Chapter 10

Petroleum Hydrocarbons

Contents

10.1. Reason for concern

Petroleum hydrocarbons have been detected in all abiotic and biotic compartments of the circumpolar Arctic environment. An evaluation of the effects of petroleum hydrocarbons in the Arctic needs to take into account various environmentally significant factors. These factors include the relative quantities of discharges, differences in single well and multi-well oil and gas exploratory and production activities, geographical extent of effects, potential for recovery, special environmental characteristics (e.g., low average temperature, low light intensity, ice cover, etc.), and presence of sensitive or vulnerable populations.

The extent and severity of ecological impacts and the potential and time scale for recovery depend on many factors. In principle, the ecological effects of petroleum hydrocarbons are similar in tropical, temperate, and Arctic environments for related or similar biological targets. The important environmental differences between ecosystem or climatic regions are those that affect the distribution, composition, physical state, and fate of the petroleum hydrocarbons. In the Arctic, the rate of biological recovery from oil damage is considered to be slower than in more temperate regions due to slower growth rates, shorter growing seasons, higher generation turnover times, and higher ages at maturity. The overall implication is that for similar events, the effects of oil pollution may be more severe and persistent in the Arctic environment than in other environments. Furthermore, experience has shown that it is generally invalid and unsuccessful simply to take the results of damage assessments or the approaches for cleaning up oil spills for regions outside the Arctic and apply these directly to Arctic situations. The environmental threats to the Arctic associated with oil and gas development, production, and transport are primarily local and/or regional and not circumpolar in scale. An important exception can occur however, for certain species of migratory animals which congregate within relatively small areas if the period of congregation overlaps with intense disturbances (e.g., large oil spills). In such cases, devastating impacts could occur at the population level.

There are various types of petroleum hydrocarbon pollutants that contribute to the overall hydrocarbon load into the Arctic environment. For the purposes of this report, the primary categories of petroleum hydrocarbon pollutants to be discussed include crude oil, refined oil products, polynuclear aromatic hydrocarbons (PAHs), and natural gas and condensates. This discussion will be focused on the petroleum hydrocarbons as they occur and affect Arctic ecosystems. It will not address the effects that the development and use of Arctic petroleum hydrocarbon can have both outside the Arctic and on global environmental issues such as climate change and global warming. Some petroleum hydrocarbons and the products of combustion of petroleum hydrocarbons, (e.g. carbon dioxide), can play a substantial role in such changes. For example, methane, the simplest petroleum hydrocarbon, is also a greenhouse gas. Its atmospheric concentration is increasing as a result of human activities related primarily to biospheric impacts (e.g., melting of permafrost) and fossil fuel production and use. Production of oil in the Arctic and its refining and use will contribute to the global release of methane. Further information concerning climate change aspects of hydrocarbon pollution can be found in IPCC (1995).

A special focus of this report is on the petroleum hydrocarbon resources (Table 10·1) of the Arctic environment. Major projects for development of oil and gas resources in the Arctic are currently underway or planned for the near future in the Barents and Kara Seas of Russia and the North Slope of Alaska (Figure 10·1). It appears likely that the first substantive offshore development in the Arctic will take place in the Barents/Pechora Sea region of the Russian Federation. Taking into account the vulnerability of the Arctic environment, these developments could increase the poten-

Table 10·1. Estimates of Arctic proved reserves; yearly production; and undiscovered, conventionally recoverable oil.

Country	Proved reserves ($\times 10^6$ m^3)	Yearly production ($\times 10^6$ m^3)	Undiscovered, conventionally recoverable oil ($\times 10^6$ m^3)
Canada	21 (1984)	1.3 (1995)	1701 (1994)
United States	915[b] (1994)	90[b] (1994)	3630[b] (1995)
Russia (Tyumen Oblast only)[a]	15700[c] (1994)	274[d] (1993)	
Nordic countries	440[e] (1994)	2[e] (1994)	695[e] (1994)

a. The Russian method of estimating reserves is quite different from that used in Western countries. Direct comparisons with Western classifications is impossible. As a rule-of-thumb Western assessments of recoverable reserves are half as large as the Russian assessments. (Kryukov *et al.* 1996).
b. Sherwood, Craig, and Cooke 1996. c. Kryukov *et al.* 1996.
d. Anon. 1994. e. NPD 1995.

tial for large-scale releases of oil with associated risks to Arctic ecosystems. Primary concerns associated with major new developments involve the increased risks of accidents and the increased difficulties of taking remedial measures in such severe environments. Other major concerns relate to the potential effects of spilled oil on commercial fisheries and the loss or alteration of habitats. This may result in economic consequences (e.g., closure of fishing areas and the tainting of seafood). The Arctic part of the Norwegian and Russian fishing areas is highly productive with an annual harvest of approximately 2-3.5 million tonnes of fish from the Barents Sea alone (Institute of Marine Research, Norway, pers. comm.).

Table 10·2 provides an overview of the types of activities associated with various phases of oil and gas developments, the corresponding types of chemicals used and pollution

Figure 10·1. Major areas of oil and gas development and potential development in the Arctic, and major shipping routes and possible new routes through Arctic waters.

Table 10·2. Oil-related environmental impact.

Activity	Type of pollution	Main chemicals	Sites affected	Potential effects targets
Exploration phase				
Rigging	Physical disturbance, noise, physical presence	None	Locally on site and along transport routes	Soils, permafrost stability, bottom sediments, vegetation, fauna, behavioral patterns
Seismics	Physical disturbance, noise	None	Locally on site	Aquatic organisms (e.g., fish larvae, mammals)
Exploratory drilling	Discharges of drill cuttings and chemicals	Water-based drilling fluids, anti-corrosion agents, scale inhibitors, cementing agents, completion chemicals, and others	Locally to regionally	Soil and sediment contamination levels, vegetation, bottom and near-bottom fauna, amenities and other environmental usage
Accidental spills (blow-outs)	Oil discharge	Hydrocarbons, dispersants	Local (on land) to long-range (rivers, lakes, and sea)	Contamination levels (soils, snow, surface waters, ice, sediments), vegetation and fauna, amenity values, and tourism
Construction phase				
Removal of vegetation	Physical disturbance, noise	None	Locally on site	Habitat diversity, quality and availability; erosion, permafrost stability, peat removal; animal behavior
Technical installations	Physical disturbance, physical presence	None	Locally on site	Habitat quality and access, permafrost stability
Excavation and infill of soils and sediments	Physical disturbance	None	On-site soils and downstream surface- and groundwater	Water courses and drainage patterns, ground- and surface-water, soil and sediment organisms
Road/trail construction	Physical disturbance, noise, physical presence	None	Locally	Access, migration routes, erosion, vegetation, animal behavior
Use of helicopters and supply vessels	Noise, exhaust discharge	Combustion products	Along routes	Contamination levels of water, soils and organisms, biotope quality, behavioral pattern
Dredging and construction of pipelines	Physical disturbance, noise, physical presence	None	Pipeline route and adjacent areas	Soils, bottom sediments, vegetation, fauna, behavioral patterns (migration)
Production phase				
Well drilling	Discharges of drill cuttings and chemicals	Drilling fluids, anti-corrosion agents, scale inhibitors, cementing agents, completion chemicals, and others	Locally to regionally	Soil and sediment contamination levels, land access, vegetation, bottom and near-bottom fauna
Well production	Discharge of production water and chemicals	Production water, scale inhibitors, flocculant agents, biocides, anti-corrosion agents, gas treatment chemicals	Local soils, local/regional surface- and groundwater, surface and shallow seawater, possibly sea floor	Contamination level of soil and waters, vegetation, land fauna and marine pelagic organisms
Other operational aqueous waste effluents	Wash and drainage water, ballast water, sanitary outlets, operation spills and leakages	Hydrocarbons, chemicals, sewage	Soils, local watersheds, shallow seawater	Contaminant levels, water vegetation and fauna, marine pelagic community, waterfowl and seabirds
Flaring, venting and purging, energy production (combustion), fire protection tests, exhaust and dust, loss of fugitive gases	Air emissions	CO_2 and CO, methane, VOC, NO_x, SO_2 and H_2S halons, ozone-depleters	Wide-range due to atmospheric transport	Greenhouse gas and ozone levels, soil, water, sediment and organism contaminant levels, human health, vegetation and fauna
Use of helicopters and supply vessels	Noise, exhaust discharge	Combustion products	Along routes	Contamination levels of water, soils and organisms, biotope quality, behavioral patterns
Accidental spills (well sites, pipelines, transport vehicles and vessels)	Oil discharge	Hydrocarbons dispersants	Local (on land) to long-range (rivers, lakes and sea) distribution	Contamination level (soils, snow, surface waters, ice, sediments), vegetation and fauna, amenity values, and tourism
Decommissioning phase				
Technical demobilization	Physical disturbance, noise	None	Locally on site and along transport routes	Soils, permafrost stability, bottom sediments, vegetation, fauna, behavioral patterns

outcomes, and the areas and environmental targets potentially affected. Although it is recognized that several of these activities (e.g., discharge of drilling fluids and other chemicals) may be important as threats to the Arctic environment in ways additional to their association with oil hydrocarbons, it is beyond the scope of this assessment to consider all of the activities listed in the table. The focus of this chapter will, therefore, be on oil hydrocarbons. Brief assessments are also made concerning drilling fluids and cuttings, and produced water, but with the emphasis on their role as sources for hydrocarbon pollution.

Polynuclear aromatic hydrocarbons (PAHs) are a special case because, in addition to being present in petroleum, they also have pyrogenic and biogenic sources and, on the basis of moderate degradability and high bioaccumulation potential, are often classified among the persistent organic pollutants. It was, however, decided that the assessment of PAHs, describing the levels and potential effects in the Arctic, was more appropriately included in this chapter than in chapter 6, which covers other persistent organic pollutants.

This report is not intended to be a literature review. Instead it is intended to provide an assessment focused on the major potential impacts from the discharge of petroleum hydrocarbons to the Arctic marine, freshwater, and terrestrial ecosystems. It includes conclusions and recommendations with regard to the reduction and prevention of these potential environmental impacts.

Section 10.2 of the report provides an overview of the regional development and transportation of petroleum resources in the Arctic; section 10.3 describes the sources of hydrocarbons; section 10.4 outlines the environmental transport and fate; sections 10.5 and 10.6 describe the levels of petroleum hydrocarbons and levels of PAHs in the Arctic environment; section 10.7 outlines environmental effects; and section 10.8 provides the conclusions and recommendations.

10.2. Regional development and transportation of petroleum resources

Exploration for oil and gas resources in recent years has indicated that the Arctic potentially contains some of the world's largest oil and gas reserves. These are located both on land and on the continental shelves. A review, by country, of the Arctic region's current and potential oil and gas activities as well as a brief discussion of environmental regulation related to controlling, monitoring, and mitigating the effects of these activities is presented below.

10.2.1. Canada

Significant hydrocarbon exploration has taken place in the Canadian Arctic, identifying considerable oil and gas reserves. Substantial reserves of gas and several small oil fields were discovered in the Sverdrup Basin as early as the 1960s, leading to considerable exploration activity over the next 20 years. Since 1980 a total of 438 hydrocarbon wells have been drilled within the Arctic ecosystem, mainly in, or offshore of, the Mackenzie Delta and in the central Arctic Islands. Of these, 315 were dry, while 123 produced some oil and/or gas in commercially non-viable quantities. Seventy of these wells were productive enough to be capped for future production.

There are no permanent production installations in the offshore in the Canadian Arctic. In the Arctic Islands, however, there was seasonal production from one well on the Bent Horn Oil Field. Each year since 1985, the Bent Horn Oil Field on Cameron Island in the High Arctic supplied oil for two to three loads of crude oil carried by a specially reinforced tanker to a refinery in Montreal. In 1993, the field produced 3.21×10^5 m^3 (2.02×10^6 barrels) of oil. Oil production at Bent Horn was, however, shut down in 1996.

The 239 exploratory wells (83 of these offshore) that were drilled in the Mackenzie Delta and adjacent nearshore Beaufort Sea (Figure 10·2) from 1969 to 1990 have resulted in about 50 productive wells on hydrocarbon fields estimated to contain $238\text{-}318 \times 10^6$ m^3 ($1.5\text{-}2.0 \times 10^9$ barrels) of oil and $0.29\text{-}0.36 \times 10^{18}$ m^3 ($10.4\text{-}12.6 \times 10^{18}$ ft^3) of gas. Although exploitation of these discoveries is considered uneconomical under present market conditions, a change in circumstances could lead to their development. Other significant reserves in the area include tar sands on northwest Melville Island, estimated to contain 17×10^6 m^3 (110×10^6 barrels) of oil, and large gas reserves nearby on Sabine Peninsula and in the offshore Hecla fields. Future development of these oil and gas reserves is dependent on world energy prices.

Oil seeps along the banks of the Mackenzie River have long been known. In 1986, construction of a 305 mm (12 inch) pipeline was completed from the Norman Wells Field on the Mackenzie River to Zama, Alberta. In 1993, production was 1.8×10^6 m^3 (11.3×10^6 barrels) with cumulative production of approximately 16×10^6 m^3 (100×10^6 barrels).

10.2.2. United States

The discovery of a major oil-producing well on Alaska's Beaufort Sea coast was announced in March 1968, and of another well, seven miles from the discovery well, in June 1968. This was the beginning of the Prudhoe Bay Oil Field, which turned out to be mammoth in size, containing over 3.1×10^9 m^3 (20×10^9 barrels) of oil (Figure 10·2). On 20 June, 1977, the Prudhoe Bay oil flowed for the first time through the 1300 km long Trans-Alaska Pipeline System (TAPS) to Port Valdez, a fjord in south-central Alaska, for shipment to markets in the contiguous United States (Hameedi 1988). In 1984, Prudhoe Bay produced 89.2×10^6 m^3 (562×10^6 barrels) of oil, and the nearby Kuparuk Oil Field produced 7.3×10^6 m^3 (46×10^6 barrels). The field's production began to decline in 1989, although a number of newly discovered oil fields and improved methods of oil recovery (for example, reinjection of natural gas into the ground at high pressure) have been successful in prolonging the life of the field. Most of the newly discovered oil fields are in the general vicinity of Prudhoe Bay. According to an estimate, over 1.2×10^9 m^3 (7.5×10^9 barrels) of oil is still commercially recoverable from the area.

The total amount of potentially recoverable hydrocarbons, including petroleum, natural gas, and natural gas liquids, for the US Arctic was estimated to be 6.9×10^9 m^3 (43.6×10^9 barrels) of oil equivalents (5.6×10^{12} ft^3 of natural gas is equivalent to 1×10^9 barrels of oil) (NPC 1981).

The average amount of undiscovered recoverable oil in the National Petroleum Reserve - Alaska (NPRA) was estimated at 0.95×10^9 m^3 (5.97×10^9 barrels) of oil and 0.32×10^{12} m^3 (11.3×10^{12} ft^3) of natural gas. Oil in the Arctic National Wildlife Refuge (ANWR) region was estimated at 0.37×10^9 m^3 (2.3×10^9 barrels), together with 0.22×10^{12} m^3 (7.8×10^{12} ft^3) of natural gas (NPC 1981). No petroleum development has taken place in either region to date.

Over the years, leasing schedules in the US Arctic have included subareas of the Beaufort, Chukchi, and Bering Seas.

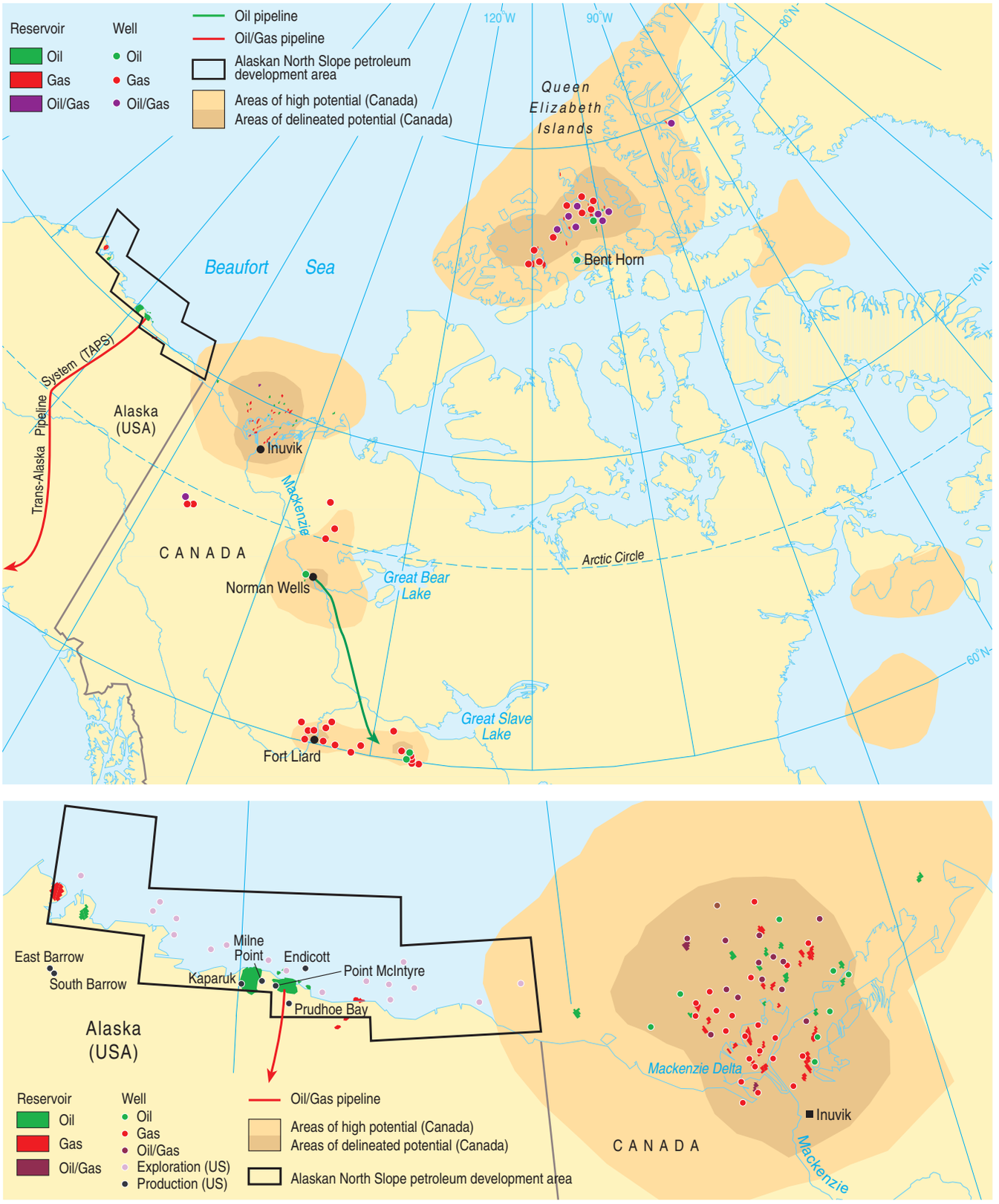

Figure 10-2. Major oil and gas development areas in Arctic Canada and on the Alaskan Beaufort Sea coast.

A number of lease sales have been held in the region since 1979, although to date no major oil discoveries have been reported. Endicott Field, 17 km northeast of Prudhoe Bay, is the only offshore producing field in US Arctic waters.

Current estimates of expected petroleum production from Alaskan offshore waters suggest a range of 0.08-0.43 10^9 m^3 (0.50-2.70 $\times 10^9$ barrels) for the Beaufort Sea and 0.05-0.24 $\times 10^9$ m^3 (0.30-1.50 $\times 10^9$ barrels) for the combined Chukchi Sea-Hope Basin region (MMS 1996). Commercial development of natural gas is not anticipated in the region under the current pricing and supply-demand scenarios.

10.2.3. **Russia**

Oil and gas activities in the marine and terrestrial environments of the Russian Federation's part of the Arctic are concentrated in the Nenets Autonomous District of the Arkhangelsk region, in the Komi Republic within the limits of the

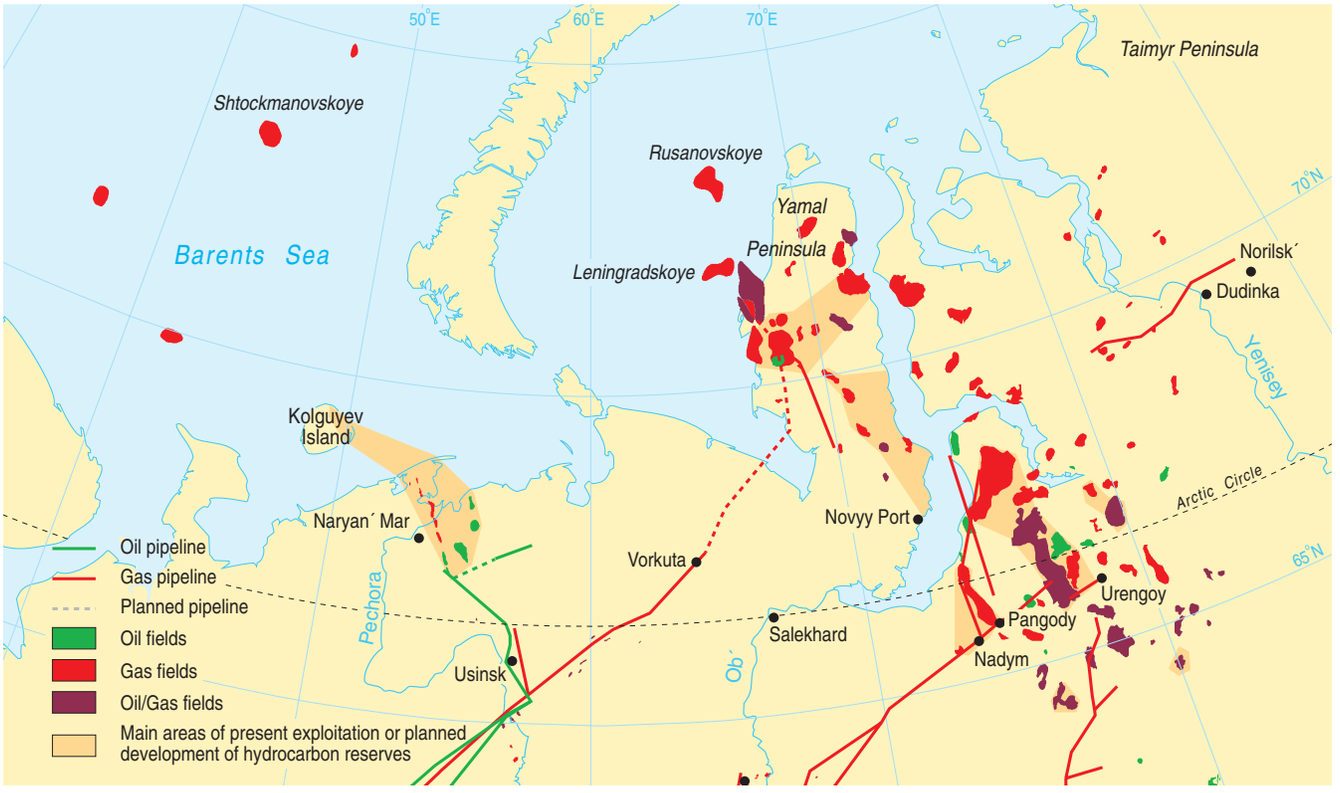

Figure 10·3. Major oil and gas development and potential development areas in Arctic Russia and the Barents Sea region.

Pechora River drainage, and in the Yamal-Nenets Autonomous District in Western Siberia (Figure 10·3). The estimated total annual volumes of the oil and gas production of the eighteen largest Russian companies and enterprises located in these three regions are 93×10^6 tonnes of crude oil, 742×10^{12} m^3 of natural gas, 3.4×10^{12} m^3 of casing-head gas, and 2×10^6 tonnes of gasoline.

Large oil and gas fields have been discovered in the Russian Arctic shelves of the Barents, Kara, and Pechora Seas. For example, the gas reserves of the Shtockmanovskoye Gas Field in the Barents Sea are estimated at approximately 2500×10^{12} m^3 and even larger fields have been discovered in the Kara Sea (i.e., Rusanovskoye and Leningradskoye Fields). The most promising oil field in the Pechora Sea is Prirazlomnoye in the southern part not far from shore.

The largest oil and gas company in the Russian Arctic is the Noyabrskneftegaz Industrial Association of Rosneftegaz Corporation, which annually produces $\leq 30 \times 10^6$ tonnes of crude oil and 190×10^{12} m^3 of natural gas. Other major producers are Kholmogorneft, OAO Rosneft-Purneftegaz, and Satorminskneft Production Administration with annual productions of 7×10^6, 10.7×10^6, and 10.6×10^6 tonnes of crude oil and 509×10^6, 1656×10^6, and 460×10^6 m^3 of casing-head gas, respectively; and OAO Surgutneftegaz with an annual production of $\approx 17 \times 10^6$ tonnes of crude oil. Major gas producers are Urengoygazprom, Yamburggazodobycha, and Nadymgazprom Stock Companies with annual productions of 290.2×10^{12}, 166.5×10^{12}, and 70.3×10^{12} m^3 of natural gas, respectively. The gas production industry on the Taimyr Peninsula is represented by Norilskgazprom Industrial Association, but its annual production is not significant. There are practically no oil and gas activities in other parts of the Russian Arctic. Oil and gas is transported from the extraction fields to processing plants by pipelines, most of which have exceeded their design life. At present, there is no significant transportation of oil and gas products by tankers.

10.2.4. Nordic countries

Significant oil and gas exploration activities in the Arctic marine areas of the Nordic countries occur along the Norwegian continental shelf and in the Barents Sea. Sweden and Finland have no Arctic coastline and do not carry out any oil and gas activities in the Arctic. Iceland has not conducted any exploration and does not produce any oil or gas. It imports all its oil via tankers. In Greenland, only exploration activities have been carried out. Four slim-hole wells were drilled during the summer of 1995. As a result of this drilling, which showed traces of oil and gas, an onshore exploratory well was drilled during the summer of 1996 on the Nuussuaq Peninsula on the west coast of Greenland.

For petroleum activity purposes, the Norwegian continental shelf is divided into three sectors: the North Sea, the Norwegian Sea (area between 62°N and 69°30'N), and the Barents Sea (area north of 69°30'N and west of 30°30'E), of which the two latter are within the Arctic (Figure 10·4). By 1995, a total of 107 and 53 wells had been drilled in the Norwegian and Barents Sea sectors, respectively, including 17 onshore wells around Svalbard.

In the Norwegian Sea sector, several oil and gas discoveries have been made, comprising a total estimated reserve of 420×10^6 m^3 of oil and natural gas liquids and 350×10^6 m^3 of gas (NPD 1995). Two fields, Draugen and Heidrun, are in production at present and five more are being developed, all between 64°N and 66°N. NPD (1995) estimates that the expected undiscovered resources in the Norwegian Sea range between 330×10^6 and 3330×10^6 m^3 oil equivalents, of which about two-thirds are expected to be gas. Large new areas down to about 2000 m depth were opened for exploration activities in 1994 and 1995.

In the Barents Sea, the first discovery was made in 1982, and up to 1996 a total reserve of 20×10^6 m^3 of oil and natural gas liquids and 285×10^6 m^3 of gas has been discovered. Most of the discoveries have been made in the Hammerfest Basin in the southwest part of the Barents Sea. None

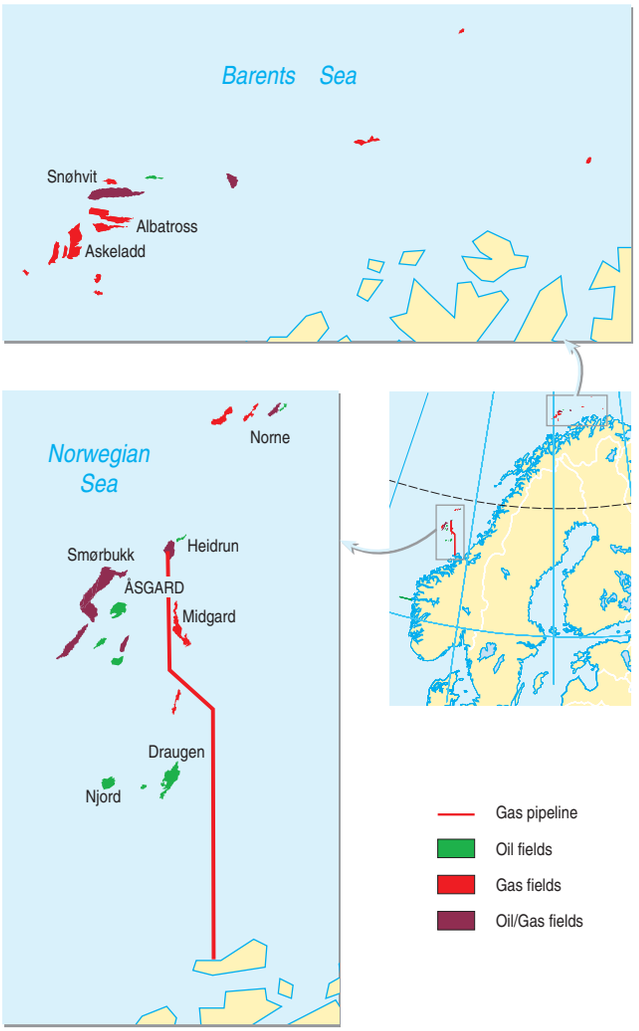

Figure 10·4. Oil and gas development areas in the Norwegian Sea and Norwegian Barents Sea regions.

of the discoveries has yet been considered profitable to develop, and the interest in the Barents Sea region has decreased in recent years. The expected undiscovered reserves, mostly in the southern part of the Barents Sea, range from 295×10^6 to 1995×10^6 m^3 oil equivalents, again with about two-thirds expected to be gas. The sparsely studied areas of the northern Barents Sea are expected to have significant potential for gas and moderate potential for oil (NPD 1995).

No onshore petroleum installations are yet in operation in the Norwegian Arctic, but one gas terminal linked to the petrochemical industry is under construction at Tjeldbergodden outside Trondheim. The terminal will receive gas, and possibly condensate, from the Norwegian Sea petroleum fields.

10.2.5. Environmental regulation related to Arctic oil and gas activities

Many legal instruments exist for Arctic countries to control, monitor, and mitigate the effects of oil and gas activities on the Arctic environment. The Protection of the Arctic Marine Environment (PAME) component of the Arctic Environmental Protection Strategy (AEPS) has produced a review of these in the PAME (1996) report and is producing Guidelines for Environmental Protection of Oil and Gas Exploration and Production in Arctic and Subarctic Offshore Regions.

Compliance with existing legal instruments, rather than the development of new ones, appears to be the main issue in the Arctic. Cooperation among the eight Arctic countries is needed to ensure that the integration of environmental protection measures into the design, construction, and operation of oil and gas exploration and production facilities takes place. Only through such cooperation will the risks of oil spill incidents, such as the Usinsk pipeline leakage and rupture which released large quantities of oil into the Komi Republic/Pechora River area of the Russian Arctic environment, be reduced.

10.3. Sources of hydrocarbons and related pollutants to the Arctic

Sources of petroleum hydrocarbons in the Arctic are both natural and anthropogenic, Table 10·3. Oil seeps from geological formations are the primary natural source. Anthropogenic sources covered in this review include chronic discharges from oil and gas development and production activities, accidental events such as oil spills from the rupture of pipelines, discharges from tankers and other ships along major routes, and atmospheric deposition. These sources vary in importance geographically across the Arctic.

It is recognized that land-based discharges and runoff have become a major source of petroleum hydrocarbons to the global marine environment (GESAMP 1993). Although there may be land-based sources of petroleum hydrocarbons to the Arctic environment, as indicated in Table 10·3, it has been difficult to obtain enough information to assess their relative importance. They are, therefore, not discussed further in this report. A brief description of the other sources of petroleum hydrocarbons to the Arctic is provided in the following sections.

Table 10·3. Sources of petroleum hydrocarbons to the Arctic environment.

Natural sources	Oil seeps. Biogenic synthesis.
Water movements	Inflow of ocean currents. Northward flowing rivers.
Atmospheric flow and deposition	Air movements from subarctic and temperate areas.
Gas and oil production	Operational discharges. Blow-outs.
Transport of crude and refined gas and oil	Accidents from tankers at sea. Tanker ballast washings. Leakage/spillage from pipelines and tanker trucks on land.
Transportation	Discharges and spills from vessels. Emissions and leakage from vehicles.
Land-based discharges and runoff	Refineries. Municipal waste water. Industrial waste water. Urban runoff. Combustion of wood and fossil fuels.

10.3.1. Natural oil seeps

Both active and inactive natural oil seeps are known to occur in the Arctic, some of them having been recognized since prehistoric times. In many instances, the presence of oil seeps has led to the discovery of commercially recoverable petroleum resources (Hunt 1979). In many other instances, this has not been the case. For example, numerous seeps in the Gulf of Alaska have prompted considerable exploration, but to date no significant petroleum discovery has been announced. Given estimates of the petroleum available for seepage over geological time scales, a reasonable world-wide seepage rate is thought to be between 0.02 and 2.0×10^6 tonnes per annum, with a best estimate of 0.2×10^6 tonnes per annum (NRC 1985).

The presence of petroleum seeps has been documented in many areas of the Arctic including the United States, Canada, and Russia. In many cases natural oil seeps originate in north-

ern-flowing rivers (e.g., Mackenzie River and the Ob River) which eventually discharge the oil into the Arctic Ocean.

The presence of hydrocarbon seepage must have been known to local inhabitants for centuries as the earliest scientific explorations of the Arctic regions describe local use of lumps and pebbles of oil shale and seepage tar by the Inuit. Of the total amount of oil per year entering the marine environment from all sources, at least 15% comes from natural oil seeps (GESAMP 1993). Estimates are not available for the Arctic region, but the rate of seepage is thought to be greater than the global average.

For example, the Mackenzie River in the Canadian Arctic contributes the largest quantities of hydrocarbons to the Beaufort Sea region. Oil seeps have also been detected in eight areas of the US Arctic, seven of which are located along the Beaufort Sea coast. Similarly, oil seeps in the Barents Sea (Doré 1995) and near Spitsbergen have been found.

10.3.2. Exploration and production activities

Oil and gas exploration and production activities can be a major source of petroleum hydrocarbons to the Arctic environment. There are various waste discharges associated with oil and gas activities; however, the major discharges to the Arctic environment are from drilling muds and cuttings, and produced water. These are described below.

10.3.2.1. Drilling muds and cuttings

Specially formulated drilling muds are used to lubricate the drill bit, control pressure in the well, seal the strata until casings are in place, support the bore hole walls, and carry drill cuttings to the surface. These drilling muds contain a variety of substances in a freshwater, seawater, or oil base, including special clays and inorganic and organic compounds, some of which are toxic to biota.

The most commonly employed drilling fluids are composed of weighting and well-stabilizing chemicals, and are used with water as the main liquid matrix. Deviated well drilling and situations with a high demand for well stabilization and lubrication require the use of oil-based drilling fluids in certain (deeper) sections of the well. Up to the early 1980s, diesel oil was the most common base fluid in oil-based drilling muds, but was then gradually replaced by low-aromatic mineral oils to reduce the environmental impact of disposing of mud and rock cuttings. More recently, fluids containing other organic base liquids such as olefins, esters, ethers, and biogenic oils, the so-called synthetic oil fluids, have partially replaced oil-based fluids, primarily in offshore drilling (SFT 1995b).

At a given well, about a dozen of the approximately 55 substances that are mixed to formulate more than 500 commercial mud products are used in drilling. A particular make-up is determined by site-specific considerations and is usually proprietary in nature. Barite, bentonite clays, caustic soda, and lignosulfonates are the primary components of water-based drilling muds. In addition, several metals (e.g., chromium, cadmium, lead, nickel, and zinc) are found in drilling muds. These are either added as metal salts or organometallic compounds, or are present as trace contaminants of major drilling-mud ingredients, such as barite and bentonite (NRC 1983).

Drill cuttings, consisting of chipped and pulverized sediment and rock, have usually been deposited on land, dumped directly into the water near rigs and platforms, or reinjected in the geological formations. The amount of drill cuttings produced is related to the drilling depth. After use, drilling muds and cuttings are separated. The muds, particularly the oil-based muds, are to some extent reused, but substantial amounts are often discharged together with the cuttings. Normal industry practice also involves routine discharges and periodic 'dumps' when the mud formulation is changed or when the end of the drilling operation is reached. Drill cuttings are heavier and coarser than drilling muds and most of the cuttings material settles out faster when discharged. In the water and on the sea bed, these cuttings are either dispersed and carried away or accumulate in place. If several wells are drilled from a single platform, as is the case for many deep-water offshore fields, large accumulations result, particularly in areas of weak water circulation.

Table 10·4 gives a summary of approximate amounts of drill muds and cuttings to be handled as waste during offshore drilling activities when oil-based muds are used. Today, oil-based cuttings are either reinjected or brought ashore for treatment, and according to SFT (1995b), no discharge has been reported from the Norwegian shelf since 1993.

Table 10·4. Typical quantities of drilling wastes discharged during offshore oil and gas exploration and production activities. (Modified from GESAMP 1993).

	Approximate average amount, tonnes per well
Exploration sites	
Drilling mud	
– periodically	200-2000[c]
– bulk at end	15-30[a] 150-400[a]
Cuttings (dry mass)	200-1000[a] 20-1000[d]
Base oil on cuttings (if oil-base mud is used)	30-120[b] 10-20[d]
Production site	
Drilling mud	900[c]
Cuttings	1000[c] 700[d]
Base oil on cuttings (if oil-base mud is used)	280[d]

a. GESAMP (1993).
b. Actual loss to environment may be higher (Chénard *et al.* 1989).
c. Neff *et al.* (1987).
d. Norwegian Shelf 1994 (SFT 1995b).

Oil and gas exploration and production wells are also drilled on land in the Arctic. The types of drilling muds and mud additives used in land drilling operations are generally similar to those used in offshore drilling. In northern Canada, for example, three types of mud systems have been used (Hardy BBT Limited and Stanley Associates Engineering Ltd. 1988). Freshwater-based muds are commonly used in holes less than 2000 m deep or in the upper 2000 m of deeper holes. Brine-based muds are usually used for drilling the portion of wells below 2000 m. Oil-based mud systems are suited for use in extremely water-sensitive shale formations, in deep salt formations, and in abnormally high pressure formations. Drilling wastes (muds plus cuttings and additives) are generally generated at the rate of approximately 0.8-1.3 m^3 per meter of hole drilled (upper 1500 m) and 1.3 m^3 per meter of hole drilled below 1500 m.

The main difference between offshore and onshore drilling practice with regard to drilling wastes is the method of disposal. On land, wastes are usually discharged to sumps. The efficiency of containment in these sumps varies widely due to differences in climatic conditions and permafrost stability. Loss of containment has been relatively common, resulting in contamination of groundwater, vegetation, soil, and biota with chemical contaminants (including hydrocar-

bons). The extent of contaminant impacts is usually local in scale (hundreds of meters) around sump locations (Hardy BBT Limited and Stanley Associates Engineering Ltd. 1988)

In conclusion, the environmental consequences of present and future drilling activities should be generally under control through strict regulations and improved waste handling technology. Efforts to reduce the cost of developing a petroleum reservoir have also resulted in recent changes in the drilling practice influencing the amount and type of waste material to be handled. At present, exploratory wells are drilled for the purpose of later production or injection, with the result that there is no distinction between exploration and production wells. As fewer wells are drilled in each field, the total amounts of drill muds and cuttings produced are reduced. Further waste reduction is achieved by drilling slimmer wells and restricting the use of oil-based or synthetic drilling fluids to as few sections of the wells as possible.

10.3.2.2. **Produced water**

Petroleum and natural gas reservoirs usually contain large quantities of water that are extracted together with the oil and gas. This produced or formation water must be separated from the hydrocarbons before they can be processed further. The proportion of produced water to crude oil or natural gas varies greatly, depending on the relative amounts in the reservoir, and will typically increase as the field is depleted. For Norwegian offshore fields a total of 33.5×10^6 m^3 of water was produced in 1994, which is about 2% by volume of the total oil and gas production in the same year (SFT 1995b, NPD 1995). It is estimated that the annual volume of produced water from Norwegian fields will increase to about 90×10^6 m^3 in the year 2000 (OLF 1991). The Trading Bay facility, which receives output from offshore oil platforms in Cook Inlet, Alaska, generates 10 650 m^3 (67 000 barrels) of oil and 9850 m^3 (62 000 barrels) of water each day. For US offshore continental shelf areas in general, it is assumed that the volume of produced water over the life of a field will be equal to 20-150% of the oil output volume (MMS 1992). For several US and North Sea fields, the produced water is reinjected into the reservoirs to facilitate enhanced petroleum recovery.

Produced water contains brine, sometimes with concentrations of total dissolved solids in excess of 300 g/L. The ionic composition generally corresponds to that of seawater, although it may vary considerably with geological formation (Collins 1975). Produced water often contains little or no dissolved oxygen and is enriched with sulfur. Produced waters also often contain small amounts of radionuclides, primarily isotopes of radium (^{226}Ra and ^{228}Ra), which are usually derived from uranium and thorium which are associated with the clay minerals and quartz sands that make up the matrix of the hydrocarbon reservoir (Reid 1983). Produced water may, with time, contain seawater and chemicals circulated downhole for well enhancement or added during the oil/water separation process.

Produced water contains substantial quantities of hydrocarbons and is treated prior to discharge to comply with regulatory limits. In the United States, current regulations require that petroleum hydrocarbon concentrations in water not exceed 72 mg/L for any one-day period or 40 mg/L for an average 30-day period. A similar regulatory limit, not to exceed 40 mg/L dispersed hydrocarbons, is in force on the Norwegian shelf. Table 10·5 provides data illustrating the petroleum hydrocarbon concentrations found in produced water effluents based on results from several Norwegian and US fields.

Table 10·5. Concentration of petroleum hydrocarbons (mg/L) in produced water effluents from Norwegian and US fields.

Site	Average concentration of hydrocarbons	Reference
Gulf of Mexico	20.6	Middleditch 1981
	20.3	Sauer 1981
Norwegian Sea (1994)	23.9	SFT 1995b
Total Norwegian shelf	24.1	SFT 1995b
Cook Inlet[a]	6.8-12.5	Boehm *et al.* 1985

a. Sum of volatile aliphatic and aromatic hydrocarbons.

Davies *et al.* (1989) show that produced water accounts for about 10% of the oil discharged by UK offshore oil exploration and production. On the Norwegian shelf, the contribution from produced water has increased from 20% of the total operational and accidental oil inputs to the sea in 1990 to 76% in 1995 (SFT 1995b). The proportion of the total oil inputs due to produced water will likely increase even further in the future for Norwegian shelf developments (OLF 1991).

10.3.3. **Oil spills**

Blow-outs, spills, and leakage during development and transportation of petroleum reserves and products pose the largest oil pollution threat to terrestrial and aquatic ecosystems in the Arctic. Pipeline ruptures and leakages, such as experienced at Usinsk, Russia in 1994-95 (Melnikov and Vlasov 1994, SFT 1995a) with 100 000 tonnes of crude oil flooding the rivers and lakes; and tanker accidents, like that of the *Exxon Valdez* in Alaska in 1989 with about 35 000 tonnes spilled, are examples of oil contamination over large areas. Oil blowouts at production sites have not yet been experienced in the Arctic.

Most oil spills are small to insignificant. For example, the 356 accidents reported in 1994 at Norwegian offshore installations spilled only 55 tonnes of oil in total, and only seven of these incidents had releases exceeding 1 m^3 (SFT 1995b). Between 1991 and 1994, Norway reported only three spills north of 70°N which exceeded 1 m^3. The total volume spilled was 20.3 m^3. In the US Arctic, no spills larger than 1 m^3 were reported during this time (PAME 1996). In the US Gulf of Mexico fields, 0.00024% of the oil produced during 1971-1978 was spilled (NRC 1985). Nevertheless, it is the rare, difficult to predict events that become environmental calamities.

To calculate the probabilities of blowouts and spills, it is assumed that future accidents can be predicted from historical data, that accidents occur independently of each other, and that accident rates depend on the volume of oil produced and transported (and indirectly on the duration of activities). Because only very limited offshore production has occurred in the Arctic, there is not presently an adequate data base to provide reliable accident rate statistics. In the United States, most such statistics are based on data from the oil platforms in the Gulf of Mexico and on tanker spills (Anderson and LaBelle 1990). These statistics must be applied with caution to the Arctic where ice provides an additional hazard. In particular, pressure ridges, which commonly scour the bottom of the inner shelf in water depths of 10-30 m, can impact platforms and sub-sea pipelines. While less commonly present in areas of potential oil development, tabular ice and icebergs have the potential to do even greater damage if they collide with bottom structures. The difficulty in carrying out mitigative procedures in the Arctic, such as recovering spilled oil or drilling relief wells, must also be considered.

Using the statistics described above and current petroleum resource estimates, it can be estimated that the number of oil spills equal to or larger than 1000 barrels (approx. 160 m^3) in the Beaufort and Chukchi Seas will be between 1 and 8 (with the probability for occurrence of one or more spills being between 58 and 99%), and that the number of spills exceeding 10 000 barrels (approx. 1600 m^3) will be between 0.3 and 2.5 (with the probability of one or more spills being between 24 and 92%). The sources of such spills are predicted to be platforms (16%), pipelines (46%), and tankers (38%). Some of the spills from tankers carrying oil produced in the Beaufort and Chukchi Seas are assumed to occur outside of the Arctic (MMS 1996).

10.3.4. Shipping

At present, shipping in Arctic waters is mostly associated with sealifts to isolated communities and industrial facilities, ice-breaker support for sealift traffic, fishing, seismic and oceanographic research, and supply of offshore drilling operations. Other types of Arctic shipping traffic include tourist ships, transport of ore concentrates from mining operations, and the transportation of crude oil by tankers from oil fields.

Shipping inevitably involves some releases of petroleum hydrocarbons to the environment. GESAMP (1993) indicates that the primary sources of petroleum hydrocarbon inputs from marine shipping on a world-wide basis are discharges from fuel oil sludges and machinery-space bilges and from oil tanker operations as well as from accidental spills discussed in the previous section. Among these, tanker accidents are the best known source of oil pollution; and, although such accidents contribute only a small percentage of the overall inputs of oil to the sea worldwide, they remain the focus of much public attention. The consequences of a tanker accident can be severe to the immediate area, particularly if the tanker is large and the spill occurs close to shore (GESAMP 1993, Spies *et al.* 1996). However, as oil tanker traffic is currently very limited in the Arctic, it seems likely that bilge and fuel oil discharges from fishing and other nontanker ships are the primary marine shipping-related sources of petroleum hydrocarbon inputs to the Arctic.

A main focus for concern regarding releases of oil to the Arctic region from shipping involves the Northern Sea Route (NSR), the system of sea lanes north of Asia, between the straits joining the Barents and Kara Seas in the west and the Bering Strait in the east (Figure 10·1). The former Soviet Union officially opened the NSR for international shipping in 1987. At present the NSR with its extension into the Barents Sea carries much more traffic than any other sea route through the Arctic Ocean and its adjacent seas. Even so, however, inputs of oil hydrocarbons along this route seem to be relatively minor with estimates of the annual releases, primarily from illegal dumping of bilge waters and accidental fuel spills, varying from 15 to 1500 tonnes, depending on the year (Melnikov *et al.* 1996). PAME (1996) estimated the legal discharges of oil from ships using the NSR to be about 1.4 tonnes/y, most of this from ordinary carriers. With significant prospects for offshore oil and gas in the Kara and Barents Seas, future expansion of ship and tanker traffic through the NSR is likely, thus increasing the probability of major tanker or other shipping-related spills.

To date, oil in the Russian Arctic has been transported primarily through pipelines. During the 1970s and 1980s an extensive pipeline network, including six trunk oil pipelines stretching over 10 000 km across Western Siberia, was constructed to transport oil and condensates from all areas currently under production and is capable of delivering approximately 400×10^6 tonnes of oil annually beyond regional boundaries. However, this current system of transport may not be sufficient in the future (Kryukov *et al.* 1996). For the northern Siberia region (particularly the northern Ob-Yenisey region) which fringes the Arctic Ocean, the natural alternative is to make use of the Northern Sea Route. The route may offer new markets in the East as well as in the West (Kryukov *et al.* 1996). Under the International Northern Sea Route Programme (INSROP), a five-year research program began in 1993 and is being coordinated by Russia, Norway, and Japan. An assessment of the feasibility of using the NSR is currently being conducted.

10.3.5. Local land-based discharges and runoff

Local pollution, connected with municipal and industrial wastewater discharges and urban surface runoff, is especially important in areas with relatively high population density and economic activity. Poor infrastructure development in Russian Arctic towns and industrial complexes, including a lack of wastewater treatment facilities, leads to significant discharges of polluted effluents into coastal areas of the Arctic Sea and the lower reaches of some Arctic rivers (Table 10·6). The largest Arctic city, Murmansk, on the Kola Peninsula, with a population of 500 000, has no municipal treatment facilities at all, and annually discharges

Table 10·6. Wastewater discharge into the coastal zones of the Siberian shelf seas.

Sea, region	Wastewater discharge, 10^3 m^3 per year		Discharge method
	Untreated	Treated	
Kara Sea, Ob Gulf	112.0	674.6	To gulf and land
Kara Sea, Vega Strait, Dikson Gulf	559.8	852.0	To gulf
Kara Sea, Yenisey Gulf	13531.1	996.2	To gulf
Kara Sea, Pyasina Gulf	10349.1	2350.0	To gulf
Laptev Sea, Khatanga Gulf	892.4	31.0	To gulf and land
Laptev Sea, Tiksi Inlet	1084.0		To the gulf
East-Siberian Sea, Pevek town and Chaun area	1909.0	12358.0	To gulf and land
Chukchi Sea, Schmidt area	1529.3	4212.0	To sea and land
Bering Sea, Provideniya area	5006.6		To sea and land
Bering Sea, Iultin area	817.1		To land and rivers
Bering Sea, Chukotka area	357.0		To land
Total	36147.7	21473.8	

65.2×10^6 m^3 of untreated wastewaters containing 56.4 tonnes of petroleum products directly into Kola Fjord (NEFCO 1995).

Information on municipal wastewater discharges from coastal towns and settlements in other parts of the Arctic has not been made available for this assessment.

10.3.6. Long-range transport of petroleum hydrocarbons into the Arctic

In addition to contamination from hydrocarbons originating from sources within the Arctic, the Arctic is exposed to petroleum hydrocarbons transported from more southerly areas. Petroleum production, transportation, and usage outside of the Arctic, especially in the temperate, heavily industrialized areas of the Northern Hemisphere, release hydrocarbons to the environment that can be transported to the Arctic as part of large-scale atmospheric and oceanic circulation patterns as well as by northward flowing rivers. The pathways that lead to such long-range transport of hydrocarbons have been discussed earlier (see chapter 3), they are complex and are generally not well understood.

Barrie *et al.* (1992) have investigated the long-range transport pathways to the Arctic for six contaminant groups. They found that, for three of these groups, including PAHs, there is little or no knowledge concerning transport to the Arctic. However, they also found that for the three groups where there is some information on long-range transport; radionuclides, acids, and metals, the main pathway is through the atmosphere. It seems probable that this is also the primary long-range transport pathway for PAHs and other hydrocarbons.

Unlike transport by ocean currents which can take years or even decades, atmospheric transport can be very fast (i.e., days). Hence, the atmosphere can deliver not only volatile and semi-volatile contaminants, but also aerosols containing particle-reactive contaminants such as PAHs (Hargrave *et al.* 1989, Patton *et al.* 1989, Bidleman *et al.* 1989, Barrie *et al.* 1992). These contaminants accumulate in and on snow, ice, and seawater and in northern drainage basins (Gregor and Gummer 1989, Welch *et al.* 1991).

Because PAHs are less prone to biomagnification than are most of the other persistent organic pollutants (see discussion in chapter 6), there is less concern about dispersed, long-range inputs from atmospheric sources. PAHs are more likely to have an impact in regions where direct inputs occur, such as near oil wells and refineries.

Based on a Norwegian global atmospheric model (NILU 1986), it was estimated that 40×10^3 tonnes per year of atmospheric petroleum hydrocarbons are added to the Arctic marine environment and 40×10^3 tonnes per year to the Arctic terrestrial environment (based on an assumed ratio between sulfur and petroleum hydrocarbons in precipitation). For the terrestrial environment, this estimate is supported by Russian measurements of petroleum hydrocarbon concentrations in snow cover in the Siberian Arctic (Melnikov *et al.* 1990), and for the marine environment by Russian measurements of petroleum hydrocarbon levels from direct monitoring of snow cover on sea ice (Table 10·7) (Melnikov *et al.* 1996). For the marine environment, this estimate translates into very low levels of PAHs (~400 parts per 10^{12}) in the upper 10 m of seawater (assuming the area of the Arctic Ocean to be approximately 1×10^7 km^2).

As discussed below (see section 10.5.2), significant amounts of hydrocarbons are transported to the Arctic seas by river flow. The major part of this flux originates from the lower reaches of the Arctic rivers. As a rule, petroleum pollution in the upper reaches is not transported to the river mouth zones due to self-purification and sedimentation processes that occur along the course of the river. For example, very little of the oil spilled in 1994 in the Usinsk area of the Pechora basin reached the river mouth (Melnikov *et al.* 1996, S.A. Melnikov 1995 pers. comm.).

Table 10·7. Mean levels of total petroleum hydrocarbons in snow cover of the Russian Arctic seas in winter, µg/L melt water (Melnikov *et al.* 1996).

Region	1990	1991	1992	1993
Kara Sea, south-western part	18.33	23.56	28.93	–
Kara Sea, Ob-Yenisey shelf	27.71	31.00	22.63	–
Kara Sea, eastern part	21.36	12.36	19.90	–
Laptev Sea, Khatanga shelf	6.00	–	21.00	52.00
Laptev Sea, Lena shelf	–	–	7.88	18.93
Laptev Sea, eastern part	17.50	–	10.63	25.17
Laptev Sea, western part	15.60	–	12.23	59.83
East-Siberian Sea, Kolyma-Indigirka shelf	24.00	9.57	10.17	62.17
East-Siberian Sea, off-shore zone	26.88	12.00	15.88	34.00
East-Siberian Sea, Chaun Bay	8.33	23.00	1.60	28.43
Chukchi Sea	27.20	10.88	9.93	72.10

10.4. Environmental transport and fate of petroleum hydrocarbons

10.4.1. Aquatic environments

10.4.1.1. Dispersion of oil in the sea

The movement of oil spilled on the sea surface is affected by physical and chemical factors including:

- Currents which are the resultant water motions from the superimposed effects of winds, pressure gradients (geostrophic flow), tides, tide-induced residual flows, mesoscale features (eddies), and large-scale circulation (chapter 3, section 3.5.4).
- Mean wind-driven current and Stokes' drift which is caused by non-linear residual orbital motion associated with the local wind-driven field.
- Interactions with ice.
- Oil spreading forced by gravity and surface tension and retarded by inertia and viscosity.
- Wind or weather patterns.
- Chemical properties of oil itself (density, viscosity, volatility) and the changes it undergoes due to weathering (evaporation, degradation, emulsification).

A large knowledge base has been developed based on experience with oil spills in temperate waters (for example, see GESAMP 1993, NRC 1985). While much of this knowledge is directly relevant to spills in the Arctic, ice provides a substantial complication. Ice affects the water motion and the interaction of winds with water (waves); and it provides a surface within, beneath, or on top of which oil can be trapped. Once oil becomes entrained in ice, subsequent transport and weathering is strongly affected by the motions and transformations that the ice undergoes during its drift (chapter 3, section 3.5.3).

If oil is released in the Arctic where the seas are clear of ice, it will behave much as it would in temperate zones, initially spreading, the lighter components evaporating, and the transport being governed by surface currents and the winds (cf. the case study of the *Exxon Valdez* by Wolfe *et al.* (1994)). Theoretically, oil on the sea surface can spread until it forms a monomolecular layer. However, in the environment this does not happen, partly because of changes in the surface tension from differential evaporation, dissolution, and photochemical reactions and partly from sea surface movements that influence the size and integrity of the oil slick as it drifts (Table 10·8 shows the spread and dispersion of a hypothetical oil spill in the Bering Sea). The movement and dimensions

Table 10·8. Estimated extent of oil spreading under a hypothetical oil spill scenario (1×10^6 gallon spill of crude oil with a composition intermediate between Prudhoe Bay and Cook Inlet crude) under summer weather conditons off the Pribilof Island in the eastern Bering Sea (from MMS 1992; based on Payne *et al.* (1984) and Ford (1985).

Day	Slick area, km^2	Median area of discontinuous slick, km^2
1	17	104
3	24	402
10	39	1743
30	59	6325
45	69	10126
60	76	14143

of the slick, particularly a thin slick (10 µm or less), are further masked by naturally occurring lipids and other organic substances at the sea surface. Movement of oil at the surface during the open-water period could place it within the offshore ice pack or on shores along the coast.

Oil released under an ice cover will rise toward the surface forming a local plume and wave ring and probably

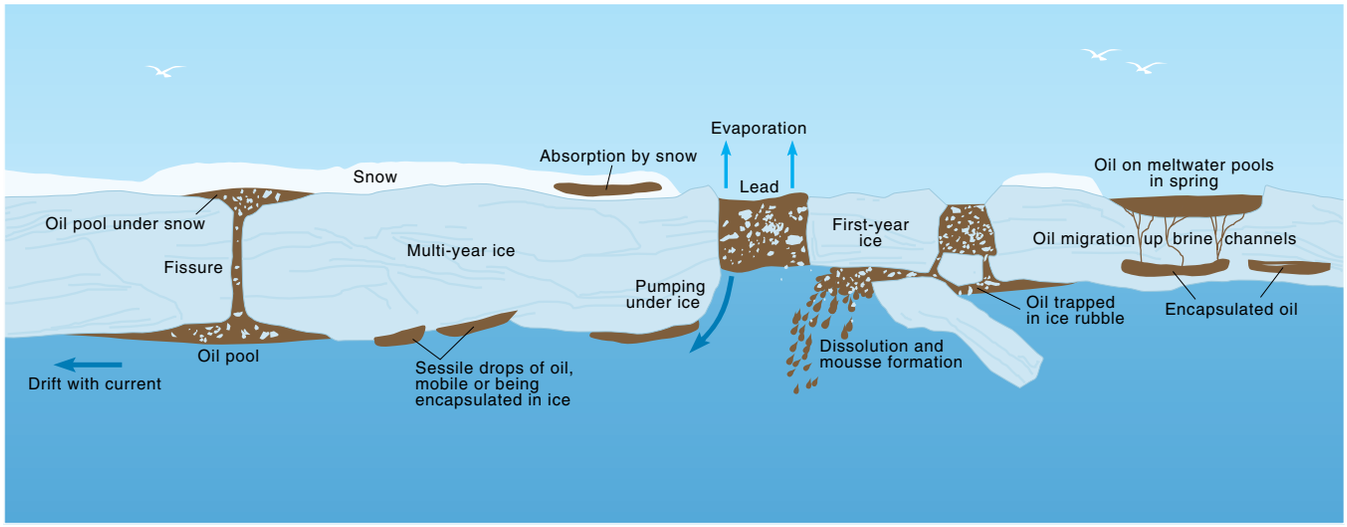

Figure 10·5. Sequence of oil-ice interaction including drops under the ice, new ice growth below the oil, oil appearing on the surface in the spring, wind herding of oil on melt pools, and the appearance of emulsified oil on top of the ice (after original figure by Bobra and Fingas).

causing partial melting due to the heat of the rising oil. Thomas (1984) calculated that under typical winter conditions in the coastal Beaufort Sea, oil at 100°C would melt about 0.5 m^3 of ice for each cubic meter of oil released. However, since the oil breaks into small droplets as it ascends through the water column, currents and mixing would probably spread the heat throughout a large volume thus limiting the oil's potential to melt ice.

Beneath the ice, oil droplets will collect until a coherent slick forms and starts to spread outward to fill the under-ice relief. The under-ice roughness determines both the amount of oil that can be trapped and the speed of oil movement relative to the ice.

In land-fast ice (chapter 3, Figure 3·22), oil may become trapped beneath immobile, low-relief ice underlain by strong water column stratification. Light components of the oil will have little opportunity to evaporate, and the oil will become frozen into the growing ice sheet. The stamukhi zone at the end of the land-fast ice (chapter 3, Figure 3·31) can retain much larger volumes of oil due to its rough, under-ice topography. Flaw leads or polynyas over the middle shelf may provide a water-air interface into which oil can spread, even in winter. Beyond the land-fast zone, oil can become incorporated in pack ice and subsequently move with its drift.

Cox *et al.* (1980) found that the mean ice roughness was 1.53 m in an untransformed, shore-fast ice zone. Thus, even relatively flat land-fast ice can retain a substantial quantity of oil. Under weak currents, trapped oil will travel with the ice. It has been shown that the current must exceed 0.20 m/s, under even the smoothest ice, to produce substantial relative motion of the oil. Since currents of this magnitude occur only for short periods of time at tidal peaks and during storm surges, it is expected that oil trapped under ice will usually move only with the ice.

Small-scale roughness (0.1-0.2 m) is enough to limit the spread of oil (Thomas 1983, Kovacs *et al.* 1981). Even beneath the smoothest ice, there is an equilibrium thickness of about 0.5-1.1 cm for the spreading oil (Thomas 1984). Therefore, large amounts of oil can be contained in a small area throughout the freezing season. For example, a spill of 32 000 m^3 (200 000 barrels) under smooth ice would be contained within an area of about 4 km^2.

During freezing, ice grows downward encapsulating oil lying beneath it (NORCOR 1975). Once the oil becomes fixed within the ice, it moves only as the ice moves (e.g., tens of meters for land-fast ice, 150 km per month at the edge of multi-year pack ice on the outer shelf). Whether motions are small or large, deformations of the ice cover cause oiled ice to be built into ridges.

Oil accumulated under the ice during winter tends to move upward within the ice and to appear essentially unweathered at the ice surface in the summer when the ice starts to melt (NORCOR 1975) (Figure 10·5). Limited lab and field experiments suggest that water-soluble components of crude oil (e.g., benzene, toluene) can migrate downward into the water column with brine formed during freezing (Payne *et al.* 1988). The timing of release of oil from the ice during spring break-up depends on the depth of the oil-contaminated ice layer on the ice sheet. Local melting, enhanced by changes in albedo due to the presence of oil, causes the oil to be released about two weeks earlier than normal mechanical break-up (NORCOR 1975, Pritchard 1981). Once the oil reaches the surface of the ice, evaporation and other weathering processes commence. As the ice cover dwindles, the oil slick begins to act like one on an open ocean containing ice floes. Since biological activity is high in early spring and the amount of open water available for birds and mammals is small, the release of oil into leads and polynyas at this time could be very damaging. This is a compelling argument for the mechanical containment and clean-up of spilled oil before spring melt.

Coastal shelf areas that export large volumes of ice would be particularly efficient at exporting spilled oil into the pack ice of the interior ocean (chapter 3, Figures 3·24 and 3·26). The large-scale ice drift (chapter 3, Figure 3·25) suggests that oil spilled in the Beaufort Sea may circulate within the Beaufort Gyre for five or more years, whereas oil spilled in the Kara or Laptev Seas could exit the Arctic with ice via the Barents Sea or Fram Strait within one to two years (chapter 3, Figure 3·23). After exiting the Arctic, the ice will melt, releasing contained oil back to the water. The potential of ice to transport material long distances is clearly illustrated by studies (e.g., Eggertsson 1994a, 1994b) that show wood from various rivers delivered to the Fram Strait.

10.4.1.2. **Dispersion of oil in freshwater**

Except for occasional, isolated oil spills from tankers, barges, and pipelines, most petroleum hydrocarbons enter freshwater (lakes, rivers, and reservoirs) chronically from municipal and industrial wastewater discharges, small spillages associated with boat and hydroplane traffic, runoff from land, and

atmospheric deposition (NRC 1985, GESAMP 1993). A recent notable exception is the Komi oil spill (see 10.5.2). Oil spills are generally contained with booms or dikes, and surface oil is usually removed effectively. In small lakes and rivers, winds often blow the oil slick to the shore on one side (Cooney 1984). Spilled oil undergoes weathering in the freshwater environment in much the same way as in the marine environment, except for minor differences related to differences in hydrocarbon solubilities.

Horowitz and Atlas (1977) described the fate of hydrocarbons during a five-week period after a gasoline spill in an Arctic lake. At the end of this time, about ten percent of the hydrocarbons remained in the sediments, and unidentified compounds related to the extensive microbial degradation were noted. Bergstein and Vestal (1978) found that the microbial flora in small tundra ponds degraded saturated hydrocarbons before the PAH fraction.

As in the marine environment, seasonal ice cover and snow on lakes and rivers affect the weathering and transport of oil. In winter, oil becomes incorporated in ice, and evaporation is reduced. Due to reduced flow and atmospheric exchange, water column concentrations of soluble hydrocarbons may be enhanced in winter. Oil frozen in ice, or covered by snow, can be released in the spring and enter water bodies with the spring flood (Melnikov and Vlasov 1994).

10.4.1.3. **Environmental alteration of oil**

When released into the terrestrial or aquatic environments, crude oil, petroleum products, or other oily wastes undergo several physical and chemical alterations, commonly named 'weathering', and caused by a wide variety of physical and biological factors. The rates and scales of some of these processes have been well studied, especially in the marine environment. It is possible to describe, and even to predict in an approximate manner, their effect on the weathering state of the spilled oil.

Reasonably good capabilities exist for predicting the evaporative losses that are responsible for the initial, and in the case of crude oil, substantial losses of mass that occur in the first 24-48 hours after a spill (MacKay *et al.* 1980, Payne *et al.* 1984, 1987, Jordan and Payne 1980). The true boiling point distillation cuts of crude oil are generally used in prediction equations for mass balance studies because component-specific data on density, vapor pressure, and other pertinent parameters are usually lacking. According to estimates for Prudhoe Bay crude oil, hydrocarbons in distillation cut 3 or lighter are nearly all lost from a slick after ten hours at a water temperature of 3.3°C and wind speed of 10.2 m/s; those in distillation cut 6 persist in small amounts for 1000 hours. Little or no evaporative losses occur in distillation cut 9 or higher (Wolfe *et al.* 1994). Algorithms have been developed for evaporative losses of different pseudo-components (distillate cuts of defined volatility) of crude oil that are estimated on the basis of the crude oil composition and properties, environmental conditions, oil-air transfer coefficients, and empirically derived relationships (Payne *et al.* 1984). Reasonably good agreement is seen between a computer-simulated estimate of loss of crude oil due to evaporation, of 23% of the mass of a hypothetical 150 000 m^3 oil spill in the northern Bering Sea (MMS 1992), and estimated evaporative loss, of 20%, in the aftermath of the *Exxon Valdez* oil spill (Wolfe *et al.* 1994). Unfortunately, hydrocarbon measurements following an oil spill are made and reported for specific components; no corroborative data are reported on pseudo-components to use for verifying modeling results.

Evaporated hydrocarbons undergo photochemical transformation in the presence of light and oxygen. This can result in the formation of aldehydes, ketones, and carboxylic acids (Payne *et al.* 1987). Half-lives of lower molecular weight compounds in the vapor phase are fairly short, of the order of days even in a cold climate. According to estimates, the half-lives of mono-aromatics, naphthalenes, and substituted naphthalenes are about one day; of biphenyl, acenaphthene, fluorene, phenanthrene, and anthracene, about two days; and of higher molecular compounds, such as pyrene, about seven days (Hanna and Drivas 1993, Wolfe *et al.* 1994).

Other physical and chemical processes affecting transformation of petroleum hydrocarbons in the marine environment include dissolution, oil-in-water dispersion, water-in-oil emulsification, oil interaction with suspended particulate material, and photo-oxidation in the liquid phase. Site-specific data are available for some of these processes from *in situ* studies, wave tank experiments, and observations after experimental and accidental oil spills (Payne *et al.* 1984, Boehm 1987, Boehm *et al.* 1987), and some empirical relationships have been formulated. However, the quantitative aspects are not sufficiently well elucidated for wider application in the subarctic and Arctic environments.

Payne *et al.* (1991) have drawn attention to several features of oil weathering that are unique to ice-covered waters compared with open-water conditions. Of importance are experimental results showing that the initiation of wave turbulence during oil polluted ice breakup may cause a very rapid rise in dissolved low molecular aromatic hydrocarbons. Concentrations increased by factors of 300 to 700 in less than one day and gradually declined over the next six days. Furthermore, they found that if an oil spill occurs during the initial stages of ice growth, dissolved aromatic hydrocarbons could be transported as conservative components along with the brine toward the bottom with minimal dilution, and where they could persist without evaporation for several months.

Microbial degradation of petroleum hydrocarbons is an important process by which substantive fractions of spilled oil are weathered or eliminated from the environment. Numerous bacteria and some filamentous fungi have been reported to degrade petroleum hydrocarbons or their derivatives (Jordan and Payne 1980, Floodgate 1984, Leahy and Colwell 1990). Seasonal distribution and diversity of bacterial populations in the Beaufort and Chukchi Seas have been described (Boyd and Boyd 1963, Bunch and Harland 1976, Kaneko *et al.* 1977, 1978). In general, the bacterial population in the coastal Beaufort Sea was found to be relatively high; higher, for example, than in the Chukchi Sea (Boyd and Boyd 1963), northern Bering Sea (Atlas and Griffiths 1984), central Arctic Ocean (Kriss 1963), or the Antarctic Ocean (Wiebe and Hendricks 1974). Viable bacterial counts, on a per gram basis, were of the order 10^2 to 10^4 for seawater, 10^5 to 10^6 for sediment, and 10^1 for ice. However, hydrocarbon-utilizing bacteria comprised less than 1% of the total counts (Atlas and Griffiths 1984).

Microbial degradation of crude oil in the Arctic appears to proceed slowly. In Beaufort Sea sediment, oil degradation was apparent only after eight months of exposure, even though the bacterial community was capable of active metabolism and growth at temperatures below 0°C (Haines and Atlas 1982). *In situ* experiments in open waters of the Barents Sea have also shown that natural microbial activity at around 0°C may be just as high as in warmer waters (Thingstad and Martinussen 1991). The slow rate of biodegradation observed in the Beaufort may have been due to limiting quantities of nitrogen and phosphorus in the medium and

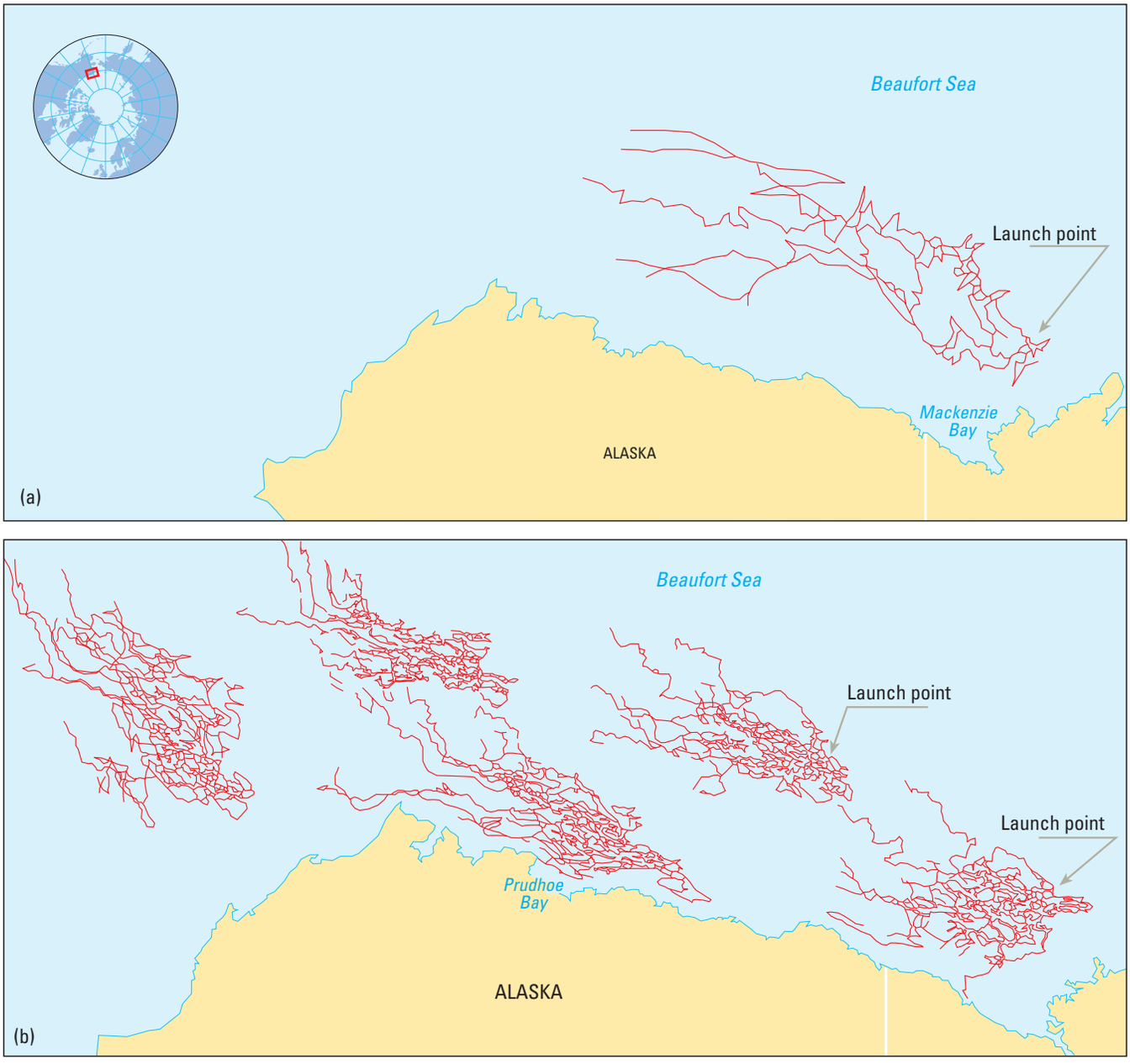

Figure 10·6. a) Trajectories of five satellite-tracked buoys launched in Mackenzie Bay, Canada (Murphy *et al.* 1981), and b) results of 30-day trajectories of oil launched from five hypothetical spill locations in the Beaufort Sea.

perhaps to low levels of dissolved oxygen, which is typical of very fine-grained sediments (silty clays) such as those in the coastal Beaufort Sea. Haines and Atlas (1982) also noted that abiotic weathering of the oil was slow and that the overall weathering was somewhat atypical in terms of the pattern of molecular degradation. Boehm *et al.* (1987) observed very little biodegradation of crude oil in subtidal areas following experimental oiling of an Arctic shoreline, whereas Eimhjellen (1982) found during the Canadian BIOS project that oil degradation in waters of 1-4°C was as rapid as in Norwegian boreal coastal waters.

The conclusion reached by Atlas (1985) after reviewing several *in situ* experimental studies is that, although the potential for hydrocarbon degradation exists in Arctic ecosystems and may be as large as in temperate regions, the actual rates of biodegradation are, in general, slow. The reason why oil degradation in cold climates seems to proceed slower than the potential for microbial activity would predict, is related to the physical and chemical conditions for biodegradation, e.g., reduced availability of the oil to degraders due to its higher viscosity and lower evaporation rate and/or to limited availability of oxygen, nitrogen, and phosphorus. Decontamination of Arctic ecosystems after an oil spill may therefore take decades rather than years, which underscores the need for special care to protect sensitive Arctic biotopes against spills.

10.4.1.4. **Oil spill modeling**

Several oil spill trajectory models have been developed to describe the pathways and landfalls of spilled oil (Huang 1983, GESAMP 1993). Inputs required for these models include environmental data (wind, current, tides, temperature, ice distribution and type, water characteristics) and oil data (physical and chemical properties, amount/rate/location of spill). For the Arctic, the lack of relevant site- and time-specific environmental data will be the weakest link in application of the models. Even where such data are available, accuracy of the output may still not be sufficient to guide response actions (Venkatesh *et al.* 1988).

Several oil spill models have been developed for ice-covered oceans (Sunde *et al.* 1995, Liu and Leendertse 1982, 1987, Colony 1986). For example, for Alaskan coastal waters, Liu and Leendertse (1987) developed a model which includes bottom topography and uses nesting, which allows spatial coverage from 1.5×10^6 km^2 to only a few hundred meters. Key model components include: 1) a 3-dimensional circulation model incorporating field observations; 2) a Stokes' drift component; 3) a model to predict the dimensions of the spilled oil; 4) a model describing ice conditions and dynamics; and 5) a 2-dimensional stochastic weather (wind) model, including a storm-track model component. The model also incorporates spreading, advection, dispersion, and weathering in estimating the oil concentration field. In addition to its use in tracking oil spills, the model has been employed to calculate back-trajectories from ecologically sensitive coasts to identify those at greater risk. An example of the model's performance compares the trajectories of five satellite-tracked buoys (Murphy *et al.* 1981) launched in Mackenzie Bay, Canada (Figure 10·6a) with results of 30-day trajectories of oil launched from five hypothetical spill locations in the Beaufort Sea (Figure 10·6b).

Sunde *et al.* (1995) have evaluated their model by comparing a five-day simulation with an oil release experiment (Figure 10·7). The release occurred in the marginal ice zone near Svalbard in April 1993; and field observations for ice, weather, and currents were used as input to the model. The authors concluded that the model results were strongly linked to the quality of the input data. Discrepancies between simulated and observed oil and ice drift were explained almost entirely by discrepancies between simulated and observed winds. This short-term test also showed that the effects of sea ice on dispersion and weathering of oil (primarily water content and secondarily evaporative loss) need to be better understood.

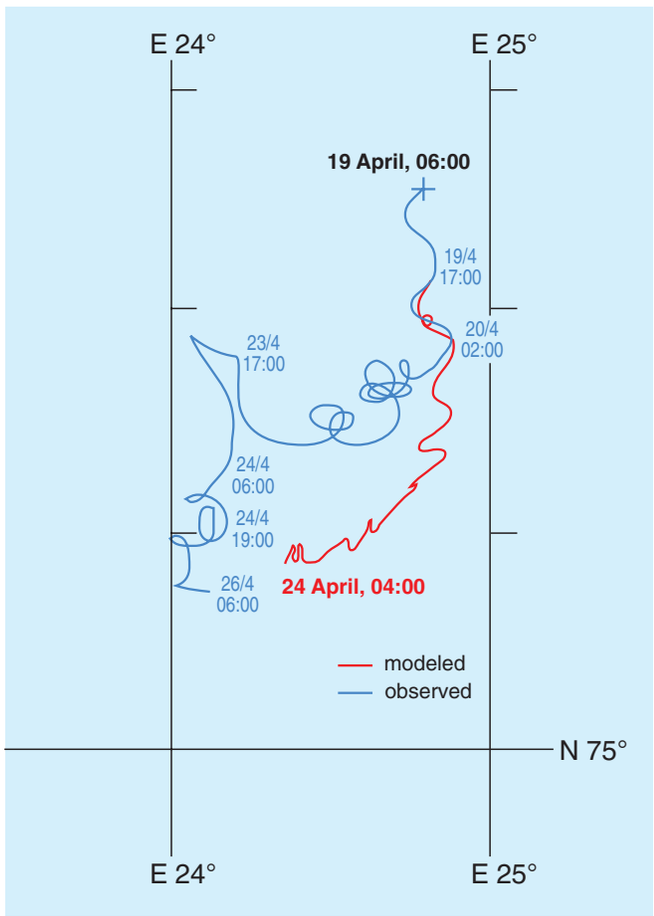

Figure 10·7. Model evaluation comparing a five-day model simulation with results of an oil release experiment in the marginal ice zone near Svalbard in April 1993 (Sunde *et al.* 1995).

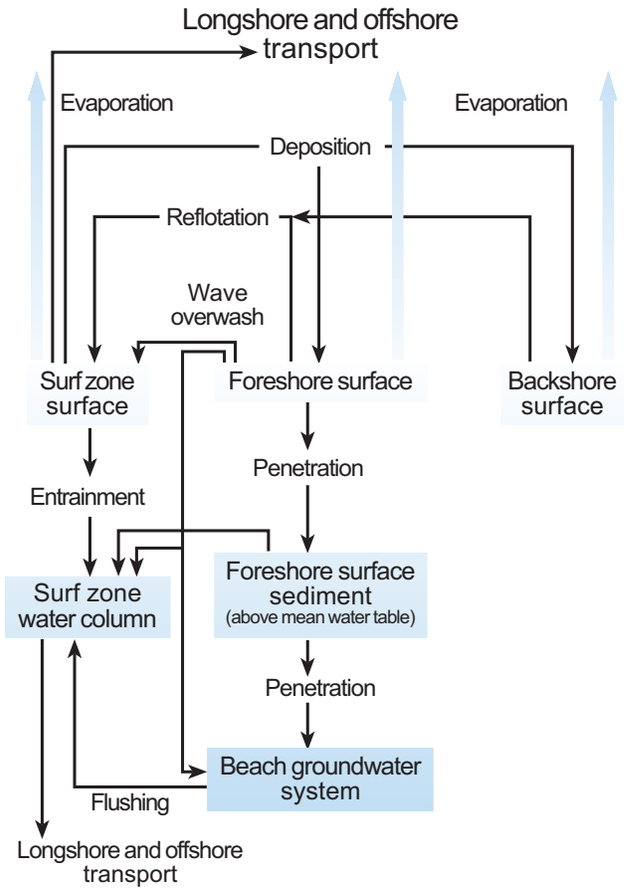

Figure 10·8. Mass transport pathways considered in the Coastal Zone Oil Spill Model (COZOIL), developed to simulate dynamic transport and weathering of oil on US Arctic coasts (from Reed *et al.* 1988).

Open-ocean models fail near the coast. Explicit representation is required for factors, such as wave characteristics, shore morphology, and sediment composition, that influence along-shore and onshore spreading of oil, penetration of oil into the sediment, and behavior of oil stranded in the fore-shore and back-shore regions of the coast. The Coastal Zone Oil Spill Model (COZOIL) has been developed specifically to simulate dynamic transport and weathering of oil on US Arctic coasts (Reed *et al.* 1988). This model, designed to accept input from an open-ocean oil trajectory model, considers the mass transport pathways shown in Figure 10·8. Penetration of oil into different sediment types and permeabilities is also considered, but the model does not include sea ice. Model output includes mass balance of spilled oil (as distillation cuts) and plots showing locations of the oil. Stochastic oil distribution estimates are produced through multiple simulations. Using data from the 1978 *Amoco Cadiz* oil spill, COZOIL reproduced the general distribution of oil, but overestimated the amount coming ashore.

The full integration of the COZOIL model with a circulation model (Spaulding *et al.* 1988), a weathering model (Payne *et al.* 1984, 1988), and a suspended particulate model (Payne *et al.* 1988) has been evaluated (Coon and Knoke 1990) and found to be impractical. However, sequential use of these models has been successfully applied to oil spill risk assessment and decision-making related to Arctic petroleum development.

10.4.1.5. The retention of oil by coasts

The adverse effects of oil on Arctic shorelines include:

- Accumulation of large amounts of oil on the beaches.
- Incorporation of nonvolatile compounds into the sediments.
- Re-release of buried oil into surface areas.
- Floating oil slicks and 'mousse' formation.
- Entrapment and entrainment of oil under ice and in coastal leads.

Several classifications have been devised to categorize shorelines according to their potential to retain oil. The oil spill vulnerability index (Hayes *et al.* 1976), combines observations on the substratum and physical processes, such as wave energy and direction, with data on retention of oil in sediment. Categories range from straight, rocky headlands with minimal oil spill retention potential (Category 1, retention for days) to protected estuarine salt marshes with maximal oil spill retention potential (Category 10, retention for 10 years). Such indices have been developed for coastlines on the US Beaufort Sea (Worbets 1979, Nummendal 1980, Alaska Clean Seas 1983, Dickins *et al.* 1987) and the US Chukchi Sea (Hayes and Ruby 1979, Robilliard *et al.* 1985). The potential effect of oil entrapped or advected under sea ice is not explicit in these indices, but consideration of this factor is recommended on a case-by-case basis.

Recently, a modified index incorporating biological and human use factors (Environmental Sensitivity Index), has been widely used to classify shorelines in the US Arctic. Environmental Sensitivity Index atlases have been prepared for most areas and videotapes are available for the Chukchi Sea. Hard-copy and digitized maps should be especially useful in guiding oil spill countermeasures by identifying priority habitats and coastal areas for protection and cleanup.

For the Canadian Beaufort Sea, geological, biological, and cultural observations are combined to produce a sensitivity number for each coastal segment (Worbets 1979). Although this facilitates comparisons among areas, it does not provide specific oil residence times for different shorelines.

An environmental sensitivity index has been developed and applied to the Norwegian coastline (Lein *et al.* 1992, Larsen *et al.* 1993) in the context of an oil spill risk analysis model (Anker-Nilssen *et al.* 1991). The model combines data on oil spill trajectories, calculates probabilities of stranding of oil in particular areas, distribution of resources in the relevant coastal areas, and resource vulnerability. The sensitivity index ranges from 0 (no detectable short- or long-term effects on macroscopic organisms) to 1 (all organisms in the impacted zone die and complete community recovery is expected to take more than 20 years). Index values have been assigned to the various sections of the coast. For hard-bottom shorelines, the index is based on prevailing wave exposure, climatic and tidal amplitudes, and the proportion of soft-bottom to hard-bottom substrates in the littoral zone. For soft-bottom shorelines, the key factors are sediment grain size distribution, geographical distance to similar benthic communities, type of biological community, areal extent of the community, and availability of effective spill countermeasures.

10.4.2. Oil spills on land

Oil spills from petroleum development facilities and transportation systems over wide expanses of the Arctic terrestrial and freshwater environments pose a significant threat to the environment, wildlife, and humans. Such threats become even more serious if the oil seeps into groundwater and becomes more widely distributed. Most often, oil spills on land result from a pipeline malfunction or rupture. However, in comparison with oil spills from accidents involving tankers and barges at sea, such spills can be quickly stopped by equipping the pipeline for automated shutdown of its pump stations and valves. For example, in the case of a rupture, only about 2226 m^3 (14 000 barrels) of oil would leak from the Trans-Alaska Pipeline during the time required to effectively shut it down (US DOI 1972). This amount represents about 1% of the daily throughput of the pipeline system and a much smaller fraction, 0.2%, of the total oil contained in the pipeline.

As in the case of the marine environment, the movements, transformations, and fates of oil spilled on land are determined by numerous factors, such as oil composition, spill volume, and prevailing environmental conditions, including the nature of the soil (surface topography, compactness, water content and permafrost) and the vegetative cover. In general, horizontal spreading of spilled oil is more retarded on land than on water, but vertical migration in the soil is usually much more rapid than through the water. The more rapid vertical migration on land also retards evaporative losses and photochemical degradation (Bossert and Bartha 1984). Heavily compacted, water-saturated, or frozen soils impede vertical migration; in such cases, horizontal spreading becomes more pronounced promoting evaporative losses. Due to their relatively high viscosities, crude oils and fuel oils tend to move horizontally, whereas gasoline and other low viscosity petroleum products tend to penetrate more quickly into the soil (McGill *et al.* 1981). A number of attempts have been made to describe analytically the areal extent of oil spreading on impermeable and permeable soil surfaces, but due to inherent assumptions in those formulae, their use has remained limited (Bossert and Bartha 1984). A simplified empirical relationship between spill area (m^2) and spill volume (m^3) appears to be a good practical alternative:

$$\text{Spill area} = 53.5\,(\text{Spill volume})^{0.89}$$

This formula is based on measurements derived from accidental and experimental spills, but the authors caution that the actual spill area can differ from the estimate by a factor of three, and in some cases, by a factor of eight (MacKay and Mohtadi 1975). Other abiotic aspects of oil weathering are discussed by McGill *et al.* (1981) and Bossert and Bartha (1984).

Numerous genera of bacteria and fungi are known to degrade hydrocarbons. The most commonly isolated bacteria from soils (for example, *Pseudomonas*, *Arthrobacter*, *Alcaligenes*, and *Flavobacterium*) are also common among those isolated from the coastal and marine environments (Leahy and Colwell 1990). Such a similarity is not obvious in the case of fungi. *Aureobasidium*, *Candida*, *Rhodotorula*, and *Sporobolomyces* are described as the most common marine isolates (Leahy and Colwell 1990) whereas *Trichoderma*, *Penicillium*, and *Aspergilus* are the more commonly isolated forms from soils (Bossert and Bartha 1984). The relative importance of bacteria and fungi in degradation of petroleum hydrocarbons is not clear at this time, although both can degrade petroleum effectively. It has been argued that filamentous fungi enhance overall microbial degradation of oil due to their mycelial growth (offering more surface contact area), but there are only limited data to support such a phenomenon (Davies and Westlake 1979).

There was an overall increase in microbial biomass and respiration when Prudhoe Bay crude oil and diesel fuel were added to soil from the Arctic coastal plain in Alaska. The in-

crease in biomass lasted for up to seven years, but increased respiration tapered off after two years (Atlas *et al.* 1978). In other cases, no significant numerical increase in the microbial population was noted in soils that were contaminated with crude oil, even though there was increased oxygen consumption (Antoniewski and Shaefer 1972). In some other cases, biomass increased after an initial decline indicating apparent toxic effects on some microbes and selective increase in growth of hydrocarbonoclastic populations (Odu 1972).

A spill of crude oil and produced water in August 1989 from a leaking value in a production pipeline near Prudhoe Bay has been well documented (Jorgenson *et al.* 1991). The spill, approx. 50 m^3 in volume, covered an area of 0.57 hectares of Arctic coastal tundra inundated with numerous small lakes and ponds. The vegetative cover was dominated by graminoids; various forbs, mosses, and lichens were present in small amounts. Most of the oil that infiltrated the soil was confined to the surface of water-saturated tundra. Monitoring of 'total petroleum hydrocarbons' showed that mean oil concentration at the soil surface decreased 79% from 1989 to 1990 (from 16 341 to 3446 ppm) and an additional 2% in 1991 (to 3169 ppm). Mean subsurface (18-23 cm deep) concentrations also showed a similar decline: i.e., 460 ppm in 1989, 36 ppm in 1990, and 37 ppm in 1991. The initial large reduction was attributed to microbial activity and photo-oxidation; the much lower rates of hydrocarbon reduction in 1990-1991 could be due to the remaining fraction of the hydrocarbons being more recalcitrant to degradation (Jorgenson *et al.* 1991). According to Jorgenson *et al.* (1991), thaw settlement (thermokarst) was stabilized by 1991, and recovery of the vegetative cover was well underway to meet the regulatory criterion of 30% of the mean percent cover of vascular plants in an adjacent, unaffected area.

Laboratory microcosm experiments to elucidate field observations after the spill showed unexpectedly poor degradation (reduction of oil), nearly negligible, after 12 weeks under aerobic conditions, but a 47% reduction by indigenous bacteria under anaerobic conditions. The best degradation rate, 60%, was observed when using a proprietary strain of bacteria. Bacteria indigenous to the site included *Moraxella osloensis*, *Acinetobacter* sp. *Rahnella aquatilis*, *Psychrobacter immobilus*, *Alcaligenes faecalis*, *Pseudomonas* sp., and *Pseudomonas palleronii*. By the end of the sixth week of the experiment, the predominant species were reduced to *Psychrobacter immobilis*, *Alcaligenes faecalis*, and *Pseudomonas palleronii*. By the end of the experiment (12 weeks) only *Pseudomonas* sp. was prevalent (Jorgenson *et al.* 1991). It is speculated that a high content of organic matter in the tundra soil may have affected biodegradation rates by absorbing hydrophobic organic solutes, providing an alternate carbon source for microbial growth and affecting the denitrifying process (Mihelic and Luthy 1988).

One of the largest crude oil spills on land occurred as a result of environmental releases from a section of pipeline in the Usinsk region of the Komi Republic in Russia. Deterioration in the condition of this pipeline led to chronic leakages and then to an accidental break. It is estimated that 37 000 to 44 000 tonnes of oil spilled from the major rupture during September 1994 (Melnikov and Vlasov 1994). However, when the chronic releases are added to the amount from the major rupture, the total amount of oil estimated to have been released to the landscape from this section of pipeline is 103 000 to 126 000 tonnes (Melnikov and Vlasov 1994). Much of this spilled oil was found to be trapped in bogs and creek beds in the area.

Considerable initial effort was devoted to containment of this oil and preventing it from reaching the Kolva, Usa, and Pechora Rivers. Very few data are available on the environmental fate (migration of oil in soil and groundwater, weathering state, and degradation) of the spilled oil. Preliminary data have shown similarity between the source hydrocarbons (Usinsk Field), and those found in soil samples from the Kolva settlement; in addition, there was qualitative correspondence between the source oil and oil patches that were entrapped in fishing nets in the Pechora River (Melnikov, Vlasov, and Ostrovsky, 1996, unpubl. data). Test burning of the trapped oil, even though it removed surface oil, caused the remaining oil to penetrate 10 cm deeper into the soil; also, ditches dug inside the dikes for oil recovery lowered the water table in the bog, resulting in deeper penetration of oil into the peat-laden soil (J. Michel 1996, pers. comm.).

Oil and gas extraction and transport activities can especially pose a threat to wetland areas. This has been found to be the situation in northwestern Siberia where vast wetland areas adjacent to extraction sites have become significantly contaminated with a wide range of pollutants. These include both naturally occurring chemicals associated with oil and gas deposits as well as anthropogenic contaminants, such as phenols, diethylene glycol, methanol, nitrogen compounds, and metals, that are used in extraction, pretreatment, and transportation of oil and gas products. Such pollutants, when released to wetland environments, are washed by spontaneous surface and ground runoff into the natural drainage system where they accumulate in landscape depressions. For example, petroleum hydrocarbon concentrations in surface runoff from the vicinity of extraction wells in northwest Siberia have been found to range from 0.5 to 5.2 g/L (Hydrochemical Institute 1992, 1993, 1994, 1996). The depressions are also widely used by oil and gas extraction activities for operational discharge of polluted and usually untreated waste waters from oil pretreatment facilities. Waste waters in this Siberian area have been found to contain petroleum hydrocarbons ranging from 0.1 to 5.0 g/L (Hydrochemical Institute 1992, 1993, 1994, 1996) as well as diethylene glycol concentrations as high as 25 g/L and methanol from 0.14 to 0.29 g/L (Tyumen Industrial Institute, Tyumen, Russia, unpubl. data).

In the northwest Siberian case, such releases have led to significant accumulation of contaminants in wetlands areas. These wetlands then serve as the source of secondary contamination of the natural systems through the periodic release of their accumulated contaminants. At most times, because of the low relief generally found in such areas and the presence of the permafrost close to the surface, rates of drainage from these contaminant concentrations are relatively low. However, during times of snow melt and large rain storms, these wetlands become flooded and so serve as large sources of pollutant discharge into the adjacent drainage system. Thus, although waste waters from northwest Siberian oil and gas extraction activities are not discharged directly into the rivers and streams, the levels of petroleum hydrocarbons and related contaminants are rather high in many of these water bodies. Petroleum hydrocarbons were found to exceed the regulatory Maximum Permissible Concentration (MPC) of 0.05 mg/L in almost all samples obtained from northwest Siberian rivers. The mean concentrations from these rivers ranged from 6 to 13 times the MPC with a maximum of 42 times the MPC. Similarly, phenol concentrations in these rivers ranged from 2 to 8 times the MPC (0.001 mg/L) with a maximum of 29 times the MPC.

10.5. Levels of petroleum hydrocarbons

An overview of the levels of total petroleum hydrocarbons, both alkanes and aromatics, in marine, freshwater, and terrestrial Arctic environments is presented in this section. This is followed by a separate section focused specifically on polynuclear aromatic hydrocarbons (PAHs), which make up a part of the aromatics group. These compounds are given separate attention because there is greater concern than for the other petroleum hydrocarbon fractions with regard to their potential effects on human health and the environment. A number of the PAHs are known carcinogens and/or mutagens. As a group, they are substantially more persistent than most other petroleum hydrocarbons and thus are more likely to be associated with long-term degradation in environmental quality.

The data which are presented in tables under this section reflect the fact that the largest quantity of data for petroleum hydrocarbons in the Arctic exists for Russia. The main reason for this is not that petroleum hydrocarbons are necessarily more problematic in Russia, but rather that Russia is the only circumpolar nation that has conducted systematic, long-term monitoring of petroleum hydrocarbons in the Arctic.

The geographical coverage of samples for petroleum hydrocarbon levels across the Arctic is generally poor because most sampling has been limited to relatively accessible locations close to sites of human activity. Samples are more commonly available from the aquatic environment (marine and freshwater) than from the terrestrial environment. There are especially few measurements of petroleum residues in biota as such measurements have not been emphasized in most sampling programs. This is because, as mentioned above, aside from the PAHs, the petroleum hydrocarbons are not generally persistent in the environment. Additionally, although many of the petroleum compounds, particularly aromatic hydrocarbons, are rapidly bioaccumulated, they are also readily metabolized by many organisms, particularly fish and other vertebrates.

Direct comparison of the values obtained by the various programs which have measured petroleum hydrocarbons in the Arctic is almost impossible because of typically wide differences in the analytical methods, instrumentation, quantification, and reporting used to produce the results. Some values, particularly those produced by nonspecific techniques such as infrared, ultraviolet/fluorescence, and gravimetry, tend to overstate the amount of petroleum hydrocarbons present in a sample. Other values, especially those which are based on the sum of individual compounds, tend to understate the total amount of petroleum hydrocarbons present. The lack of consistency among the data sets severely limits their usefulness for analyses of spatial and temporal trends.

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 increased by other important factors, such as sampling representativeness, sample contamination, etc. Data known to be erroneous have been excluded from the data tables. However, as few details describing the sampling techniques, sample storage, sample processing, and sample analysis (calibration, estimation of precision/accuracy, quantification, etc.), specific to each individual data set were available to review (if they exist at all for many data sets), no data quality assessment was made of the remaining data. Consequently, all data should be viewed with caution. This situation has led to the strong recommendation made in this chapter that actions be taken to establish data quality protocols, so that all future data for petroleum hydrocarbons are intercomparable.

Under these circumstances, while the accuracy and representativeness of any individual data value in the data sets are probably arguable, taken as a whole, the data reported can, nonetheless, be used to identify the general range of values for petroleum hydrocarbons in various compartments of the Arctic ecosystem.

It should also be noted that the comparisons made in this section for petroleum hydrocarbon levels in sediments among different geographical areas should be considered only inexact, semiqualitative estimates. This is because the concentrations of petroleum hydrocarbons in a sediment sample are known to vary in relation to sedimentary particle grain size and organic carbon content. Very few of the data available for the Arctic are accompanied by such information, and so the data can not be normalized to remove the confounding influence of these factors.

A further complication arises because few of the compounds found in petroleum are unique to petroleum. Many have terrestrial biogenic, marine biogenic, pyrolytic, or diagenetic sources. Consequently, values reported as petroleum residues often include compounds from non-petrogenic sources. In the more recent literature (e.g., Yunker *et al.* 1993, 1994, Steinhauer and Boehm 1992), information is usually provided to evaluate source. A number of diagnostic parameters can aid in such an evaluation. These include:

- *Carbon Preference Index* (CPI): 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 do petroleum hydrocarbons, which have a roughly equal distribution of odd- and even-numbered n-alkane chains (CPI app.1).
- *Pristane/Phytane Ratio*: 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 very much larger than 1.0 (typically 3-5). When the value of the ratio is close to 1.0, a strong contribution of petroleum hydrocarbons is indicated.
- *Alkane Compositional Ratio*: 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, while it approaches 1.0, depending on the type of petroleum input, in those contaminated with petroleum.
- The presence of naphthalene and its alkyl-substituted homologues in sediments is characteristic of unweathered petroleum. These compounds are rarely found at detectable concentrations in 'clean' sediments.
- *Phenanthrene*: Phenanthrene and its alkylated homologues can be petrogenic, pyrogenic, or diagenetic. The presence of the more highly alkylated homologues usually indicates a petrogenic source.
- *Dibenzothiophene*: Dibenzothiophene homologues and its C_1 to C_3 alkyl are distinct components of many fresh crude oils.
- *Relative Abundance of PAHs*: Four- and five-ring PAHs are produced primarily from the combustion of fossil fuels or wood on land. The relative abundance of these PAHs to two- and three-ring PAHs can be used to help distinguish between petrogenic and pyrogenic sources. Pyrogenic PAH assemblages are enriched in three- to five-ring PAH compounds whereas uncombusted fossil fuels are enriched in two- and three-ring PAH compounds.

In addition, combustion favors the less stable kinetic isomer over the more stable thermodynamic isomer of the same molecular mass (e.g., anthracene favored over phenanthrene).

- *Phenanthrene/Dibenzothiophene Ratio*: As petroleum inputs increase, the amount of dibenzothiophene increases relative to phenanthrene, and the value of the ratio approaches the value in the oil (typically 1-2). In unoiled sediments, the ratio is very high and can range between 10 and 100 or even higher.
- *Naphthalene/Phenanthrene*: This ratio is particularly useful for detecting the presence of fresh petroleum. Because the presence of naphthalene compounds is characteristic of fresh crude, this ratio is much greater than 1.0 for most petroleum types and decreases to approximately 0.2 to 1.5 in 'clean' sediments.
- *The Fossil Fuel Pollution Index* (FFPI) (Boehm and Farrington 1984, Steinhauer and Boehm 1992): This is a ratio designed to quantify the approximate percentage of fossil PAHs relative to total PAHs.

$$FFPI = \frac{N + P + (\frac{1}{2}[P + C_1P]) + D}{\sum PAH} \times 100$$

where

N = naphthalenes
P = phenanthrenes
C_1P = methyl phenanthrenes
D = dibenzothiophenes
$\sum$PAH = total PAHs (sum of two- to five-ring PAHs).

Typically, many of the above parameters must be used together to evaluate a data set in a qualitative sense, and conclusions regarding sources (petrogenic vs. biogenic, petrogenic vs. pyrogenic, natural vs. anthropogenic, etc.) are made on a weight-of-evidence basis. A more rigorous approach is to use principal components analysis (see for example, Yunker *et al.* 1991a).

10.5.1. Levels in the marine environment

Summaries of the data on concentrations of the petroleum hydrocarbons in water, suspended particulates, bottom sediments, and biota of the Arctic marine environment are presented in Annex Tables 10·A3, 10·A4, 10·A6, and 10·A9, respectively.

The Regional Center 'Monitoring of the Arctic' in St. Petersburg has conducted the most extensive sampling to date of the petroleum hydrocarbons in the nearshore and marine waters of the Russian Arctic (RCMA 1994, 1995) (Annex Table 10·A3). The results are highly variable as might be expected for so vast an area. In general, as discussed above, the concentrations of dissolved petroleum hydrocarbons (detected by infrared) were found to be higher in river estuaries than in offshore locations. The concentrations were also found to be generally higher in surface than in subsurface waters. Both of these trends strongly suggest anthropogenic sources.

In the shelf seas, petroleum hydrocarbon concentrations are usually highest just off of the river mouths, and tend to decrease with increasing distance from these areas. Reported concentrations of dissolved petroleum hydrocarbons in marine waters of the Russian Arctic are generally much higher than those found in North American areas (Annex Table 10·A3). Much of this can be explained by differences in analytical methodology, as the Russian data include results from both petroleum hydrocarbons and hydrocarbon-like compounds. For example, most of the North American data are based on gas chromatography detection methods, which typically include aliphatic compounds up to nC_{36}. The Russian infrared method can lead to much higher values when samples contain highly weathered particulate oil or tarball material containing primarily aliphatic compounds heavier than nC_{36}. Some of the difference, however, can be attributed to greater pollution.

The highest concentrations of petroleum hydrocarbons are observed in the vicinity of seaports, such as in Kola Fjord near the port of Murmansk. During winter, when the waters of the Kola Fjord are relatively stagnant, concentrations of total hydrocarbons in surface water and bottom water exceed the maximum permissible concentration (MPC) of 50 μg/L by factors of 150 and 36-126, respectively. The average concentrations of paraffin oil in surface water and bottom water are five times the MPC and 35 times the MPC, respectively. In contrast, during summer, when water movements throughout the fjord are greatly increased, concentrations of total hydrocarbons seldom exceed two times the MPC (ACOPS 1995).

The concentrations of total petroleum hydrocarbons in marine sediments from the Russian Arctic are highly variable. The highest concentrations have been reported from the Ob and Yenisey estuaries, and the Baydaratskaya Gulf (Annex Table 10·A6). On the basis of data reported by the Regional Center 'Monitoring of the Arctic' (RCMA 1994, 1995) for the above three areas, maximum values for total hydrocarbons were much higher in 1994 than 1995, as shown in Table 10·9. The explanation for this is not known at the present time.

Table 10·9. Comparison of total petroleum hydrocarbon concentrations in marine sediments in the Russian Arctic in 1994 with concentrations in 1995.

	Range, μg/g dry weight				Mean, μg/g dry weight	
	N	1994	N	1995	1994	1995
Baydaratskaya Gulf	17	12-498	5	11-30	154	22
Ob Gulf	17	27-1680	11	21-40	431	27
Yenisey Gulf	8	59-330	8	15-40	158	25

Farther offshore in the Russian Arctic, concentrations are lower. There is good agreement between concentrations of petroleum hydrocarbons reported for the offshore Kara Sea by RCMA (1995) (<2-23 μg/g) and those reported by dos Santos *et al.* (1996a) (1-22 μg/g).

In contrast to the practice in Russia, the other circumpolar countries do not monitor total petroleum hydrocarbons in water to assess oil pollution, but instead make measurements in bottom sediments. For this reason, data on the levels of hydrocarbons in water from the other sectors of the Arctic Ocean are scarce and mostly limited to low molecular weight aliphatic hydrocarbons.

Macdonald (1976) measured concentrations of low molecular weight hydrocarbons in the Beaufort Sea during 1974 and 1975 (Annex Table 10·A3). He reported concentrations of methane (72-617 nL/L in 1974 and 15-1151 nL/L in 1975) that were considered high for marine waters, but very low concentrations of ethane, ethene, propane, and n-butane. Profiles of dissolved methane as a function of depth indicated that concentrations were highest in a zone within about 10 m of the bottom, suggesting that the sediments were the methane source. Cline (1977) and Cline *et al.* (1978) measured similar concentrations of low molecular weight hydrocarbons in the Chukchi and southeast Bering Seas and in Norton Sound (Annex Table 10·A3). The concentrations of methane, ethane, and ethylene they found in these areas were approximately two (ethane, ethylene) to three (methane)

times higher in near-bottom samples than in surface samples, also indicating a sedimentary source for these compounds. Since high ethane, propane, and n-butane levels were not found in conjunction with the high methane concentrations in the Alaskan and Canadian Beaufort Sea samples, petrogenic contamination by seeps or hydrocarbon exploration-related activities was considered unlikely. In contrast to the levels discussed above for Canadian and Alaskan areas, Swinnerton and Lamontagne (1974) reported a range of 65-75 nL/L for the sum of methane, ethane, butane, and propane in the Norwegian Sea.

Additional evidence of sediments as the source of methane is provided by work by Kvenvolden *et al.* (1981, 1993). This study showed that methane concentrations in the interstitial waters of Bering Sea sediments were much higher than concentrations of other low molecular weight hydrocarbons, and that the methane concentration increased with depth in the sediments.

Very few measurements have been made of alkanes in seawater because background concentrations are generally well below historical detection limits. Using large volume samplers (sample volumes of 100-200 L), however, Thomas (1988) measured individual alkanes in the waters of Tuktoyaktuk Harbour, Northwest Territories, Canada (Annex Table 10·A3). All n-alkanes occurred at concentrations between the detection limit (0.1-4 ng/L depending on the compound) and 8 nL/L. Total alkane (nC_{12} to nC_{38} + isoprenoids) concentrations ranged from 10 to 162 ng/L (Annex Table 10·A3). Patterns attributable to plant wax sources of alkanes were evident. Corresponding particulate alkane concentrations occurred in the range 27-1100 ng/g (Annex Table 10·A4) (Thomas 1988) and showed an alkane pattern similar to that found in the water samples.

Similar values (52-74 ng/L) were reported by Ahnoff *et al.* (1983) for waters off Faeringehavn, Greenland. Somewhat higher values for total dissolved alkanes, in the range 0.5-3.0 μg/L, have been reported by Cretney *et al.* (1987b) and Levy (1981) for Baffin Island coastal waters (Annex Table 10·A3).

Taken as a whole, the reported concentrations for alkanes, aliphatics, and petroleum hydrocarbons in Beaufort Sea sediments are highly variable (Annex Table 10·A6). Much of the variability can be attributed to differences in analytical methodology. In particular, high values are reported for 'hexane extractable compounds' (1-2570 μg/g) and 'oil and grease' (45-5200 μg/g) (Thomas *et al.* 1990). The hexane extractable fraction in 244 out of 249 sediment samples collected during Beaufort Sea coastal surveys in 1982-1984, often in areas of offshore drilling and shipping, exceeded the limit of 10 μg/g set by Canada's Ocean Dumping Control Act, and was often many times this limit (Wainwright and Humphrey 1988). Another important factor in the variability, which is characteristic for data on bottom sediments, is that the values have not been normalized against particle size fraction or organic content. Varying amounts of biogenic lipids are undoubtedly present in the samples.

Wong *et al.* (1976) reported that the distribution of n-paraffins in marine sediments from the southern Beaufort Sea was characteristic of sediments containing both marine and terrestrial biological and detrital materials. All sediments contained a series of n-paraffins (n-C_{17} to n-C_{32}), the isoprenoids pristane and phytane, and an unresolved group of non-polar hydrocarbons. The concentrations of n-paraffins ranged from 2.6 to 23.6 μg/g, whereas the levels of the unresolved hydrocarbons varied from 28.3 to 136.7 μg/g. Peake *et al.* (1972) reported a similar range of 1-9 μg/g for n-paraffins from ten Beaufort Sea sediment samples. By comparison, Brown *et al.* (1972) found that the range of total n-paraffins in Saanich Inlet (British Columbia) surface sediments was 2-33 μg/g.

Wong *et al.* (1976) noted an odd-carbon predominance, which is characteristic of terrestrial plant material, in most of the marine sediments they examined from the southern Beaufort Sea, suggesting an influx of detritus from the Mackenzie River. Similar characteristics were observed at the Tarsiut petroleum exploration site in the Canadian section of the southern Beaufort Sea (Thomas *et al.* 1982). There, the aliphatic hydrocarbon content varied between 0.3-38.7 μg/g and the unresolved complex mixture from 0.9-44.7 μg/g.

Thomas *et al.* (1983) found the alkane content of Beaufort Sea beach sediments (0.08-26 μg/g) to be within the lower end of the range for marine sediments (Annex Table 10·A6). Cretney *et al.* (1987a) report a similar range of <0.3-2.3 μg/g for Baffin Island beach sediments.

Detailed studies of sedimentary hydrocarbon assemblages from the US and Canadian parts of the Beaufort Sea strengthen the conclusion that the hydrocarbon content of coastal and shelf sediments usually reflects multiple sources. For example, Steinhauer and Boehm (1992) note that sediments from the Beaufort Sea coast in Alaska are uniformly characterized by odd-carbon dominance in the nC_{23} to nC_{34} range. This strongly suggests that inputs comprising primarily terrestrial plant materials dominate the higher molecular weight portion of this hydrocarbon assemblage. Yet, the presence of substantial amounts of nC_{10} to nC_{20} alkanes suggests that these sediments also contain petroleum hydrocarbons.

The presence of pristane in almost all the Beaufort Sea sediment samples reflects a mixed contribution of marine biogenic and petrogenic sources. Yunker *et al.* (1993) and Cretney *et al.* (1987a) provide similar arguments using the relative abundance and pattern of alkanes to identify terrestrial biogenic and petrogenic sources of hydrocarbons in sediments on the Mackenzie Shelf and nearshore Baffin Island, respectively. By contrast, the terrestrial higher plant n-alkanes are less important in Barents Sea sediments and petrogenic isoprenoids are more abundant indicating a much more important contribution of petrogenic hydrocarbons to the total (Yunker *et al.* 1996).

Samples of fish tissues from the southern Beaufort Sea show percentages of unresolved total hydrocarbons of 23-67% compared with 12-16% for fish from the northeast Pacific Ocean and over 80% for Sargasso Sea fish (Wong *et al.* 1976). These characteristics can be used as a measure of petroleum contamination, since northeast Pacific Ocean fish are considered uncontaminated and Sargasso Sea fish contaminated. On this basis, southern Beaufort Sea fish may be considered marginally contaminated by petroleum hydrocarbons, although the small number of samples analyzed, the biological variability, and the analytical uncertainties associated with this unresolved hydrocarbon envelope technique limit interpretation of these data. Wong *et al.* (1976) reported concentrations of non-polar hydrocarbons in Beaufort Sea fish at 2-8 μg/g wet weight, which is quite similar to the range of 0.4-11.8 μg/g (wet weight) reported for a variety of fishes, including flounder, dogfish, cod, and herring, from Atlantic waters by Clark and MacLeod (1977). However, Thomas *et al.* (1990) reported higher concentrations of n-alkanes in liver tissue, i.e., ND-350.6 μg/g (wet weight), for flounder from Tuktoyaktuk Harbor (Annex Table 10·A9).

Wong *et al.* (1976) found that most mixed zooplankton samples from the southern Beaufort Sea contained C_{16} to C_{38} n-alkanes The gas chromatographic pattern showed the maximum n-paraffin content at C_{26}, with a slight predominance of odd carbon-number paraffins and a significant en-

velope of unresolved hydrocarbons. Total hydrocarbons ranged from 2 to 203 μg/g (wet weight) with 0-7 μg/g (wet weight) being unresolved hydrocarbons. The authors were unable to determine whether these relatively minor indications of the presence of petroleum hydrocarbons were truly indicative of the incorporation of such compounds in the samples of plankton biomass or whether they were an artifact of the sampling technique which included collection of both living and non-living particles. Data from studies conducted in Massachusetts, Scotland, and elsewhere in the British Isles indicate n-paraffin concentrations in mixed plankton ranging from 0.3 to 159 μg/g (wet weight) (Clark and MacLeod 1977).

Biota from Alaska show indications of some contamination with petroleum hydrocarbons. Snyder-Conn and Lubinsky (1993) found a range of 0.4-1.8 μg/g (wet weight) for petroleum hydrocarbons (nC_{12}-nC_{20} + isoprenoids) in Arctic flounder and 0.04-0.93 μg/g (wet weight) in four-horn sculpin. Steinhauer and Boehm (1992) report generally low concentrations, ranging from 1.6-26 μg/g (wet weight), of total petroleum hydrocarbons for five species of invertebrates from Beaufort Sea (Annex Table 10·A9).

Based on determinations of C_{12}-C_{35} hydrocarbons, concentrations in the bottom sediments of the Norwegian part of the open Barents Sea ranged from 5 to 60 μg/g (dos Santos *et al.* 1996b). Total petroleum hydrocarbon levels in sediments from 14 harbors along the Norwegian Arctic coast in 1994 were found to vary considerably and were, in general, higher than the average for all Norwegian harbors (Konieczny 1996). Levels exceeding 1000 μg/g were found in 5 of the 14 harbors. The highest level, 7000 μg/g was found in Hammerfest, presumably caused by the large number of bunker oil storage sites around the harbor.

10.5.2. Levels in the freshwater environment

Summaries of concentrations of the petroleum hydrocarbons in water, suspended matter, sediments, and biota of the Arctic freshwater environment are presented in Annex Tables 10·A1, 10·A2, 10·A5, and 10·A8, respectively.

It is well documented that the Russian Arctic rivers, especially in areas of oil and gas exploration and production such as the lower reaches of the Ob River, are heavily contaminated with petroleum hydrocarbons. Data from the Russian Federal monitoring network show that mean concentrations of hydrocarbons in water samples obtained from this river, in the areas of the most extensive oil extraction activities, and determined with IR-spectrophotometry, exceeded the established standards for freshwater (MPC = 50 μg/L) by 12-16 times. Although this analytical method is widely used in routine monitoring (APHA 1989, WHO 1987), it can, depending on the crude oil composition and levels of biogenic hydrocarbons in the water, give significant errors (Chapman and Kimstach 1996). Additionally, in the opinion of the experts from the Regional Centre 'Monitoring of the Arctic', the sampling and analytical procedures used by this routine monitoring network violate significantly the sampling and analytical protocols recommended by the Russian Federation Guidelines (Semenov 1977), and thus usually lead to a systematic overestimation of the concentrations. In spite of this, the concentrations obtained correlate well with the areas of oil and gas activities. It should also be noted that pollution of the eastern Siberian rivers (e.g., Kolyma River) has increased in recent years.

When oil fields and other relevant pollution sources are situated on the upper rather than the lower reaches of a river, the area near the river mouth is not usually heavily contaminated. This is due to intensive self-purification processes that occur along the course of the river. In spite of intense, chronic and acute petroleum hydrocarbon pollution in the upper part of the Pechora Basin, the river mouth area has only moderate levels of contamination, with mean values that do not exceed the MPC. However, the cumulative transport of the contaminants by river flow is indicated by elevated concentrations of total petroleum hydrocarbons in the bottom sediments of the Pechora Basin relative to concentrations in bottom sediments from background water bodies (Taimyr Peninsula). Decreased concentrations of total petroleum hydrocarbons in the Ob River downstream of the last observation station of the Federal monitoring network along the river (at Salekhard) also indicate that rivers have high self-purification capacity, and that oil pollution is primarily a local environmental stress factor, with only limited transport over long distances by river flows (Annex Tables 10·A1, 10·A5).

Mean levels of petroleum hydrocarbons in bottom sediments at the mouths of Russian Arctic rivers in 1991-1993 were in the range 29-58 μg/g. These concentrations are comparable with those observed in the Mackenzie River Delta in Canada (Table 10A·5). The highest levels were found at the mouths of the Pechora and Ob Rivers.

A survey of petroleum hydrocarbon contamination of the bottom sediments in the Pechora River Basin was made in 1994-1995, in connection with the Usinsk oil spill. In the immediate vicinity of the oil spill site (Palmer-Shor Brook), a concentration of 14 mg/g was documented. In the mouth of the Kolva River, the hydrocarbon concentration decreased to 0.52 mg/g. However, farther downstream, where the Usa River flows into the Pechora (Ust-Usa settlement on the right bank), the bottom sediments were strongly contaminated with hydrocarbons (16 mg/g). This observation can be explained by the hydrological conditions of the river at this site which are favorable for accumulation of hydrocarbons. Farther downstream, the concentrations did not exceed 100 μg/g (Melnikov and Vlasov 1994). This example indicates that the distribution of petroleum hydrocarbons in bottom sediments is extremely variable, and is strongly dependent on both the particle composition of the sediments and hydrological conditions.

Data on gross fluxes of petroleum hydrocarbons discharged by Russian rivers to their estuarine zones are presented in Table 10·10. It should be noted that the routine river monitoring network does not allow for reliable estimation of riverine fluxes of contaminants, especially hydrophobic contaminants (Gordeev and Tsirkunov in press), both because it does not provide sufficient data, and because the monitoring sites farthest downstream from monitored cross-sections on the large rivers are situated too far from the estuarine zones. In addition, as discussed above, the IR-spectrophotometric method used by the Russian Federal (Roshy-

Table 10·10. Estimates of petroleum hydrocarbons gross fluxes by the Russian Arctic rivers (Hydrochemical Institute 1994, 1996).

River	1991		1992		1993	
	Discharge, km^3	Flux, 10^3 t	Discharge, km^3	Flux, 10^3 t	Discharge, km^3	Flux, 10^3 t
Onega	17.4	0.54	15.0	0.40	1.82	0.42
Northern Dvina	113	1.58	103	2.37	129	3.23
Mezen	20.4	0.41	18.2	0.42	24.9	0.42
Pechora	174	1.57	139	2.50	150	4.50
Ob	256	219	347	286	435	626
Yenisey	631	446	656	472	552	205
Lena	482	15.4	508	2.54	479	33.0
Indigirka	57.1	5.19	48.9	3.13	36.0	1.44
Kolyma	76.6	4.59	80.7	4.76	84.2	10.5

dromet) monitoring network is not reliable for accurate determination of petroleum hydrocarbons. For these reasons, the data presented in the table can give only a preliminary overall picture of riverine fluxes, and should not be considered precise estimates. A special research and monitoring program should be established to assess riverine fluxes of petroleum hydrocarbons.

Information on oil pollution in freshwater environments in the other parts of the Arctic is scarce. Dissolved alkanes in the Mackenzie River occur at sub-nanogram to low nanogram per liter concentrations (Erickson and Fowler 1987, Nagy *et al.* 1987, Morgan *et al.* 1987, Yunker *et al.* 1989, 1991a, 1991b, 1994) which are typical of background. The major fraction of the n-alkanes found in the Mackenzie River suspended particulates (Annex Table 10·A2) indicates a terrestrial plant source (Erickson and Fowler 1987, Yunker *et al.* 1996). The lower molecular weight n-alkanes and the isoprenoids indicate the presence of a petroleum hydrocarbon fraction. Particulate alkanes appear to be higher in the Mackenzie River Delta than upstream, and higher in summer than in winter. This observation may be real, but may also be an artifact related to analytical variations or differences in particulate loading along the river. Snyder-Conn and Lubinsky (1993) reported alkanes (nC_{12}-nC_{20} + pristane and phytane) for ponds and lakes in Alaska (Annex Table 10·A1). All values were essentially below or at the detection limit of 10 μg/g, a detection limit at least three orders of magnitude too high to allow quantification of most background concentrations.

The alkane content reported for sediments from the lower Mackenzie River is approximately 0.2-37 μg/g (Annex Table 10A·5) (Thomas *et al.* 1990, Morgan *et al.* 1987, Nagy *et al.* 1987). Farther upstream near Norman Wells, where active oil exploration has occurred in the Mackenzie River, the range is 4-148 ng/g.

Steinhauer and Boehm (1992) report similar values (3-32 μg/g) to those found in the lower Mackenzie for Alaskan rivers along the Beaufort Sea coast (Annex Table 10·A5). Values reported for the nC_{12}-nC_{20} fraction in sediments from Alaskan ponds and lakes (Snyder-Conn and Lubinsky 1993) are also within this range when the moisture content and limited carbon range reported are taken into account.

The data available for petroleum hydrocarbon levels in freshwater biota are very limited. There are only measurements for levels in three crustaceans and one emergent aquatic plant from Alaska (Snyder-Conn and Lubinsky 1993). The levels in these animals are quite similar, with a range of 0.22-2.26 μg/g (wet weight) for the crustaceans and 0.7-10 μg/g for the plant (Annex Table 10·A8).

10.5.3. Levels in the terrestrial environment

Data on the concentrations of petroleum hydrocarbons in soils are only available for Russia. Background levels appear to be approximately 10-40 μg/g (Annex Table 10·A7). A great potential for pollution of the Russian Arctic terrestrial environment is associated with spills and leakage from oil pipelines. An example is the 1994 Komi spill in the drainage area of the Pechora River. A survey of the spill site made in the autumn of 1994 showed that the total area affected by accidental and operational releases, was 1320 000 m^2, with a mean level of soil contamination of 3.25 mg/g. Oil concentration in the soil five meters from the main leakage site was 15% of the mass (150 mg/g).

Since most of pipelines in this region are old and in poor condition, other oil spills in the region can be expected.

10.6. Levels of polynuclear aromatic hydrocarbons (PAHs)

Polynuclear aromatic hydrocarbons (PAHs) comprise, in a narrow sense, all aromatic hydrocarbon molecules containing three or more benzene rings. Often di-aromatic compounds such as naphthalenes and biphenyls are also included in the PAHs. In general, the two main contributors to PAHs in the environment are fossil fuels, including crude oils, and incomplete combustion of organic materials, such as wood, coal, and oil. Most crude oils contain small amounts (<1-10%) but differing proportions of more or less the same set of PAHs. The PAH content of shale oils and coal-derived synthetic oils may be as high as 15% (Neff 1985). After refining, PAHs are more abundant in the heavy fraction (bunker oil, asphalt) than in the lighter products (gasoline, diesel oil).

Many naturally synthesized compounds have a polynuclear aromatic basic structure. Such substances are especially found in bacteria, fungi, and higher plants and in the pigments of certain insects. These oxygen- or hydroxy-substituted compounds are readily reduced to PAHs under anaerobic conditions in bogs and sediments. Steroids and terpenoids have also been found to act as precursors for PAHs. Among the PAHs formed by natural processes are perylene, retene, and phenanthrene homologues. PAHs are also formed naturally during forest fires and volcanic eruptions.

The relative composition of the PAHs in a sample may give some indications as to origin. As a rule, alkylated homologues are more abundant than unsubstituted compounds in crude and refined oils, while the opposite is true for PAHs formed by combustion.

Although more soluble in water than the paraffins, PAHs have a very low solubility in water. Aqueous solubilities of PAHs decrease with increasing molecular weight and ring size, increase with temperature (about two- to five-fold) over the range 5-30°C, and decrease with increasing salinity or ionic strength (salting out effects about a factor of two between 0 and 36 parts per thousand). Increases in levels of dissolved and colloidal organic matter in natural waters tend to increase the apparent solubility of PAHs, and similarly their levels in sediments tend to increase with increasing total organic matter. Due to their hydrophobic nature, PAHs in natural aquatic systems rapidly tend to become associated with particulates, and consequently, sediments are the most important reservoir of PAHs in the environment. Affinity for the particulate phase generally increases with molecular weight/size of the PAH molecule. An important fate process of PAHs is photo-oxidation in the atmosphere and in the upper portions of the water column. The combined effects of low air and water temperatures (low rates of volatilization and biodegradation, particularly for low molecular weight PAHs) and reduced solar radiation lead to higher persistence of PAH compounds in the Arctic than in more temperate regions of the Earth.

PAHs are formed in three general ways: 1) high temperature pyrolysis of organic materials; 2) low to moderate temperature diagenesis of sedimentary organic material to fossil fuels; and 3) direct biosynthesis by bacteria, plants, and fungi. Major industrial sources of PAHs include production of acetylene from natural gas; pyrolysis of wood to form charcoal, wood tars, and carbon black; pyrolysis of kerosene to form benzene, toluene, and other organic solvents; manufacture of electrolytic aluminum using graphite electrodes; coke production; gas production from petroleum; coal gasification; production of synthetic alcohol; and oil refinery operations. Emissions rich in PAHs are also produced by incineration of

industrial and domestic wastes, forest and grass fires, power generation from fossil fuels, and the combustion of fuels in internal combustion engines. The sources of PAHs influence their bioavailability for uptake by organisms, with the PAHs present in oil being more bioavailable than those with a pyrogenic origin (McElroy *et al.* 1989).

Anthropogenic activities are generally accepted as the most important source of PAHs released into the environment. These anthropogenic releases can reach levels at which the environmental concentrations of PAHs are of concern with regard to threats to the ecosystem and to human health. For example, in recent years anthropogenic releases of the PAH benzo[a]pyrene from vehicle emissions, the burning of fossils fuels for heating, and other sources have led to this compound becoming one of the main pollutants threatening the air quality in Russian cities (Anon. 1992). This threat is especially noticeable in cities with unfavorable meteorological conditions, such as the Arctic cities on the Kola Peninsula where, as illustrated in Table 10·11,

Table 10·11. Levels of benzo[a]pyrene (ng/m^3) in the air of Russian cities on the Kola Peninsula in 1991 and 1993 (NEFCO 1995).

City/Town	Value	1991	1993
Apatity	Mean[a]	0.5	–
	Maximum[b]	1.4	2.7
Kandalaksha	Mean	2.2	–
	Maximum	5.8	9.5
Kovdor	Mean	0.8	–
	Maximum	2.5	1.8
Monchegorsk	Mean	2.2	–
	Maximum	8.6	8.1
Murmansk	Mean	1.1	–
	Maximum	4.0	3.4
Nikel	Mean	0.5	–
	Maximum	2.9	2.2

Maximum Permissible Concentration (MPC) = 1 ng/m^3.
a. Mean = annual average.
b. Maximum = the highest concentration measured during the year over a 20-minute period.

Table 10·12. Annual inputs of benzo[a]pyrene and PAHs to the aquatic environment from various sources.

Source	Input in tons (US) per year B[a]P	Total PAHs
Biosynthesis	25	2700
Petroleum spillage	25-30	170000
Domestic/industrial wastes	29	4400
Surface runoff	118	2940
Fallout/rain	500	50000
Total	697-702	230040

the regulatory Maximum Permissible Concentration (MPC) for benzo[a]pyrene is commonly exceeded, at times by quite a large amount.

Table 10·12 gives rough estimates of the inputs of benzo[a]pyrene and total PAHs to the aquatic environment from various sources (from Neff 1979). Petroleum spillage is the largest single source primarily due to the enrichment of PAH compounds in petroleum and the chronic and inevitable spillage of petroleum. Crude oil typically contains about 7-34% by weight of total aromatics with about 1% of this being in the naphthalene homologous series.

Most environmental compartments contain a complex mixture of literally hundreds of PAH compounds. The most important ones studied have been those known or thought to be carcinogenic to mammals and aquatic life (not all PAHs are carcinogenic). Of the approximately 16 PAH compounds shown to act as carcinogens, the top five in order of potency are benzo[a]pyrene > benz[a]anthracene > benzo[b]fluoranthene and ideno[1,2,3-cd]pyrene > benzo[ghi]perylene.

PAHs bioaccumulate in fish and other aquatic organisms, with sediment apparently the major source (McElroy *et al.* 1989). However, PAHs are less prone to bioaccumulation or biomagnification than the organochlorines, partly because of metabolic degradation of PAHs in top predators and their prey (Macdonald and Bewers 1996).

10.6.1. PAHs in the marine environment

Concentrations of PAHs measured in Arctic marine water, suspended particulates, bottom sediments, fishes, and invertebrates are presented in Annex Tables 10·A11, 10·A12, 10·A14, 10·A19, and 10·A20, respectively.

In measurements made during the Beaufort Sea Project, the concentrations of PAHs in the waters of the southern Beaufort Sea were found to vary from 13.0 to 45.0 ng/L (chrysene equivalents), and inversely with salinity (Wong *et al.* 1976). PAH concentrations in nearshore waters were much more variable than PAHs in deeper waters, possibly due to the influence of the Mackenzie River which flows through regions with known fossil fuel deposits and natural seepages. In Tuktoyaktuk Harbour, Northwest Territories, Thomas (1988) used an *in situ* pumping system to obtain high-volume samples for PAH analyses. Values for total PAHs ranged from 10 to 105 ng/L. Naphthalene was the most abundant PAH indicating fresh petroleum, probably from industrial activities in the harbor, as the source.

Erickson *et al.* (1983) measured dissolved PAH concentrations at a drilling location in the southern Beaufort Sea. Near-surface values ranged between 11 and 369 ng/L; near-bottom (16 m) samples were also highly variable (5-278 ng/L). Perylene was the most prominent PAH compound in most samples.

Wong *et al.* (1976) reported that total PAHs (defined as the sum of chrysene, benzanthracene, perylene, phenanthrene, and pyrene and certain isomers) in marine sediments from the Beaufort Sea ranged from 120 to 2890 ng/g. This compares with ranges of 160-1030 ng/g for total PAHs in coastal and offshore Alaskan sediments (Steinhauer and Boehm 1992) and 8-7800 ng/g for various locations across the Beaufort Sea shelf in both 'background' areas and areas near offshore oil and gas developments (Thomas *et al.* 1990). Peake *et al.* (1972) reported PAHs in quantities ranging from 100 to 1100 ng/g in surficial sediments over an area of approximately 10 000 km^2 immediately adjacent to the Mackenzie Delta. Concentrations of benzo[a]pyrene in the sediments from the Beaufort Sea, Barents Sea, and Mackenzie Shelf are relatively consistent (Annex Table 10·A14).

Examination of the PAH assemblages in sediment samples from various Arctic locations is consistent with multiple sources. Petrogenic, diagenetic, and pyrolytic sources of PAHs are indicated, but the relative importance of each source varies with location. For example, the Alaskan sediments are dominated by alkylated naphthalenes and phenanthrenes and have relatively low amounts of the four- and five-ring PAHs, indicating that petroleum hydrocarbons are an important source of the PAHs in these sediments (Steinhauer and Boehm 1992). By contrast, there is a strong contribution from combustion sources to the PAHs in Barents Sea sediments (Yunker *et al.* 1996).

In the Canadian Beaufort Sea, the prominence of the four- and five-ring PAHs at generally much lower concentrations indicates mixed pyrogenic/petrogenic sources of PAHs (Erickson *et al.* 1983, Yunker *et al.* 1993). Often the petrogenic signal is masked by that of the pyrogenic PAHs.

The concentrations of total PAHs in Russian Arctic marine waters range from the detection limit to 88.4 ng/L (An-

nex Table 10·A11). Phenanthrene and naphthalene are the most abundant PAH compounds in samples collected in the Pechora, White, and Kara Seas. Naphthalene is most common in river estuaries and coastal areas where some anthropogenic influence is most likely. In the Bering and Chukchi Seas, measurements of benzo[e]pyrene, benzo[a]pyrene, benzo[k]fluoranthene, and benzo[b]fluoranthene indicate enrichment of these compounds in the surface layer relative to near-bottom waters (Irha *et al.* 1992).

PAH concentrations in sediments from the Russian marine environment are generally much lower than in those from the Canadian or American Beaufort Sea. Concentrations range from 4 to 3422 ng/g, but most are less than 500 ng/g (Annex Table 10·A14). The most abundant PAHs are generally those with five or more aromatic rings and alkylated naphthalenes, indicating a mixed pyrolytic/petrogenic source of PAHs.

PAH profiles for two cores from the Kara Sea (Loring *et al.* 1996) suggest little variation in the top 20 cm. All depths along the cores, (except for the surface (0-1 cm) layer where naphthalene/alkyl naphthalene were most abundant), were dominated by perylene.

Based on 1989-1995 monitoring data from the Regional Centre 'Monitoring of the Arctic' (Melnikov *et al.* 1996), the Ob-Yenisey Shelf of the Kara Sea is the most contaminated area of the Russian Arctic seas for PAHs. Among 20 individual PAH compounds monitored in the Russian marine environment, 13 compounds were detected in all samples of bottom sediments from this area. Benzo[a]pyrene, the most carcinogenic PAH, was detected in 60% of the samples. However, it should be noted that the mean concentration of benzo[a]pyrene in the Pechora Sea bottom sediments was 2.5 times higher than in Ob-Yenisey Shelf sediments (0.61 ng/g vs. 0.25 ng/g).

Sediments near Spitsbergen are also enriched with PAHs compared with those of the Russian marine environment. Concentrations of total PAHs varied between 1565 and 8092 ng/g and are probably due to contamination by petroleum or petroleum products as indicated by the dominance of alkyl-substituted naphthalenes in the PAH profiles.

Total PAHs in surficial sediments from several locations along the northern Norwegian coast occur in a wide concentration range, 44 - 46 200 ng/g, with the high molecular weight (combustion) PAHs being the most abundant. Total PAH concentrations measured along two cores (approx. 50 cm) from this region provide uniformly low values (3-39 ng/g) below an enriched surface (0-2 cm) layer (44-274 ng/g) (Næs *et al.* 1995).

A qualitative assessment of the potential biological effects of marine sediments containing PAHs can be made by using the guidelines set out by Long and Morgan (1990) and Long *et al.* (1995). For a number of contaminants, including 13 PAHs, these researchers provide concentrations at which it is judged that biological effects are possible, but rare, and levels where effects are likely. These levels are obtained by listing in ascending order all available concentrations for a specific contaminant at which biological effects in marine sediments have been observed. The level of possible effects (the 'Effects Range-Low') is estimated as the lower 10th percentile concentration for each contaminant; the level of probable effects (the 'Effects Range-Medium') is estimated as the 50th percentile concentration. It should be noted that combustion-derived PAH components are much less bioavailable (low desorption from the soot particles) than petroleum-derived PAHs (Butler and Crossley 1981, McGroddy and Farrington 1995).

These guidelines have been compared with the PAH concentrations in marine sediments summarized in Annex Table 10·A14. No values over the Effects Range-Medium value were found. However, there are a number of observations that exceed the Effects Range-Low values (Table 10·13). These exceedances are limited to only four of the PAHs evaluated, (naphthalene, acenaphthylene. fluorene, and phenanthrene), and to locations in the Barents Sea, Svalbard, the Beaufort Sea, and Tuktoyaktuk Harbour.

Table 10·13. The Effects Range - Low (ER-L) guideline values for each of 13 PAHs (from Long *et al.* 1995), the number of observations that exceed these values, and the total number of observations.

PAH Compound	ER-L guideline value, ng/g dry wt	Number of observations > ER-L	Total number of observations
Naphthalene	160	22	250
2-methyl naphthalene	70	0	91
Acenaphthene	16	0	239
Acenaphthylene	44	18	238
Fluorene	19	32	279
Phenanthrene	240	28	265
Anthracene	85	0	265
Fluoranthene	600	0	281
Pyrene	665	0	281
Benz[a]anthracene	261	0	281
Chrysene	384	0	258
Benzo[a]pyrene	430	0	279
Dibenz[a,h]anthracene	63	0	279

On a world-wide basis, background levels for PAHs in sediments appear to be in the range from a few tens to approximately 500 ng/g. The total PAH content of northwest Atlantic sediments farther than 1000 km offshore was 18-97 ng/g (Windsor and Hites 1979), whereas total PAH levels were about 500 ng/g in Amazon River sediments (Laflamme and Hites 1978) and about 100 ng/g in the sediments of King Edward Cove on the subantarctic island of South Georgia (Platt and Mackie 1979). In an unpolluted Ontario lake, eighteen PAHs were found at concentrations ranging from 6 to 38 ng/g (Brown and Starnes 1978) and 5 ng/g of benzo-[a]pyrene were reported on the west coast of Greenland (Mallet *et al.* 1963).

Consequently, several areas of the Arctic have an elevated PAH baseline relative to global background concentrations. The Beaufort Sea Shelf is one such area of high PAHs in the Arctic region. The main source of these PAHs appears to be the Mackenzie River and certain Alaskan rivers that flow through regions with known fossil fuel deposits, natural hydrocarbon seepages, and burned-over areas. Also, several relatively small bay and fjord areas associated with local anthropogenic sources, such as Longyearbyen and Barentsburg on Svalbard (Holte *et al.* 1994), and harbors along the Norwegian Arctic coast (Konieczny 1996), have high levels in relation to global background concentrations. Taking into account the high levels of petroleum hydrocarbons pollution found in the Kola Fjord, it is likely that this area also has a high level of PAH contamination; however, assessment of this area was prevented by lack of data.

Total PAHs (defined as the sum of chrysene, benzanthracene, perylene, phenanthrene, and pyrene and certain isomers) in the tissue of southern Beaufort Sea fish (pomfret, least cisco, and Arctic cisco) averaged 21 ng/g wet weight and ranged from 9 to 31 ng/g wet weight (Wong *et al.* 1976). It is difficult to compare these baseline data with those from other areas and from more recent studies because of differences in the ages, species, migration patterns, food habits, and chemical compounds involved, and because of changes in analytical techniques and methods of quantification.

Additional data are presented for various species in the Arctic regions of Canada, Norway, Russia, and the United States in Annex Table 10·A19. Consistently high concentrations were noted in Arctic and starry flounder

from Tuktoyaktuk Harbour by Thomas (1988). The most abundant PAH compound in the tissues of these fishes was naphthalene, probably reflecting the chronic exposure to petroleum hydrocarbons in the water and sediment of the harbor.

The total PAH concentrations found in various benthic invertebrates and plankton/neuston of the Arctic are summarized in Annex Table 10·A20. Concentrations are highly variable. The highest accumulation of PAHs in benthos generally occurs in areas where PAHs are enriched in the sediments. Consequently the highest values are reported for Alaska and Siberian coastal waters. The high molecular-weight PAHs were the most abundant PAH compounds in most benthic samples and in the plankton/neuston.

The values observed for total PAHs in Arctic marine biota are similar to those reported for biota from locations thought to reflect background conditions outside the Arctic. For example, total PAHs in fish caught from waters near Ocean Weather Station 'P' (pristine open-ocean location) were 20-82 ng/g wet weight for salmon and 184 ng/g wet weight for tuna. Sullivan (1974) gives a range of 0-230 μg/g wet weight for benzo[a]pyrene in shrimp, mollusks, and oysters.

10.6.2. PAHs in the freshwater environment

Concentrations of PAHs measured in Arctic freshwater suspended particulates, freshwater bottom sediments, and freshwater fishes are presented in Annex Tables 10·A10, 10·A13, and 10·A17, respectively.

Very few data are available on concentrations of PAH compounds in freshwater. Erickson and Fowler (1987) found <30 ng/L total PAHs in Mackenzie River water. Nagy *et al.* (1987) also analyzed Mackenzie River water, and report total values of 50-1800 ng/L. Peters *et al.* (1995) analyzed ice from the Agassi Ice Cap, Ellesmere Island, representing the past 36 years of deposition. Total PAH concentrations (only naphthalene, acenaphthene, fluorene, phenanthrene, fluoranthene, pyrene, and chrysene could be detected) ranged from 36 to 660 ng/L. Naphthalene was the most abundant PAH compound present in all cases, accounting for an overall mean of 88% of the total PAHs. No PAHs with five or more aromatic rings were detected. Gregor (1989) also measured PAHs (7 compounds) in ice from the Agassiz Ice Cap and reported concentrations in the range of the detection limit, 125 ng/L. McNeely and Gummer (1984) analyzed surface snow samples from coastal Ellesmere Island for 17 PAH compounds. None was found to be above the detection limit of 20 ng/L.

In freshwater sediments, total PAH concentrations in the Arctic are highly variable and probably reflect a combination of long-range transport and local industrial and natural sources occurring in each watershed. Ranges reported for Canadian, Finnish, Norwegian, Russian, and US lakes and rivers are 3-530 ng/g, 81-1050 ng/g, 116-6975 ng/g, <10-1674 ng/g, and <10-708 ng/g, respectively.

Burbot muscle and liver tissues have been analyzed for PAHs in Canada, Russia, and Finland. Concentrations in liver are generally higher than in muscle. The highest values are reported for fish caught at Norman Wells, NWT (muscle, not detected-1114 ng/g; liver, not detected-2423 ng/g) (Lockhart *et al.* 1987, Morgan *et al.* 1987). The concentrations of total PAHs reported in burbot liver from Finland and Russia are 445-727 and 26 ng/g, respectively; a muscle tissue level of 41 ng/g was reported from Finland.

Small numbers of whitefish have been analyzed for total PAHs in the major rivers of Arctic Russia. Values for muscle and liver are 3-76 ng/g and 5-305 ng/g, respectively. The most abundant PAH compound is anthracene, indicating a chronic source of hydrocarbons at most locations sampled.

Phenanthrene, anthracene, fluoranthene, and pyrene were found to be present in suspended matter of practically all large rivers of the Russian Arctic, with a the range of mean concentrations of 0.06-10.3 ng/mg. Benzo[a]pyrene was found only in the Ob River (mean = 0.4 ng/mg). It should be noted that this toxicant was detected in 63% of the fish tissue samples from this river, but was not detected in fish samples from the other Russian rivers. The mean benzo[a]pyrene concentration in livers of salmonids from the Ob River was 0.54 ng/g (Melnikov *et al.* 1996).

10.6.3. PAHs in the terrestrial environment

Information on PAHs levels in the terrestrial environment of the Arctic is extremely scarce. The only terrestrial mammal for which PAH data are available in the Arctic is the reindeer (*Rangifer tarandus*) in Russia. Total PAH concentrations are uniformly low (0.5-68 ng/g) and decrease in the order liver > kidney > muscle (Annex Table 10·A15).

The concentrations of total PAHs in Arctic birds in Russia range from 2 to 29 ng/g in muscle and from detection limit to 444 ng/g in liver. Naphthalene and alkylated naphthalenes are the most abundant PAHs in most samples, for both muscle and liver. Total PAHs in bird carcasses and eggs from Alaska are in the range <10-260 ng/g (Annex Table 10·A18).

Total PAH data for a limited number of plant and mushroom samples in Russia and the USA indicate a range of detection limit to 431 ng/g (Annex Table 10·A16).

10.7. Environmental effects

The following section is directed at giving a generalized overview of documented effects of petroleum pollution on various types of Arctic ecosystems. Emphasis has been placed on the effects observed from accidental or experimental exposure to petroleum contamination directly in the natural environment or in artificial ecosystems (mesocosms). The large body of information available from toxicity testing of oil and individual hydrocarbons with single test organisms is not treated.

It is apparent that the wealth of information on environmental effects of petroleum stems mainly from the marine system where the massive oil spills have occurred, and most of this is from lower latitudes. This information has been extensively reviewed, some of the more recent being GESAMP (1993) and SFT (1990). For other compartments of the Arctic environment, the information is less comprehensive and systematic. In particular, documentation is scarce concerning the effects of oil on Arctic terrestrial ecosystems, and especially on terrestrial fauna.

10.7.1. Effects in terrestrial ecosystems

These ecosystems comprise all communities on dry land and along the shores of rivers and lakes. Bogs, swamps, and other wetland types are somewhat between terrestrial and limnetic systems, but will be classified among the former in this context, primarily because the vegetation and animal life tend to be dominated by terrestrial forms. The most widespread of the Arctic terrestrial ecosystem types are tundra and taiga.

Factors influencing the type and duration of oil spill effects in Arctic environments are the severe, long, dark win-

ters and the short, cool summers. The Arctic and much of the northern subarctic region are also characterized by very wet soils underlain at shallow depth by permafrost containing large quantities of ice. The vegetative cover is almost complete, but shallow, and extremely vulnerable to surface damage (Linkins *et al.* 1984). Productivity is low and nutrient recycling and energy flow are limited. These conditions have a strong influence on the mobility of spilled oil on and in the ground, on photo- and bio-degradation of the oil hydrocarbons, and on revegetation and recovery of damaged communities.

An oil spill on land will, in general, be more confined than oil spilled in aquatic ecosystems. Oil spilled on tundra soils may, however, migrate both laterally and vertically (Linkins *et al.* 1984). The rate and extent of oil flow over the surface will depend on a number of factors, such as type and amount of oil, plant cover, slope, soil texture and water content, temperature, wind, and sorptive capability of the ground for oil. Low temperatures and high oil viscosity as well as chemical weathering of the oil will reduce spreading. When petroleum is introduced to terrestrial systems, several modifying processes start working on the hydrocarbons, such as evaporation, photochemical degradation, bacterial mineralization, and mixing with and burial in soil and snow.

Absorption/adsorption on the soil affects both horizontal and vertical movement. This factor is dependent on plant cover. Various studies show vast differences in absorptive capacity of different plant covers. Mosses have been found to have high absorption capacity. MacKay *et al.* (1974) found that 160 gallons (0.606 m^3) of crude oil contaminated a surface area of 4.7 m^2 of moss-covered tundra compared with up to 101 m^2 where there was no moss cover.

Cracks in the soil above the permafrost may lead oil all the way down to the permafrost and horizontal spreading in deeper soil layers may occur (Linkins *et al.* 1984). The water content of the soils also greatly affects absorption and vertical movement. Water-logged soils over permafrost, typical for Arctic tundra, reduce the degree of oil penetration. Oil movement on snow cover is very different. MacKay *et al.* (1975) found that crude oil at 0°C spilled on snow was readily absorbed by the snow. However, a spill of hot oil, for example from a pipeline rupture, tended to form channels in the snow transporting the oil along the ground underneath and contaminating relatively large areas.

Reported damage of oil to terrestrial vegetation varies considerably from one spill to another. Wein and Bliss (1973) found that all actively growing plant tissues were destroyed in Arctic wetland plant communities after oil spills. Mosses were eliminated almost entirely, whereas sedges recovered rapidly. Similar results were found during a tundra oil spill by Freedman and Hutchinson (1976). It appears that if oil exposure is limited to above-ground plant parts, as for example in a spray spill, the damage can be severe, but is generally of short duration because resprouting from protected buds and roots occurs (cf. Linkins *et al.* 1984 and references therein). If, however, the oil penetrates the soil and contaminates the root system, vegetation death and/or reduced regrowth in the following seasons occur. These effects are often due to declines in plant nitrogen content, reduced viable root biomass, and changes in mycorrhizal structure (Linkins *et al.* 1984). Sensitivity of vegetation to oil pollution is, therefore, to some extent, related to the type of root system it possesses. Plants with shallow, sparsely developed root systems are expected to be more vulnerable (Klokk 1986) than those having a well-developed stock root (Baker 1979). Deneke *et al.* (1975), however, report examples of species with stock roots being sensitive to oil, presumably because they are incapable of regrowth by root ramification to uncontaminated sites.

The specific causes for long-term decline or lack of revegetation after oil spills are not clear, but are believed to be caused by oil-associated changes in edaphic factors influencing plant regrowth (Everett 1978). In addition the residence time of toxic oil fractions in tundra soils may be very long, up to 30 years, according to references given by Linkins *et al.* (1984). Hence the effects of oil in the soil on plant regrowth and seedling establishment most probably are due to a combination of oil toxicity and altered edaphic factors. Some structural (increased root biomass) and functional (microsomal aryl hydrocarbon hydroxylase in the roots) adaptations of Arctic plants to long-term oil exposure have also been reported (cf. Linkins *et al.* 1984).

Studies after an oil spill along the Trans-Alaskan Pipeline have shown that revegetation on the tundra may be enhanced by rehabilitation (Brendel 1985). Of the stimulative means applied, heavy application of a fertilizer high in nitrogen and phosphorus combined with soil tilling gave the best grass regrowth. Soil content of oil appeared to be reduced significantly through fertilization and aeration.

Effects on terrestrial animals due to exposure to oil contamination are poorly understood. Mammals, like birds, are likely to be affected mainly by the soiling effect of oil on the pelage. There is no written assessment of the potential risk from drinking of contaminated water, and there are, in general, few reports of wild, cold-climate mammals being harmed by oil contamination. Heavy mortality of muskrats (*Ondatra zibethicus*) has been reported after a major oil spill along the St. Lawrence River (Palm 1979, Alexander *et al.* 1981). Experiments have shown that oiling caused immediate significant increase in heat production with accompanying elevated food intake to compensate for loss of insulation (McEwan *et al.* 1974). It is unlikely that muskrats exposed in this way would survive under natural low temperature conditions. One must also assume that the same will be the case for other mammals relying on fur for thermal insulation.

10.7.2. Effects in freshwater ecosystems

These systems comprise creeks, rivers, lakes, and the inner freshwater parts of estuaries. Published assessments of the ecological effects of oil spills in Arctic freshwater ecosystems are still limited. Most of the work has concentrated on phytoplankton and zooplankton in lakes and ponds (cf. references in Miller *et al.* 1986) and on biodegradation rates in Arctic and subarctic lakes (references in Miller *et al.* 1986). Experiments by Snow and Scott (1975), Hellebust *et al.* (1975), and others indicate that the direct toxic impact of oil on lake microalgae, in general, is modest, but secondary effects may be significant. These effects may include:

- Increased primary production as a result of death, mineralization, and nutrient release from more sensitive organisms.
- Increased primary production due to heterotrophic dominance accompanied by nitrogen fixation.
- Decreased primary production when heterotrophic dominance is not accompanied by nitrogen fixation.

Stream periphyton has also been reported to increase in biomass on oil contaminated stones (Roeder *et al.* 1975, Rosenberg and Wiens 1976).

Oil spillage may cause widespread damage to freshwater macrophyte communities. Several studies (mostly in cold temperate regions) have shown that perennial plants, including growth forms such as emergent, surface floating, and

submerged, are more tolerant to, and recover more rapidly from, oil pollution than annuals (Baker 1971, Burk 1977). In a review of a large number of freshwater spills, Baca *et al.* (1985) concluded that oil effects on macrophytes in running water are minimal compared with effects on macrophytes in lakes and ponds. In rivers, spills often result in oiling of outer fringing vegetation only, and oil residence times are relatively short. In an oiling experiment in tundra ponds, Barsdate *et al.* (1980) found little damage to the dominant *Carex aquatilis* when oil contaminated only the stems, but severe and repeated damage to leaves when smothered with oil, along with reduced growth of plants in the following years.

Long-term changes in macroinvertebrates of tundra ponds after controlled spills have been documented by Miller *et al.* (1977), Mozley and Butler (1978), and Barsdate *et al.* (1980). The latter reported the results of an artificial oil spill in a Canadian tundra pond. With time, the remains of the oil on the surface adhered to emergent plants, but little oil was found in the sediments or in wet litter. Remaining oil retarded autumn freezing and enhanced spring thawing significantly, but only during the first year. The major effect of the spill was a rapid kill of the pond zooplankton. During the subsequent three years, annually recruited zooplankton were also killed by the small amounts of oil still present in and around the pond.

Sensitivity varies among types of zooplankton, with fairy shrimp and daphnids being most sensitive, and cyclopoid copepods least (O'Brien 1978). Insects seem not to be particularly sensitive to oil toxicity, but considerable mortality by entrapment in surface oil films has been documented (Barsdate *et al.* 1980, Snow and Rosenberg 1975, Shales *et al.* 1989). Several authors (references in Miller *et al.* 1986) have studied invertebrate compositional changes following experimental spills in small streams. Nauman and Kernodle (1975) found a marked decrease in total abundance and species richness along the side of an Arctic creek affected by a prolonged diesel oil seepage. Chironomids were the most affected. Rosenberg and Wiens (1976) documented a dramatic increase in macrofauna drift in a creek in the northern Yukon after a small experimental oil spill. Densities of various invertebrate groups were reduced by 80-100% following the spill. Short-term drift of macroinvertebrates was also documented following controlled oil spills under the ice in an Arctic and a subarctic stream (Miller *et al.* 1986). No mortality was observed, but several species responded by drifting from the oil-impacted areas for the first few days. Colonization on rocks was also reduced during the winter following the spill.

The effects of oil on freshwater vertebrate species (fish, birds, and mammals) have received far less attention than effects on these species in marine ecosystems. Oiling of birds and mammals in freshwater ecosystems is also likely to be much more localized than in the marine system.

There are numerous reports on mortality, various sublethal effects, and fish tainting associated with oil spillages, but nearly none from Arctic freshwater environments. To our knowledge, no gross mortality of freshwater fish due to oil pollution has been reported. Lockhart and Danell (1992) investigated two incidences of tainting of fish in Canadian Arctic rivers, one by effluent from a synthetic crude oil production plant and one by diesel fuel spilled when a truck overturned. These cases both resulted in complaints that downstream populations of whitefish were tainted with an oily taste. In both of these instances, supplies of the spilled material were available, and laboratory exposures to fish confirmed the ability of the spilled materials to produce tainting.

Oil spills in areas of congregating birds may be particularly damaging. Large numbers of ducks, geese, and herons were reported killed following an oil spill on the St. Lawrence River (Alexander *et al.* 1981). Oil damage to birds stems mainly from plumage fouling, causing reduced insulation, and from ingestion of oil during plumage cleaning and consumption of oil contaminated food. Following the *Exxon Valdez* oil spill, numerous bald eagles (*Haliaeetus leucocephalus*), nesting in the affected area, were reported killed from ingestion of oil-contaminated dead birds and sea otters (*Enhydra lutris*) (Townsend and Heneman 1989). Other predatory birds assumed to be threatened included peregrine falcons (*Falco peregrinus*), which also nested in the area. Oil spills in smaller streams and ponds are likely to cause only modest and local damage due to the fact that the number of birds utilizing such areas is usually low.

10.7.3. Effects in coastal and marine ecosystems

By far the largest amount of information available on the effects of oil on Arctic organisms and ecosystems is concerned with marine systems. Coastal areas in general have received almost all the largest oil spills recorded as well as numerous smaller spills, plus continuous discharge of oil-containing effluents, and oil from natural seeps. A number of reports and reviews have been produced describing effects on pelagic systems (mainly from experimental spills), on shoreline and benthic hard- and soft-bottom communities, and on fish, birds, and mammals. Several of these are relevant to the Arctic. For more comprehensive treatments of the impact of oil on marine systems in general we refer to the review by GESAMP (1993) and references therein.

Overviews specifically related to the Arctic marine environment have been presented by Malins (1977), Wells and Percy (1985), Thingstad (1990), and Sakshaug (1992). In general, biological effects are similar in tropical, temperate, and Arctic marine environments for related or similar targets (GESAMP 1993). The important environmental differences are those that affect the distribution, composition, and physical state of the petroleum hydrocarbons in space and time, i.e., their bioavailability. Also the rate of biological recovery from oil damage must be expected to be slower due to the slow growth rates, low generation turnover, and high age at maturity which is characteristic of many important Arctic organisms.

10.7.3.1. Plankton communities and productivity

Seasonal variations in light conditions and ice cover are the main factors governing the primary production of the Arctic pelagic ecosystem. This is illustrated in the Barents Sea where primary production may be divided into three seasons (Thingstad 1990):

- Winter, with very little light due to low sun as well as ice cover, and almost no production.
- Spring/summer, when light conditions and surface water stratification combine with high nutrient levels to form the basis for very intense primary production, which provides the basis for intensive zooplankton growth, which in its turn supports intensive grazing by fish.
- Summer, when the nutrient levels are very low and production is modest and based on nutrients remineralized in the photic zone itself.

A special feature of the biology of Arctic seas is the synoptic north-south production sequence along the ice edge as it retreats northward in the spring/early summer period. This se-

quence starts in the north with modest prebloom phytoplankton growth under the ice followed by an intensive surface bloom just along the edge of the melting ice, and then a deep postbloom biomass maximum overlain by oligotrophic water south of the edge where the most intensive zooplankton grazing occurs, which in turn triggers northward feeding migration by fish such as capelin (Thingstad 1990). The most important spring and ice edge bloom algae are diatoms, whereas summer production is dominated by small flagellates.

The Arctic pelagic food chain is characterized by few species, very high individual densities and biomasses, and a strong lipid transfer along the food chain from phytoplankton to fish and mammals, the 'lipid wave' (Falk-Petersen *et al.* 1990). The production season is short and intense, and energy is stored as lipids in higher trophic levels for the long, dark, unproductive season (Lee 1975, Percy and Fife 1980). It can be postulated that petroleum hydrocarbons, being strongly lipophilic, may follow the transferred lipids and, through toxicity or other interference, affect lipid conversion and utilization at higher trophic levels. Such oil effects have, however, not been documented or in other ways verified.

From studies in lower latitudes, it can be concluded that the damage of oil spills to primary producers is likely to be rather modest and of short duration. Major negative effects on primary production have been reported from experimental work (cf. reviews by Vandermeulen and Ahren 1976, O'Brien and Dixon 1976, Johnson 1977), but have not been verified from *in situ* studies. Reduction in primary production has been documented at total oil concentrations of 200-300 mg/L from plastic bag experiments (Anon. 1984), but mainly as an indirect effect of reduced nutrient excretion and shading by the oil slick. The relatively low concentrations of oil found in the productive surface layers of the water after a spill have in some cases stimulated rather than hampered primary production, either directly as found by Gordon and Prouse (1973) or indirectly by reducing zooplankton grazing (Johansson 1980). Laboratory studies have shown differences in oil sensitivity among phytoplankton groups (Daling and Davenport 1982, Dahl *et al.* 1983, Mahoney and Haskins 1980).

The above findings may be relevant for Arctic plankton subjected to oil spills, but direct documentation is sparse. Hsiao (1978) found inhibition of Arctic phytoplankton growth, but also signs of stimulated growth, during several days exposure to 10 ppm of a range of crude oils. Plankton studies are generally not given priority after oil spills because the oil content of the water is usually so low that effects are unlikely (Moe *et al.* 1993).

Field observations on zooplankton have also been made at numerous accidental and experimental spills (Wells and Percy 1985), including some few from Arctic regions. Collectively these studies show that negative biological effects and changes can occur after a spill, but they appear to be short-lived. Zooplankton populations and communities in open temperate waters appear to recover rapidly, largely because of their wide distributions and rapid regeneration rates. Observations during the *Potomac* spill off Western Greenland revealed external contamination and oil in the gut contents of copepods and amphipods, but no apparent biological effects (Maurer and Kane 1978, Petersen 1978). Members of the Arctic zooplankton, which are often primarily copepods, usually have longer life spans than individuals of the same or similar species in temperate waters (Wells and Percy 1985). Hence if any negative effects occur, regeneration must be expected to be slower in the Arctic than in temperate waters.

10.7.3.2. **Benthic communities**

Shoreline and shallow subtidal communities are prime targets of concern during most coastal oil spills. As opposed to open waters, where the concentrations of hydrocarbons are rapidly reduced, massive stranding and contamination may occur. On low energy beaches especially, the loss of harmful components may be very slow. Oil with little weathering may be trapped for decades and slowly released to the ecosystem. Polar intertidal areas are often biologically barren due to frequent ice scouring, which means that the communities that do occur in such areas are, by nature, transitory. Any effects of stranded oil may, therefore, be regarded as just another instance in the recurring pattern of community destruction. The persistent presence of oil may, however, further reduce the community by preventing colonization between the scouring periods.

In more subarctic regions, such as along the northern Norwegian coast, ice scouring is infrequent and the intertidal communities are of a boreal type with more or less dense cover of macroalgae, barnacles, mussels, and several other organisms on hard substrates. In these communities, the impact of stranded oil will be similar to more temperate regions.

A thorough investigation of the immediate impact and recovery of the intertidal communities after the *Exxon Valdez* crude oil spill in Alaska (Stoker *et al.* 1992) showed that, even on the most heavily impacted shores, survivors of the four main groups of organisms, seaweeds (fucoids), barnacles, mussels, and periwinkles, were present just after the spill. One year later the abundance of adults of these organisms on the heavily oiled shores was somewhat less than on lightly oiled shores, but new recruits showed higher densities, possibly in response to an increase in available substrate space resulting from the loss of the organisms that were killed. This demonstrated a very strong recovery potential of the shoreline community through recruitment from nearby unaffected sites. Two years after the spill most oiled shorelines appeared healthy and in a state typical for the prespill communities. Stoker *et al.* (1992) concluded that the rate of recovery of rocky shores after the *Exxon Valdez* accident appeared to be more rapid than on similar shores affected by the *Torrey Canyon* and the *Amoco Cadiz* accidents.

The impact of oil on cold-water, rocky-shore communities may be regarded as one of several factors of disturbance which can destabilize the ecosystem and drive the community in the direction of monoculture (Southward 1982). The effects normally vary greatly among different species. Many organisms resist desiccation during low tide by closing up their shell or outer protection, e.g., mussels and barnacles, and they may survive short-term oil smothering in the same way. Still, they will be suffocated by thick layers of oil (Wikander 1982). Mobile organisms such as crustaceans may escape by seeking deeper water, but escape responses can also cause animals to get stuck in the oil (Bonsdorff and Nelson 1981). In general, crustaceans, and in particular amphipods, have been found to be sensitive to oil spills (den Hartog and Jacobs 1980, Sanders *et al.* 1980, Elmgren *et al.* 1983, Teal and Howarth 1984, Cross *et al.* 1987a, Kingston *et al.* 1995, and others). Large moving populations of scavenging amphipods and mysids are often present in Arctic waters (Wells and Percy 1985, Sakshaug 1992) and may possibly play a similar important trophic role as the krill in the Antarctic (Sakshaug 1992). Oil damage to amphipods may, therefore, have more severe ecological consequences in the Arctic than in warmer regions.

Intertidal macroalgae are considered relatively resistant to oil due to their mucoid surface protection, but heavy mortal-

ities have been reported. After a spill of 1000 tonnes of bunker oil on the Arctic coast of Norway in 1981, all algal vegetation died in the heavily smothered areas (Wikander 1982). On lightly polluted shores, the smothered parts of the algae showed strongly retarded growth during the following spring, but new sprouts on the same plants were apparently healthy. Experience with other cold-water oil spills confirms that the recovery potential of hard-bottom intertidal organisms is high, but that the recovery time is also dependent on other factors, such as wave exposure and substrate topography and texture, which mainly influence retention time of the oil in the system. Oil may become stranded on backshores by storms or spring tide. The residence time of this oil may be extremely long due to gradual asphalt formation (Sergy 1985).

Oil impacts on sedimentary shorelines have been reported from several accidental and experimental spills at lower latitudes (e.g., Sanders *et al.* 1980, Farke *et al.* 1985, Kuiper *et al.* 1983). On exposed shores with gravel or sand, the oil may penetrate to considerable depth during low tide, but as the flora and fauna of these shores are poor, the apparent effects are, in general, minor. On shores affected by patches of oil, redistribution of sediments by strong wave action may remove considerable amounts of the oil over short periods of time (Sergy 1985), but if contamination is considerable, redistribution will only move not remove the oil. On sheltered sandy and muddy shores, the effects can be more pronounced. In addition to initial mortality, delayed mortality related directly or indirectly to the oil being buried in place has been demonstrated (Kuiper *et al.* 1983). Two years after an upper shore diesel oil spill at Spitsbergen, substantial amounts of oil were still present both at the surface and 40-50 cm down in the shore sediments (Gulliksen and Taasen 1982). The effect of the spill on the soft shore community was negligible, however, mainly because the shore sediments naturally contained almost no flora and fauna. Nonetheless, the only significant animal species present, the amphipod *Gammarus setosus*, disappeared from the oiled sediments. Little oil penetration occurs if the sediments are water-saturated (Farke *et al.* 1985), but even then the bioturbation by animals, such as lugworms (*Arenicola marina*), may transport oil to significant depths (Kuiper *et al.* 1983).

Most of our knowledge on the effects of oil on subtidal sea floor biota stems from spills or chronic oily discharges (e.g., oil contaminated drill cuttings) in temperate waters, but the immediate effects of oil on Arctic biota will probably not differ greatly from those observed in comparable temperate habitats (Wells and Percy 1985). The severity of impact will depend on essentially the same physical and biological variables.

Reported effects on subtidal benthic communities differ strongly from one spill to another. After the BIOS experimental oil spill, no apparent short-term effects on infauna community structure were revealed (Cross and Thomson 1987). Slight effects on some few species were detected on a longer term from oil transported down with particles, (Cross *et al.* 1987a). Biomass, diversity, or reproduction in subtidal macroalgae were not affected by the oil (Cross *et al.* 1987b). A survey four months after the *Braer* accident off Shetland in 1993 (Kingston *et al.* 1995) revealed little or no impact on the benthic animal community, even at oil levels far above those that had led to significant effects in other situations. The most likely explanation was that the *Braer* oil was of such low toxicity as to not disrupt the community structure, or that the survey was carried out too soon after the spill to enable the full effects to be manifested. In contrast, the *Florida* accident in Buzzards Bay (USA) in 1969 caused far more severe impact (Sanders *et al.* 1980). Significant animal mortality occurred within hours after the spill, and community structure instability persisted for between five and ten years. There is reason to assume that the time scale of impact and recovery from oil contamination in Arctic marine bottoms will be longer than in warmer regions. As outlined above, the removal of oil by natural processes of dispersion and degradation, necessary prior to biological recovery, will in general proceed more slowly in the Arctic than elsewhere. Furthermore, the slow generation turnover and long life span of Arctic benthic organisms suggest that the recovery rate itself may be lower than in warmer regions.

10.7.3.3. **Sub-ice communities**

A characteristic and unique feature of the Arctic marine ecosystem is the 'inverted' benthic community established on the underside of the polar ice cover. Thorough descriptions of this system are given by Dunbar (1985) and Sakshaug (1992). It contains all trophic levels from bacteria and specialized ice algae via crustaceans to fish and is an important nutrient basis for seabirds, seals, and polar bear. Several of the organisms have their entire life cycle tied to the ice.

Oil spilled under multiyear ice will remain practically unchanged until the ice thaws, causing a far longer exposure period to toxic components for the sub-ice community when it is impacted by an oil spill than for an open-water or shoreline community exposed to spilled oil (Sakshaug 1992). The uneven underside of the ice will confine the oil and to some extent prevent extensive spreading. If further freezing seals off the oil, any effects may be delayed until the ice thaws. Release of almost unweathered oil and of toxic photo-oxidation products of the hydrocarbons during thawing may seriously impact the important ice edge production sequence (Thingstad 1990, Sakshaug 1992).

Documented effects of oil on the ice community are sparse. Effects on algae and meiofauna of a sub-ice ecosystem were studied in the BIOS project. No adverse effects were detected for algal density, biomass, or productivity after a moderate under-ice application of crude oil (Cross 1987). Also, all the major faunal groups, the polychaetes, copepods, and nematodes, appeared unaffected by the crude oil alone, but the two former groups showed strongly reduced densities when the oil was chemically dispersed at application (Cross and Martin 1987). Exposure of the ice amphipod, *Gammarus wilkitskii*, to water-accommodated fractions of crude oil gave only modest effects (Killie and Gulliksen 1994). However, since amphipods generally are considered particularly sensitive to oil and easily get caught in an oil film (Bonsdorff and Nelson 1981), it is expected that oil under ice may well cause high amphipod mortality through trapping and smothering (Bushdosh 1978, Cross 1982).

10.7.3.4. **Fish, birds, and marine mammals**

There are numerous studies dealing with the lethal and sublethal effects of oil on fish, mostly from laboratory experiments (cf. reviews related to the Arctic, e.g., Rice 1985, Moe *et al.* 1993, Evenset and Hansen 1994). Very little of the available information deals directly with the effects from spills in Arctic waters. However, somewhat more is available from laboratory experiments on fish species whose distributions include the Arctic seas. These studies document the toxicity of oil and its components to a range of functions in individual fish. They give little reason to conclude that the sensitivity of Arctic fish to oil is different than that of fish from temperate waters (Rice 1985).

Field studies, even after one of the very large oil spills, have generally failed to document any widespread effects on fish, presumably due to some combination of several factors including avoidance reactions to oil, the relatively low content of toxic hydrocarbons beneath oil slicks, and the highly dynamic nature of fish stocks reducing the possibilities of identifying changes caused by the oil. However, adverse effects have been recorded during chronic exposure of winter flounder (*Pseudopleuronectes americanus*) to sediments contaminated with petroleum at concentrations commonly found to occur under oil spill conditions (GESAMP 1993).

Fish eggs and larvae are, in general, more vulnerable to oil spills than adult fish, partly due to their intrinsically higher sensitivity to oil toxicity, and partly to their higher probability of exposure. The eggs and larvae of many species, such as cod eggs and larvae and herring larvae, develop near the water surface where the concentrations of toxic components from an oil slick are highest. It is also during these stages that their ability to avoid oil by active swimming is low.

In larval cod and capelin, Føyn and Serigstad (1989) found very low tolerance, which was manifested as irreversibly reduced oxygen uptake, even to short (2-24 hour) exposures to 50 μg/L of the water soluble fraction of crude oil. The implication of such an impact on the population level remains a function of the degree of overlap between the fresh oil slick and the shoals of larvae, and this will differ from one spill to another. In general, large proportions of the larvae will have to be destroyed to affect recruitment (Reed and Spaulding in GESAMP 1993).

Effects of oil on seabirds have attracted strong attention and public concern during most oil spills. Clear individual damage has been manifested by loss of thermal insulation capabilities due to fouling of plumage, by toxic effects caused by ingestion of oil with food or during preening of plumage, and by fouling of eggs causing embryonic mortality.

Dose-dependent reduction in thermal insulation of plumage and compensatory increase in metabolic heat production has been demonstrated for several species of seabirds during moderate oil smothering (*e.g.*, Hartung 1967, McEwan and Koelink 1973, Lambert *et al.* 1982, Zacchariassen 1989). There are, however, limits to how much birds are able to compensate for loss of insulation, and in a harsh Arctic environment where the struggle for food is vital, this disadvantage may be fatal for the birds.

Oil ingested from feeding or preening of feathers may impair reproduction manifested as, for example, reduced egg laying, reduced shell thickness, and disturbance of normal estrogen cycles (cf. references in Leighton *et al.* 1985). Depressed growth in young birds ingesting oil has been reported several times for a variety of species (Hartung 1964, Szaro *et al.* 1978, Miller *et al.* 1979). Depressed growth causing the young to fledge late or at a smaller size than normal may be a fatal disadvantage in the short summer of the Arctic (Leighton *et al.* 1985). Other effects of ingested oil reported are increase in body temperature and changes in hormone and detoxification enzyme levels (Zacchariassen 1989), and impaired osmoregulation, anaemia, and adrenal exhaustion (Leighton *et al.* 1985).

Seabirds are particularly at risk from damage from oil because of their social behavior. Arctic regions have specific times and places with very large aggregations of seabirds in connection with breeding, molting, overwintering, and preparation for migration. A single Arctic breeding colony may contain a large proportion of the total standing stock of a certain bird species (cf. Leighton *et al.* 1985 for an overview), and an oil spill in the vicinity of such a colony may, therefore, cause a disproportionally large amount of damage. There is, however, controversy concerning the long-term population level effects on seabirds, and it is highly uncertain whether or not observed oil mortalities are substantial in relation to natural mortality. Because many oil pollution incidents and much of the chronic oil pollution affect birds while they are at sea, it is very difficult to assess the size and the provenance of the populations of various species that are actually at risk (Dunnet 1982, 1987).

There appears to be little relation between the size of an oil spill and the number of seabird casualties. In the *Amoco Cadiz* oil spill, releasing about 250 000 tonnes of oil, only about 4500 birds were killed (Baker *et al.* 1990), whereas the *Exxon Valdez* spill of about 35 000 tonnes of oil is believed to have killed about 250 000 birds (Platt and Ford 1996). Also, a nearly inconspicuous spill on the east coast of Finnmark in 1979 was estimated to have killed 10 000-20 000 birds, primarily guillemots (Barrett 1979). The impact of an individual spill will also differ from one species of seabird to another, as a function of behavioral patterns and sensitivity. A simplified system to rank birds in terms of vulnerability to oil pollution, developed by King and Sanger (1979), identifies alcids, in general, to be most vulnerable, with Atlantic puffin (*Fratercula arctica*), common murre (*Uria aalge*), thick-billed murre (*Uria lomvia*), razorbill (*Alca torda*), and northern gannet (*Sula bassanus*) as particularly vulnerable. Common eiders (*Somateria mollissima*) are also considered vulnerable, whereas gulls and jaegers appear to be the most tolerant.

The effects of petroleum on marine mammals also has received a great deal of attention, particularly due to the high public interest in these animals. The polar seas, and especially the Arctic, are the habitat for a large proportion of the marine mammals of the world (Engelhardt 1985, GESAMP 1993), primarily seals and whales, but also sea otters and polar bears. Being dependent on air breathing, all marine mammals must stay in close contact with the air-water interface and hence easily come in contact with a surface oil slick. This is particularly the case in ice-filled areas where the restricted open surfaces are crucial for the animals and are also the areas where the oil becomes concentrated. Still the documented cases of oil pollution incidents affecting marine mammals are few and questionable.

Information on the effects of oil on whales is limited, but the general impression is that whales are relatively unharmed by contact with oil. There are no records of oil fouling the skin of free-living whales. The reason for this is assumed to be either that whales have the ability to avoid oil slicks or that the quality of the skin prevents adhesion of oil. Reports indicating active avoidance are conflicting (Engelhardt 1985), but several observations in spill situations have suggested that whales do not take notice of oil slicks. In a review of the available literature, Griffiths *et al.* (1987) concluded that both large whales and dolphins are not threatened by transient contact with floating surface oil. While smaller toothed whales were able to avoid thicker layers of oil but not sheens, larger whales have been seen swimming and feeding in oil-stained water. Geraci and St. Aubin (1982) have shown that even prolonged skin contact is not likely to be harmful. The effects of ingested oil on whales remain unknown, but filtering efficiency of baleen whales may be significantly reduced by oil fouling the baleen hairs (Braithwaite *et al.* 1983). However such accumulations seem not to be important to the long-term feeding ability of baleen whales (Geraci and St. Aubin 1982).

The effects of oil on seals are better known. True or phocid seals, including all seals in the northwest Atlantic, as

well as sea lions and walrus, have a relatively coarse and short fur and rely on dermal and subcutaneous blubber for insulation. The danger of heat loss due to oil fouling is, therefore, small. On the other hand, fouling by viscous oil may be a hindrance to swimming, especially in pups, and exhaustion may have been the reason for death in some cases (Warner 1969, Davis and Anderson 1976). There is also some evidence that long-term exposure may cause dermatitis in phocid seals. In fur seals, the pelt forms an important layer of insulation between the skin and the water, and evidence has shown that oiling of the fur significantly increases the heat flux across the pelt (Engelhardt 1985). The most significant pathological effect of exposure to oil recorded in seals is, however, eye lesions, but the recovery rate from these conditions when transferred to clean water seems to be very rapid.

Like fur seals, sea otters and polar bears rely on fur for insulation and also for buoyancy when swimming. Oil contamination may, therefore, be particularly damaging to these animals. Sea otters are peripheral in their occurrence in the Arctic and have been shown to be particularly sensitive with respect to being at the northern limit of their distribution (Kooyman *et al.* 1977, Costa and Kooyman 1982). Even light oiling has detrimental effect on thermoregulation in these animals. In the *Exxon Valdez* accident approximately 2-3000 of the 10-11 000 sea otters in the area were killed on a short term, and probably many more have subsequently died (Waldichuk 1990). Regarding polar bears, severe heat loss and elevated compensatory metabolism have been observed after experimental oiling of their fur (Hurst *et al.* 1982). Furthermore, extensive grooming causes ingestion of oil leading to tissue elevation of hydrocarbons and gradual development of dysfunction and lethal damage in several internal organs. Renal failure has been proposed to be the ultimate cause of death (Øritsland *et al.* 1981).

10.7.3.5. Effects of petroleum activity on marine organisms and systems

The two main threats to marine systems from petroleum activities, apart from oil spills, are discharges from drilling operations and of produced water (see section 10.3.2). Although these discharges involve contaminants which are treated in detail in other chapters of this assessment (e.g., metals), they represent a significant part of the environmental threat arising from oil and gas development and production activities and hence an overview of their effects is given here.

Acute lethal and sublethal biological effects, including *in situ* changes in biological communities due to drill cuttings, drilling fluids, produced waters, and other discharges associated with petroleum development, have been studied extensively since the early 1950s in the United States, and since the late 1970s in northern Europe. A panel of the US National Research Council on Assessment of Fate and Effects of Drilling Fluids and Cuttings in the Marine Environment published a detailed review of scientific studies on the subject (NRC 1983). Neff (1987) also reviewed data on the subject in the context of long-term biological effects of offshore petroleum development. Most of the reported studies in these reviews were carried out in the Gulf of Mexico; only a few dealt with the Arctic environment (e.g., Sifred 1976, Tornberg *et al.* 1980, Northern Technical Services 1981).

For North Sea conditions, several recent reviews of environmental impacts of petroleum activity have been made, (e.g., Gray *et al.* 1990, Kingston 1992, Olsgard and Gray 1995, and Bakke *et al.* 1995). The main difference between the North Sea and the US situations is the extensive use and discharge of oil-based drilling fluids in the former. For Norwegian fields, these discharges peaked around 1985 and ceased completely in 1993 due to legislative restrictions. The focus of concern both in regulations and monitoring has been the effects on bottom communities and fishing grounds around platform sites.

Exploratory drilling with discharge of primarily water-based muds has been performed at several locations along the Arctic part of the Norwegian shelf and in the Barents Sea, but production in the Arctic region has only just started in two locations, the Draugen and Heidrun oil fields in the Norwegian Sea. In the Russian Arctic, the Prirazlomnoye Field in the eastern Pechora Sea is expected to be the first to go into production, at the earliest in 1998.

Monitoring around the Heidrun and Draugen oil fields has given some indications of the effects to expect during drilling at offshore fields in the Barents Sea under current legislation. The water depths at Draugen and Heidrun are around 280 and 350 m respectively, corresponding to the depths to be encountered in the Barents Sea. At Draugen, only water-based muds have been discharged, but still slight bottom fauna changes have been detected in an area of about 15 km^2 size around the drilling site (Myhrvold *et al.* 1995). For Heidrun, where discharges of ether-based muds occurred in 1993 and 1994, slight faunal changes and reduced diversity were detected in an area of about 2-3 km^2 in 1995 (DNV 1996).

These results do not deviate systematically from findings in the deeper, northern region of the North Sea. Around fields which have been in production for several years, and where considerable amounts of oil-based muds and cuttings have been discharged to the sea floor, the chemically contaminated bottom areas have normally covered between 10 and 120 km^2 (Olsgard and Gray 1995, Bakke *et al.* 1995). The situation seems reasonably stable from one year to another. Around younger fields, either under exploration or where production has just started, the contaminated areas are generally less than 30 km^2.

The corresponding areas where biological effects on benthic fauna have been detected, are from nil to 80 km^2 around older fields and generally less than 15 km^2 around younger fields. The first biological changes recorded in such areas have been increased abundance of some species and slight changes in presence and absence patterns of rare species (Gray *et al.* 1990). Dominance by opportunistic species appears only at a later stage. The faunal changes seem to be stable for several years after cessation of drill cuttings discharge in spite of significant reduction in hydrocarbon content of the sediments, suggesting that barite and other compounds associated with the discharges also have an impact (Olsgard and Gray 1995).

There is a general consensus that due to mixing, dispersion, and settling, water-based drilling discharges with their associated high content of suspended solids, metals, and certain organic toxicants are rapidly diluted. Harmful biological effects or exceedances of water quality standards occur only in the immediate vicinity of the discharge. Under most open-water conditions, dilution of drilling fluids far above 1000-fold can be attained within a few 100 m downcurrent of the discharge pipe and within 2-3 hours of discharge. Studies have shown 10 000-fold dilution in trace metal concentration only 100 m downcurrent from the discharge point (US DOI 1992).

The National Research Council concluded in its review that of the five principal ingredients in water-based drilling fluids (barite, bentonite, lignite, chrome-lignosulfonate, and sodium hydroxide), barite, bentonite, and lignite can be clas-

sified as practically non-toxic, i.e., having LC_{50} values greater than 10 000 ppm (NRC 1983). Further, it was concluded that the toxic effects of sodium hydroxide, as determined in laboratory experiments, could be attributed to elevation in pH, a situation that is not likely to occur when drilling fluids are discharged to the sea. Predictably, larval and juvenile forms of test animals were found to be more sensitive to the effects of drilling fluids than the adults.

Field experiments have been conducted in the Beaufort Sea to determine the effects of water-based drilling fluids and cuttings on benthic fauna (Crippen *et al.* 1980, Northern Technical Services 1981). Off Prudhoe Bay, Alaska, it was shown that drilling fluids and cuttings discharged below the ice at sites 6-8 m deep and on the ice in 7 m of water were quickly swept away from the study sites (Northern Technical Services 1981). This study also indicated that the abundance of amphipods and bivalves changed at the reference and test sites during the three-month study period. It should be noted that epibenthic animals, notably mysids and amphipods, undergo extensive onshore-offshore migrations intermittently throughout the open-water period, contributing to very high spatial variability (Griffiths and Dillinger 1981). At the disposal site on the ice, the disposed material accumulated on the bottom as the ice melted. In this bottom area the numbers of polychaetes and harpacticoid copepods were significantly lower than at the reference site. However, sediment grain size distribution was quite different between the two sites and some experimental trays had been disturbed during the course of the experiment. These factors may have affected the observed difference between test and reference sites.

The increasing use and discharge in recent years of synthetic drilling fluids, where esters, ethers, or refined oils from plants or animals have replaced the base oil, should be a matter of concern. Such drilling fluids must be expected to be applied in future drilling in the Arctic region too. It appears that these fluids have reasonably low acute toxicities, but some of them (mainly those based on esters) are rapidly degraded and may cause adverse bottom effects similar to organic enrichment, i.e., oxygen deficiency and reduced sediments (Schaanning *et al.* 1996). Studies indicate that these fluids are distributed almost as rapidly over the fields as water-based fluids (Bakke *et al.* 1995).

Produced water is the largest volume waste stream in the entire exploration and production process. Over the lifetime of a field, the volume of produced water may exceed the volume of hydrocarbons produced by up to ten times (Stephenson 1991). The basic components in the produced water may be grouped into five categories, petroleum hydrocarbons, heavy metals, radionuclides, treating chemicals, and salts.

Concentrations of a number of metals are often much higher in produced water than in the surrounding water into which it is discharged. Iron can be 560-2340 times higher in produced water, and cadmium 4-41 times higher (Menzie 1982). Produced water is generally very low in dissolved oxygen or anoxic, and somewhat warmer than the surrounding water.

Only very few studies relating to produced water impacts have been carried out in the Arctic, and there is considerable concern related to the possible effects of the large volumes that may have to be discharged at sites of petroleum production. Based on information for realistic concentrations and toxicities of the natural constituents and added chemicals in produced water discharges, and on the prevailing mixing processes at offshore discharge sites, one may conclude that it is not likely that produced water from responsibly operated platforms will have detectable negative effects offshore (Stephenson 1991). However, coastal embayments with restricted circulation may be vulnerable, and the potential for gradual accumulation of contaminants, such as metals, to harmful levels in these areas has not been fully assessed.

10.7.4. Biological effects of PAHs

The ubiquitous distribution of many PAH compounds in all environmental compartments provides a route of exposure to virtually all biota. The most plentiful data which occur for the aquatic environment clearly indicate bioaccumulation of various PAH compounds in aquatic plants, plankton, mollusks, crustaceans, echinoderms, annelids, and fish (Neff 1979, Varanasi 1989). Uptake of PAHs by aquatic organisms from the water column, from sediments, and from their diet varies widely among organisms and among individual PAH compounds and is influenced by a large number of both exogenous and endogenous factors. For example, decreasing temperature tends to increase the bioaccumulation of PAHs; the presence of dissolved organic matter can both enhance and reduce the bioaccumulation of specific PAH compounds. PAH body burdens tend to be higher in organisms which cannot metabolize them efficiently or at all, and in organisms rich in lipid. For example, fish tend to contain far lower body burdens of PAHs than do most of the invertebrates from the same environment due to their greater capability for metabolizing these compounds.

Bioaccumulation is generally positively correlated with physical/chemical properties of PAHs, such as molecular weight and octanol/water partition coefficients. The degree to which an organism will bioaccumulate PAHs, can therefore be roughly predicted knowing its physical/chemical properties. The bioconcentration factor, particularly in fish, tends to increase with increasing molecular weight. Biomagnification of PAHs via trophic pathways has not been observed in aquatic systems (McElroy *et al.* 1989).

Release of PAHs from contaminated organisms varies with each compound and organism. It can be passive, merely reflecting an equilibrium between the aqueous phase and the site of accumulation, or it can be active, involving the metabolic transformation of PAHs to polar water-soluble metabolites which are more readily excreted.

In general, there has been far less attention given to the study of the biological effects of PAHs than of compounds such as PCBs and chlorinated pesticides because PAHs tend to be less persistent than such halogenated compounds and are effectively metabolized by vertebrates. Most of the data for acute and sublethal toxicity have been acquired in relation to oil spill studies.

The biological effects of PAHs are related to size and structure of the molecules. In general, for aquatic organisms, the 2- and 3-ring aromatic PAHs (naphthalenes, fluorenes, phenanthrenes, and anthracenes) have significant acute toxicity. On the other hand, the 4- to 7-ring aromatics do not have any significant acute toxicity, though all of the proven PAH carcinogens are from this group. PAHs containing more than seven fused rings generally exhibit neither acute toxicity nor carcinogenic activity. Highly angular configurations are more carcinogenic than either linear or highly condensed ring systems (Neff 1979). Acute toxicity increases with increasing alkyl substitution of 2- and 3-ring PAHs.

PAHs can interact with cells in two ways to cause toxic responses. Firstly, they can bind reversibly to lipophilic sites in the cell, thereby interfering with several cellular processes. Secondly, the hydrophilic metabolic products of PAHs can interact with DNA to create covalently bound products known as adducts. Formation of these adducts is thought to

be one of the initial steps in the tumor induction caused by the carcinogenic PAH compounds. In mammals and fish, carcinogenic, teratogenic, and mutagenic effects of PAHs occur only after metabolic activation. Consequently, the capability of various animals to metabolize PAHs is an important factor in assessing possible long-term biological effects. Polar metabolites are more mobile and reactive in the aquatic environment because they are more soluble than their parent compounds. They can also be more acutely toxic than the parent compound.

Acute toxicity (LC_{50}) of naphthalene and alkyl-substituted naphthalenes to a variety of shrimp, copepods, polychaetes, and fish ranges from 8 to 5100 ppb. The embryonic and larval stages of fish were the most sensitive. LC_{50} values for similar biota for fluorene, phenanthrene, and fluoranthene are generally above 300 ppb.

Chronic exposure to low concentrations of PAHs in water, sediment, and food leads to a wide range of observable sublethal effects. Some PAHs at concentrations of 1-20 μg/L caused progressive inhibition of algal growth. Anderson and Gossett (1986) examined information on the biological effects of PAHs bound to sediment and concluded that threshold levels of PAH contamination might be between 5 μg/g (dry weight) and approximately 200-500 μg/g total organic carbon.

Representative sublethal effects on aquatic animals include: (1) reduction in feeding rate of copepods (Berdugo *et al.* 1977); (2) decrease in filtration rate in mollusks (Haranghy 1956); (3) gill hyperplasia and hemorrhages of the gill filaments in the mummihog (Di Michele and Taylor 1978); (4) reduction in feeding rate/growth of mollusks (Widdows *et al.* 1982); and (5) reduction in scope for growth of mollusks (Stickle *et al.* 1985).

High incidence of neoplasms has been associated with elevated concentrations of PAHs in sediment in a number of areas including Vancouver Harbour (British Columbia) (Goyette and Boyd 1989), and Puget Sound (Washington), eastern Lake Erie (USA), and the Black River (Ohio) (Baumann 1989). As noted above, the development of tumors in aquatic animals may be related to the ability of the organism to metabolize PAHs. Most invertebrates inherently have low metabolic capability. Among the invertebrates, the data suggest that higher invertebrates such as echinoderms, arthropods, and annelids can metabolize PAHs, whereas lower invertebrates such as protozoans can not. The ability of mollusks to metabolize PAHs is questionable; if it exists at all, it is at extremely low rates. Vertebrates, however, can effectively metabolize PAHs.

10.8. Conclusions and recommendations

1. Petroleum hydrocarbons have been detected in all biotic and abiotic compartments of the circumpolar Arctic environment. However, with the exception of the concentrations in a few areas exposed to catastrophic releases from oil pipeline ruptures or prolonged chronic releases, hydrocarbon levels in the Arctic associated with anthropogenic inputs have been found to be relatively low and not of ecological consequence.
2. Some severe local and regional problems have occurred recently in the Arctic associated with oil and gas exploration, development, and transportation. These include, for example, the pipeline ruptures in the Usinsk area in the Pechora River Basin, and the oil spill along the coasts of Alaska from the grounding of the *Exxon Valdez* oil tanker.
3. New major projects for development of oil and gas resources are presently underway or planned for the near future in a number of Arctic areas, including the lower Ob River Basin, the Yamal Peninsula, and Kara Sea in Russia, the Barents Sea in Russia and Norway, and the North Slope in Alaska. These developments will increase the potential for large-scale releases of oil to the Arctic with associated risks to Arctic ecosystems.
4. Major potential anthropogenic sources of oil pollution to the Arctic, in addition to oil and gas development, are releases from marine shipping; local concentrations around human settlements from burning of fossil fuels for energy, heat, and transportation; and long-range atmospheric transport from temperate and subarctic areas.
5. The most highly contaminated areas in the Arctic are certain rivers, estuaries, and harbors close to human settlements, and industrial or military areas; and in terrestrial/freshwater environments, areas where accidental and operational spills have occurred (e.g., in the Komi Republic/Pechora River Basin affected by the Usinsk pipeline rupture).
6. Inadequate monitoring data, and a lack of knowledge concerning the fate of petroleum hydrocarbons in river and estuarine environments, prevents the development of a reliable assessment of the impact of river input on marine contamination. However, preliminary results indicate that, due to the high self-purification capacity of rivers, they may not be a major pathway of long-range transport of contamination to the Arctic Ocean.
7. Polynuclear aromatic hydrocarbons (PAHs) are widespread in the Arctic environment and come from a variety of sources. These sources are either pyrogenic (from combustion), petrogenic, or biogenic. However, even where high natural PAH background concentrations have been found (e.g., the Canadian Beaufort Sea), the measured levels are, in almost all cases, below those thought to be required for observable effects on biota.
8. The effects of oil pollution may be more severe and persistent in the Arctic than for corresponding situations in other regions. This is due to the effects of the extreme Arctic climatic conditions on the distribution, composition, and physical state of petroleum hydrocarbons as well as on the rates of biological factors, such as growth rates, productive season lengths, generation turnover times, and ages at maturity, that determine the rate of ecosystem recovery from oil damage.
9. The environmental threats to the Arctic associated with oil and gas development, production, and transport are primarily local and/or regional and not circumpolar in scale. An important exception can occur, however, for certain species of migratory animals which congregate within relatively small areas if the period of congregation overlaps with intense disturbances (e.g., large oil spills). In such cases, devastating impacts could occur at the population level.
10. Primary concerns associated with the major new Arctic oil and gas developments involve the increased risks of accidents and the increased difficulties of taking remedial measures in such severe environments. These concerns are intensified because these environments are often of great significance to the indigenous peoples of the Arctic and others for the harvesting of wildlife, such as marine mammals, birds, and some fish species.
11. It is recommended that steps be taken to harmonize the monitoring for petroleum hydrocarbon levels and effects being conducted in regions of existing or developing oil and gas exploitation and transportation in the Arctic to

assure that consistent, internationally intercomparable data are collected. It is further recommended that a coordinated, circumpolar network of a relatively few long-term reference monitoring sites be established across the Arctic to provide background information on long-term trends in PAH and other petroleum hydrocarbons levels throughout this region.

12. Compliance with existing legal instruments appears to be the main issue rather than the development of new ones. It is therefore recommended that the eight Arctic countries cooperate to ensure that the integration of environmental protection measures into the design, construction, and operation of oil and gas exploration, production, and transport facilities and into harbor and other facilities along Arctic shipping routes takes place.
13. It is strongly recommended that the work of the International Northern Sea Route Programme (INSROP) be supported in order to assess the feasibility and potential international consequences of using the Northern Sea Route for the transport of oil via tankers.
14. It is recommended that environmental sensitivity mapping be improved and completed for all Arctic coastlines to aid in the selection of oil spill countermeasures.
15. Activities associated with oil and gas development can lead to severe environmental problems if not properly managed. These are caused by both acute (e.g., oil spills) and chronic discharges (e.g., produced water). In the Arctic situation, additional challenges arise due to extreme conditions of cold, ice cover, and periods of darkness. It is recommended that these extremes conditions of the Arctic be rigorously taken into account in the regulatory process for Arctic oil and gas exploration and development.
16. It is recommended that a research program on the pathways and fates of petroleum hydrocarbons in the terrestrial, riverine, and estuarine environments in the Arctic area be established to develop reliable estimates of fluxes to the Arctic Ocean.

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Annex

Contents