy of the science, it does require that it proceed through an evaluative process where checks and balances apply that can reduce bias or eliminate research which has used flawed methodology. Readers should be cautious in their interpretation of scientific opinions quoted in the assessment from non-peer reviewed sources.

There is a paucity of data on the effects of oil and gas components on indigenous populations in the Arctic. This is not surprising as the population is small, exposure is likely to be minimal, and statistical power for a study is low. There are no peer reviewed exposure data or health-related data available for Russia, which makes a pan-Arctic assessment of the health implications of oil and gas activity incomplete.

5.5.6.2. Risk analysis/assessment methodology for health impacts

Several useful models exist which could be applied to risk assessment of specific oil and gas activities in the circumpolar Arctic (Harwell et al., 1992; Rajaretnam and Spitz, 2000; USAID, 2000; Chen et al., 2003; Health Canada, 2004b). These models evaluate the likelihood of an adverse event, size of exposed population, pre-condition of the population from the point of view of general health/disease/nutritional status, availability of alternative areas or sources of food or drinking water to avoid exposure, reproductive status, at-risk groups by age and gender and ethnicity, and other confounding factors such as alcohol/drugs/smoking and the type and duration of likely exposure. These models all require reliable and good quality data, both of which are lacking for Arctic populations.

USAID (2000) reported an excellent approach to a detailed risk assessment process for specific drilling or transfer sites which takes into account type of activity, characteristics of the activity, possible target groups and likely exposure pathways. It ranked risk from very high (5) to none/very low (0) based on several parameters. Probability of a stressor occurring is ranked from 'will occur' to 'never occurs'. Magnitude of exposure is ranked from a 10^{-3} risk of cancer (one-in-one-thousand risk) or a hazard quotient of 100 to a cancer risk of less than 10^{-6} (one-in-one-million risk) or a hazard quotient of less than one. The number of exposed persons is ranked from more than 10 000 to less than 10. Severity and reversibility of effects are ranked from premature death (irreversible outcome) to little or no damage to health. Uncertainty in the data is ranked by 'no data or estimates only' to 'direct high quality observations/measurements/data'.

Two case studies in the Caspian Sea area used the method to evaluate risks (USAID, 2000). The authors used surrogate concentration data (expressed in terms of benzo[*a*] pyrene equivalencies) taken from a U.S. Food and Drug Administration risk assessment of crude oil contaminants in seafood samples following the *Exxon Valdez* oil spill, to estimate the kind of concentrations/effects of these contaminants that may occur after an oil spill. The 'equivalency' approach is a technique for assessing the risk from a number of different compounds that are present in a single exposure source. It uses the potency of a single PAH to express the cancer potencies of the other PAHs in the mixture. Table 5.33 summarises the information for one of the evaluation case studies.

Chen et al. (2003) evaluated some existing models for estimating risk and proposed an additional model. The authors applied their findings to a water contamination problem (toluene) from a sub-Arctic gas plant. Emphasis was placed on a thorough knowledge of toluene concentrations, the exposed population (receptor), and the full exposure pathway. While the mathematical model used by Chen

Table 5.33. Summary scores for potential human health risks for offshore OKIOC drilling on the Caspian Sea (USAID, 2000).

Potential pollution source	Chemicals of potential concern (considered in case study)	Potential receptors	Potential exposure pathways	Probability of stressor occurrence		Magnitude of exposure		Number of persons exposed		Severity and/ or reversibility of effects		Data uncertainty		Combined risk score[a]	Risk based order (higher value means higher risk)
				risk level	risk score[a]	risk level	risk score[a]	risk level	risk score[a]	risk level	risk score[a]	risk level	risk score[a]		
Releases associated with formation gases															
minor release of H_2S	H_2S	Rig workers (120–150 persons)	Inhalation	medium	3	very low	1	medium	3	low	2	high	4	72	2
major release of H_2S	H_2S	Rig workers (120–150 persons)	Inhalation	low	2	low	2	medium	3	medium	3	high	4	144	3
catastrophic release of H_2S	H_2S	Rig workers (120–150 persons)	Inhalation	very low	1	high	4	medium	3	very high	5	high	4	240	5
Used drilling fluids/cuttings															
	Barium	People in and around Bautino who drink water contaminated with drilling mud or drilling fluid constituents (6000 persons)	Consumption of contaminated ground/surface water	medium	3	low	2	high	4	low	2	high	4	192	4
Minor oil spills (≤1 barrel)															
	PAHs	General population in and around Atyrau who consume fish products (450 000 persons)	Consumption of contaminated fish and shellfish	high	4	none to very low	1	none to very low	1	very low	1	high	4	16	1

[a]See section 5.5.6.2 for the bases of the 'risk scores'.

and co-workers differs from that used by the authors of the USAID report, both rely on site specific assessments and the need for a good knowledge of operational issues which could relate to exposure. The Harwell et al. (1992) model focused on three ecosystem factors: the extent of the stress (biosphere, region, or ecosystem), the medium affected (air, water, or land), and the recovery time of the ecosystem (short, medium, or long). The model dealt more with impacts on the environment upon which human populations depend than with individual human population impacts.

Health Canada (2004b,c) has provided extensive documentation on a step-wise approach to considering both hazards and risks of contaminated sites for different ages and genders living on, near, or visiting the site. Intake factors for soil, water, food, and air, and toxicological reference values were used to aid the assessment. This approach provides sound methodology once a site is identified as requiring a characterization and an evaluation of the risk it poses to the local population. Specific attention is given to indigenous people. However, use of the approach for assessment of Arctic oil and gas activity sites is limited because no specific contaminated sites have sufficient characterization for such an evaluation.

5.5.6.3. Concluding comments

In general, the overall risk to the health of Arctic populations from oil and gas, their components and products *per se* appears to be very low, even though it is not quantifiable without more population-specific exposure data from across the Arctic. Following accidental oil spills and gas emissions, exposures to most substances which have the potential to be toxic appear to be short term (rapid evaporative losses, wind dispersed, rapid clean-up, or containment), low in concentration (below health guidelines), and most often avoidable (unless airborne). Most effects which have been reported among local populations occur after a major spill and are transitory. Local temperature inversions could increase respiratory health risks for populations living near large gas and particulate sources; however, none have been reported to date among Arctic populations and probably none have been investigated. Low-level exposure and short duration of exposure to PAHs, benzene, and other volatile substances make risk of cancer in most Arctic populations living near oil and gas sites very low now and in the future. If small increases in cancer rates were to occur, they would probably be undetectable due to the small sample size of the exposed population.

Non-peer reviewed information on population health effects in parts of Russia – the Tyumen Oblast and the Nizhnevartovsk District in the Khanty-Mansi Autonomous Okrug – are suggestive of significant health risks associated with oil and gas contamination in this region. If the effects reported in these regions are valid (e.g., sustained eye, nose and throat irritation, reproductive and mental health problems, and cancer), they would be consistent with animal and human findings following sustained high-level exposure to oil and gas elements (fresh oil, volatile aromatic hydrocarbons, SO_2, and H_2S). These findings need to be corroborated, and if confirmed, acted upon.

Psychological trauma, expressed as PTSD has been documented in some Arctic and sub-Arctic communities following spills. The risk of psychological injury for individuals near accident sites where there is significant environmental damage is high. PTSD should be considered as a serious 'health' outcome in populations which are dependent on the land, cooperative hunting, and food sharing.

The risk of communicable disease is likely to increase moderately in Arctic populations as further oil and gas activities (and exploration/extraction for other resources) expand. Migrant workers, crowded conditions, re-infection of home communities, loss of access to traditional foods, and social structure and access to alcohol and drugs will all play a part in this process. It is not possible to quantify this risk as it will vary from location to location and is entirely dependent on documented physical health outcomes. Population and worker disease outcomes are usually difficult to identify in the Arctic owing, for example, to small communities, lack of standard disease registries, transient populations, lack of available medical services, and confidentiality of patient records. Future efforts to provide good quality health care, exposure reduction advice, and disease management strategies in communities will counterbalance some of the increased risk of disease related to oil and gas and other industrial activities in the Arctic.

While there are methods available for detailed analyses of the risks posed by contaminated oil and gas sites, these apply to site-specific situations where data have been collected around the site or estimated from other data sources. Data are lacking for major oil-contaminated sites in the Arctic and there is no comprehensive list of pan-Arctic oil-contaminated sites available, hence, quantitative or semi-quantitative risk assessment is not possible for major oil-contaminated zones.

Future contamination of some areas of the Arctic land mass and the marine environment by oil is inevitable. Increasing quantities of oil and gas will be extracted and shipped from Arctic well heads. Considering the extent and age of some sections of the Arctic oil pipeline network and of the flow lines and gathering lines from some older facilities, it is likely that transfer and transport accidents will result in some significant spills or gas emissions. Climate change may exacerbate these losses as the permafrost, used as the structural base for much of the oil transport pipeline system (storage, pumping, compressing), thaws. Use of oil tankers may also increase in Arctic waters as ice thickness decreases. Tanker spills may expose coastal populations to oil, may contaminate fish and wildlife used for food, and may reduce access to traditional food species which move away from the contaminated zone. Oil in the Arctic environment is likely to degrade more slowly and remain a source of PAH exposure longer than in sub-Arctic regions. Counterbalancing these pressures are actions by Arctic jurisdictions to set regulations, to enforce guidelines and standards, and to require the most sustainable practices available.

Oil and gas companies appear to be working to apply best practices, to implement regulations, and to work more closely with communities near well sites and after accidents. Better process controls and equipment, better training of oil and gas workers and emergency measures personnel, improved maintenance of pipelines and remediation of the worst sites, can reduce risks of damage to the ecosystem upon which health depends (i.e., contamination of the air people breathe, the water they drink, and the food they eat). Because there are multiple and connected determinants of health, measuring the benefits of safer and more informed oil and gas activity may be impossible. Nevertheless, reducing human exposure to petroleum hydrocarbons, gases, metals, and other substances which may enter the human environment is a good strategy and will always optimize health.

5.5.7. Concluding comments and recommendations on the impacts of oil and gas activities on human populations

5.5.7.1. Effects on human populations

Well-related contamination is not likely to pose a threat to human populations living within the vicinity of the sites.

- Contamination from offshore wells is highly unlikely to pose any threat to health unless there is a large blowout or oil spill. Human populations live many kilometres away from these wells and, despite their toxic properties, exposures to volatile chemicals, well gases, PAHs, and other contaminants would be very low.
- Contamination from onshore wells is also unlikely to lead to any significant exposures unless operational releases of exhausts, well-head gases, and sustained flaring combine with climatic conditions to enable airborne contaminants to build up within the human breathing zone for several days. A large land spill may lead to contamination of lake or river water used for drinking purposes; however, exposure would be small as local consumers would probably seek other uncontaminated water sources for consumption where they exist.

Considering the toxicity characteristics of several oil and gas components alone it is clear that oil and gas, their components and products have the potential to affect human health if exposure is high enough. In general, there is more evidence that various components of crude oil can cause acute and chronic health risks than crude oil by itself.

- Human reactions to acute exposure to petroleum components are mainly transient and short lived. Accidental contact with petroleum or exposure to its vapour causes skin irritation and stinging and redness of the eyes. Prolonged or repeated exposure to low concentrations of volatile components of oil can produce nausea, dizziness, headaches, drowsiness, and depression.
- There is ample evidence (animal and human) that inhalation of some individual petroleum components (such as benzene, toluene, xylene, and *n*-hexane) and gases (such as H_2S and SO_2) can produce both acute and chronic toxic effects. The concentrations at which effects occur vary by component. Most prevalent and characterized are eye, nose and throat irritations, headaches, and lassitude. These are all common outcomes following exposure to volatile hydrocarbons and tend to be reversible once exposure ends. Inhalation of oil droplets only occurs under rare conditions (oil spray, high wind in a spill or blowout zone) and is not known to have lasting health effects if the duration and amount of exposure are small.
- Inhalation exposure resulting from flaring at the well head and open burning of spilled oil can be very dangerous as the particulate matter is easily inhaled, retained in the lungs and can contain high concentrations of bioavailable contaminants (such as PAH, sulphurous compounds, dioxins, furans, and metals). Most Arctic gas has a low sulphur content (<1%) and would not form high concentrations of particulates. Nevertheless, flaring of gas and open burning of oil (rare events in the Arctic as a result of better management and control techniques) need to be avoided whenever possible.
- Dermal exposure of humans and animals to crude oil has been shown to produce toxic effects. Dermatitis is a concern but tends to be limited to direct contact with oil (on drill sites, as a result of a spill, during clean-up operations) and the effects are transitory. Once exposure ceases, most skin inflammation disappears in a week to ten days. Photosensitization of skin following oil exposure can cause enhanced skin burn during periods of exposure to direct sunlight (UV radiation).
- Oral exposures to oil-related components are likely to be minimal and are not likely to cause adverse effects in populations living near well sites, oil-contamination sites, or spill zones. Human populations tend to avoid eating contaminated food which is clearly tainted (taste or odour detection). Ingested PAHs, some of which are known carcinogens when inhaled or applied to skin, are not thought to contribute significantly to ill health. Smoked and open fire-roasted foods have higher PAH content than most foods obtained from oil-contamination zones and deemed suitable for consumption based on taste and odour. Drinking water may be contaminated with volatile organics, metals and oil; but concentrations are usually very low and effects, if any, tend to be transitory.

Increases in human cancer risk related to oil and gas activity in the Arctic are unlikely.

- Although some volatile components of oil and gas and PAHs can cause cancer in animals and humans under specific conditions of exposure route, duration, and concentration, most of these conditions are not met in local communities in the Arctic (very few people are exposed and exposures are extremely low to most of the substances of concern). Benzene, a known carcinogen, tends to occur at low concentrations in the vicinity of oil and gas operations. Due to its vapour pressure, benzene volatilizes to the atmosphere after a spill and airborne levels decline rapidly (<24 hr) to concentrations below a level of concern. The duration of benzene exposure of local populations in the Arctic from all sources (spills, seeps, emissions) is very short. PAHs consumed in food tend to have low cancer potency and food that is clearly tainted with oil products is rarely eaten by local communities.
- A study from one region in the southern hemisphere reported elevated cancer rates among an exposed indigenous population close to oil and gas activities; however, these results have not been supported by similar studies in other regions and other predisposing conditions may have been involved. Anecdotal information related to excess cancer among the population in the Tyumen Oblast and the Nizhnevartovsk District in the Khanty-Mansi Autonomous Okrug of Russia has not been peer reviewed and needs to be confirmed.

Psychological damage appears to be a consistent impact of oil spill situations (and other major environmental events) and is of concern to health and wellbeing because it can lead to domestic violence, lost income, and loss of community structure.

- Psychological damage which may be manifest as diagnosed PTSD or some other measure of effect can be sustained for up to, and beyond, two years in some studies of affected populations. Effects are felt by both non-indigenous and indigenous communities; however, in the case of published oil spill investigations, more strongly by indigenous communities and by women within those communities. Tainting of food as a result of oil contamination can lead to dietary changes and poorer nutrition in the community and among individuals and can result in economic and psychosocial impacts.

There are likely to be some beneficial impacts of oil and gas activity on health (physical, mental, and social wellbeing).

- Revenue and employment associated with oil and gas activities can increase the availability of disposable income, improve quality of, and access to, health care and enable better local and distance transportation systems for communities (also discussed in Chapter 3).

5.5.7.2. Adequacy of data for assessment

While there are adequate data related to the toxicity of oil and gas and their components and products, there are insufficient exposure data and epidemiology data for the pan-Arctic region for a full assessment of risk.

- There are no peer-reviewed studies of health outcomes in non-occupationally exposed Arctic populations, although there are several cross-sectional, case control, and cohort studies of sub-Arctic populations exposed to oil and its components following spills or to some gas emissions.
- Some of the reasons for the lack of health impact information for Arctic populations include: the limited number of spills near Arctic communities; the difficulty in conducting meaningful and statistically reliable health studies in small communities with small populations; the lack of baseline data for local populations; and the lack of scientific capacity or protocols to launch investigations rapidly after a spill.

It is very difficult to predict 'oil and gas activity related' future risks to health in the Arctic without a clear picture of the specific aspects of each event and population (specific location and duration of spill; type and quantity of oil or product spilled; presence or absence of gas; weather conditions; population water and food use patterns; and pre-existing population health status).

5.5.7.3. Complexity of health impacts and assessment

Physiological, psychological, and social impacts are all linked. There is some evidence in this assessment that oil- and gas-related events, such as spills, in combination with other 'determinants' of health (e.g., education, access to health care, nutritional status, family income, social status, genetic and biological endowment, personal lifestyle choices) can affect the physical, mental, and social health of individuals and a community.

- Indigenous people are more vulnerable to the impacts of oil and gas activity as a result of their general health, socio-economic status, and reliance on traditional country food. PTSD is a significant health risk following an oil or gas spill close to a community or its hunting territory.
- Cumulative impacts of environmental exposures are important. Local populations in the vicinity of oil and gas activities may be impacted in subtle ways (physical or psychological) by small spills, low concentrations of airborne particulates and gases, and tainted food and water They may also be impacted by persistent contaminants and metals arriving from sources outside the Arctic and accumulating in country foods. Food stress, which can affect nutritional status, may come from pressure to change diets due to contamination, lost access to prey, or social pressure. Substitute store-bought food containing high carbohydrate content and less omega-3 fatty acids may not be as healthy as traditional food and may predispose the population to disease (e.g., obesity, diabetes, cardiovascular injury). The need for income to pay for store-bought food may lead to family disruption, change in lifestyle and traditional values, and to jobs that are prone to the transfer of infectious diseases. Lack of recognition of disease or the availability of local health care may exacerbate adverse health outcomes in this cycle of combined and interacting impacts.
- Health assessments undertaken in communities affected or likely to be affected by oil and gas activities may not be able to detect or interpret small shifts in biomarkers, disease, or social change in small groups of individuals because benchmark health status data are unavailable in most communities.
- Local employment and migrant workers can spread disease into communities. Poor early detection and health service response will compound and confound the combined effects of health stressors.

5.5.7.4. Areas of concern which may impact health

There appear to be very few, reliable, peer-reviewed data on human exposure levels, contaminant concentration data related to oil and gas activities for media and food, disease rates, or other health outcomes from contaminated sites or high production sites in the Arctic.

- There is no up-to-date list of characterized contamination sites related to oil and gas activity in the Arctic. Once identified and characterized, sites close to settlements or villages are the areas most likely to yield information on population health impacts.
- Specific areas of Arctic Russia (e.g., the Tyumen Oblast, the Nizhnevartovsk District in the Khanty-Mansi Autonomous Okrug, the Komi Region) are likely to be contaminated as a result of oil and gas activities and some findings, even though not peer reviewed, indicate adverse health outcomes are occurring in some of these sites.

5.5.7.5. Future impacts on health

Oil spills in the Arctic are likely to continue, but spills are likely to be smaller in volume and better managed due to better maintenance, response, and containment. The impacts on health are uncertain.

- Oil extraction will increase in volume in the Arctic over the next 20 years, politically motivated sabotage may increase, some transmission lines are quite old and are also vulnerable to the impacts of climate change on pipeline structures (change in the permafrost), and

more oil tanker traffic may occur in Arctic waters as ice becomes thinner. Oil transportation events may expose larger areas of the Arctic to oil contamination and could then expose more Arctic people to a range of oil and gas contaminants.

- Health and safety protocols will become increasingly important in the prevention and mitigation of oil spills. Stringent regulations, guidelines and standards and their enforcement are essential to ensure the most sustainable practices are in place in the oil and gas sector.
- Reducing human exposure to petroleum hydrocarbons, gases, metals, and other substances which may enter the human environment is beneficial to health. Efforts to provide good quality health care, exposure reduction advice and disease management strategies in communities will counterbalance some of the increased risk of disease related to oil and gas and other industrial activities in the Arctic.

5.5.7.6. Recommendations

Studies are needed to identify, characterize, and prioritize current oil and gas contamination areas in the Arctic, and especially in Russia.

- Direct application of a combination of the risk methodologies cited in section 5.5.6.2 and others which are available at three to five of the most contaminated sites in the Arctic would be very worthwhile for estimating risks under Arctic conditions and for understanding and mitigating health effects which may have occurred or may be occurring in these locations.
- Action is required to remediate those sites which are found to be badly polluted in order to significantly reduce or prevent threats to health of populations living in the area.
- Studies are required to validate information from the Tyumen Oblast and the Nizhnevartovsk District in the Khanty-Mansi Autonomous Okrug of Russia regarding the impacts of oil and gas activities on population health. If confirmed, health authorities need to work with other governmental agencies and the oil and gas companies in the region to improve the health outcomes.

Pan-Arctic monitoring of health status including psychological impacts and contaminant levels in ambient air, water, food, and human tissues would be very helpful in assessing current population exposure and health (bench marking) and changes in health status which might result from increased oil and gas activity in the future.

- Special focus should be placed on community assessments and on children and women of reproductive age within those communities.
- Health assessments undertaken in communities affected or likely to be affected by oil and gas activities should take into account multiple determinants of health and a variety of the most significant likely biological, psychological, and social markers.
- The assessments need to be carefully constructed with community input to ensure that the scientific and data expectations are capable of being met.
- Better monitoring of infectious disease among the workforce at oil and gas facilities would enable more prompt and effective treatment of the occupational cohort. It would also reduce the transfer of disease from workers to communities even as oil and gas activities expand in the Arctic.

5.6. Assessment of the effects of oil and gas activity on the environment and human health in the Arctic

The effects of oil and gas activity on human health and the environment in the Arctic has been assessed in this chapter using information published mostly in the peer-reviewed scientific literature, supplemented by some 'grey' literature (e.g., consulting reports, website articles). Despite the large amount of information available, significant gaps in knowledge remain, limiting the ability to assess ongoing impacts and to provide guidance on impacts likely to be caused by an expanding oil and gas industry in the Arctic.

This section describes several overall themes which have arisen in most or all of the detailed sections of this report relating to impacts of oil and gas activity on fish and aquatic organisms (see section 5.3), terrestrial species (see section 5.4) and human populations (see section 5.5). These common themes are not intended to replace the conclusions and recommendations presented at the end of each of the sections named above; they are to reinforce some common understandings related to the effects of oil and gas activity on human health and the environment and areas of future concern.

5.6.1. Research

This assessment identified several areas where further research is required. For example, there are virtually no studies on the exposure of the general population in the Arctic to chemical and metal compounds released from oil and gas activities; this information would allow scientists and regulators to determine potential health effects. Similarly, no data were available relating to the exposure of terrestrial birds and mammals to oilfield chemicals and releases from production sites. In contrast, there is a great deal of information on the effects of hydrocarbons on aquatic organisms, although this mostly concerns temperate or sub-Arctic species and its appropriateness for truly Arctic settings is limited. While there are many, strong national research programs, there is a need for improved coordination of research across the Arctic using, where possible, common methodologies, species and publication guidelines. This applies to research on the health of human populations, wildlife and aquatic/marine species.

The funding of research remains an issue with at least the appearance that interpretation of research results may be biased in the direction of the funding source. This can affect the credibility of the results and conclusions. It seems desirable to establish a funding mechanism that operates at arms length from industry, government or environmental organizations although these organizations may provide some or all of the funds.

5.6.2. Monitoring

Assessment of impacts in the Arctic environment with the onset and expansion of oil and gas activity relies on rigorous, consistent monitoring of environmental media and

health impacts. One of the impediments in the current assessment was the difficulty in obtaining published data from monitoring programs because few of the findings are published in the primary literature. The development of consistent, rigorous monitoring programs using measures that can be applied throughout much of the Arctic would allow detection of changes in the environment and human health. New tools, such as biological markers of exposure and effects are emerging and offer promise for future monitoring programs. Their development, using rigorous, internationally agreed methodologies, should be a priority. Specimen banks to store biotic (and sometimes abiotic) samples prior to the development of oil and gas projects will permit future retrospective analyses as new techniques emerge (chemical, biochemical, genetic etc.). Some agreed type of data management and archiving of qualitative information such as microscope slides and images of tissues are needed. A dedicated, multinational institute would be an appropriate mechanism for this.

5.6.3. Methodology

Analysis of hydrocarbons has changed markedly over the years. There has been a general increase in sensitivity and greater separation of individual compounds with improvements in analytical equipment. However, these changes make it difficult to determine changes in the concentrations of hydrocarbons among locations or to compare past levels with current levels. It is vital that methods of analysis be agreed upon and standardized to allow comparison throughout the Arctic and over time. As new methods of analysis emerge, for example, two-dimensional gas chromatography with MS detection, efforts need to be made to calibrate the newer methods against the older methods so that comparisons between historical measurements made with old technology can be compared with new data. The same applies to the biological measures such as biochemical measures, histological preparations and genetic measures. Statistically based standards of analytical quality need to be agreed upon and stated in reports.

5.6.4. Data quality

A huge body of information is present in the 'grey' literature in the form of industry, consultant and summary reports, and these often contain excellent information. However, the lack of peer review leaves the reader with questions about quality and reduces confidence in these reports. Scientists should be encouraged to publish results from environmental and human health programs in peer-reviewed journals to promote access to and confidence in the information.

5.6.5. Risk assessment

This assessment was conducted using peer-reviewed scientific literature and sometimes 'grey' literature; however, there were areas where rigorous scientific studies were missing. In general, there is a lack of information on the environmental and human health impacts of oil and gas activities in Russia. Russia has a diverse, rigorous scientific community and has the greatest production and transportation of oil and gas in the Arctic, yet few environmental or health studies were available for this region. Anecdotal evidence suggests that environmental impacts, particularly from pipelines and spills, may be substantial in some areas of the country. As authoritative risk assesment studies become more accessible, and as the quality of the information is verified, conclusions about impacts of the industry in the Arctic may have to be altered. No quantitative risk assessments can be completed without site-specific and population-specific study data.

5.6.6. Contaminated sites

Contaminated sites within the Arctic need to be located, mapped and characterised (size, history, content, particle-size, geology, etc.) to determine their potential impact on humans and biota in the surrounding environment. Although some remediation and revegetation research is occurring, there appears to be little success in restoring the sites to their natural state. Follow-up at many of the early sites with new methods of analysis would also provide a wealth of knowledge on the rates of biodegradation of the hydrocarbons and other industry-related contamination that can be used as oil and gas activity expands. There is also a need to locate contaminated sediments to determine the full extent of impacts of past oil and gas activity. Reports of adverse impacts on animals and human populations near contamination sites can only be verified if there is full identification and characterization of these sites and careful research on population health status.

5.6.7. Natural seeps

Natural seeps are like ongoing spills and provide valuable sites for basic research on the transport and transformation of hydrocarbons, on the physical and chemical changes to hydrocarbons under Arctic conditions, and on adaptations by the local benthic biological community to long-term exposure. The metabolism of oil and formation of metabolic by-products, rates of transformation of hydrocarbons, and colonization by bacteria and other biota offer important insights on hydrocarbon chemistry in the Arctic.

5.6.8. Improving technologies

Major advances have been made in the technology used to locate and extract oil and gas in the Arctic. One of the most visible examples of change is in the conversion of seismic testing from steel-tracked equipment during the summer months to balloon-tired vehicles operated largely in winter. Similar improvements have been made in the production and transportation of products and in waste handling. However, the application of the new methods, technologies and policies across the Arctic is difficult to judge. Despite these improvements in management of oil and gas activities, major expansion of the industry is likely to lead to ongoing impacts due to the volume of activity predicted in the Arctic.

5.6.9. Regulations

Detailed regulations used to control oil and gas activities are present in all Arctic countries, but their application and enforcement is not uniformly applied. For example, an oil pipeline in Alaska which ruptured in 2006 was not regulated because of various loopholes, leading to poor monitoring by the operator. A similar pipeline leak apparently occurred at roughly the same time in the Komi district of Russia although no details on the size of the leak are available. It is imperative that governments examine their regulations to ensure that all facilities are equally regulated and that enforcement is consistent throughout the Arctic. Guidelines for soil, air and water quality, which have usually been developed for the protection of the environment and human

heath, also need to be standardized throughout the Arctic to the extent that this is possible.

5.6.10. Complexity of assessment

This assessment reviewed the scientific literature to assemble a scientific consensus on the effects of oil and gas activities. Diverse oil and gas activities and various chemical inputs are expected to make innumerable changes in an Arctic setting, and it is not possible to predict the cumulative effects of the physical, chemical and biological changes. The excellent research following the *Exxon Valdez* spill shows the difficulties in predicting changes within a large ecosystem; some of the impacts were not evident until a decade or more after the spill, and many had a high level of uncertainty. The effects of changes within the Arctic ecosystem, which is generally more poorly understood than sub-Arctic and temperate ecosystems, cannot be predicted, particularly within the context of a warming climate. Issues such as the psychological and social effects on people living near spills, ecological and behavioural changes in Arctic wildlife in response to development, and toxicology in Arctic aquatic and terrestrial systems are still poorly known. Further research on indicators of the cumulative effects of oil and gas activities will help to document the extent of changes.

Appendix 5.1. Ecological impacts and population health effects of Arctic and some sub-Arctic oil spills

A5.1. Introduction

Given the difficulties in extrapolation from laboratory experiments, much has been learned from studies following accidental spills. Oil spills occur frequently and number in the hundreds to thousands per year but most are small (Chapter 4). Details of several significant Arctic/sub-Arctic oil spills are provided in other chapters of this report.

As the Arctic ice becomes less extensive during summer there is likely to be increased ship traffic and more frequent spills. This Appendix describes some of the effects of oil spills on aquatic and terrestrial ecosystems and human health in cases where they seem instructive for potential Arctic events. Effects of oil spills on aquatic organisms have been reviewed on several occasions (e.g., Vandermeulen, 1982; Teal and Howarth, 1984; Vandermeulen and Hrudey, 1987; Baker et al., 1990; Pearce, 1993; Birtwell and McAllister, 2002; Ikävalko, 2005). Information is provided chronologically from earlier to more recent spills. The information includes a range of spill types: from tanker spills, to terminal and pipeline spills, storage tanks, and blowouts.

The causes of the oil releases to the environment identified in this Appendix range from accidents to lack of maintenance, and from natural disasters to sabotage. Damage is reported in most cases to the aquatic and terrestrial ecosystems and sometimes to human health. While some authors have emphasized the 'recovery' from impacts of oil spills (Baker et al., 1990), most (e.g., Vandermeulen, 1982; Pearce, 1993) have noted effects that sometimes persist for many years, experience confirmed by ongoing research for more than a decade after the *Exxon Valdez* spill (e.g., Peterson et al., 2003). For aquatic ecosystems, the most obvious damage is to seabirds, particularly those species in contact with a surface slick, and sometimes furbearers like otters; less obvious damage occurs to taxa found in shallow, intertidal zones and soft-bottom sub-tidal areas. Relatively few observations describe effects far offshore. With improvements in capacity to identify and measure hydrocarbons at low concentrations has come the growing realization that PAHs persist in sediments, particularly in low-energy areas. These lingering PAHs can have subtle effects on sediment-dwelling organisms or on their predators for years after spill events. However, sediments are complex mixtures that contain numerous contaminants and it is not always straightforward to attribute the effects elicited by them to specific compounds found in them (e.g., Sundberg et al., 2006).

Overall, the study of spills offers a picture of growing sophistication in research to document environmental behaviour of oil components and to describe the biological consequences for organisms exposed to them. Effects being detected are more and more subtle as means of detection improve. There has been an expansion in interest beyond early documentation of the occurrence of oil slicks and immediate kills of biota or closures of fisheries due to oil tainting, to studies of increasingly sensitive environmental chemistry and toxicology. An example is the emergence of technologies to detect changes in DNA. These technologies are detecting effects on the DNA of individuals exposed in contaminated habitat but the implications for those individuals are not clear. In one experiment, rats fed mussels contaminated by the Erika spill suffered 'moderate' DNA breakage in liver and bone marrow but the interpretation of these effects in terms of risks to the rats or to any other consumers of seafood will require future work. In spite of the growing sophistication of spill research, old problems remain with efforts to detect population impacts and to distinguish them from normal variation in population measures, and efforts to understand the meaning of diverse sub-lethal effects and pathologies observed in individuals for those individuals and especially for populations of them.

The 'extra-Arctic' experiences in the following sections are instructive for the Arctic because oil is likely to persist for longer periods than elsewhere, the period of biological productivity is relatively short and often concentrated in space, and sensitivities to toxic effects of oil seem similar to those found in organisms from more temperate areas.

A5.2. Arctic spills

Tanker spill, S.S. Johnathan Harrington, Elson Lagoon, Alaska, August 1944

Incident: The S.S. *Johnathan Harrington*, an oil tanker with a U.S. Navy convoy, became stranded on a sand bar off Doctor Island in Elson Lagoon southeast of Barrow, Alaska (approx 71°20′ N, 156°30′ W) in August, 1944. To re-float the ship, about 25 000 gallons of oil were released into the sea. This account has been abstracted from the personal observations of Mr. Thomas Brower Sr (*Qiniqtuagaksrat Utuqqanaat Inuuniagninisiqun*, 1980).

Ecological impacts: The oil formed a mass several inches thick on top of the water and covered both sides of the Plover Islands. A solid mass of oil 6 to 10 inches thick surrounded the islands and extended out about 60 feet on the seaward side although a slick extended much further. Seals and seabirds that swam in the (oil-coated) water were reported to be blinded and suffocated by contact with the oil. It took approximately four years for the oil to disappear and during this time the bowhead whales (*Balaena mysticetus*) that normally migrate close to the islands in the autumn made a wide detour out to sea. Indigenous hunters who customarily hunted whales at this location could not do so during the period.

This is the only account of biological observations found by the authors following a spill during the Second World War although there must have been others.

Health effects: No reports available.

Shore tank destruction by avalanche, Deception Bay, Canada, 6 June 1970

Incident: A spill of around 367 000 gallons of Arctic diesel oil and 58 000 gallons of gasoline occurred at Deception Bay, Quebec, (62°08′ N, 74°41′ W) between 6 and 8 June, 1970 (Ramseier et al., 1973). In this instance, an avalanche of slush destroyed a storage tank farm and spilled the contents onto permafrost and sea ice. Diesel recovered from an open pool about ten days after the spill had lost only 2–4% of its lightest fraction indicating that little evaporation had taken place. The reason for this lack of evaporation was hypothesized to be transfer of heat from oil to ice. When the oil absorbs energy, it loses that energy to the ice and so rather than evaporation of oil, the ice melts. Little evaporation was evident from pools of oil on Deception Bay because oil from these pools had almost the same boiling range as the original diesel; a loss of 3–4% of low-boiling compounds would account for such an altered boiling range.

Ecological impacts: Relatively little biological damage was caused beyond the immediate setting. About 90% of the biota in the sub-tidal slide area was destroyed by the physical effects of the slide; as much as half the populations of certain bivalves were killed by oil in inshore tidal areas. Very little (perhaps 5%) polychaete fauna could have been killed. *Fucus* and *Mytilus* were affected the most but probably more by heat from burning the oil on the surface than by exposure to the oil. There were no effects observed on the plankton. Overall the damage was described as slight and localized.

Health effects: No reports available.

Pipeline break, Barrow, Alaska, 5 August, 1976

Incident: Spillage of leaded gasoline from a pipeline buried under a gravel pad at Barrow, Alaska, was discovered on 5 August, 1976 (Horowitz and Atlas, 1977). The amount spilled was estimated to have been 55 000 gallons of which about half was recovered. The gasoline moved through the gravel pad, over the permafrost and into a small, nearby lake which served as the drinking water supply to the Arctic Naval Research Laboratory.

Ecological impacts: Studies reported by Horowitz and Atlas (1977) showed that bacterial populations in water and sediment were greatly enhanced by the influx of gasoline, especially for taxa capable of using the gasoline as a substrate. The bacterial population contributed to the biodegradation of the gasoline such that by the time the lake began to freeze some 10% of extractable material remained in the sediment. Nutrient supplementation and inoculation with hydrocarbon-utilizing bacteria enhanced biodegradation and reduced the level of extractable material to only 3%.

Health effects: No reports available.

Tanker spill, USNS Potomac, Melville Bay, west coast of Greenland, 5 August 1977

Incident: The U.S. naval supply ship USNS *Potomac* bound for Thule struck a small iceberg on 5 August, 1977 and spilled about 100 000 gallons of Bunker C fuel oil in Melville Bay, Greenland at 74°53′ N, 61°13′ W (Maurer and Kane, 1978).

Ecological impacts: From 8 to 16 days after the spill, water samples were taken from the R/V *Adolf Jensen* in an effort to describe the extent of any surface contamination with oil and depth profiles were taken to examine vertical mixing (Ahnoff and Eklund, 1978). The concentrations in surface water ranged from 0.3–2 µg/L and these decreased with depth. Two weeks after the spill, personnel skilled in oil spill studies reached the site and collected samples of plankton for analyses. Zooplankton samples from the surface were dominated by the amphipod *Parathemisto libellula*, while those from the water column consisted mostly of the large copepod *Calanus hyperboreus*. There was no indication that the abundance or species composition of the plankton were affected but plankton sampled nearest the slick area were more highly contaminated with hydrocarbon residues.

Health effects: None reported.

Pipeline sabotage, Steele Creek, near Fairbanks, Alaska, 15 February 1978

Incident: Intentional sabotage resulted in a spill of 500 000 to 672 000 gallons of crude oil from the Trans Alaska Pipeline near Fairbanks on 15 February, 1978 (Buhite, 1979).

Ecological impacts: The spill contaminated about 2.1 acres of land but some 15 500 barrels were recovered over a 63-day clean-up period (Buhite, 1979). Remaining vegetation and soil contaminated with about 500 barrels of oil were pushed away from the pipeline and some 1.9 acres were burned in a controlled manner. The original vegetation was destroyed but the burned area was fertilized with high-phosphorus fertilizer and allowed to revegetate naturally. About half the site experienced regrowth in the first growing season. By coincidence, an experiment was conducted almost exactly two years earlier to investigate the effects of a spill on spruce forest (Jenkins et al., 1978) at a site about 48 km from Fairbanks. The damage to plants was described in three categories: immediate, within the initial growing season, and cumulative after the initial growing season. Contact with oil was lethal quickly to most above-ground plants, particularly lichens and mosses growing in low-elevation areas. Black spruce (*Picea mariana*) became chlorotic, turned brown and abscised during the second growing season. There was no recovery or revegetation by seedlings over the two years of observation.

Health effects: No human health data.

Shore tank leakage, Spitzbergen, spring 1978

Incident: Approximately 130 m^3 of diesel fuel was spilled into Van Mijenfjord, Spitzbergen in spring 1978 from tanks onshore near the Svea coal mine (Gulliksen and Taasen, 1982).

Ecological impacts: Investigations on the effects of the spill began three weeks after ice cleared, from July 21–30, 1978. These were continued in July 1979 and 1980. Sediment samples taken near the spill site in 1980 contained considerable quantities of diesel. The highest level was 5892 mg/kg in surface sediment from the site nearest the tanks, compared to levels of 13–23 mg/kg at unoiled sites. In spite of this, the biological impact was described as negligible mainly because the area had little flora or

fauna regardless of the spill. During summer 1978 there was no sign of marine life in the littoral zone just below the tank site but at similar unoiled sites nearby algae and amphipods were found.

Health effects: No reports available.

Fuel Spillage, Cameron River, near Yellowknife, Northwest Territories, Canada, March 1983

Incident: In March, 1983, a fuel truck loaded with diesel overturned about 15 m from the bank of the ice-covered Cameron River near Yellowknife, Northwest Territories, Canada (Lockhart et al., 2002a). Most of the diesel was spilled the following day during attempts to recover the truck and its cargo. A sample of diesel was taken at that time. The spilled diesel penetrated the snow (60 cm deep) and migrated along the frozen surface of the soil toward the Cameron River. Attempts were made to burn the oil but some entered the river. Eight days later some diesel was recovered when a hole in the river ice was made about 25 m downstream from the point where the diesel entered the river. The study largely reported the comparison of the two samples with respect to chemistry, toxicology and capacity to taint fish.

Ecological impacts: Chromatograms of the diesel from the two samples were essentially identical indicating little change in composition over the eight days. Water-soluble fractions were prepared from the two diesel samples and LC_{50} values for larval whitefish (*Coregonus clupeaformis*) were both about 1 ppm (nominal concentration) indicating retention of the toxic components for the eight-day period. Several months after the spill, a fisherman complained that whitefish taken about 500 m downstream from the spill site were inedible due to their oily taste and odour. The fish in that reach of the river would have been present at the time of the spill since they were confined between two impassable waterfalls 16 km apart. These observations prompted a laboratory examination of the ability of the two diesel samples to taint whitefish in the laboratory. Both samples produced about the same intensity of oily flavours in the fish during an exposure of 2–5 or 5 hours indicating little or no loss of tainting components during the eight days in the river.

Health effects: None reported.

Contaminated soils, Nizhnevartovsk Region (western Siberia), Russia, 1980 to present

Incident: Between 700 000 and 840 000 ha of polluted soil in western Siberia and about 2% (6500 ha) of land in the oil fields in the Nizhnevartovsk Region are polluted by oil and oil-related activity (Ingram and Willemse, 2001). The soil pollution has been caused largely by oil pipeline and well spills, but also by oily muds, drilling and production waste, chemical waste disposal and leaking storage sites, saline production water, operational discharges and leakage, and oil production site drainage. The information below is all from unrefereed information (Ingram and Willemse, 2001).

Ecosystem impacts: Groundwater and aquifer pollution has occurred in the Nizhnevartovsk Region, particularly in the older oil fields such as *Samotlor*, where oil concentrations around 0.1 mg/L have been found in groundwater layers up to 200 m below the surface. Surface water pollution is extensive in western Siberia, particularly in water bodies around oil fields, and especially near older fields in the Nizhnevartovsk Region. The main pollutants are oil products, reaching between 5 to 50 times the Russian maximum permissible concentration (PDK values).

Over 50% of the fished rivers in the region are contaminated with oil products, resulting in reduced fish health and breeding ability, changed spawning behaviour, and are possibly linked to declining fish populations. These negative impacts may be caused by pollution, urbanization and industrialization of the region in which the oil sector is the major industry.

Health effects: A high percentage (97%) of drinking water extracted from the River Vakh over a five-year period was polluted with oil exceeding the PDK value. Whilst treated surface and groundwater presents a low risk to human health, the large number of water supply stations and many, unofficial, private wells providing untreated drinking water presents a considerable, unquantified health hazard.

The Chief Doctor of the Oncological Clinic of the Tyumen Oblast, Dr. N. Naumov, reported that the total number of cancer cases in the Tyumen Oblast in 1999 was 5659. This is 25.2% more than in 1989. Children accounted for less than 1% of the total number of cases, whereas people between 15 and 59 years of age accounted for 60% of the total. Between 1989 and 1999, the cancer rate increased from 143 to 194 per 100 000 in the age category 15–59 years and from 173 to 351 per 100 000 in children. Increases by gender over the ten-year period were 28.5% for men (rectal, 28.5%; lungs, 28.5%) and 35.5% for women (rectal, 53.9%; cervical, 53.9%; colon, 42.5%; breast, 34.6%).

The Regional Ecological Committee of Khanty Mansiysk indicated in their 1998 report (see Ingram and Willemse, 2001) that the main groups of illnesses were: diseases of the respiratory system, traumas and poisoning, and infectious and parasite diseases. There is a tendency toward increased tuberculosis and TB-related death. Alcoholism is 1.5 times higher in Khanty Mansiysk compared to the Russian population average.

There is a suspected (but unproven) relationship between the 'poor environment' in Nizhnevartovsk and poor health statistics, that is, there is a higher than average level of certain oncological, cardiological and endocrinological diseases and a shorter life expectancy compared to the rest of the Russian population. Nizhnevartovsk has been on the list of Russian cities having the highest indicators for diseases caused by a poor environment since 1990.

Oncological disease rates in Nizhnevartovsk are reported by Ingram and Willemse (2001) as having exceeded the Russian population average since 1994 for persons of 15–19 years and 50–59 years of age. Rates of endocrinological disease exceeded the average for the Russian population by 1.17–1.87 times for the age groups 20–29, 40–49 and 50–59, and 2.52 times for children. In 1995, there were reports of increases in tumours and rheumatic and skin diseases among adults and increases in endocrinological, opthalmological, ear, and skin diseases among children. The increases were sustained in 1996. Ninety percent of expectant mothers experienced a complicated pregnancy and 35% of infants were born with anomalies in Nizhnevartovsk in 1995.

Data from 'AGIS – Environment and Health' (cited by Ingram and Willemse, 2001) indicate that illnesses related to ecological problems in the city of Nizhnevartovsk

during the period 1991 to 1996 were between 1.3 and 3.6 times higher than the average for the Russian population. The total rate of illness in Nizhnevartovsk increased significantly from 1991 to 1998. In 1991, the total disease rate (morbidity) was 144 per 100 000 compared to 366 per 100 000 in 1993. The incidence rate of oncological disease is 2 to 3 times higher than anywhere else in Russia within the age group of 15–49 years. Cardiological disease morbidity exceeded the average rate in Russia by 9 times, while endocrinological disease morbidity exceeded the average Russian rate by 2 to 5 times.

The Tyumen Regional Oncological Centre has reported research that it has been conducting on the environmental effects of oil pollution and the relationship between oil pollution and disease (information as cited by Ingram and Willemse, 2001). The centre has been collecting information from other hospitals within the Tyumen Region for about 20 years, when health effects thought to be related to the oil industry were first noticed. According to Dr. N. Raycov, the radioactivity of oil (radon) has increased because of several nuclear activities within the region during Soviet times. Dr. Raycov drew the following conclusions: total population oncological diseases have increased 150% in the Tyumen Oblast over the last 20 years; oncological diseases of people aged 25–69 years have increased from 30 to 60% of the total in the last 20 years; oncological diseases in children up to 25 years of age have increased by 100% in the last 20 years; testicular cancer has increased by 300% in the last 20 years; total rectal cancer has increased by 80% in the last 5 years; oncological disease in Nizhnevartovsk, Surgut, Nishnirovgorod and Langepas regions are 30–40% higher than in surrounding regions; oncological diseases in the towns of Nizhnevartovsk, Radoesjnie, Megion, Langepas and Gagalin have increased in the last 5 years by almost 100%.

Leaking pipeline, Usinsk (Komi Republic), Russia, 1994 (see Chapter 4, section 4.4.6.2)

Incident: The largest Arctic spill to date is that from several pipeline leaks in northern Russia, often collectively called the 'Komi' spills. The largest spill occurred following several ruptures of a pipeline in the Usinsk region of the Komi Republic in 1994. A quantity of crude oil, variously estimated in the 100 000 to 200 000 tonne range, was spilled into marshland.

Ecological impacts: Investigations from 1995 to 1999 identified over 300 polluted sites and 745 ha were described as 'severely impacted.' Some oil reached the Kolva and Ussa rivers, tributaries to the Pechora River, and moved into the Pechora River some 600 km from the estuary. Ecologically, the Pechora basin is unique because it comprises the western limit for Siberian species and the eastern limit of European species of fish belonging to the salmon-whitefish complex. A number of ecological investigations have been reported over several years following the spill and further descriptive publications are anticipated. Biological observations included changes in the trophic structure of the lower Kolva River with increased numbers of predators, deformed pike in Lake Shutchye and decreased populations of 'minnows' and European grayling (*Thymallus thymallus*).

Health effects: Local villagers have suffered for years from the effects of the petroleum pollution from the many oil spills in the region. Most local people are worried about the fish living in the Kolva River. "The river used to have lots of fish, now there are hardly any at all and when we cook them they smell bad...people here survive but they are worried about the future."

A5.3. Sub-Arctic spills

Tanker spill, Palva, Finnish southwest archipelago, 1 May, 1969

Incident: The MT *Palva* ran aground on 1 May, 1969, in the southern part of the Finnish Archipelago Sea (approximately 59°50′ N, 21°25′ E). An estimated 150 t of Russian crude oil was spilled and this spread at least 25 nautical miles from the spill site (Heino, 1972).

Ecological impacts: Efforts were made to clean up the oil with booms, dispersants and by burning but oil patches were still found on shores in the area a year later. Several biological studies indicated that the most serious effect was the death of approximately 25–33% (2400–3000 individuals) of the eider ducks (*Somateria mollissima*) nesting in the polluted area (Soikkeli and Virtanen, 1972). Studies in 1969 and 1970 revealed no obvious effects on littoral fauna (Pelkonen and Tulkki, 1972) or benthos (Mustonen and Tulkki, 1972). There was some localized damage to shoreline vascular plants due to patches of oil or to attempts to burn the oil. Algae were undamaged, although these had not begun annual growth at the time of the spill (Ravanko, 1972).

Health effects: No reports available.

Tanker spill, Irini, southern Stockholm archipelago, 6 October 1970

Incident: The small tanker *Irini* ran aground in the southern part of the Stockholm archipelago on 6 October, 1970, and spilled about 1000 t of medium and heavy fuel oil over the following five days. Wind forced almost half the spilled oil into a small bay, Gastviken. The Swedish Coast Guard used booms, skimmers, and dispersants (Corexit 7674 and Finasol SC) to protect the shores. Clean-up operations were completed in May, 1971.

Ecological impacts: No oil-related effects were evident on macrophytic algae; bladder wrack (*Fucus vesiculosus*) was resistant to oil and was chosen for sampling. Fauna on samples of *Fucus* taken from Gastviken from 1971 to 1976 were compared with similar samples from a nearby, unaffected bay (Notini, 1978). Some 24 species of animals were found on *Fucus* from Gastviken over the period June to November, 1971; of these 5 were found in June, 10 in July, 16 in August, and 14 in November. This indicates a large initial reduction in most of these animals over the first winter with some recovery over the following summer. With the exception of chironomids, *Fucus* from the reference bay had higher numbers of all animal taxa. Blue mussels (*Mytilus edulis*) were completely missing from *Fucus* in Gastviken during the first three sampling periods of 1971 but recolonized the bay to levels of about 50 individuals per 100 g of *Fucus* over the following years. More mobile animals such as insects, fish and swimming crustaceans were able to recolonize the bay faster than less mobile animals. Four years after the spill, Gastviken showed no further evidence of detrimental effects on the animals.

Health effects: No reports available.

Tanker spill, Drupa, near Stavanger, Norway, 14 February 1976

Incident: The *Drupa* was grounded at Klakken (approx. 58°31′ N, 5°42′ E) when approaching Stavanger, Norway, on 14 February, 1976, spilling 2000 to 2400 t of Iranian crude oil. The oil spread over calm seas and contaminated the shoreline and some fishing nets. The amounts of oil that had 'disappeared' from the water surface and shoreline over time at several locations ranged from 28% over 6 days to 65% over 50 days, with no further loss over the following month (Grahl-Nielsen et al., 1976).

Ecological impacts: The area where two seine nets had been set was oiled. One net contained 150 t of saithe (*Gadus virens*) and the second, 50 t of saithe. These fish were not marketed but some were taken for organoleptic analysis and for chemical analysis for hydrocarbons; both procedures found the fish to be untainted (Grahl-Nielsen et al., 1976). Several species of benthic invertebrates were analyzed for residues of aromatic hydrocarbons present in the oil and were found to have significant levels up to a maximum of 28 μg/g of tissue. These levels declined rapidly for the next 2–3 months although they had still not disappeared completely from all species a year later (Grahl-Nielsen et al., 1976).

Health effects: No reports are available; tainted fish were kept from the market.

Tanker spill, Imperial St. Clair, Lake Huron, Canada, 23 December 1976

Incident: This is the only example cited in this Appendix of a spill in ice-covered freshwaters. The *Imperial St. Clair* grounded in Lake Huron on 23 December, 1976, spilling about 57 000 gallons of diesel fuel and gasoline into Lake Huron near Parry Sound, Ontario, Canada (Beckett, 1979). The ice conditions at the time made work unsafe on the surface. The only option judged feasible to control the spread of the oil was burning and numerous burns were conducted over the period from late December 1976, until the end of March 1977 (these efforts included exploding dynamite to make holes in the ice to create a place for oil to collect and be burned). Several significant observations were recorded at the time:

- Oil froze into the ice and did not remain in large pockets. Freshwater ice contained up to 60% oil but this ice was weak and melted earliest in the spring.
- Burning was most efficient but should be conducted immediately after the spill or at the spring thaw.
- Burning was ineffective in very cold or windy conditions.
- Heat generated by burning oil appeared to induce a capillary action on oil remaining in the ice thus perpetuating the initial burn. In this setting oil dispersed little under the ice.
- Heavy snow prevented burning of surface oil.
- Blasting was effective in determining if pockets of oil were present under the ice. Blasted areas melted first and allowed water and oil to collect in depressions.
- All clean-up methods were ineffective when the ice was moving in floes.

Ecological impacts: Not reported.

Health effects: Not reported.

Platform Bravo blowout, Ekofisk, North Sea, 22 April 1977

Incident: A blowout occurred in the *Ekofisk* oil field at platform Bravo in the North Sea on 22 April 1977. A mixture of oil and gas was ejected to a height some 50 m above sea level and at a temperature of ~75–90 °C. The amount of oil involved was estimated to be between 20 000 and 30 000 t over 7.5 days resulting in a slick of some 4000 km^2 (Mackie et al., 1978). Recovery of oil from the slick area was unsuccessful due to gale-force winds.

Ecological impacts: Much of the environmental monitoring consisted of surveys of hydrocarbon contamination in water, fish, and sediments. Grahl-Nielsen (1978) used naphthalenes, phenanthrenes and dibenzothiophenes (NPD) as tracers for blowout oil and mapped the distribution of these hydrocarbons in the water for several weeks after the blowout. At a station near the platform, 4.1 μg/L of NPD and 250 μg/L of oil-in-water emulsion were found. The most extensive distribution of NPD in seawater was found two weeks after the spill. The highest concentration of NPD was reported to be 8 μg/L in water samples close to the platform during and immediately after the blowout.

Law (1978) used GC/MS to determine aliphatic hydrocarbon concentrations in water samples collected at 1-m depth on 28–29 April. These fell in the range 0.05–3.9 μg/L and aromatic hydrocarbons fell in the range 0.04–1.7 μg/L.

Despite the presence of hydrocarbons in the water, analyses of several samples of fish collected on 28–29 April revealed no contamination attributable to the blowout (Law, 1978). Hydrocarbons were identified by fluorescence in material collected in sediment traps indicating loss to particulate material that settled to the bottom. Samples collected in July showed loss of oil during the interval from May to July. In May, hydrocarbons were identified in gut and stomach contents of some fish (mackerel, haddock, plaice) and taste panels found that some of the fish were tainted although hydrocarbons were not found in muscle (Mackie et al., 1978).

In May 1977, Johnson et al. (1978) sampled sediments at 41 locations over a square grid some 100 miles to a side with its centre at platform Bravo. The whole area was one of intense petroleum production and hydrocarbons were identified in sediments routinely. Johnson and co-workers attempted to discriminate the hydrocarbons from the blowout from those derived from other sources and concluded that concentrations from 5 to 8 ppm at seven stations could be derived from the blowout.

Health effects: No reports available.

Tanker spill, Tsesis, Baltic Sea, 26 October 1977

Incident: The Soviet tanker *Tsesis* grounded in the archipelago south of Stockholm, Sweden on 26 October, 1977. Seven cargo tanks containing a total of 6400 t of fuel oil opened on grounding; bunker tanks were also damaged. It was estimated that about 1100 t were spilled, mostly No.5 fuel oil, and about 600–700 t were recovered, leaving about 400 t in the environment. Booms were deployed within a few hours but oil that escaped the

booms was found along 18 km of shoreline (Linden et al., 1979). Oil visibly contaminated an area of about 34 km^2. This spill was investigated thoroughly and a report on several follow-up investigations is available (Kineman et al., 1980). Damage to birds and to the littoral zone was minimal due of the timing of the spill.

Ecological impacts: An investigation of the effects of acute exposure on pelagic planktonic organisms revealed an apparent increase in phytoplankton primary production and biomass, possibly caused by decreased grazing by zooplankton. About half the zooplankton specimens examined were observed to be visibly contaminated with oil droplets. Bacterial abundance increased (Linden et al., 1979). Sediment traps were used to estimate the amount of oil that settled to the bottom. Two days before the spill the area was subject to strong winds which resuspended particulate matter from the bottom and this, along with faecal pellets from zooplankton, may have provided sites to adsorb oil and transport it to the bottom. Material taken in sediment traps contained as much as 0.7% oil, probably mostly adsorbed onto the recently resuspended sediment particles (Johansson, 1980).

The major biological damage was observed where the oil reached shorelines. Johansson (1980) reported that phytoplankton species composition remained unchanged by dissolved oil (the concentration two to five days after the accident was 50–60 µg/L). Microflagellates dominated the community before and after the spill. Phytoplankton biomass and productivity increased after the accident, which was probably due to depressed zooplankton grazing. Although planktonic bacteria biomass increased notably after the oil spill, the populations of their grazers (ciliates, rotifers) did not (Johansson et al., 1980). The abundance of larger zooplankton changed shortly after the accident, probably due to narcosis and/or avoidance reactions. The spill had virtually no effect on bladder wrack (dormant at the time of the accident) but had a major effect on the macrofauna associated with it (Notini, 1980) and on sediment-dwelling fauna (Notini, 1980; Elmgren et al., 1983). In some areas, littoral crustaceans were reduced drastically but the populations recovered within a year, presumably by immigration of new individuals from elsewhere.

Amphipods (*Pontoporeia*) as well as the polychaete *Harmothoe sarsi* Kinberg were reduced to less than 5% of pre-spill biomass at the most highly contaminated site (Elmgren et al., 1983). All meiofauna except nematodes were greatly reduced and a large kill of ostracods was suspected. Neither the macrofauna nor the meiofauna showed any sign of recovery over the nine to ten months following the spill (Elmgren et al., 1980) but recovery became evident by the second summer after the spill although *Pontoporeia* biomass was still depressed after three years (Elmgren et al., 1983).

Clams (*Macoma balthica*), became contaminated to about 2000 µg hydrocarbons /g but suffered little or no mortality in spite of this; levels fell by about 50% by the second summer following the spill.

Liver and muscle of flounder (*Pleuronectes flesus*) from soft-bottom areas where they prey on *Macoma* were highly contaminated with petroleum hydrocarbons (>50 ppm) about a year after the spill (Linden et al., 1979).

The area of the spill included habitat used by herring (*Clupea harengus membras* L.) and sprat (*Strattus sprattus*) and spawning areas for herring. Spawning sites for herring were less frequent in the spill area than in reference sites and hatching success was lower. The low hatching success may have been due to factors other than the oil; the authors hypothesized that the disappearance of Gammarids from the contaminated area may have led to increased fungal infection of the herring eggs and reduced hatching success. Surveys detected no change in the abundance of pelagic fish in the area and chemical analysis of herring showed no contamination with oil (Nellbring et al., 1980).

Health effects: No reports available.

Terminal spill, Esso Bernicia, Shetland Islands, December 1978

Incident: 1200 t of bunker C fuel oil were spilled at the newly opened Sullom Voe Oil terminal in the Shetland Islands in December 1978.

Ecological impacts: The amount of oil spilled was relatively small but effects on seabirds were significant. The British Marine Life Study Society estimated that more than 4000 birds were oiled and that great northern divers (*Gavia immer*) were affected most (http://www.glaucus.org.uk/oilbirds.htm). Surveys of dead birds on Shetland Island beaches found a higher proportion of oiled carcasses in 1979 than in any other year up to 2004 (Heubeck, 2006). Otters (*Lutra lutra*) were seen in contact with the oil, making no apparent attempt to avoid it. Thirteen otter corpses were recovered between the time of the spill and 22 February, 1979. Complete post mortem examinations were conducted on five of the thirteen otters found and all showed gastro-enteropathy with evidence of ingested oil, apparently as a result of grooming oiled fur (Baker, 1981).

Health effects: No reports available.

Tanker spill, Antonio Gramsci, Gulf of Finland, 27 February 1979

Incident: The Soviet oil tanker *Antonio Gramsci* grounded off Venstpils, Latvia, on 27 February, 1979. An estimated 5000–6000 t of crude oil escaped into the sea. Approximately 500 t were collected over an area of about 1300 km^2, but most of the oil migrated north. By early May, oil clumps had stranded on the shores of the Åland Islands. Clean-up work was completed in July to August (Itämeren öljyvahinko, 1979; Finnish Ministry of the Interior, 1980; Bonsdorff, 1981).

Ecological impacts: Observed effects on fish were restricted to predominance of young sprat (*Clupea sprattus*) with tail abnormalities. Spawning by fish did not seem to have been affected by the oil. The acute phase of the oil accident was prior to the spawning period of sprat, and thus the oil toxicity was lower by the time the oil reached the spawning areas. Oil effects were thus mainly indirect, that is, through the food chain.

In the *Cladophora* belt (the littoral zone, 0.5–1.5 m depth) fauna remained largely unaffected by oil. Blue mussels, gammarid amphipods (*Gammarus* spp.) and barnacles were not harmed. In the *Fucus* belt (3–6 m depth), blue mussel, the snail *Theodoxus fluviatilis*, and isopod (e.g., *Idotea balthica*) populations decreased drastically in spring. In deep soft bottoms, no dramatic long-term changes of the fauna (*Macoma balthica*, *Pontoporeia affinis*, Chironomidae, Ostracoda) were observed.

No studies on hydrocarbon levels had been made in these areas prior to the accident, so the controls, background values, and exact information about

population sizes (e.g., of benthic animals) in the contaminated areas were missing.

The accident occurred in the wintering areas of eiders, and an estimated 31 000 eiders died due to the effects of oil. In the Åland Islands, about 1200 dead and 500 oil-stained birds were collected in April to May 1979. Clean-up activities disturbed nesting birds, and lead to decreased eider reproduction. Only a small number of oiled seals were seen after the spill.

Health effects: No reports available.

Spill, Ryuyo Maru No. 2, fishing vessel, Pribiloff Islands, 8 November 1979

Incident: The fishing vessel *Ryuyo Maru No. 2* ran aground at Village Cove, Pribiloff Islands on 8 November 1979, and an estimated 40 000 gallons of fuel were spilled (Reiter, 1981).

Ecological impacts: Fifty percent of the micro-organisms and higher invertebrate animals in the saltwater lagoon were estimated to have been killed. Dead, oiled seals were found but it was not known whether they were oiled before or after death. The vessel was subsequently destroyed by demolition and the greatest risk to seals then became not the oil but fishing nets that were either washed overboard or released by the demolition. Seals became tangled in the nets and drowned. Caution was taken to keep the noise of demolition to a minimum but there was evidence of seals and birds killed by concussion in the immediate area of the demolition.

Health effects: No reports available.

Tanker spill, M/S Eira, north Baltic Sea, 31 August 1984

(The following description was prepared by Dr. Johanna Ikävalko, Helsinki, from documents published in Finnish.)

Incident: The M/S *Eira* grounded in the Quark (northern Baltic Sea) on 31 August 1984. Approximately 200 t of heavy fuel oil (POR 180) were spilled into the sea. The Quark is a very shallow (in some parts <20 m depth) area between Bothnian Bay (northernmost Baltic) and the Baltic Proper (central Baltic).

Ecological impacts: After the oil spill, hydrocarbon concentrations in water were elevated for about a month. In sediments, the hydrocarbon content was not notably higher than normal, while in soft bottoms (the littoral zone), slightly elevated concentrations were recorded over large areas in the Quark. Concentrations of hydrocarbons were elevated in *Macoma balthica* and blue mussel in autumn 1984, but by the following summer had decreased to normal background levels.

The oil spilled from the M/S *Eira* appeared to be only slightly acutely toxic to some *Gammarus* species. Many oiled specimens of zooplankton were recorded. The abundance of *Gammarus duebeni* and the snail *Lymnea palustris* decreased in the most severely contaminated areas. These species live at the high water level, and thus had direct contact with stranded oil. Effects of oil were much less severe on benthos from deeper habitat and on littoral zone organisms. (Note that there is no tide in the Baltic Sea. All water level fluctuations are due to inflow from rivers, precipitation, evaporation, outflow through the Danish Straits in the west, air pressure, and prevailing winds.) Littoral fauna on hard bottoms showed a decrease in abundance in summer 1985, which may have been a consequence of the *Eira* oil spill. Effects of the oil spill in soft bottom communities were minor. A few *Pontoporeia affinis* specimens with oil-like spots on their body were observed, but no changes in population structure were detected.

Fishing was affected for about two months (September to October 1984) in the contaminated area, partly due to the oil itself, but also because fish left the area. The odour and flavour of fish were, however, normal. Perch (*Perca fluviatilis*) suffered from skin diseases, and the size of liver and gonads changed, clearly indicating that the perch were under physiological stress. The mortality of perch eggs was higher than in normal years. Juvenile herring and goby (*Gobiidae*) were smaller than during a normal summer, and the rear part of their body was short and curved. Effects were also observed in their skeleton and jaws. Long-term effects of the *Eira* oil spill were not studied. About four months after the spill, perch were collected from oiled and unoiled areas for comparison of mixed-function oxygenase activities and these were found to be slightly elevated in perch from the oiled areas (Lindström-Seppa, 1988).

During autumn 1984, 1914 seabirds were recorded dead due to the oil spill (1528 black guillemots, *Cepphus grylle*; 259 eiders, *Somateria mollissima*; and 117 other bird species). The accident had the strongest negative effect on the black guillemot populations: in summer 1985 the population in the Quark had decreased by 500 pairs, and between summer 1985 and summer 1987 by 700 (28.5%). Auks and terns were not affected by the accident as the oil did not reach the auks' nesting areas, and the oil combating and cleaning of beaches in the tern nesting areas was very effective. However, in 1985 the egg shells of both black guillemots and auks were thinner than in normal years, but this may have been partially due to the late spring (i.e., cold weather in early spring).

Health effects: No reports available.

Tanker spill, Antonio Gramsci, south coast of Finland, 6 February 1987

Incident: The *Antonio Gramsci* had a second, smaller spill, this time grounding off the south coast of Finland on 6 February 1987 (Hirvi, 1989, 1990). About 570 t of Soviet Export Blend (SEB) crude oil were spilled in severe ice conditions. Oiled ice drifted uncontrolled, expanding the contaminated area. Attempts were made to collect oiled ice but with little success; approximately 100 t of oil were collected from the sea in February.

Ecological impacts: During ice break-up in May, the oil was released into the water column and formed an oily film on the surface. In mid-May, the oil drifted ashore in the southern archipelago of Finland. Because the oil existed as small patches in the open sea, attempts to collect it were stopped in mid-May, but activities were continued in the archipelago area. About 38 t of oil were collected from Finnish shorelines.

Of the 570 t of crude oil spilled, about 110 t were recovered by oil countermeasures, and 185 t were eliminated by weathering, mainly through evaporation. An estimated 270 t remained in the marine environment (Hirvi, 1989). Oil effects on benthos were studied in 1987 and 1988. *Pontoporeia* sp. had distinct oily spots on its body

in the open sea area, and in the coastal area, *Pontoporeia* sp., *Gammarus* sp., and the littoral gastropod *Lymnea* sp. were clearly contaminated by oil. *Gammarus* sp. still suffered from oil in 1988. Hydrocarbon concentrations were analyzed in the isopod *Saduria entomon* (*Mesidotea entomon*), *Lymnaea* sp. and two bivalves, *Macoma balthica* and blue mussel. Elevated concentrations of hydrocarbons were found especially in the second year after the accident.

The impact on seabirds was relatively small. Since the oil was trapped in ice for months after the accident, many nesting areas remained unaffected during the most sensitive period of nesting and hatching. Seals were also unaffected. However, perch and flounder caught in the oil-affected areas had higher concentrations of hydrocarbons in liver than fish from the reference area in 1987; by 1988 the situation had improved. The oil probably affected the health of the fish in 1987; they had lower levels of vitamin C in caudal fins, increased incidences of parasites and diseases, and changes in the sizes of liver and maturing gonads. Some catches of herring and salmon were tainted with an oily taste and odour.

Health effects: No reports available.

Tanker spill, Exxon Valdez, Prince William Sound, Alaska, 24 March 1989

Incident: On 24 March 1989, an accident involving the super-tanker *Exxon Valdez* resulted in a spill of 11 million gallons of crude oil into the waters of Prince William Sound, Alaska. This spill and its effects have been described by the most extensive series of studies of any northern spill to date.

Ecological impacts: The literature on this spill is vast. Several books have been published describing the effects of this single spill (e.g., Loughlin, 1994; Wells et al., 1995; Rice et al., 1996). Summary and synthesis papers have since appeared (e.g., Peterson et al., 2003). The studies following this spill combined chemical analyses for hydrocarbons with biological sampling to describe impacts on a range of taxa. The most immediate response was closure of several fisheries (black cod, *Anoplopoma fimbria*; shrimp; salmon; herring). Spies et al. (1996) summarized a wide range of chemical and biological studies. Biologically, populations of intertidal organisms were affected most. Stekoll et al. (1996) reported effects on intertidal invertebrates and plants, notably *Fucus*, limpets, mussels, barnacles, littorine gastropods and oligochaetes. Sampling in 1990 and 1991 showed that recovery varied at different locations but generally remained incomplete. In contrast, oligochaete populations were enhanced at oiled sites. Recolonization of oiled shorelines by algae and invertebrates was inhibited by oil coatings on intertidal rocks. Growth of juvenile pink salmon (*Oncorhynchus gorbuscha*) was reduced in 1989 but not 1990 (Wertheimer and Celewycz, 1996; Willette, 1996). Dolly Varden (*Salvelinus malma*) and cutthroat trout (*Oncorhynchus clarki*) were similarly affected (Hepler et al., 1996). Around a third of the local population of sea otters was lost (Garrott et al., 1993). In 1989 and 1990, one of several pods of killer whales (AB pod) suffered the highest mortality rates of any in the North Pacific, but the cause could not be proven to be the spill (Dahlheim and Matkin, 1994). This pod frequented some of the most heavily oiled sites in Prince William Sound; other pods did not show this unusual mortality. Some of the apparent effects on salmon eggs in spawning streams were still being debated ten or more years after the spill (Brannon et al., 2001), showing the difficulty of establishing clear cause/effect relationships. Peterson et al. (2003) reviewed the effects on Prince William Sound after more than a decade and found several effects that were still apparent, probably due to the long-term persistence of PAHs in the sediments.

Health effects: Several studies have been undertaken by Palinkas et al. (1992, 1993, 2004) on the human health effects of the *Exxon Valdez* spill.

Alaska Natives were more likely than Euro-Americans to have participated in clean-up activities and to report damage to commercial fishing areas and effects on non-commercial hunting, fishing and gathering activities.

The post-spill (i.e., 1 year) prevalence of Generalized Anxiety Disorder (GAD) and Post Traumatic Stress Disorder (PTSD) as defined by Palinkas et al. (1993, 2004) for the study communities with all degrees of exposure was 20.2% and 9.4% respectively. When compared with the unexposed group, members of the high-exposure group were 3.6 times more likely to have GAD and 2.9 times more likely to have PTSD.

Women exposed to this event were particularly vulnerable to these conditions and Alaska Natives were particularly vulnerable to depressive symptoms after the oil spill.

Possible explanations for the vulnerability of Alaska Natives to this event include pre-disaster psychiatric condition as evidenced by high rates of alcohol abuse, psychiatric inpatient admission, and suicide, and the oil spill's effect on subsistence activities that provide the foundation for social support and community cohesion. Poor well-being and functioning are also of policy interest because of the societal costs due to loss of productivity, increased family burdens, and any associated use of health services. Palinkas et al. (1993) believed that the consistent pattern of increasing rates of psychiatric diagnosis with increasing exposure to the spill and subsequent clean-up efforts suggest a dose-response relationship.

One year after the *Exxon Valdez* spill, the prevalence of PTSD was associated with low family support among Alaska Natives but not among Euro-Americans (Palinkas et al., 2004). It was also associated with reported declines in subsistence activities and amount of social disruption in both ethnic groups. Among Alaska Natives, PTSD was significantly associated with participation in clean-up activities, reports of property lost or damaged as a result of the spill, and effects on hunting, fishing and gathering. Among Euro-Americans, PTSD was associated only with effects on hunting, fishing and gathering.

Alaska Natives reported unpleasant memories, attempts to avoid thinking about the past, avoidance of feelings, feeling cut off from others, inability to feel, trouble concentrating, and physical reactions. Compared to Euro-Americans, Alaska Natives were more likely to report unpleasant memories, nightmares and other bad dreams, attempts not to think about the past, having trouble concentrating and physical reactions. According to Palinkas et al. (2004) an event like the *Exxon Valdez* oil spill is sufficiently traumatic to warrant a diagnosis of PTSD if it threatens one's 'social integrity' through increased social conflict and isolation, even if the event does not result in loss of life. In this instance, the diagnosis transforms a social experience of conflict and isolation into a personal experience of disease.

Six years after the *Exxon Valdez* oil spill, symptoms of depression, anxiety and PTSD were still detectable in communities and associated with local conditions, loss of resources and avoidance coping strategies (Arata et al., 2000). Arata and co-workers reported clinically significant levels of anxiety (males, 23%; females, 13%), depression (males, 39%; females, 20%), and PTSD (male fishers, 34%; female fishers, 40%). Resource loss variables accounted for 30% of the variance in anxiety symptoms, with changes in physical health, changes in relations with non-relatives and investment without gain making significant contributions to the model. The high rates of distress observed suggest that there have been significant long-term effects of the *Exxon Valdez* oil spill on the mental health of commercial fishers in the sample. Arata et al. (2000) concluded that these results were consistent with research suggesting that technological disasters produce chronic social disruption.

Russell et al. (1996) concluded that survey and ethnographic data show a significant relationship between measures of exposure to the spill and clean-up and measures of GAD, PTSD, symptoms of depression, substance abuse, and domestic violence. Their analysis also indicated a significant relationship between exposure and social disruption following the oil spill and clean-up. There was much uncertainty about the present and future of indigenous and non-indigenous traditional ways of life after the spill. Fishers and others began to look to the clean-up as one means to cope with the economic consequences of a lost fishing and harvesting season. Some boat owners received contracts for clean-up and others did not. Some businesses lost money while others gained large profits. In some of the smaller, mostly indigenous, communities almost all able-bodied adults worked on the clean-up, disrupting local government functions and the usual cycle of daily life. Wages paid by Exxon were far greater than the average salaries of people in the area. This resulted in a 'collective trauma' defined as damage to the bonds that connect individuals within a social unit.

Indigenous communities had the highest overall exposure scores. Individuals were exposed to the effects of the spill if their fishing grounds were exposed, if they worked on the clean-up, or if in some other fashion they came into direct contact with the oil spill or its effect. Following the spill, 20% of people experienced GAD, 9.7% PTSD and 16.6% depressive symptoms. In general, indigenous people had higher rates of psychiatric conditions than non-indigenous people. Perceptions of drug and alcohol abuse and domestic violence, as well as perceptions of problems associated with these psycho-social conditions were significantly associated with exposure to the oil spill and clean-up.

According to Russell and co-workers, individuals felt threatened by another oil spill and its consequences. They relied upon community support systems for recovery from personal distress. As a result, harm to these social support systems impeded or delayed individual recovery. Damage to the ties that bind people together impedes cooperative action for the protection and safety of an entire town or village and may also impede the process of community recovery from the event itself. In this case, rather than cooperative pulling together, there was often conflict and divisiveness as well as charge and counter-charge among neighbours and friends about how some individuals were benefiting at the expense of others.

Tanker spill, M.V. Braer, Shetland Islands, 5 January 1993

Incident: The *Braer*, en route from Norway to Canada, was grounded on 5 January 1993, at Garth's Ness, Shetland Islands and spilled 85 000 t of light Gulfaks oil and 1600 t of fuel oil over the following week (Ritchie, 1997). This is the largest spill from a ship in northern waters reported to date. Oil was dispersed rapidly due to the violent weather but some of the oil came into contact with pens of Atlantic salmon (*Salmo salar*). The volatile nature of the oil and the extreme weather conditions resulted in it being blown over the land to the north and east of the vessel, exposing soil, crops, buildings, animals, and humans to hydrocarbons as vapour, droplets and oil-water emulsions. Dispersants were sprayed and fell over land as well as sea. Some of the constituents of the oil were known to be potentially carcinogenic.

Ecological impacts: Environmental studies were conducted including measurement of hydrocarbons in the water and in various organisms and examination of numerous species for indications of injury (Davies and Topping, 1997). Oil spray and vapour were carried inland resulting in land contamination, air pollution, and deposits on houses and cars (Harris, 1997). Fish and invertebrate animals became tainted and contaminated with PAHs at levels above those from reference sites. Fish lost the contaminants within a few weeks but some of the invertebrates remained contaminated for over a year. Penned salmon near the spill site were contaminated and tainted, with the tainting persisting for up to 30 weeks in the most affected locations. The mixed-function oxygenase system in the salmon was induced for a few weeks at sites near the spill. A few dead otters were found (half of them hit by cars) but oil was not established as the cause of death. Some seals had eye and nasal discharges and a small number may have had existing health conditions exacerbated by the oil.

Health effects: Campbell et al. (1993, 1994) studied acute symptoms among the population living within 4.5 km of the grounding (and subsequent break up) of the *Braer* in rural areas of the Shetlands. With the exception of somewhat higher incidences of headaches, throat irritation, and itchy eyes among the target population compared with the control population located 95 km from the scene of the spill, no significant differences were found for any of the biological markers investigated. Exposed participants also reported significant differences in mood change compared to before the spill. Differences of lower orders were found for tiredness, diarrhoea, nausea, wheezing, coughing, and chest ache.

The examinations for volatile substances in blood showed only butanol and tertiary butanol in all samples and ethanol in three exposed people. The higher haemoglobin concentrations in those exposed outside might be associated with their greater physical activities. Urinary hippurate:creatinine ratios greater than 1:1 suggest that the possibility of exposure to toluene (or other solvents) should be investigated.

A follow-up study five months after the oil spill, found that the exposed group was more likely to report being in poor or declining health than the controls, and that the exposed group also reported a higher incidence of weakness (Campbell et al., 1994).

Peak expiratory flow rate in schoolchildren living close to the *Braer* oil spill were measured and were within the normal range provided by the manufacturer of the peak flow meter (Crum, 1993). No deterioration was seen over the study period (3 days and 9–12 days after the ship wreck). Even children with asthma did not have an abnormal reading.

Tanker spill, Sea Empress, Milford Haven, Wales, 15 February 1996

Incident: On the evening of 15 February 1996, the *Sea Empress* laden with more than 130 000 t of light crude oil ran aground on rocks at the entrance to Milford Haven harbour in south-west Wales. Over the next week an estimated 72 000 t of crude oil and 360 t of heavy fuel oil were released into the sea, contaminating about 200 km of coastline.

Ecological impacts: The spill contaminated about 200 km of coastline and resulted in the precautionary closure of fishing for fish, shellfish, crustaceans and edible seaweeds for periods ranging from about 3 months (fish) to about 18 months (mussels) (Edwards and White, 1999). Sea bass (*Dicentrarchus labrax*) was the most important commercial species in the area and sampling young-of-the-year in 1996 revealed few of these fish in Milford Haven, nevertheless any effects of the oil were thought to be less significant than natural year-to-year variation (Lancaster et al., 1998). Data on plankton abundance in the southern Irish Sea since 1970 were available from previous monitoring and these revealed relatively little impact of the oil on these organisms (Batten et al., 1998). There were no obvious mass mortalities of fish after the spill but large numbers of dead or moribund bivalve molluscs, starfish, and heart-urchins were washed ashore. Amphipods disappeared from areas of the sea bed and were replaced largely by polychaetes (Edwards and White, 1999). Five years later, however, the amphipod fauna was recovering (Nikitik and Robinson, 2003). One particularly rare starfish, the cushion starfish (*Asterina phylactica*) had been studied in detail and was reduced from a pre-spill population of 150 in rock pools in West Angle Bay to a post-spill population of only 13; subsequent counts have indicated slow recovery (Edwards and White, 1999).

The most visible impacts were on seabirds: about 7000 oiled birds were collected onshore and an unknown number died at sea. The species most affected were the guillemots and scoter ducks. The numbers of birds using the area were greatly reduced in the following year (Edwards and White, 1999).

Blenny (*Lipophrys pholis*), plaice (*Pleuronectes platessa*) and the sand (common) dab (*Limanda limanda*) from oiled areas had higher levels of DNA adducts than fish from a clean area (Lyons et al., 1997; Harvey et al., 1999). Similar analyses of the invertebrates *Halichondria panacea* and blue mussel (*Mytilus edulis*), however, revealed no excess DNA adducts.

Health effects: Living in areas exposed to the crude oil spill was significantly associated with higher anxiety and depression scores, worse mental health, and self-reported headache, sore eyes, and sore throat after adjusting for age, sex, smoking status, anxiety, and the belief that oil had affected health. People living in exposed areas reported higher rates of physical and psychological symptoms than in control areas. Symptoms significantly associated with exposure after adjustment for anxiety and health beliefs were those expected from the known toxicological effect of oil, suggesting a direct health effect on the exposed population.

Whether the psychological effects are because of a toxic effect of the oil or are a generalized response to an environmental disaster with potential impacts on health, employment, income, and perception of the environment is unknown.

Tanker spill, Nakhodka, Oki Islands, Japan, 2 January 1997

Incident: On 2 January 1997, the Russian tanker, *Nakhodka*, carrying 19 000 t of Bunker C fuel oil broke up northeast of the Oki islands in the Sea of Japan. Most of the ship sank spilling more than 6000 t of oil into the sea (Morita et al., 1999). The ship broke up some 100 km offshore and crane barges with mechanical grabs were used in the recovery of oil at sea (Ansell et al., 2001). The bow section did not sink immediately but continued to drift and stranded on the rocky coast of Mikuni, Fukui Prefecture (Hayashi et al., 2000). Oil came ashore along more than 1000 km of shoreline (Ansell et al., 2001), much of it in rocky places where clean-up had to be done by hand because mechanized equipment could not operate there.

Ecological impacts: Surveys of benthic animals and algal communities in shallow sub-littoral rocky areas began about three months after the spill (Hayashi et al., 2000). This design was obviously limited by lack of comparative pre-spill data, but did allow detailed description and assessment of algal and benthic communities present after the spill. The populations appeared to be in good health indicating no drastic changes as a result of the spill. Some studies described bacterial strains found in the impacted shoreline and the use of selected bacteria as aids in the remediation of the sites. One such study identified several strains of bacteria isolated from heavily oiled coastal areas and noted that these could use long-chain aliphatic hydrocarbons but not aromatic hydrocarbons (Chaerun et al., 2004).

Health effects: Exposure to the oil and the subsequent clean-up efforts were suggested to inflict acute health problems on local residents. These included lower back and leg pain, headache, and symptoms related to the eyes (i.e., sore eyes, jiggling of vision, teary eyes) and throat (sore throat, scratchy throat, etc.). The proportion of people experiencing at least one symptom was 56.7% for men and 78.7% for women. Many symptoms started after the beginning of clean-up activities. The occurrence of lower back and leg pain and symptoms associated with the eyes decreased when clean-up operations were suspended. The greater the number of working days, the more subjects complained of symptoms, the more types of symptoms reported and the longer the duration of the symptoms. The average number of days worked on clean-up activities was 4.7 for men and 4.4 for women. Forty percent participated in cleanup for 1 to 2 days with 17% involved for more than 10 days. Older people participated in clean-up activities more frequently than the younger people. Only three people, out of 97 tested, showed an elevated level of hippuric acid; these elevations returned to normal in the second examination.

Components of hydrocarbons in the air in Anto were analyzed on three days and found to be very low and far below the occupationally acceptable limits for total

butane (500 ppm), ethylbenzene (100 ppm), *n*-pentane (300 ppm), xylene (100 ppm), benzene (10 ppm), and *n*-octane (300 ppm). Suspended particles and oil mist levels were also very low and below the occupationally acceptable levels (i.e., suspended particles, 2 mg/m^3; oil mist, 3 mg/m^3). Therefore, it was unlikely that these compounds from the oil had caused the physical symptoms directly by toxicological mechanisms. The concentration of benzo[*a*] pyrene and the level of mutagenicity in the extracts from the oil which reached the coast of Mikuni were reported to be higher than the average in soil samples obtained in Tokyo.

Direct exposure to oil was mainly from oil sticking to the face and hands of those on clean-up operations. Many of the symptoms that disappeared easily (e.g., flushed face and irritation of mucosa and skin) were considered to be acute responses to exposure to the oil. Other symptoms also disappeared quickly (e.g., musculoskeletal symptoms such as lower back pain) as they were the result of tiredness from clean-up jobs and bad weather. On the other hand, general symptoms such as fatigue, nausea, and headache, did not improve immediately.

Tanker spill, Erika, Belle-Ile, France, 12 December 12 1999

Incident: The tanker *Erica* broke in half about 70 km off the coast of Brittany on 12 December 1999, and the two halves sank about 10 km apart spilling around 19 000 t of heavy fuel oil into the sea. The oil drifted for several months and first reached the coast of France on December 23. Much of the oil was recovered, initially at sea but mainly through intensive shoreline clean-up operations that extended throughout 2000 and 2001 (Laubier et al., 2004).

Ecological effects: Several studies have described the dispersal and degradation of oil from the *Erika* (e.g., Tronczynski et al., 2004). The most obviously affected fauna were the seabirds wintering in the Bay of Biscay. Laubier et al. (2004) reported that between 64 000 and 125 000 common guillemots (*Uria aalge*) were killed, with around a third less than a year old. Studies of birds were quite rigorous because of monitoring data available from 1980 to 1999 for comparison with post-spill data. Despite the kill of guillemots, population surveys did not detect a drop in abundance of this species; indeed an increase was suggested in the northern part of the Bay of Biscay although it was not significant statistically. Abundance Figures for the more polluted north of the Bay of Biscay fell significantly for loons (*Gavia* spp.), common scoters (*Melanitta nigra*), razorbills (*Alca torda*) and fulmars (*Fulmarus glacialis*). No species increased significantly in the north. Loons and kittiwakes (*Rissa tridactyla*) decreased in the central area but fulmars, guillemots and great skua (*Stercorarius skua*) increased. In the unpolluted south, only gannets (*Sula bassana*) changed in abundance and they increased (Castege et al., 2004). It was suggested that birds may have moved among different areas in response to levels of oil pollution.

Marine mammals are relatively rare in the Bay of Biscay with the exception of dolphins (*Dolphinus delphis*) and no overt symptoms of oil exposure were observed. The numbers of marine mammals washed up on shorelines were within normal ranges and personnel involved in shoreline clean-up would almost certainly have noticed if unusual numbers had occurred. There was no indication of oiling of coast-dwelling otters or the consequent hypothermia observed elsewhere although porphyrin levels were increased in otter faeces (Ridoux et al., 2004), a possible biomarker of effects.

The spill occurred at a time when sole (*Solea solea*) aggregate on spawning grounds at 40–100 m depth and so the young produced from that spawning (the 2000 year class) were the most studied (Claireaux et al., 2004). No effects on abundance or growth of the 2000 year class were demonstrated although some sub-lethal responses were shown.

There was widespread mortality of sea urchins and the contamination of bivalves prompted experiments by Lemiere et al. (2005) who took mussels (*Mytilus* sp.) contaminated to varying degrees by oil from the *Erika* and fed them to rats. After feeding periods of 15 to 30 days, the rats were examined for evidence of DNA damage. Moderate DNA damage (breaks by comet assay) was demonstrated in liver and to a lesser extent in bone marrow but not in peripheral blood. The implications of these results are speculative but clearly merit further research.

Health effects: CIRE-Ouest (2000) reported that approximately 75% of clean-up workers had skin contact with fuel oil, despite the protective clothing (gloves, waterproof clothing and boots) worn by more than nine out of ten people. Of the 1465 individuals studied, 110 (5.6% of the volunteers and 11% of the professionals) reported having been injured in the clean-up work and 772 (50% of the volunteers and 57% of the professionals) reported that they developed one or more health problems during or immediately after their activities. The three most frequently cited health problems were lumbar pain, headaches and skin irritations. In most cases, these problems were not serious. Of the population studied, only 2 people were hospitalized (for joint injuries).

Among the professional clean-up workers, there was a higher risk of injury for women than men; among the volunteers, there was a higher risk of lower back pain and sympathetic nervous system problems for women than men. Age was also an explanatory factor for variation in the occurrence of some pathologies among the professionals: for professionals over 30 years old, there was an increased risk of skin and eye irritations and of sympathetic nervous system problems.

For both volunteers and professionals, the duration of the activities (in days) was identified as a risk factor for all the health problems that arose. Carrying out multiple activities promoted the occurrence of skin irritations and sympathetic nervous system problems among the volunteers. For volunteers, specific activities entailing a higher risk of injury were cleaning of wharfs, jetties, equipment, clothing and machinery; for professionals, they were the handling of birds and heavy equipment. The cleaning of sandy and rocky beaches increased the risk of lower back pain and skin irritations for the professionals. The cleaning of wharfs, jetties, equipment, clothing and machinery and the use of heavy equipment also led to an increased risk of skin irritations for the professionals.

Discomfort as a result of odours was a risk factor for the occurrence of eye irritations and sympathetic nervous system problems among the volunteers, and of sympathetic nervous system and skin problems among the professionals. Skin contact with fuel oil on hands and arms meant an increased risk of skin irritations for all workers.

Contact on the head also meant an increased risk of eye irritations among professionals. An increased risk of injury, lower back pain and skin irritations was identified for the volunteers who reported that they had not been informed of the on-site precautions to take.

While this study did not reveal serious health problems in connection with the clean-up activities; a large number of professional and volunteer workers experienced minor problems. Prevention measures could be improved with better communication of exposure safety information to all workers (the need to avoid skin contact with fuel oil and to limit the duration of activity, especially in the case of women). People with pre-existing health problems, including osteoarticular problems, should be required to consult their attending physician before undertaking this type of activity.

Tanker spill, Baltic Carrier, Falster Island (western Baltic), Denmark, 29 March 2001

Incident: The oil tanker *Baltic Carrier* collided with the freighter *Tern* off the Danish island of Falster (west Baltic Sea) on 29 March 2001 after the failure of a steering mechanism in the *Baltic Carrier*. About 2700 t of bunker oil escaped into the sea and quickly drifted ashore on 30–50 km of Danish shorelines, mostly shingle beaches and wetlands and including a bird sanctuary. A European task force on site reported that about 2135 t of oil had been recovered by 4 April, about half of this at sea (Vincent et al., 2001). A good description of the event was provided by Pecséli et al. (2002).

Ecological impacts: The toxicity of the oil from the *Baltic Carrier* was measured with the copepod *Acartia tonsa* in experimental exposures that resulted in significant acute mortality of test animals at concentrations below 1 mg/L (Pecséli et al., 2002). Exposures of benthic organisms to oil-treated sediments did not cause acute mortality but did result in chronic mortality and sub-lethal effects. Environmental concentrations of a range of PAHs were measured in seawater, sediment, mussels and other biota. The levels of ΣPAH in water were initially highly elevated, 641–12 833 ng/L in April, but fell to 28.9–641 ng/L by June. Levels in sediment declined more slowly, from 89.8–4119 μg/kg (dw) in April, to 45.8–1304 μg/kg dw in June, and 81.1–1018 μg/kg dw in December. In blue mussels, levels ranged from 132 to 1031 μg/kg ww (mean 304 μg/kg ww) in April and were little changed at 185–894 μg/kg (mean 207 μg/kg) in December. This may also be partly due to an earlier observed natural increase in polycyclic aromatic compound levels in mussels from summer to winter. However, mussels also showed elevated concentrations of alkylated dibenzothiophenes, indicative of recent petroleum pollution. In shrimp, the total PAH levels were first elevated but then decreased 93% during the first year after the oil spill. In particular, the water soluble fraction of the *Baltic Carrier* oil (i.e. the light and alkylated PAH compounds) decreased in shrimp tissue during 13 months after the accident. Flounders had increased concentrations of PAHs long after the accident. After one year, the PAH levels had dropped by about 50%, far less than for shrimps. As fish are known to have an effective PAH metabolism, this finding indicates continuous exposure of these substances. High PAH levels were also measured in the sediment, the habitat of the flounder. In contrast, farmed trout remained unaffected by oil.

Pecséli et al. (2002) collected 1750 dead or moribund birds and estimated that the actual number may have been as much as ten times higher. The European task force reported that 2300 dead birds were found at sea or onshore within the first few days. Most locally breeding bird populations recovered within a year except for breeding waders at the most heavily oiled site.

Health effects: No reports available.

References

Aas, E., G. Jonsson, R.C. Sundt, S. Westerlund and S. Sanni, 2007. Monitoring of PAH metabolites and metals in bile from caged Atlantic cod (*Gadus morhua*) and wild pelagic fish along expected contaminant gradients in the North Sea. In: K. Hylland, T. Lang, D. Vethaak (Eds.). Biological Effects of Contaminants in Marine Pelagic Ecosystems, pp. 263-276. SETAC special publication.

Aastrup, P., 2000. Responses of West Greenland caribou to the approach of humans on foot. Polar Research, 19:83-90.

ACIA, 2004. Impacts of a Warming Arctic. Arctic Climate Impact Assessment. Cambridge University Press, 139p.

ACIA, 2005. Arctic Climate Impact Assessment. Cambridge University Press, v+1042p.

Adams, W.A., 1975. Light intensity and primary productivity under sea ice containing oil. Department of the Environment, Victoria, B.C., Canada, Beaufort Sea Technical Report 29, 156p.

Addison, R.F. and A.J. Edwards, 1988. Hepatic microsomal monooxygenase activity in flounder *Platichthys flesus* from polluted sites in Langesundjford and from mesocosms experimentally dosed with diesel oil and copper. Marine Ecology Progress Series, 46:51-54.

Addison, R.F., K. Doe and A. Edwards, 1984. Effects of oil-based drilling mud cuttings on winter flounder (*Pseudopleuronectes americanus*): absence of acute toxicity or mixed function oxidase induction. Canadian Technical Report of Fisheries and Aquatic Sciences, 1307, 19p.

Ahnoff, M. and G. Eklund, 1978. Oil contamination of Melville Bay water after the *Potomac* accident in August 1977. Report XX, Department of Analytical and Marine Chemistry, Chalmers University of Technology and University of Goteborg, 12p + figs and tables.

Ainley, D.G., C.R. Grau, T.E. Roudybush, S.H. Morrell and J.M. Utts, 1981. Petroleum ingestion reduces reproduction in Cassin's auklets. Marine Pollution Bulletin, 12:314-317.

Akkanen, J. and J.V.K. Kukkonen, 2003a. Measuring the bioavailability of two hydrophobic organic compounds in the presence of dissolved organic matter. Environmental Toxicology and Chemistry, 22:518-524.

Akkanen, J. and J.V.K. Kukkonen, 2003b. Biotransformation and bioconcentration of pyrene in *Daphnia magna*. Aquatic Toxicology, 64:53-61.

Alaska Department of Fish and Game, 2003. Caribou Management Report of survey-inventory activities. 1 July 2000 - 30 June 2002. In: C. Healy (Ed.). Wildlife Management Report. Juneau, Alaska. 203p.

Albers, P.H., 1980. Transfer of crude oil from contaminated water to bird eggs. Environmental Research, 22:307-314.

Albers, P.H., 2003. Petroleum and individual polycyclic aromatic hydrocarbons. In: D.J. Hoffman, B.A. Rattner, G.A. Burton Jr., J. Cairns Jr. (Eds.). Handbook of Ecotoxicology, pp. 341-371. Second Edition.

Albert, T., 1997. Comments by subsistence hunters regarding impacts of seismic noise to fall migrating bowhead whales and to the subsistence hunt. In: T. Newbury, K.L. Mitchell, C.T. Mitchell, P. Barton (Eds.). Arctic Seismic Synthesis and Mitigating Measures Workshop, pp. 27-44. U.S. Department of the Interior, Minerals Management Service, Anchorage, Alaska.

Alerstam, T. and G. Högstedt, 1982. Bird migration and reproduction in relation to habitats for survival and breeding. Ornis Scandinavica, 13:25-37.

AMAP, 1998. AMAP Assessment Report: Arctic Pollution Issues. Arctic Monitoring and Assessment Programme, Oslo, Norway. xii+859p.

AMAP, 2003. AMAP Assessment 2002: Human Health in the Arctic. Arctic Monitoring and Assessment Programme, Oslo, Norway. xiv+137p.

AMAP, 2004a. AMAP Assessment 2002: Radioactivity in the Arctic. Arctic Monitoring and Assessment Programme, Oslo, Norway. xi+100p.

AMAP, 2004b. AMAP Assessment 2002: Persistent Organic Pollutants in the Arctic. Arctic Monitoring and Assessment Programme, Oslo, Norway. xvi+310p.

AMAP, 2004c. Persistent Toxic Substances, Food Security and Indigenous Peoples of the Russian North. Final Report. Arctic Monitoring and Assessment Programme, Oslo, Norway. 192p.

AMAP, 2005. AMAP Assessment 2002: Heavy Metals in the Arctic. Arctic Monitoring and Assessment Programme, Oslo, Norway. xvi+265p.

Amstrup, S.C., 1993. Human disturbances of denning polar bears in Alaska. Arctic, 46:246-250.

Amstrup, S.C., 2000. Polar bear. In: J.C. Truett, S.R. Johnson (Eds.). The Natural History of an Arctic Oil Field. Development and the Biota, pp. 133-157. Academic Press, San Diego.

Anderson, J.W., J.M. Neff, B.A. Cox, H.E. Tatem and G.M. Hightower, 1974. Characteristics of dispersions and water-soluble extracts of crude and refined oils and their toxicity to estuarine crustaceans and fish. Marine Biology, 27:75-88.

Anderson, J.W., D.B. Dixit, G.S. Ward and R.S. Foster, 1977. Effects of petroleum hydrocarbons on the rate of heart beat and hatching success of estuarine fish embryos. In: F.J. Vernberg, A. Calabrese, F.P. Thurberg, W.B. Vernberg (Eds.). Physiological Responses of Marine Biota to Pollutants, pp. 241-258. Academic Press.

Anderson, J.W., R. Riley, S. Kiesser and J. Gurtisen, 1987. Toxicity of dispersed and undispersed Prudhoe Bay crude oil fractions to shrimp and fish. In: 1987 Oil spill conference: Prevention, behavior, control, cleanup, pp. 235-240. American Petroleum Institute, Washington.

Andres, B.A., 1999. Effects of persistent shoreline oil on breeding success and chick growth in black oystercatchers. Auk, 116:640-650.

Angliss, R.P. and R.B. Outlaw, 2005. Alaska Marine Mammal Stock Assessments, 2005. NOAA Technical Memorandum, NMFS-AFSC-161. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Alaska Fisheries Science Center, National Marine Mammal Laboratory. Seattle Washington.

Anker-Nilssen, T., 1987. Metoder til konsekvensanalyser olje/sjøfugl. Viltrapport 44, Norsk Institutt For Naturforskning, Trondheim.

Anker-Nilssen, T. and O.W. Røstad, 1982. Oljekatastrofen i Skagerak ved Årsskiftet 1980/81 omfang og undersøkelser. Vår Fågelfauna, 5:82-90.

Ankley, G.T., R.J. Erickson, G.L. Phipps, V.R. Mattson, P.A. Kosian, B.R. Sheedy and J.S. Cox, 1995. Effects of light intensity on the phototoxicity of fluoranthene to a benthic macroinvertebrate. Environmental Science and Technology, 29:2828-2833.

Ansell, D.V., B. Dicks, C.C. Guenette, T.H. Moller, R.S. Santner and I.C. White, 2001. A review of the problems posed by spills of heavy fuel oils. Presented at the 2001 International Oil Spill Conference, Tampa, Florida, 16p.

API, 2001. Chemical Human Health Hazards Associated with Oil Spill Response. Regulatory Analysis and Scientific Affairs. American Petroleum Institute, Washington, D.C. Publication No. 4689. 40p.

Arata, C.M., J.S. Picou, G.D. Johnson and T.S. McNally, 2000. Coping with technological disaster: an application of the conservation of resources model to the *Exxon Valdez* oil spill. Journal of Traumatic Stress, 13:23-39.

Arctic Council, 2002. Arctic Offshore Oil and Gas Guidelines. National Oceanic and Atmospheric Administration, U.S. Department of Commerce, 64p.

Arfsten, D.P., D.J. Schaeffer and D.C. Mulveny, 1996. The effects of near ultraviolet radiation on the toxic effects of polycyclic aromatic hydrocarbons in animals and plants: A review. Ecotoxicology and Environmental Safety, 33:1-24.

Arhelger, S.D., B.R. Robertson and D.K. Button, 1977. Arctic hydrocarbon biodegradation. In: D.A. Wolfe (Ed.). Fate and Effects of Petroleum Hydrocarbons in Marine Organisms and Ecosystems, pp. 270-275. Pergamon Press.

Armsworthy, S.L., P.J. Cranford and K. Lee, 2005a. Offshore Oil and Gas Environmental Monitoring. **Approaches and Technologies.** Battelle Press. 631p.

Armsworthy, S.L., P.J. Cranford, K. Lee and T. King, 2005b. Chronic effects of synthetic drilling mud on sea scallops (*Placopecten magellanicus*). In: S.L. Armsworthy, P.J. Cranford, K. Lee (Eds.). Offshore Oil and Gas Environmental Effects Monitoring. Approaches and Technologies, pp. 243-265. Battelle Press.

Astrup, J. and B. Mohl, 1993. Detection of ultrasound by the cod *Gadus morhua*. Journal of Experimental Biology, 182:71-80.

Atlas, R.M., 1977. Studies on petroleum biodegradation in the Arctic. In: D.A. Wolfe (Ed.). Fate and Effects of Petroleum Hydrocarbons in Marine Organisms and Ecosystems, pp. 261-269. Pergamon Press.

Atlas, R.M., 1981. Microbial degradation of petroleum hydrocarbons: an environmental perspective. Microbiological Reviews, 45:180-209.

Atlas, R.M., 1985. Effects of hydrocarbons on microorganisms and petroleum biodegradation in arctic ecosystems. In: F.R. Englehardt (Ed.). Petroleum Effects in the Arctic Environment, pp. 63-99. Elsevier Science.

Atlas, R.M., A. Horowitz and M. Busdosh, 1978. Prudhoe crude oil in arctic marine ice, water and sediment ecosystems: degradation and interactions with microbial and benthic communities. Journal of the Fisheries Research Board of Canada, 35:585-590.

ATSDR, 1995. Toxicological profile for polycyclic aromatic hydrocar-

bons. Agency for Toxic Substances and Disease Registry. U.S. Department of Health and Human Services. 470p.

ATSDR, 2007. Toxicological Profile Information Sheet. Agency for Toxic Substances and Disease Registry. http://www.atsdr.cdc.gov/toxpro2.html

Auerbach, N.A., M.D. Walker and D.A. Walker, 1997. Effects of road dust disturbance on substrate and vegetation properties in arctic tundra. Ecological Applications, 7:218-235.

Baelum, J., L.B. Andersen, G.R. Lundqvist, L. Molhave, O.F. Pedersen and M. Vaeth, 1985. Responses of solvent-exposed printers and unexposed controls to six-hour toluene exposure. Scandinavian Journal of Work, Environment and Health, 11:271-280.

Bailey, E.P. and C.H. Davenport, 1972. Dieoff of common murres on the Alaskan Peninsula and Unimak Island. Condor, 74:214-219.

Baine, M., 2002. The North Sea rigs-to-reefs debate. ICES Journal of Marine Science, 59(suppl.):s277-s280.

Baird, P.H., 1990. Concentrations of seabirds at oil-drilling rigs. The Condor, 92:768-771.

Baker, J.M., 1970. The effects of oils on plants. Environmental Pollution, 1:27-44.

Baker, J.R., 1981. Otter *Lutra lutra* L. mortality and marine oil pollution. Biological Conservation, 20:311-321.

Baker, J.M., R.B. Clark, P.F. Kingston and R.H. Jenkins, 1990. Natural recovery of cold water marine environments after an oil spill. Presented at the 13th Arctic and Marine Program Technical Seminar, 111p.

Bakke, T., J.A. Berge, K. Naes, F. Oreld, L.-O. Reiersen and K. Bryne, 1988. Long-term recolonization and chemical change in sediments contaminated with oil-based drill cuttings. In: F.R. Engelhardt, J.P Ray, A.H. Gillam (Eds.). Drilling Wastes, pp. 521-544. Elsevier Applied Science.

Ballachey, B.E., J.L. Bodkin and A.R. DeGange, 1994. An overview of sea otter studies. In: T.R. Loughlin (Ed.). Marine Mammals and the *Exxon Valdez*, pp. 47-59. Academic Press.

Ballard, W.B., M.A. Cronin, M.D. Robards and E.H. Follmann, 2000. Body sizes, ages, reproductive status, and sex ratios of arctic foxes, *Alopex lagopus*, in the Prudhoe Bay oil field, Alaska. Canadian Field-Naturalist, 114:493-494.

Banfield, A.W.F., 1974. The Mammals of Canada. University of Toronto Press. 439p.

Barron, M.G., T. Podrabsky, S. Ogle and R.W. Ricker, 1999. Are aromatic hydrocarbons the primary determinant of petroleum toxicity to aquatic organisms? Aquatic Toxicology, 46:253-268.

Barron, M.G., M.G. Carls, J.W. Short and S.D. Rice, 2003. Photo-enhanced toxicity of aqueous phase and chemically dispersed weathered Alaska North Slope oil to Pacific herring eggs and larvae. Environmental Toxicology and Chemistry, 22:650-660.

Barsiene, J., J. Lazutka, J. Syvokiene, V. Dedonyte, A. Rybakovoas, E. Bagdonas, A. Bjornstad and O.K. Andersen, 2004. Analysis of micronuclei in blue mussels and fish from the Baltic and North Seas, Environmental Toxicology, 19:365-371.

Baskin, L.M., unpubl. MS. Dynamics of wild and domestic reindeer numbers in Russia in the late 20th century. Institute of Ecology and Evolution, Russian Academy of Sciences, Moscow.

Batten, S.D., R.J.S. Allen and C.O.M. Wotton, 1998. The effects of the *Sea Empress* oil spill on the plankton of the southern Irish Sea. Marine Pollution Bulletin, 36:764-774.

Baumann, P.C., 1984. Cancer in wild freshwater fish populations with emphasis on the Great Lakes. Journal of Great Lakes Research, 10:251-253.

Baumann, P.C., W.D. Smith and W.K. Parland, 1987. Tumor frequencies and contaminant concentrations in brown bullheads from an industialized river and a recreational lake. Transactions of the American Fisheries Society, 116:79-86.

Baussant, T., S. Sanni, G. Jonsson, A. Skadsheim and J.F. Borseth, 2001a. Bioaccumulation of polycyclic aromatic compounds, 1: Bioconcentration in two marine species and in semi-permeable membrane devices during chronic exposure to dispersed crude oil. Environmental Toxicology and Chemistry, 20:1175-1184.

Baussant, T., S. Sanni, A. Skadsheim, J. Johsson, J.F. Borseth and B. Gaudebert, 2001b. Bioaccumulation of polycyclic aromatic compounds, 2: Modeling bioaccumulation in marine organisms chronically exposed to dispersed oil. Environmental Toxicology and Chemistry, 20:1185-1195.

Becker, S., C.J. Halsall, W. Tych, H. Hung, S. Attewell, P. Blanchard, H. Li, P. Fellin, G. Stern, B. Billeck and S. Friesen, 2006. Resolving the long-term trends of polycyclic aromatic hydrocarbons in the Canadian arctic atmosphere. Environmental Science and Technology, 40:3217-3222.

Beckett, Capt. C.J., 1979. The grounding of the *Imperial St. Clair* – a case history of contending with oil in ice. 1979 Oil Spill Conference (Prevention, Behaviour, Control, Cleanup), pp. 371-375. American Petroleum Institute, Publication No. 4284.

Beckett, K.J., R.J. Aulerich, L.K. Duffy, J.S. Patterson and S.J. Bursian, 2002. Effects of dietary exposure to environmentally relevant concentrations of weathered Prudhoe Bay crude oil in ranch-raised mink (*Mustela vison*). Bulletin of Environmental Contamination and Toxicology, 69:593-600.

Ben-David, M., T.M. Williams and O.A. Ormseth, 2000. Effects of oil ingestion on exercise physiology and diving behavior in river otters: A captive study. Canadian Journal of Zoology, 78:1380-1390.

Ben-David, M., L.K. Duffy and R.T. Bowyer, 2001. Biomarker responses in river otters experimentally exposed to oil contamination. Journal of Wildlife Diseases, 37:489-508.

Bengtsson, L. and H. Berggren, 1972. The bottom fauna in an oil-contaminated lake. Ambio, 1:141-144.

Benville, P.E. Jr., T.G. Yocom, P. Nunes and J.M. O'Neill, 1981. Simple, continuous-flow systems for dissolving the water-soluble components of crude oil into seawater for acute or chronic exposure of marine organisms. Water Research, 15:1197-1204.

Berge, J.A., K.I. Johannessen and L.-O. Reiersen, 1983. Effects of the water soluble fraction of North Sea crude oil on the swimming activity of the sand goby, *Potatoschistus minutus* (Pallas). Journal of Experimental Marine Biology and Ecology, 68:159-167.

Berger, J.T.R., 1983. Energy, resource development and human values. Canadian Journal of Community Mental Health, special supplement no. 1:21-31.

Berger, J., A. Hoylman and T. Weber, 2001. Perturbation of vast ecosystems in the absence of adequate science: Alaska's Arctic refuge. Conservation Biology, 15:539-541.

Bergerud, A.T., 1978. Caribou. In: J.L. Schmidt, D.L. Gilbert (Eds.). Big Game of North America, Ecology and Management, pp. 83-102. Stackpole Books.

Berman, M.S. and D.R. Heinle, 1980. Modification of the feeding behaviour of marine copepods. Marine Biology, 56:59-64.

Berner, J., 2003. Health Status of the Arctic Residents. In: AMAP Assessment 2002: Human Health in the Arctic, pp. 10-20. Artic Monitoring and Assessment Programme, Oslo, Norway.

Berry, J.A. and P.G. Wells, 2004. Integrated fate modelling for exposure assessment of produced water on the Sable Island bank (Scotian Shelf, Canada). Environmental Toxicology and Chemistry, 23:2483-2493.

Berry, J.A. and P.G. Wells, 2005. Environmental modelling of produced water dispersion with implications for environmental effects monitoring design. In: S.L. Armsworthy, P.J. Cranford, K. Lee (Eds.). Offshore Oil and Gas Environmental Effects Monitoring. Approaches and Technologies, pp. 111-129. Battelle Press.

Berry, W.O., J.D. Brammer and D.E. Bee, 1978. Uptake of water-soluble gasoline fractions and their effect on oxygen consumption in aquatic stages of the mosquito (*Aedes aegypti* (L.)). Environmental Pollution, 15:1-22.

Beserati, N., 2001. Molecular dosimetry studies of smoking - induced carcinogenesis in target and surrogate tissues of humans. Ph.D. Dissertation, University of Maastricht.

Bevanger, K., 1998. Biological and conservation aspects of bird mortality caused by electrical power lines: A review. Biological Conservation, 86:67-76.

Bickham, J.W., J.A. Mazet, J. Blake, M.J. Smolen, Y.G. Lou and B.E. Ballachey, 1998. Flow cytometric determination of genotoxic effects of exposure to petroleum in mink and sea otters. Ecotoxicology, 7:191-199.

Biggar, K.W., S. Haidar, M. Nahir and P.M. Jarrett, 1998. Site investigations of fuel spill migration into permafrost. Journal of Cold Regions Engineering, 12:84-104.

Bilbao, E., A. Ibabe, B. Zaldibar, M. Soto, M.P. Cajaraville, I. Cancio and I. Marigómez, 2007a. Cell- and tissue-level biomarkers of pollution in mussels (*Mytilus edulis*) and cod (*Gadus morhua*) caged along a pollution gradient in Statfjord (North Sea). In: K. Hylland, A.D. Vethaak, T. Lang (Eds.). Biological Effects of Contaminants in Marine Pelagic Ecosystems, pp 215-164. SETAC special publication.

Bilbao, E., M. Soto, M.P. Cajaraville, I. Cancio and I. Marigómez, 2007b. Cell and tissue-level biomarkers of pollution in feral pelagic fish, herring (*Clupea harengus*) and saithe (*Pollachius virens*) from the North Sea. In: K. Hylland, A.D. Vethaak, T. Lang (Eds.). Biological Effects of Contaminants in Marine Pelagic Ecosystems, pp. 121-142. SETAC special publication.

Binderup, M.-L., L. Duedahl-Olesen, S. Einarsson, B. Fabech, A.-K. Lundebye Haldorsen, H. Johnsson, H.K. Knutsen, A.K. Müller, P.J. Vuorinen and M.L. Wiborg, 2004. The effect of oil spills on seafood safety. An example of the application of the Nordic risk analysis model. Nordic Council of Ministers, Copenhagen, 214p.

Birtwell, I.K. and C.D. McAllister, 2002. Hydrocarbons and their effects

on aquatic organisms in relation to offshore oil and gas exploration and oil well blowout scenarios in British Columbia, 1985. Canadian Technical Report of Fisheries and Aquatic Sciences, 2391, 52p.

Birtwell, I.K., R. Fink, D. Brand, R. Alexander and C.D. McAllister, 1999. Survival of pink salmon (*Oncorhynchus gorbuscha*) fry to adulthood following a 10-day exposure to the aromatic hydrocarbon water-soluble fraction of crude oil and release to the Pacific Ocean. Canadian Journal of Fisheries and Aquatic Sciences, 56:2087-2098.

Bjarti, T., 2002. An experiment on how seismic shooting affects caged fish. Faroese Fisheries Laboratory (University of Aberdeen), Aug 16, 2002. 41p.

Bjerregaard, P., 2003. Ethno-cultural adaptation of the peoples of the Arctic region. In: AMAP Assessment 2002: Human Health in the Arctic, pp. 6-9. Artic Monitoring and Assessment Programme, Oslo, Norway.

Bjorge, R.R., 1987. Bird kill at an oil industry flare stack in northwest Alberta. Canadian Field-Naturalist, 101:346-350.

Black, J.J. and P.C. Baumann, 1991. Carcinogens and cancers in freshwater fishes. Environmental Health Perspectives, 90:27-33.

Blaxter, J.H.S., J.A.B. Gray and E.J. Denton, 1981. Sound and startle responses in herring shoals. Journal of the Marine Biological Association of the UK, 61:851-859.

Blenkinsopp, S., P. Bolieau, D. Kyle, G. Sergy and M. Fingas, 1996. How to prepare water accommodated fractions from petroleum hydrocarbons for use in aquatic toxicity testing – the basics. Proceedings of the 19th Arctic and Marine Oil spill Program, pp. 515-528. Technical Seminar.

BLM, 2002. Final Environmental Impact Statement: Renewal of the Federal Grant for the Trans-Alaska Pipeline System Right-of-Way. U.S. Department of the Interior Bureau of Land Management, November 2002. BLM/AK/PT-03/005+2880+990. 3113 p. http://tapseis.anl.gov/documents/eis/index.cfm

BLM, 2005. Northeast National Petroleum Reserve – Alaska. Final. Amended Integrated Activity Plan/ Environmental Impact Statement. U.S. Department of the Interior, Bureau of Land Management. www.blm.gov/ak/nenpraeis/index.html

Blumer, M., G. Souza and J. Sass, 1970. Hydrocarbon pollution of edible shellfish by an oil spill. Marine Biology, 5:195-202.

Blundon, D., M. Dale, S. Goudey and J. Hoddinott, 1987. Effects of oil and oil spill chemicals on shoreline plants of northern freshwater ecosystems. In: J. Vandermeulen, S. Hrudey (Eds.). Oil in Freshwater: Chemistry, Biology, Countermeasure Technology, pp. 403-409. Pergamon Press.

Bodkin, J.L., B.E. Ballachey, T.A. Dean, A.K. Fukuyama, S.C. Jewett, L. McDonald, D.H. Monson, C.E. O'Clair and G.R. VanBlaricom, 2002. Sea otter population status and the process of recovery from the 1989 *Exxon Valdez* oil spill. Marine Ecology Progress Series, 241:237-253.

Boehm, P.D., G.S. Douglas, J.S. Brown, D.S. Page, A.E. Bence, W.A. Burns and P.J. Mankiewicz, 2000. Comment on 'natural hydrocarbon background in benthic sediments of Prince William Sound, Alaska: oil vs. coal'. Environmental Science and Technology, 34:2064-2065.

Boersma, P.D., E.M. Davies and W.V. Reid, 1988. Weathered crude oil effects on chicks of fork tailed storm petrels (*Oceanodroma furcata*). Archives of Environmental Contamination and Toxicology, 17:527-531.

Boertmann, D., A. Mosbech, K. Falk and K. Kampp, 1996. Seabird Colonies in Western Greenland (60º - 79º 30´ N. lat.). National Environmental Research Institute, NERI Technical Report no. 170. 148p.

Bohle, B., 1986. Avoidance of petroleum hydrocarbons by the cod (*Gadus morhua*). Fiskeridirektoratets Skrifter. Serie Ernaering, 18:97-112.

Boitsov, S., S. Meier, J. Klungsoyr and A. Svardal, 2004. Gas chromatography-mass spectrometry analysis of alkylphenols in produced water from offshore oil installations as pentafluorobenzoate derivatives. Journal of Chromatography A, 1059:131-141.

Bonsdorff, E., 1981. The *Antonio Gramsci* oil spill. Impact on the littoral and benthic ecosystems. Marine Pollution Bulletin, 12:301-305.

Boogaard, P.J., and N.J. van Sittert, 1995. Urinary 1-hydroxypyrene as a biomarker of exposure to PAHs in workers in petrochemical industries: baseline values and dermal uptake. Science of the Total Environment, 163:203-209.

Booman, C., J. Dalen, H. Leivestad, A. Levson, T. van der Meeren and K. Toklum, 1996. Effecter av luftkanonshyting pa egg, larver og yngel. Fisken og Havet, 1996:1-83. (Norwegian with English summary)

Borseth, J.F. and K.F. Tollefsen, 2004. Water column monitoring 2003. Summary Report. Report RF 2004/039, RF Akvamiljo, Norway, 28p.

Bosetti, C., P. Boffetta and C. La Vecchia, 2006. Occupational exposures to polycyclic aromatic hydrocarbons, and respiratory and urinary tract cancers: a quantitative review to 2005. Annals of Oncology, 8:431-446.

Bourne, W.R.P., 1968. Oil pollution and bird populations. In: J.D. Carthy, D.R. Arther (Eds.). The Biological Effects of Oil Pollution on Littoral Communities, pp. 99-121. Suppl. vol. 2. Field Studies Council, London.

Bowling, J.W., G.J. Leversee, P.F. Landrum and J.P. Giesy, 1983. Acute mortality of anthracene-contaminated fish exposed to sunlight. Aquatic Toxicology, 3:79-90.

Bowyer, R.T., J.W. Testa, J.B. Faro, C.C. Schwartz and J.B. Browning, 1994. Changes in diets of river otters in Prince William Sound, Alaska: Effects of the *Exxon Valdez* oil spill. Canadian Journal of Zoology, 72:970-976.

Bowyer, R.T., J.W. Testa and J.B. Faro, 1995. Habitat selection and home ranges of river otters in a marine environment: Effects of the *Exxon Valdez* oil spill. Journal of Mammalogy, 76:1-11.

Bowyer, R.T., G.M. Blundell, M. Ben-David, S.C. Jewett, T.A. Dean and L.K. Duffy, 2003. Effects of the *Exxon Valdez* oil spill on river otters: Injury and recovery of a sentinel species. Wildlife Monographs, 153:1-53.

Braddock, J.F. and K.A. McCarthy, 1996. Hydrologic and microbiological factors affecting persistence and migration of petroleum hydrocarbons spilled in a continuous-permafrost region. Environmental Science and Technology, 30:2626-2633.

Braddock, J.F., J.E. Lindstrom and R.C. Prince, 2003. Weathering of a subarctic oil spill over 25 years: The Caribou-Poker Creeks research watershed experiment. Cold Regions Science and Technology, 36:11-23.

Braddock, J.F., K.A. Gannon and B.T. Rasley, 2004. Petroleum hydrocarbon-degrading microbial communities in Beaufort-Chukchi Sea Sediments. Report OCS Study MMS 2004-061, 43p.

Brandal, P.O., O. Grahl-Nielsen, T. Neppelbert, K.H. Palmork, K. Westrheim and S. Wilhelmsen, 1976. Oil-tainting of fish, a laboratory test on salmon and saithe. ICES report C.M. 1976/E:33.

Brannon, E.L., L.L. Moulton, L.G. Gilbertson, A.W. Maki and J.R. Skalski, 1995. An assessment of oil spill effects on pink salmon populations following the *Exxon Valdez* oil spill – Part 1: Early life history. In: P.G. Wells, J.N. Butler, J.S. Hughes (Eds.). E*xxon Valdez* oil spill: Fate and effects in Alaskan waters, pp. 548-584. American Society for Testing and Materials, ASTM STP 1219.

Brannon, E.L., K.C.M. Collins, L.L. Moulton and K.R. Parker, 2001. Resolving allegations of oil damage to incubating pink salmon eggs in Prince William Sound. Canadian Journal of Fisheries and Aquatic Sciences, 58:1070-1076.

Bratton, G.R., C.B. Spainhour, W. Flory, M. Reed and K. Jayko, 1993. Presence and potential effects of contaminants. In: J.J. Burns, J.J. Montague, C.J. Cowles (Eds.). The Bowhead Whale, pp. 700-744. Society for Marine Mammalogy, Lawrence, Kansas.

Brendehaug, J., S. Johnsen, K.H. Byrne, A.L. Gjøse, T.H. Eide and E. Aamot, 1992. Toxicity testing and chemical characterization of produced water – A preliminary study, pp. 245-256. In: J.P. Ray, F.R. Engelhardt (Eds.). Produced Water Technological/ Environmental Issues and Solutions. Plenum Press.

Briggs, K.T., S.H. Yoshida and M.E. Gershwin, 1996. The influence of petrochemicals and stress on the immune system of seabirds. Regulatory Toxicology and Pharmacology, 23:145-155.

Briggs, K.T., M.E. Gershwin and D.W. Anderson, 1997. Consequences of petrochemical ingestion and stress on the immune system of seabirds. ICES Journal of Marine Science, 54:718-725.

Brocksen, R.W. and H.T. Bailey, 1973. Respiratory response of juvenile chinook salmon and striped bass exposed to benzene, a water-soluble component of crude oil. Proceedings of the Joint Conference on the Prevention and Control of Oil Spills, pp. 783-791. American Petroleum Institute.

Brodersen, C., 1987. Rapid narcosis and delayed mortality in larvae of king crabs and kelp shrimp exposed to the water-soluble fraction of crude oil. Marine Environmental Research, 22:233-239.

Brown, J., 2005. ANIMIDA Task 1. Core contractor program management, logistics, database and reporting. Phase II Final summary report. Report by Exponent to U.S. Department of the Interior, Minerals Management Service. Report on Contract No. 143501-99-CT-30998 TO 10901, 47p.

Brown, E.D., T.T. Baker, J.E. Hose, R.M. Kocan, G.D. Marty, M.D. McGurk, B.L. Norcross and J. Short, 1996. Injury to the early life history stages of Pacific herring in Prince William Sound after the *Exxon Valdez* oil spill. In: S.D. Rice, R.B. Spies, D.A. Wolfe, B.A. Wright (Eds.). Proceedings of the Exxon Valdez Oil Spill Symposium. Symposium 18:448-462. American Fisheries Society.

Brueggeman, J.J., C.I. Malme, R.A. Grotefendt, D.P. Volsen, J.J. Burns, D.G. Chapman, D.K. Ljungblad and G.A. Green, 1990. 1989 Walrus

Monitoring Program, Klondike, Burger, and Popcorn Prospects in the Chukchi Sea. Shell Western E&P Inc., 121p + app.

Bryazgin, 1973. In PINRO, 2002.

Buccafusco, R.J., S.J. Ells and G.A. LeBlanc, 1981. Acute toxicity of priority pollutants to bluegill (*Lepomis macrochirus*). Bulletin of Environmental Contamination and Toxicology, 26:446-452.

Buchanan, R.A., J.R. Christian, V.D. Moulton, B. Mactavish and S. Dufault, 2004. 2004 Laurential 2-D seismic survey environmental assessment. Report prepared by LGL Limited, St. John's NL, and Canning & Pitt Associates Inc., St John's NL, for Conoco Phillips Canada Resources Corporation, Calgary, Alberta, 274p.

Buchman, M.F., 1999. NOAA Screening Quick Reference Tables, NOAA HAZMAT Report 99-1. Coastal Protection and Restoration Division, National Oceanic and Atmospheric Administration. 12p.

Buck, B.M. and C.R. Greene, 1979. Source level measurements of an arctic sea ice pressure ridge. Journal of the Acoustical Society of America, 66(suppl. 1):s24-s26.

Budelmann, B.U., 1992. Hearing in crustacean. In: D.B. Webster, R.R. Fay, A.N. Popper (Eds.). The Evolutionary Biology of Hearing, pp 232-239.

Bue, B.G., S. Sharr, S.D. Moffitt and A.K. Craig, 1996. Effects of the *Exxon Valdez* oil spill on pink salmon embryos and pre-emergent fry. In: S.D. Rice, R.B. Spies, D.A. Wolfe and B.A. Wright (Eds). Proceedings of the Exxon Valdez Oil Spill Symposium. Symposium 18:619-627. American Fisheries Society.

Bugbee, S.L. and C.M. Walter, 1973. The response of macroinvertebrates to gasoline pollution in a mountain stream. Proceedings of the Joint Conference on the Prevention and Control of Oil Spills, pp. 725-731. American Petroleum Institute.

Buhite, T.R., 1979. Cleanup of a cold weather terrestrial pipeline spill. 1979 Oil Spill Conference (Prevention, Behaviour, Control, Cleanup), pp. 367-370. American Petroleum Institute. Publication No. 4284.

Buist, I., R. Belore, A. Lewis, F. Bercha, M. Cerovsek, J. Wilson and B. Rinelli, 2003. Persistence of crude oil spills on open water. Final Report. Project 1435-01-02-RP-85091. U.S. Minerals Management Service, Anchorage, Alaska, 74p.

Bunch, J.N., 1987. Effects of petroleum releases on bacterial numbers and microheterotrophic activity in the water and sediment of an arctic marine ecosystem. Arctic, 40(suppl 1), 172-183.

Bunch, J.N. and R.C. Harland, 1976. Biodegradation of crude petroleum by the indigenous microbial flora of the Beaufort Sea. Beaufort Sea Technical Report 10, 52p.

Burdick, G.E., E.J. Harris, H.J. Dean, T.M. Walker, J. Skea and D. Colby, 1964. The accumulation of DDT in lake trout and the effect on reproduction. Transactions of the American Fisheries Society, 93:127-136.

Burger, A.E., 1993. Estimating the mortality of seabirds following oil spills: effects of spill volume. Marine Pollution Bulletin, 26:140-143.

Burgess, R.M., 2000. Mammals of an Arctic Oil Field. Arctic fox. In: J.C. Truett, S.R. Johnson (Eds.). The Natural History of an Arctic oil field, pp. 159-178. Academic Press.

Burnham, K. and M. Rahman, 1992. Effects of petrochemicals and ultraviolet radiation on epidermal IA expression in vitro. Journal of Toxicology and Environmental Health, 35:175-185.

Burton, G.A. Jr., D.L. Denton, K. Ho and D.S. Ireland, 2003. Sediment toxicity testing: issues and methods. In: D.J. Hoffman, B.A. Rattner, G.A. Burton Jr., J. Cairns Jr. (Eds.). Handbook of Ecotoxicology, pp. 111-150. Second Edition. CRC Press.

Busdosh, M., 1981. Long-term effects of the water soluble fraction of Prudhoe Bay crude oil on survival, movement and food search success on the arctic amphipod *Boeckosimus* (=*Onisimus*) *affinis.* Marine Environmental Research, 5:167-180.

Busdosh, M. and R.M. Atlas, 1977. Toxicity of oil slicks to arctic amphipods. Arctic, 30:85-92.

Cairns, D.K. and R.D. Elliot, 1987. Oil spill impact assessment for seabirds: The role of refugia and growth centres. Biological Conservation, 40:1-9.

Cairns, J. Jr., A.G. Heath and B.C. Parker, 1975. Temperature influence on chemical toxicity to aquatic organisms. Journal Water Pollution Control Federation, 47:267-280.

Caldwell, R.S., E.M. Caldarone and M.H. Mallon, 1977. Effects of a seawater-soluble fraction of Cook Inlet crude oil and its major aromatic components on larval stages of the Dungeness crab, *Cancer magister* Dana. In: D.A. Wolfe (Ed.). Fate and Effects of Petroleum Hydrocarbons in Marine Ecosystems and Organisms, pp. 210-220. Pergamon Press.

Cameron, R.D., K.R. Whitten, W.T. Smith and D.D. Roby, 1979. Caribou distribution and group composition associated with construction of the trans-Alaska pipeline. Canadian Field-Naturalist, 93:155-162.

Cameron, R.D., W.T. Smith, S.G. Fancy, K.L. Gerhart and R.G. White, 1993. Calving success of female caribou in relation to body weight. Canadian Journal of Zoology, 71:480-486.

Cameron, R.D., W.T. Smith, R.G. White and B. Griffith, 2002. The Central Arctic caribou herd. In: D.C. Douglas, P.E. Reynolds, E.B. Rhode (Eds.). Arctic Refuge Coastal Plain Terrestrial Wildlife Research Summaries, pp. 38-45. U.S. Geological Survey, Biological Resources Division. USGS/BRD/BSR-2002-0001.

Cameron, R.D., W.T. Smith, R.G. White and B. Griffith, 2005. Central Arctic caribou and petroleum development: distributional, nutritional, and reproductive implications. Arctic 58:1-9.

Campbell, D., D. Cox, J. Crum, K. Foster, P. Christie and D. Brewster, 1993. Initial effects of the grounding of the tanker *Braer* on health in Shetland. British Medical Journal, 307(6914):1251-1255.

Campbell, D., D. Cox, J. Crum, K. Foster and A. Riley, 1994. Later effects of grounding of tanker *Braer* on health in Shetland. British Medical Journal, 309(6957):773-774.

Camphuysen, C.J. and M. Heubeck, 2001. Marine oil pollution and beached bird surveys: the development of a sensitive monitoring instrument. Environmental Pollution, 112:443-461.

Camus, L., M.B. Jones, J.F. Borseth, F. Regoli and M.H. Depledge, 2002. Heart rate, respiration and total oxyradical scavenging capacity of the arctic spider crab, *Hyas araneus*, following exposure to polycyclic aromatic compounds via sediment and injection. Aquatic Toxicology, 61:1-13.

Caprino, L. and G.I. Togna, 1998. Potential health effects of gasoline and its constituents: A review of current literature (1990-1997) on toxicological data. Environmental Health Perspectives, 106:115-125.

Capuzzo, J.M. and B.A. Lancaster, 1981. Physiological effects of South Louisiana crude oil on larvae of the American lobster (*Homarus americanus*). In: J. Vernberg, A. Calabrese, F.P. Thurberg and W.B. Vernberg (Eds.). Biological Monitoring of Marine Pollutants, pp. 405-423. Academic Press.

Capuzzo, J.M. and B.A. Lancaster, 1982. Physiological effects of petroleum hydrocarbons on larval lobsters (*Homarus americanus*); hydrocarbon accumulation and interference with lipid metabolism. In: W.B. Vernberg, A. Calabrese, F.P. Thurberg and F.J. Vernberg (Eds.). Physiological Mechanisms of Marine Pollutant Toxicity, pp. 477-501. Academic Press.

Carls, M.G., A.C. Wertheimer, J.W. Short, R.M. Smolowitz and J.J. Stegeman, 1996. Contamination of juvenile pink and chum salmon by hydrocarbons in Prince William Sound after the Exxon Valdez oil spill. In: S.D. Rice, R.B. Spies, D.A. Wolfe and B.A. Wright (Eds.). Proceedings of the Exxon Valdez Oil Spill Symposium. Symposium 18:593-607. American Fisheries Society.

Carls, M.G., S.D. Rice and J.E. Hose, 1999. Sensitivity of fish embryos to weathered crude oil: Part I. Low-level exposure during incubation causes malformations, genetic damage, and mortality in larval Pacific herring (*Clupea pallasi*). Environmental Toxicology and Chemistry, 18:481-493.

Carlson, R.I., J.A. Wilson, R.L. Lochmiller, D.M. Janz, J.L. Schroder and N.T. Basta, 2003. Ecotoxicological risks associated with land treatment of petrochemical wastes. II. Effects on hepatic phase I and phase II detoxification enzymes in cotton rats. Journal of Toxicology and Environmental Health A, 66:327-343.

Casal Lareo, A. and S. Medina, 2002. Vertidos de petroleo y salud publica (Oil spills and public health). (Editorial). Gaceta Sanitaria, 17:93-95.

Castege, I., G. Hemery, N. Roux, J. d'Elbee, Y. Lalanne, F. D'Amico and C. Mouches, 2004. Changes in abundance and at-sea distribution of seabirds in the Bay of Biscay prior to, and following the *Erika* oil spill. Living Aquatic Resources, 17:361-367.

CCME, 2000. Canada-wide Standards for Petroleum Hydrocarbons (PHCs) in Soil: Scientific Rationale – Supporting Technical Document. Canadian Council of Ministers of the Environment. 340p.

CCME, 2001a. Canada-wide Standards for Petroleum Hydrocarbons in Soil: Technical Supplement. Canadian Council of Ministers of the Environment. 23p.

CCME, 2001b. Canada-wide Standards for Petroleum Hydrocarbons in Soil. Canadian Council of Ministers of the Environment. 8p.

CCME, 2004a. Canadian Environmental Quality Guidelines. Canadian Council of Ministers of the Environment.

CCME, 2004b. Canadian Soil Quality Guidelines for the Protection of Environmental and Human Health. Summary Table, Updated 2004. Canadian Council of Ministers of the Environment. 6p.

CCME, 2005. Canadian Water Quality Guidelines for the Protection of Aquatic Life: Summary Table, Updated October, 2005. Canadian Council of Ministers of the Environment.

CEPA, 1993. Benzene-PSL1. Environment Canada/Health Canada. Canadian Environmental Protection Act. http://www.hc-sc.gc.ca/ewh-semt/pubs/contaminants/psl1-lsp1/index_e.html

CEPA, 1994. Polycyclic Aromatic Hydrocarbons-PSL1. Environment Canada/Health Canada. Canadian Environmental Protection Act.

http://www.hc-sc.gc.ca/ewh-semt/pubs/contaminants.

Chaerun, S.K., K. Tazaki, R. Asada and K. Kogure, 2004. Bioremediation of coastal areas 5 years after the *Nakhodka* oil spill in the Sea of Japan: isolation and characterization of hydrocarbon-degrading bacteria. Environment International, 20:911-922.

Chaîneau, C.H., J.L. Morel and J. Oudot, 1997. Phytotoxicity and plant uptake of fuel oil hydrocarbons. Journal of Environmental Quality, 26:1478-1483.

Chapman, P.M. and R.C. Loehr, 2003. Relevant environmental science. Environmental Toxicology and Chemistry, 22:2217-2218.

Chen, Z., G.H. Huang and A. Chakma, 2003. Hybrid fuzzy-stochastic modeling approach for assessing environmental risks at contaminated groundwater systems. Journal of Environmental Engineering, 129:79-88.

Choi, J. and J.T. Oris, 2000. Evidence of oxidative stress in bluegill sunfish (*Lepomis macrochirus*) liver microsomes simultaneously exposed to solar ultraviolet radiation and anthracene. Environmental Toxicology and Chemistry, 19:1795-1799.

Christian, J.R., A. Mathieu, D.H. Thomson, D. White and R.A. Buchanan, 2003. Effect of seismic energy on snow crab (*Chionoecetes opilio*). LGL report to Environmental Studies Research Fund (ESRF), Calgary, Alberta. 56p + app.

Chromov and Prokhorova, 1979. In PINRO, 2002.

CIRE-Ouest, 2000. Etude épidémiologique des troubles de santé survenus à court terme chez les personnes ayant participé au nettoyage des sites pollués par le fioul de l'Erika. Cellule Inter régionale d'Epidémiologie Ouest, Institut de Veille Sanitaire, Directions Départementales des Affaires Sanitaires et Sociales de la Loire Atlantique, de la Vendée, du Morbihan et du Finistère. 56p.

Claireaux, G., Y. Desaunay, F. Akcha, B. Auperin, G. Bocquene, H. Budzinski, J.-P. Cravedi, F. Davoodi, R. Galois, C. Goanvec, D. Guerault, N. Imbert, O. Mazeas, L. Nonnotte, P. Prunet, P. Sebert and A. Vettier, 2004. Influence of oil exposure on the physiology and ecology of the common sole *Solea solea*: experimental and field approaches. Aquatic Living Resources, 17:335-351.

Clark, R.B., 1984. Impact of oil pollution on seabirds. Environmental Pollution A, 33:1-22.

Clark, R.B., 1987. Summary and conclusions: Environmental effects of North Sea oil and gas developments. Philosophical Transactions of the Royal Society of London. Series B, 316:669-677.

Cleveland, L., E.A. Little, R.D. Calfee and M.G. Barron, 2000. Photoenhanced toxicity of weathered oil to *Mysidopsis bahia.* Aquatic Toxicology, 49:63-76.

Collier, T.K., C.A. Krone, M.M. Krahn, J.E. Stein, S-.L. Chan and U. Varanasi, 1996. Petroleum exposure and associated biochemical effects in subtidal fish after the *Exxon Valdez* oil spill. In: S.D. Rice, R.B. Spies, D.A. Wolfe, B.A. Wright (Eds.). Proceedings of the Exxon Valdez Oil Spill Symposium. Symposium 18:671-683. American Fisheries Society.

Collins, C.M., C.H. Racine and M.E. Walsh, 1994. The physical, chemical, and biological effects of crude oil spills after 15 years on a black spruce forest, interior Alaska. Arctic, 47:164-175.

Conn, J.S., C. Behr-Andres, J. Wiegers, E. Meggert and N. Glover, 2001. Remediation of arctic tundra following petroleum or salt water spills. Polar Record, 37:264-266.

Connell, D.W., 1971. Kerosene-like tainting in Australian mullet. Marine Pollution Bulletin, 2:188-190.

Consonni, D., A.C. Pesatori, A. Tironi, I. Bernucci, C. Zocchetti and P.A. Bertazzi, 1999. Mortality study in an Italian oil refinery: Extension of the follow-up. American Journal of Industrial Medicine, 35:287-294.

Cook, S.V., A. Chu and R.H. Goodman, 2002. Leachability and toxicity of hydrocarbons, metals and salt contamination from flare pit soil. Water Air and Soil Pollution, 133:297-314.

Coombs, S. and J.C. Montgomery, 1999. The enigmatic lateral line system. In: R.R. Fay, A.N. Popper (Eds.). Comparative Hearing: Fish and Amphibians, pp 319-362. Springer Handbook of Auditory Research, Springer Verlag.

Coppock, R.W., 1997. Sources of contamination identified by oilfield operation industrial process and operating practices. In: G.E. Chalmers (Ed.). A Literature Review and Discussion of the Toxicological Hazards of Oilfield Pollutants in Cattle, pp. 9-16. Alberta Research Council, Alberta, Canada. ARCV97-R2.

Coppock, R.W., M.S. Mostrom, A.A. Khan and S.S. Semalulu, 1995. Toxicology of oil-field pollutants in cattle – a review. Veterinary and Human Toxicology, 37:569-576.

Coppock, R.W., M.S. Mostrom, E.L Stair and S.S. Semalulu, 1996. Toxicopathology of oilfield poisoning in cattle: a review. Veterinary and Human Toxicology, 38:36-42.

Corwin, J.T., 1981. Audition in elasmobranches. In: W.N. Travolga, A.N. Popper, R.R. Fay (Eds.). Hearing and Sound Communication in Fishes, pp 81-105. Springer-Verlag.

Cosens, S.E. and L.P. Dueck, 1987. Responses of migrating narwhal and beluga to icebreaker traffic at the Admiralty Inlet ice-edge, N.W.T. in 1986. Proceedings of the Ninth International Conference on Port and Ocean Engineering Under Arctic Conditions, pp. 39-54. Fairbanks, Alaska.

Cosens, S.E. and L.P. Dueck, 1993. Icebreaker noise in Lancaster Sound, N.W.T., Canada: Implications for marine mammal behavior. Marine Mammal Science, 9:285-300.

COSEWIC, 2003. Assessment and update status report on the wolverine *Gulo gulo*, in Canada. Committee on the Status of Endangered Wildlife in Canada. vi +41p.

Costa, D.P. and G.L. Kooyman, 1982. Oxygen consumption, thermoregulation, and the effect of fur oiling and washing on the sea otter, *Enhydra lutris.* Canadian Journal of Zoology, 60:2761-2767.

Cott, P.A., B.W. Hanna and J. Dahl, 2003. Discussion on seismic exploration in the Northwest Territories 2000-2003. Canadian Manuscript Report of Fisheries and Aquatic Sciences, 2648, vi+36p.

Couillard, C.M. and F.A. Leighton, 1990. Sequential study of the pathology of Prudhoe Bay crude oil in chicken embryos. Ecotoxicology and Environmental Safety, 19:17-23.

Couillard, C.M. and F.A. Leighton, 1991a. Bioassays for the toxicity of petroleum oils in chicken embryos. Environmental Toxicology and Chemistry, 10:533-538.

Couillard, C.M. and F.A. Leighton, 1991b. Critical period of sensitivity to petroleum toxicity in the chicken embryo. Environmental Toxicology and Chemistry, 10:249-253.

Coulon, F., E. Pelletier, R. St. Louis, L. Gourhant and D. Delille, 2004. Degradation of petroleum hydrocarbons in two sub-antarctic soils: influence of an oleophilic fertilizer. Environmental Toxicology and Chemistry, 23:1893-1901.

Craddock, D.R., 1977. Acute toxic effects of petroleum on arctic and subarctic marine organisms. In: D.C. Malins (Ed.). Effects of Petroleum on Arctic and Subarctic Marine Environments and Organisms. Vo1. II. Biological Effects, pp. 1-94. Academic Press.

Cranford, P.J., D.C. Gordon, K. Lee, S.L. Armsworthy and G.-H., Tremblay, 1999. Chronic toxicity and physical disturbance effects of water- and oil-based drilling fluids and some major constituents on adult sea scallops (*Placopecten magellanicus*). Marine Environmental Research, 48:225-256.

Cranford, P.J., S.L. Armsworthy, S. McGee, T. King and K. Lee, 2005. Scallops as sentinel organisms for offshore environmental effects monitoring. In: S.L. Armsworthy, P.J. Cranford, K. Lee (Eds.). Offshore Oil and Gas Environmental Effects Monitoring. Approaches and Technologies, pp. 267-296. Battelle Press.

Crawford, R.B. and J.D. Gates, 1981. Effects of a drilling fluid on the development of a teleost and an echinoderm. Bulletin of Environmental Contamination and Toxicology, 26:207-212.

Crête, M. and J. Huot, 1993. Regulation of a large herd of migratory caribou: summer nutrition affects calf growth and body reserves of dams. Canadian Journal of Zoology, 71:2291-2296.

Crocker, F.H., W.F. Guerin and S.A. Boyd, 1995. Bioavailability of naphthalene sorbed to cationic surfactant-modified smectite clay. Environmental Toxicology and Chemistry, 29:2953-2958.

Cronin, M.A. and J.W. Bickham, 1998. A population genetic analysis of the potential for a crude oil spill to induce heritable mutations and impact natural populations. Ecotoxicology, 7:259-278.

Cronin, M.A. and A.W. Maki, 2004. Assessment of the genetic toxicological impacts of the *Exxon Valdez* oil spill on pink salmon (*Oncorhynchus gorbuscha*) may be confounded by the influence of hatchery fish. Ecotoxicology, 13:495-501.

Cronin, M.A., S.C. Amstrup, G.M. Durner, L.E. Noel, T.L. McDonald and W.B. Ballard, 1998. Caribou distribution during the postcalving period in relation to infrastructure in the Prudhoe Bay Oil Field, Alaska. Arctic, 51:85-93.

Cross, W.E., 1986. Cameron Island benthos – Preliminary baseline community description. Report to Indian Affairs and Northern Development, Ottawa, LGL Ltd, King City, Ontario. 47p.

Cross, W.E., 1987. Effects of oil and chemically treated oil on primary productivity of high arctic ice algae studied *in situ.* Arctic, 40(suppl. 1):266-276.

Cross, W.E. and C.M. Martin, 1987. Effects of oil and chemically treated oil on nearshore under-ice meiofauna studied *in situ*. Arctic, 40(suppl. 1):258-265.

Cross, W.E. and D.H. Thomson, 1987. Effects of experimental releases of oil and dispersed oil on arctic nearshore macrobenthos. I. Infauna. Arctic, 40(suppl. 1):184-200.

Cross, W.E., C.M. Martin and D.H. Thomson, 1987a. Effects of experimental releases of oil and dispersed oil on arctic nearshore macrobenthos. II. Eipibenthos. Arctic, 40(suppl. 1):201-210.

Cross, W.E., R.T. Wilce and M.F. Fabijan, 1987b. Effects of experimental releases of oil and dispersed oil on arctic nearshore macrobenthos. III. Macroalgae. Arctic, 40(suppl. 1):211-219.

Croxall, J.P. and P. Rothery, 1991. Population regulation of seabirds: Implications of their demography for conservation. In: C.M. Perrin, J.D. Lebreton, G.J.M. Hirons (Eds.). Bird Population Studies, pp. 272-296. Oxford University Press.

Crum, J.E., 1993. Peak expiratory flow rate in schoolchildren living close to the *Braer* oil spill. British Medical Journal, 307:23-24.

Curatolo, J.A. and S.M. Murphy, 1986. The effects of pipelines, roads, and traffic on the movement of caribou, *Rangifer tarandus*. Canadian Field-Naturalist, 100:218-224.

Custer, T.W., C.M. Custer, R.K. Hines, D.W. Sparks, M.J. Melancon, D.J. Hoffman, J.W. Bickham and J.K. Wickliffe, 2000. Mixed function oxygenases, oxidative stress, and chromosomal damage measured in lesser scaup wintering on the Indiana Harbor Canal. Archives of Environmental Contamination and Toxicology, 38:522-529.

Dagg, T.G., K.P. Satin, W.J. Bailey, O. Wong, L.L. Harmon and RT.E. Swencicki, 1992. An updated cause specific mortality study of petroleum refinery workers. British Journal of Industrial Medicine, 49:203-212.

Dahlheim, M.E. and C.O. Matkin, 1994. Assessment of injuries to Prince William Sound killer whales. In: T.R. Loughlin (Ed.). Marine Mammals and the Exxon Valdez, pp. 163-171. Academic Press.

Dalen, J. and A. Raknes, 1985. Scaring effects on fish from three dimensional seismic surveys. Institute for Marine Research Report FO 8504/8505, Bergen, Norway. (In Norwegian, English summary)

Dalen, J. and G.M. Knutsen, 1986. Scaring effects in fish and harmful effects on eggs, larvae and fry by offshore seismic explorations. In: H.M. Merklinger (Ed.). Progress in Underwater Acoustics, pp 93-102. Plenum Press.

Dalen, J., E. Ona, A.V. Soldal and R. Saetre, 1996. Seismiske undersokelser til havs: en vundering av konsekvenser for fisk og fiskerier [Seismic investigations at sea; an evaluation of consequences for fish and fisheries]. Fisken og Havet 1996: 1-26. (In Norwegian, English summary)

Dales, R.E., W.O. Spitzer, M.T. Schechter and S. Suissa, 1989a. The influence of psychological status on respiratory symptom reporting. American Review of Respiratory Disease, 139:1459-1463.

Dales, R.E., W.O. Spitzer, M.T. Schechter, S. Suissa, M.T. Schechter, P. Tousignant, and N. Steinmetz, 1989b. Respiratory health of a population living downwind from natural gas refineries. American Review of Respiratory Disease, 139:595-600.

Danischewski, D., 2007. AChE and GST activities as biomarkers in cod, *Gadus morhua*, caged along two North Sea transects. In: K. Hylland, A.D. Vethaak, T. Lang (Eds.). Biological Effects of Contaminants in Marine Pelagic Ecosystems, pp. 301-310. SETAC special publication.

Darville, R.G., H.J. Harmon, M.R. Sanborn and J.L. Wilhm, 1983. Effect of naphthalene on the hemolymph ion concentrations of *Chironomus attenuatus* and the possible mode of action. Environmental Toxicology and Chemistry, 2:423-429.

Dauphine, T.C., 1976. Biology of the Kaminuriak population of barrenground caribou. Part 4. Growth, reproduction and energy reserves. Canadian Wildlife Service Report Series 38. 71p.

Davies, J.M. and G. Topping, 1997. The Impact of an Oil Spill in Turbulent Waters: The *Braer*. The Stationery Office Limited, Edinburgh. 263p.

Davies, J.M., J.M. Addy, R.A. Blackman, J.R. Blanchard, J.E. Ferbrache, D.C. Moore, H.J. Somerville, A. Whitehead and T. Wilkinson, 1984a. Environmental effects of the use of oil-based drilling muds in the North Sea. Marine Pollution Bulletin, 15:363-370.

Davies, J.M., J.S. Bell and C. Houghton, 1984b. A comparison of the levels of hepatic aryl hydrocarbon hydroxylase in fish caught close to and distant from North Sea oil fields. Marine Environmental Research, 14:23-45.

Davis, H.K., 1995. Depuration of oil taint and muscle pigment from fish. Water Science and Technology, 31:23-28.

Davis, J.E. and S.S. Anderson, 1976. Effects of oil pollution on breeding grey seals. Marine Pollution Bulletin, 7:115-118.

Davis, R.W., T.M. Williams, J.A. Thomas, R.A. Kastelein and L.H. Cornell, 1988. The effects of oil contamination and cleaning on sea otters (*Enhydra lutris*). II. Metabolism, thermoregulation, and behavior. Canadian Journal of Zoology, 66:2782-2790.

Davis, H.K., C.F. Moffat and N.J. Shepherd, 1997. A study of tainting of fish by chemically dispersed petroleum products. Report FC 96/62. Scottish Office Agriculture, Environment and Fisheries Department, Marine Laboratory, Aberdeen, Scotland. 81p + figs.

Day, R.H., S.M. Murphy, J.A. Wiens, G.D. Hayward, E.J. Harner and B.E. Lawhead, 1997a. Effects of the *Exxon Valdez* oil spill on habitat use by birds along the Kenai Peninsula, Alaska. The Condor, 99:728-742.

Day, R.H., S.M. Murphy, J.A. Wiens, G.D. Hayward, E.J. Harner and L.N. Smith, 1997b. Effects of the *Exxon Valdez* oil spill on habitat use by birds in Prince William Sound, Alaska. Ecological Applications, 7:593-613.

Dean, T.A., J.L. Bodkin, A.K. Fukuyama, S.C. Jewett, D.H. Monson, C.E. O'Clair and G.R. VanBlaricom, 2002. Food limitation and the recovery of sea otters following the *Exxon Valdez* oil spill. Marine Ecology Progress Series, 241:255-270.

DeGraeve, G.M., R.G. Elder, D.C. Woods and H.L. Bergman, 1982. Effects of naphthalene and benzene on fathead minnows and rainbow trout. Archives of Environmental Contamination and Toxicology, 11:487-490.

Delcourt, H.R. and P.A. Delcourt, 1988.Quaternary landscape ecology: relevant scales in space and time. Landscape Ecology, 2:23-44.

Depledge, M.H., 1984. Changes in cardiac activity, oxygen uptake and perfusion indices in *Carcinus maenas* (L.) exposed to crude oil and dispersant. Comparative Biochemistry and Physiology. C, 78:461-466.

Derocher, A.E. and I. Stirling, 1991. Oil contamination in polar bears. Polar Record, 27:56-57.

Devenis, P.K., 1995. Komi Oil Spill – An assessment by a multi-national team. Arctic Marine Oilspill Program (AMOP). Environment Canada. Ottawa, Canada. 18b:1147-1161.

Devlin, E.W., J.D. Brammer and R.L. Puyer, 1982. Acute toxicity of toluene to three age groups of fathead minnows (*Pimephales promelas*). Bulletin of Environmental Contamination and Toxicology, 29:12-17.

Devon, 2004. Comprehensive Study Report: Devon Beaufort Sea Exploration Drilling Program. Report submitted to National Energy Board by Devon Canada Corporation, Calgary, Alberta, August 2004.

Dewailly, E. and O. Weihe, 2003. The effects of Arctic pollution on population health. In: AMAP Assessment 2002: Human Health in the Arctic, pp. 95-105. Artic Monitoring and Assessment Programme, Oslo, Norway.

Dillon, T.M., J.M. Neff and J.S. Warner, 1978. Toxicity and sublethal effects of no.2 fuel oil on the supralittoral isopod *Lygia exotica*. Bulletin of Environmental Contamination and Toxicology, 20:320-327.

DiMichele, L. and M.H. Taylor, 1978. Histopathological and physiological responses of *Fundulus heteroclitus* to naphthalene exposure. Journal of the Fisheries Research Board of Canada, 35:1060-1066.

Divine, B.J. and V. Barron, 1987. Texaco Mortality Study: III. A cohort study of producing and pipeline workers. American Journal of Industrial Medicine, 11:189-202.

Divine, B.J. and C.M. Hartman, 2000. Update of a study of crude oil production workers: 1946-94. Occupational and Environmental Medicine, 57: 411-417.

DnV, 2001. Environmental monitoring Region VI Haltenbanken 2000. Det Norske Veritas Report no 2001-0379, 248p. (In Norwegian with English summary)

Dokholyan et al., 1979. In PINRO, 2002.

Dokholya et al., 1980. In PINRO, 2002.

Dowlatabadi, H, M. Boyle, S. Rowley and M. Kandlikar, 2004. Bridging the gap between project-level assessments and regional development dynamics: A methodology for estimating cumulative effects. Paper submitted to the Canadian Environmental Assessment Agency.

Doyle, L., 2006. Oil gushes into the Arctic Ocean from BP pipeline. The Independent, London, England. March 21, 2006.

Duesterloh, S., J.W. Short and M.G. Barron, 2002. Photoenhanced toxicity of weathered Alaska North Slope crude oil to the calanoid copepods *Calanus marshallae* and *Metridia okhotensis*. Environmental Science and Technology, 36:3953-3959.

Duffy, L.K., R.T. Bowyer, J.W. Testa and J.B. Faro, 1993. Differences in blood haptoglobin and length-mass relationships in river otters (*Lutra canadensis*) from oiled and non-oiled areas of Prince William Sound, Alaska. Journal of Wildlife Diseases, 29:353-359.

Duffy, L.K., R.T. Bowyer, J.W. Testa and J.B. Faro, 1994. Chronic effects of the *Exxon Valdez* oil spill on blood and enzyme chemistry of river otters. Environmental Toxicology and Chemistry, 13:643-647.

Duncan, P.B. and A.J. Hooten, 1996. Influence of residual and applied oil on intertidal algal recruitment. In: S.D. Rice, R.B. Spies, D.A. Wolfe, B.A. Wright (Eds.). Proceedings of the Exxon Valdez Oil Spill Symposium. Symposium 18:238-248. American Fisheries Society.

Dunnet, G.M., 1982. Oil pollution and seabird populations. Philosophical Transactions of the Royal Society of London. Series B, 297:413-427.

Dunnet, G.M., 1987. Seabirds and North Sea Oil. Philosophical Transactions of the Royal Society of London. Series B, 316:513-524.

Dunning, D.J., Q.E. Ross, P. Geoghegan, J.J. Reichle, J.K. Menezes and J.K. Watson, 1992. Alewives avoid high-frequency sound. North American Journal of Fisheries Management, 12:407-416.

Dunstan, W.M., L.P. Atkinson and J. Natoli, 1975. Stimulation and in-

hibition of phytoplankton growth by low molecular weight hydrocarbons. Marine Biology, 31:305-310.

Dyer, S.J., J.P. O'Neill, S.M. Wasel and S. Boutin, 2002. Quantifying barrier effects of roads and seismic lines on movements of female woodland caribou in northeastern Alberta. Canadian Journal of Zoology, 80:839-845.

Eberhardt, L.E., R.A. Garrott and W.C. Hanson, 1983. Winter movements of arctic foxes, *Alopus lagopus*, in a petroleum development area. Canadian Field-Naturalist, 97:66-70.

Edwards, N.T., 1983. Polycyclic aromatic hydrocarbons (PAHs) in the terrestrial environment – A review. Journal of Environmental Quality, 12:427-441.

Edwards, W.C. and D.G. Gregory, 1991. Livestock poisoning from oil field drilling fluids, muds and additives. Veterinary and Human Toxicology, 33:502-504.

Edwards, R. and I. White, 1999. The *Sea Empress* oil spill: environmental impact and recovery. Proc. 1999 International Oil Spill Conference, pp. 97-102. 7-12 March 1999, Seattle, USA.

Edwards, W.C., R.W. Coppock and L.L. Zinn, 1979. Toxicoses related to the petroleum industry. Veterinary and Human Toxicology, 21:328-337.

Eggen, R.L., R. Behra, P. Burkhardt-Holm, B.I. Escher and N. Schweigert, 2004. Challenges in ecotoxicology. Environmental Science and Technology, 38:57A-64A.

Eide, I., 1990. A review of exposure conditions and possible health effects associated with aerosol and vapour from low-aromatic oil-based drilling fluids. Annals of Occupational Hygiene, 34:149-157.

Eide, I. and K. Zahlsen, 1996. Inhalation experiments with mixtures of hydrocarbons: Experimental design, statistics, and interpretation of kinetics and possible interactions. Archives of Toxicology, 70:397-404.

Eisler, R., 1987. Polycyclic aromatic hydrocarbon hazards to fish, wildlife, and invertebrates: A synoptic review. Biological Report 85(1.11). U.S. Fish and Wildlife Service. 81p.

Ellingsen, K.E. and J.S., Gray, 2002. Spatial patterns of benthic diversity: Is there a latitudinal gradient along the Norwegian continental shelf? Journal of Animal Ecology, 71:373-389.

Elmgren, R., S. Hansson, U. Larsson and B. Sundelin, 1980. Impact of oil on deep soft bottoms. In: J.J. Kineman, R. Elmgren, S. Hansson (Eds.). The *Tsesis* Oil Spill, pp. 97-126. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, Boulder, Colorado.

Elmgren, R., S. Hansson, U. Larsson, B. Sundelin and P.D. Boehm, 1983. The *Tsesis* oil spill: acute and long-term impact on the benthos. Marine Biology, 73:51-65.

Eltohami, E.S., M.A. Adewumi and M. Erdogmus, 2004. PCB migration and cleanup scenarios in natural gas pipelines. Journal of Energy Resources Technology, 126:105-111.

Engås, A., S. Løkkenborg, E. Ona and A.V. Soldal, 1996. Effects of seismic shooting on local abundance and catch rates of cod (*Gadus morhua*) and haddock (*Melanogrammus aeglefinus*). Canadian Journal of Fisheries and Aquatic Sciences, 53:2238-2249.

Engelhardt, F.R., 1981. Oil pollution in polar bears: Exposure and clinical effects. In: Proceedings of the 4th Arctic Marine Oilspill Program, pp. 139-179. Technical seminar. Edmonton, Alta. Environmental Protection Service, Ottawa.

Engelhardt, F.R., 1983. Petroleum effects on marine mammals. Aquatic Toxicology, 4:199-217.

Engelhardt, F.R., 1985. Petroleum Effects in the Arctic Environment. Elsevier Applied Science. 281p.

Englehardt, F.R., 1994. Limitations and innovations in the control of environmental impacts from petroleum industry activities in the Arctic. Marine Pollution Bulletin, 29:334-341.

Engelhardt, F.R., J.R. Geraci and T.G. Smith, 1977. Uptake and clearance of petroleum hydrocarbons in the ringed seal, *Phoca hispida*. Journal of the Fisheries Research Board of Canada, 34:1143-1147.

Engelhardt, F.R., M.P. Wong and M.E. Duey, 1981. Hydromineral balance and gill morphology in rainbow trout, *Salmo gardneri*, acclimated to fresh and sea water, as affected by petroleum exposure. Aquatic Toxicology, 1:175-186.

Engelhardt, F.R., J.P. Ray and A.H. Gillam, 1989. Drilling Wastes. Elsevier Applied Science. 867p.

Environment Canada, 1994. Priority Substances List assessment report: Polycyclic aromatic hydrocarbons. Environment Canada. 61p.

Environment Canada, 2006. Topic 7: Migratory birds and the Kendall Island Bird Sanctuary. Submission to the Joint Review Panel for the Mackenzie Gas Project.

Epstein, P.R. and J. Selber (Eds.), 2002. Oil: A Life Cycle Analysis of its Health and Environmental Impacts. Center for Health and the Global Environment, Harvard Medical School, Boston. 73p.

EPT, 2003. Attachment 1-1. A critical review of methods for developing ecological soil quality guidelines and criteria. Guidance for developing ecological Soil Screening Levels (Eco-SSLs). American Petroleum Institute. Washington, D.C. OSWER Directive 92857-55. 91p.

ERCB, 1984. Lodgepole Blowout Inquiry, Phase 1. Report to the Lieutenant Governor in Council, with respect to an Inquiry held into the blowout of the well, Amoco Dome Brazeau River 13-12-48-12. Energy Resources Conservation Board, Alberta. 4 volumes.

Erickson, W.P., G.D. Johnson and D.P. Young, 2005. A summary and comparison of bird mortality from anthropogenic causes with an emphasis on collisions. Third International Partners in Flight Conference, pp. 1029-1042. USDA Forest Service. General Technical Report PSW-GTR-191.

Eriksson, M., J.-O. Ka and W.W. Mohn, 2001. Effects of low temperature and freeze-thaw cycles on hydrocarbon biodegradation in arctic tundra soil. Applied and Environmental Microbiology, 67:5107-5112.

Eriksson, M., E. Sodersten, Z. Yu, G. Dalhammar and W.W. Mohn, 2003. Degradation of polycyclic aromatic hydrocarbons at low temperature under aerobic and nitrate-reducing conditions in enrichment cultures from northern soils. Applied and Environmental Microbiology, 69:275-284.

Ernst, R.J., W.M.N. Ratnayake, T.E. Farquharson, R.G. Ackman, W.G. Tidmarsh and J.A. Carter, 1989. Tainting of Atlantic cod (*Gadus morhua*) by petroleum hydrocarbons. In: F.R. Engelhardt, J.P. Ray, A.H. Gillam (Eds.). Drilling Wastes, pp. 827-839.Elsevier Applied Science.

Erwin, R.M., D.R. Cahoon, D.J. Prosser, G.M. Sanders and P. Hensel, 2006. Surface elevation dynamics in vegetated *Spartina* marshes versus unvegetated tidal ponds along the mid-Atlantic coast, USA, with implications to waterbirds. Estuaries and Coasts, 29:96-106.

ESG International Inc., 2003. Toxicity of petroleum hydrocarbons to soil organisms and the effects on soil quality: Phase I. Fraction-specific toxicity of crude oil. Petroleum Technology Alliance Canada, Calgary, Alberta. Report 98372. 236p.

Estes, J.A., 1991. Catastrophes and conservation: Lessons from sea otters and the *Exxon Valdez*. Science, 254:1596.

Estes, J.A., M.T. Tinker, T.M. Williams and D.F. Doak, 1998. Killer whale predation on sea otters linking oceanic and nearshore ecosystems. Science, 282:473-476.

EU, 1998. Council Directive 98/83/EC of the European Parliament and of the Council of 3 November 1998 on the Quality of Water intended for Human Consumption. European Union. *Official Journal L 330, 05/12/1998 pp. 32-54.*

EU, 2000. Council Directive 2000/69/EC of the European Parliament and of the Council of 16 November 2000 relating to limit values for benzene and carbon monoxide in ambient air.

EU, 2004. Council Directive 2004/107/EC of the European Parliament and of the Council of 15 December 2004 relating to arsenic, cadmium, mercury, nickel and polycyclic aromatic hydrocarbons in ambient air. Official Journal L 023, 26/01/2005. pp. 3-16.

Evans, W.E., 1983. A study to determine if gray whales detect oil. In: J.R. Geraci, D.J. St. Aubin (Eds.). Study of the effects of oil on cetaceans, pp. 47-61. Report by University of Guelph for Bureau of Land Management. NTIS No. PB83-152-991.

Everett, K.R., 1978. Some effects of oil on the physical and chemical characteristics of wet tundra soils. Arctic, 31:260-276.

Fairbrother, A., J. Smits and K.A. Grasman, 2004. Avian immunotoxicology. Journal of Toxicology and Environmental Health, Part B, 7:105-137.

Faksness, L.G., P.G. Grini and P.S. Dalings, 2004. Partitioning of semi-soluble organic compounds between the water phase and oil droplets in produced water. Marine Pollution Bulletin, 48:731-742.

Falk, M.R. and M.J. Lawrence, 1973. Seismic exploration: its nature and effect on fish. Department of the Environment, Fisheries and Marine Service, Fisheries Operations Directorate, Central Region, Winnipeg, Technical Report Series No. CEN / T–73-9. 51p.

Fall, J.A. and L.J. Field, 1996. Subsistence uses of fish and wildlife before and after the Exxon Valdez oil spill. In: S.D. Rice, R.B. Spies, D.A. Wolfe, B.A. Wright (Eds.). Proceedings of the *Exxon Valdez* Oil Spill Symposium. Symposium 18:819-836. American Fisheries Society.

Fancy, S.G., 1983. Movements and activity budgets of caribou near oil drilling sites in the Sagavanirktok River Floodplain, Alaska. Arctic, 36:193-197.

Fancy, S.G., K.R. Whitten and D.E. Russell, 1994. Demography of the porcupine caribou herd 1983-1992. Canadian Journal of Zoology, 72:840-846.

Faulkner, B.C. and R.L. Lochmiller, 2000. Ecotoxicity revealed in parasite communities of *Sigmodon hispidus* in terrestrial environments contaminated with petrochemicals. Environmental Pollution,

110:135-145.

Fay, R.R. and A.N. Popper, 1999. Comparative Hearing: Fish and Amphibians. Springer-Verlag.

Fedorov, 1967. In PINRO, 2002.

Feist, S.W., J. Bignell, G.J. and G.D. Stentiford, 2007. Histological changes in caged blue mussel (*Mytilus edulis*) and cod (*Gadus morhua*) at contaminant gradients in the German Bight and the Statfjord offshore oil industry area in the North Sea. In: K. Hylland, A.D. Vethaak, T. Lang (Eds.). Biological Effects of Contaminants in Marine Pelagic Ecosystems, pp. 311-328. SETAC special publication.

Felix, N.A., M.K. Raynolds, J.C. Jorgenson and K.E. DuBois, 1989. Resistance and resilience of tundra plant communities to disturbance by winter seismic vehicles. Arctic and Alpine Research, 24:69-77.

Fernandes, M.B. and Sicre, M.-A. 1999. Polycyclic aromatic hydrocarbons in the Arctic: Ob and Yenisi estuaries and Kara Sea Shelf. Estuarine, Coastal and Shelf Science 48:725-737.

Fernandes, M.B. and M.-A. Sicre, 2000. The importance of terrestrial organic inputs on Kara Sea shelves as revealed by *n*-alkanes, OC and $\delta^{13}C$ values. Organic Geochemistry, 31:363-374.

Fingas, M.F., 1995a. Oil spills and their clean-up. Chemistry and Industry, 24:1005-1008.

Fingas, M.F., 1995b. The Newfoundland offshore burn experiment – NOBE. Proceedings of the 1995 International Oil Spill Conference, pp. 123-132. Washington, D.C. American Petroleum Institute.

Finley, K.J. and R.A. Davis, 1984a. Reactions of beluga whales and narwhals to ship traffic and ice-breaking along ice edges in the eastern Canadian high Arctic: 1982-1984. An Overview. LGL Limited, King City, Ontario, for Canada Department of Indian Affairs and Northern Development, Ottawa, 42p.

Finley, K.J. and R.A. Davis, 1984b. Reactions of beluga whales and narwhals to ship traffic and ice-breaking along ice edges in the eastern Canadian high Arctic: 1982-1984. LGL Limited, King City, Ontario, for Canada Department of Indian Affairs and Northern Development, Ottawa, 42+108+117+34p.

Finley, K.J. and R.A. Davis, 1986. Reactions of beluga whales and narwhals to ship traffic and ice-breaking along ice edges in the eastern Canadian high Arctic: 1982-1984. An Overview. Department of Indian Affairs and Northern Development, Environmental Studies 37, Sec 1, 42p.

Finneran, J.J., C.E. Schlundt, D.A. Carder, J.A. Clark, J.A. Young, J.B. Gaspin and S.H. Ridgway, 2000. Auditory and behavioral responses of bottlenose dolphins (*Tursiops truncatus*) and beluga whale (*Delphinapterus leucas*) to impulsive sounds resembling distant signatures of underwater explosions. Journal of the Acoustical Society of America, 108:417-431.

Finneran, J.J., C.E. Schlundt, R. Dear, D.A. Carder and S.H. Ridgway, 2002. Temporary shift in masked hearing thresholds in odontocetes after exposure to single underwater impulses from a seismic watergun. Journal of the Acoustical Society of America, 111:2929-2940.

Finnish Ministry of the Interior, 1980. The 1979 Baltic Oil Spill. Environmental Studies A:2. Department for Environmental Protection, Government Printing Centre, Helsinki. 53p.

Flickinger, E., 1981. Wildlife mortality at petroleum pits in Texas. Journal of Wildlife Management, 45:560-564.

Follman, E.H. and J.L. Hechtel, 1990. Bears and pipeline construction in Alaska. Arctic, 43:103-109.

Forbes, B.C., 1992. Tundra disturbance studies. I: Long-term effects of vehicles on species richness and biomass. Environmental Conservation, 19:48-58.

Forbes, B.C., 1993. Aspects of natural recovery of soils, hydrology and vegetation at an abandoned high arctic settlement, Baffin Island, Canada. In: Proceedings of the Sixth International Permafrost Conference, pp. 176-181. Lanzhou Institute of Glaciology and Geocryology, South China University of Technology Press.

Forbes, B.C., 1995. Tundra disturbance studies. III. Short-term effects of aeolian sand and dust, Yamal region, northwest Siberia. Environmental Conservation, 22:335-344.

Forbes, B.C., 1997. Tundra disturbance studies. IV. Species establishment on anthropogenic primary surfaces, Yamal Peninsula, northwest Siberia, Russia. Polar Geography, 21:79-100.

Forbes, B.C., 1999. Reindeer herding and petroleum development on Poluostrov Yamal: sustainability or mutually incompatible uses? Polar Record, 35:317-322.

Forbes, B.C., 2004. Impacts of energy development in polar regions. In: Encyclopedia of Energy, pp. 93-105. Academic Press.

Forbes, B.C. and R.L. Jefferies, 1999. Revegetation of disturbed arctic sites: constraints and applications. Biological Conservation, 88:15-24.

Forbes, B.C. and O.I. Sumina, 1999. Comparative ordination of low arctic vegetation recovering from disturbance: reconciling two contrasting approaches for field data collection. Arctic, Antarctic, and Alpine Research, 31:389-399.

Forbes, B.C., J.J. Ebersole and B. Strandberg, 2001. Anthropogenic disturbance and patch dynamics in circumpolar arctic ecosystems. Conservation Biology, 15:954-969.

Ford, R.G., J.A. Wiens, D. Heinemann and G.L. Hunt, 1982. Modelling the sensitivity of colonially breeding marine birds to oil spills: guillemot and kittiwake populations on the Pribiloff Islands, Bering Sea. Journal of Applied Ecology, 19:1-31.

Fouchécourt, M.O. and J.-L. Riviére, 1996. Activities of liver and lung cytochrome P450-dependent monooxygenases and antioxidant enzymes in laboratory and wild Norway rats exposed to reference and contaminated sites. Archives of Environmental Contamination and Toxicology, 30:513-522.

Fouchécourt, M.O., M. Arnold, P. Berny, B. Videmann, B. Rether and J.L. Riviere, 1999. Assessment of the bioavailability of PAHs in rats exposed to a polluted soil by natural routes: Induction of EROD activity and DNA adducts and PAH burden in both liver and lung. Environmental Research, 80:330-339.

Fowler, A.C. and P.L. Flint, 1997. Persistence rates and detection probabilities of oiled king eider carcasses on St. Paul Island, Alaska. Marine Pollution Bulletin, 34:522-526.

Foy, M.G., 1979. Acute lethal toxicity of Prudhoe Bay crude oil and Corexit 9527 to arctic marine invertebrates and fish from Frobisher Bay, N.W.T. Proceedings of the Arctic Marine Oil Spill Program, pp. 72-80. Technical Seminar.

Foy, M.G., 1982. Acute lethal toxicity of Prudhoe Bay crude oil and Corexit 9527 to arctic marine fish and invertebrates. Environment Canada, Technology Development Report EPS 4-EC-82-3, 62p.

Frantzis, A., 1998. Does acoustic testing strand whales. Nature, 392:29.

Fremling, C.R., 1981. Impact of a spill of No. 6 fuel oil on Lake Winona. In: 1981 Oil Spill Conference (Prevention, Behavior, Control, Cleanup), pp. 419-421. American Petroleum Institute, Pub. 4334.

Frisch, J., N.A. Øritsland and J. Krog, 1974. Insulation of furs in water. Comparative Biochemistry and Physiology A, 47:403-410.

Frost, K.J., C.-A. Manen and T.L. Wade, 1994. Petroleum hydrocarbons in tissues of harbor seals from Prince William Sound and the Gulf of Alaska. In: T.R. Loughlin (Ed.). Marine Mammals and the Exxon Valdez, pp. 331-370. Academic Press.

Frost, T.K., R. Hagemann, E. Furuhoft and S. Johnsen, 2002. Calculation of PNEC values applied in environmental risk management of produced water discharges. Statoil document 200212100003 (Appendix 1).

Fry, D.M. and L.J. Lowenstine, 1985. Pathology of common murres and Cassin's auklets exposed to oil. Archives of Environmental Contamination and Toxicology, 14:725-737.

Fukuyama, A.K., G. Shigenaka and R.Z. Hoff, 2000. Effects of residual *Exxon Valdez* oil on intertidal *Protothaca staminea*: mortality, growth, and bioaccumulation of hydrocarbons in transplanted clams. Marine Pollution Bulletin, 40:1042-1050.

Furness, R.W. and P. Monaghan, 1987. Seabird Ecology. Blackie. 164p.

Galbraith, H., R. Jones, R. Park, J. Clough, S. Herrod-Julius, B. Harrington and G. Page, 2002. Global climate change and sea level rise: potential losses of intertidal habitat for shorebirds. Waterbirds, 25:173-183.

Gallego, A., L.H. Cargill, M.R. Heath, S.J. Hay and T. Knutsen, 1995. An assessment of the immediate effect of the *Braer* oil-spill on the growth of herring larvae using otolith microstructure analysis. Marine Pollution Bulletin, 30:536-542.

Gamberg, M., B. Braune, E. Davey, B. Elkin, P.F. Hoekstra, D. Kennedy, C. Macdonald, D. Muir, A. Nirwal, M. Wayland and B. Zeeb, 2005. Spatial and temporal trends of contaminants in terrestrial biota from the Canadian Arctic. Science of the Total Environment, 351-352:148-164.

Gardiner, R., 2003. Overview and characteristics of some occupational exposures and health risks on offshore oil and gas installations. Annals of Occupational Hygiene, 47:201-210.

Garrott, R.A., L.L. Eberhardt and D.M. Burn, 1993. Mortality of sea otters in Prince William Sound following the *Exxon Valdez* oil spill. Marine Mammal Science, 9:343-359.

Garshelis, D.L., 1997. Sea otter mortality estimated from the carcasses collected after the *Exxon Valdez* oil spill. Conservation Biology, 11:905-916.

Garshelis, D.L. and J.A. Estes, 1997a. Response to critique by Garshelis and Estes of *Exxon Valdez* sea otter mortality estimate. Marine Mammal Science, 13:351-354.

Garshelis, D.L. and J.A. Estes, 1997b. Sea otter mortality from the *Exxon Valdez* oil spill: evaluation of an estimate from boat-based surveys. Marine Mammal Science, 13:341-351.

Gaston, A.J. and I.L. Jones, 1998. The Auks. Oxford University Press. 349p.

George, S.G., J.S. Christiansen, B. Killie and J. Wright, 1995a. Dietary crude oil exposure during sexual maturation induces hepatic mixed function oxygenase (CYP1A) activity at very low environmental temperatures in polar cod *Boreogadus saida*. Marine Ecology Progress Series, 122:307-312.

George, S.G., J. Wright and J. Conroy, 1995b. Temporal studies of the impact of the *Braer* oil spill on inshore feral fish from Shetland, Scotland. Archives of Environmental Contamination and Toxicology, 29:530-534.

Georghiou, G.P., 1986. The magnitude of the resistance problem, National Academy Press.

Geraci, J.R., 1990. Physiologic and toxic effects on cetaceans. In: J.R. Geraci, D.J. St. Aubin (Eds.). Sea Mammals and Oil Confronting the Risks, pp. 167-197. Academic Press.

Geraci, J.R. and T.G. Smith, 1976. Direct and indirect effects of oil on ringed seals (*Phoca hispida*) of the Beaufort Sea. Journal of the Fisheries Research Board of Canada, 33:1976-1984.

Geraci, J.R. and D. St. Aubin, 1990. Sea Mammals and Oil: Confronting the Risks, Academic Press. 282p.

Geraci, J.R., D.J. St. Aubin and R.J. Reisman, 1983. Bottlenose dolphin, *Tursiops truncates*, can detect oil. Canadian Journal of Fisheries and Aquatic Sciences, 40:1516-1522.

Geraci, J.R., D.J. St. Aubin and B.D. Hicks, 1986. The epidermis of odontocetes: a view from within. Part 1. Anatomy and physiology. In: M.M. Bryden, R. Harrison (Eds.). Research on Dolphins, pp. 3-21. Clarendon Press.

Giles, R.C., L.R. Brown and C.D. Minchew, 1978. Bacteriological aspects of fin erosion in mullet exposed to crude oil. Journal of Fish Biology, 13:113-117.

Gilman, A., 2003. Risk reduction strategies for Arctic Peoples. In: AMAP Assessment 2002: Human Health in the Arctic, pp. 106-113. Artic Monitoring and Assessment Programme, Oslo, Norway.

Gilroy, D.J., 2000. Derivation of shellfish harvest reopening criteria following the New Carissa oil spill in Coos Bay, Oregon. Journal of Toxicology and Environmental Health, A60:317-329.

Glegg and Koevening, 1974. In PINRO, 2002.

Goksøyr, A. and L. Forlin, 1992. The cytochrome P-450 system in fish, aquatic toxicology and environmental monitoring. Aquatic Toxicology, 22:287-312.

Golet, G.H., P.E. Seiser, A.D. McGuire, D.D. Roby, J.B. Fischer, K.J. Kuletz, D.B. Irons, T.A. Dean, S.C. Jewett and S.H. Newman, 2002. Long-term direct and indirect effects of the *Exxon Valdez* oil spill on pigeon guillemots in Prince William Sound, Alaska. Marine Ecology Progress Series, 241:287-304.

Gomer, C.J. and D.M. Smith, 1980. Acute skin phototoxicity in hairless mice following exposure to crude shale oil or natural petroleum oil. Toxicology, 18:75-85.

Goodale, D.R., M.A. Hyman and H.E. Winn, 1981. Cetacean responses in association with the Regal Sword oil spill. Cetacean and Turtle Assessment Program Annual Report. Report from the University of Rhode Island-Providence for the Minerals Management Service. (Available from the Alaska Outer Continental Shelf Office, 949 E 36th Ave. Rm 110, Anchorage, AK 99508).

Goodlad, D. and I. Napier, 1997. Assessment of the Shetland sandeel fishery – 1996. North Atlantic Fisheries College, Shetland Fisheries Training Centre Trust, Fisheries Development Note, 4p.

Gordon, D.C. Jr., L.D. Griffiths, G.V. Hurley, A.L. Muecke, D.K. Muschenheim and P.G. Wells, 2000. Understanding the environmental effects of offshore hydrocarbon development. Canadian Technical Report of Fisheries and Aquatic Sciences, 2311, 82p + app.

Gordon, J., D. Gillespie, J. Potter, A. Frantzis, M.P. Simmonds, R. Swift and D. Thompson, 2004. A review of the effects of seismic surveys on marine mammals. Marine Technology Society Journal, 37:16-34.

Gough, M.A. and S.J. Rowland, 1990. Characterization of unresolved mixtures of hydrocarbons in petroleum. Nature, 344:648-650.

Grahl-Nielsen, O., 1978. The *Ekofisk Bravo* blowout. Petroleum hydrocarbons in the sea. In: P.L. Russell (Ed.). Proc. Conference on Assessment of the Ecological Impacts of Oil Spills, pp. 477-487. 14-17 June 1978, Keystone, Colorado.

Grahl-Nielsen, O., 1987. Hydrocarbons and phenols in discharge water from offshore operations. Fate of the hydrocarbons in the recipient. Sarsia, 72:375-382.

Grahl-Nielsen, O., T. Neppelberg, K.H. Palmork, K. Westrheim and S. Wilhelmsen, 1976. The *Drupa* oil spill, investigation concerning oil, water and fish. ICES Fisheries Improvement Committee, C.M.1976/E:34, 18p.

Grant, A. and A.D. Briggs, 2002. Toxicity of sediments from around a North Sea oil platform: are metals or hydrocarbons responsible for ecological impacts? Marine Environmental Research, 53:95-116.

Grasman, K.A. 2002. Assessing immunological function in toxicological studies of avian wildlife. Integrative and Comparative Biology, 42:34-42.

Gratzfeld, J., 2003. Extractive industries in arid and semi-arid zones: Environmental planning and management. Ecosystem Management Series, No.1. The World Conservation Union. 112p.

Gray, J.S., K.R. Clarke, R.M. Warwick and G. Hobbs, 1990. Detection of initial effects of pollution on marine benthos: an example from the Ekofisk and Eldfisk oilfields, North Sea. Marine Ecology Progress Series, 66:285-299.

Green, R.H., 2005. Marine coastal monitoring: designing an effective offshore oil and gas environmental effects monitoring program. In: S.L. Armsworthy, P.J. Cranford, K. Lee (Eds.). Offshore Oil and Gas Environmental Effects Monitoring. Approaches and Technologies, pp. 373-397. Battelle Press.

Green, R.H. and P. Montagna, 1996. Implications for monitoring: study designs and interpretation of results. Canadian Journal of Fisheries and Aquatic Sciences, 53:2629-2636.

Greene, C.R., Jr. and W.J. Richardson, 1988. Characteristics of marine seismic survey sounds in the Beaufort Sea. Journal of the Acoustical Society of America, 83:2246-2254.

Griffith, B., D.C. Douglas, N.E. Walsh, D.D. Young, T.R. McCabe, D.E. Russell, R.G. White, R.D. Cameron and K.R. Whitten, 2002. The Porcupine caribou herd. In: D.C. Douglas, P.E. Reynolds, E.B. Rhode (Eds.). Arctic Refuge Coastal Plain Terrestrial Wildlife Research Summaries, pp. 8-37. U.S. Geological Survey, Biological Resources Division. USGS/BRD/BSR-2002-0001.

Grose, P.L. and J.S. Matson, 1977. Summary fact sheet: The Argo Merchant spill: a preliminary scientific report. National Oceanic and Atmospheric Administration, Environmental Research Laboratory, Boulder, CO. NTIS No. PCA02/MF A 01. 133p.

Gross, T.S., B.S. Arnold, M.S. Sepulveda and K. McDonald, 2003. Endocrine disrupting chemicals and endocrine active agents. In: D.J. Hoffman, B.A. Rattner, G.A. Burton Jr., J. Cairns Jr. (Eds.). Handbook of Ecotoxicology, pp. 1033-1098. Second Edition. Lewis Publishing.

Gruber, J.A., 1981. Ecology of the Atlantic bottlenose dolphin (*Tursiop truncates*) in the Pass Cavallo area of Matagorda Bay, Texas. M.S. Thesis, Texas A&M University. 182p.

Gschwend, P.M. and R.A. Hites, 1981. Fluxes of polycyclic aromatic hydrocarbons to marine and lacustrine sediments in the northeastern United States. Geochimica et Cosmochimica Acta, 45:2359-2367.

Gulliksen, B. and J.P. Taasen, 1982. Effect of an oil spill in Spitzbergen in 1978. Marine Pollution Bulletin, 13:96-98.

Haensly, W.E., J.M. Neff, J.R. Sharp, A.C. Morris, M.F. Bedgood and P.D. Boem, 1982. Histopathology of *Pleuronectes platessa* L. from Aber Wrac'h and Aber Benoit, Brittany, France: Long-term effects of the *Amoco Cadiz* crude oil spill. Journal of Fish Diseases, 5:365-391.

Hahn, M.E., 2002. Aryl hydrocarbon receptors: diversity and evolution. Chemico-Biological Interactions, 141:131-160.

Halldórsson, H.P., G. Ericson and J. Svavarsson, 2004. DNA strand breakage in mussels (*Mytilus edulis*) deployed in intertidal and subtidal zone in Reykjavik harbour. Marine Environmental Research, 58:763-767.

Halsall, C.J., L.A. Barrie, P. Fellin, D.C.G. Muir, B.N. Billeck and Lockhart, W.L., 1998. Spatial and temporal variation of polycyclic aromatic hydrocarbons in the arctic atmosphere. Environmental Science and Technology, 31:3593-3599.

Hanna, S.R., and P.J. Drivas, 1993. Modeling VOC emissions and air concentrations from the *Exxon Valdez* oil spill. Journal of the Air & Waste Management Association, 43:298-300.

Hanna, B.M., J.A. Hellebust and T.C. Hutchinson, 1975. Field studies on the phytotoxicity of crude oil to subarctic aquatic vegetation. Verhandlungen Internationale Vereinigung fur Theoretische and Angewandte Limnologie, 19:2165-2171.

Hannah, J.B., J.E. Hose, M.L. Landolt, B.S. Miller, S.P. Felton and W.T. Iwaoka, 1982. Benzo[*a*]pyrene-induced morphologic and developmental abnormalities in rainbow trout. Archives of Environmental Contamination and Toxicology, 11:727-734.

Hansen, J., 1998. Pollution and human health. In: AMAP Assessment Report: Arctic Pollution Issues, pp. 775-844. Arctic Monitoring and Assessment Program, Oslo, Norway.

Hanson, H.P., 1975. Photochemical degradation of petroleum hydrocarbon surface films on seawater. Marine Chemistry, 3:183-195.

Harris, C., 1997. The *Braer* incident: Shetland Islands, January 1993. In: J.M. Davies, G. Topping (Eds.). The impact of an oil spill in turbulent waters: The *Braer*, pp. 13-25. The Stationery Office Limited.

Harris, R.P., V. Berdugo, E.D.S. Corner, C.C. Kilvington and S.C.M. O'Hara, 1977. Factors affecting the retention of a petroleum hydrocarbon by marine planktonic copepods. In: D.A. Wolfe (Ed.).

Fate and Effects of Petroleum Hydrocarbons in Marine Organisms and Ecosystems, pp. 286-304. Pergamon Press.

Harris, R.E., G.W. Miller and W.J. Richardson, 2001. Seal responses to airgun sounds during summer seismic surveys in the Alaskan Beaufort Sea. Marine Mammal Science, 17:795-812.

Harrison, W., M.A. Winnik, P.T.Y. Kwong and D. Mackay, 1975. Disappearance of aromatic and aliphatic components from small sea-surface slicks. Environmental Science and Technology, 9:231-234.

Hartley, J., R. Trueman, S. Anderson, J. Neff, P. Dando and K. Fucik, 2003. Drill cuttings initiative – food chain effects literature review. Report UKOOA Drill Cuttings JIP.

Hartung, R. and G.S. Hunt, 1966. Toxicity of some oils to waterfowl. Journal of Wildlife Management, 30:564-570.

Harvey, J.S., B.P. Lyons, T.S. Page, C. Stewart and J.M. Parry, 1999. An assessment of the genotoxic impact of the *Sea Empress* oil spill by the measurement of DNA adduct levels in selected invertebrate and vertebrate species. Mutation Research, 441:103-114.

Harwell, M.A., W. Cooper and R. Flaak, 1992. Prioritizing ecological and human welfare risks from environmental stresses. Environmental Management, 16:451-464.

Haskell, S.P., R.M. Nielson, W.B. Ballard, M.A. Cronin and T.L. McDonald, 2006. Dynamic responses of calving caribou to oilfields in northern Alaska. Arctic 59:179-190.

Hassel, A., T. Knutsen, J. Dalen. S. Lokkeborg, K. Skaar, O. Ostensen, E.K. Haughland, M. Fonn, A. Hoines and O.A. Musund, 2003. Reaction of sandeel to seismic shooting: A field experiment and fishery statistical study. Institute of Marine Research, Bergen, Norway.

Hawkes, J.W., 1977. The effects of petroleum hydrocarbon exposure on the structure of fish tissues. In: D.A. Wolfe (Ed.). Fate and Effects of Petroleum Hydrocarbons in Marine Organisms and Ecosystems, pp. 115-128. Pergamon Press.

Hawkes, J.W. and C.M. Stehr, 1982. Cytopathology of the brain and retina of embryonic surf smelt (*Hypomesus pretiosus*) exposed to crude oil. Environmental Research, 27:164-178.

Hayashi, I., T. Kono and H. Yamakawa, 2000. Distributional characteristics of benthic organisms in shallow sublittoral rocky areas of Minuni, Fukui Prefecture: Part of the survey on the effects of the *Nakhodka* oil spill. Bulletin of the Japan Sea National Fisheries Research Institute, 50:43-137.

Health Canada, 1996. Health-based tolerable daily intake/concentrations and Tumourigenic doses/concentrations for Priority Substances. Health Canada.

Health Canada, 2004a. Exposure Guidelines for Residential Indoor Air Quality. Health Canada.

Health Canada, 2004b. Federal Contaminated Site Risk Assessment in Canada. Part 1. Guidance on Human Health Preliminary Quantitative Risk Assessment (PQRA). Part 2. Health Canada Toxicological Reference Values (TRVs).

Health Canada, 2004c. Canadian Handbook On Health Impact Assessment - Volume 1: The Basics.

Health Canada, 2007. Guidelines for Canadian Drinking Water Quality - Supporting Documents.

Hedtke, S.F. and F.A. Puglisi, 1982. Short-term toxicity of five oils to four freshwater species. Archives of Environmental Contamination and Toxicology, 11:425-430.

Heino, A., 1972. The *Palva* oil tanker disaster in the Finnish southwestern archipelago. I. The extent of oil pollution in the sea area between Uto and Kokar in 1969 and 1970. Aqua Fennica, 1972:116-121.

Helder, T., 1980. Effects of 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (TCDD) on early life stages of the pike (*Esox lucius*). Science of the Total Environment, 14:255-264.

Hellou, J., 1996. Polycyclic aromatic hydrocarbons in marine mammals, finfish, and molluscs. In: N.W. Beyer, G.H. Heinz, A.W. Redmon-Norwood (Eds.). Environmental Contaminants in Wildlife. Interpreting Tissue Concentrations, pp. 229-250. CRC Press.

Hellou, J., G. Stenson, I-H. Ni and J.F. Payne, 1990. Polycyclic aromatic hydrocarbons in muscle tissue of marine mammals from the Northwest Atlantic. Marine Pollution Bulletin, 21:469-473.

Hellou, J., C. Upshall, D. Taylor, P. O'Keefe, V. O'Malley and T. Abrajano, 1994. Unsaturated hydrocarbons in muscle and hepatopancreas of two crab species, *Chionoectes opilio* and *Hyas coarctatu.,* Marine Pollution Bulletin, 28:482-488.

Hellou, J., D. Mackay and B. Fowler, 1995. Bioconcentration of polycyclic aromatic hydrocarbons from sediments to muscle of finfish. Environmental Science and Technology, 29:2555-2560.

Hellstrom, T. and K.B. Doving, 1983. Perception of diesel oil by cod (*Gadus morhua* L). Aquatic Toxicology, 4:303-315.

Hendricks, J.D., T.R. Meyers, D.W. Shelton, J.L. Casteel and G.S. Bailey, 1985. Hepatocarcinogenicity of benzo[*a*]pyrene to rainbow trout by dietary exposure and intraperitoneal injection. Journal of the National Cancer Institute, 74:839-851.

Hepler, K.R., P.A. Hansen and D.R. Bernard, 1996. Impact of oil spilled from the *Exxon Valdez* on survival and growth of dolly varden and cutthroat trout in Prince William Sound. In: S.D. Rice, R.B. Spies, D.A. Wolfe, B.A. Wright (Eds.). Proceedings of the Exxon Valdez Oil Spill Symposium. Symposium 18:645-658. American Fisheries Society.

Herman, D.C., C.I. Mayfield and W.E. Inniss, 1991. The relationship between toxicity and bioconcentration of volatile aromatic hydrocarbons by the alga *Selenastrum capricornutum.* Chemosphere, 22:665-676.

Hessel, P.A., F.A. Herbert, L.S. Melenka, K. Yoshida and M. Nakaza, 1997. Lung health in relation to hydrogen sulfide exposure in oil and gas Workers in Alberta, Canada. American Journal of Industrial Medicine, 31:554-557.

Heubeck, M., 2006. The Shetland beached bird survey, 1979-2004. Marine Ornithology, 34:123-127.

Highsmith, R.C., T.L. Rucker, M.S. Stekoll, S.M. Saupe, M.R. Lindenberg, R.N. Jenne and W.P. Erickson, 1996. Impact of the *Exxon Valdez* oil spill on intertidal biota. In: S.D. Rice, R.B. Spies, D.A. Wolfe, B.A. Wright (Eds.). Proceedings of the Exxon Valdez Oil Spill Symposium. Symposium 18:212-237. American Fisheries Society.

Hilborn, R., 1996. Detecting population impacts from oil spills: a comparison of methodologies. In: S.D. Rice, R.B. Spies, D.A. Wolfe, B.A. Wright (Eds.). Proceedings of the Exxon Valdez Oil Spill Symposium. Symposium 18:639-644. American Fisheries Society.

Hinkel, K.M. and J.K. Hurd, Jr., 2006. Permafrost destabilisation and thermokarst following snow fence installation, Barrow, Alaska. Arctic, Antarctic and Alpine Research, 38:530-539.

Hinkle-Conn, C., J.W. Fleeger, J.C. Gregg and K.R. Carman, 1998. Effects of sediment-bound polycyclic aromatic hydrocarbons on feeding behavior in juvenile spot (*Leiostomus xanthurus* Lacepede: Pisces). Journal of Experimental Marine Biology and Ecology, 227:113-132.

Hirst, A.G. and P.G. Rodhouse, 2000. Impacts of geophysical seismic surveying on fishing success. Reviews in Fish Biology and Fisheries, 10:113-118.

Hirvi, J.-P., 1989. Summary of the environmental effects of the *Antonio Gramsci* oil spill in the Gulf of Finland in 1987. Arctic and Marine Oilspill Program Technical Seminar, 13:179-195.

Hirvi, J.-P., 1990. Suomenlahden öljyvahinko 1987. Vesi- ja ympäristöhallinnon julkaisuja A 51. Helsinki. (In Finnish)

Hites, R.A., R.E. LaFlamme and J.W. Farrington, 1977. Sedimentary polycyclic aromatic hydrocarbons: the historical record. Science, 198:829-831.

Hodgins, H.O., B.B. McCain and J.W. Hawkes, 1977. Marine fish and invertebrate diseases, host disease resistance, and pathological effects of petroleum. In: D.C. Malins (Ed.). Effects of Petroleum on Arctic and Subarctic Marine Environments and Organisms. Vol. II, Biological Effects, pp. 95-173. Academic Press.

Hoffman, D.J. and P.H. Albers, 1984. Evaluation of potential embryotoxicity and teratogenicity of 42 herbicides, insecticides, and petroleum contaminants to mallard eggs. Archives of Environmental Contamination and Toxicology, 13:15-27.

Holcombe, G.W., G.L. Phipps, M.L. Knuth and T. Felhaber, 1984. The acute toxicity of selected substituted phenols, benzenes and benzoic acid esters to fathead minnows *Pimephales promelas*. Environmental Pollution A, 35:367-381.

Holdway, D.A., 2002. The acute and chronic effects of wastes associated with offshore oil and gas production on temperate and tropical marine ecological processes. Marine Pollution Bulletin, 44:185-203.

Holliday, D.V., R.E. Piper, M.E. Clarke and C.F. Greenlaw, 1987. The effects of airgun energy release on the eggs, larvae and adults of the northern anchovy (*Engraulis mordax*). American Petroleum Institute, Washington, DC. Tracer Applied Sciences.

Holmes, W.N. and J. Cronshaw, 1977. Biological effects of petroleum on marine birds. In: D.C. Malins (Ed.). Effects of Petroleum on Arctic and Subarctic Marine Environments and Organisms. Volume II. Biological Effects, pp. 359-398. Academic Press.

Holmes, W.N., K.P. Cavenaugh and J. Cronshaw, 1978. The effect of ingested petroleum on oviposition and some aspects of reproduction in experimental colonies of mallard ducks (*Anas platyrhynchos*). 54:335-347.

Holmes, W.N., J. Gosline and J. Cronshaw, 1979. Effects of mild cold stress on the survival of seawater-adapted mallard ducks (*Anas platyrhynchos*) maintained on food contaminated with petroleum. Environmental Research, 20:425-444.

Holsbeek, L., C.R. Joiris, V. Debacker, I.B. Ali, P. Roose, J.-P. Nellissen, S. Gobert, J.-M. Bouquegneau and M. Bossicart, 1999. Heavy metals, organochlorines and polycyclic aromatic hydrocarbons in

sperm whales stranded in the southern North Sea during the 1994/95 winter. Marine Pollution Bulletin, 38:304-323.

Holzer, T.L. and D.L. Galloway, 2005. Impacts of land subsidence caused by withdrawal of underground fluids in the United States. In: J. Ehlen, W.C. Haneberg, R.A. Larson (Eds.). Humans as Geologic Agents. Reviews in Engineering Geology. XVI:87-99.

Horowitz, A. and R.M. Atlas, 1977. Response of microorganisms to an accidental gasoline spillage in an Arctic Freshwater Ecosystem. Applied and Environmental Microbiology, 33:1252-1258.

Horowitz, A., A. Sexstone and R.M. Atlas, 1978. Hydrocarbons and microbial activities in sediment of an arctic lake one year after contamination with leaded gasoline. Arctic, 31:180-191.

Hose, J.E., J.B. Hannah, D. DiJulio, M.L. Landolt, B.S. Miller, W.T. Iwaoka and S.P. Felton, 1982. Effects of benzo[*a*]pyrene on early development of flatfish. Archives of Environmental Contamination and Toxicology, 11:167-171.

Hose, J.E., M.D. McGurk, G.D. Marty, D.E. Hinton, E.D. Brown and T.T. Baker, 1996. Sublethal effects of the *Exxon Valdez* oil spill on herring embryos and larvae: morphological, cytogenetic and histopathological assessments, 1898-1991. Canadian Journal of Fisheries and Aquatic Sciences, 53:2355-2365.

Howgate, P., P.R. Mackie, K.J. Whittle, J. Farmer, A.D. McIntyre and A. Eleftheriou, 1977. Petroleum tainting in fish. Rapp. P.-v. Reun. Cons. int. Explor. Mer, 171:143-146.

Hrudey, S.E., 1979. Sources and characteristics of liquid process wastes from arctic offshore hydrocarbon exploration. Arctic, 32:3-21.

Hsiao, S.I.C., 1978. Effects of crude oils on the growth of Arctic marine phytoplankton. Environmental Pollution, 17:93-107.

Hsiao, S.I.C., D.W. Kittle and M.G. Foy, 1978. Effects of crude oils and the oil dispersant corexit on primary production of arctic marine phytoplankton and seaweed. Environmental Pollution, 15:209-221.

Huang, X-D., G.D. Dixon and B.M. Greenberg, 1993. Impacts of UV radiation and photomodification of the toxicity of PAHs to the higher plant *Lemna gibba* (duckweed). Environmental Toxicology and Chemistry, 12:1067-1077.

Hubbs, C.L. and A.B. Rechnitzer, 1952. Report of experiments designed to determine effects of underwater explosions on fish life. California Fish and Game, 38:333-366.

Hudson, P.J., 1985. Population parameters for the Atlantic alcidae. In: D.N. Nettleship, T.R. Birkhead (Eds.). The Atlantic Alcidae, pp. 233-261. Academic Press.

Huggett, R.J., R.A. Kimerle, P.M. Mehrle Jr., H.L. Bergman, 1992. Biomarkers Biochemical, Physiological and Histological Markers of Anthropogenic Stress. Lewis Publishers. 347p.

Huggett, R.J., J.J. Stegeman, D.S. Page, K.R. Parker, B. Woodin and J.S. Brown, 2003. Biomarkers in fish from Prince William Sound and the Gulf of Alaska: 1999-2000. Environmental Science and Technology, 37:4043-4051.

Hughes, M.R., C. Kasserra and B.R. Thomas, 1990. Effect of externally applied Bunker Fuel on body mass and temperature, plasma concentration, and water flux of glaucous winged gulls, *Larus glaucescens*. Canadian Journal of Zoology, 68:716-721.

Humphries, L., 2001. A review of relative sea level rise caused by mining-induced subsidence in the coastal zone: some implications for increased coastal recession. Climate Research, 18:147-156.

Hunt, G.L., 1987. Offshore oil development and seabirds: The present status of knowledge and long-term research needs. In: D.F. Boesch, N.N. Rabalais (Eds.). Long-term Environmental Effects of Oil and Gas Development, pp. 539-586. Elsevier.

Hurtig, A. and M. San Sebastian, 2002. Geographical differences in cancer incidence in the Amazon Basin of Ecuador in relation to residence near oil fields. International Journal of Epidemiology, 31:1021-1027.

Hutchinson, T.C. and W. Freedman, 1978. Effects of experimental crude oil spills on subarctic boreal forest vegetation near Norman Wells, N.W.T. Canada. Canadian Journal of Botany, 56:2424-2433.

Hutchinson, T.C. and J.A. Hellebust, 1978. Vegetational Recovery in the Canadian Arctic after Crude and Diesel Oil Spills. Ottawa, ON. Minister of Supply and Services. Ottawa, ON Canada. ALUR 75-76-83QS-8185-000-EE-A1. 62p.

Hutchinson, T.C., J.A. Hellebust, D. Mackay, D. Tam and P. Kauss, 1979. Relationship of hydrocarbon solubility to toxicity in algae and cellular membrane effects. In: 1979 Oil Spill Conference (Prevention, Behavior, Control, Cleanup), pp. 541-547. American Petroleum Institute, Washington, Special Publication No. 4308.

Hylland, K., G. Becker, J. Klungsøyr, T. Lang, A. McIntosh, B. Serigstad, J.E. Thain, K.V. Thomas, T.I.R. Utvik, A.D. Vethaak and W. Wosniok, 2002a. An ICES workshop on biological effects in pelagic ecosystems (BECPELAG): summary of results and recommendations. ICES ASC CM:2002/X:3, 6p.

Hylland, K., G. Becker, J. Klungsøyr, T. Lang, A. McIntosh, B. Serigstad, J.E. Thain, K.V. Thomas, T.I.R. Utvik, A.D. Vethaak and W. Wosniok, 2002b. An ICES workshop on biological effects in pelagic ecosystems (BECPELAG): overview of the programme. ICES ASC CM:2002/X:02, 6p.

Hylland, K., J. Beyer, M. Berntssen, J. Klungsøyr, T. Lang and L. Balk, 2006. May persistent organic pollutants affect fish populations in the North Sea? Journal of Toxicology and Environmental Health. Part A, 69:125-138.

Hylland, K, T. Lang, M. McIntosh, J.E. Thain, T.I.R. Utvik, A.D. Vethaak and W. Wosniok, 2007. Biological effects of contaminants in pelagic ecosystems – the BECPELAG workshop. In: K. Hylland, A.D. Vethaak, T. Lang (Eds.). Biological Effects of Contaminants in Marine Pelagic Ecosystems, pp. 3-8. SETAC special publication.

IARC, 1982. Monographs on the evaluation of carcinogenic risk of chemicals to humans. In: Some Industrial Chemicals and Dyestuffs. International Agency for Research on Cancer, Lyons, 29:93-148.

IARC, 1985. Polynuclear aromatic hydrocarbons. Part 4, Bitumens, coal-tars and derived products, shale-oils and soots. Summaries of data reported and evaluation. International Agency for Research on Cancer, volume 35. 271p.

IARC, 1987. Monographs on the evaluation of carcinogenic risks to humans. Overall evaluations of carcinogenicity. International Agency for Research on Cancer. Supplement 7. 440p.

IARC, 1989. Monographs on the evaluation of carcinogenic risk to humans. In: Crude Oils. International Agency for Research on Cancer, 45:119-158.

ICES, 2007. Report of the Working Group on Biological Effects of Contaminants, Alessandria, Italy, March 19-23. Report CM 2007/MHC:03. 129p.

Ikävalko, J., 2005. Effects of oil spills on arctic marine ecosystems. Finnish Institute of Marine Research, Report D4.2.3.2. 42p.

Ikävalko, J., G. Dieckmann and B. Gerdes, 2004. An experimental study on the effects of Statfjord crude oil, and application of Inipol and fish meal on the sea ice biota in Svalbard in February-April 2004. Research Report, www.arcop.fi

IMG-Golder Corp., 2002. Behavioural and physical response of riverine fish to airguns. Report prepared for WesternGeco, Calgary, Alberta.

Ingebrigtsen, K., J.S. Christiansen, O. Lindhe and I. Brandt, 2000. Disposition and cellular binding of ^{3}H-benzo[*a*]pyrene at subzero temperatures: studies in an aglomerular arctid teleost fish - the polar cod (*Boreogadus saida*). Polar Biology, 23:503-509.

Ingram, V. and R. Willemse, 2001. West Siberia oil industry environmental and social profile. Report prepared by IWACO Consultants for Water and Environment for the Dutch, German and Russian Greenpeace offices. Netherlands, 145p.

Ioppolo-Armanios, M.I., R. Alexander and R.I. Kagi, 1992. Identification and analysis of C_0-C_3 phenol in Australian crude oils. Organic Geochemistry, 18:603-609.

Irving, L., 1972. Arctic life of birds and mammals including man. Zoophysiology and Ecology, Vol. 2. Springer-Verlag. 192p.

Itämeren öljyvahinko, 1979. Sisäasiainministeriö. Ympäristönsuojeluosasto 1980 A:2. Valtion painatuskekus. 299p. (in Finnish)

James, M.O., 1989. Biotransformation and disposition of PAH in aquatic invertebrates. In: U. Varanasi (Ed.). Metabolism of Polycyclic Aromatic Hydrocarbons in the Aquatic Environment, pp. 69-91. CRC Press.

James, M.O. and J.R. Bend, 1980. Polycyclic aromatic hydrocarbon induction of cytochrome P-450-dependent mixed-function oxidases in marine fish. Toxicology and Applied Pharmacology, 54:117-133.

Jenkins, T.F., L.A. Johnson, C.M. Collins and T.T. McFadden, 1978. The physical, chemical and biological effects of crude oil spills on black spruce forest, interior Alaska. Arctic, 31:305-323.

Jensen, J. and L.E. Sverdrup, 2003. Polycyclic aromatic hydrocarbon ecotoxicity data for developing soil quality criteria. Reviews of Environmental Contamination and Toxicology, 179:73-97.

Jenssen, B.M., 1994. Review article – Effects of oil pollution, chemically treated oil, and cleaning on the thermal balance of birds. Environmental Pollution 86:207-215.

Jenssen, B.M., M. Ekker and K. Zahlsen, 1990. Effects of ingested crude oil on thyroid function and on the mixed function oxidase system in ducks. Comparative Biochemistry and Physiology C, 95:213-216.

Jepson, P.D., M. Arbelo, R. Deaville, I.A.P. Patterson, P. Castro, J.R. Baker, E. Degollada, H.M. Ross, P. Herraez, A.M. Pocknell, F. Rodgiquez, F.E. Howie, A. Espinosa, R.J. Reid, J.R. Jaber, V. Martin, A.A. Cunningham and A. Fernandez, 2003. Gas-bubble lesions in stranded cetaceans. Nature, 425:575-576.

Jobling, S. and J.P. Sumpter, 1993. Detergent components in sewage effluent are weakly oestrogenic to fish: An *in vitro* study using rainbow trout (*Oncorhynchus mykiss*) hepatocytes. Aquatic toxicology, 27:361-372.

Johannessen, K.I., 1976. Effects of seawater extract of Ekofisk oil on hatching success of Barents Sea capelin. ICES 1976/E:29. 12p.

Johansson, S., 1980. Impact of oil on the pelagic ecosystem. In: J.J. Kineman, R. Elmgren, S. Hansson (Eds.). The *Tsesis* oil spill, pp. 61-80. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, Boulder, Colorado.

Johansson, S., O. Larsson and P. Boehm, 1980. The *Tsesis* oil spill. Marine Pollution Bulletin, 11:284-293.

Johnson, F.G., 1977. Sublethal biological effects of petroleum hydrocarbon exposures: bacteria, algae and invertebrates. In: D.C. Malins (Ed.). Effects of Petroleum on Arctic and Subarctic Marine Environments and Organisms, pp. 371-318. Volume II. Biological Effects. Academic Press.

Johnson, S.R., 2002. Marine mammal mitigation and monitoring program for the 2001 Odoptu 3-D seismic survey, Sakhalin Island Russia: Executive summary. Report from LGL Ltd, Sidney, B.C., for Exxon Neftegas Ltd., Yuzhno-Sakhalinsk, Russia. 49p. (Also available as working paper SC/02/WGW/19, International Whale Commission, Western Gray Whale Working Group Meeting, Ulsan, South Korea, 22-25 October 2002. 48p.)

Johnson, J.H., P.W. Brooks, A.K. Aldridge and S.J. Rowland, 1978. Presence and sources of oil in the sediment and benthic community surrounding the *Ekofisk* field after the blowout at Bravo. In: P.L. Russell (Ed.). Proc. Conference on Assessment of the Ecological Impacts of Oil Spills, pp. 488-513. 14-17 June 1978, Keystone, Colorado.

Johnson, L.L., E. Casillas, T.K. Collier, B.B. McCain and U. Varanasi, 1988. Contaminant effects on ovarian development in English sole, *Parophrys vetulus*, from Puget Sound, Washington. Canadian Journal of Fisheries and Aquatic Sciences, 45:2133-2146.

Johnson, C.J., M.S. Boyce, R.L. Case, H.D. Cluff, R.J. Gau, A. Gunn and R. Mulders, 2005. Cumulative effects of human developments on arctic wildlife. Wildlife Monographs, 160:1-37.

Joly, K., C. Nelleman and I. Vistnes, 2006. A reevaluation of caribou distribution near an oilfield road on Alaska's North Slope. Wildlife Society Bulletin, 34:866-869.

Jones, K.C., J.A. Stratford, K.S. Waterhouse, E.T. Furlong, W. Giger, R.A. Hites, C. Schaffner and A.E. Johnston, 1989. Increases in the polynuclear aromatic hydrocarbon content of an agricultural soil over the last century. Environmental Science and Technology, 23:95-101.

Jordan, M.J., J.E. Hobbie and B.J. Peterson, 1978. Effect of petroleum hydrocarbons on microbial populations in an arctic lake. Arctic, 31:170-179.

Jørgensen, E.H. and J. Wolkers, 1999. Effect of temperature on the P4501A1 response in winter- and summer-acclimated arctic char (*Salvelinus alpinus*) after oral benzo[*a*]pyrene exposure. Canadian Journal of Fisheries and Aquatic Sciences, 56:1370-1375.

Jørgensen, E.H., M. Celander, A. Goksøyr and M. Iwata, 2001. The effect of stress on toxicant-dependent cytochrome-P450 enzyme responses in the arctic charr (*Salvelinus alpinus*). Environmental Toxicology and Chemistry, 20:2523-2529.

Jorgensen, T., S. Lokkeborg and A.V. Soldal, 2002. Residence of fish in the vicinity of a decommissioned oil platform in the North Sea. ICES Journal of Marine Science (suppl.), 59:s288-s293.

Jorgenson, M.T. and M.R. Joyce, 1994. Six strategies for rehabilitating land disturbed by oil development in arctic Alaska. Arctic, 47:374-390.

Kamp, K., D.N. Nettleship and P.G.H. Evans, 1994. Thickbilled murres of Greenland: status and prospects. In: D.N. Nettleship, J. Burger, M. Gochfeld (Eds.). Seabirds on Islands, pp. 133-154. BirdLife International. BirdLife Conservation Series No. 1.

Kanter, R., D. Straughan and W.N. Jesse, 1971. Effects of exposure to oil on *Mytilus californianus* from different localities. In: Proceedings of the Joint Conference on the Prevention and Control of Oil Spills, pp. 485-488.

Kaplan, I., S.-T. Lu, R.-P. Lee and G. Warrick, 1996. Polycyclic hydrocarbon biomarkers confirm selective incorporation of petroleum in soil and kangaroo rat liver samples near an oil well blowout site in the western San Joaquin Valley, California. Environmental Toxicology and Chemistry, 15:696-707.

Kauss, P.B. and T.C. Hutchinson, 1975. Effects of water-soluble petroleum components on the growth of *Chlorella vulgaris* Beijerinck. Environmental Pollution, 9:157-175.

Kenaga, E., 1982. Predictability of chronic toxicity from acute toxicity of chemicals in fish and aquatic invertebrates. Environmental Toxicology and Chemistry, 1:347-358.

Kenyon, T.N., F. Ladich and H.Y. Yan, 1998. A comparative study of hearing ability in fishes: the auditory brainstem response approach. Journal of Comparative Physiology, A182:307-318.

Kerr, J.B. and C.T. McElroy, 1993. Evidence for large upward trends of ultraviolet-B radiation linked to ozone depletion. Science, 262:1032-1034.

Kershaw, G.P., 1990. Movement of crude oil in an experimental spill on the SEEDS simulated pipeline right-of-way, Fort Norman, N.W.T. Arctic, 43:176-183.

Kertell, K., 1996. Response of Pacific loons (*Gavia pacifica*) to impoundments at Prudhoe bay, Alaska. Arctic, 49:356-366.

Keshava, C., R.L. Divi, D.L. Whipkey, B.L. Frye, E. McCanlies, M. Kuo, M.C. Poirier and A. Weston, 2005. Induction of CYP1A1 and CYP1B1 and formation of carcinogen-DNA adducts in normal mammary epithelial cells treated with benzo[*a*]pyrene. Cancer Letters, 221:213-224.

Kevan, P.G., B.C. Forbes, S.M. Kevan and V. Behan-Pelletier, 1995. Vehicle tracks on high Arctic tundra: their effects on the soil, vegetation, and soil arthropods. Journal of Applied Ecology, 32:655-667.

Kewen, T., G.E. Grisak and L.T.L. Callow, 1985. An investigation of ground water quality at a decommissioned sour gas plant in Alberta. In: Hazardous Wastes in Ground Water: A Soluble Dilemma, pp. 219-224.

Khan, R.A. and J. Kiceniuk, 1983. Effects of crude oils on the gastrointestinal parasites of two species of marine fish. Journal of Wildlife Diseases, 19:253-258.

Khan, R.A. and J. Kiceniuk, 1984. Histopathological effects of crude oil on Atlantic cod following chronic exposure. Canadian Journal of Zoology, 62:2038-2043.

Khan, R.A. and J.F. Payne, 2005. Influence of a crude oil dispersant, corexit 9527, and dispersed oil on capelin (*Mallotus villosus*), Atlantic cod (*Gadus morhua*), longhorn sculpin (*Myoxocephalus octodecemspinosus*) and cunner (*Tautogolabrus adspersus*). Bulletin of Environmental Contamination and Toxicology, 75:50-56.

Khan, A.A., R.W. Coppock, M.M. Schuler, L.Z. Florence, L.E. Lillie and M.S. Mostrom, 1996. Biochemical effects of Pembina Cardium crude oil exposure in cattle. Archives of Environmental Contamination and Toxicology, 30:349-355.

Khan, A.A., R.W. Coppock, M.M. Schuler and L. Geleta, 2002. Biochemical changes as early stage systemic biomarkers of petroleum hydrocarbon exposure in rats. Toxicology Letters, 134:195-200.

Khitun, O., 1997. Self-recovery after technogenic and natural disturbances in the central part of the Yamal Peninsula (Western Siberian Arctic). In: R.M.M. Crawford (Ed.). Disturbance and Recovery in Arctic Lands, pp. 531-552. NATO ASI Series. Series 2: Environment. Vol. 25. Kluwer Academic Publishers.

Kim, S., E.L. Stair, R.L. Lochmiller, D.P. Rafferty, J.L. Schroder, N.T. Basta, J.W. Lish and C.W. Qualls, Jr., 2001. Widespread risks of dental fluorosis in cotton rats (*Sigmodon hispidus*) residing on petrochemical waste sites. Journal of Toxicology and Environmental Health, A, 62:107-125.

Kineman, J.J., R. Elmgren and S. Sansson, 1980. The *Tsesis* oil spill. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, Boulder, Colorado, 296p.

King, J.M. and K.S. Coley, 1985. Toxicity of aqueous extracts of natural and synthetic oils to three species of *Lemna*. In: R.C. Bahner, D.J. Hansen (Eds.). Aquatic Toxicology and Hazard Assessment, pp. 302-309. Eighth Symposium, American Society for Testing and Materials, ASTM STP 891.

King, M.A. and C.A. Lefever, 1979. Effects of oil transferred from incubating gulls to their eggs. Marine Pollution Bulletin, 10:319-321.

Kingston, P.F., J.M.T. Dixon, S. Hamilton and D.C. Moore, 1995. The impact of the *Braer* oil spill on the macrobenthic infauna of the sediments off the Shetland Islands. Marine Pollution Bulletin, 30:445-459.

Kinloch, D., H. Kuhnlein and D.C.G. Muir, 1992. Inuit foods and diet: A preliminary assessment of benefits and risks. Science of the Total Environment, 122:247-278.

Kirshenblat, 1965. In PINRO, 2002.

Knox, E.G. and E.A. Gilman, 1997. Hazard proximities of childhood cancers in Great Britain from 1953-1980. Journal of Epidemiology and Community Health, 51:151-159.

Knox, E.G. and E.A. Gilman, 1998. Migration patterns of children with cancer in Britain. Journal of Epidemiology and Community Health, 52:716-726.

Ko, J.-Y. and J.W. Day, 2004. A review of ecological impacts of oil and gas development on coastal ecosystems in the Mississippi Delta. Ocean and Coastal Management, 47:597-623.

Kocan, R.M., J.E. Hose, E.D. Brown and T.T. Baker, 1996. Pacific herring (*Clupea pallasi*) embryo sensitivity to Prudhoe Bay petroleum hydrocarbons: laboratory evaluation and in situ exposure at oiled

and unoiled sites in Prince William Sound. Canadian Journal of Fisheries and Aquatic Sciences, 53:2366-2375.

Kokelj, S.V. and I. GeoNorth, 2002. Drilling mud sumps in the Mackenzie Delta region: construction, abandonment and past performance. Indian and Northern Affairs. Yellowknife, NWT, Canada. 55p.

Kooyman, G.L., R.W. Davis and M.A. Castellini, 1977. Thermal conductance of immersed pinniped and sea otter pelts before and after oiling with Prudhoe Bay crude. In: D. Wolfe (Ed.). Fate and Effects of Petroleum Hydrocarbons in Marine Organisms and Ecosystems, pp. 151-157. Pergamon Press.

Korn, S., N. Hirsch and J.W. Struhsaker, 1976. Uptake, distribution, and depuration of 14C-Benzene in northern anchovy, *Engraulis mordax*, and striped bass, *Morone saxatilis*. Fishery Bulletin, 74:545-551.

Korn, S., D.A. Moles and S.D. Rice, 1979. Effects of temperature on the median tolerance limit of pink salmon and shrimp exposed to toluene, naphthalene, and Cook Inlet crude oil. Bulletin of Environmental Contamination and Toxicology, 21:521-525.

Konstantinov, 1986. In PINRO, 2002.

Koroleva, 1977. In PINRO, 2002.

Korte, F., G. Kvesitadze, D. Ugrekdelidze, M. Gordeziani, G. Khatisashvili, O. Buadze, G. Zaalishvili and F. Coulston, 2000. Organic toxicants and plants. Ecotoxicology and Environmental Safety, 47:1-26.

Kosheleva, V.V., I.P. Migalofsky, M.A. Novikov, E.A. Gorbacheva and A.M. Lapteva, 1997. Response of aquatic organisms to environmental pollution during oil and gas field development on the Barents Sea shelf, PINRO Press, Murmansk. 92p. (In Russian, English translation supplied)

Kostianoy, A.G., S.A. Lebedev, K.T. Litovchenko, S.V. Stanichny and O.E. Pichuzhkina, 2004. Satellite remote sensing of oil spill pollution of the southeastern Baltic Sea. Gayana (Concepción), 68:327-332.

Kostyuchenko, L.P., 1972. Effects of elastic waves generated in marine seismic prospecting on fish eggs in the Black Sea. Hydrobiology Journal, 9:45-48.

Kotov, 1976. In PINRO, 2002.

Krahn, M.M., M.S. Myers, D.G. Burrows and D.C. Malins, 1984. Determination of metabolites of xenobiotics in the bile of fish from polluted waterways. Xenobiotica, 14:633-646.

Krahn, M.M., L.D. Rhodes, M.S. Myers, L.K. Moore, W.D. MacLeod Jr. and D.C. Malins, 1986. Associations between metabolites of aromatic compounds in bile and the occurrence of hepatic lesions in English sole (*Parophrys vetulus*) from Puget Sound, Washington. Archives of Environmental Contamination and Toxicology, 15:61-67.

Kravitz, M.J., J.O. Lamberson, S.P. Ferraro, R.C. Swartz, B.L. Boese and D.T. Specht, 1999. Avoidance response of the estuarine amphipod *Eohaustorius estuarius* to polycyclic aromatic hydrocarbon-contaminated, field-collected sediments. Environmental Toxicology and Chemistry, 18:1232-1235.

Kühn, R., M. Pattard, K.D. Pernak and A. Winter, 1989. Results of the harmful effects of water pollutants to *Daphnia magna* in the 21-day reproduction test. Water Research, 23:501-510.

Kuljukka-Rabb, T., L. Nylund, R. Vaaranrinta, K. Savela, P. Mutanen, T. Veidebaum, M. Sorsa, A. Rannug and K. Peltonen, 2002. The effect of relevant genotypes on PAH exposure-related biomarkers. Journal of Exposure Analysis and Environmental Epidemiology, 12:81-91.

La Bella, G., C. Frglia, A. Modica, S. Ratti and G. Rivas, 1996. First assessment of effects of air gun seismic shooting on marine resources in the central Adriatic Sea. Society of Petroleum Engineers, Inc. International Conference on Health, Safety and Environment. New Orleans, Louisiana, 9-12 June.

Lacroix, D.L., R.B. Lanctot, J.A. Reed and T.L. McDonald, 2003. Effect of underwater seismic surveys on molting male long-tailed ducks in the Beaufort Sea, Alaska. Canadian Journal of Zoology, 81:1862-1875.

Lancaster, J.E., M.G. Pawson, G.D. Pickett and S. Jennings, 1998. The impact of the *Sea Empress* oil spill on seabass recruitment. Marine Pollution Bulletin, 36:677-688.

Lance, B.K., D.B. Irons, S.J. Kendall and L.L. McDonald, 2001. An evaluation of marine bird population trends following the *Exxon Valdez* oil spill, Prince William Sound, Alaska. Marine Pollution Bulletin, 42:298-309.

Landrum, P.F., J.P. Giesy, J.T. Oris and P.M. Allred, 1987. Photoinduced toxicity of polycyclic aromatic hydrocarbons to aquatic organisms. In: J.H. Vandermeulen, S.E. Hrudey (Eds.). Oil in Freshwater: Chemistry, Biology, Countermeasure Technology, pp. 304-318. Pergamon Press.

Laubier, L., M. Le Moigne, P. Flammarion, E. Thybaud and D. Cossa, 2004. The monitoring programme of the ecological and ecotoxicological consequences of the *Erika* oil spill. Aquatic Living Resources, 17:239-241.

Lavrentyev, P.J., 1994, Anthropogenic stress in ciliate communities: a short-term study on arctic tundra lakes. Ergebnisse der Limnologie, 40:149-153.

Law, R.J., 1978. Determination of petroleum hydrocarbons in water, fish and sediments following the *Ekofisk* blow-out. Marine Pollution Bulletin, 9:321-324.

Lawhead, B.E., S.C. Bishop and W.A. Stubblefield, 1992. Caribou, drilling muds, and geophagy: a literature review and synthesis. BP Exploration (Alaska) Inc. Anchorage. 20p.

Lawson, D.E., 1986. Response of permafrost terrain to disturbance: a synthesis of observations from northern Alaska. Arctic Alpine Research, 18:1-17.

Le Boeuf, B.J., 1971. Oil contamination and elephant seal mortality: a 'negative' finding. In: D. Straughan (Ed.). Biological and Oceanographical Survey of the Santa Barbara Channel Oil Spill, pp. 277-285. Allan Hancock Foundation, Univ. of Southern California.

Lee, R.F. and J.W. Anderson, 2005. Significance of cytochrome P450 system responses and levels of bile fluorescent aromatic compounds in marine wildlife following oil spills. Marine Pollution Bulletin, 50:705-723.

Lee, W.Y. and J.A.C. Nicol, 1981. A study of the recovery of a marine isopod (*Sphaeroma quadridentatum*) from petroleum-induced sensitivity. In: F.J. Vernberg, A. Calabrese, F.P. Thurberg, W.B. Vernberg (Eds.). Biological Monitoring of Marine Pollutants, pp. 467-482. Academic Press.

Lee, R.F., R. Sauerheber and G.H. Dobbs, 1972. Uptake, metabolism and discharge of polycyclic aromatic hydrocarbons by marine fish. Marine Biology, 17:201-208.

Lee, K., K. Azetsu-Scott, S.E. Cobanli, J. Dalziel, S. Niven, G. Wohlgeschaffen and P. Yeats, 2005. Overview of potential impacts from produced water discharges in Atlantic Canada. In: S.L. Armsworthy, P.J. Cranford, K. Lee (Eds.). Offshore Oil and Gas Environmental Effects Monitoring. Approaches and Technologies, pp. 319-371. Battelle Press.

Leighton, F.A., D.B. Peakall and R.G. Butler, 1983. Heinz-Body hemolytic anemia from the ingestion of crude oil – a primary toxic effect in marine birds. Science, 220:871-873.

Leighton, F.A., R.G. Butler and D.B. Peakall, 1985. Oil and arctic marine birds: an assessment of risk. In: F.R. Englehardt (Ed.). Petroleum Effects in the Arctic Environment, pp. 183-215. Elsevier Science.

Lemiere, S., C. Cossu-Leguille, A. Bispo, M.J. Jourdain, M.C. Lanhers, D. Burnel and P. Vasseur, 2004. Genotoxicity related to transfer of oil spill pollutants from mussels to mammals via food. Environmental Toxicology, 19:387-395.

Lemiere, S., C. Cossu-Leguille, A. Bispo, M-J. Jourdain, M-C. Lanhers, D. Burnel and P. Vasseur, 2005. DNA damage measured by the single-cell gel electrophoresis (Comet) assay in mammals fed with mussels contaminated by the *Erika* oil-spill. Mutation Research, 581:11-21.

Leppanen, M.T. and J.V.K. Kukkonen, 2000a. Effect of sediment-chemical contact time on availability of sediment-associated pyrene and benzo[*a*]pyrene to oligochaete worms and semi-permeable membrane devices. Aquatic Toxicology, 49:227-241.

Leppanen, M.T. and J.V.K. Kukkonen, 2000b. Fate of sediment-associated pyrene and benzo[*a*]pyrene in the freshwater oligochaete *Lumbriculus variegatus* (Muller). Aquatic Toxicology, 69:199-212.

Levy, E.M., 1983. Commentary – What impact will the oil industry have on seabirds in the Canadian Arctic? Arctic, 36:1-4.

Levy, E. and M. Ehrhardt, 1981. Natural seepage of petroleum at Buchan Gulf, Baffin Island. Marine Chemistry, 10:355-364.

Lewis, S.J. and R.A. Malecki, 1984. Effects of egg oiling on larid productivity and population dynamics. Auk, 101:584-592.

Lewis, R.J., A.R. Schnatter, A.M. Katz, F.S. Thompson, N. Murray, G. Jorgensen and G. Theriault, 2000. Updated mortality among diverse operating segments of a petroleum company. Occupational and Environmental Medicine, 57:595-604.

Lewis, R.J., A.R. Schnatter, I. Drummond, N. Murray, F.S. Thompson, A.M. Katz, G. Jorgensen, M.J. Nicolich, D. Dahlman and G. Theriault, 2003. Mortality and cancer morbidity in a cohort of Canadian petroleum workers. Occupational and Environmental Medicine, 60:918-928.

Lin, Q., I.A. Mendelssohn, N.P. Bryner and W.D. Walton, 2005a. In-situ burning of oil in coastal marshes. 1. Vegetation recovery and soil temperature as a function of water depth, oil type, and marsh type. Environmental Science and Technology, 39:1848-1854.

Lin, Q., I.A. Mendelssohn, K. Carney, S.M. Miles, N.P. Bryner and W.D. Walton, 2005b. In-situ burning of oil in coastal marshes. 2. Oil spill cleanup efficiency as a function of oil type, marsh type and water depth. Environmental Science and Technology, 39:1855-1860.

Linden, O., 1975. Acute effects of oil and oil/dispersant mixture on larvae of Baltic herring. Ambio, 4:130-133.

Linden, O., 1976a. Effects of oil on the amphipod *Gammarus oceanicus*.

Environmental Pollution, 10:239-250.

Linden, O., 1976b. The influence of crude oil and mixtures of crude oil/dispersants on the ontogenetic development of the Baltic herring (*Clupea harengus membras* L.). Ambio, 5:136-140.

Linden, O., R. Elmgren and P. Boehm, 1979. The *Tsesis* oil spill; its impact on the coastal ecosystem of the Baltic Sea. Ambio, 8:248-253.

Lindström-Seppa, P., 1988. Biomonitoring of oil spill in a boreal archipelago by xenobiotic biotransformation in perch (*Perca fluviatilis*). Ecotoxicology and Environmental Safety, 15:162-170.

Linkins, A.E., R.M. Atlas and P. Gustin, 1978. Effect of surface applied crude oil on soil and vascular plant root respiration, soil cellulase, and hydrocarbon hydroxylase at Barrow, Alaska. Arctic, 31:355-365.

Lipscomb, T.P., R.K. Harris, R.B. Moeller, J.M. Pletcher, R.J. Haebler and B.E. Ballachey, 1993. Histopathologic lesions in sea otters exposed to crude oil. Veterinary Pathology, 30:1-11.

Ljungblad, D.K., B. Wursig, S.L. Swartz and J.M. Keene, 1988. Observations on the behavioural responses of bowhead whales (*Balaena mysticetus*) to active geophysical vessels in the Alaskan Beaufort Sea. Arctic, 41:183-194.

Lockhart, W.L., R.W. Danell and D.A.J. Murray, 1987. Acute toxicity bioassays with petroleum products: Influence of exposure conditions. In: J.H. Vandermeulen, S.E. Hrudey (Eds.). Oil in Freshwater: Chemistry, Biology, Countermeasure Technology, pp. 335-344. Pergamon Press.

Lockhart, W.L., D.A. Metner, D.A.J. Murray, R.W. Danell, B.N. Billeck, C.L. Baron, D.C.G. Muir and K. Chang-Kue, 1989. Studies to determine whether the condition of fish from the lower Mackenzie River is related to hydrocarbon exposure. Environmental Studies No. 61, Department of Indian Affairs and Northern Development. 84p.

Lockhart, W.L., P. Wilkinson, B.N. Billeck, G.J. Brunskill, R.V. Hunt and R. Wagemann, 1993. Polycyclic aromatic hydrocarbons and mercury in sediments from two isolated lakes in central and northern Canada. Water Science and Technology, 28:43-52.

Lockhart, W.L., D.A. Duncan, B.N. Billeck, R.A. Danell and M.J. Ryan, 1996. Chronic toxicity of the 'water-soluble fraction' of Norman Wells crude oil to juvenile fish. Spill Science and Technology Bulletin, 3:259-262.

Lockhart, W.L., B.N. Billeck, R.W. Danell, D.A.J. Murray, Y.K. York, D. Tilden, K. Hall and G.A. Stern, 2002a. Deleterious properties retained by diesel fuel spilled in northern Canada in winter. In: C.A. Brebbia (Ed.). Oil and Hydrocarbon Spills III. Modelling, Analysis and Control, pp. 374-382. WIT Press.

Lockhart, W.L., R.W. Danell, G.A. Stern and R.K. York, 2002b. Field and experimental tainting of arctic freshwater fish by crude and refined petroleum products. In: C.A. Brebbia (Ed.). Oil and Hydrocarbon Spills III. Modelling, Analysis and Control, pp. 329-337. WIT Press.

Løkkeborg, S., 1991. Effects of geophysical survey on catching success in longline fishing. ICES CM B:40: 9p.

Lokkeborg, S., O-B. Humborstad, T. Jorgensen and A.V. Soldal, 2002. Spatio-temporal variations in gillnet catch rates in the vicinity of North Sea oil platforms. ICES Journal of Marine Science, 59(suppl.):s294-s299.

Longpre, D., K. Lee and V. Jarry, 2000. The response of *Scirpus pungens* to crude oil-contaminated sediments. In: D.C. Gordon Jr., L.D. Griffiths, G.V. Hurley, A.L. Muecke, D.K. Muschenheim, P.G. Wells (Eds.). Understanding the Environmental Effects of Offshore Hydrocarbon Development. Canadian Technical Report of Fisheries and Aquatic Sciences, 2311, 34.

Lorentsen, S.-H. and T. Anker-Nilssen, 1993. Behaviour and oil vulnerability of fulmars *Fulmaris glacialis* during an oil spill experiment in the Norwegian Sea. Marine Pollution Bulletin, 26:144-146.

Loughlin, T.R., 1994. Marine Mammals and the *Exxon Valdez*. Academic Press, 395p.

Lowe, D.M. and M.N. Moore, 1979. The cytology and occurrence of granulocytomas in mussels. Marine Pollution Bulletin, 10:137-141.

Lyngs, P., 1994. Geirfuglen, et 150 år minde. Dansk Orn. Foren. Tidsskr. 88:49-72.

Lyons, B.P., J.S. Harvey and J.M. Parry, 1997. An initial assessment of the genotoxic impact of the *Sea Empress* oil spill by the measurement of DNA adduct levels in the intertidal teleost *Lopophrys pholis*. Mutation Research, 390L:263-268.

Lyons, R.A., J.M.F. Temple, D. Evans, D.L. Fone and S.R. Palmer, 1999. Acute health effects of the *Sea Empress* oil spill. Journal of Epidemiology and Community Health, 53:306-310.

Lyver, P.O. and Lutsel K'e First Nation, 2005. Monitoring barren-ground caribou body condition. Arctic, 58:44-54.

Macdonald, R.W. and J.M. Bewers, 1996. Contaminants in the arctic marine environment: priorities for protection. ICES Journal of Marine Science, 53:537-563.

Macdonald, R.W., T. Harner and J. Fyfe, 2005. Recent climate change in the Arctic and its impact on contaminant pathways and interpretation of temporal trend data. Science of the Total Environment, 342:5-86.

Mackay, D., 1985. Physical and chemical fate of spilled oil. In: F.R. Engelhardt (Ed.). Petroleum Effects in the Arctic Environment, pp. 37-61. Elsevier Applied Science.

Mackie, P.R., R. Hardy and K.J. Whittle, 1978. Preliminary assessment of the presence of oil in the ecosystem at Ekofisk after the blowout, April 22-30 1977. Journal of the Fisheries Research Board of Canada, 35:544-551.

Mageau, C., F.R. Engelhardt, E.S. Gilfillian and P.D. Boehm, 1987. Effects of short-term exposure to dispersed oil in arctic invertebrates. Arctic, 40(suppl. 1):162-171.

Maher, W.A., 1982. Preparation and characterization of water-soluble fractions of crude and refined oils for use in toxicity studies. Bulletin of Environmental Contamination and Toxicology, 29:268-272.

Majewski, H.S. and E. Scherer, 1985. Uptake and depuration of two polycyclic aromatic hydrocarbons, fluorene and benzo[*a*]pyrene, in the mussel *Mytilus edulis* and the diatom *Phaeodactylum tricurnutum*. Water Pollution Research Journal of Canada, 20:29-35.

Maki, A.W., 1992. Of measured risks – the environmental impacts of the Prudhoe Bay, Alaska, oil field. Environmental Toxicology and Chemistry, 11:1691-1707.

Maki, A.W., E.J. Brannon, L.G. Gilbertson, L.L. Moulton and J.R. Skalski, 1995. An assessment of oil spill effects on pink salmon populations following the *Exxon Valdez* oil spill – Part 2: Adults and escapement. In: P.G. Wells, J.N. Butler, J.S. Hughes (Eds.). Exxon Valdez Oil Spill: Fate and Effects in Alaskan waters, pp. 585-625. American Society for Testing and Materials, ASTM STP 1219.

Malins, D.C., 1977. Effects of petroleum on arctic and subarctic marine environments and organisms. Vol. II. Biological effects. Academic Press. 500p.

Malins, D.C., B.B. McCain, D.W. Brown, S-L. Chan, M.S. Myers, J.T. Landahl, P.G. Prohaska, A.J. Friedman, L.D. Rhodes, D.G. Burrows, W.D. Gronlund and H.O. Hodgins, 1984. Chemical pollutants in sediments and diseases of bottom-dwelling fish in Puget Sound, Washington. Environmental Science and Technology, 18:705-713.

Malme, C.I. and P.R. Miles, 1985. Behavioural responses of marine mammals (gray whales) to seismic discharges. In: G.D. Greene, F.R. Engelhardt, R.J. Paterson (Eds.). Proc. workshop on effects of explosives use in the marine environment, pp 253- 280. Technical Report 5, Canadian Oil and Gas Lands Administration, Environmental Protection Branch, Ottawa, 398p.

Malme, C.I., P.R. Miles, C.W. Clark, P. Tyack and J.E. Bird, 1984. Investigations of the potential effects of underwater noise from petroleum industry activities on migrating gray whale behavior. Phase II: January 1984 migration. Report from Bolt Beranek & Newman Inc., Cambridge, MA, for U.S. Minerals Management Service, Anchorage, AK. BBN Rep. 5586. NTIS PB86-218377.

Malme, C.I., P.R. Miles, P. Tyack, C.W. Clark and J.E. Bird, 1985. Investigation of the potential effects of underwater noise from petroleum industry activities on feeding humpback whale behavior. Report from Bolt Beranek & Newman Inc., Cambridge, MA, for U.S. Minerals Management Service, Anchorage, AK. BBN Rep. 5851; OCS Study MMS 85-0019. NTIS PB86-218385.

Malme, C.I., B. Würsig, J.E. Bird and P. Tyack, 1986. Behavioral responses of gray whales to industrial noise: feeding observations and predictive modeling. Outer Continental Shelf Environmental Assessment Program, Final Report. BBN Rep. 6265. 600p. OCS Study MMS 88-0048; NTIS PB88-249008.

Malme, C.I., B. Wursig, J.E. Bird and P. Tyack, 1987. Observations of feeding gray whale responses to controlled industrial noise exposure. Proceedings of the Ninth International Conference on Port and Ocean Engineering Under Arctic Conditions, pp. 55-73. The Geophysical Institute, University of Alaska.

Malme, C.I., B. Würsig, J.E. Bird and P. Tyack, 1988. Observations of feeding gray whale responses to controlled industrial noise exposure. In: W.M. Sackinger, M.O. Jeffries, J.L. Imm, S.D. Treacy (Eds.). Port and Ocean Engineering under Arctic Conditions, vol. II, pp. 55-73. Geophysical Institute, University of Alaska, Fairbanks.

Mann, D.A., D.M. Higgs, W.N. Tavolga, M.J. Souza and A.N. Popper, 2001. Ultrasound detection by clupeiform fishes. Journal of the Acoustical Society of America, 109:3048-3054.

Marty, G.D., J.E. Hose, M.D. McGurk, E.D. Brown and D.E. Hinton, 1997a. Histopathology and cytogenetic evaluation of Pacific herring larvae exposed to petroleum hydrocarbons in the laboratory or in Prince William Sound, Alaska, after the *Exxon Valdez* oil spill. Canadian Journal of Fisheries and Aquatic Sciences, 54:1846-1857.

Marty, G.D., R.A. Heintz and D.E. Hinton, 1997b. Histology and tera-

tology of pink salmon larvae near the time of emergence from gravel substrate in the laboratory. Canadian Journal of Zoology, 75:978-988.

Marty, G.D., J.W. Short, D.M. Dambach, N.H. Willits, R.A. Heintz, S.D. Rice, J.J. Stegeman, J.J. and D.E. Hinton, 1997c. Ascites, premature emergence, increased gonadal cell apoptosis, and cytochrome P4501A induction in pink salmon larvae continuously exposed to oil-contaminated gravel during development. Canadian Journal of Zoology, 75:989-1007.

Marty, G.D., M.S. Okihiro, E.D. Brown, D. Hanes and D.E. Hinton, 1999. Histopathology of adult Pacific herring in Prince William Sound, Alaska, after the *Exxon Valdez* spill. Canadian Journal of Fisheries and Aquatic Sciences, 56:419-426.

Mathieu, A., W. Melvin, B. French, M. Dawe, E.M. DeBlois, F. Power and U. Williams, 2005. Health effect indicators in American plaice (*Hippoglossoides platessoides*) from the Terra Nova development site, Grand Banks, NL, Canada. In: S.L. Armsworthy, P.J. Cranford, K. Lee (Eds.). Offshore Oil and Gas Environmental Effects Monitoring. Approaches and Technologies, pp. 297-317. Battelle Press.

Matishov, G.G., 1992. The reaction of bottom fish larvae to airgun pulses in the context of the vulnerable Barent Sea ecosystem. Fisheries and Offshore Petroleum Exploitation, 2nd International Conference, Bergen, Norway, 6-8 April.

Matishov et al., 2003. In PINRO, 2002.

Matkin, C.O., G.M. Ellis, M.E. Dahlheim and J. Zeh, 1994. Status of killer whales in Prince William Sound 1985-1992. In: T.R. Loughlin (Ed.). Marine Mammals and the Exxon Valdez, pp. 141-162. Academic Press.

Matson, C.W., J.C. Christian Franson, T. Hollmén, M. Kilpid, M. Harioe, P.L. Flint and J.W. Bickham, 2004. Evidence of chromosomal damage in common eiders (*Somateria mollissima*) from the Baltic Sea. Marine Pollution Bulletin, 49:1066-1071.

Maurer, R. and J. Kane, 1978. Zooplankton in the vicinity of the USNS *Potomac* oil spill (Baffin Bay, August 5, 1977). National Marine Fisheries Service, Northeast Fisheries Center, Naragansett, Rhode Island, Report 78-07, 12p.

Maxim, L.D. and R.W. Niebo, 2001. Appendix B. Oil spill analysis for North Slope oil production and transportation operations. Environmental Report for Trans Alaska Pipeline System Right-of-Way Renewal. Trans Alaska Pipeline System Owners. 36p.

Mayer, F.L. and M.R. Ellersieck, 1986. Manual of acute toxicity: Interpretation and data base for 410 chemicals and 66 species of freshwater animals. U.S. Departments of the Interior, Fish and Wildlife Service Research Publication, 160. 579p.

Mazmanidi, N.D. and G.I. Kovaleva, 1975. The effect of dissolved petroleum products on some of the aspects of carbohydrate metabolism in fishes and invertebrates. Ichtheology, 15:811-817.

Mazmanidi and Kotov, 1981. In PINRO, 2002.

McCabe, T.R., 1994. Assessing values of arctic wildlife and habitat subject to potential petroleum development. Landscape and Urban Planning, 28:33-45.

McCain, B.B., H.O. Hodgins, W.D. Gronlund, J.W. Hawkes, D.W. Brown, M.S. Myers and J.H. Vandermeulen, 1978. Bioavailability of crude oil from experimentally oiled sediments to English sole (*Parophrys vetulus*), and pathological consequences. Journal of the Fisheries Research Board of Canada, 35:657-664.

McCart, P. and J. Denbeste, 1987. Colonization of experimentally oiled substrates by periphyton and benthic macroinvertebrates in two arctic streams. In: J.H. Vandermeulen, S.E. Hrudey (Eds.). Oil in Freshwater: Chemistry, Biology and Countermeasure Technology, pp. 388-402. Pergamon Press.

McCarthy, J.F. and L.R. Shugart, 1990. Biomarkers of Environmental Contamination. Lewis Publishers. 457p.

McCarthy, J.J., O.F. Canzini, N.A. Leary, D.J. Dokken and K.S. White, 2001. Climate Change 2001: Impacts, Adaptation, and Vulnerability. Contributions of Working Group II to the Third Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press. 967p.

McCarthy, K., L. Walker and L. Vigoren, 2004. Subsurface fate of spilled petroleum hydrocarbons in continuous permafrost. Cold Regions Science and Technology, 38:43-54.

McCauley, R.D., M.N. Jenner, C. Jenner, K.A. McCabe and J. Murdoch, 1998. The response of humpback whales (*Megaptera novangliae*) to offshore seismic survey noise: preliminary results of observations about a working seismic vessel and experimental exposures. APPEA (Australian Petroleum Production & Exploration Association.) Journal 38:692-707.

McCauley, R.D., J. Fewtrell, A.J. Duncan, C. Jenner, M-N. Jenner, J.D. Penrose, R.I.T. Prince, A. Adhitya, J. Murdoch and K. McCabe, 2000a. Marine seismic surveys: Analysis of airgun signals; and effects of air gun exposure on humpback whales, sea turtles, fishes and squid. Rep. from Centre for Marine Science and Technology, Curtin Univ., Perth, W.A., for Austral. Petrol. Prod. Assoc., Sydney. 188p.

McCauley, R.D., J. Fewtrell, A.J. Duncan, C. Jenner, M.-N. Jenner, J.D. Penrose, R.I.T. Prince, A. Adhitya, J. Murdoch and K. McCabe., 2000b. Marine seismic surveys – a study of environmental implications. Australian Petroleum Production & Exploration Association, 40:692-708.

McCauley, R.D., J. Fewtrell and A.N. Popper, 2003. High intensity anthropogenic sound damages fish ears. Journal of the Acoustical Society of America, 113:638-642.

McCloskey, J.T. and J.T. Oris, 1993. Effect of anthracene and solar ultraviolet radiation exposure on gill ATPase and selected hematologic measurements in the bluegill sunfish *Lepomis macrochirus*. Aquatic Toxicology, 24:207-218.

McCourt, K.H., J.D. Feist, D. Doll and H.J. Russell, 1974. Disturbance studies of caribou and other mammals in the Yukon and Alaska 1972. Canadian Arctic Gas Study Ltd.. Biological Report Series. Vol. 5. 246p.

McDonald, B.G. and P.M. Chapman, 2002. PAH phototoxicity – an ecologically irrelevant phenomenon? Marine Pollution Bulletin, 44:1321-1326.

McElroy, A.E., J.W. Farrington and J.M. Teal, 1989. Bioavailability of polycyclic aromatic hydrocarbons in the aquatic environment. In: U. Varanasi (Ed.). Metabolism of Polycyclic Aromatic Hydrocarbons in the Aquatic Environment, pp. 1-39. CRC Press.

McEwan, E.H. and P.E. Whitehead, 1972. Reproduction in female reindeer and caribou. Canadian Journal of Zoology, 50:43-46.

McGurk, M.D. and E.D. Brown, 1996. Egg-larval mortality of Pacific herring in Prince William Sound, Alaska, after the *Exxon Valdez* oil spill. Canadian Journal of Fisheries and Aquatic Sciences, 53:2343-2354.

McIntosh, A.D., L.C. Massie, J.M. Davies, P. Howgate and P.R. Mackie, 1990. Assessment of fish from the northern North Sea for oil taint. ICES report CM 1990/E:24. 9p.

McIntosh, A.D., K. Hylland, B.T.G. Gowland and I.M. Davies, 2007. The assessment of hepatic 7-ethoxyresorufin *O*-deethylase (ER-OD) activity and CYP1A concentration in herring (*Clupea harengus*) and saithe (*Pollachius virens*) from two areas of the North Sea. In: K. Hylland, A.D. Vethaak, T. Lang (Eds.). Biological Effects of Contaminants in Marine Pelagic Ecosystems, pp. 93-102. SETAC special publication.

McKendrick, J.D., 1999. Arctic tundra recovery from crude oil after 24 years, Prudhoe Bay. Agroborealis, 31:28-35.

McKendrick, J.D. and W.M. Mitchell, 1978. Effects of burning crude oil spilled onto six habitat types in Alaska. Arctic, 31:277-295.

McLoughlin, P.D., M.K. Taylor, H.D. Cluff, R.J. Gau, R. Mulders, R.L. Case and F. Messier, 2003. Population viability of barren-ground grizzly bears in Nunavut and the Northwest Territories. Arctic, 56:185-190.

McMurray, S.T., R.L. Lochmiller, K. McBee and C.W. Qualls Jr., 1999. Indicators of immunotoxicity in populations of cotton rats (*Sigmodon hispidus*) inhabiting an abandoned oil refinery. Ecotoxicology and Environmental Safety, 42:223-235.

Metelev et al., 1971. In PINRO, 2002.

Mikhailova et al., 1986. In PINRO, 2002.

Mikhailova et al. 2001. In PINRO, 2002.

Miller, F.L. and E. Broughton, 1974. Calf mortality on the calving ground of Kaminuriak caribou. Canadian Wildlife Service. Ottawa. Report Series Number 26. 25p.

Miller, F.L. and A. Gunn, 1979. Caribou and muskoxen response to helicopter harassment, Prince of Wales Island 1976-77. Indian and Northern Affairs Canada. Ottawa, ON. AIPP report 1978; ESCOM report AI-30. 176p.

Miller, F.L. and A. Gunn, 2003. Catastrophic die-off of Peary caribou on western Queen Elizabeth Islands, Canadian High Arctic. Arctic, 56:381-390.

Miller, M.C., V. Alexander and R.J. Barsdate, 1978. The effects of oil spills on phytoplankton in an arctic lake and ponds. Arctic, 31:192-218.

Miller, M.C., J.R. Stout and V. Alexander, 1986. Effects of a controlled under-ice oil spill on invertebrates of an arctic and a sub-arctic stream. Environmental Pollution A, 42:99-132.

Miller, F., E. Broughton and A. Gunn, 1988. Mortality of migratory barren-ground caribou on the calving grounds of the Beverly herd, Northwest Territories 1981-83. Canadian Wildlife Service Occasional Paper 66. 26p.

Miller, G.W., R.E. Elliott, W.R. Koski, V.D. Moulton and W.J. Richardson, 1999. Whales. In: W.J. Richardson (Ed.). Marine Mammal and Acoustical Monitoring of Western Geophysical's Open-water Seismic Program in the Alaskan Beaufort Sea, 1998, pp. 5-1 to 5-109.

LGL Rep. TA2230-3. Report from LGL Ltd., King City, Ont., and Greeneridge Sciences Inc., Santa Barbara, CA, for Western Geophysical, Houston, TX and U.S. Nat. Mar. Fish. Serv., Anchorage, AK, and Silver Spring, MD. 390p.

Miller, G.W., V.D. Moulton, R.A. Davis, M. Holst, P. Millman, A. MacGillivray and D. Hannay, 2005. Monitoring seismic effects on marine mammals – southeastern Beaufort Sea, 2001-2002. In: S.L. Armsworthy, P.J. Cranford, K. Lee (Eds.). Offshore Oil and Gas Environmental Effects Monitoring, pp. 511-542. Approaches and Technologies. Battelle Press.

Minchew, C.D. and J.D. Yarbrough, 1977. The occurrence of fin rot in mullet (*Mugil cephalus*) associated with crude oil contamination of an estuarine pond-ecosystem. Journal of Fish Biology, 10:319-323.

Minchew, C.D., L.R. Brown and C.M. Ladner, 1979. The occurrence of 'white eye syndrome' in shrimp (*Penaeus aztecus*). In: Oil Spill Conference (Prevention, Behavior, Control, Cleanup), pp. 537-539. American Petroleum Institute, Special Publication No. 4308.

Mitchell, W.W., T.E. Loynachan and J.D. McKendrick, 1979. Effects of tillage and fertilization on persistence of crude oil contamination in an Alaskan soil. Journal of Environmental Quality, 8:525-532.

Mitson, R.B., 1995. Underwater noise of research vessels review and recommendations. ICES Cooperative Research Report 209, 61p.

MMS, 2002. Liberty Development and Production Plan, Final Environmental Impact Statement. Volume I (Executive Summary, Sections I through V). U.S. Department of the Interior, Minerals Management Service, Alaska Outer Continental Shelf Region, OCS EIS/EA, MMS 2002-019. 567p.

MMS, 2003. Beaufort Sea Planning Area Oil and Gas Lease Sales 186, 195, and 202. Final Environmental Impact Statement. Volume I (Executive Summary, Sections I through VI). U.S. Department of the Interior, Minerals Management Service, Alaska Outer Continental Shelf Region, OCS EIS/EA MMS 2003-001. 1526p.

Moen, B.E., B.E. Hollund, M. Berntsen, R. Flo, K.R. Kyvik and T. Riise, 1995. Exposure of the deck crew to carcinogenic agents on oil product tankers. Annals of Occupational Hygiene, 39:347-361.

Moles, A., 1998. Sensitivity of ten aquatic species to long-term crude oil exposure. Bulletin of Environmental Contamination and Toxicology, 61:102-107.

Moles, A., S.D. Rice and S. Korn, 1979. Sensitivity of Alaskan freshwater and anadromous fishes to Prudhoe Bay Crude Oil and benzene. Transactions of the American Fisheries Society, 108:408-414.

Moles, A., S. Bates, S. Rice and S. Korn, 1981. Reduced growth of coho salmon fry exposed to two petroleum components, toluene and naphthalene, in freshwater. Transactions of the American Fisheries Society, 110:430-436.

Monson, D.H., D.F. Doak, B.E. Ballachey, A. Johnson and J.L. Bodkin, 2000. Long-term impacts of the *Exxon Valdez* oil spill on sea otters, assessed through age-dependent mortality patterns. Proceedings of the National Academy of Sciences U.S.A. 97:6562-6567.

Monteiro, P.R., M.A. ReisHenriques and J. Coimbra, 2000. Plasma steroid levels in female flounder (*Platichthys flesus*) after chronic dietary exposure to single polycyclic aromatic hydrocarbons. Marine Environmental Research, 49:453-467.

Moore, S.F. and R.L. Dwyer, 1974. Effects of oil on marine organisms: a critical assessment of published data. Water Research, 8:819-827.

Morita, M., Y. Kusaka, Y. Deguchi, A. Moriuchi, Y. Nakanaga, M. Iki, S. Miyazaki and K. Kawahara, 1999. Acute health problems among the people engaged in the cleanup of the Nakhodka oil spill. Environmental Research A, 81:185-194.

Mosbech, A. and D. Boertmann, 1999. Distribution, abundance and reaction to aerial surveys of post-breeding king eiders (*Somateria spectabilis*) in western Greenland. Arctic, 52:18-203.

Mosbech, A. and C. Glahder, 1991. Assessment of the impact of helicopter disturbance on moulting pink-footed geese *Anser brachyrhychus* and barnacle geese *Branta leucopis* in Jameson Land. Ardea, 79:233-238.

Moulton, V.D. and J.W. Lawson, 2002. Seals, 2001. In: W.J. Richardson, J.W. Lawson (Eds.). Marine Mammal Monitoring of WesternGeco's open-water seismic program in the Alaskan Beaufort Sea, 2001, pp. 3-1 to 3-46. LGL Rep. TA2564-4. Report from LGL Ltd., King City, Ont., for WesternGeco LLC, Anchorage, AK; BP Explor. (Alaska) Inc., Anchorage, AK; and Nat. Mar. Fish. Serv., Anchorage, AK, and Silver Spring, MD. 95p.

Mozley, S.C. and M.G. Butler, 1978. Effects of crude oil on aquatic insects of tundra ponds. Arctic, 31:229-241.

Mulcahy, D.M. and B.E. Ballachey, 1994. Hydrocarbon residues in sea otter tissues. In: T.R. Loughlin (Ed.). Marine Mammals and the Exxon Valdez, pp. 313-330. Academic Press.

Muraveiko et al., 1994. In PINRO, 2002.

Murphy, S.M. and J.A. Curatolo, 1987. Activity budgets and movement rates of caribou encountering pipelines, roads, and traffic in northern Alaska. Canadian Journal of Zoology, 65:2483-2490.

Murphy, S.M. and B.E. Lawhead, 2000. Mammals of an arctic oil field. Caribou. In: J.C. Truett, S.R. Johnson (Eds.). The Natural History of an Arctic oil field, pp. 59-84. Academic Press.

Murray, D.A.J., W.L. Lockhart and G.R.B. Webster, 1984. Analysis of the water-soluble fraction of crude oils and petroleum products by gas chromatography. Oil and Petrochemical Pollution, 2:39-46.

Mustonen, M. and P. Tulkki, 1972. The *Palva* oil tanker disaster in the Finnish southwestern archipelago. IV. Bottom fauna in the oil polluted area. Aqua Fennica, 1972:137-141.

Myers-Smith, I.H., R.M. Thompson and F.S. Chapin, 2006. Cumulative impacts on Alaska arctic tundra of a quarter centrury of road dust. Ecoscience, 13:503-510.

Myhre, P., T. Bausant, R. Sundt, S. Sanni, R. Vabo, H.R. Skjoldal and J. Klungsoyr, 2004. Risk assessment of reproductive effects of alkyl phenols in produced water on fish stocks in the North Sea. Report AM-2004/018, RF-Akmiljo. 76p.

Myhrvold, A., J.P. Aabel, S.J. Cripps, O.H.J. Christie and O. Grahl-Nielsen, 1995. Draugen seabed monitoring survey 1994. Rogaland Research, Norway, Report no RF-95/012. 50p + app.

Nagel, N.B., 2001. Compaction and subsidence issues within the petroleum industry: from Wilmington to Ekofisk and beyond. Physical Chemical Earth – Part A, 26:3-14.

NAS, 2004. Cumulative environmental effects of oil and gas activities on Alaska's North Slope. National Academy of Sciences. National Academies Press. 304p.

National Assessment Synthesis Team, 2000. Potential consequences of climate variability and change for Alaska. In: Climate Change Impacts on the United States: The Potential Consequences of Climate Variability, pp. 283-312. Cambridge University Press.

Natural Resources Canada, 2006. Subsidence due to hydrocarbon extraction and flooding at Taglu and Niglintgak. Submission to the Joint Review Panel for the Mackenzie Gas Project. http://www.jointreviewpanel.ca/ (accessed June 2007).

Neff, J.M., 1987. Biological effects of drilling fluids, drill cuttings and produced water. In: D.F. Boesch, N.N. Rabalais (Eds.). Long-term Environmental Effects of Offshore Oil and Gas Development, pp. 469-538. Elsevier Applied Science.

Neff, J.M., 2002. Bioaccumulation in marine organisms: Effect of contaminants from oil well produced water. Elsevier. 452p.

Neff, J.M. and M. Gnatek, 2004. ERMS drilling wastes Task 1 – Toxicity. Summary of toxicity of drilling mud chemicals. Unpublished Report, Environmental Risk Management System Project, Statoil Norway, 38p.

Neff, J.M. and W.A. Stubblefield, 1995. Chemical and toxicological evaluation of water quality following the *Exxon Valdez* oil spill. In: P.G. Wells, J.N. Butler, J.S. Hughes (Eds.). Exxon Valdez Oil Spill: Fate and Effects in Alaskan Waters, pp. 141-177. American Society for Testing and Materials, Special Technical Publication 1219.

Neff, J.M., R.F. Hillman, R.S. Carr, R.L. Buhl and J.I. Lahey, 1987. Histopathologic and biochemical responses in arctic marine bivalve molluscs exposed to experimentally spilled oil. Arctic, 40(suppl. 1):220-229.

Neff, J.M., S. Ostazeski, W. Gardiner and I. Stejskal, 2000. Effects of weathering on the toxicity of three offshore Australian crude oils and a diesel fuel to marine animals. Environmental Toxicology and Chemistry, 19:1809-1821.

Nellbring, S., S. Hansson, G. Aneer and L Westin, 1980. Impact of oil on local fish fauna. In: J.J. Kineman, R. Elmgren, S. Hansson (Eds.). The *Tsesis* Oil Spill, pp. 193-201. Department of Commerce, National Oceanic and Atmospheric Administration, Boulder, Colorado.

Nellemann, C. and R.D. Cameron, 1996. Effects of petroleum development on terrain preferences of calving caribou. Arctic, 49:23-28.

Nellemann, C. and R.D. Cameron, 1998. Cumulative impacts of an evolving oil field complex on the distribution of calving caribou. Canadian Journal of Zoology, 76:1425-1430.

Nellemann, C., P. Jordhoy, O.G. Stoen and O. Strand, 2000. Cumulative impacts of tourist resorts on wild reindeer (*Rangifer tarandus tarandus*) during winter. Arctic, 53:9-17.

Nelson, F.E., O.A. Anisimov and N.I. Shiklomanov, 2002. Climate change and hazard zonation in the circum-Arctic permafrost regions. Natural Hazards, 26:203-225.

Nelson-Smith, 1977. In PINRO, 2002.

Nestler, J.M., G.R. Ploskey, J. Pickens, J. Menezes and C. Schilt, 1992. Responses of blueback herring to high-frequency sound and implications for reducing entrainment at hydropower dams. North American Journal of Fisheries Management, 12:667-683.

Nicolas, J.-M., 1999. Vitellogenesis in fish and the effects of polycyclic aromatic hydrocarbon contaminants. Aquatic Toxicology, 45:77-90.

Niederlehner, B.R., J. Cairns, Jr. and E.P. Smith, 1997. Modeling acute

and chronic toxicity of nonpolar narcotic chemicals and mixtures to *Ceriodaphnia dubia*. Ecotoxicology and Environmental Safety, 39:136-146.

Nikitik, C.C. and A.W. Robinson, 2003. Patterns in benthic populations in the Milford Haven waterway following the *Sea Empress* oil spill with special reference to amphipods. Marine Pollution Bulletin, 46:1125-1141.

NIOSH, 2005. NIOSH Pocket Guide to Chemical Hazards. Department of Health and Human Services, Centers for Disease Control, National Institute for Occupational Health and Safety. Pub no. 2005-149. 454p.

Noel, L.E., R.H. Pollard, W.B. Ballard and M.A. Cronin, 1998. Activity and use of active gravel pads and tundra by caribou, *Rangifer tarandus granti*, within the Prudhoe Bay oil field. Alaska. Canadian Field-Naturalist, 112:400-409.

Noel, L.E., K.R. Parker and M.A. Cronin, 2004. Caribou distribution near an oilfield road on Alaska's North Slope, 1978-2001. Wildlife Society Bulletin, 32:757-771.

Noel, L.E., K.R. Parker and M.A. Cronin, 2006. Response to Joly et al. (2006): A reevaluation of caribou distribution near an oilfield road on Alaska's North Slope. Wildlife Society Bulletin, 34:870-873.

Norcross, B.L., J.E. Hose, M. Frandsen and E.D. Brown, 1996. Distribution, abundance, morphological condition, and cytogenetic abnormalities of larval herring in Prince William Sound, Alaska, following the *Exxon Valdez* oil spill. Canadian Journal of Fisheries and Aquatic Sciences, 53:2376-2387.

Norwegian Ministry of Health and Care Services, 2001. Water Supply and Drinking Water Regulations. FOR 2001-12-04-1372. Oslo, Norway.

Norwegian Ministry of the Environment, 2005. The Government's Environmental Policy and the State of the Environment in Norway. Report No. 21 to the Storting (2004-2005). http://odin.dep.no/md/english/doc/white_paper/022001-040026/dok-bn.html

Notini, M., 1978. Long-term effects of an oil spill on *Fucus macrofauna* in a small Baltic bay. Journal of the Fisheries Research Board of Canada, 35:745-753.

Notini, M., 1980. Impact of oil on the littoral ecosystem. In: J. Kinemann, R. Elmgren, S. Hansson (Eds.), The *Tsesis* oil spill, pp. 193-201. US Department of Commerce, NOAA, Washington DC.

Novikov, 1992. In PINRO, 2002.

NTP, 2005. Report on Carcinogens, Eleventh Edition. U.S. Department of Health and Human Services, Public Health Service, National Toxicology Program.

Nunes, P. and P.E. Benville Jr., 1978. Acute toxicity of the water-soluble fraction of Cook Inlet crude oil to the Manila clam. Marine Pollution Bulletin, 9:324-331.

Nuttall, M., 2005. Hunting, herding, fishing, and gathering: indigenous peoples and renewable resource use in the Arctic. In: Arctic Climate Impact Assessment, pp.649-690. Cambridge University Press.

Obana, H., S. Hori, A. Nakamura and T. Kashimoto, 1983. Uptake and release of polynuclear aromatic hydrocarbons by short-necked clams (*Tapes japonica*). Water Research, 17:1183-1187.

O'Brien, W.J., 1978. Toxicity of Prudhoe Bay crude oil to Alaskan arctic zooplankton. Arctic, 31:219-226.

Ogata, M. and Y. Miyake, 1973. Identification of substances in petroleum causing objectionable odour in fish. Water Research, 7:1493-1504.

Olsgard, F. and J.S. Gray, 1995. A comprehensive analysis of the effects of offshore oil and gas exploration and production on the benthic communities of the Norwegian continental shelf. Marine Ecology Progress Series, 122:277-306.

Øritsland, N.A., F.R. Engelhardt, F.A. Juck, R.J. Hurst and P.D. Watts, 1981. Effect of crude oil on polar bears. Department of Indian Affairs and Northern Development, Ottawa. Environmental Studies, No. 24. Report No. QS-8283-020-EE-A1. 268p.

O'Rourke, D. and S. Connolly, 2003. Just Oil? The distribution of environmental and social impacts of oil production and consumption. Annual Review of Environment and Resources, 28:587-617.

Osborne, J. and C. Leeder, 1989. Acute and chronic toxicity of base oil and cuttings from three wells drilled in the Beaufort Sea. In: F.R. Engelhardt, J.P. Ray, A.H. Gillam (Eds.). Drilling Wastes, pp. 481-494. Elsevier Applied Science.

OSPAR, 1997. Agreed Ecotoxicological Assessment Criteria for Trace Metals, PCBs, PAHs, TBT and some Organochlorine Pesticides. Report of the Joint Meeting of the Oslo and Paris Commissions, Brussels, 2-5 September 1997, Annex 6. OSPAR Agreement 97-15. OSPAR Commission, London.

OSPAR, 2000. OSPAR Decision 2000/3 on the Use of Organic-Phase Drilling Fluids (OPF) and the Discharge of OPF-Contaminated Cuttings. OSPAR Commission, London.

OSWER, 2005. Office of Solid Waste and Emergency Response. Guidance for developing ecological soil screening levels. U.S. Environmental Protection Agency. Washington, DC. OSWER Directive 9285.7-55. 85p.

Ott, F.S., R.P. Harris and S.C.M. O'Hara, 1978. Acute and sublethal toxicity of naphthalene and three methylated derivatives to the estuarine copepod, *Eurytemora affinis*. Marine Environmental Research, 1:49-58.

Owen, R.E., 1984. Oil spill from the British tanker *Alvenas*. National Marine Fisheries Service, Southwest Fisheries Center, Miami, FL. 3p.

Paine, R.T., J.L. Ruesink, A. Sun, E.L. Soulanille, M.J. Wonham, C.D.G. Harley, D.R. Brumbaugh and D.L. Secord, 1996. Trouble on oiled waters: Lessons from the *Exxon Valdez* oil spill. Annual Review of Ecology and Systematics, 27:197-235.

Palinkas, L.A. J. Russel, M.A. Downs and J.S. Petterson, 1992. Ethnic differences in stress, coping, and depressive symptoms after the *Exxon Valdez* oil spill. Journal of Nervous and Mental Disease, 180:287-295.

Palinkas, L.A., J.S. Petterson, J.C. Russell and M.A. Downs, 1993. Community patterns of psychiatric disorders after the *Exxon Valdez* oil spill. American Journal of Psychiatry, 150:1517-1523.

Palinkas, L.A., J.S. Petterson, J.C. Russell and M.A. Downs, 2004. Ethnic differences in symptoms of post-traumatic stress after the *Exxon Valdez* oil spill. Prehospital and Disaster Medicine, 19:102-112.

Palm, R.C. Jr., D.B. Powell, A. Skillman and K. Gotfredsan, 2003. Immunocompetence of juvenile chinook salmon against *Listonella anguillarum* following dietary exposure to polycyclic aromatic hydrocarbons. Environmental Toxicology and Chemistry, 22:2986-2994.

Palmork, K.H. and J.E. Solbakken, 1980. Accumulation and elimination of radioactivity in the Norway lobster (*Nephrops norvegicus*) following intragastric administration of [9-^{14}C]phenanthrene. Bulletin of Environmental Contamination and Toxicology, 25:668-671.

Palmork, K.H. and J.E. Solbakken, 1981. Distribution and elimination of [9-^{14}C]phenanthrene in the horse mussel (*Modiola modiolus*). Bulletin of Environmental Contamination and Toxicology, 26:196-201.

Palmroth, M.R.T., J. Pichtel and J.A. Puhakka, 2002. Phytoremediation of subarctic soil contaminated with diesel fuel. Bioresource Technology, 84:221-228.

Pan, B.J., Y.J. Hong, G.C. Chang, M.T. Wang, F.F. Cinkotai and Y.C. Ko, 1994. Excess cancer mortality among children and adolescents in residential districts polluted by petrochemical manufacturing plants in Taiwan. Journal of Toxicology and Environmental Health, 43:117-129.

Park, J.M. and M.G. Holliday, 1998. Occupational-health aspects of marine oil-spill response. Michael Holliday and Associates, Ottawa, Canada. 37p.

Parker, M.W., S.W. Kress, R.T. Golightly, H.R. Carter, E.B. Parsons, S.E. Schubel, J.A. Boyce, G.J. McChesney and S.M. Wisely, 2007. Assessment of social attraction techniques used to restore a common murre colony in central California. Waterbirds, 30:17-28.

Parlee, B., M. Manseau and Lutsel K'e Dene First Nation, 2005. Using traditional knowledge to adapt to ecological change: Denesoline monitoring of caribou movements. Arctic, 58:26-37.

Patin, S., 1999. Environmental Impact of the Offshore Oil and Gas Industry. EcoMonitor Publishing. 425p. (Translated from Russian by Elena Cascio).

Patten, B.G., 1977. Sublethal biological effects of petroleum hydrocarbon exposures: Fish. In: D.C. Malins (Ed.). Effects of Petroleum on Arctic and Subarctic Marine Environments and Organisms, pp. 319-335. Vol. II. Biological Effects. Academic Press.

Patterson, B.R., B.T. Olsen and D.O. Joly, 2004. Population estimate for the bluenose-east caribou herd using post-calving photography. Arctic, 57:47-58.

Patton, J.F. and M.P. Dieter, 1980. Effects of petroleum hydrocarbons on hepatic function in the duck. Comparative Biochemistry and Physiology C, 65:33-36.

Payne, J.F., 1976. Field evaluation of benzopyrene hydroxylase induction as a monitor for marine petroleum pollution. Science, 191:945-946.

Payne, J.R. and G.D. McNabb Jr., 1984. Weathering of petroleum in the marine environment. Marine Technology Society Journal, 18:24-42.

Payne, J.F., J.W. Kiceniuk, W.R. Squires and G.L. Fletcher, 1978. Pathological changes in a marine fish after a 6-month exposure to petroleum. Journal of the Fisheries Research Board of Canada, 35:665-667.

Payne, J.F., L. Fancey, J. Kiceniuk, U. Williams, J. Osborne and A. Rahimtula, 1985. Mixed-function oxygenases as biological monitors around petroleum hydrocarbon development sites: potential for induction by diesel and other drilling mud base oils containing

reduced levels of polycyclic aromatic hydrocarbons. Marine Environmental Research, 17:328-332.

Payne, J.F., L. Fancey, J. Kiceniuk, U. Williams, A. Rahimtula, S. Khan and J. Osborne, 1987. Oil-based drilling fluids: are they an environmental risk? In: J. Kuiper, W.J. Van den Brink (Eds.). Fate and Effects of Oil in Marine Ecosystems, pp. 279-289. Martinus Nijhoff Publishers.

Payne, J.F., J. Kiceniuk, L.L. Fancey, U. Williams, G.L. Fletcher, A. Rahimutula and B. Fowler, 1988. What is a safe level of polycyclic aromatic hydrocarbons for fish: subchronic toxicity study on winter flounder (*Pseudopleuronectes americanus*). Canadian Journal of Fisheries and Aquatic Sciences, 45:1983-1993.

Payne, J.F., L.L. Fancey, J. Hellou, M.J. King and G.L. Fletcher, 1995. Aliphatic hydrocarbons in sediments: a chronic toxicity study with winter flounder (*Pleuronectes americanus*) exposed to oil well drill cuttings. Canadian Journal of Fisheries and Aquatic Sciences, 52:2724-2735.

PCF, 2000. Sour Gas: Questions and Answers. Petroleum Communication Foundation, Canadian Centre for Energy Information, Calgary, Alberta. 36p.

Peakall, D.B., D. Hallett, D.S. Miller and W.B. Kinter, 1980. Effects of ingested crude oil on black guillemots – combined field and laboratory study. Ambio, 9:28-30.

Peakall, D.B., J. Tremblay, W.B. Kinter and D.S. Miller, 1981. Endocrine dysfunction in seabirds caused by ingested oil. Environmental Research, 24:6-14.

Peakall, D.B., D.J. Hallett, J.R. Bend, G.L. Foureman and D.S. Miller, 1982. Toxicity of Prudhoe Bay crude oil and its aromatic fractions to nestling herring gulls. Environmental Research, 27:206-215.

Peakall, D.B., D.A. Jeffrey and D.S. Miller, 1985. Weight loss of herring gulls exposed to oil and oil emulsion. Ambio, 14:108-110.

Pearce, F., 1993. What turns an oil spill into a disaster? New Scientist 137:11-13.

Pearson, W.H., J.R. Skalski and C.I. Malme, 1992. Effects of sounds from a geophysical survey device on behaviour of captive rockfish (*Sebastes* spp.). Canadian Journal of Fisheries and Aquatic Sciences, 49:1343-1356.

Pearson, W.H., J.R. Skalski, S.D. Sulkin and C.I. Malme, 1994. Effects of seismic energy releases on the survival and development of zoeal larvae of Dungeness crab (*Cancer magister*). Marine Environmental Research, 38:93-113.

Pearson, W.H., R.A. Elston, R.W. Bienert, A.S. Drum and L.D. Antrim, 1999. Why did the Prince William Sound, Alaska, Pacific herring (*Clupea pallasi*) fisheries collapse in 1993 and 1994? Review of hypotheses. Canadian Journal of Fisheries and Aquatic Sciences, 56:711-737.

Pecséli, M., G. Pritzl, O. Andersen, G. Banta, A.B. Hansen, J. Christensen, T. Hviid, L. Malmborg, K. Johansen and J.L. Sorensen, 2002. The *Baltic Carrier* Oil Spill Monitoring and Assessment of Environmental Effects in Gronsund (DK), Storstroms AMt Teknik-og Miljoforvaltningen, Denmark, 158p.

Pelkonen, K. and P. Tulkki, 1972. The *Palva* oil tanker disaster in the Finnish southwestern archipelago. III. The littoral fauna of the oil polluted area. Aqua Fennica, 1972:129-136.

Pelletier, M.C., R.M. Burgess, K.T. Ho, A. Kuhn, R.A. McKinney and S.A. Ryba, 1997. Phototoxicity of individual polycyclic aromatic hydrocarbons and petroleum to marine invertebrate larvae and juveniles. Environmental Toxicology and Chemistry, 16:2190-2199.

Penrose, W.R., 1982. Effects of oil hydrocarbons and dispersants on fish. In: J.B. Sprague, J.H. Vandermeulen, P.G. Wells (Eds.). Oil and Dispersants in Canadian Seas: Research Appraisal and Recommendations, pp. 81-86. Environment Canada, EPA 3-EC-82-2.

Percy, J.A. and T.C. Mullin, 1975. Effects of crude oils on arctic marine invertebrates. Beaufort Sea Technical Report 11. Canada Department of the Environment, Victoria, B.C., 167p.

Percy, J.A. and T.C. Mullin, 1977. Effects of crude oil on the locomotory activity of arctic marine invertebrates. Marine Pollution Bulletin, 8:35-40.

Percy, J.A. and P.G. Wells, 1984. Effects of petroleum in polar marine environments. Marine Technology Society Journal, 18:51-61.

Perkins, R.A., S. Rhoton and C. Behr-Andres, 2005. Comparative marine toxicity testing: A cold-water species and standard warm-water test species exposed to crude oil and dispersant. Cold Regions Science and Technology, 42:226-236.

Petersen, G. and P. Kristensen, 1998. Bioaccumulation of lipophilic substances in fish early life stages. Environmental Toxicology and Chemistry, 17:1385-1395.

Peterson, C.H. and L. Holland-Bartels, 2002. Chronic impacts of oil pollution in the sea: risks to vertebrate predators. Marine Ecology Progress Series, 241:235-236.

Peterson, C.H., L.L. McDonald, R.H. Green and W.P. Erickson, 2001. Sampling design begets conclusions: the statistical basis for detection of injury to and recovery of shoreline communities after the *Exxon Valdez* oil spill. Marine Ecology Progress Series, 210:255-283.

Peterson, C.H., S.D. Rice, J.W. Short, D. Esler, J.L. Bodkin, B.E. Ballachey and D.B. Irons, 2003. Long-term ecosystem response to the *Exxon Valdez* oil spill. Science, 302:2082-2086.

Pfau, R.S., K. McBee and R.A. van Den Bussche, 2001. Genetic diversity of the major histocompatibility complex of cotton rats (*Sigmodon hispidus*) inhabiting an oil refinery complex. Environmental Toxicology and Chemistry, 20:2224-2228.

Pfeiphere and Platipira, 1986. In PINRO, 2002.

Piatt, J.F. and R.G. Ford, 1996. How many seabirds were killed by the *Exxon Valdez* oil spill? In: S.D. Rice, R.B. Spies, D.A. Wolfe, B.A. Wright (Eds.). Proceedings of the Exxon Valdez Oil Spill Symposium. Symposium 18:712-720. American Fisheries Society.

Piatt, J.F., C.J. Lensink, W. Butler, M. Kendziorek and D.R. Nysewander, 1990. Immediate impact of the *Exxon Valdez* oil spill on marine birds. Auk, 107:387-397.

Pickering, Q.H. and C. Henderson, 1966. Acute toxicity of some important petrochemicals to fish. Journal Water Pollution Control Federation, 38:1419-1429.

Picou, S.J. and D.A, Gill, 1996. The *Exxon Valdez* oil spill and chronic psychological stress. American Fisheries Society Symposium, 18:879-893.

Pomazovskaya, 1979. In PINRO, 2002.

PINRO, 2002. Contribution to AMAP prepared by Knipovich Polar Research Institute of Marine Fisheries and Oceanography (PINRO), Murmansk.

PINRO, 2006. Contribution to the AMAP Oil and Gas Assessment prepared by Knipovich Polar Research Institute of Marine Fisheries and Oceanography (PINRO), Murmansk.

Pinto, J.M., W.H. Pearson and J.W. Anderson, 1984. Sediment preferences and oil contamination in the Pacific sand lance *Ammodytes hexapterus*. Marine Biology, 193-204.

Poland, J.S., M.J. Riddle and B.A. Zeeb, 2003. Contaminants in the Arctic and the Antarctic: a comparison of sources, impacts, and remediation options. Polar Record, 39:369-383.

Pollard, R.H., W.B. Ballard, L.E. Noel and M.A. Cronin, 1996a. Parasitic insect abundance and microclimate of gravel pads and tundra within the Prudhoe Bay Oil Field, Alaska, in relation to use by caribou, *Rangifer tarandus granti*. Canadian Field-Naturalist, 110:649-658.

Pollard, R.H., W.B. Ballard, L.E. Noel and M.A. Cronin, 1996b. Summer distribution of caribou, *Rangifer tarandus granti*, in the area of the Prudhoe Bay oil field, Alaska 1990-1994. Canadian Field-Naturalist, 110:659-674.

Poole, C., N.A. Dreyer, M.H. Satterfield, L. Levin and K.J. Rothman, 1993. Kidney cancer and hydrocarbon exposures among petroleum refinery workers. Environmental Health Perspectives, 101(suppl.6):53-62.

Popper, A.N., M. Salmon and K.W. Horch, 2001. Acoustic detection and communication by decapod crustaceans. Journal of Comparative Physiology A, 187:83-89.

Pukkala, E., 1998. Cancer incidents among Finish oil refinery workers, 1971-1994. Journal of Occupational and Environmental Medicine, 40:675-679.

Qiniqtuagaksrat Utuqqanaat Inuuniagninisiqun: The Traditional land Use Inventory for the Mid-Beaufort Sea, 1980. North Slope Borough Commission on History and Culture, 1:107-108.

Raabe, G.K. and O. Wong, 1996. Leukemia mortality by cell type in petroleum workers with potential exposure to benzene. Environmental Health Perspectives, 104:1381-1392.

Rafferty, D.P., R.L. Lochmiller, K. McBee, C.W. Qualls and N.T. Basta, 2001. Immunotoxicity risks associated with land-treatment of petrochemical wastes revealed using an in situ rodent model. Environmental Pollution, 112:73-87.

Rajaretnam, G. and H.B. Spitz, 2000. Effect of leachability on environmental risk assessment for naturally occurring radioactive materials in petroleum oil fields. Health Physics, 78:191-198.

Ramachandran, S.D., P.V. Hodson, C.W. Khan and K. Lee, 2004. Oil dispersant increases PAH uptake by fish exposed to crude oil. Ecotoxicology and Environmental Safety, 59:300-308.

Ramseier, R.O., G.S. Gantcheff and L. Colby, 1973. Oil spill at Deception Bay, Hudson Strait, Environment Canada, Inland Waters Directorate, Water Resources Branch, Ottawa, Scientific Report Series 29.

Ravanko, O., 1972. The *Palva* oil tanker disaster in the Finnish southwestern archipelago. V. The littoral and aquatic flora of the polluted area. Aqua Fennica, 1972:142-144.

Ray, J.P. and F.R. Engelhardt, 1992. Produced water. Technological/ Environmental Issues and Solutions. Plenum Press. 616p.

Reiersen, L.-O. and J.A. Berge, 1985. Increased swimming activity of

the sand goby, *Pomatoschistus minutus*, exposed to low concentrations of water soluble fraction of North Sea crude oil. In: J.S. Gray, M.E. Christiansen (Eds.). Marine Biology of Polar Regions and Effects of Stress on Marine Organisms, pp. 379-387. John Wiley & Sons.

Reiter, Lt. G.A., 1981. Cold weather response F/V *Ryuyo Maru No. 2* St. Paul, Pribiloff Islands, Alaska, In: 1981 Oil Spill Conference (Prevention, Behavior, Control, Cleanup), American Petroleum Institute, pp. 227-231. Report no. 4334.

Ren, L., L.F. Zeiler, D.G. Dixon and B.M. Greenberg, 1996. Photoinduced effects of polycyclic aromatic hydrocarbons on *Brassica napus* (Canola) during germination and early seedling development. Ecotoxicology and Environmental Safety, 33:73-80.

Reynaud, S. and P. Deschaux, 2006. The effects of polycyclic aromatic hydrocarbons on the immune system of fish: a review. Aquatic Toxicology, 7:229-238.

Rice, S.D., 1973. Toxicity and avoidance tests with Prudhoe Bay oil and pink salmon fry. In: Proceedings of the Joint Conference on Prevention and Control of Oil Spills, pp. 667-670. American Petroleum Institute.

Rice, S.D., D.A. Moles and J.W. Short, 1975. The effect of Prudhoe Bay crude oil on survival and growth of eggs, alevins, and fry of pink salmon *Oncorhynchus gorbuscha*. In: Proceedings of the 1975 Conference on Prevention and Control of Oil Spills, pp. 503-507. American Petroleum Institute.

Rice, S.D., R.E. Thomas and J.W. Short, 1977. Effect of petroleum hydrocarbons on breathing and coughing rates and hydrocarbon uptake-depuration in pink salmon fry. In: F.J. Vernberg, A. Calabrese, F.P. Thurberg, W.B. Vernberg (Eds.). Physiological Responses of Marine Biota to Pollutants, pp. 259-277. Academic Press.

Rice, S.D., R.B. Spies, D.A. Wolfe and B.A. Wright, 1996. Proceedings of the *Exxon Valdez* Oil Spill Symposium. Symposium 18. American Fisheries Society. 932p.

Richardson, W.J., 1997. Bowhead responses to seismic, as viewed from aircraft. In: T. Newbury, K.L. Mitchell, C.T. Mitchell, P. Barton (Eds.). Arctic Seismic Synthesis and Mitigating Measures Workshop, pp. 15-26. U.S. Departments of the Interior, Minerals Management Service, Anchorage, Alaska.

Richardson, W.J., 1999. Marine mammal and acoustical monitoring of Western Geophysical's openwater acoustic program in the Alaskan Beaufort Sea, 1998. Report from LGL Ltd., King City, Ont., and Greenridge Sciences, Inc., Santa Barbara, CA, for Western Geophysical, Houston, TX, and NMFS, Anchorage, AK, and Silver Springs, MD. LGL Report TA2230-3. 390p.

Richardson, W.J. and C.I. Malme, 1993. Man-made noise and behavioural responses. In: J.J. Burns, J.J. Montague, C.J. Cowles (Eds.). The Bowhead Whale, pp. 631-700. Spec. Publ. 2. Soc. Mar. Mammal., Lawrence, Kansas.

Richardson, W.J., B. Würsig and C.R. Greene, 1986. Reactions of bowhead whales, *Balaena mysticetus*, to seismic exploration in the Canadian Beaufort Sea. Journal of the Acoustical Society of America, 79:1117-1128.

Richardson, W.J., R.A. Davis, C.R. Evans, D.K. Ljungblad and P. Norton, 1987. Summer distribution of bowhead whales, *Balaena mysticetus*, relative to oil industry activities in the Canadian Beaufort Sea, 1980-84. Arctic, 40:93-104.

Richardson, W.J., C.R. Greene, J.P. Hickie, R.A. Davis and D.H. Thomson, 1989. Effects of Offshore Petroleum Operations on Cold Water Marine Mammals: A Literature Review. Second edition. Report from LGL Ltd., King City, Ont., for the American Petroleum Institute, Washington, D.C. 385p.

Richardson, W.J., C.R. Greene, Jr., C.I. Malme and D.H. Thomson, 1995. Marine Mammals and Noise. Academic Press. 576p.

Richardson, W.J., G.W. Miller and C.R. Greene Jr., 1999. Displacement of migrating bowhead whales by sounds from seismic surveys in shallow waters of the Beaufort Sea. Journal of the Acoustical Society of America, 106(4, Pt. 2):2281.

Rickard, W.E. and J. Brown, 1974. Effects of vehicles on arctic tundra. Environmental Conservation, 1:55-62.

Ridoux, V., L. Lafontaine, P. Bustamante, F. Caurant, W. Dabin, C. Delcroix, S. Hassani, L. Meynier, V. Pereira da Silva, S. Simonin, M. Robert, J. Spitz and O. Van Canneyt, 2004. The impact of the *Erika* oil spill on pelagic and coastal marine mammals: combining demographic, ecological, trace metals and biomarker evidences. Aquatic Living Resources, 17:379-387.

Riksheim, H. and S. Johnsen, 1994. Determination of produced water contaminants in the marine environment. Society of Petroleum Engineers, SPE-27151:479-484.

Riley, R.T. and M.C. Mix, 1981. The effects of naphthalene on glucose metabolism in the European flat oyster *Ostrea edulis*. Comparative Biochemistry and Physiology, 70CL:13-20.

Ritchie, W., 1997. The *Braer* oil spill: Environmental circumstances and the remit of the Ecological Steering Group on the oil spill in Shetland (ESGOSS). In: J.M. Davies, G. Topping (Eds.). The Impact of an Oil Spill in Turbulent Waters: The *Braer*, pp. 1-12. The Stationery Office Limited, Edinburgh.

Ritchie, W. and M. O'Sullivan, 1994. The environmental impact of the wreck of the *Braer*. The Ecological Steering Group on the oil spill in Shetland. The Scottish Office.

Ritchie, G.D., M.Y.V. Bekkedal, A.J. Bobb and K.R. Still, 2001. Biological and health effects of JP-8 exposure. Report No. TOXDET 01-01. Naval Health Research Center Detachment (Toxicology). 44p.

Røe, T.I. and S. Johnsen, 1996. Discharges of produced water to the North Sea. In: M. Reed, S. Johnsen (Eds.). Produced water 2: Environmental Issues and Mitigation Technologies, pp. 13-25. Plenum Press.

Røe Utvik, T.I.R., 1999. Chemical characterization of produced water from four offshore oil production platforms in North Sea. Chemosphere, 39:2593-2606.

Roesijadi, G., D.L. Woodruff and J.W. Anderson, 1978, Bioavailability of naphthalenes from marine sediments artificially contaminated with Prudhoe Bay crude oil. Environmental Pollution, 15:223-229.

Rogerson, A., W.Y. Shiu, G.L. Huang, D. Mackay and J. Berger, 1983. Determination and interpretation of hydrocarbon toxicity to ciliate protozoa. Aquatic Toxicology, 3:229-238.

Rolfes, J. and J.T. Andersson, 2001. Determination of alkylphenols after derivatization to ferrocenecarboxylic acid esters with gas chromatography-atomic emission detection. Analytical Chemistry, 73:3073-3082.

Roos, P.H., S. Tschirbs, F. Pfeifer, P. Welge, A. Hack, M. Wilhelm and H.M. Bolt, 2004. Risk potentials for humans of original and remediated PAH-contaminated soils: application of biomarkers of effects. Toxicology, 205:181-194.

Rosenberg, D.M. and A.P. Wiens, 1976. Community and species responses Chironomidae (Diptera) to contamination of fresh waters by crude oil and petroleum products, with special reference to the Trail River, Northwest Territories. Journal of the Fisheries Research Board of Canada, 33:1955-1963.

Rosenberg, D.M. and V.H. Resh, 1993. Freshwater Biomonitoring and Benthic Macroinvertebrates. Chapman and Hall. 488p.

Rosenberg, D.M., A.P. Wiens and J.F. Flannagan, 1980. Effects of crude oil contamination on ephemeroptera in the Trail River, Northwest Territories, Canada. In: J.F. Flannagan, K.E. Marshall (Eds.). Advances in Ephemeroptera Biology, pp. 443-455. Plenum Publishing.

Roseneau. D.G. and P.M. Stern, 1974. Distribution and movements of the Porcupine Caribou Herd in northeastern Alaska 1972. Canadian Arctic Gas Study Ltd. Biological Report Series. Volume Seven. 209p.

Ross, P.I., 2002. Update. COSEWIC status report on the grizzly bear *Ursus arctos*. Environment Canada. Ottawa, Canada.

Rossi, S.S. and J.W. Anderson, 1978. Petroleum hydrocarbon resistance in the marine worm *Neanthes arenaceodentata* (Polychaeta: Annelida) induced by chronic exposure to No. 2 fuel oil. Bulletin of Environmental Contamination and Toxicology, 20:513-521.

Roubal, W.T., T.K. Collier and D.C. Malins, 1977. Accumulation and metabolism of carbon-14 labeled benzene, naphthalene and anthracene by young coho salmon (*Oncorhynchus kisutch*). Archives of Environmental Contamination and Toxicology, 5:513-529.

Routledge, E.J. and J.P. Sumpter, 1997. Structural features of alkylphenolic chemicals associated with estrogenic activity. Journal of Biological Chemistry, 272:3280-3288.

Rowe, D.W., J.B. Sprague, T.A. Heming and I.T. Brown, 1983. Sublethal effects of treated liquid effluent from a petroleum refinery. II. Growth of rainbow trout. Aquatic Toxicology, 3:161-169.

Rushton, L., 1993a. A 39-year follow-up of the U.K. oil refinery and distribution center studies: results for kidney cancer and leukemia. Environmental Health Perspectives, Supplements 101(Suppl. 6):77-84.

Rushton, L., 1993b. Further follow-up of mortality in a United Kingdom oil refinery cohort. British Journal of Industrial Medicine, 50:549-560.

Rushton, L. and H. Romaniuk, 1997. A case control study to investigate the risk of leukemia associated with exposure to benzene in petroleum marketing and distribution workers in the United Kingdom. Occupational and Environmental Medicine, 54:152-166.

Russell, D.E. and P. McNeil, 2002. Summer Ecology of the Porcupine Caribou Herd. Porcupine Caribou Management Board. Whitehorse, YT, Canada. 15p.

Russell, D.E., A.M. Martell and W.A.C. Nixon, 1993. Range ecology of the Porcupine caribou herd in Canada. Rangifer, 8:1-168.

Russell, J.C., M.A. Downs, J.S. Petterson and L.A. Palinkas, 1996. Psychological and social impacts of the *Exxon Valdez* oil spill and

cleanup. American Fisheries Society Symposium 18:867-878.

Russell, D.E., G. Kofinas and B. Griffith, 2002. Proceedings of the barren-ground caribou calving ground workshop. Canadian Wildlife Service. Technical Report Series No. 390. 47p.

Ryan, M.J., 1999. The effects of petroleum exposure on the feeding behaviour of rainbow trout using the water-soluble fraction of Norman Wells crude oil. M.Sc. Thesis, University of Manitoba, 80p.

Ryan, P.G., and W.R. Siegfried, 1994. The viability of small populations of birds: an empirical investigation of vulnerability. In: H. Remmert (Ed.), Minimum Animal Populations, pp. 3-22. Ecological Studies 106, Springer-Verlag.

Sabo, J. and J.J. Stegeman, 1977. Some metabolic effects of petroleum hydrocarbons in marine fish. In: F.J. Vernberg, A. Calabrese, F.P. Thurberg, W.B. Vernberg (Eds.). Response of Marine Biota to Pollutants, pp. 279-287. Academic Press.

Saetre, R. and E. Ona, 1996. Seismike undersokelser og pa fiskeegg og larver en vurdering av mulige effecter pa bestandsniva [Seismic investigations and damages on fish eggs and larvae; an evaluation of possible effects on stock level]. Fisken og Havet 1996:1-17, 1-8. (In Norwegian, English summary)

Samiullah, Y., 1985. Biological effects of marine oil pollution. Oil and Petrochemical Pollution, 2:235-264.

Samuels, W.B. and A. Ladino, 1983. Calculation of seabird population recovery from potential oilspills in the mid-Atlantic region of the United States. Ecological Modelling, 21:63-84.

Sand, O., 1981. The lateral-line and sound reception. In: W.N. Travolga, A.N. Popper, R.R. Fay (Eds.). Hearing and Sound Communication in Fishes, pp 459-480. Springer-Verlag.

San Sebastián, M. and A.K. Hurtig, 2004. Oil exploitation in the Amazon basin of Ecuador: a public health emergency. Pan-American Journal of Public Health, 15:205-211.

San Sebastián, M. M. Armstrong, J.A. Cordoba and C. Stephens, 2001. Exposure and cancer incidence near oil fields in the Amazon basin of Ecuador. Occupational and Environmental Medicine, 58:517-522.

San Sebastián, M., M. Armstrong and C. Stephens, 2002. Outcome of pregnancy among women living in the proximity of oil fields in the Amazon basin of Ecuador. International Journal of Occupational and Environmental Health, 8:312-319.

Sandercock, B.K., K. Martin and S.J. Hannon, 2005. Demographic consequences of age-structure in extreme environments: population models for arctic and alpine ptarmigan. Oecologia, 146:13-34.

Sanni, S., K.B. Oysaed, V. Hoivangli and B. Gaudebert, 1998. A continuous flow system (CFS) for chronic exposure of aquatic organisms. Marine Environmental Research, 46:97-101.

Santulli, A., A. Modica, C. Messina, L. Ceffa, A. Curatolo, G. Rivas, G. Fabi and V. D'Amelio, 1999. Biochemical responses of European sea bass (*Dicentrarchus labrax* L.) to the stress induced by offshore experimental seismic prospecting. Marine Pollution Bulletin, 38:1105-1114.

Satin, K.P., O. Wong, L.A. Yuan, W.J. Bailey, K.L. Newton, C-P. Wen, R.E. Swencicki, 1996. A 50-year mortality follow-up of a large cohort of oil refinery workers in Texas. Journal of Occupational and Environmental Medicine, 38:492-506.

Scanes, C.G. and F.M.A. Mcnabb, 2003. Avian models for research in toxicology and endocrine disruption. Avian and Poultry Biology Reviews, 14:21-52.

Scannell, P.W., D. Dasher, L. Duffy, R. Perkins and T. O'Hara, 2005. Acute and chronic toxicity of hydrocarbons in marine and fresh water with emphasis on Alaska species. A review of the literature. Alaska Department of Environmental Conservation. 201p.

Schaanning, M.T. and T. Bakke, 2006. Remediation of sediments contaminated with drill cuttings. A review of field monitoring and experimental data for validation of the ERMS sediment module. NIVA Report No. 5188-2006. Norwegian Institute for Water Research.

Schechter, M.T., W.O. Spitzer, M.E. Hutcheon, R.E. Dales, L.M. Eastridge, N. Steinmetz, P. Tousignant and C. Hobbs, 1989. Cancer downwind from sour gas refineries: the perception and the reality of an epidemic. Environmental Health Perspectives, 79:283-290.

Schekkerman, H., I. Tulp, T. Piersma and G.H. Visser, 2003. Mechanisms promoting higher growth rate in arctic than temperate shorebirds. Oecologia, 134:332-342.

Schindler, D.W., 1987. Detecting ecosystem responses to anthropogenic stress. Canadian Journal of Fisheries and Aquatic Sciences, 44(suppl. 1):6-25.

Schnatter, A.R., T.W. Armstrong, L.S. Thompson, M.J. Nicolich, A.M. Katz, W.W. Huebner and E.D. Pearlman, 1996. The relationship between low level benzene exposure and leukemia in Canadian petroleum distribution workers. Environmental Health Perspectives, 104:1375-1379.

Schroder, J.L., N.T. Basta, D.P. Rafferty, R.L. Lochmiller, S. Kim, C.W. Qualls and K. McBee, 1999. Soil and vegetation fluoride exposure pathways to cotton rats on a petrochemical-contaminated landfarm. Environmental Toxicology and Chemistry, 18:2028-2033.

Schroder, J.L., N.T. Basta, M. Payton, J.A. Wilson, R.I. Carlson, D.M. Janz and R.L. Lochmiller, 2003. Ecotoxicological risks associated with land treatment of petrochemical wastes. I. Residual soil contamination and bioaccumulation by cotton rats (*Sigmodon hispidus*). Journal of Toxicology and Environmental Health A, 66:305-325.

Schultz, M.E. and R.J. Schultz, 1982. Induction of hepatic tumors with 1,12-dimethylbenz[*a*]anthracene in two species of viviparous fishes (genus *Poeciliopsis*). Environmental Research, 27:337-351.

Schwarz, A.L. and G.L. Greer, 1984. Responses of Pacific herring, *Clupea harengus pallasi*, to some underwater sounds. Canadian Journal of Fisheries and Aquatic Sciences, 41:1183-1192.

Scott, B.F. and D.B. Shindler, 1978. Impact of crude oil on planktonic freshwater ecosystems. In: P.L. Russell (Coordinator). Proceedings of the Conference on the Assessment of Ecological Impacts of Oil Spills, pp. 712-733. American Institute of Biological Sciences, Keystone, Colorado.

Scott, A.P., S.I. Kristiansen, I. Katsiadaki, J. Thain, K-E. Tollefsen, 2007. Assessment of estrogen exposure in cod (*Gadus morhua*) and saithe (*Pollachius virens*) in relation to their proximity to an oil field. In: K. Hylland, A.D. Vethaak, T. Lang (Eds.). Biological Effects of Contaminants in Marine Pelagic Ecosystems, pp. 329-340. SETAC special publication.

SDWG, 2005. Arctic Children and Youth Health Indicators (Draft). Web site of the Sustainable Development Working Group of the Arctic Council. <www.sdwg.org>

Sergy, G.A. and P.J. Blackall, 1987. Design and conclusions of the Baffin Island Oil Spill Project. Arctic, 40(Suppl. 1):1-9.

Servheen, C., S. Herrero and B. Peyton, 1999. Status survey and conservation action plan. Bears. International Union for the Conservation of Nature. Bear Specialist Group. Gland, Switzerland. 320p.

Servos, M.R., 1999. Review of the aquatic toxicity, estrogenic responses and bioaccumulation of alkylphenols and alkylphenol polyethoxylates. Water Quality Research Journal of Canada, 34:123-177.

Severgazprom, 2007. Information on gas condensate plants. www.severgazprom.ru/english/infosgp/gasprocess.htm

Sexstone, A., P. Gustin and R.M. Atlas, 1978. Long term interactions of microorganisms and Prudhoe Bay crude oil in tundra soils at Barrow, Alaska. Arctic, 31:348-354.

SFT, 1999. Guidelines for the Risk Assessment of Contaminated Sites. Norwegian Pollution Control Authority. Report TA-1691.

Shallenberger, L.G., J.F. Aquavella and D. Donaleski, 1992. An updated mortality study of workers in three major United States refineries and chemical plants. British Journal of Industrial Medicine, 49:345-354.

Shane, S.H. and D.J. Schmidly, 1978. The population biology of the Atlantic bottlenose dolphin, *Tursiops truncatus*, in the Aransas Pass area of Texas. Report by Texas A & M Universtiy Texas Agricultureal Experiment Station, Department of Fisheries and Wildlife Sciences, College Station, TX 77843, for the Marine Mammal Commission, Washington, DC. NTIS No PB 283-393. 130p.

Shepherd, N.J. and C.F. Moffat, 1998. Sensory assessment – its role in monitoring hydrocarbon contamination in the marine environment. Presented at the Western European Fish Technologists Association meeting, Tromso, Norway, October 4-7. 9p.

Shideler, R. and J. Hechtel, 2000. Mammals of an arctic oil field. Grizzly bear. In: J.C. Truett, S.R. Johnson (Eds.). The Natural History of an Arctic Oil Field, pp. 105-132. Academic Press.

Shindler, D.B., B.F. Scott and D.B. Carlisle, 1975. Effect of crude oil on populations of bacteria and algae in artificial ponds subject to winter weather conditions and ice formation. Verhandlungen Internationale Vereinigung fur Theoretische and Angewandte Limnologie, 19:2138-2144.

Short, J.W. and P.M. Harris, 1996. Chemical sampling and analysis of petroleum hydrocarbons in near-surface seawater of Prince William Sound after the *Exxon Valdez* oil spill. In: S.D. Rice, R.B. Spies, D.A. Wolfe, B.A. Wright (Eds.). Proceedings of the Exxon Valdez Oil Spill Symposium. Symposium 18:17-28. American Fisheries Society.

Short, J.W., K.A. Kvenvolden, P.R. Carlson, F.D. Hostettler, P.J. Rosenbauer and B.A. Wright, 1999. Natural hydrocarbon background in benthic sediments of Prince William Sound, Alaska: oil vs. coal. Environmental Science and Technology, 33:34-42.

Short, J.W., B.A. Wright, K.A. Kvenvolden, P.R. Carlson, F.D. Hostettlerr and R.J. Rosenbauer, 2000. Response to comment on 'natural hydrocarbon background in benthic sediments of Prince William Sound, Alaska: oil vs. coal.' Environmental Science and Technology, 34:2066-2067.

Short, J.W., S.D. Rice, R.A. Heintz, M.G. Carls and A. Moles, 2003.

Long-term effects of crude oil on developing fish: lessons from the *Exxon Valdez* oil spill. Energy Sources, 25: 509-517.

Short, J.W., M.L. Lindeberg, P.M. Harris, J.M. Maselko, J.J. Pella and S.D. Rice, 2004. Estimate of oil persisting on the beaches of Prince William Sound 12 years after the *Exxon Valdez* oil spill. Environmental Science and Technology, 38:19-25.

Shugart, L.R., C.W. Theodorakis, A.M. Bickham and J.W. Bickham, 2003. Genetic effects of contaminant exposure and potential impacts on animal populations. In: D.J. Hoffman, B.A. Rattner, G.A. Burton Jr., J. Cairns Jr. (Eds.). Handbook of Ecotoxicology. Second Edition, pp. 1129-1147. Second Edition. Lewis Publishers.

Shulyakovski, 1985. In PINRO, 2002.

Sijm, D.T.H.M. and A. Opperhuizen, 1989. Biotransformation of organic chemicals by fish: enzyme activities and reactions. In: O. Hutzinger (Ed.). Handbook of Environmental Chemistry. Vol. 2E, Reactions and Processes, pp. 163-235. Springer Verlag.

Simpson, M.G. and T.H. Hutchinson, 1992. Toxicological pathology of dab *Limanda limanda* along pollution gradients in the southern North Sea. Marine Ecology Progress Series, 91:155-161.

Sims, R.C. and M.R. Overcash, 1983. Fate of polynuclear aromatic hydrocarbon compounds (PNAs) in soil-plant systems. Residue Reviews, 88:1-68.

Sindermann, C.J., 1979. Pollution-associated diseases and abnormalities of fish and shellfish: A review. Fishery Bulletin, 76:717-749.

Siniff, D.B., T.D. Williams, A.M. Johnson and D.L. Garshelis, 1982. Experiments on the response of sea otters *Enhydra lutris* to oil contamination. Biological Conservation, 23:261-272.

Siron, R., E. Pelletier, D. Delille and S. Roy, 1993. Fate and effects of dispersed crude oil under icy conditions simulated in mesocosms. Marine Environmental Research, 35:273-302.

Siron, R., E. Pelletier and C. Brochu, 1995. Environmental factors influencing the biodegradation of petroleum hydrocarbons in cold seawater. Archives of Environmental Contamination and Toxicology, 28:406-416.

Skadsheim, A., J.F. Borseth, A. Bjornstad, L. Camus, E. Aas and S. Sanni, 2005. Hydrocarbons and chemicals: Potential effects and monitoring in the deep sea. In: S.L. Armsworthy, P.J. Cranford, K. Lee (Eds). Offshore Oil and Gas Environmental Effects Monitoring. Approaches and Technologies, pp. 71-93. Battelle Press.

Skalski, J.R., W.H. Pearson and C.I. Malme, 1992. Effects of sounds from a geophysical survey device on catch-per-unit-effort in a hook-and-line fishery for rockfish (*Sebastes* spp.). Canadian Journal of Fisheries and Aquatic Sciences, 49:1357-1365.

Skjoldal, H.R., T. Dale, H. Haldorsen, B. Pengerud, T.F. Thingstad, K. Thessem and A. Aaberg, 1982. Oil pollution and plankton dynamics 1. Controlled ecosystem experiment during the 1980 spring bloom in Lindaspollene, Norway. Netherlands Journal of Sea Research, 16:511-523.

Smith, R.L. and J.A. Cameron, 1979. Effect of water soluble fraction of Prudhoe Bay crude oil on embryonic development of Pacific herring. Transactions of the American Fisheries Society, 108:70-75.

Smith, D.W. and T.D.W. James, 1985. Sump Studies V – Ecological changes adjacent to sumps at exploratory wellsites in the Mackenzie Delta and northern Yukon. Minister of Supply and Services Canada. Ottawa, ON. Environmental Studies No. 35QS-83700-000-EF-A1. 53p.

Smith, K.E.C., G.O. Thomas and K.C. Jones, 2001. Seasonal and species differences in the air-pasture transfer of PAHs. Environmental Science and Technology, 35:2156-2165.

Smits, J.E., M.E. Wayland, M.J. Miller, K. Liber and S. Trudeau, 2000. Reproductive, immune, and physiological end points in tree swallows on reclaimed oil sands mine sites. Environmental Toxicology and Chemistry, 19:2951-2960.

Smolders, R, K. Cooreman, P. Roose, R. Blust and W. de Coen, 2007. Cellular energy allocation in caged and pelagic free-living organisms along two North Sea pollution gradients. In: K. Hylland, A.D. Vethaak, T. Lang (Eds.). Biological Effects of Contaminants in Marine Pelagic Ecosystems, pp. 235-252. SETAC special publication.

Snow, N.B. and B.F. Scott, 1975. The effect and fate of crude oil spilt on two arctic lakes. In: 1975 Conference on Prevention and Control of Oil Pollution, pp. 527-534. American Petroleum Institute.

Soikkeli, M. and J. Virtanen, 1972. The *Palva* oil tanker disaster in the Finnish southwestern archipelago. II. Effects of oil pollution on the eider (*Somateria mollissima*) population in the archipelagoes of Kokar and Foglo, southwestern archipelago. Aqua Fennica, 1972:122-128.

Solbakken, J.E. and K.H. Palmork, 1981. Metabolism of phenanthrene in various marine animals. Comparative Biochemistry and Physiology C, 70:21-26.

Sorensen, P.W., R.J. Medved, M.A.M. Hyman and H.E. Winn, 1984. Distribution and abundance of cetaceans in the vicinity of human activities along the continental shelf of the northwestern Atlantic. Marine Environmental Research, 12:69-81.

Spies, R.B., S.D. Rice, D.A. Wolfe and B.A. Wright, 1996. Effects of the *Exxon Valdez* oil spill on the Alaskan coastal environment, pp. 1-16. American Fisheries Society, Symposium 18.

Spitzer, W.O., R.E. Dales, M.T. Schechter, S. Suissa, P. Tousignant, N. Steinmetz and M.E. Hutcheon, 1989. Chronic exposure to sour gas emissions: meeting a community concern with epidemiologic evidence. Canadian Medical Association Journal, 141:685-691.

Sprague, J.B., J.H. Vandermeulen and P.G. Wells, 1982. Oil and dispersants in Canadian seas: research appraisal and recommendations. Environment Canada, EPA 3-EC-82-2, 185p.

St. Aubin, D., 1990a. Physiologic and toxic effects on pinnipeds. In: J.R. Geraci, D.J. St. Aubin (Eds.). Sea Mammals and Oil: Confronting the Risks, pp. 103-127. Academic Press.

St. Aubin, D.J., 1990b. Physiologic and toxicologic effects on polar bears. In: J.R. Geraci, D.J. St. Aubin (Eds.). Sea Mammals and Oil: Confronting the Risks, pp. 235-239. Academic Press.

Stagg, R.M., A. McIntosh and P. Mackie, 1995. Elevation of hepatic monooxygenase activity in the dab (*Limanda limanda* L.) in relation to environmental contamination with petroleum hydrocarbons in the northern North Sea. Aquatic Toxicology, 33:245-264.

Stagg, R.M., J. Rusin, M.E. McPhail, A.D. McIntosh, C.F. Moffat and J.A. Craft, 2000. Effects of polycyclic aromatic hydrocarbons on expression of CYP1A in salmon (*Salmo salar*) following experimental exposure and after the *Braer* oil spill. Environmental Toxicology and Chemistry, 19:2797-2805.

Stebbing, A.R.D., 1982. Hormesis – the stimulation of growth by low levels of inhibitors. Science of the Total Environment, 22:213-234.

Stegeman, J.J., 1978. Influence of environmental contamination on cytochrome P-450 mixed-function oxygenases in fish: implications for recovery in the Wild Harbor marsh. Journal of the Fisheries Research Board of Canada, 35:668-674.

Stegeman, J.J. and J.J. Lech, 1991. Cytochrome P-450 monooxygenase systems in aquatic species: carcinogen metabolism and biomarkers for carcinogen and pollutant exposure. Environmental Health Perspectives, 90:101-109.

Stegeman, J.J., B.R. Woodin and A. Goksoyr, 1988. Apparent cytochrome P-450 induction as an indication of exposure to environmental chemicals in the flounder *Platichthys flesus*. Marine Ecology Progress Series, 46:55-60.

Stekoll, M.S., L.E. Clement and D.G. Shaw, 1980. Sublethal effects of chronic oil exposure on the intertidal clam *Macoma balthica*. Marine Biology, 57:51-60.

Stekoll, M.S., L. Deysher, R.C. Highsmith, S.M. Saupe, Z. Guo, W.P. Erickson, L. McDonald and D. Strickland, 1996. Coastal habitat injury assessment: Intertidal communities and the *Exxon Valdez* oil spill. In: S.D. Rice, R.B. Spies, D.A. Wolfe, B.A. Wright (Eds.). Proceedings of the Exxon Valdez Oil Spill Symposium. Symposium 18:177-192. American Fisheries Society.

Stepanjan, 2002. In PINRO, 2002.

Stephenson, M.T., 1992. A survey of produced water studies. In: J.P. Ray, F.R. Engelhardt (Eds.). Produced water. Technological/Environmental Isues and Solutions, pp. 1-11. Plenum Press.

Stewart, D.B., 2003. Possible impacts on overwintering fish of trucking granular materials over lake and river ice in the Mackenzie Delta area. Canada/Inuvialuit Fisheries Joint Management Committee Report 2003-1, v+12p.

Stirling, I., 1990. Polar bears and oil: Ecologic perspectives. In: J.R. Geraci, D.J. St. Aubin (Eds.). Sea Mammals and Oil: Confronting the Risks, pp. 233-234. Academic Press.

Stirling, I., 2002. Polar bears and seals in the Eastern Beaufort Sea and Amundsen Gulf: A synthesis of population trends and ecological relationships over three decades. Arctic, 56(suppl. 1):59-76.

Stone, C.J., 2003. The effects of seismic activity on marine mammals in UK waters 1998-2000. Joint Nature Conservancy, Report 323. Aberdeen, Scotland. 43p.

Stowe, T.J. and L.A. Underwood, 1984. Oil spillages affecting seabirds in the United Kingdom, 1966-1983. Marine Pollution Bulletin, 15:147-152.

Stroganov, 1971. In PINRO, 2002.

Strosher, M., 1996. Investigations of flare gas emissions in Alberta. Final Report to Environment Canada (Conservation and Protection), the Alberta Energy and Utilities Board and the Canadian Association of Petroleum Producers.

Stubblefield, W.A., G.A. Hancock, W.H. Ford and R.K. Ringer, 1995. Acute and subchronic toxicity of naturally weathered *Exxon-Valdez* crude oil in mallards and ferrets. Environmental Toxicology and Chemistry, 14:1941-1950.

Sturdevant, M.V., A.C. Wertheimer and J.L. Lum, 1996. Diets of juvenile pink and chum salmon in oiled and non-oiled hearshore habitats in Prince William Sound 1989 and 1990. In: S.D. Rice, R.B. Spies,

D.A. Wolfe, B.A. Wright (Eds.). Proceedings of the Exxon Valdez Oil Spill Symposium. Symposium 18:578-592. American Fisheries Society.

Sundberg, H., R. Ishaq, U. Tjarnlund, G. Akerman, K. Grunder, C. Bandh, D. Broman and L. Balk, 2006. Contributions of commonly analyzed polycyclic aromatic hydrocarbons (PAHs) to potential toxicity in early life stages of rainbow trout (*Oncorhynchus mykiss*). Canadian Journal of Fisheries and Aquatic Sciences, 63:1320-1333.

Sundt, R.C. and T. Baussant, 2003. Uptake, tissue distribution and elimination of C4-C7 alkylated phenols in cod. Dietary vs. waterborne exposure. Report-AM 2003/001, Akvamiljo, 27p.

Sutcliffe, D.W., 1994. Water quality and stress indicators in marine and freshwater systems: linking levels of organization. The Freshwater Biological Association, UK, 182p.

Sverdrup, A., E. Kjellsby, P.G. Kruger, R. Floysand, F.R. Knudsen, P.S. Enger, G. Serck-Hanssen and K.B. Helle, 1994. Effects of experimental seismic shock on vasoactivity of arteries, integrity of the vascular endothelium and on primary stress hormones in the Atlantic salmon. Journal of Fish Biology, 45:973-995.

Sydnes, L.K., T.H. Hemmingsen, S. Skare, S.H. Hansen, I-B. Falk-Petersen, S. Lonning and K. Ostgaard, 1985. Seasonal variations in weathering and toxicity of crude oil on seawater under arctic conditions. Environmental Science and Technology, 19:1076-1081.

Syroechkovski, E.E., 1990. Reindeer in the USSR: problems of protection and rational use. Rangifer, Special Issue 3:423-432.

Szaro, R.C., G. Hensler and G.H. Heinz, 1981. Effects of chronic ingestion of No. 2 fuel oil on mallard ducklings. Journal of Toxicology and Environmental Health, 7:426-436.

Tagatz, M.E., 1961. Reduced oxygen tolerance and toxicity of petroleum products for juvenile American shad. Chesapeake Science, 2:65-71.

Teal, J.M. and R.W. Howarth, 1984. Oil spill studies: A review of ecological effects. Environmental Management, 8:27-44.

Thomas, P. and L. Budiantara, 1995. Reproductive life-history stages sensitive to oil and naphthalene in Atlantic croaker. Marine Environmental Research, 39:147-150.

Thomas, R.E. and S.D. Rice, 1975. Increased opercular rates in pink salmon *Oncorhynchus gorbuscha* fry after exposure to the water-soluble fraction of Prudhoe Bay crude oil. Journal of the Fisheries Research Board of Canada, 32:2221-2224.

Thomas, P., H.W. Wofford and J.M. Neff, 1981. Biochemical stress responses of striped mullet (*Mugil cephalus* L.) to fluorene analogs. Aquatic Toxicology, 1:329-342.

Thomas, R.E., S.D. Rice, M.M. Babcock and A. Moles, 1989. Differences in hydrocarbon uptake and mixed function oxidase activity between juvenile and spawning adult coho salmon (*Oncorynchus kisutch*) exposed to Cook Inlet crude oil. Comparative Biochemistry and Physiology C, 93:155-159.

Thomas, K.V., J. Balaam, M.R. Hurst and J.E. Thain, 2004a. Bio-analytical and chemical characterisation of offshore produced water effluents for estrogen receptor (ER) agonists. Journal of Environmental Monitoring, 6:593-598.

Thomas, K.V., J. Balaam, M.R. Hurst and J.E. Thain, 2004b. Identification of *in vitro* estrogen and androgen receptor agonists in North Sea offshore produced water discharges. Environmental Toxicology and Chemistry, 23:1156-1163.

Tibbetts, P.J.C., L.T. Buchanan, L.J. Gawel and R. Large, 1992. A comprehensive determination of produced water composition. In: J.P. Ray, F.R. Engelhardt (Eds.). Produced Water. Technological/Environmental Isues and Solutions, pp. 97-112. Plenum Press.

Tkachenko and Zvylev, 1985. In PINRO, 2002.

Topping, G., J.M. Davies, P.R. Mackie and C.F. Moffat, 1997. The impact of the *Braer* spill on commercial fish and shellfish. In: J.M. Davies, G. Topping (Eds.). The Impact of an Oil Spill in Turbulent Waters: The *Braer*, pp. 121-143. The Stationery Office Limited, Edinburgh.

Trail, P.W., 2006. Avian mortality at oil pits in the United States: a review of the problem and efforts for its solution. Environmental Management, 38:532-544.

Troisi, G.M. and L. Borjesson, 2005. Development of an immunoassay for the determination of polyaromatic hydrocarbons in plasma samples from oiled seabirds. Environmental Science and Technology, 39:3748-3755.

Tronczynski, J., C. Munschy, K. Heas-Moisan, N. Guiot, I. Truquet, N. Olivier, S. Men and A. Furaut, 2004. Contamination of the Bay of Biscay by polycyclic aromatic hydrocarbons (PAHs) following the T/V *Erika* oil spill. Aquatic Living Resources, 17:243-259.

Truett, J.C., M.E. Miller and K. Kertell, 1997. Effects of arctic Alaska oil development on brant and snow geese. Arctic, 50:138-146.

Truscott, B., D.R. Idler and G.L. Fletcher, 1992. Alteration of reproductive steroids of male winter flounder (*Pleuronectes americanus*) chronically exposed to low levels of crude oil in sediments. Canadian Journal of Fisheries and Aquatic Sciences, 49:2190-2195.

Trust, K.A., D. Esler, B.R. Woodin and J.J. Stegeman, 2000. Cytochrome P450 1A induction in sea ducks inhabiting nearshore areas of Prince William Sound, Alaska. Marine Pollution Bulletin, 40:397-403.

Tsai, S.P., E.L. Gilstrap, S.R. Cowles, P.J. Snyder and C.E. Ross, 1993. A cohort mortality study of two Californian refinery and petrochemical plants. Journal of Occupational Medicine, 35:415-421.

Tsai, S.P., E.L. Gilstrap, S.R. Cowles, P.J. Snyder and C.E. Ross, 1996. Long-term follow-up mortality study of petroleum refinery and chemical plant employees. American Journal of Industrial Medicine, 29:79-96.

Tsunina, N.M., 2002. Evaluation of health risks of residents of Novokuibyshevsk. Gig Sanit. 4:15-17. (In Russian)

Tsunina, N.M., 2003. Hygienic assessment of the environment of a territorial industrial complex. Gig Sanit. 2:62-63. (In Russian)

Twerdok, L.E., 1999. Development of toxicity criteria for petroleum hydrocarbon fractions in the Petroleum Hydrocarbon Criteria Working Group approach for risk-based management of total petroleum hydrocarbons in soil. Drug and Chemical Toxicology, 22:275-291.

Ugrekhelidze, D., F. Korte and G. Kvesitadze, 1997. Uptake and transformation of benzene and toluene by plant leaves. Ecotoxicology and Environmental Safety, 37:24-29.

UNEP, 2001. In: C. Nellemann, L. Kullerud, I. Vistnes, B.C. Forbes, E. Husby, G.P. Kofinas, B.P. Kaltenborn, J. Rouaud, M. Magomedova, R. Bobiwash, C. Lambrechts, P.J. Shei, S. Tveitdal, O. Gron and T.S. Larsen. GLOBIO. Global Methodology for Mapping Human Impacts on the Biosphere. The Arctic 2050 Scenario and Global Application. United Nations Environment Programme. UNEP/DEWA/TR.01-3. 55p.

USAID, 2000. Comparative risk assessment in the Caspian Sea region. Volume III: Human Health Comparative Risk Assessment Case of Two Oil and Gas Development Activities. United States Agency for International Development. Report prepared by Abt Associates, Cambridge, MA. 65p.

USEPA, 2005. Final Ocean Discharge Criteria Evaluation of the Arctic NPDES General Permit for Oil and Gas Exploration (Permit No.: AKG280000). United States Environmental Protection Agency, Region 10, Seattle, Washington, USA. (October 15)

USEPA, 2007. National Primary Drinking Water Regulations, EPA 816-F-03-016, June 2003. U.S. Environmental Protection Agency. http://www.epa.gov/safewater/contaminants/index.html;

U.S. National Research Council, 1985. Oil in the Sea. Inputs, Fates and Effects. National Academies Press. 601p.

U.S. National Research Council, 2003. Oil in the Sea III. Inputs, Fates and Effects. National Academies Press. 265p.

Utvik, T.I.R. and L. Gärtner, 2007. Concentration of polycyclic aromatic hydrocarbons in seawater: Comparison of results from dispersion modelling with measured data from blue mussels and SPMD residues. In: K. Hylland, A.D. Vethaak, T. Lang (Eds.). Biological Effects of Contaminants in Marine Pelagic Ecosystems, pp. 29-42. SETAC special publication.

Valkenburg, P., D.G. Kelleyhouse, J.L. Davis and J.M. Van Hoef, 1994. Case history of the Fortymile caribou herd 1920-1990. Rangifer, 14:11-22.

Van de Wiele, T., L. Vanhaecke, C. Boeckaert, K. Peru, J. Headley, W. Verstraete and S. Siciliano, 2005. Human colon microbiota transform polycyclic aromatic hydrocarbons to estrogenic metabolites. Environmental Health Perspectives, 113:6-10.

van Oostdam, J. and N. Tremblay, 2003. Biological monitoring: human tissue levels of environmental contaminants. In: AMAP Assessment 2002: Human Health in the Arctic, pp. 31-56. Artic Monitoring and Assessment Programme, Oslo.

van Overbeek, J. and R. Blondeau, 1954. Mode of action of phytotoxic oils. Weeds, 3:55-65.

Van Pelt, T.I. and J.F. Piatt, 1995. Deposition and persistence of beachcast seabird carcasses. Marine Pollution Bulletin, 30:794-802.

van Schooten, F.J., L.M. Maas, E.J.C. Moonen, J.C.S. Kleinjans and R. van der Oost, 1995. DNA dosimetry in biological indicator species living on PAH-contaminated soils and sediments. Ecotoxicology and Environmental Safety, 30:171-179.

Van Tamelen, P.G. and M.S. Stekoll, 1996. Population response of the brown alga *Fucus gardneri* and other algae in Herring Bay, Prince William Sound, to the *Exxon Valdez* oil spill. In: S.D. Rice, R.B. Spies, D.A. Wolfe and B.A. Wright (Eds.). Proceedings of the Exxon Valdez Oil Spill Symposium. Symposium 18:193-211. American Fisheries Society.

Vandermeulen, J.H., 1982. Some conclusions regarding long-term biological effects of some major oil spills. Philosophical Transactions of the Royal Society of London B 297:335-351.

Vandermeulen, J.H., 1987. Toxicity and sublethal effects of petroleum hydrocarbons in freshwater biota. In: J.H. Vandermeulen, S.E. Hrudey (Eds.). Oil in Freshwater: Chemistry, Biology, Countermeasure Technology, pp. 267-303. Pergamon Press.

Vandermeulen, J.H. and D.C. Gordon Jr., 1976. Re-entry of 5-year-old stranded bunker C fuel oil from a low-energy beach into the water, sediments, and biota of Chedabucto Bay, Nova Scotia. Journal of the Fisheries Research Board of Canada, 33:2002-2010.

Vandermeulen, J.H. and S.E. Hrudey, 1997. Oil in Freshwater: Chemistry, Biology, Countermeasure Technology. Pergamon Press.

Varanasi, U. and Malins, D.C., 1977. Metabolism of petroleum hydrocarbons: Accumulation and biotransformation in marine organisms. In D.C. Malins (Ed.). Effects of Petroleum on Arctic and Subarctic Marine Environments and Organisms, pp. 175-270. Vo1. II. Biological Effects. Academic Press.

Verma, D.K., D.M. Johnson and J.D. McLean, 2000. Benzene and total hydrocarbon exposures in the upstream petroleum oil and gas industry. American Industrial Hygiene Association Journal, 61:255-263.

Veselovski and Veselova, 1990. In PINRO, 2002.

Vilchek, G.E. and A.A. Tishkov, 1997. Usinsk oil spill: environmental catastrophe or routine event? In: R.M.M. Crawford (Ed.). Disturbance and Recovery in Arctic Lands, pp. 411-420. NATO ASI Series Series 2: Environment. Vol. 25. Kluwer Academic Publishers.

Vincent, G., Le Guen, B. and Le Floche S., 2001. Accident of the oil tanker *Baltic Carrier* off the Danish coastline. Final Report, European Task Force in Denmark, European Commission, 10p.

Vistnes, I. and C. Nellemann, 2001. Avoidance of cabins, roads, and power lines by reindeer during calving. Journal of Wildlife Management, 65:915-925.

Vistnes, I., C. Nellemann, P. Jordhoy and O. Strand, 2001. Wild reindeer: impacts of progressive infrastructure development on distribution and range use. Polar Biology, 24:531-537.

Vistnes, I., C. Nellemann, P. Jordhoy and O. Strand, 2004. Effects of infrastructure on migration and range use of wild reindeer. Journal of Wildlife Management, 68:101-108.

Vives, I., J.O. Grimalt, P. Fernandez and B. Rosseland, 2004. Polycyclic aromatic hydrocarbons in fish from remote and high mountain lakes in Europe and Greenland. Science of the Total Environment, 324:67-77.

Voropayev et al., 1986. In PINRO, 2002.

Votier, S.C., B.J. Hatchwell, A. Beckerman, R.H. McCleery, F.M. Hunter, J. Pellatt, M. Trinder and T.R. Birkhead, 2005. Oil pollution and climate have wide-scale impacts on seabird demographics. Ecology Letters, 8:1157-1164.

Waldner, C.L., C.S. Ribble and E.D. Janzen, 1998. Evaluation of the impact of a natural gas leak from a pipeline on productivity of beef cattle. Journal of the American Veterinary Medical Association, 212:41-48.

Waldner, C.L., C.S. Ribble, E.D. Janzen and J.R. Campbell, 2001a. Associations between oil- and gas-well sites, processing facilities, flaring and beef cattle reproduction and calf mortality in western Canada. Preventative Veterinary Medicine, 50:1-17.

Waldner, C.L., C.S. Ribble, E.D. Janzen and J.R. Campbell, 2001b. Associations between total sulfation, hydrogen sulphide deposition and beef cattle breeding outcomes in Western Canada. Preventative Veterinary Medicine, 50:19-33.

Walker, D.A. and K.R. Everett, 1987. Road dust and its environmental impact on Alaskan taiga and tundra. Arctic and Alpine Research, 19:479-489.

Walker, A.H. and L.J. Field, 1991. Subsistence fisheries and the *Exxon Valdez*: human health concerns. In: Proceedings of the 1991 Oil Spill Conference, pp. 441-446. American Petroleum Institute, Report No. 4529.

Walker, D.A. and M.D. Walker, 1991. History and pattern of disturbance in Alaskan arctic terrestrial ecosystems: A hierarchical approach to analysing landscape change. Journal of Applied Ecology, 28:244-276.

Walker, D.A., P.J. Webber, K.R. Everett and J. Brown, 1978. Effects of crude and diesel oil spills on plant communities at Prudhoe Bay, Alaska, and the derivation of oil spill sensitivity maps. Arctic, 31:242-259.

Walker, D.A., D. Cate, J. Brown and C. Racine, 1987a. Disturbance and recovery of arctic Alaskan tundra terrain. A review of recent investigations. Cold Regions Research and Engineering Laboratory. Hanover, New Hampshire. CCREL Report 87-11. 63p

Walker, D.A., P.J. Webber, E.F. Binnian, K.R. Everett, N.D. Lederer, E.A. Nordstrand and M.D. Walker, 1987b. Cumulative impacts of oil fields on northern Alaskan landscapes. Science, 238:757-761.

Walker, T.R., P.D. Crittenden, S.D. Young and T. Prystina, 2006. An assessment of pollution impacts due to the oil and gas industries in the Pechora basin, north-eastern European Russia. Ecological Indicators, 6:369-387.

Wang, Z., M. Fingas, S. Blenkinsopp, G. Sergy, M. Landriault, L. Sigouin and P. Lambert, 1998. Study of the 25-year-old Nipisi oil spill: Persistence of oil residues and comparisons between surface and subsurface sediments. Environmental Science and Technology, 32:2222-2232.

Ward, E.M., C.A. Burnett, A. Ruder and K. Davis-King, 1997. Industries and cancer. Cancer Causes and Control, 8:356-370.

Wardle, C.S., T.J. Carter, G.G. Urquhart, A.D.F. Johnstone, A.M. Ziolkowski, G. Hampton and D. Mackie, 2001. Some effects of seismic air guns on marine fish. Continental Shelf Research, 21:1005-1027.

Warner, R.E., 1969. Environmental effects of oil pollution in Canada. Evaluation of problems and research needs. Brief prepared for the Canadian Wildlife Service. 20p.

Weatherhead, B., A. Tanskanen and A. Stevermer, 2005. Ozone and ultraviolet radiation. In: Arctic Climate Impact Assessment, pp. 151-182. Cambridge University Press.

Webster, L., L. Angus, G. Topping, E.J. Dalgarno and F. Colin, 1997. Long-term monitoring of polycyclic aromatic hydrocarbons in mussels (*Mytilus edulis*) following the *Braer* oil spill. Analyst, 122:1491-1495.

Weideman, A., 1996. Regulation of produced water by the US EPA. In: M. Reed, S. Johnsen (Eds.). Produced Water 2: Environmental Issues and Mitigation Technologies. International Produced Water Symposium, Plenum Press.

Weimerkirsch, H., 2002. Seabird demography and its relationship with the marine environment. In: E.A. Schreiber, J. Burger (Eds.). Biology of Marine Birds, pp. 115-136. CRC Press.

Weinstein, J.E., J.T. Oris and D.H. Taylor, 1997. An ultrastructural examination of the mode of UV-induced toxic action of fluoranthene in the fathead minnow. Aquatic Toxicology, 39:1-22.

Weisman, W.H., 1998. Total petroleum hydrocarbon criteria working group: A risk-based approach for the management of total petroleum hydrocarbons in soil. Journal of Soil Contamination, 7:1-15.

Weller, D.W., Y.V. Ivashchenko, G.A. Tsidulko, A.M. Burdin and R.L. Brownell, Jr., 2002. Influence of seismic surveys on western gray whales off Sakhalin Island, Russia in 2001. Working Paper SC/54/BRG14. International Whaling Commission, Western Gray Whale Working Group Meeting, Ulsan, South Korea, 22-25 October 2002. 12p.

Wellings, S.R., C.E. Alpers, B.B. McCain and B.S. Miller, 1976. Fin erosion disease of starry flounder (*Platichthys stellatus*) and English sole (*Parophrys vetulus*) in the estuary of the Duwamish River, Seattle, Washington. Journal of the Fisheries Research Board of Canada, 33:2577-2586.

Wells, P.G., J.N. Butler and J.S. Hughes (Eds.), 1995. *Exxon Valdez* Oil Spill: Fate and Effects in Alaskan Waters. American Society for Testing and Materials, Philadelphia.

Wertheimer, A.C. and A.G. Celewycz, 1996. Abundance and growth of juvenile pink salmon in oiled and non-oiled locations of western Prince William Sound after the *Exxon Valdez* oil spill. In: S.D. Rice, R.B. Spies, D.A. Wolfe, B.A. Wright (Eds.). Proceedings of the Exxon Valdez Oil Spill Symposium. Symposium 18:518-532. American Fisheries Society.

Westlake, D.W.S., 1982. Microorganisms and the degradation of oil under northern marine conditions. In: J.B. Sprague, J.H. Vandermeulen, P.G. Wells (Eds.). Oil and Dispersants in Canadian Seas – Research Appraisal and Recommendations, pp. 47-53. Environment Canada, Ottawa, EPS 3-EC-82-2.

Westlake, D.W.S. and F.D. Cook, 1975. Biodegradability of Northern Crude Oils. Indian and Northern Affairs. Ottawa, ON Canada. ALUR 74-75-81QS-8048-000-EE-A1. 72p.

Westworth, D.A., 1977. Impact of seismic activity on muskrat populations on the Mackenzie Delta. Indian and Northern Affairs. Ottawa, ON. Supply and Service Catalogue No. R71-19/1. QS-8139-000-EE-A1. 70p.

Whipple, J.A., M.B. Eldridge and P. Benville Jr., 1981. An ecological perspective of the effects of monocylcic aromatic hydrocarbons on fishes. In: F.J. Vernberg, A. Calabrese, F.P. Thurberg, W.B. Vernberg (Eds.). Biological Monitoring of Marine Pollutants, pp. 483-551. Academic Press.

White, P.A. and L.D. Claxton, 2004. Mutagens in contaminated soil: a review. Mutation Research, 567:227-345.

Whitehouse, B.G., 1984. The effects of temperature and salinity on the aqueous solubility of polynuclear aromatic hydrocarbons. Marine Chemistry, 14:319-332.

Whitten, K.R. and R.D. Cameron, 1983. Movements of collared caribou, *Rangifer tarandus*, in relation to petroleum development on the arctic slope of Alaska. Canadian Field-Naturalist, 97:143-146.

Whittle, K.J., D.A. Anderson, P.R. Mackie, C.F. Moffat, N.J. Shepherd and A.H. McVicar, 1997. The impact of the *Braer* oil on caged salmon. In: J.M. Davies, G. Topping (Eds.). The impact of an oil spill in turbulent waters: The *Braer*, pp. 144-160. The Stationery Office Limited, Edinburgh.

Wickstrom, M.L., D. Bechtel, M. Kallas and C. Waldner, 2006a. Effects of oil and gas field emissions on DNA integrity and immune function in domestic and wild mammals. 27th Annual Meeting of the Society of Environmental Toxicology and Contamination. Montréal, QC, Canada.

Wickstrom, M.L., C. Waldner, D. Bechtel and T. Clark, 2006b. Association between exposure to emissions from the oil and gas industry in western Canada and health of neonatal beef calves. 27th Annual Meeting of the Society of Environmental Toxicology and Contamination. Montréal, QC, Canada.

Wiebe, A.H., 1935. The effect of crude oil on freshwater fish. Transactions of the American Fisheries Society, 65:324-331.

Wiens, J.A., 1995. Recovery of seabirds following the *Exxon Valdez* oil spill: An overview. In: P.G. Wells, J.N. Butler, J.S. Hughes (Eds.). *Exxon Valdez* Oil Spill: Fate and Effects in Alaskan Waters, pp. 854-894. American Society for Testing and Materials, Special Technical Publication.

Wiens, J.A., 1996. Oil, seabirds, and science. BioScience, 46:587-597.

Wiens, J.A., T.O. Crist, R.H. Day, S.M. Murphy and G.D. Hayward, 1996. Effects of the *Exxon Valdez* oil spill on marine bird communities in Prince William Sound, Alaska. Ecological Applications, 6:828-841.

Wiens, J.A., T.O. Crist, R.H. Day, S.M. Murphy and G.D. Hayward, 2001. A canonical correspondance analysis of the effects of the *Exxon Valdez* oil spill on marine birds. Ecological Applications, 11:828-839.

Wiese, F.K. and G.J. Robertson, 2004. Assessing seabird mortality from chronic oil discharges at sea. Journal of Wildlife Management, 68:627-638.

Wiese, F.K. and P.C. Ryan, 2003. The extent of chronic marine oil pollution in southeastern Newfoundland waters assessed thorugh beached bird surveys. Marine Pollution Bulletin, 46:1090-1101.

Wiese, F.K., W.A. Montevecchi, G.K. Daroven, F. Huettmann, A.W. Diamond and J. Linke, 2001. Seabirds at risk around offshore oil platforms in the north-east Atlantic. Marine Pollution Bulletin, 42:1285-1290.

Wiese, F.K., G.J. Robertson and A.J. Gaston, 2004. Impacts of chronic marine oil pollution and the murre hunt in Newfoundland on thick-billed murre *Uria lomvia* populations in the eastern Canadian Arctic. Biological Conservation, 116:205-216.

Wilder, D.G., 1970. The tainting of lobster meat by Bunker C oil alone or in combination with the dispersant Corexit. Fisheries Research Board of Canada, Manuscript Report Series, No. 1087, 25p.

Wilkinson, P., B. Thakrar and P. Walls, 1999. Lympho-haematopoietic malignancy around all industrial complexes that include major oil refineries in Great Britain. Occupational and Environmental Medicine, 56:577-580.

Willette, M., 1996. Impacts of the *Exxon Valdez* oil spill on the migration, growth, and survival of juvenile pink salmon in Prince William Sound. In: S.D. Rice, R.B. Spies, D.A. Wolfe, B.A. Wright (eds.). Proceedings of the Exxon Valdez Oil Spill Symposium. Symposium 18:533-550. American Fisheries Society.

Williams, T.M. and D.C. Heard, 1986. World status of wild *Rangifer tarandus* populations. Rangifer, 1:19-28.

Williams, U.P., J.W. Kiceniuk and J.R. Botta, 1985. Polycyclic aromatic hydrocarbon accumulation and sensory evaluation of lobsters (*Homarus americanus*) exposed to diesel oil at Arnold's Cove, Newfoundland. Canadian Technical Report of Fisheries and Aquatic Sciences, 1402. 13p.

Williams, T.M., R.A. Kastelein, R.W. Davis and J.A. Thomas, 1988. The effects of oil contamination and cleaning on sea otters (*Enhydra lutris*). I. Thermoregulatory implications based on pelt studies. Canadian Journal of Zoology, 66:2776-2781.

Wilson, R.D., P.H. Monaghan, A. Osanik, L.C. Price and M.A. Rogers, 1973. Estimate of annual input of petroleum to the marine environment from natural seepage. Gulf Coast Association of Geological Societies, Transactions, 23:182-193.

Wilson, J.A., R.I. Carlson, D.M. Janz, R.L. Lochmiller, J.L. Schroder and N.T. Basta, 2003. Ecotoxicological risks associated with land treatment of petrochemical wastes. III. Immune function and hematology of cotton rats. Journal of Toxicology and Environmental Health A, 66:345-363.

Winters, K., C. Van Baalen and J.A.C. Nicol, 1977. Water soluble extractives from petroleum oils: Chemical characterization and effects on microalgae and marine animals. Rapp. P-v. Reun. Cons. int. Explor. Mer, 171:166-174.

WISSA, 2006. The western Canada study of animal health effects associated with exposure to emissions from oil and natural gas facilities. Western Interprovincial Scientific Studies Association.

Wolfe, D.A., M. Hameedi, J.A. Galt, G. Watabaysahi, J. Short, C. O'Claire, S. Rice, J. Michel, J.R. Payne, J. Braddock, S. Hanna and D. Sale, 1994. The fate of the oil spilled from the *Exxon Valdez*. Environmental Science and Technology, 28:561A-568A.

Wolfe, D.A., M.M. Krahn, E. Casillas, S. Sol, T.A. Thompson, J. Lunz and K.J. Scott, 1996. Toxicity of intertidal and subtidal sediments contaminated by the *Exxon Valdez* oil spill. In: S.D. Rice, R.B. Spies, D.A. Wolfe, B.A. Wright (Eds.). Proceedings of the Exxon Valdez Oil Spill Symposium. Symposium 18:121-139. American Fisheries Society.

Wolkers, J., E.H. Jørgensen, S.M. Nijmeijer and R.F. Witkamp, 1998. Dose and time dependency of Cytochrome P4501A induction in liver and kidney of B(a)P exposed arctic charr (*Salvelinus alpinus*). Marine Environmental Research, 46:117-120.

Woltering, D.M., 1984. The growth response in fish chronic and early life stage toxicity tests: a critical review. Aquatic Toxicology, 5:1-21.

Wong, O., F. Harris and T.J. Smith, 1993. Health effects of gasoline exposure. II. Mortality patterns of distribution workers in the United States. Environmental Health Perspectives, 101(suppl. 6):63-76.

Woodward, D.F., P.M. Mehrle Jr. and W.L. Mauk, 1981. Accumulation and sublethal effects of a Wyoming crude oil in cutthroat trout. Transactions of the American Fisheries Society, 110:437-445.

Woodward, D.F., R.G. Riley and C.E. Smith, 1983. Accumulation, sublethal effects, and safe concentration of a refined oil as evaluated with cutthroat trout. Archives of Environmental Contamination and Toxicology, 12:455-464.

Wooller, R.D., J.S. Bradley and J.P. Croxall, 1992. Long-term population studies of seabirds. Trends in Ecology and Evolution, 7:111-114.

Wraige, E.J., 1997. Studies of the synthesis, environmental occurrence and toxicity of unresolved complex mixtures (UCMs) of hydrocarbons. Ph.D. Thesis. University of Plymouth.

Wright, P.J., R.M. Stagg and A.D. McIntosh, 1997. The impact of the *Braer* oil spill on sandeels around Shetland. In: J.M. Davies, G. Topping (Eds.). The impact of an oil spill in turbulent waters: The *Braer*, pp. 161-181. The Stationery Office Limited, Edinburgh.

Wright, J.F., D.W. Sutcliffe and M.T. Furse, 2000. Assessing the biological quality of fresh waters. Freshwater Biological Association, U.K. 373p.

Würsig, B., 1990. Cetaceans and oil: Ecologic perspectives. In: J.R. Geraci, D.J. St. Aubin (Eds.). Sea Mammals and Oil: Confronting the Risks, pp. 129-165. Academic Press.

Würsig, B.G., D.W. Weller, A.M. Burdin, S.H. Reeve, A.L Bradford, S.A. Blokhin and R.L Brownell Jr., 1999. Gray whales summering off Sakhalin Island, Far East Russia: July-October 1997. A joint U.S.-Russian scientific investigation. Final Report by Texas A&M Univ., College Station, TX, and Kamchatka Institute of Ecology and Nature Management, Russian Acadamy of Sciences, Kamchatka, Russia, for Sakhalin Energy Investment Co. Ltd and Exxon Neftegaz Ltd, Yuzhno-Sakhalinsk, Russia. 101p.

WWF, 2005. Submission to the Joint Review Panel for the Mackenzie Gas Project by the World Wildlife Fund. http://www.ngps.nt.ca/jrp/ (accessed July 2007).

Yang, C-Y., H-F. Chiu, J-F. Chiu, W-Y Kao, S-S. Tsai and S-J. Lan, 1997. Cancer mortality and residence near petrochemical industries in Taiwan. Journal of Toxicology and Environmental Health, 50:265-273.

Yang, C-Y., M.F. Cheng, J-F. Chiu and S-S Tsai, 1999. Female lung cancer and petrochemical air pollution in Taiwan. Archives of Environmental Health, 54:180-185.

Young, T.K. and Bjerregaard, P., 2008. Health Transitions in Arctic Populations. University of Toronto Press, 485p.

Young, D.D., T.R. McCabe, R. Ambrose, G.W. Garner, G.J. Weiler, H.V. Reynolds, M.S. Udevitz, D.J. Reed and B. Griffith, 2002. Predators. In: D.C. Douglas, P.E. Reynolds, E.B. Rhode (Eds.). Arctic Refuge Coastal Plain Terrestrial Wildlife Research Summaries, pp. 51-53. U.S. Geological Survey, Biological Resources Division. USGS/BRD/BSR-2002-0001.

Yunker, M.B. and R.W. Macdonald, 1995. Composition and origins of polycyclic aromatic hydrocarbons in the Mackenzie River and on the Beaufort Sea shelf. Arctic, 48:118-129.

Zbanyszek, R. and L.S. Smith, 1984. The effect of water-soluble aromatic hydrocarbons on some haematological parameters of rainbow trout, *Salmo gairdneri* Richardson, during acute exposure. Journal of Fish Biology, 24:545-552.

Zelick, R., D. Mann and A.N. Popper, 1999. Acoustic communication in fishes and frogs. In: R.R. Fay, A.N. Popper (Eds.). Comparative Hearing: Fish and Amphibians, pp 363-411. Springer Handbook of Auditory Research, Springer-Verlag.

Zhou, T. and K.V. Wong, 1997. Modeling airborne levels of some vola-

tile components in oil spills. Proceedings of the 1997 International Oil Spill Conference, pp. 101-108. Washington, D.C.

Zitko, V, L.E. Burridge, M. Woodside and H. Akagi, 1984. Low contamination of fish by hydrocarbons from the Uniake G-72 (Shell Oil, Vinland) wellsite blowout in February 1984. Canadian Technical Report of Fisheries and Aquatic Sciences, 1305. 43p.

Zobell, C.E., 1971. Sources and biodegradation of carcinogenic hydrocarbons. Proceedings of the Joint Conference on Prevention and Control of Oil Spills, pp. 441-451. American Petroleum Institute.

Zock, J-P, G. Rodriguez-Trigo, F. Pozo-Rodriguez, J.A. Barbera, L. Bouso, Y. Torralba, J.M. Anto, F.P. Gomez, C. Fuster, H. Verea and the SEPAR-*Prestige* Study Group, 2007. Prolonged respiratory symptoms in clean-up workers of the *Prestige* oil spill. American Journal of Respiratory and Critical Care medicine, 176: 610-616.

Zoltai, S.C. and G.P. Kershaw, 1995. Large volume oil spill on land surface: the Vozey Oil Field, Russia. Arctic Marine Oilspill Program (AMOP). Environment Canada. Ottawa, Canada. Vol 18b:1177-1186.

Chapter 7
Scientific Findings and Recommendations

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7.1. Introduction

This assessment was designed to build upon and update the previous AMAP assessments of oil and gas activities conducted in 1997 and 1998 as part of larger assessments of the state of the Arctic environment. The previous assessments looked primarily at the specific issue of hydrocarbon pollution from oil and gas activities and its effects. Since 1998 there has been growing interest in oil and gas in the Arctic. Concomitant with this escalating interest has been the development of new and improved techniques and technology, updated environmental legislation, regulations, and industry practices, and an increased public awareness of the Arctic. The current assessment, by taking a broader view of 'activities,' is more comprehensive providing a history and near-term projection of the oil and gas industry for each Arctic country. It also considers a wider range of effects. In addition to hydrocarbon pollution, this assessment looks at effects of polycyclic aromatic hydrocarbons (PAHs) and other substances, noise, and physical disturbances on individual organisms, populations, habitats, ecosystems and human health. A major addition to this assessment directed by the Arctic Ministers is a consideration of the social and economic effects, and potential effects, of oil and gas activities in the Arctic. The assessment also addresses the vulnerability of Arctic species and ecosystems to oil and gas activities, including accidental oil spills. The findings of this assessment are more thorough than previous assessments but in some cases are still incomplete due to lack of certain information.

For onshore oil and gas activities, the main issue identified is physical impacts, disturbances and habitat fragmentation of the terrestrial environment. Early oil and gas activities caused long-term effects in the terrestrial ecosystem, such as scarring on the tundra due to a general lack of understanding of the sensitivity of the Arctic environment and the slow rates of recovery. Even more recent operations have caused impacts such as changes in drainage, changes in distribution of wildlife populations, and local effects from large oil spills such as the Komi spill in Russia. New technology and methods have significantly reduced damage caused by operations, but these changes are cumulative and as activities expand or overlap, the impact may still be long term and in some cases may even be increasing.

For the marine and freshwater aquatic environments, the main issue of concern is the risk and potentially large impact of accidental oil spills. The *Exxon Valdez* oil spill in Prince William Sound in the Gulf of Alaska happened outside the area considered for this assessment, but did affect sub-Arctic populations of birds, mammals, fish and other organisms and demonstrated the extent of damage that could be caused by a large oil spill in Arctic marine waters. So far, no large oil spills have occurred within the Arctic and sub-Arctic marine area addressed in this assessment. Arctic animal populations are often highly aggregated during breeding, feeding or migration, and an oil spill could potentially affect a large fraction of populations of seabirds, fish and marine mammals. Thus, a large oil spill in ice-covered waters could represent a threat to populations and even to species.

Arctic oil and gas activities have had both positive and negative effects on socio-economic conditions of communities near the activities. The social effects are generally greatest at the local level, while economic effects are often also evident at the regional and national levels. The assessment concludes that adaptive development of management and supervisory systems and evolving advances in technology and best practices have lessened the effects of oil and gas activities. Careful planning, diligent application of rules with necessary control and enforcement, use of best technology and techniques, and continued adaptation to changing conditions may reduce the effects of current and future activities. Even so, the cumulative effects are of growing concern as Arctic activities expand. Evidence shows that accidents will happen and best practices will not always be followed.

The vast majority of the Arctic environment, away from local sources from human populations and activities, is largely pristine with regard to oil hydrocarbons and PAHs. Concentrations are low and close to natural background levels, although these levels are elevated in some areas from natural sources such as oil seeps (e.g., the Mackenzie Valley and Buchan Gulf in Canada) and erosion of coal-containing bedrocks around Svalbard. Even though no regional (large-scale) effects on the environment or clear population level effects on fauna or flora have been documented and no effects on human populations in the Arctic have been substantiated, crucial data are missing. This assessment has been limited by the lack of detailed information on inputs of contaminants from point sources of oil and gas activities such as oil and gas fields, and on the concentrations and gradients in contaminants in the vicinity of such point sources. Thus it has been difficult to assess the degree and areal extent of pollution effects at the local level around such facilities. This has also affected the ability to assess exposures of humans and wildlife populations in areas with onshore oil and gas activities. There has also been a lack of information on the status and trends in animal populations in areas of oil and gas activities and no comprehensive or reliable studies of Arctic populations that may have been exposed to oil and gas pollution.

This assessment distinguishes between effects occurring on local and regional scales. Local is taken to mean the area near point sources of pollution, infrastructure or disturbances from oil and gas activities. This can be oil fields, various facilities, villages, cities, airports, etc. The local scale would typically be smaller than 1000 km^2. Regional is used to mean larger areas such as a whole region of a country, for instance the North Slope region of Alaska, the Mackenzie Delta region, the Yamal region, the Pechora Sea region, etc. The regional scale would typically be of the order of 10 to 100 thousand km^2.

7.2. Main findings

The main findings of the assessment follow. These are presented as short statements with supporting text that frame the issues and provide the main justification for the findings. There are ten main findings, each broken down into a short set of sub-findings. The ten findings cover a range of topics. Finding 1 addresses oil and gas resources and the history and future projections of oil and gas activities in the Arctic. Finding 2 covers socio-economic effects, while Findings 3 and 4 consider the sources and levels of pollution by hydrocarbons and other substances from oil and gas activities. Findings 5, 6 and 7 deal with environmental effects: physical impacts on the terrestrial environment are presented in Finding 5, the potential effects from oil spills in aquatic environments are presented in Finding 6, while toxicological and other effects are presented in Finding 7. Finding 8 addresses effects on human health. The two last findings cover technology and use of best practices (Finding 9) and governance (Finding 10).

Finding 1: Oil and gas activity in the Arctic is likely to increase

F1.A. Hydrocarbon resources in the Arctic are substantial

Arctic production accounts for as much as about 10% and 25% of the world's total oil and gas production respectively. The present data indicate that Russia has produced 80% of all Arctic oil and 99% of all Arctic gas, with lesser production from the United States, Norway and Canada. Many estimates of oil and gas resources from the Arctic exist, and although they vary according to the different methods and criteria used to calculate them, all indicate that a significant percentage of the world's discovered oil and gas reserves and remaining undiscovered oil and gas resources are in the Arctic. One estimate of discovered oil and gas has 5% of the world's oil and 22% of the world's gas in the Arctic. The present data indicate that northern Russia has 75% of known oil reserves and 90% of known gas reserves in the Arctic. Although highly uncertain, some estimates indicate that up to 25% of the world's undiscovered oil and gas resources reside in the Arctic. While these estimates are disputable, the Arctic certainly contains a large amount of undiscovered resources. Russia, Norway and the United States are thought to have the largest amount of undiscovered Arctic oil resources, while Russia, the United States and Canada are thought to have the majority of undiscovered Arctic gas. Offshore, currently available information indicates that Russia, the United States and Norway have the largest undiscovered Arctic resources on their continental shelves. Offshore shelf areas in Canada have had some positive exploration results from drilling in the 1970s and 1980s, while Greenland and the Faroe Islands are in the initial stages of evaluation. These areas should not be discounted as possible major sources of future production.

F1.B. There is a long history of oil and gas exploration and production in the Arctic

Oil seeps were known to indigenous people and the early explorers for a long time. Commercial Arctic oil and gas activities have been occurring onshore for many decades in Russia, the United States, and Canada, first beginning in the 1920s. Early exploration efforts in the 1940s to 1960s used poorly adapted technology and methods and were characterized by an initial lack of understanding concerning the environmental consequences of the activities. Offshore exploration started in the 1970s and early 1980s in all Arctic countries with petroleum provinces. As new techniques were developed, exploration activities both onshore and offshore accelerated, mainly through the 1980s but in some areas into the early 1990s. These exploration activities covered large areas of previously unexplored Arctic lands and seas. Hundreds of thousands of line kilometers of 2-D seismic data were collected and large numbers of exploratory wells were drilled in this period. However, of the discoveries made very few were large enough to justify their development. Since most of the activities did not result in discoveries, large areas of the Arctic were subsequently taken out of consideration for development due to the lack of potential economic resources.

During the 1980s and continuing into the present time, the methodologies employed by much of the Arctic oil and gas industry have changed in two important ways. First, there has been an increase in development drilling in known fields and in smaller accumulations adjacent to existing oil and gas transportation infrastructure. Second, there is an increased reliance on seismic data to evaluate potential geological targets, reducing the required number of wildcat wells to discover oil and gas. New large-scale infrastructure projects are either under construction or in advanced stages of evaluation in Alaska, Canada, Norway, and Russia, indicating that high levels of seismic and drilling activity will continue in many parts of the Arctic for the foreseeable future.

Oil and gas were discovered in the northwestern parts of Russia as early as the 1930s, and have been produced from the northern Timan-Pechora and western Siberian provinces since the 1960s and 1970s. Activity in these areas remains high. Many thousands of kilometers of pipelines have been built to transport oil and gas from northwestern Russia to other regions of the former Soviet Union. Exploration for oil in Alaska began in the 1920s, and in 1967 the large *Prudhoe Bay* oil field was discovered. Following the construction of the 1300-km Trans-Alaska Pipeline in 1977, oil production and large-scale infrastructure development began and is still ongoing in northern Alaska. Canadian Arctic exploration efforts resulted in the discovery of oil in the Norman Wells region from which seasonal production began in the 1920s. Production expanded in the 1980s resulting in the construction of a 900-km pipeline south to Alberta. Oil was discovered in the Arctic Islands at Bent Horn in 1976 and small amounts of oil were produced and shipped by tanker from 1985 until 1997. Significant amounts of gas and some oil were discovered after an intense exploration effort in the Mackenzie Delta and Beaufort Sea region, with smaller discoveries of oil or gas in other parts of the Canadian Arctic. The Mackenzie Valley pipeline, which is being proposed to ship gas from the Mackenzie Delta south to Alberta, is currently undergoing environmental assessment and regulatory review.

Norway's Arctic exploration efforts, which began in the early 1980s, led to production from the *Draugen* field in the Norwegian Sea in 1993 and then from several other fields. The *Snøhvit* gas field in the Barents Sea is under development with production scheduled for 2007. Very high levels of exploratory activity are currently being conducted in the Norwegian part of the Barents Sea. Exploration seismic and drilling efforts have also taken place and are ongoing in Greenland and the Faroe Islands and seismic exploration has taken place offshore of Iceland. While exploration has not yet yielded oil or gas discoveries in these areas, and is being conducted at relatively modest levels, efforts to date have produced results warranting further evaluation.

F1.C. Levels of oil and gas activities in the Arctic are affected by many factors

Many factors ultimately control whether and when Arctic oil and gas development activities will take place. These include international political factors such as energy security for developed countries and demand for energy from emerging economies. Other factors include resource potential and the chemical, geological and physiographic nature of the deposit; long-term trends in oil and gas prices; legal, regulatory, and economic controls; lands made available for activities; environmental, political and economic risk; technological development; and capacity of existing infrastructure or development of new supporting infrastructure.

Operating costs of activities in the Arctic must account for harsh and challenging working conditions such as limited or non-existent infrastructure, low temperatures, seasonal darkness, permafrost, sea ice, changing climate, and high transportation costs, as well as increasingly complex regulatory controls to protect the environment and people living and working in the Arctic.

The lead time from discovery to development is usually equal to or longer than that for other parts of the world. A dedicated program for onshore development may take ten years or more between discovery and production. Offshore development and development in smaller, more remote and/or more environmentally sensitive areas onshore, may take 15 to 30 years to develop – or may never be developed.

F1.D. Oil and gas activities are likely to expand into new areas

Areas thought to have high resource potential, whether previously explored or unexplored, are being considered for more focused exploration activities. Throughout the Arctic, areas are being made available for exploration licensing and leasing. If development results, it will lead to increased capital investment and expanded infrastructure.

Plans are in place for near-term (<10 years) and mid-term (10–15 years) future development and further exploration for oil and gas in the Arctic. In Russia, oil and gas production activities will grow in the northern Timan-Pechora and West Siberia provinces and in the Kara and Barents seas. This development is likely to include the construction of a major oil pipeline for Arctic oil transport to the Pacific Rim, and several new marine terminals and subsequent Arctic tanker traffic to markets, including Arctic tanker routes. In Alaska, oil production will continue in the Arctic Alaska province on the North Slope and Federal onshore and offshore lands and may include gas production if a major pipeline is constructed to transport gas to the lower 48 states. In Canada, expansion of oil and gas exploration is likely to occur in onshore and offshore areas including the Mackenzie Delta, and gas production will increase with the construction of the Mackenzie Valley gas pipeline. Norway is planning continued exploration and development activities in the Norwegian and Barents seas with associated offshore pipeline and tanker transport.

In the mid- and far-term (15–25 years) exploration is likely to continue and extend into new offshore Arctic shelf areas, and onshore exploration activities around existing fields and in new areas are likely to take place in Alaska, Canada, Greenland, eastern Siberia and northeastern Russia. Development as a result of these activities is, however, unlikely to occur within the mid- to far term due to the typically long lead time between exploration and development. New pipelines and marine terminals are likely.

On the horizon (>25 years), it is possible that unconventional oil and gas resources may be developed in Arctic areas. These deposits include viscous or 'heavy' oil, coal-bed methane, and potentially vast methane hydrate deposits both onshore and offshore.

F1.E. Arctic oil and gas transportation systems will expand

Existing transportation infrastructure for oil and gas in the Arctic includes pipelines, tankers, vehicles, and railcars. In Russia, transportation of oil and gas to refineries and users is accomplished by a combination of pipelines, coastal barges, shuttle tankers, large tankers, supertankers, railcars, and trucks. By some estimates, Russia's pipeline system comprises approximately 150 000 km of gas- and 50 000 km of oil-product lines and a significant number of kilometers of oil and gas collection and gas distribution lines, but as yet there are minimal trunk lines in the Arctic. There has been an increase in the volume of oil transported by tankers along the Norwegian coast from Russia. In 2002 the volume was 4.7 million m^3; in 2004, 14 million m^3; in 2005, 11 million m^3; and over 12 million m^3 in 2006. By some estimates, Russia may have the capacity to ship more than 46.6 million m^3 of oil per year by 2010 and over 115 million m^3 of oil per year by 2015. In the United States, oil is transported from the North Slope of Alaska by the 1300-km Trans-Alaska Pipeline to southern Alaska and transferred to tankers for export. In Canada, oil from the *Norman Wells* field is transported by pipeline 900 km south to Alberta and then on to southern markets. In Arctic offshore Norway, oil is transported to shore by tankers and gas is transported by subsea pipelines to the mainland. In Alaska and Russia, many of the Arctic pipelines are at or near their operational life expectancy.

Several major pipeline projects are planned in the near- to mid-term in the Arctic. In Russia, many new pipelines are being built to augment an aging system. Two major projects being planned are the Eastern Siberia-Pacific Ocean pipeline system (ESPO) that will carry oil from the eastern Siberian oil fields to the Pacific coast of Russia for regional export. The other major project, now on indefinite hold, is an oil pipeline thousands of kilometers long from the fields in Timan-Pechora and western Siberia to an Arctic port in the Murmansk area, where oil will be shipped by tanker to Europe and the United States. There are new projects underway to expand port capacities or to construct new ports for loading of tankers in the Russian North. Should Canada approve the 1200-km Mackenzie Valley gas pipeline, this will allow the first production from gas fields of the Mackenzie Delta and the Central Mackenzie, with eventual development of Beaufort Sea discoveries. The pipeline will connect to existing pipeline systems in southern Canada. The Alaska natural gas pipeline is likely to be built, connecting the Alaska North Slope with pipeline transport to Canada and the United States mainland, allowing gas to be commercially produced for the first time in northern Alaska. Depending on the final route, this pipeline could be 2600 to 3400 km long.

In areas of new discoveries where infrastructure does not exist, such as the Chukchi or East Siberian seas, new transportation infrastructure will need to be built.

Finding 2: Oil and gas activities are major drivers of social and economic change

Oil and gas activities are a catalyst for the growth of the regional and community market economy and infrastructure, and frequently also for the introduction of new decision-

making systems and values. But there are many other concurrent causes of social and economic change in the Arctic, and it remains nearly impossible to separate the role of oil and gas activities or their proportional contribution to specific effects. Every region is unique in its particular resources, geographic context, political and economic institutions, culture and history, and stage of oil and gas development, so the opportunity to generalize is limited.

F2.A. Social and economic effects of oil and gas activities are mitigated by the planning, regulatory and allocation functions of governments

Effective governance includes the ability to plan for and respond to societal impacts of development, strong environmental regulation and supervisory responsibilities, public involvement in decision-making, and a pragmatic working relationship among industry, government, and the public. The distribution of power is a major factor in shaping the degree to which unevenness creates tensions and negative effects. When local organizations and institutions lack power, local interests are likely to be neglected, so that costs are borne disproportionately by local residents while benefits accrue primarily at the regional and national levels.

The initial presence of and prospects of additional activities by the oil and gas industry have heightened the desire of local governments and regional indigenous groups to be involved in the regulation and monitoring of industrial activities, to receive sufficient funding to cope with increased program and service workloads, and, in some areas, to share in the wealth created. In Alaska, the North Slope Borough was created. In Canada, land claim settlements introducing co-management boards and self-governance were established and land claims continue to be settled. One feature of local participation is the use of local boards that recognize the value of both local and traditional knowledge in the decision-making process. In Canada, the federal government undertakes consultations on oil and gas activities to identify decisions or actions that could infringe on indigenous peoples' rights under the Canadian constitution and, wherever possible, to accommodate the concerns expressed by indigenous peoples. In Russia, the degree of local involvement in governance has typically been lower, firstly due to the centralized structure of the Soviet State and later due to the political and economic upheaval at the start of the post-Soviet era. More recently, local groups such as *Yerv* and *Yamal Potomkam!* have begun organizing themselves, in part to gain a greater share of the benefits of oil and gas activity while reducing negative socio-economic and environmental impacts. In Norway, and in Greenland's approach to planning so far, governance regarding oil and gas has been concentrated at the national level, emphasizing the retention of earnings by government to be used for the common good. Norway and Alaska have both used oil and gas revenues to establish trust funds for long-term benefit.

F2.B. Socio-economic effects vary according to the scale and 'life-cycle' stage of oil and gas activity

'Life-cycle' stages of the oil and gas industry vary in scale and area, ranging from a particular field or prospect, to a larger development area, to an entire region or country. Many Arctic regions are at early stages in the oil and gas 'life cycle' and are experiencing the initial effects of large development projects. Social and economic effects tend to increase and be more local at the exploration and construction phase, then to stabilize and be more regional in the production stage.

The remote and technologically intensive nature of Arctic development has focused industry on larger reserves to finance the high capital costs of bringing the resource to market. This intensity increases the social and economic effects at the various stages. Construction will bring local employment, business and market economy effects. For example, employment is highest during the construction stage, which typically entails employing available local labour and bringing large numbers of workers to the region. If workers are highly mobile, they can re-locate to the site of the next large construction project. If workers are not mobile, their employment opportunity is short-lived. Also during construction, employment normally shows seasonally high levels of transient labour.

By contrast, public revenues often come from royalties or taxes, and thus are more evenly spread through the production stage. During this stage, production and revenues peak and begin a long decline, although revenues are linked to oil prices and so are highly variable. While production is high, employment is relatively low and steady. Production jobs are high skill, high pay, and workers tend to be non-local.

F2.C. Indigenous Arctic people are becoming more involved and more affected as oil and gas development expands in the Arctic

Indigenous people are becoming more active participants in oil and gas activity in the Arctic, as decision-makers, owners, employees and community service providers. Land claim settlements in Alaska and northern Canada have resulted in indigenous people becoming private land-owners, co-management participants and business participants in the industry.

Much of the Arctic land areas, and all of the continental shelves, are owned by and managed by national or regional governments. Private ownership of land in the north is less prevalent than in the southern zones of the same countries. In North America, most of the privately owned land belongs to indigenous corporations established by land claims agreements, meaning that it is owned in common by the indigenous inhabitants. Some of these agreements resulted from the desire of governments to settle indigenous claims in order to make areas available for oil and gas activities and pipeline rights-of-way.

The interaction between the oil and gas industry and indigenous people is facilitating social and economic change. Protecting cultural heritage is a high priority throughout the Arctic, from local initiatives to national legislation and international conventions. A crucial part of indigenous cultures is connection to place, increasing their vulnerability to dislocation by industrial and other activities that can separate them from their lands. At the same time, Arctic indigenous peoples have developed great flexibility to deal with the inherent variability of the Arctic environment, increasing their resilience to change.

The Arctic countries are now largely market economies, with varying degrees of state intervention in their markets. In regions of Greenland (Denmark), North America and Russia, elements of subsistence economies still exist. Wages and cash connect indigenous people to the modern market economy, but at the same time acquiring food from the land and sea and sharing or bartering of foods and other goods and services provide a major part of households' production and consumption. This non-market sector mitigates the high cost of living and the limited array of consumer goods in remote areas, buffers the volatility in the wage economy, and

maintains cultural identity and social capital in Northern communities.

Industry expansion across the Arctic has increased the overlap between traditional use of the land and oil and gas activity. Techniques are being employed to use traditional knowledge in project planning, environmental assessments and regulatory decision-making.

F2.D. The economic value of oil and gas activities plays a significant role in national, regional and local-level effects

Oil and gas activities can be harnessed to stimulate broader economic growth and to increase financial capital that provides lasting benefits. These activities are likely to form the largest sector of the Arctic Gross Domestic Product (GDP). The value of the oil and gas activities is also a factor in regional and national policy-making. For example, the decline in oil production in Alaska is highlighting the need to diversify and plan for other sources of revenue. The value of Norwegian production has facilitated national policies to distribute benefits and invest trust revenues. Oil and gas activities in northern Canada have been limited to several substantial exploration booms and a limited amount of oil and gas production. Construction of the proposed Mackenzie Valley pipeline can be considered as a basin-opening project that would lead to a new round of exploration, development and production.

Oil and gas revenues have brought wealth and associated improvements in public health, education, and other services to a generation of Arctic residents in some regions. However, in other regions, development has had adverse effects on the environment and disrupted local social and cultural systems, leaving a legacy of negative effects that reduce the potential for sustainability.

Oil and gas revenues can also support the retention of cultural practices. Harnessing these revenues to create economic stimulus and overall gains in employment and services may offer longer-term benefits than aiming for jobs in industry. In some instances revenues from oil and gas may directly benefit the national government but often not the local government. In contrast, the social effects are primarily local.

Finding 3: Contamination from oil and gas activities in the Arctic is relatively minor compared to inputs from natural seepages

F3.A. Natural oil seeps comprise the majority of the total input of oil hydrocarbons to the Arctic environment

Hydrocarbons normally found in petroleum have been detected in all compartments of the Arctic environment. However, many hydrocarbon compounds can be produced by processes other than those leading to the formation of petroleum (petrogenesis). Hydrocarbons in the environment are formed through four distinct processes: pyrogenesis (combustion), diagenesis (natural degradation in sediments), biogenesis (biological production) or petrogenesis. Each process produces a characteristic profile of individual compounds although there is considerable overlap. A selection of 20 to 30 parent and alkylated polycyclic aromatic hydrocarbons provide an excellent diagnostic tool for identifying petrogenic contributions. These compounds have been used as a surrogate for establishing the presence of petrogenic hydrocarbons. In order to quantify the relative contribution from the different sources, it is necessary to have datasets that quantify individual parent and alkylated compounds. Such information is limited for the Arctic. It is therefore difficult to quantify the petrogenic contribution on an Arctic-wide scale with certainty. However, some datasets are available and suitable for quantifying petrogenic hydrocarbons in some regions.

This assessment has attempted to estimate the relative magnitude and importance of different sources of petroleum hydrocarbons and PAHs influencing the Arctic environment. On an Arctic-wide basis and using incomplete quantification of individual contributors to each input pathway, it has been estimated that natural oil seeps are contributing most to the total input of oil hydrocarbons. Oil spills may be the second largest source, greater than non- oil and gas related industrial activities, atmospheric deposition, and activities related to oil and gas excluding spills.

F3.B. Several sources of emissions and discharges of petroleum hydrocarbons and related contaminants exist in the Arctic, and result in local pollution in some areas

Important anthropogenic sources of oil hydrocarbons and petrogenic PAHs in the Arctic include the general use of refined petroleum products, oil and gas production, and loss in transportation via pipelines, railways, and ships. Other sources include shipping accidents, runoff from land, rivers, industrial activities (e.g., oil and gas terminals, refineries, smelters, and mines), and routine discharges from ships and fishing vessels. Inputs of oil hydrocarbons to the Arctic also derive from long-range transport from the heavily industrialized northern hemisphere, especially pyrogenic PAHs which result from the combustion of almost any carbon-based fuel. Long-range transport to the Arctic takes place through the atmosphere, by river flows and by ocean currents. The atmospheric transport pathway contributes only small amounts of petroleum hydrocarbons, but significant amounts of pyrolytic PAHs. On land most of the hydrocarbon contaminants will be contained within the immediate vicinity of the discharge or spill. Atmospheric emissions can affect much larger areas.

Several river basins in Canada and Russia are petroleum enriched. These north-flowing rivers carry oil hydrocarbons to the adjacent seas. Currently, the source for most of this transport is natural. These rivers may also contribute oil hydrocarbons to the northern seas as a result of shipping accidents, blowouts, pipeline ruptures, runoff from towns, cities and industrial facilities, and from long-range atmospheric transport to snow within the catchment area.

F3.C. Arctic oil and gas activities are currently a minor source of oil hydrocarbons and PAHs on a regional scale, but can be important locally

On a regional scale, the current discharges and emissions associated with the oil and gas industry are estimated to contribute a relatively small fraction of the total input of oil hydrocarbons. This could rise to a much higher fraction when oil and gas production in the Arctic peaks. In the past, oil and gas activities have led to pollution on a local scale, sometimes severe pollution. An example is the Komi oil spill in 1994, which actually comprised several minor spills from a pipeline over a period of several months. An estimated 100 000 tonnes were released that polluted 280 hectares (2.8 km^2) of marshland/freshwater. An evaluation of the extent and importance of local pollution around point sources from oil and gas activities has generally not been possible due to a lack of detailed information. Waste disposal methods and procedures employed by the oil and gas industry are central

issues in evaluating the effects associated with the industry. In some areas, sumps were used to store process water and drilling fluids and were sealed and abandoned after drilling ceased. Studies have shown that a number of these sumps are leaking and more may lose their integrity as climate change causes erosion of the permafrost.

Oil spills information from the North Slope of Alaska for 1995 to 2002 indicates that around half the spills were from the petroleum industry in the area. Transport and spills associated with the general use of petroleum were the source of the remainder of spills. Similar sources of spills are likely in Russia and other Arctic regions.

Modern technology and improved practices have raised expectations for dramatic improvements in Arctic land-based and offshore discharges and emissions. All Arctic countries have stopped discharges of oil-based drilling mud. Most countries now use water-based drilling fluids, and synthetic-based muds have replaced oil-based muds in most cases where such fluids are necessary. Spent muds and cuttings are disposed of in approved disposal sites onshore or re-injected into approved underground reservoirs. Onshore, treated produced water is still discharged to surface waters and land in Russia, but re-injection of produced water and wastes is becoming standard practice in the Arctic. Most countries are moving toward re-injecting produced water from offshore production. The use of 'environmentally-friendly' chemicals is being encouraged. There is continuous improvement in waste handling procedures. Improved technology, more stringent standards, and heightened awareness of the need to reduce emissions have resulted in significant environmental benefits.

F3.D. Oil and gas exploration, production and transportation have the greatest potential for large-scale accidental or long-term releases of contamination to land and sea

Drilling of oil and gas wells is much more extensive in the development and production phase than during exploration, while the types of emissions and discharges are similar. Specific to the production phase are emissions from flaring, venting and production testing, and the potential releases of production chemicals, contaminated production water and wastes from drilling. The transport phase of activity has the greatest potential to release oil and hazardous substances into the environment. Tankers and pipelines are the main potential sources of spills.

Finding 4: Levels of oil hydrocarbons and PAHs in the Arctic environment are generally low, except in some local areas

F4.A. Low background concentrations of oil hydrocarbons and PAHs occur in the Arctic marine environment

Levels of oil hydrocarbons are generally low in the Arctic marine environment, and fall within ranges normally considered to be background. For all sea areas, levels in sediments are generally well studied. Less information is available on levels in sea water and marine biota.

In the northern North Atlantic Ocean, Barents Sea, Russian northern seas, sea areas around Alaska and the Queen Elizabeth Islands in Arctic Canada, concentrations of oil hydrocarbons and PAHs in sediments are low, and levels found can usually be attributed to natural sources. In the Eurasian northern seas the highest background levels of petroleum hydrocarbons and PAHs are found around the Svalbard archipelago. The main source of the PAHs is probably the erosion and weathering of coal-rich sediments, while marine oil seeps make a smaller contribution. Elevated baseline levels also occur in sediments in the vicinity of the Mackenzie Delta and Beaufort Sea Shelf, mostly due to inputs from natural seeps along the Mackenzie River and offshore. Elevated concentrations are found in some coastal areas such as Buchan Gulf (Canada) and river estuaries of Russia. The sources to the latter are both natural and anthropogenic.

Natural petroleum seeps from land and rivers and natural marine seeps are important contributors to the background levels of oil hydrocarbons and PAHs found in large parts of the Arctic marine environment. Offshore petroleum industry in the Arctic is limited, and has only a local impact close to the oil and gas production fields.

F4.B. Information on oil hydrocarbons and PAHs on land and in freshwater systems is more limited than for the marine environment, but indicates low levels in areas distant from human activities

The assessment of oil hydrocarbons and PAHs in freshwater systems and on land is not as good as for the marine environment, due to limited information available for this assessment. Only Russia has conducted extensive systematic, long-term monitoring of petroleum in the terrestrial and freshwater systems of the Arctic. However, the information available from these studies has been condensed to the point that it is difficult to identify contamination hot spots or to establish concentration gradients. Another factor limiting data comparability and the value of the data for establishing trends is the use of several analytical methods for which quality assurance data are insufficient or lacking entirely. For Canada and Alaska, information from a limited number of studies was used. For Greenland, Iceland, the Faroe Islands and Norway, very few studies have been carried out on land. For most Arctic countries the measurement of petroleum hydrocarbons and PAHs in biota has not been a high priority in most monitoring programs because concentrations in biota are usually low and input to the environment is assumed to be low. PAHs are also readily metabolized by many organisms and thus do not reflect exposure history through accumulation. An important exception is that PAHs are measured in biota after oil spill events in order to document the level of bioaccumulation and exposure of the biota.

In areas remote from human populations the concentrations of oil hydrocarbons and PAHs in ice, freshwater, soil and biota generally fall within the lower end of the global range.

F4.C. Elevated concentrations of oil hydrocarbons and PAHs are usually found close to industrial and urban communities in the Arctic

Areas where higher concentrations of oil hydrocarbons and PAHs are found are usually associated with point sources of human activities. In the oil and gas fields of northwestern Russia there are reports of many cases of local contamination from accidental spills and discharges, overloading of treatment facilities, and leakage from waste deposits such as sludge pits. Soils in some areas have accumulated large amounts of oil (up to 10% by weight) that will last for many decades under conditions of natural degradation, representing long-term pollution. In offshore oil fields on the Norwegian shelf to the Norwegian Sea, discharges of drill cuttings have contaminated

the sediments in near-zones around the discharge points, totalling about 80 km^2 of bottom habitat for all the fields combined. Monitoring has shown disturbance of the benthic fauna over an area of about 14 km^2 and that the area is decreasing due to natural recovery. Monitoring of abandoned onshore drill sites on the North Slope of Alaska has shown some local contamination but generally of limited extent.

The data sets available for this assessment have, with the exception of the Norwegian offshore fields, generally not been detailed enough to establish the areal extent of local pollution or to establish quantitative relationships between sources and concentrations in the environment.

Finding 5: Physical impacts, disturbances and habitat fragmentation are the main issues for terrestrial environments

F5.A. Oil and gas activities leave a physical 'footprint' on land

Exploration and development of oil and gas fields on land leave a physical 'footprint' in the terrestrial Arctic environment. Infrastructure in the form of gravel pads, buildings, waste sumps, roads, airstrips and pipelines transforms the areas occupied. Gravel extraction for the construction of pads and roads, for example, may leave physical scars or disturb stream habitats. Oil and gas infrastructure causes direct physical impacts on the Arctic tundra and other habitats.

More diffuse physical near-zone impacts are also associated with the infrastructure. Dust from roads may affect the physical conditions and vegetation out to a few hundred meters. Roads and other constructions may also influence the hydrology of flat tundra landscapes. Pipelines and roads may impede migrations of animals, and traffic and human presence may cause avoidance in some species, while others may be attracted. Avoidance effects may extend out several kilometers. There are also positive effects from roads, gravel pads and structures as they can provide relief from insect harassment for caribou/reindeer. Pregnant female caribou/reindeer, and those with calves, appear to avoid structures and human activity to a greater extent than other caribou/reindeer. The long-term effect of these changes on caribou/reindeer, and other species that might show similar behaviours, are complex and unclear and require more research in the light of possible expanding oil and gas activity.

Networks of roads, pipelines, settlements and human presence contribute to habitat fragmentation that can affect wildlife. Oil and gas activities and development are but one sector of several contributors to habitat fragmentation. The spatial configuration of infrastructure and disturbances is an important aspect for the degree of environmental impact. Roads, pipelines and clearings for seismic transects are linear configurations that can have disproportionately large effects either by impeding migration for instance for caribou/reindeer, or by providing migration corridors for predators like wolves. Design of the infrastructure (e.g., elevated or buried pipelines) and traffic control may significantly reduce the environmental impacts. On a larger scale, removal of oil and gas from geological formations may lead to the subsidence of the ground surface and this may affect wetland habitat and the nesting of shorebirds and waterfowl.

F5.B. The physical impact of past activities has affected varying proportions of tundra environments

Tundra environments are fragile and vehicles leave tracks that remain for long periods. Summer travel on thawed ground, although a practice abandoned by most countries, is particularly damaging. Varying proportions of tundra environments have been damaged by tundra travel and construction of infrastructure related to oil and gas exploration and development. On the North Slope of Alaska, infrastructure and gravel mines occupy an area of about 100 km^2, which is less than 0.1% of the area of the coastal plain tundra. Tundra travel may have impacted an area of similar extent. Larger areas are affected in the oil and gas regions of northern Russia, reflecting the more extensive activities there. In the Yamalo-Nenets Autonomous Okrug, the total area of disturbed land is over 1500 km^2, constituting about 0.13% of the okrug area. On the Taz Peninsula, more than 6000 km^2 of tundra have been disturbed, equivalent to about 1.5% of the total area. In the Arctic, the rate of recovery of disturbed terrestrial systems is very slow due to the severe climate and low availability of nutrients.

F5.C. Modern oil and gas activities leave smaller physical 'footprints' than corresponding activities in the past

Improvements in drilling technology allow fewer and more compact drilling pads. This reduces the area occupied by infrastructure, the amount of gravel needed, and the scale of construction. Tundra travel for seismic activities and other purposes occurs on frozen ground in winter with specially designed vehicles that leave little or no impact on the vegetation and the ground. Movement of equipment and supplies on ice roads in winter reduces the need for permanent roads that may disturb wildlife in the summer period.

The physical 'footprint' from infrastructure and travel is now substantially smaller using current technology and best practices than it was during the earlier period of oil and gas activities. Real-time monitoring of wildlife combined with traffic control and other mitigating measures has also substantially reduced the disturbance from oil and gas activities in some areas. Although new oil and gas activities leave a smaller footprint, this is sometimes additional to impacts from earlier activities. The cumulative effects from past and present activities may continue to increase, albeit at a slower rate than in previous years.

F5.D. Oil spills on land have limited spatial extent compared to oil spills at sea but may have long-lasting impact

Oil spills on land are a potential source of impacts to the Arctic environment. Old mixed-product pipelines or gathering lines are prone to leak from corrosion, and substantial amounts of oil and other substances have been spilled in terrestrial environments. Even low-pressure product pipelines thought to be less prone to leak, have shown substantial corrosion as at Prudhoe Bay, Alaska in August 2006. There are reports that in the Russian north, substantial amounts of oil have been lost from pipelines during transport. The Komi oil spill in Usinsk in 1994 drew large attention and spread into the freshwater system of the Pechora River. Despite the large amount of oil spilled (>100 000 tonnes), the environmental impacts appeared to be limited to a moderately restricted area. About 7 km^2 of land was designated as severely impacted by oil and an area perhaps ten times as large was impacted to some degree.

In contrast to the open marine environment where spilled oil can spread and affect large areas and long coastlines, terrestrial oil spills are less mobile. The soil of the active layer above the permafrost may become saturated and relatively small areas may hold large amounts of spilled oil. The oil may resist bacterial decomposition because of low levels of oxygen and retain its toxic components for decades. The oil can get into freshwater systems, as did the Komi oil spill, but the spread is more limited than for marine spills. Terrestrial spills are amendable for containment, recovery, and remediation actions. For these reasons, oil spills on land are of less environmental concern than marine oil spills.

This is not to say that terrestrial oil spills are unimportant. Large amounts of oil have been spilled and persist in the Arctic, contaminating considerable areas of tundra and wetland environments. This represents a threat to wildlife and a risk of contaminated food and water and of health impacts on humans in the affected areas. Also, the clean-up response to terrestrial oil spills can have physical effects on wildlife, terrestrial environments, archaeological resources, human health, and local socio-economics. However, the occurrence of natural oil seeps in most areas of terrestrial Arctic oil operations is a far greater source of oil than that spilled. There is insufficient information to judge what impacts may be associated with these seeps.

Finding 6: Oil spills have the greatest potential to impact aquatic environments

F6.A. Small oil spills are relatively frequent while large spill events are rare

Spills are inevitably associated with exploration, production, transportation, storage and use of oil, gas and/or their refined products. The frequency of spills decreases with increasing size of the spill, and small spills (a few to some hundred liters) occur relatively frequently while large spills are quite rare. This is illustrated by global spill statistics, which predict a frequency of about 2 spills greater than 10 000 tonnes annually from tanker accidents and less than 0.1 spills from blowout. In the Environmental Impact Statement for the Alaskan Beaufort Sea, anticipated annual occurrence rates of spills larger than 160 m^3 (1000 barrels) were 0.13 and 0.10 spills standardized per billion barrels (160 million m^3) of oil produced, for spills from platforms and pipelines, respectively.

Oil spill statistics indicate the probabilities of spills but not when and where they will occur. This is addressed by risk assessments. There have been no major oil spills in ice-covered waters within the area of the current assessment since spills resulting from actions during the Second World War. This is partly due to the still limited extent of offshore operations in the Arctic, but also to better supervisory and operational practices. Globally, the frequency of oil spills from tankers, platforms and pipelines has been declining, reflecting improvements in technology and operational performances. This no doubt also benefits operations in the Arctic. At the same time, the harsh environmental conditions and the presence of sea ice make operations in the Arctic inherently more risky than at more southerly latitudes.

F6.B. Seabirds and fur-bearing marine mammals are vulnerable to oiling

Seabirds and some marine mammals are vulnerable to oil on the sea surface. Their sensitivity results from the physical coating of feathers or fur with oil. This causes a loss of thermal insulating properties and body heat, leading to hypothermia and often death. This is particularly serious for organisms living in cold environments. Seabirds that spend most of their time swimming or diving are more vulnerable than those spending most of their time airborne or snatching food from the surface. Arctic seabirds generally have long-range spring and autumn migrations, are colonial or semi-colonial, and have a slow reproductive capacity with delayed maturity of adults, low fecundity and high adult survival, characteristics which make some species particularly sensitive to oiling. Physical oiling was the mechanism responsible for the high acute mortality of seabirds and sea otters (*Enhydra lutris*) in Prince William Sound following the *Exxon Valdez* spill, an event that occurred outside the geographical area for the current assessment. In this case, for several seaduck populations, the high mortality caused by physical oiling was followed by effects that lingered for more than a decade after the spill. Harlequin ducks feed in the intertidal zone, and at oiled coasts they had lower survival and lower body mass and showed a decline in densities compared to stable numbers on shores that had not been oiled. The effect was caused by long-term contamination from persistent subsurface reservoirs of unweathered spilled oil in the intertidal zone.

Seabird eggs are particularly sensitive to oiling, with minute amounts of some oils causing death or mutation of developing embryos. Studies have shown that oil transported back to the nest on the breast feathers of adult birds is sufficient to cause mortality in incubated eggs. Hence, any release of oil to the Arctic marine environment from oil and gas activities could cause death to adult birds from direct oiling or mortality in incubating eggs. The ingestion of oil by seabirds, whether from preening of oil-coated feathers or ingestion of contaminated food items, can lead to changes in reproductive hormones and the immune system. These changes could ultimately reduce the rate of reproduction. Also, fur-bearing animals, including polar bears (*Ursus maritimus*), can increase the dosage they receive by ingesting oil when attempting to clean fur by licking it.

Marine oil spills have the potential to cause decline in seabird populations, and single seabird colonies may be deserted. Some species that concentrate in a few areas can suffer a greater population decline after a single spill than more dispersed populations. Seabirds are slow- reproducing species with long population recovery times. Populations can have some resilience to catastrophic high mortality because there can be pools of non-breeding birds that become active breeders when the breeding population is lost through natural or man-made causes. Hence populations of some species recover more rapidly than others owing to natural mechanisms of response to catastrophic events. However, not all species have such characteristics and a major oil spill would undoubtedly affect some species for very long periods of time.

Seals, in particular young pups, are also vulnerable to oiling. Harp seals (*Phoca groenlandica*) congregate in the Gulf of St. Lawrence to bear their pups on the ice each spring. A spill of about 600 m^3 of Bunker C oil occurred there in 1969, and it was reported that several thousand adult seals and pups were oiled and an unknown number killed. Although the Gulf of St. Lawrence lies outside the area of the current assessment, it is ice-covered in winter, and the impact of such a spill is relevant to the Arctic situation. Whelping areas, where large numbers of seals aggregate annually on the ice, are found in the Davies Strait region (hooded seals, *Cystophora cristata*), in the southern Greenland Sea/ Jan Mayen area (hooded and harp seals), and at the entrance to the White Sea (harp seals). Whelping areas for spotted

(*Phoca largha*) and ribbon (*Phoca fasciata*) seals are found in the southern part of the annual pack ice in the Bering Sea.

F6.C. Whales have low vulnerability to oiling in general but their vulnerability could be higher in ice-covered waters

Whales depend on their layer of blubber and not on pelage for thermal insulation, and hence physical oiling is less acutely harmful. This is also the case for many Arctic seals and the walrus (*Odobenus rosmarus*). Whale skin is very tough and tolerant to oil, although mucous membranes of the eyes and respiratory pathways may be irritated and damaged. Inhalation of vapours from a fresh spill could potentially harm whales that do not escape from the scene of a spill.

There is limited evidence for effects of oil on whales in spill situations. Observations of killer whales (*Orcinus orca*) in Prince William Sound suggested that there were effects on the social behaviour of at least one group of individuals and fifteen killer whales disappeared between autumn 1988 and summer 1990. These whales were so well known that they were recognizable as individuals, and this loss represented an unprecedented change for these animals. Fouling of baleen may represent a special threat to the large baleen whales. Bowhead whales (*Balaena mysticetus*) may skim feed on Arctic zooplankton in the surface layer in leads and openings in the ice. This species may be more vulnerable should oil concentrate in this special environment.

Bowhead, beluga (*Delphinapterus leucas*), narwhal (*Monodon monoceros*), and walrus have seasonal migrations from wintering areas in the southern parts of the sea-ice distribution in winter into the high Arctic in summer. The spring migrations take place early before the general ice break-up and follow systems of leads and polynyas in the ice. This is also the time when these migrating mammals give birth to their calves. Owing to the confined nature of their migration habitats where they depend on the openings in the ice for breathing, they could potentially be sensitive to oiling in this situation.

F6.D. Even small spills can affect many animals if they occur at times and places where the animals have congregated in large numbers

The circumstances of an oil spill are often more important for the extent of damage to animal populations than the size of the spill. Large numbers of seabirds have been reported dead following relatively small spills, whereas some large spills have been associated with relatively low mortality. The decisive factor has been the concentrated occurrence of animals or their vulnerable life stages and the overlap with the spilled oil.

Seabirds and marine mammals can be highly aggregated in the Arctic. Seabird colonies may range in size from hundred thousands to millions of individuals that feed in the vicinity of the colonies. Seabirds also aggregate to feed in polynyas and leads in the spring prior to breeding, for molting at sea after breeding, and for wintering in polynyas and the marginal ice zone in the low Arctic or in open sub-Arctic waters. In these situations they are highly vulnerable to oil spills.

Marine mammals also congregate in confined areas for purposes such as pupping and molting, or live normally in social groups. Harp and hooded seals aggregate in large numbers at specific sites to give birth to their pups on the ice. The northern fur seal (*Callorhinus ursinus*) is very concentrated at breeding colonies, the largest being at the Pribilof Islands. A spill near such concentrations of marine mammals can have an impact disproportionate to its size.

Bowhead and beluga have seasonal migrations between wintering areas in the northern Bering Sea and the Davis Strait region, into the high Arctic during summer. Narwhal have similar migrations from their wintering area in the Davis Strait. These migrations follow recurrent systems of shore leads and polynyas that develop in late winter and spring. During migration and feeding the whales may occur concentrated in the restricted areas of open water in the ice where they may be sensitive to oil spills.

Several species of migratory waterfowl and shorebirds have staging areas in coastal habitats. Some of these areas are very important because a large fraction of the population aggregates there and feeds to fuel for the southward autumn migration. These areas are very sensitive to oil spills. Examples include the Yukon-Kuskokwim Delta in western Alaska, Izembek and other lagoons on the northern side of the Alaska Peninsula, estuaries in Yamal and Gydan and the Lena Delta in northern Russia, and the Mackenzie Delta and sites along the southern shores of the Coronation Gulf and Queen Maud Gulf in northern Canada.

The ice edge at the transition between ice-covered and open waters has a special biological significance. The ice provides resting places for birds and seals, while the ice edge phytoplankton bloom and concentrated occurrence of zooplankton and small fish may provide good feeding conditions. Polar bears also frequent the marginal ice zone hunting for ringed seals (*Phoca hispida*). The ice edge may be a particularly vulnerable zone due to the concentrated occurrence of marine life and the possible accumulation there of drifting oil.

F6.E. Small cod-fishes that spawn under the ice are sensitive components of Arctic marine ecosystems

Polar cod (*Boreogadus saida*), Arctic cod (*Arctogadus glacialis*), saffron cod (*Eleginus gracilis*), and navaga (*Eleginus navaga*) are Arctic cod-fishes that spawn under the ice in winter. The eggs have a long incubation time and hatch when the ice starts to melt and spring growth of plankton resumes. These small cod-fish species are important in coastal and offshore parts of the Arctic marine ecosystems. There is limited information on their population structure and their specific spawning areas.

Potentially, an oil spill in winter could overlap with the spawning areas of these cod-fishes and impact on the eggs or larvae. This could affect the recruitment and size of the populations and have repercussive effects on other parts of the ecosystems due to the roles of these fishes as prey for animals at higher trophic levels.

Other fish species are also sensitive to oil spills. Pacific herring (*Clupea pallasi*) and capelin (*Mallotus villosus*) spawn on sandy beaches and in shallow subtidal waters where the eggs adhere to the bottom substrate. Several coregonid whitefish species spawn under the ice in rivers and lakes where the eggs incubate to hatch in spring. Pacific (*Hippoglossus stenolepis*), Atlantic (*H. hippoglossus*), and Greenland (*Reinhardtius hippoglossoides*) halibuts spawn along continental slopes in the Bering Sea, Norwegian Sea, and off Iceland, western Greenland, and Labrador. The eggs have a long incubation time and could be sensitive to oil spills from deep blowouts in these areas.

F6.F. An oil spill in ice-covered waters could have a large ecological impact

The impact of an oil spill in ice-covered waters depends in part on the amount and fate of the oil and its overlap with

sensitive organisms. The ice cover would tend to preserve the hydrocarbons in the oil. Exposure to volatile components could therefore be longer, and higher concentrations could be maintained than under open-water conditions because the evaporative pathway is much reduced.

Large effects could possibly occur if the oil overlapped a spawning area of polar cod or other small cod-fishes that spawn under ice. The same applies to congregations of seabirds or marine mammals where they have come to feed or, in the case of marine mammals, to breathe. If the oil spread along the lead systems that bowhead, beluga, narwhal, and walrus use on their seasonal migrations, the potential conflict could be large. The same is the case if the oil spread along ice edges with concentrated occurrences of seabirds and seals. In worst-case scenarios, large fractions of populations could be affected and killed.

The extent of ice cover is a key determinant for the impact of a spill in ice-covered water and for the possible remedial actions that can be taken. One hundred percent ice cover of sufficient thickness to support clean-up equipment and crews using terrestrial clean-up techniques allows highly efficient spill remediation and limited subsequent impact in some spill situations. Thin ice or broken ice prevents the use of either terrestrial or open-water marine techniques, limiting response to burning in place. The worst-case scenario for an Arctic marine spill is a low volatile crude spilled in broken ice conditions at a time and place when animals have congregated.

Finding 7: Oil and gas activities have had environmental effects locally but long-term changes to Arctic wildlife populations have not been documented

F7.A. Pollution effects due to oil and gas activities in the Arctic are local

Oil hydrocarbons can be acutely toxic to aquatic life forms if they occur at high concentrations. Low-boiling aromatic compounds in the low parts-per-million concentration range in the water kill fish if they are exposed for periods of a few days. Variations in sensitivity occur among species and life stages, with sensitivity to acute effects of oil generally greater in larval stages. At lower concentrations than those causing acute effects, chronic responses may occur. Chronic effects are most often of concern with exposure via sediments because hydrocarbons can persist there for decades or more, and studies have shown that such exposure can result in tumours and genetic changes. Chronic effects are relevant in the Arctic because degradation rates are slower there. Studies to estimate 'no observable effect concentrations' (NOECs) from chronic exposure studies indicate that these fall in the low μg/L range. Sunlight enhances greatly the toxicity of several PAHs to aquatic organisms, especially those organisms that are translucent and occupy shallow habitats where light can penetrate.

There are a wide range of mechanisms whereby oil hydrocarbons and PAHs exert their toxicity. These include effects on cell membranes such as anatomical and physiological effects on gills, effects on the central nervous system such as narcosis and behavioural changes, biochemical effects including changes in levels of enzymes and hormones, induction of tumours, and others. The limited information available on truly Arctic species suggests sensitivity similar to that of temperate species. However, because petroleum hydrocarbons degrade more slowly under cold and dark conditions, Arctic organisms may be exposed to higher levels for longer periods. Also, the clearance of hydrocarbons taken up in cold-blooded organisms through metabolism and excretion is slower in colder temperatures, effectively extending retention of hydrocarbons within body tissues. This could lead to more prolonged toxicological effects under Arctic conditions compared to temperate conditions.

The concentrations of oil hydrocarbons and PAHs in the Arctic are generally low and close to background levels. Biological effects on aquatic and terrestrial animals are therefore not expected away from locally contaminated areas. There is considerable input of oil from natural seeps in some areas of the Arctic and it has not been clearly established whether or not there are biological effects associated with these inputs.

Local contamination occurs in some areas due to spills and other inputs of hydrocarbons and PAHs from oil and gas activities and other sources. Concentrations may exceed those where effects can occur. Documented effects on communities of benthic animals were found in cases such as in a near-zone where contaminated drill cuttings had been released from offshore platforms in the Norwegian Sea. Similar detailed information from other parts of the Arctic (e.g., Russia and Alaska) was not provided for this assessment. This has prevented an assessment of the current levels of exposure of terrestrial plants, mammals and birds to chemicals produced and used by oil and gas activities in the vicinity of facilities. Chemicals of potential concern include volatile compounds and vented gases, oil, petroleum products and oilfield chemicals. Exposure in mammals and birds occurs by the inhalation of volatile components, ingestion of contaminated food and water, and the absorption of chemicals through the skin. Domestic animals have been exposed to oilfield chemicals in temperate regions and have shown a wide range of biological effects, although their significance is unknown. Small mammals near petrochemical facilities have also shown genetic effects due to hydrocarbon and metal exposure but the extent of these exposures in not known in the Arctic.

The data on concentrations of petroleum hydrocarbons from Russian terrestrial environments are aggregated for larger geographical areas and the inter-comparability of field data with concentrations reported in toxicological studies is uncertain. It has therefore not been possible to evaluate the extent of areas of local contamination, and whether these might reach the scale of a regional problem rather than a local problem.

F7.B. Physical impacts in the marine environment are local

Physical impacts in the marine environment may occur during drilling and the construction of infrastructure for drilling, production, and transportation. Constructing gravel islands necessarily covers a finite area of the seafloor with gravel and thus buries organisms in the affected area. Plumes of sediment often form down-current from the island construction operation and may affect plankton and benthos. If the gravel islands were for exploration wells, they were often left in place to erode or persist. Abandoned gravel islands may also provide new habitat for organisms. Temporary ice islands have a local and transient effect on the seafloor and seafloor organisms. Limited physical disturbance of the seafloor can be caused by drill ships, which leave anchor depressions around the vessel, bottom-founded drilling structures, and seafloor excavations.

Discharge of drill cuttings in the marine environment where permitted, will form a pile of cuttings that will cover a relatively small area, although fine particles may spread to be thinly distributed over a wider area. Organisms would be affected under the pile of cuttings and possibly by plumes of mud from unwashed cuttings. Such effects are usually limited to a few km^2 of seafloor at most.

Marine pipelines may be laid directly on the seafloor in waters not affected by destructive ice or oceanographic processes, or they may be buried, such as the Northstar pipeline in the U.S. Beaufort Sea. Pipelines laid on the seafloor may disturb mobile bottom dwelling species and have a limited effect on organisms directly beneath them. Pipelines may disturb archaeological resources such as shipwrecks if laid on top. Surveys using remote sensing devices or television will usually mitigate such disturbances. Trenching the seafloor for burying pipelines or for installing sub-sea completion systems will disturb the areas trenched and possibly areas down-current if plumes of sediment are released during trenching or backfilling the trench.

F7.C. Noise associated with oil and gas activities is a source of disturbance

The highest sound levels from oil and gas activities in onshore areas are from aircraft. Low-flying aircraft may disturb Arctic wildlife and can have a negative impact particularly on species that are aggregated during important stages in their life cycles. This includes aggregations of flightless geese and seaducks during their annual molt, aggregations of geese, ducks and shorebirds at staging areas where they feed to fuel their migrations, and seabirds at their breeding colonies. It also includes walrus haul-outs and calving areas for caribou/reindeer.

In offshore areas, the noise source related to oil and gas activities with the highest potential for affecting marine life is airguns used in seismic surveys, and the animals most at risk are marine mammals. Seismic surveys are monitored to ensure that close contact with marine mammals is avoided, but how this distance is determined varies from country to country and species to species. The effects from sound are generally local to a zone near the source. This may extend tens of kilometers in the case of offshore seismic surveys where avoidance effects on whales such as bowheads have been observed. However, this would affect a limited fraction of the total population and be transitory, and there is no evidence that local effects have scaled up to have a significant effect on populations. Disturbance may also occur from traffic by ships, icebreakers, and tankers associated with oil and gas activities. The many different sources of sound associated with oil and gas activity in the onshore and offshore Arctic regions, combined with rising ambient sound levels from other human sound sources, may lead to cumulative effects from all sources.

F7.D. There is little evidence that Arctic oil and gas activities have caused long-term changes to Arctic wildlife populations

Oil and oil and gas activities have the potential to affect Arctic animal populations, as was clearly demonstrated in the case of the *Exxon Valdez* oil spill in Prince William Sound although this event happened outside the Arctic. Many examples of local effects on individual animals have been documented in the Arctic. However, it is not known whether local effects, such as acute mortality or chronic effects on individuals from oil and hydrocarbons, and disturbances and displacement of individuals from infrastructure and activities, have accumulated to produce a clear and detectable effect on the size or status of Arctic animal populations. Evidence suggests that any such effects are unlikely to have occurred at a scale where they have represented a threat to populations or species. Whether this is true in Russia is not known because there is insufficient information to enable an evaluation.

Arctic species are often widely distributed and many have high intraspecific variation in the form of sub-species in different areas. Fish, birds and mammals also form distinct populations or stocks that occur in a spatial and geographical context and may be reproductively separated from other populations or stocks. The status of populations or sub-species may vary, some being large and healthy while others are small or declining and considered to be endangered or threatened. The vulnerability of populations or sub-species is related to their status because sub-species with small populations and/or endangered status may be more vulnerable to additional stress than sub-species in better status.

Documenting effects at the population level for fish, birds and mammals is confounded by their large natural variation and the lack of knowledge and monitoring of population changes. The caribou populations on the North Slope of Alaska have been particularly well monitored in relation to oil and gas activities. While some disturbance and change to caribou distributions may have occurred, any negative effects were not enough to prevent a large increase in the populations from the 1970s to the 1990s. It is not known how important any negative effects would be in herds that are currently in decline.

Although evidence of population effects is currently limited, subtle effects could still be occurring and could become more expressed in future under different climatic conditions and in combination with effects from other activities. Monitoring and documenting such effects may be a challenge that will require considerable effort.

F7.E. Exposure to oils affects the quality of fishery products for human consumption by imparting undesirable tastes and odours

Hydrocarbons taken up by fish and distributed to muscle can impart tastes and odours that render the fish unfit for human consumption. This is usually the most important economic loss when spills overlap areas used for commercial or subsistence fisheries. Tainting occurs in exposures that last for minutes to hours in fish with high levels of fats in muscle (e.g., salmonids, coregonids) but is less important for lean species like cod or saithe (*Pollachius virens*). Tainting of salmonid fish can be retained for periods of weeks to months. The components responsible for tainting are likely to be the low-boiling aromatic compounds. Based on experience from the *Braer* spill in Shetland, retention of tainting may be even longer for some invertebrate species. In subsistence fisheries, tainting or fear of contamination has been responsible for large declines in uses of subsistence foods. The only known example of tainting within the Arctic is the tainting of whitefish following a small spill of diesel fuel into the Cameron River, NWT, Canada.

Finding 8: Human health in the Arctic can potentially be affected by oil and gas activities but there is limited information to assess if effects have occurred to date

F8.A. Some components of crude oil have the potential to cause acute and chronic human health impacts

Oil and gas exploration, extraction and transportation activities, regardless of where they occur, have the potential to cause short-term and long-term adverse effects on human health. Such effects, if and when they occur, are likely to be local with respect to the source, whether near the drill or production site or as a result of oil spills.

Human exposure to volatile substances associated with oil and gas activity in the Arctic is primarily via inhalation, and only slightly by skin contact and by oral intake. Exposure to less volatile compounds like oil and some PAHs, is primarily by oral exposure and skin contact and less by inhalation. Exposure duration to volatile substances (mainly after an accidental spill) is short-term (acute) and may only last for a few hours. In general, human reactions to acute exposure to petroleum components are mainly transient. Exposures to oil and PAH contamination, which may result from an accidental spill or local contamination at a production site, tend to be longer-term (sub-chronic or chronic).

Inhalation exposure resulting from flaring at the well head and open burning of spilled oil can be dangerous because the particulate matter which may result is easily inhaled, retained in the lungs, and can contain high concentrations of bio-available contaminants (PAH, sulfurous compounds, dioxins, furans, and metals). While flaring of gas and open burning of oil are now only occasional events in most of the Arctic region (Alaska, Canada and Norway), there is information that flaring is a regular activity in parts of Russia. Short-term flaring in remote areas (away from local populations) will not affect health because the exposure concentrations in the air of the toxicants emitted will be very low and well below those reported in most urban environments.

Dermal exposure (via skin contact) of humans and animals to crude oil has been shown to produce toxic effects. Most skin-related effects are transitory, for example, once exposure ceases, skin inflammation is likely to clear up in seven to ten days. However, repeated skin exposure in mice to some PAHs has resulted in skin cancer and highlights a need to limit repeated human skin exposure to oil during exploration, production and spill clean-up.

Several PAHs can be ingested in food and water although human populations tend to avoid eating contaminated food or surface water which is clearly tainted (visible oil, taste or odour detection). Ingested PAHs, some of which are known carcinogens when inhaled or applied to skin, are not thought to contribute significantly to ill health, and the risk of excess cancer related to ingested PAH is very low. Smoked and open fire-roasted foods have a higher PAH content than most foods obtained from oil contamination zones where they are considered unsuitable for personal consumption based on their taste and odour. In the Arctic, oral exposure of populations living on or near contamination sites or spill zones to fresh oil is rare. Volatile aromatic compounds evaporate rapidly and degrade quickly in sunlight. They do not appear at concentrations of concern in food and surface water. Groundwater may be contaminated with volatile organic compounds, metals and oil; however, concentrations are usually very low and effects, if any, tend to be transitory.

F8.B. Cancer risks in regions of oil and gas activity have not been established

Based on animal studies, both benzene and some PAHs have been classified as carcinogens. One study from a region in the southern hemisphere has reported elevated cancer rates among an exposed population; however, these results have not been supported by similar studies in other regions. In general, it is unlikely that there is much excess cancer in most of the Arctic region as a result of oil and gas activity because exposures to known carcinogens are likely to be very low and very few people are likely to be exposed. A statistically significant identification of excess cancer under these conditions (low exposure and low numbers of exposed individuals) would be difficult, if not impossible. Benzene, a human carcinogen, tends to occur at very low concentrations in the vicinity of oil and gas operations. Due to its vapour pressure, benzene volatilizes to the atmosphere after a spill and airborne levels decline rapidly (less than 24 hours) to concentrations below the levels of concern specified by health agencies in several Arctic countries.

F8.C. Psychological damage appears to be a consistent impact of oil spill situations

Psychological damage which may be manifest as diagnosed post-traumatic-stress-disorder (PTSD) or other measures of effect has been sustained for up to, and beyond, two years in some studies of spill-affected human populations in sub-Arctic regions. Effects are felt by individuals in both non-indigenous and indigenous communities; however, in the case of published oil spill investigations, the effects have been more strongly felt by indigenous communities and by women within those communities. These psychological outcomes are important and can lead to social and economic impacts such as domestic violence, lost income and loss of community structure. Tainting of food as a result of oil contamination or fear of tainting, can lead to dietary changes in the community and among individuals and can also result in psycho-social impacts.

F8.D. There are insufficient human exposure and epidemiological data available for the Arctic region to conclude whether non-occupational population groups are currently affected and to undertake a robust risk assessment

In general, there is no good characterization of contamination sites in the Arctic region and a complete lack of exposure information for Arctic populations living near to oil and gas activities. While few people appear to live in the vicinity of most areas of activity, there are several persistent references to serious land contamination in some parts of Russia and anecdotal reports of poor health in populations living near these areas. However, there are no peer-reviewed studies of health outcomes in non-occupationally exposed Arctic populations.

Sub-Arctic populations have experienced transient effects related to volatile off-gases (nausea, headaches, lethargy, irritability, runny noses, sore throats and eyes, depression) and oil (skin irritation, cough). Risk assessments for Arctic populations could make use of epidemiological studies of these sub-Arctic populations exposed to oil and its components following spills, but only if detailed information on exposure, living conditions, health status and other parameters are available for the Arctic populations. The lack of health impact information for Arctic populations reflects the limited number of spills near Arctic communities, and the difficulty of conducting meaningful and statistically

reliable health studies in small communities with small populations.

Physiological, psychological, and social impacts are all linked. It is clear that oil and gas activities by themselves and in combination with other 'determinants' of health (e.g., education, access to health care, nutritional status, family income, social status, genetic and biological endowment, personal lifestyle choices) could affect the physical, mental and social health of individuals and communities in the Arctic. Furthermore, cumulative impacts of environmental exposures are important. Local populations in the vicinity of oil and gas activity may be affected by exposures to spills, particulates, gases, and sanitary waste. They may also be affected by traditional foods contaminated by persistent contaminants and metals which arrive in the Arctic via long-range atmospheric transport from sub-Arctic regions. Food stress, which can affect nutritional status, may come from pressure to change diets due to contamination, lost access to prey, or social pressure.

It is likely that indigenous people would be more affected by oil and gas activity than non-indigenous individuals as a result of their proximity to oil and gas exploration/extraction areas, their poorer general health and socio-economic status, and their greater vulnerability to changes in traditional ways of life (living off the land, community cohesion, food sharing). Local employment and migrant workers associated with oil and gas activities could introduce and spread infectious diseases into indigenous communities, affecting family health and community structure and operation. Crowded conditions, re-infection of home communities, loss of access to traditional foods and social structure, and access to alcohol and drugs will all play a part in this process. It is not possible to quantify this risk as it will vary from location to location and is dependant on documented physical health outcomes. Poor early detection and health service response will compound and confound the combined effects of health stressors.

F8.E. Oil and gas activity in the Arctic can have a positive impact on health

There are likely to be some beneficial impacts of oil and gas activity on health (i.e., physical, mental and social well-being). These relate to more employment and availability of disposable income, better sanitation and access to health care and improved transportation.

Finding 9: Technology and use of best practices have lowered environmental impacts, but additional risks may occur as conditions change or new areas are explored and developed

F9.A. Technology and practices have adapted and evolved to deal with Arctic operating and environmental conditions

Experience with effectiveness and impacts of past activities, and growing concern by industry, governments, interest groups, the general public, and local residents, have led to the development of new technologies and practices for Arctic oil and gas activities. These are specifically adapted to the Arctic region and continue to evolve. Very few modern Arctic oil and gas facilities are operated as they were twenty years ago; large and capital-intensive developments such as the *Snøhvit* field in the Barents Sea were just not possible twenty years ago. Technological advancements have been applied to Arctic oil and gas exploration and development activities and these have significantly reduced the impact on the environment.

Impacts on the environment and on biological resources can be mitigated or reduced by Arctic specific technology. This is demonstrated by the use of low impact seismic techniques in the boreal forest, tundra and wetland areas. Reduction in environmental impacts results from the increased use of vibrator vehicles, development and use of light-weight vehicles to reduce ground pressures, and reduced breadth and necessity of cut lines. Offshore, new air-gun technology and improved operating procedures have reduced the impact on the marine environment. Three-dimensional seismic surveys are more focused and able to image the subsurface more accurately, thereby reducing the number of wells that need to be drilled to define a possible deposit.

Well drilling technology and well design have undergone significant changes in the past 20 years. New exploration wells are drilled in the winter, and technology using ice roads or roadless access, and drill pads made of ice, leave virtually no footprint and have minimal impact. Changes to rig design and well drilling methods have reduced the size of development drill pads by 30% to 40% compared to the size of earlier pads. Deep well injection of waste drill cuttings/muds, other drilling wastes, and produced water from oil fields plays an important role in reducing surface impacts.

However, improved practices are not consistent across the Arctic and damage from development continues in some areas.

F9.B. Changing conditions at existing fields and exploration and development in new areas may introduce additional risks for potential impacts on sensitive Arctic species and habitats

Management and maintenance of infrastructure in aging Arctic oil and gas fields need diligent application of best technologies and practices for monitoring and integrity assessment. Equipment will need to be upgraded and infrastructure elements may need repair or replacement. As the end of life approaches for some of the largest and oldest fields, decommissioning activities may take place. These activities are relatively new to the Arctic and are expensive to address. Even though the cost of producing diminishing oil reserves grows enormously as fields age, the use and development of the best available technology and best practices is essential and should not be neglected.

Oil and gas activities are likely to expand into new areas of the Arctic. This expansion may require, and be facilitated by, further adaptation and improvement in technology. Some of these new areas will be further into the high Arctic, and the expansion may represent additional risk for potential impacts to sensitive Arctic species and habitats.

A warming climate has many implications for Arctic technology and practices. In addition to adapting current technology and practices to the effects of changing climatic conditions, new technologies and practices may need to be developed. The effects of thawing permafrost can erode coastal and riverbank facilities, damage field and transportation infrastructure, and compromise well integrity and therefore require the urgent application of adaptive engineering, and the development of new design solutions. A longer ice-free season in the Arctic seas may allow petroleum and transportation industry access to areas in deeper water farther out on the Arctic shelves and Arctic Ocean, but also requires improving technology and

practices for emergency response at ever more remote and distant locations. Longer open-water seasons also cause accelerated wave erosion of the coast, loss of ice and ice-edge wildlife habitat, and disruption of local indigenous communities that depend on a subsistence way of life. If landfast ice becomes less stable or is lost for longer periods, an important platform for winter transport and operations will be lost and increased broken ice conditions near shore will also make it harder to respond effectively to spills.

While new ways to reach and produce oil and gas efficiently and safely is a driver for the development of new technology and best practices, environmental, emergency response, and engineering requirements for mitigation, prevention or elimination of adverse environmental or cultural impacts are also major considerations. These technologies and practices must continue to evolve and to adapt to changing and new conditions.

Finding 10: Governance, regulatory systems, and international standards are important aspects of the performance of the oil and gas industry in the Arctic

F10.A. The use of international standards and best practices are contributing to a reduction in negative environmental and socio-economic effects from oil and gas activities

Operational requirements related to financial considerations are working to bring common operating standards to Arctic projects. Increasingly, Arctic operations are conducted either by publicly traded international companies and/or are financed at least in part from international financial organizations. Large companies have increasing interest in adopting accepted international standards, technology, techniques, and practices due to shareholder and stakeholder concerns and through requirements set by most international finance institutions like the World Bank. This has increasingly led to industry conforming to a number of internationally accepted standards in order to fully participate in the worldwide petroleum market. These standards include common reporting requirements for reserves, petroleum quantity and quality measurement, safety procedures and environmental protection.

Government regulators are also communicating between countries and across jurisdictions through various forums and thus have increasingly incorporated accepted international standards, technology, techniques and practices into their rules, regulations, and guidance documents. All Arctic nations including those with more traditionally prescriptive regimes contain goal-setting or performance-based features in their regulatory systems, which allow industry to use the latest and best technology and techniques as they become available.

Internationalization of the petroleum industry and the increasing use of international standards and practices by Arctic nations have contributed to the reduction in potential negative environmental and socio-economic effects from oil and gas activities.

F10.B. Arctic national oil and gas legal regimes are relatively stable, modern and designed to protect human health, rights of indigenous residents and the environment, but in some cases regulatory systems are outdated, incomplete, or enforcement is inadequate

All Arctic nations with oil and gas activities have a legislative and regulatory base within which these activities are allowed. There are many similarities among these regimes; they allow for access to the resource, they regulate the activities associated with exploration, development, production, transportation, and decommissioning, and they protect national interests including security, financial, environmental, social and cultural interests. All these legislative and regulatory regimes have undergone some degree of change over time and are still changing. Although some aspects of these systems change due to new data, evolving technologies and practices, and as a result of accidents or events, most Arctic nations have stable and predictable regulatory regimes. Russia has undergone the greatest shift, having evolved from a non-commercial oriented central economy system to a modern legislative and regulatory regime in a very short period of time. The Russian system is still in a state of transition.

Transparent and clear regulations and laws allow industry and government to plan and conduct operations in a predictable and systematic manner while keeping all stakeholders, including Arctic residents, informed and involved. All Arctic countries have systems that accommodate public involvement and allow some form of legal challenge and intervention in projects that do not meet required standards.

All Arctic nations are politically stable compared to many of the other oil and gas producing countries. The clarity and transparency of regulatory regimes are important factors in protecting not only industry and government interests, but also for protecting the environment, human health and socio-economic well-being of Arctic residents. Under such governance, industry is able to make financial commitments and proceed with planning activities, while adhering to even very complex regulatory requirements for safety, environmental protection, and conservation of resources. Governments can also set out comprehensive controls and standards and can plan accordingly to identify and address concerns over proposed activities. Interested parties and the public may also be afforded enough time and information in order to have input or plan for these activities at a local and individual level.

Even though all Arctic countries thought to have petroleum potential have modern legal regimes for oil and gas activities, there must be effective control of these activities. No matter how complete and comprehensive the legal system, without adequate regulations to implement these laws or without confidence that these regulations are adequate or evenly applied to activities, the protection of occupational safety, and economic, environmental, and cultural security are compromised. To ensure that regulations are adequate and consistently applied they should be continually reviewed and assessed for their effectiveness. The regulatory system should allow for changing and updating of rules and required standards to adapt to new conditions and technology. Governments should ensure that the authorities are properly trained and staffed to enforce regulations and to ensure that compliance, inspection, auditing and monitoring procedures are followed. Regulatory control should continue to be based on the best available science, technology and practices.

7.3. Recommendations

This assessment has been conducted to provide a basis for the development of policy and management measures. The assessment is scientific and is the responsibility of the team of authors that was given the task to carry out the assessment. It is not the task of the scientific authors to provide detailed advice on policy or management measures; this advice is prepared by the AMAP Working Group based on the scientific findings of the assessment. However, the scientific authors have provided some general recommendations relating to the management of oil and gas activities in the Arctic, which reflect the main findings.

The findings of this assessment build upon a large amount of scientific information from monitoring and research that was compiled and examined, and documented in the preceding chapters of this report. The present section provides recommendations that aim to improve the basis for future assessments by generating comparable data from all Arctic countries. One aspect is the lack of information on particular topics and particular areas. It is likely that in many cases this information did exist, but that for various reasons it was not accessible or made available for the assessment. A second topic is the need for monitoring to update existing information. A third issue is the need to address certain gaps in knowledge by research.

Each of the preceding chapters of this report concludes with a set of recommendations. The summary recommendations listed in this section reflect these recommendations, and the individual chapters should be consulted for further details, background, and the context to the statements made here.

7.3.1. Managing oil and gas activities in the Arctic

Prevention of oil spills

R1. In all aspects of oil and gas activities, attention should be focused on the prevention of oil spills. The highest priority should be to prevent oil spills in ice-infested marine waters. In this respect it is recommended that consideration be given to:

- the conduct of risk assessments in association with all means of transport of oil and gas;
- the use of best practices and technology in transport and storage of oil;
- seasonal restrictions on oil and gas activities;
- the need for protected areas closed to oil and gas activities;
- strengthened capabilities and improved coordination of oil spill prevention, preparedness, and response; including
- rapid availability of adequate oil spill response equipment and well-trained personnel.

Use of best practices

R2. Oil and gas activities have the potential to cause significant impacts on Arctic ecosystems and peoples, but this can to a large extent be prevented by use of the best and most appropriate technologies and practices. Arctic countries should require the use of best available technologies and practices, including where appropriate:

- appropriate consultations and collaboration with communities that may be affected, to develop strategies for avoiding negative impacts while harnessing economic and other opportunities;
- closed-loop drilling systems where drilling wastes are re-injected or cleaned and safely deposited;
- transportation, including pipelines, and other infrastructure to be built or modernized, and maintained according to the highest industry and international standards;
- 'roadless' development techniques to reduce the physical impacts of roads;
- conduct of activities on frozen land in winter months to avoid physical impacts on the ground and vegetation;
- seasonal restrictions on activities to avoid disturbance to wildlife in sensitive periods and areas; and
- monitoring of wildlife to regulate activities to reduce disturbance and impacts, including the use of marine mammal observers aboard seismic, icebreaking, and other ships to avoid close approaches and disturbance.

R3. Clear and flexible regulations should continue to be used that are goal-oriented and supported by appropriate guidance to reduce the risk of accidents and the extent of environmental effects, and to improve safety. Emphasis should be given to compliance monitoring of infrastructure and practices to ensure that standards and regulations are effectively and consistently followed. Arctic countries should establish a mechanism through which to share experiences, and should coordinate and cooperate concerning their methods of risk and impact assessments and management of the oil and gas industry. Arctic countries should use an adaptive management approach to ensure that new information can be incorporated into the management and decision-making processes and changes in conditions can be accommodated or mitigated.

Pollution prevention

R4. It is possible to operate oil and gas activities with little or no discharge of contaminants to the environment. A policy aiming for zero routine discharge of harmful substances should be adopted in order to prevent pollution of the Arctic environment. This should include:

- reducing or ending the flaring of associated natural gas (except in emergencies and for safety reasons);
- reducing or eliminating discharges to the terrestrial and aquatic environment and ending the use of sumps and pits for the disposal of spent muds and cuttings from onshore drilling and production operations; and
- using material and chemicals that are environmentally manageable and techniques that conserve, recycle and reuse waste.

R5. Action should be taken to clean up and remediate sites that are badly polluted, including old or abandoned sites, in order to significantly reduce or prevent threats to the health of human populations and wildlife living in the area.

7.3.2. Lack of information for assessment

A certain amount of the information required for this assessment was probably available in principle, but was not provided for the assessment or could not be obtained through

ordinary means. The recommendations in this section are intended to provide a better basis for future assessments.

Point sources of pollution and concentration gradients
R6. To provide the basis for an assessment of the quantities of waste from Arctic oil and gas activities and the treatment of such waste, it is recommended that better reporting procedures be developed for:

- waste management measures, including re-injection or discharge, and recycled or reused volumes and the chemical composition of wastes; in particular
- produced water discharge volumes, disposal methods and locations, and chemical composition including polar components such as alkylated phenols and other substances.

R7. Point sources of operational discharges and emissions from oil and gas activities need to be identified and the types and amounts of contaminants released should be monitored and reported. These data should be stored and made available from regulatory agencies in the Arctic countries. This should also be the case for accidental spills and releases. Concentrations of the released contaminants in the environment should be monitored and reported, allowing assessment of gradients and areas affected by contamination. This would enable assessment of the degree and area of pollution.

R8. Contaminated sites from spills and releases from past oil and gas activities should be identified and monitored to determine the degree of pollution in terms of concentrations, areas affected, and risks to humans and wildlife (see also R14).

Habitat fragmentation
R9. Information on pipelines and other infrastructure that can act as impediments to wildlife movements, or as attractors to wildlife, should be provided. The information should include an inventory of line length (kilometers), locations, types, placement (whether on ground, raised, or buried) and age of all pipelines (both transmission and in-field pipelines) in the circumpolar regions.

R10. In order to assess the degree of habitat fragmentation, information on roads and associated traffic should be provided. This should include all roads and traffic in the areas of oil and gas activities, to contribute to assessment of cumulative effects.

R11. Information on air traffic related to oil and gas activities should be provided, in particular low-flying helicopters and fixed-wing aircraft that disturb wildlife.

Socio-economic conditions and human health
R12. Countries should be encouraged to collect and compile comparable Arctic oil- and gas-related socio-economic statistics. A circumpolar assessment of the socio-economic effects of oil and gas activities in the Arctic requires the collection and compilation of intercomparable information and associated collection protocols on a number of parameters including employment, wages, gender, industry expenditure, GDP contribution, social infrastructure, use of subsistence resources, cultural practices, consultations, mitigation, and occupational health and safety.

Standards and regulations
R13. Given the large volume of detailed national regulatory laws, standards, guidelines, and procedures for oil and gas activities in force in the Arctic countries, it is recommended that a compilation be made by the Arctic Council and its working groups and periodically updated.

7.3.3. Monitoring to improve the basis for assessment

Monitoring is an important source of data and information in the conduct of assessments. Based on this assessment, the recommendations below have been developed to improve and enhance the development and implementation of monitoring programs and the use of the resulting data.

Contaminated and polluted areas
R14. Contaminated sites within the Arctic from past and current oil and gas activities should be located, mapped, and characterized (size, history, contamination, geology, biology) to determine their potential impact on humans and biota in the surrounding environment. Reports of adverse impacts on animals and human populations near contamination sites can only be verified if there is a full identification and characterization of these sites.

R15. A selection of contaminated sites should be chosen for long-term monitoring of the degradation and fate of spilled oil, petroleum hydrocarbons, PAHs, and other oil-related substances. This would assist in the prioritization of and decisions for remediation and rehabilitation of sites, as well as in assessing future risks of environmental and human health effects.

Compliance monitoring
R16. Monitoring of facilities is necessary to assess industry compliance with legal and technical standards applicable to the oil and gas industry in Arctic countries; this should include, in particular, monitoring of pipeline integrity, drilling and well work-over activities, and the construction of facilities, as well as groundwater reservoirs and areas near onshore wells and pipelines to ensure that there is no evidence of leakage from the oil or gas facility and that environmental quality standards are being met.

Integrated monitoring and assessment
R17. Consistent, rigorous monitoring programs should be developed using measures that can be applied throughout much of the Arctic to allow the detection of changes in the environment, society, and human health. Such programs should include new tools, such as biological markers of exposure and effects and sociological indicators of change.

R18. Monitoring programs should measure physical, chemical, biological, and socio-economic conditions that may be impacted by oil and gas activities, and should be based on internationally agreed protocols for chemical and biological monitoring. Before petroleum activities commence, monitoring should begin with a comprehensive baseline investigation, which should incorporate existing information, and comprise as a minimum all monitoring sites and variables planned to be used in the long-term monitoring program.

R19. Chemical and biological monitoring should be conducted using appropriate quality assurance in relation to the design of the sampling program, the collection of samples, and the analytical procedures used. Where possible, participation

in Arctic-wide or international intercomparison exercises on analytical measurements is recommended to promote comparability in the results among the institutes and laboratories conducting monitoring in the Arctic. In addition, a systematic approach to the handling, evaluation, and reporting of all types of data to be used in assessments should be developed for use on a pan-Arctic basis.

R20. The specific compounds of petroleum hydrocarbons that elicit most of the toxicological effects recorded in laboratory experiments (e.g., volatile aromatic compounds) are usually excluded from environmental monitoring programs; such programs tend to focus on the more persistent, high-molecular components. To bridge this gap and allow assessment of the environmental concentrations of these more toxic compounds would require the monitoring programs to pay particular attention to the compatibility and comparability of the data.

R21. Monitoring should be conducted so as to enable a distinction between impacts due to oil and gas activities and impacts from other sources, including natural climate variability and climate change and social changes from causes unrelated to the oil and gas industry. Monitoring needs to be tailored according to the type of oil and gas activity and to the nature of the potentially affected environment, and should be coordinated in regional ecosystems (e.g., LMEs [Large Marine Ecosystems] for the marine environment) so that interactions and cumulative effects from multiple activities may be examined.

Animal populations
R22. High taxonomic diversity within many Arctic animal species, divided into several subspecies and stocks, is related to a high degree of site fidelity and local adaptation of breeding populations. This is an important aspect of the functional integrity of Arctic ecosystems, which must be taken into account in impact assessments. There is a need for better identification of population structures and for the monitoring of populations and subspecies of Arctic mammals, birds, and fish species.

Human health
R23. Pan-Arctic monitoring of human health status should be conducted in relation to oil and gas activities, including psychological impacts as well as the levels of contaminants in ambient air, water, food, and human tissues, to allow an assessment of current population exposure and health, with special focus on community assessments and on children and women of reproductive age.

Environmental impact assessment
R.24. A large number of regional project-specific environmental impact assessments and statements (EIA and EIS) have been carried out in relation to oil and gas activities in various parts of the Arctic. These summarize knowledge and use results from monitoring programs. There is scope for better use and streamlining of the production of EIA/EISs as well as of pan-Arctic assessments such as the current assessment. As a first step this should be explored through an exchange of information and experiences among the Arctic countries.

7.3.4. Gaps in knowledge

This assessment has identified many gaps in knowledge of the impacts of Arctic oil and gas activities on the environment, biota, and human populations of the Arctic. This is partly due to an incomplete understanding of environmental conditions in the relevant areas of the Arctic and of the species and populations of the many plants and animals that live there as well as their ecological interactions. There is also incomplete knowledge of the socio-economic and health effects on the human populations of the development of the oil and gas industry in the often remote areas. The recommendations in this section illustrate some of the areas in which research and other studies are needed to provide a better overall picture.

Research to improve technology
R25. The Arctic countries should facilitate and cooperate on research to improve technology in relation to oil and gas exploration and development. In particular, research into less impacting drilling and seismic technologies should be continued.

Oil spill clean-up
R26. Given the great difficulties encountered in responding to oil spills under Arctic conditions, research should continue into oil spill clean-up technology, and response strategies and techniques for Arctic waters, including spills on ice, under ice, and in broken ice.

Comparative studies of socio-economic effects
R27. To enhance understanding of socio-economic effects of Arctic oil and gas activities, it is recommended that future studies include the following:

- compilation of Arctic oil- and gas-related socio-economic statistics on a circumpolar basis;
- comparative studies of the effectiveness of socio-economic mitigation and opportunity measures; and
- comparative and case studies on the effects on access to and availability and quality of subsistence resources (e.g., fish, game, caribou/reindeer).

Human health
R28. To enable scientists and regulators to determine potential health effects, studies should be conducted on the exposure of the general human population in the Arctic to chemical elements and compounds released from oil and gas activities, and on the impacts of these substances on human health. Future quantitative risk assessments must include site-specific and population-specific study data.

R29. Given the lack of information in the peer-reviewed scientific literature on the environmental and human health impacts of oil and gas activities in Russia, which has the largest production and transportation of oil and gas in the Arctic, studies should be conducted on the impacts of oil and gas activities on human health in Russia and the results be made available for future assessments.

Contaminated sites (e.g., previous spill sites) and natural seeps
R30. Sites of accidental oil spills, as well as the sites of experimental studies in various regions of the Arctic during the 1970s and 1980s, should be revisited on a regular basis and monitored using modern detailed chemical and biological techniques to determine the fate, persistence, and long-term effects of toxicologically significant petroleum hydrocarbon components. Soils and plant species should also be monitored for evidence of toxic metabolic by-products from the

degradation of the hydrocarbons. Monitoring the recovery of these sites should include small mammals, birds, and any other species present.

R31. Natural seeps are sources of ongoing hydrocarbon inputs and thus provide valuable research sites. Such sites should be used for basic research on the transport and fate of hydrocarbons under Arctic conditions, as well as for research on biological effects and adaptations by local biological communities to long-term exposure. The metabolism of oil and formation of metabolic by-products, rates of transformation of hydrocarbons, and colonization by bacteria and other biota offer important insights into hydrocarbon chemistry and the fate and effects of hydrocarbons in the Arctic. Mechanisms of adaptation, if any, by local biological communities may offer insights for sites that may become contaminated chronically by the oil and gas industry.

R32. In relation to research on remediation and revegetation of oil-contaminated areas, follow-up studies should be conducted at many of the previously contaminated sites using new methods of analysis in order to provide information on the rates of degradation of the hydrocarbons and other industry-related contamination, with the ultimate aim of improving success in returning the sites to their natural state.

Behaviour and fate of oil in sea ice

R33. There is still limited information on the behaviour and fate of the many crude and refined oils and petroleum hydrocarbon components in ice-infested Arctic marine waters. Continued research on this topic should be high priority. Better knowledge is essential to improve assessments of the transport, fate and effects of spilled oil in ice-covered waters, including oil under ice carried by currents and oil drifting with the sea ice.

Exposure and toxicology

R34. There is little information available relating to the exposure of terrestrial birds and mammals to oilfield chemicals and releases from production sites, and most of the large amount of information on the effects of hydrocarbons on aquatic organisms concerns temperate or sub-Arctic species. Research is required on a wide range of the potential biological effects of these chemicals under conditions and with species and life stages appropriate to the Arctic, including, among others, studies of acute and chronic toxicity, genetic effects, and combined effects with, for example, exposure to sunlight. This includes studies of linkages between the diverse sub-lethal effects and the risks they pose to individuals and populations of Arctic animals.

Animal populations and ecosystems

R35. Fish populations are intermediate links between lower and higher trophic levels in aquatic ecosystems. Polar cod (*Boreogadus saida*) and Arctic cod (*Arctogadus glacialis*) are two species that play key roles in high Arctic marine ecosystems. There is need for research on the basic biology of these species, and their populations should be identified and mapped.

R36. There is a general need for better knowledge about population structure, both genetically and geographically, of Arctic mammals, birds, fish, and key invertebrate species. This is basic information required for better description and assessment of the status and vulnerability of the Arctic biota and ecosystems. Priority should be given to threatened species, subspecies, or populations, and to species that are important food resources to Arctic inhabitants.

R37. Further research is needed on the fundamental ecological interactions between Arctic species and the possible changes in these interactions as oil and gas development in the Arctic expands and as the ecosystem changes, particularly in response to a warming climate. Studies should be conducted to provide basic information on relationships and changes in key ecological processes, activities, and habitats, including features such as nesting, molting and staging waterfowl, shorebirds and seabirds, spawning areas for fish, calving or pupping and molting areas for marine mammals, migration corridors for fish and marine mammals, and changes in predatory species, prey, diseases, and parasites.

Sensitive areas

R38. Ecologically sensitive areas should be mapped, and oil spill trajectory models should be further improved and used to determine areas most at risk from oil spills. This would improve the basis for attempts to decrease or eliminate the probability of an oil spill affecting sensitive areas, for setting priorities for response strategies and deployments, as well as for helping to determine the shipping routes least at risk from spills.

Coordination of research

R39. This assessment has drawn on data and information from many national research programs in Arctic countries. Given the need for such data and information to be obtained on a comparable basis among the various participating organizations in the Arctic, so that results can be compared across the entire area, it is recommended that the coordination of research across the Arctic be improved to provide for the use of, where possible, common methodologies, species, and publication guidelines. This applies to research on contaminant levels and biological effects in terrestrial and aquatic species and the health of human populations. Furthermore, as new methods emerge, they should be calibrated against earlier methods if appropriate, so that comparisons may be made for data obtained over a long period of time. Statistically-based standards of analytical quality should be agreed upon and stated in reports.

Glossary

Σ16 USEPA PAHs	Benz[*a*]anthracene, benzo[*a*]pyrene, benzo[*b*]fluoranthene, benzo[*k*] fluoranthene, chrysene, dibenz[*a,h*] anthracene, indeno[1,2,3-*cd*]pyrene, acenapthylene, anthracene, benzo[*g,h,i*] perylene, fluoranthene, fluorene, naphthalene, phenanthrene and pyrene
ΣPAH	Sum of PAHs
•OH	Hydroxyl radical
$\bullet NO_3$	Nitrate radical
AHH	Aryl hydrocarbon hydroxylase
AMAP	Arctic Monitoring and Assessment Programme
BCF	Bioconcentration factor
Bq	Becquerel
BTEX	Benzene, toluene, ethylbenzene and xylene
BTX	Benzene, toluene, and xylene
CAD	Canadian dollars
CH_4	Methane
CO_2	Carbon dioxide
CPI	Carbon Preference Index
dB	Decibel
DOM	Dissolved organic matter
dw	Dry weight
EAC	Ecological Assessment Criterion
EC_{50}	Concentration causing 50% of the effect
EF	Emission Factor
ER	Estrogen receptor
EROD	Ethoxyresorufin O-deethylase
EU	European Union
FFPI	Fossil Fuel Pollution Index
GAD	Generalized anxiety disorder
GST	Glutathione-S-transferase
H_2S	Hydrogen sulphide
ha	Hectare
HPLC	High performance liquid chromatography
Hz	Hertz
IC_{50}	Inhibitory concentration for 50% of the population
KCl	Potassium chloride
K_{ow}	Octanol/water partition coefficient
LAC	Low-molecular weight aromatic compounds
LC_{50}	Concentration causing 50% mortality
LD_{50}	Dose causing 50% mortality
LOAEL	Lowest observed adverse effect level
LOEL	Lowest observable effect level
MFO	Mixed-function oxygenase
MPC	Maximum Permissible Concentration (Russia)
MROD	Methoxyresorufin O-deethylase
MW	Molecular weight
NAO	Nenets Autonomous Okrug
nm	Nautical mile
nmVOCs	Non-methane volatile organic compounds
NOEC	No observable effect concentration
NOEL	No observable effect level
NO_X	Nitrogen oxides
NPRA	National Petroleum Reserve - Alaska
OWD	Oil-water dispersion
Pa	Pascal
PAHs	Polycyclic aromatic hydrocarbons
PCBs	Polychlorinated biphenyls
PEL	Probable effect level
$PM_{2.5}$	Particulates of 2.5 μm or less
PNEC	predicted no effect concentration
ppb	Parts per billion
ppm	Parts per million
ppmv	Parts per million volume
ppt	Parts per thousand
PTSD	Post traumatic stress disorder
Ra	Radium
SO_2	Sulfur dioxide
sq ft	Square feet
T	Tonne (1000 kg)
TPH	Total petroleum hydrocarbon
TLm	Median tolerance limit
UV	Utraviolet
VOCs	Volatile organic chemicals
WAF	Water-accommodated fraction
w/o emulsion	Water-in-oil emulsion
WSF	Water-soluble fraction
ww	Wet weight
YNAO	Yamalo-Nenets Autonomous Okrug

Oil and Gas Industry Conversions

(after http://www.eppo.go.th/ref/UNIT-OIL.html)

Petroleum hydrocarbons, and the refined products made from crude oil are generally quantified either by volume or by weight. In the United States, the basic units of volume are (US) barrels or (US) gallons; and for weight, (metric) tonnes or US (short) tons. In other countries, the SI system is generally applied, with cubic metres (m^3) and (metric) tonnes as the most commonly used units for volume and weight, respectively. The relationship between volume and weight is usually determined by density (the alternative measures being relative density or specific gravity).

Oil equivalents (o.e.) are used to express quantities of oil and natural gas in units that can be combined / compared.

In the United States in particular, oilmen reckon quantities of oil produced, moved or processed in barrels per day (bpd or b/d). A loose but simple rule of thumb for conversion is that a barrel a day is roughly 50 tonnes a year, but the relationship varies according to density and so according to product.

Conversion factors for volumes

Base unit	Equivalent
1 cubic metre	1000 litres
	6.2898 (US) barrels
	264.17 (US) gallons
	219.97 Imperial gallons
	35.315 cubic feet
1 litre	0.001 cubic metres
	1000 cubic centimetres
	0.26417 US gallons
	0.035314 cubic feet
1 (US) barrel	0.15899 cubic metres
	158.984 litres
	42 (US) gallons
	34.9726 Imperial gallons
	5.6146 cubic feet
1 (US) gallon	0.0037854 cubic metres
	3.7854 litres
	0.133681 cubic feet
	0.0238095 (US) barrels
	0.83268 Imperial gallons
1 Imperial gallon	0.004561 cubic metres
	0.028594 (US) barrels
	1.20094 (US) gallons
	0.160544 cubic feet
1 cubic foot	0.028317 cubic metres
1 gross ton	100 cubic feet = 2.83 cubic metres of (shipping) permanently enclosed space

Conversion factors for weights

Base unit	Equivalent
1 metric tonne	1000 kilograms
	1.10231 US (short) tons
	0.98421 Imperial (long) tons
1 kilogram	0.001 (metric) tonnes
	2.20462 lbs (pounds)
1 US (short) ton	0.907186 metric tonnes
	0.892857 Imperial (long) tons
	2000 lbs (pounds)
1 Imperial (long) ton	1.01605 metric tonnes
	1.12 US (short) tons
	2205 lbs (pounds)

Conversions based on the assumption that all weights are weights in air, as used for computing bulk commercial quantities of petroleum.

Conversion factors for volume to/from weight

Base unit	Equivalent
1 cubic metre of oil	0.855 tonnes
1 barrel oil	0.136 tonnes
1 tonne oil	1.1696 cubic metre
	7.3529 barrels
1 barrel oil equivalent	1 barrel crude oil
	160 cubic metres gas
	5487 cubic feet gas (based on average energy equivalent of TOTAL gas reserves)
1 m^3 oil equivalent	1008 cubic metres of gas
	35600 cubic feet of gas
1 million standard cubic feet of natural gas	172.3 barrels crude oil equivalent
1 (US short) ton LNG	1.22 tonnes crude oil (energy equivalent)
	52300 standard cubic feet of gas

Conversion factors for flow rates

Base unit	Equivalent
Gas: 1 normal cubic metre per day	37.33 standard cubic feet per day
Oil: 1 barrel per day	approximately 50 tonnes crude oil per year

Liquefied methane

Base unit	Equivalent
1 ton of liquefied methane	approximately 16 barrels
	approximately 50 000 cubic feet (1400 cubic meters) of natural gas, depending on methane content

Product specific gravity ranges

	Specific gravity	Barrels per metric tonne
Crude oils	0.80 – 0.97	8.0 – 6.6
Aviation gasolines	0.70 – 0.78	9.1 – 8.2
Motor gasolines	0.71 – 0.79	9.0 – 8.1
Kerosines	0.78 – 0.84	8.2 – 7.6
Gas oils	0.82 – 0.90	7.8 – 7.1
Diesel oils	0.82 – 0.92	7.8 – 6.9
Lubricating oils	0.85 – 0.95	7.5 – 6.7
Fuel oils	0.92 – 0.99	6.9 – 6.5
Asphaltic bitumens	1.00 – 1.10	6.4 – 5.8

Interfuel conversion factor

While individual crude and gases vary widely in quality, certain standard qualities are often assumed for statistical purposes:

Reference fuel	Barrel of oil equivalent	Ton of oil equivalent	1000 cubic feet of natural gas
Calorific value	5.8×10^6 Btu gross	43×10^6 Btu gross	1×10^6 Btu gross
Conversion factors	1	0.14	5.8
	7.41	1	43.0
	0.17	0.02	1

Based on these qualities, the following equivalent rates of consumption can be used with reasonable accuracy:

Liquefied natural gas, tonnes per year	Natural gas, 10^6 million cubic feet per day, 10^9 million normal cubic metres per year	Oil, tons of oil equivalent per year	Oil, barrels of oil equivalent per year
1	1.41	1.22	25
0.71	1	0.87	18
0.82	1.15	1	20
0.04	0.056	0.049	1

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