Contaminants in the arctic marine environment: priorities for protection

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We assess evidence for significant adverse effects on the arctic marine environment associated with global and regional releases of chemical and radioactive contaminants from human activities. The sources, mode of transport, and biological effects of persistent organic compounds, lead, cadmium, mercury, artificial radionuclides, and oil are reviewed. An outline of the physical and biogeochemical processes occurring within the Arctic Ocean is used as a basis for identifying contaminant transport, accumulation, and exposure pathways. Where significant anthropogenic impacts can be inferred or suspected, we evaluate opportunities for human intervention through the introduction of new source controls, or other restrictions, to reduce impacts. We conclude that additional controls are justified only in the cases of persistent artificial organic compounds and petroleum exploitation and transport. Artificial radionuclides pose threats that are largely nominal. Protection of the arctic environment and human health can be achieved through universal compliance with existing international standards. Threats posed by cadmium and lead from anthropogenic sources are relatively minor. Trends in the anthropogenic mobilization of mercury are not currently defined with sufficient confidence to assess the associated dangers posed to animal and human health in the Arctic. Nevertheless, the properties of mercury and its current environmental levels suggest that the Arctic is sensitive to increased anthropogenic mobilization of this metal. There is also evidence that mercury levels in the Arctic are increasing due to anthropogenic activities. Accordingly, priority should be given to determining temporal trends of mercury within the Arctic.

Key words: Arctic Ocean, assessment, bioaccumulation, contaminants, controls, effects, marine environment, metals, pathways, pollution, polycyclic aromatic hydrocarbons, organochlorines, radionuclides, regulation, risks, sources, toxicity, waste.

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Introduction

This paper reviews the current status of the effects of human activities on the arctic marine environment. Its purpose is to identify legitimate concerns about the condition of arctic coastal and ocean environments arising from contemporary and potential disturbances with a focus on contaminants i.e. persistent organics, metals, radionuclides, polycyclic aromatic hydrocarbons, and oil. The approach taken is, first, to outline the behaviour of contaminants and how they interact with the major physical, chemical, and biological processes in the Arctic Ocean. The contemporary status of contaminants is then reviewed to identify concerns and priorities. The priority issues are then examined to determine whether there are important deficiencies in existing control measures. The paper has been prepared using the scientific literature and the results, to date, of scientific investigations carried out within the Arctic Monitoring and Assessment Programme (AMAP). The intention of this review is not to suggest legislation or regulation; rather, it is to provide sufficient understanding to determine where additional legislation or regulatory controls might be warranted.

An explanation is first required of terms that are commonly misinterpreted but essential to the discrimination between concerns based on evidence from the natural sciences and those of perception, which are based on societal factors and human behaviour. First, we distinguish between the terms contamination and...
pollution. We use the term pollution as internationally defined by the United Nations Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection (GESAMP, 1986; Windom, 1991):

"Pollution means the introduction by man, directly or indirectly, of substances and energy into the marine environment (including estuaries) resulting in such deleterious effects as harm to living resources, hazards to human health, hindrance to marine activities including fishing, impairment of quality of use of sea water and reduction of amenities."

This definition is used, with some minor revision, in various international agreements including the United Nations Convention on the Law of the Sea (UNCLOS). Its importance is that pollution is equated to an adverse effect on the marine environment, its resources, and/or amenities. In contrast, the term “contamination” implies that the characteristics of the marine environment have been modified as a result of anthropogenic activities but without inference that these modifications are in any way deleterious. This latter term has been less widely defined but the definition of the Advisory Committee on Marine Pollution of the International Council for the Exploration of the Seas (ICES, 1989) is consistent with the use of the term in most international scientific fora, including GESAMP:

"Contamination is used to describe the situation which exists where either the concentration of a natural substance (e.g. a metal) is clearly above normal, or the concentration of a purely man-made substance (e.g. DDT) is readily detectable, but where no judgment is passed as to the existence of pollution (i.e. adverse effects)."

Our ability to recognize contamination continually improves with the evolution of technology. The sensitivity of electron-capture detectors, for example, has assured the detection of organochlorine compounds in virtually every environmental compartment. In contrast, pollution depends not on our ability to measure a contaminant but rather to identify associated harm. The distinction between these two terms is important because it permits discrimination between instances in which disturbance by human activities can be detected (contamination) and those in which adverse effects occur as a result of these disturbances (pollution). In turn, the difference between these terms implies that there exists a capacity for change in the marine environment without adverse effects occurring. This concept of environmental (or assimilative) capacity has been discussed at length by GESAMP (1986). The terms pollution, contamination, and environmental capacity are widely misunderstood and misinterpreted. This often leads to inferences of adversity resulting from anthropogenic activities based on perception rather than science.

In general, when scientists are asked to assess an activity and its potential effects on the environment they will adopt what is termed a conservative approach. This is not always the equivalent of making worst-case assumptions for the assessment but usually makes allowance for the greater part of the uncertainty associated with a projection or prognosis. Thus, prior assessment of the consequences of a particular development usually does, and must, make such allowances to ensure that the estimate of expected damage is a pessimistic one. Defining the scale of possible damage in this way permits the licensing authority to understand fully the consequences of a decision to authorize an activity. If the actual consequences are less than those projected, the associated damage is less than that deemed originally tolerable when the project was authorized.

In recent years, a new term has come into prominence - the so-called precautionary principle ("precautionary approach" or "principle of precautionary action"). Most commonly, it is couched in a context similar to the following text derived from the Ministerial Declaration of the Second International Conference on the North Sea (London Conference, 1987):

"Accepting that, in order to protect the North Sea from possible damaging effects of the most dangerous substances, a precautionary approach is necessary which may require action to control inputs of such substances even before a causal link is established by absolutely clear scientific evidence."

"[The Governments] therefore agree to: accept the principle of safeguarding the marine ecosystem of the North Sea by reducing pollution emissions of substances that are persistent, toxic and liable to bio-accumulate at source by the use of the best available technology and other appropriate measures. This applies especially when there is reason to assume that certain damage or harmful effects on the living resources of the sea are likely to be caused by such substances, even where there is no scientific evidence to prove a causal link between emissions and effects ("the principle of precautionary action")."

The philosophical difficulty here is that if precautionary action is to be invoked irrespective of scientific evidence, the principle becomes one of management or policy and not one of science. However, if it is looked on in this way, it in no way detracts from its effect or purpose but becomes an additional element of precaution to that of the conservatism commonly used in scientific evaluations. In general, it is accepted (Gray, 1994) that the precautionary principle is indeed one of management or policy and should not be justified on scientific grounds.

The contaminants of concern

The priority contaminants in the Arctic have been defined under the Arctic Monitoring and Assessment
Programme (AMAP) of the International Agreement on the Arctic Environmental Protection Strategy and may be grouped into four categories:
1. Persistent organochlorine compounds. These include pesticides, industrial products, byproducts of incineration, and byproducts of chemical processes.
2. Artificial radionuclides. These include radioactive elements released to the environment by nuclear weapons testing, nuclear fuel reprocessing, radioactive waste disposal, and accidents.
3. Metals. Here, we focus on the highest priority toxic metals, lead, cadmium, and mercury.
4. Polycyclic aromatic hydrocarbons (PAHs). These are components of petroleum and byproducts of combustion.

Factors controlling the behaviour of contaminants in the Arctic Ocean

Global biogeochemical considerations

Many substances released to the environment through human activities (e.g. metals, PAHs) enter natural cycles that have been operating globally throughout recent geological history. For the most part, human activities have enhanced the mobilization of chemical compounds rather than altered the basic constituents of the natural environment. Human activities may have perturbed natural cycles by, for example, mobilizing metals through mining or producing PAHs as a byproduct of the combustion of fossil fuels but, due to the underlying natural cycle, it is often difficult to distinguish the anthropogenic contribution and its effect on the ecosystem. These naturally-occurring substances may enter the marine environment through weathering, forest fires, or biological processes and be removed to marine sediments by sequestering onto particles. Transport takes place in the atmosphere (i.e. aeolian transport), in runoff from the continents and in ocean currents. Contaminants can be in dissolved, particulate, and/or gaseous phases depending on the physical and chemical properties of the chemical. Mercury, for example, is transported in all three phases, gaseous transport constituting a major pathway. Less volatile elements, such as uranium, are transported predominantly in particulate and dissolved forms. High naturally-occurring backgrounds of some substances (e.g. Hg or PAHs) may make a particular location all the more vulnerable to exceeding some “effects threshold” due to added burdens imposed by human activities (cf. Unkler and Macdonald, 1995).

Some contaminants such as organochlorine pesticides and artificial radionuclides are, for all practical purposes, produced solely by human activities. Once released to the environment these compounds enter biogeochemical cycles according to their chemical and physical properties. Finding these contaminants in the ecosystem, therefore, unequivocally implicates humans. These compounds, although produced in small quantities relative to naturally-occurring compounds, often have properties that make them particularly hazardous (e.g. toxicity, longevity).

Scientists refer to the distribution of a chemical among phases as partitioning. Partitioning, which depends primarily on the properties of the chemical, is the major physical/chemical control on the transport and fate of chemicals in the environment. For example, elements that bind strongly to particles in the ocean (e.g. particle-reactive elements like Pb, Pu) are susceptible to scavenging and sedimentation and therefore tend to be transported over comparatively short distances in the ocean (Kershaw et al., 1995). Elements that favour the dissolved phase (e.g. ⁹⁹Tc, I) tend to remain in ocean waters for long periods and can be transported over greater distances. These latter elements are often referred to as “conservative” because they remain in the water and are redistributed predominantly by mixing.

Partitioning also affects the way chemicals interact with aquatic biota. Some compounds exhibit a relationship between their concentrations in an organism and those in the surrounding environment, e.g. water, that can be treated as partitioning. This provides the basis for terms such as biological concentration factor (BCF), which reflects the equilibrium ratio of the concentration of a substance in an organism to that in the surrounding medium (i.e. accumulation from water). The terms bioconcentration, bioaccumulation, and biomagnification have been used widely, but with varied meanings, to describe processes whereby biota accumulate contaminants. Here, we use these terms as defined by Macek et al. (1979):

“Bioconcentration refers to that process whereby chemical substances enter aquatic organisms through the gills or epithelial tissue directly from the water; bioaccumulation is a broader term referring to a process which includes bioconcentration but also any uptake of chemical residues from dietary sources. Biomagnification refers to a process by which the tissue concentrations of bioaccumulated chemical residues increase as these materials pass up the food chain through two or more trophic levels.”

The arctic marine ecosystem is particularly vulnerable to fat-soluble contaminants because it is simple (low diversity) and has long food chains supporting top predators. The top predators tend to be long-lived species with low fecundity and a high proportion of fat in their bodies. Within the interior of the Arctic Ocean, organic carbon is conserved near the ocean’s surface by efficient recycling as witnessed by relatively high bacterial production (P. Wheeler, pers. comm.). This implies that the organochlorine compounds, which
tend to follow carbon, will also be recycled efficiently. Taken together, the Arctic Ocean provides an efficient system both to capture and biomagnify many fat-soluble compounds in top predators. A boraline peoples who depend heavily on food from the sea, particularly in the Canadian and Greenland Arctic, occupy the niche of top-predator in the marine ecosystem.

The distinguishing characteristic of the Arctic is its comparatively low temperature. Because low temperatures reduce volatilization, semi-volatile compounds can be preferentially transported to the Arctic by a “global distillation” process in much the same way as there exists a net transport of heat from equatorial to polar regions (Ottar, 1982; Mackay and Wania, 1995; Wania and Mackay, 1995). The primary medium of such transport for semi-volatile and insoluble substances is the atmosphere rather than the sea, whose temperature range is comparatively narrow (Barrie et al., 1992). For more soluble and less volatile substances, however, oceanic transport may be more important.

The effects of substances on biological organisms involve two factors: exposure and toxicity. Exposure reflects the degree to which an organism is exposed to the substance in the environment, either directly or through the medium of food. Toxicity of a chemical depends wholly on its physical and chemical properties. This allows a distinction between the terms hazard and risk, with the former circumscribing the properties, including toxicity, of a substance and the latter depending on the combination of hazard and exposure. The potential effects of pollutants on biological organisms range from the biochemical level to the population level. At the biochemical level, effects can range from the induction of enzymes to hormone dysfunction and possible genetic damage, to death. At the population level, effects can be observed as reproductive failure and alteration in the ecosystem structure. In general, cause and effect are harder to link as one moves from the biochemical level toward the population level. In obviously polluted areas (e.g. some temperate, industrialized harbours) it is possible to relate contaminant distribution to biochemical, reproductive, and community changes (cf. Hansen et al., 1985; Varanasi et al., 1987; Addison and Edwards, 1988; Warwick et al., 1990). For the arctic marine ecosystem, however, it will be much more difficult to show that contaminants are producing these kinds of effect and to date no such effect has been demonstrated of which we are aware. Risk assessments for the arctic marine environment are really ecological risk assessments in which the properties of a contaminant, including its chronic and acute toxicity, are evaluated together with food web dynamics and exposure. The focus of such assessments has been the risks to human health associated with the consumption of country foods rather than effects on organisms or the ecosystem itself (e.g. Dietz et al., 1990; Hansen et al., 1991; Kinloch et al., 1992; Wheatley and Paradis, 1995).

An interesting evaluation of the comparative risks of fatal cancer induction in human consumers of seafood containing radionuclides and a number of organic chemicals suspected of being carcinogens has been carried out by the International Atomic Energy Agency (IAEA, 1993). Its conclusions reinforce the need for greater attention to assessing the risks and detriment arising from potentially carcinogenic substances in seafood. It further notes that this “would enhance public appreciation of the potential of the risks associated with the dissemination of such chemicals into the environment and also provide an improved perspective with which to judge the risks associated with sea disposal of radioactive waste”. Not all contaminants end up in the Arctic. Reactive substances introduced into the atmosphere and the sea are continually being captured and removed from circulation by biota and freshwater and marine sedimentation – the latter can eventually remove them from the biosphere by sequestering them permanently into sea-floor sediments (i.e. for geological time scales). Neither do contaminants that reach the Arctic Ocean simply accumulate there indefinitely. In addition to sedimentation, other physical/chemical/biological processes such as radioactive decay, reduction, oxidation, metabolism, hydrolysis, and photolysis remove or transform contaminants. Lastly, contaminants may enter the Arctic Ocean and then simply leave again in exported ice, in outflowing water currents, and by atmospheric exchange (outgassing).

The task here is to define, first, where human activities have caused a major change in the incidence and distribution of chemicals in the arctic marine environment (contamination) and, second, to assess whether these are associated with adverse effects (i.e. pollution), giving rise to legitimate concerns about present or future damage. Besides the release of chemicals to the environment there are other human activities that cause marine environmental damage. These include physical destruction of habitat through coastal development, dam construction, dredging, mining, and the breaking of ice for marine transport. Although not specifically covered by the GEASMP definition of pollution, the adverse effects of these activities produce harm to the environment and, therefore, technically constitute “pollution”.

Vertical structure of the Arctic Ocean

The Arctic Ocean is vertically stratified into layers that owe their origin to water masses entering from the Pacific and the Atlantic (Fig. 1). These layers are subsequently modified within the Arctic particularly at the margins (i.e. on the shelves) by sea-ice formation and runoff. Together, stratification and processes on the shelves are the primary physical oceanographic controls.
of contaminant distributions. Contaminant distributions are further influenced by partitioning onto inorganic and biological particles for which the shelves are, once again, important locations.

The surface layer, or polar mixed layer (PML), includes approximately the top 50 m. This layer cycles seasonally; brine produced by ice formation in winter tends to destabilize the water column, allowing it to mix, while in summer, melting ice and freshwater runoff produces stratification with a fresher surface layer (5–10 m). Therefore, it is the PML that is in immediate (annual) communication with the atmosphere and ice, and it is here and within the ice that most of the biological primary production occurs. Beneath the PML is found a region of increasing salinity (the halocline); the important function of this region is that it acts as a barrier between the deeper ocean and the upper ocean. Within the Canada Basin (Fig. 1) the halocline is dominated by water from the Pacific Ocean which has been modified as it passes over the Chukchi Shelf; it is found between about 50 m and 250 m (Macdonald et al., 1989). The Atlantic layer, which pervades both arctic basins, is found at depths from about 200 m to greater than 1000 m. At the top of the Atlantic layer there is a complex region thought to be supported by insertion of water masses produced on the shelves (cf. Aagaard et al., 1981; Jones and Anderson, 1986). This region has been called the lower halocline to distinguish it from the Pacific layer, but it is probably a composite water mass produced by brine drainage from the shelves, the Barents Sea shelf being a key one (Schlosser et al., 1995).

The time scale of transport or replacement of water masses varies among the layers: the surface layer has a residence time of about 10 years in the Arctic (Ostlund, 1982). The haloclines also have residence times of this order but, as we proceed deeper into the Atlantic layer, the residence time increases to perhaps 30 years. The basin waters have the longest time scale, measured in centuries (Schlosser et al., 1994).

Horizontally, the Arctic Ocean can be subdivided into two main basins separated by the Lomonosov Ridge (Fig. 2). These basins are to some degree decoupled from one another allowing distinct differences in freshwater content (Aagaard and Carmack, 1989), water-mass structure (Carmack, 1990; M'Laughlin et al., 1995), and current structure (Aagaard, 1989; Rudels et al., 1992). By inference, differences in contaminant distribution between the two major basins are also to be expected.

**Modes of contaminant entry**

There are five modes of contaminant entry into the Arctic Ocean. These are: (1) inflowing ocean currents, (2) deposition from the atmosphere, (3) northward flowing rivers, (4) direct runoff from the land, and (5) direct disposal into the ocean.

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**Figure 1.** The vertical structure of the Arctic Ocean showing the polar mixed layer (approximately the top 50 m), the regions of density gradients between 50 m and 200 m (the haloclines and Pacific layer), the Atlantic Layer (approximately 200–1000 m), and the basin waters. It should be noted that our understanding of the structure of the Arctic Ocean as shown in this figure is continuing to undergo significant revision following recent high-quality tracer transects of the Arctic Ocean.
Ocean currents

The Arctic Ocean is often referred to as a “Mediterranean sea” because it is surrounded by land and communicates with oceans to the south only through restricted passages (Fig. 2). Large quantities of sea water flow into the Arctic from the Atlantic via the Fram Strait and the Barents Sea, and from the Pacific via the Bering Strait. These water masses outcrop in the oceans of origin, but they subduct within the Arctic Ocean or before they reach it to produce the vertical structure outlined above. The inflowing water potentially carries with it those contaminants received along its flow path before it enters the Arctic. Contaminants prone to this mode of entry tend to fall into two categories. First, those tending not to partition onto solid phases can be transported over long distances and times because they are not readily scavenged. Radioactivity, although somewhat particle reactive, discharged by European reprocessing plants, is the most clearly documented example of such a contaminant; within 5-6 years of its disposal into the Irish Sea this radionuclide enters the Atlantic layer of the Nansen Basin in the Arctic Ocean (Livingston, 1984; Smith et al., 1990; Dahlgaard, 1995; Kershaw and Baxter, 1995). Second, contaminants that deposit from the atmosphere into the waters south of the Arctic Ocean and northward flowing currents simply complete the journey; HCH is a good example (cf. Hinckley et al., 1991; Gaul, 1994).

Atmosphere

Atmospheric pathways into the Arctic are undoubtedly more complex than ocean currents. Winter and summer flow patterns are very different (Barrie et al., 1992). In winter, air flows mainly from the Eurasian continent into the Arctic and then out over North America (Fig. 3). It is this flow pattern that produces arctic haze in spring, and which has been linked to direct, rapid inputs of contaminants from heavily industrialized areas of eastern and northern Europe and Asia (cf. Barrie, 1986; Hentzenberg, 1989; Pacyna, 1991; Welch et al., 1991;
Klungsøyr et al., 1995). In summer, the Eurasian flow reverses and a weak north to south transport is found. At this time, airflow into the Arctic is frequently from the North Pacific and North Atlantic. Although the transport path from eastern and northern Europe is a well-documented and dominant source of contaminants, episodic transport can also occur from other regions including North America, Southern Europe/Northern Africa, and Asia (Pacyna, 1991; Raatz, 1991; Welch et al., 1991). Unlike transport by ocean currents, which can take years or even decades, atmospheric transport can be very fast (days). Hence, the atmosphere can deliver not only volatile and semi-volatile contaminants, but also aerosols containing particle-reactive contaminants such as Pb, highly chlorinated PCBs, and PAHs (cf. Bidleman et al., 1989; Hargrave et al., 1989; Patton et al., 1991; Barrie et al., 1992; Akeredolu et al., 1994). These contaminants end up in snow, on ice, in sea water, and in northern drainage basins (Gregor and Gupper, 1989; Welch et al., 1991).

Rivers and runoff

Total runoff into the Arctic Ocean contributes about 3300 km$^3$ fresh water each year. This inflow occurs mainly through four major rivers (Yenisey (603 km$^3$ a$^{-1}$), Ob (530 km$^3$ a$^{-1}$), Lena (520 km$^3$ a$^{-1}$), Mackenzie (340 km$^3$ a$^{-1}$); Aagaard and Carmack, 1989). The remainder of the freshwater inflow comes from intermediate-sized rivers (totalling 590 km$^3$ a$^{-1}$) and numerous other smaller rivers (totalling 720 km$^3$ a$^{-1}$). Contaminants discharged into northern drainage basins or into the rivers may subsequently be delivered to the inner shelves. During their transit, contaminants will have the opportunity to interact with inorganic and organic particles. Because deltas, estuaries, and large shelves tend to trap most of the sediments brought in by rivers (cf. GESAMP, 1987, 1992), particle-reactive contaminants entering in this manner will tend also to be initially trapped in these places. However, dissolved contaminants will be free to travel further, undergoing dilution but not removal to the sediments. Many rivers entering the Arctic Ocean form deltas on the shelf proper but two of the largest, the Yenisey and the Ob, discharge via long estuaries (Fig. 2; Pfirman et al., 1995b). It is to be expected that the delivery of particles and particle-reactive contaminants, their trapping in coastal sediments, and their vulnerability to subsequent resuspension and transport, will depend on exactly how a river enters the sea. This will be particularly important for sediment transport mechanisms involving resuspension, direct entrainment of turbid river water into ice, and the entrainment of bottom sediments into ice either by direct bottom contact or by suspension freezing which will be discussed below (cf. Reimnitz et al., 1994; Pfirman et al., 1995b).

Direct disposal

The former Soviet Union (FSU) has discharged and dumped radionuclides onto shelves in the Barents and Kara Seas during the past several decades (Yablokov et al., 1993). This disposal was augmented by underwater testing of nuclear devices near Novaya Zemlya. Direct disposal of PCBs into soils occurred at Distant Early Warning (DEW) line sites in the Canadian Arctic and
contamination from these sites has been identified in the nearby marine environment (Bright et al., 1995). Chronic and acute losses from oil exploration and production in offshore developments occurred on the Canadian Beaufort shelf (Thomas et al., 1986). Oil development on other shelves will similarly result in marine disposal. Of all the modes of entry, direct disposal is clearly the easiest to regulate.

Transport of contaminants within the Arctic Ocean

As illustrated schematically in Figure 2, the various water masses entering the Arctic Ocean tend to be steered by topography into boundary currents. Replacement of water in the interior of the basins occurs too but it is slower. The vertical layers in the ocean have motions distinct from one another. Much of the water entering from the Pacific escapes through the Canadian Archipelago while some of it exits at the Fram Strait. One branch of the Atlantic layer passes over the Barents Shelf, where it is modified, and then rejoins a less modified branch which enters the Arctic in the West Spitsbergen current (Fig. 2; Rudels et al., 1992; Klungsøyr et al., 1995). The Atlantic layer circulates counter-clockwise around the basins while being modified by shelf water and by mixing to escape eventually in the East Greenland current.

In the context of contaminants interacting with living systems the surface currents are most important involving water to depths of 30–50 m. The motion of ice which, due to extensive drifter programs (cf. Gloerson et al., 1992; Rigor, 1992) is well-understood (Fig. 4), is the best indicator of surface currents. On a
large scale, the Arctic Ocean surface circulation is organized into two distinct zones. The Beaufort Gyre in the Canada Basin is seen as a clockwise field of motion connecting the Chukchi, East Siberian, and Beaufort Seas. In the vicinity of the Lomonosov Ridge, the organization of surface circulation undergoes a transition with the Transpolar Drift (Fig. 4). Within the Transpolar Drift, surface water and ice from the Kara and Barents Seas tend to cross the central Arctic Ocean to exit at the Fram Strait. Thus, if contaminants are introduced to the Arctic Ocean by rivers discharging onto the Eurasian Shelf, a significant portion of them can be expected to leave by this route.

The special role of ice
Ice acts as a surface upon which atmospheric contaminants impinge (Melnikov, 1991) and within which contaminated sediments can be incorporated from the shelves (Pfirman et al., 1995). Therefore, ice is potentially an important vehicle for transporting contaminants over long distances with little alteration. Of equal importance, the ice provides a vital surface for arctic marine biota and therefore interaction with contaminants at the water-ice-air boundary is potentially a critical step of the pathway into the food web. The period of melting, at which time contaminants in ice are most likely to be released, coincides with the period of maximum primary productivity further enhancing the coupling of ice-borne contaminants with the food web. Ice cover also acts as a lid on the sea preventing or slowing down the exchange between the atmosphere and the ocean. In the event that atmospheric concentrations of volatile contaminants are reduced, as in the case of HCH (cf. Jantunen and Bidleman, 1995), ice cover may slow the natural process of outgassing in the interior ocean.

Ice tends to be produced on shelves and exported from them to the interior ocean. Turbulence and freezing in the water column during ice formation (suspension freezing) can effectively entrain bottom sediments by frazil and anchor ice formation (e.g. Pfirman et al., 1990; Riemnitz et al., 1994 and references therein). Ice-drift trajectories (Fig. 4) show that, on a large scale, ice has predictable sources and lifetimes within the Arctic. Contaminants picked up by ice in the Kara Sea, for example, will either enter the Barents Sea within one to two years through the straits between Novaya Zemlya, Frans Josef Land, and Svalbard or will exit the Arctic through the Fram Strait within a couple of years. In contrast, contaminants in ice of the Beaufort Gyre may have 5 or more years to accumulate as the ice goes through annual melt-freeze cycles. As the ice drifts within the Arctic Ocean, the melt-thaw cycle tends to ablate the ice surface in summer and add ice to the bottom of multi-year floes in winter. This process moves particles and particle-reactive contaminants to the ice surface within two to three years. Soluble (conservative) contaminants, on the other hand, will be rejected from the ice during the melt-thaw cycle along with brine. Therefore, it is to be expected that old, multi-year ice will contain small amounts of conservative contaminants and most of its particle-reactive contaminants at the ice surface. These particles will ultimately be released at locations where the ice completely melts (i.e. the Barents and Greenland Seas).

In the special case of oil and oilspills, ice will be the main agent of long-distance transport. Oil will end up on the surface of the ice because it can enter melt ponds or leads in summer by floating up through brine channels in the ice.

The special role of shelves
Arctic shelves process freshwater runoff and sediments entering from rivers and produce brine in regions of diverging ice cover in winter (Macdonald et al., 1995). In spring and summer, rivers discharge most of their sediment and freshwater loads, biological production occurs, and large regions of the shelf tend to clear of ice (Fig. 5a). In winter, freshwater inputs are vastly reduced and constrained by ice to the nearshore (Fig. 5b). On the middle and outer shelf, brine produced by the rapid formation of ice in flaw leads and polynyas destabilizes the water column and can produce dense water which flows along the shelf bottom to enter the ocean interior (Melling and Lewis, 1982). From the perspective of contaminant transport, the contrast between summer and winter may be summarized as follows. In summer, contaminants enter the shelf from rivers, from ice melt and from the atmosphere. The conservative contaminants will remain near the surface due to the strong seasonal stratification which prevents mixing and may be exported from the shelf. Bio-active or particle-reactive contaminants enter the food web or become scavenged by particles to settle toward the shelf bottom. In winter, the supply of contaminants through ice melt or rivers is much reduced as is the supply of particles to scavenge them. However, convection of the water at this time of year has the potential to move contaminants into deeper layers of the ocean interior (e.g. into and through the halocline) either through sinking of surface water or through entrainment of contaminated sediments in offshore flowing water.

Priority pollution concerns
Persistent organic contaminants
Recent excellent reviews of these contaminants in the Arctic have been given by Muir and Norstrom (1994),
Norstrom and Muir (1994), Muir et al. (1992), Lockhart et al. (1992) and Barrie et al. (1992). The persistent organic contaminants include:

1. The chlorinated hydrocarbon pesticide family, e.g. chlordane, toxaphene (a mixture of polychlorinated camphenes (PCC) or bornanes), hexachlorocyclohexanes (HCH), hexachlorobenzene (HCB), and DDT. Some of these pesticides are currently being used in circumpolar countries (e.g. endosulfan, methoxychlor, lindane, trifluralin, atrazine, dacthal, and others).

2. Chlorinated industrial products, byproducts, and combustion products; e.g. polychlorinated biphenyls (PCBs), polychlorinated dibenzo-p-dioxins and dibenzofurans (PCDD, PCDF), and other chlorinated contaminants as yet unidentified (e.g. recently

Figure 5. Shelf processes affecting the transport of dissolved and particulate matter in (a) summer and (b) winter.
found tris-(4-chlorophenyl)-methanol (TCP) which is now known to be a byproduct of DDT manufacture.

(3) Polynuclear aromatic hydrocarbons (PAHs).
This list, which contains the predominant organic contaminants found in the arctic marine environment, is not complete because new compounds are being manufactured each year. However, the environmental behavior of new compounds can now be reasonably well-predicted by multi-media models using only the physical and chemical properties of the compounds (e.g. Wania and Mackay, 1995). Therefore, in principle, there is a basis to determine whether or not a new synthetic compound poses a threat to the global environment.

Organochlorine compounds

Starting in the 1940s, organochlorine compounds were produced and released predominantly in the temperate land areas of the northern hemisphere. Because they have given rise to evident adverse effects in the environment close to sources, many of these compounds (chlordane, DDT, PCB) have already been banned, or restricted, in North America and western Europe (Pacyna, 1995) and measures are presently being undertaken in some countries to reduce emissions by secondary sources such as incinerators (Anon., 1995). A significant proportion of the releases from past production, however, continues to cycle in the global environment and contributes a large share of the organochlorine compounds presently entering the Arctic. As pointed out by Goldberg (1991), large quantities of organochlorine pesticides continue to be used in less-developed countries, particularly in the southern hemisphere. Unfortunately, there is little quantitative information on the amounts being used, nor are good data available for the former Soviet Union or China, both of whom have been major contributors to the global budget (Norstrom and Muir, 1995). It is essential that we acquire and document consistent and reliable statistics on the production and use of organochlorine compounds if we are to develop regulations aimed at protecting the global environment.

The properties that make many organochlorine compounds particularly hazardous are their toxicity, volatility, persistence in the environment, and affinity for fat. Organochlorines exhibit toxicity in a variety of ways. Some are relatively innocuous (e.g. αHCH), some are carcinogenic (e.g. toxaphene to animals), some disrupt hormones (e.g. DDE), some are highly toxic to mammals, fish and birds (e.g. 2,3,7,8-TCDD and planar PCBs) while most of the organochlorine pesticides are neurotoxins. Their mode of entry into the Arctic Ocean has occurred predominantly via the atmosphere although, due to their relatively high partitioning from the gas phase into sea water, HCH and possibly some of the components of toxaphene, may have been delivered in significant amounts through ocean currents. PCBs, DDT, and other contaminants have reportedly also entered the Russian shelves via contaminated rivers, particularly the Yenisey, Ob, Lena, and Kolyma rivers (Pfirman et al., 1993). Due to the strong affinity of highly chlorinated compounds for particles, a large proportion of these contaminants will become trapped in shelf and slope sediments. Some of these sediment-bound contaminants may later enter ice through suspension freezing or ice grounding.

The organochlorines are readily incorporated into phytoplankton which are in direct contact with the water and have high surface area per unit volume. Entry of organochlorines into the food chain is most efficient when the lipid/water solubility ratio exceeds $10^4$ as it does for PCBs, chlordane, and HCB. The relatively high concentrations in marine phytoplankton (compared to terrestrial primary producers) result in higher values in the marine food web (Fig. 6). For example, bioconcentration factors (BCF) from water to zooplankton range from about $7 \times 10^4$ to toxaphene to $2 \times 10^6$ for PCBs (Muir et al., 1992); bioaccumulation in cod ranges from about $2 \times 10^5$ to toxaphene to $2 \times 10^8$ for PCBs and, finally, the biomagnification for top predators relative to water is about $10^7$ for toxaphene and $10^8$ for PCBs. However, this is not the entire story because substantial changes in the composition of organochlorine compounds within organisms are mediated by excretion and/or metabolism which varies among species.

Our ability to assess temporal trends of organochlorine contaminants in marine biota from the Arctic is limited by the sparseness of data; at present, usually only two or three sampling periods are available from which to infer trends. Efforts to detect such trends in marine animals are further hampered by other sources of variance including the evolution of analytical methods, seasonal and spatial variations, difference among species, and differences related to both the sex and age of the animals. Organochlorine concentrations in seabird eggs from Prince Leopold Island (Noble, 1990) and in seabirds from the Barents Sea (Savinova et al., 1995) suggest that burdens have declined since the mid 1970s. Two of the more toxicologically significant organochlorine compound classes, PCBs and DDT, were measured in ringed seals between 1972 and 1981 by Addison et al. (1986). They found that concentrations of both compound classes declined during that time interval and that PCBs appeared to be decreasing faster. This finding was confirmed by Muir et al. (1988) and has been extended with samples collected in 1989 (Addison, pers. comm). An attractive explanation for more rapid declines in PCBs is their complete production phase-out by the end of the 1970s whereas DDT continues to be used in some developing countries (cf. Pacyna, 1995; Voldner and Li, 1995). However, a review by Kurtz...
of the organochlorine data up to 1993 for both ringed seals and polar bears (Table 1) clearly illustrates the danger of generalizing contaminant trends using the present data base for marine mammals. Well-designed programs are urgently required to reveal how organochlorine contaminant burdens in arctic predators are responding to the various changes in production and use of the organochlorines.

The four major organochlorine contaminants found in marine mammals are DDTs, PCBs, chlordane, and PCCs (Norstrom and Muir, 1994). A substantial, but geographically incomplete, data set has been accumulated for marine mammals including bears, seals, and whales (Norstrom and Muir, 1994 and references therein). These data show variations in concentration among species and from location to location within the Arctic. Contamination of the Canadian arctic marine ecosystem by DDT and PCBs is generally 10–50 times less than the marine environment at temperate latitudes of the northern hemisphere and well below the threshold known to produce, for example, hyperadrenocortical disease syndrome or reproductive failure in seals (Norstrom and Muir, 1994). However, caution is warranted as elevated concentrations of organochlorines have recently been found in bears and seals from the vicinity of Svalbard (Norheim et al., 1992; Skaare et al., 1994; Norstrom et al., 1995). Contaminant burdens may also be very high locally near the Russian coast (Melnikov and Vlasov, 1992) but data are lacking to make a valid assessment of this region.

The top predators (e.g. small-toothed whales, polar bears, birds, humans) are at the greatest risk from organochlorines. However, precise cause and effect relationships between the organochlorine compounds and biological disorders remain elusive due to many factors including the complexity of the compounds themselves.
(e.g. PCBs alone include 200 different congeners each with a specific toxicity). Nevertheless, Norstrom and Muir (1994) point out that where comparisons can be made, the levels of PCBs and DDTs in arctic marine mammals are below the effects levels (bear populations at Svalbard may be an exception - cf. Norheim et al., 1992). Polar bears and seals, both of which undergo large seasonal changes in fat content, may be more at risk than mean organochlorine burdens would suggest due to the release of fat-associated contaminants during periods of fasting. Cod, which undergo large changes in fat content during spawning, may similarly be at greater risk during that time.

High concentrations of organochlorine contaminants in arctic marine mammals pose a risk to humans who consume large quantities of country foods (Dewailly et al., 1989). In a study at Broughton Island, NWT, Klinchok et al. (1992) found that 12% of women and 22% of men exceeded the medium-term tolerable daily intake of 1 µg kg⁻¹ day⁻¹ PCBs established by the Canadian Government. Despite this disturbing finding, the authors concluded that altering consumption patterns to avoid these contaminants was not warranted: nutritional and cultural benefits of these country foods outweighed the risks of contamination.

Polynuclear aromatic hydrocarbons
Unlike the organochlorines, PAHs are widespread products of natural cycles (forest fires, plant decomposition, diagenesis, petrogenesis) and therefore anthropogenic contamination is more difficult to identify. Contaminant sources for the Arctic include petroleum exploration and production (Thomas et al., 1986) and long-range atmospheric transport (Patton et al., 1991; Lockhart et al., 1992; Kawamina and Suzuki, 1994; Mascal and Hoyau, 1994; Yinuker and Macdonald, 1995; Klungsøyr et al., 1995). PAHs, especially the more soluble low molecular weight compounds, can bioconcentrate in fish from sediments (cf. Hellou et al., 1995 and references therein). 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. The toxicity of PAHs derives from their potential, during metabolism, to form carcinogenic and mutagenic diols and epoxides that react with DNA (Zedeck, 1980). The potential of contaminant PAHs to produce toxic effects must be viewed in the context of a variable, naturally occurring background (Yinuker and Macdonald, 1995).

Even where high natural PAH background and concentrations have been found (the Canadian Beaufort Sea), Yinuker and Macdonald (1995) concluded that the measured levels were below the threshold required for observable effects on biota. Because PAHs are not prone to biomagnification, there is less concern about dispersed, long-range inputs. PAHs are more likely to have an effect in regions where direct inputs occur (e.g. creosote applications, oil spills) especially in circumstances where there is confined, long-term contact between the organism and PAHs (e.g. flatfish associated with contaminated sediments; cf. Malins et al., 1984; Hellou et al., 1995).

Radionuclides and radioactivity
This section assesses the sources and effects of radionuclides in the Arctic leading to an evaluation of the need for additional regulatory action. Much of the information presented is based on the review of Aarkrog (1993).

Sources of radionuclides
The predominant radionuclides derived from anthropogenic sources measured in the arctic marine environment comprise ⁹⁹Tc, ¹²⁹I, ¹³⁷Cs, ⁹⁰Sr, ²³⁸Pu, ²³⁹⁺²⁴⁰Pu, ²⁴¹Pu, and ²³⁸U. To these can be added a range of short-lived fission and activation products derived from the Chernobyl accident which are now mostly below normal limits of detection. The important sources of radionuclides are: atmospheric fallout (both local and global) from weapons tests and satellite power unit burnup; nuclear fuel reprocessing installations in Europe; discharges from the Ob, Yenisey, and Lena rivers; and releases from dumped Soviet and Russian radioactive wastes and accidents. The only sources giving rise to detectable quantities of radionuclides in the arctic marine environment generally are global nuclear weapons fallout and nuclear fuel reprocessing operations in Europe, particularly prior releases from Sellafield. The relative importance of these sources varies among radionuclides. For example, the vast majority of the ⁹⁹Tc in the Arctic seas originates from European reprocessing plants (Dahlgaard, 1995a) while the correlation of ⁹⁰Sr with salinity suggests that the dominant contemporary source for this nuclide is land runoff containing global atmospheric fallout (Dahlgaard, 1995b). In contrast, ¹³⁷Cs derives from atmospheric fallout, discharges from European reprocessing plants, re-mobilization from Irish Sea sediments, and releases from the Chernobyl accident, partly via the outflow from the Baltic Sea (Dahlgaard, 1995b). With limited exceptions, such as the ²³⁸Pu from the radionuclide thermolectric power unit in the US Navy navigation satellite Transit 5BN that burned up over the Indian Ocean in 1964, introductions of artificial radionuclides from other sources are regionally undetectable against the background of those present from the three principal source categories. On local scales, however, such as in proximity to rivers delivering radionuclides from human activities on land (e.g. the Ob and Yenisey rivers), in locations at, or near to, nuclear weapons explosion sites (e.g. Chernaya Bay, Novaya Zemlya) or in the immediate vicinity of dumped or lost solid radioactive waste.
sources, radionuclides from other sources can be both expected and occasionally detected. The following discussion of sources provides the basis for assessing whether they are amenable to control – an essential prerequisite to determining the utility of any additional regulatory controls.

Fallout from weapons tests
Early nuclear weapons tests between 1945 and mid 1952 involved only fission weapons and there was no significant large-scale transport of radioactive contamination to the Arctic. The fallout from the major tests at Bikini, Nevada, and Semipalatinsk was deposited at the test sites and within the corresponding latitude bands. Thermoneutral weapons testing began in October 1952, when the US tested its first thermonuclear device at Eniwetok Island, and this began the period of significant global radionuclide fallout (Fig. 7). The total direct fallout into the arctic marine environment up to 1993 corresponded to 9 PBq of $^{90}$Sr and 15 PBq of $^{137}$Cs (Arkrog, 1993) (PBq = peta Becquerel = $10^{15}$ disintegrations per second). In addition, some 1.5 PBq of fallout $^{90}$Sr and 0.5 PBq of fallout $^{137}$Cs (decay corrected to 1993) has entered the Arctic Ocean via river transport (Arkrog, 1993).

Between 1955 and 1962, 87 atmospheric test explosions took place at Novaya Zemlya. These explosions released between 100 and 370 PBq of $^{90}$Sr and 155 and 560 PBq of $^{137}$Cs as fallout, predominantly global but some of it regional. Arkrog (1993) gives an upper estimate for the latter as 20 PBq $^{90}$Sr and 30 PBq $^{137}$Cs.

Atmospheric nuclear weapons test explosions have almost totally ceased as a result of the Partial Nuclear Test Ban Treaty covering testing in the atmosphere, space, and water that entered into force on 10 October 1963. This is reflected in declining atmospheric fallout since that time (Fig. 7).

Nuclear fuel reprocessing in Western Europe
Releases to the ocean from nuclear fuel reprocessing plants at Dounreay, Cap de la Hague, and Windscale/Sellafield have all contributed to the introduction of radionuclides by water transport into the Arctic (Kershaw and Baxter, 1995). Sellafield is historically and currently the most important source among these. Maximum releases from Sellafield occurred in about 1975 when approximately 5 PBq of $^{137}$Cs were discharged to the Irish Sea (Gray et al., 1995 and see Figure 7b for the $^{137}$Cs discharge record). Releases have continually been reduced since that time.
and are now of the order of 0.1 PBq per year (excluding $^{3}$H). Marine transport of $^{137}$Cs to the Arctic Ocean takes about 5–7 years (Aarkrog et al., 1987) but a substantial proportion of the radionuclides previously discharging from Sellafield, currently about 10–15 PBq, resides in the Arctic Ocean (Aarkrog, 1993; Kershaw and Baxter, 1995). These radionuclides are found in the Atlantic layer (Fig. 1) and therefore are not immediately accessible to the biological community in surface waters.

Radioactive wastes from the Mayak reprocessing Association, Chelyabinsk, and at plants in Tomsk and Krasnoyarsk. During the period 1949–1956, liquid radioactive wastes from the Mayak reprocessing plant were released to the Techa-Iset-Tobol rivers, which are within the Ob drainage basin, including about 100 PBq of $^{90}$Sr and 4.6 PBq $^{137}$Cs. Additional releases were made from the Tomsk plant to a tributary of the Ob and from the Krasnoyarsk plant to the Yenisey River. The outflows from the Ob, Yenisey, and Lena rivers contain artificial radionuclides from fallout and possibly activities within the respective drainage basins. Considerable research is being extended to quantify and characterize the radionuclides in these outflows under the United States Atomic Nuclear Waste Assessment Program (ANWAP) that is funded by the Office of Naval Research. Presently, there appears to be little evidence of radionuclide sources other than fallout to the lower reaches of the Ob and Yenisey rivers and to the Kara Sea (Baskaran et al., 1995) or from transport from the FSU dump sites to distant regions in the Arctic Ocean (Baskaran and Naidu, 1995).

While the industrial sources of contemporary discharges are amenable to control, radionuclides previously released into the environment have created secondary sources that are more appropriately the subject of intervention under the terms of the System of Radiological Protection (ICRP, 1991).

Accidents (Chernobyl, submarine losses and aircraft accidents)

Accidents inevitably occur. Chernobyl, perhaps the best known example, was preceded by other accidents in the nuclear industry (Windscale, Three Mile Island). At Chernobyl, 100 PBq of $^{137}$Cs was released to the atmosphere, most of which was deposited throughout the northern hemisphere. Of this amount, about 1 PBq was deposited between 70–90$^\circ$N (i.e. in the Arctic). This has been augmented by the transport of approximately a further 5 PBq of $^{137}$Cs from the north-east Atlantic into the Arctic through ocean advection (Aarkrog, 1993).

In 1968, a B52 aircraft carrying four nuclear weapons crashed on the sea-ice 11 km west of Thule, Greenland. The conventional explosives in the weapons detonated and distributed the warhead plutonium (total ~6 kg) onto the wreckage and into the surrounding environment (the atmosphere, on to sea ice and into and underneath the sea ice at the point of impact). Between 2.8 and 4.2 kg of plutonium was estimated to be distributed in and around the crash site in addition to unquantified radionuclides in contaminated wreckage (Aarkrog, 1995). The wreckage and ice-deposited plutonium were recovered as far as possible, leaving an estimated 0.2–0.6 kg of plutonium in the environment. Some of this plutonium was subsequently deposited into the underlying marine sediments. The redistribution of sedimentary plutonium at the site has been the subject of periodic investigations and, for an assessment of the transport of this residual plutonium and dose consequences, see Aarkrog (1995) and Smith et al. (1994).

The major submarine loss in the Arctic Ocean was that of the Komsomolets in the Norwegian Sea. The hull of this submarine contains a nuclear reactor and two torpedoes with nuclear warheads. While there is some agreement that the total plutonium activity of the warheads is ~15 TBq this has been differently equated to 3.5 kg (Livovsky et al., 1995) and 6 kg (NATO, 1995) in units of mass. Because of its extreme affinity for particulate material (that is, a high sediment–water partition coefficient) plutonium released from torpedo warheads is likely to be retained in sediments in the vicinity of the wreck. The warheads probably also contain a larger quantity of $^{235}$U (NATO, 1995). The current status of this wreck and the consequences of corrosion and release of radionuclides have been studied by a NATO/CCMS Pilot Study (NATO, 1995). The low probability of catastrophic failure and limited extent of likely contamination of the environment make this a source of limited environmental concern. The rather corroded state of the submarine hull makes its recovery impractical; however, some consideration is being given to encapsulating the wreck in an absorptive blanket.

Dumped radioactive sources

The former Soviet Union and the Russian Federation carried out dumping at sea of liquid and solid radioactive wastes both prior to and, after the Soviet Union became a contracting party to the London Convention 1972 (Yablokov, 1993). Some of these dumping activities took place in contravention of the provisions of the London Convention (IMO, 1972). The most important dumped wastes are those of submarines and marine nuclear reactor assemblies containing spent fuel. A assessment of the rates of release of radionuclides and the resultant consequences in terms of risks to the environment and human health are being carried out as part of both the US Arctic Nuclear Waste Assessment Program and the IAEA’s International Arctic Seas Project (IASAP), to be completed in early 1997.
Dose consequences and associated risks

The main criterion for assessing the severity of risks posed by radionuclides is exposure, or dose, and the associated risk to health. Thus, it is not the presence, levels, or sources of radionuclides in the environment that are of immediate importance. Rather, it is the exposure of animals and humans to those radionuclides that is paramount. The System of Radiological Protection (ICRP, 1991) embodies an assumption of a linear no-threshold dose-response relationship at low doses such as those generally received from environmental sources. The ability to convert radiation dose to a risk of health defects allows risks associated with low doses to be estimated and compared with risks posed by other activities and practices (cf. IAEA, 1993).

An example of the contributions to dose made by radionuclides to arctic inhabitants is presented in Figure 8 (after Beak, 1995). The estimated contributions made by external exposures and by specific natural and artificial radionuclides through inhalation (mainly $^{222}$Rn) and ingestion pathways are given for average adults and children based on pooled data for five northern communities (Broughton Island, Baker Lake, Rae-Edzo, Makkovik, and Old Crow). The estimated contributions to total individual dose for members of more exposed communities consuming large amounts of country foods, particularly caribou meat, are depicted under the heading “critical group”. With the exception of $^{137}$Cs, which is produced artificially, all of the radionuclides specifically identified in this figure are natural.

As can be seen in Table 2 and Figure 8, the largest doses and risks are associated with natural radionuclides, predominantly $^{210}$Po, $^{210}$Pb, $^{228}$Th, and $^{232}$Th. Among the radionuclides derived from anthropogenic sources, $^{137}$Cs is the major contributor to dose but represents only 2–3% of the total and derives predominantly from land-mammal, and not seafood, consumption.

To provide a context against which to consider the risks presented in Table 2, the annual probabilities of death from a variety of causes to individuals in the United Kingdom and United States populations are provided in Table 3 (after IAEA, 1993).

None of the doses and risks associated with artificial radionuclides at the currently observed levels in the Arctic Ocean are large enough to warrant immediate concern. Thus, the major concerns relate to practices and sources that threaten to give rise to exposures that would not adequately protect potentially exposed populations. Among these are previously dumped high-level wastes, which are the subject of the IAEA’s International Arctic Seas Assessment Project (IASAP). This study will include an examination of contemporary and future risks to determine if they are high enough to warrant intervention (i.e. remediation). The only manner of minimizing both the probabilities and effects of accidents is to employ the same...
Table 2. Doses and risks to adults in the Broughton Island population.

<table>
<thead>
<tr>
<th>Radionuclide</th>
<th>Mixed diet (Total dose) (Sv a⁻¹)</th>
<th>Incremental risk serious health defect (a⁻¹)</th>
<th>Country food diet dose (Sv a⁻¹)</th>
<th>Incremental risk serious health defect (a⁻¹)</th>
<th>Percent of total dose (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>¹³⁴Cs⁧</td>
<td>6.14 × 10⁻⁷</td>
<td>3.1 × 10⁻⁸</td>
<td>8.55 × 10⁻⁷</td>
<td>4.3 × 10⁻⁸</td>
<td>0.02-0.03</td>
</tr>
<tr>
<td>¹³⁷Cs⁧</td>
<td>5.91 × 10⁻⁵</td>
<td>3.0 × 10⁻⁶</td>
<td>8.28 × 10⁻⁵</td>
<td>4.1 × 10⁻⁶</td>
<td>2.35-2.57</td>
</tr>
<tr>
<td>⁹⁶Sr⁧</td>
<td>2.25 × 10⁻⁷</td>
<td>1.1 × 10⁻⁶</td>
<td>2.25 × 10⁻⁷</td>
<td>1.1 × 10⁻⁸</td>
<td>0.01</td>
</tr>
<tr>
<td>²²⁶Ra*</td>
<td>1.54 × 10⁻⁵</td>
<td>7.7 × 10⁻⁷</td>
<td>2.10 × 10⁻⁵</td>
<td>1.1 × 10⁻⁶</td>
<td>0.61-0.65</td>
</tr>
<tr>
<td>²¹⁰Pb*</td>
<td>6.78 × 10⁻⁵</td>
<td>3.4 × 10⁻⁶</td>
<td>9.58 × 10⁻⁵</td>
<td>4.8 × 10⁻⁶</td>
<td>2.69-2.97</td>
</tr>
<tr>
<td>²¹²Pb⁦</td>
<td>1.45 × 10⁻³</td>
<td>7.2 × 10⁻⁵</td>
<td>1.99 × 10⁻³</td>
<td>1.0 × 10⁻⁴</td>
<td>57.6-61.6</td>
</tr>
<tr>
<td>⁷Be†</td>
<td>2.00 × 10⁻¹⁰</td>
<td>1.0 × 10⁻¹¹</td>
<td>2.00 × 10⁻⁹</td>
<td>1.0 × 10⁻¹⁰</td>
<td>0</td>
</tr>
<tr>
<td>²³⁰Thⱪ</td>
<td>6.66 × 10⁻⁵</td>
<td>3.3 × 10⁻⁶</td>
<td>8.60 × 10⁻⁵</td>
<td>4.3 × 10⁻⁶</td>
<td>2.64-2.66</td>
</tr>
<tr>
<td>²³⁷ThІ</td>
<td>2.36 × 10⁻⁶</td>
<td>1.2 × 10⁻⁷</td>
<td>3.01 × 10⁻⁸</td>
<td>1.5 × 10⁻⁷</td>
<td>0.09</td>
</tr>
<tr>
<td>²³⁴U§</td>
<td>1.11 × 10⁻⁴</td>
<td>5.6 × 10⁻⁶</td>
<td>1.54 × 10⁻⁴</td>
<td>7.7 × 10⁻⁶</td>
<td>4.39-4.76</td>
</tr>
<tr>
<td>²³⁵U†</td>
<td>1.27 × 10⁻⁶</td>
<td>6.4 × 10⁻⁸</td>
<td>1.64 × 10⁻⁶</td>
<td>8.2 × 10⁻⁸</td>
<td>0.05</td>
</tr>
<tr>
<td>²³⁶U⁦</td>
<td>2.95 × 10⁻⁶</td>
<td>1.5 × 10⁻⁷</td>
<td>4.10 × 10⁻⁶</td>
<td>2.1 × 10⁻⁷</td>
<td>0.12-0.13</td>
</tr>
<tr>
<td>²³⁸Uundance</td>
<td>1.02 × 10⁻⁶</td>
<td>5.1 × 10⁻⁸</td>
<td>1.30 × 10⁻⁶</td>
<td>6.5 × 10⁻⁸</td>
<td>0.04</td>
</tr>
<tr>
<td>Total</td>
<td>2.36 × 10⁻³</td>
<td>1.2 × 10⁻⁴</td>
<td>3.0 × 10⁻³</td>
<td>1.5 × 10⁻⁴</td>
<td></td>
</tr>
</tbody>
</table>

* Dose is primarily associated with land mammal consumption.
† Inhalation pathway only.
‡ Calculation for water consumption only although this may not be the predominant pathway of exposure.
§ Dose is primarily associated with aquatic foodstuff consumption.

Metals (cadmium, lead and mercury)
Cadmium (Cd), lead (Pb), and mercury (Hg) are toxic at high enough concentrations and none has any known nutritive benefit. Metals released by anthropogenic activities enter natural geochemical cycles, making it difficult to assess incremental effects due to human mobilizations. Each of these metals has unique properties that give rise to differing biogeochemical behaviour. For example, Pb tends to become strongly attached to particles and hence to be sedimented with little tendency to recycle; Cd favours the dissolved phase and enters marine biological cycles, thereby correlating strongly with the nutrient phosphate (Boyle et al., 1976). When sedimented, Cd tends to become remobilized (Gobeil et al., 1987). Hg becomes attached to organically rich particles but exhibits several forms (elemental, ionic, methylated) each of which has different toxicity and mobility (cf. Weiner and Spry, 1995; Linqvist and Rodhe, 1985).

Undoubtedly, the Arctic has been contaminated by metals as reflected in Pb and Cd distributions in Greenland ice cores (e.g. Rosman et al., 1994; Boutron, 1995) and Hg profiles in dated sediment cores (Lockhart et al., 1995; Lucotte et al., 1995; Slemr and Langer, 1992). The predominant inputs of anthropogenic metals to the Arctic Ocean are from long-range transport via the atmosphere (Rahn, 1981; Jickells et al., 1990; Barrie et al., 1992; Akeredolu et al., 1994; Pacyna, 1995), locally from heavily industrialized regions in the Russian Arctic (Matishov, 1993; Klungsøy et al., 1995; Pacyna,
1995), or from mine sites within the Arctic (Nanisivik in Strathcona Sound, Muir et al., 1992; Blackangel in Greenland, Asmund, 1992). Rivers may also be an important route of entry for contaminant metals (Melnikov, 1991). However, recent data on the three major Russian River estuaries (Yenisei, Ob, Lena) suggest that they are pristine with respect to a suite of metals including Pb, Cd (Dai and M artin, 1995; Martin et al., 1993), and Hg (Coquery et al., 1995). The largest North American river, the Mackenzie, is reportedly also pristine (Thomas et al., 1986). Smaller rivers draining heavily industrialized areas of northern Europe and Asia may provide locally important heavy metal inputs to the Russian shelves but there are no reliable data that would permit an evaluation.

Sufficient data are available to make preliminary estimates of the atmospheric fluxes of anthropogenically-derived Pb, Cd, and Hg to the Arctic Ocean. The total annual atmospheric fluxes of contaminant Pb and Cd to the Arctic from Eurasia have been estimated (for 1979 to 1980) at 2400 tonnes a$^{-1}$ and 47 tonnes a$^{-1}$, respectively (Akeredolu et al., 1994). Based on stable-isotope data, atmospheric contaminant Pb is derived predominantly from eastern Europe and Russia (Sturges et al., 1993). The evidence suggests that the fluxes of contaminant Pb are significant when compared to other sources (cf. Rahn, 1981) whereas those of contaminant Cd are not (Rahn, 1981; Yeats and Westerlund, 1993). Pacyna and K eeler (1995) estimate that 60-80 tonnes a$^{-1}$ of contaminant Hg is presently deposited from the atmosphere in the Arctic. Atmospheric deposition on ice close to source regions (e.g. the Kola Peninsula) has the greatest potential to produce metal concentrations that affect, for example, ice biota. This would be most likely to occur over the inner Russian shelves (cf. Melnikov, 1991). For Hg, the predominant anthropogenic sources have been the burning of coal, mining and extraction of Hg, smelting of other metals, chlor-alkali plants, and municipal waste incineration. The total anthropogenic emissions of Hg to the atmosphere, estimated for 1983 at about $3.6 \times 10^5$ tonnes a$^{-1}$ (Nriagu and Pacyna, 1988), exceed those from natural sources of $2.6 \times 10^3$ tonnes a$^{-1}$ (for estimated Hg releases see also Linqvist and Rodhe, 1985; Fitzgerald, 1986; Nriagu and Pacyna, 1988; Linqvist et al., 1991; Fitzgerald, 1995; Pacyna and K eeler, 1995). Evidence from sediment cores collected in remote northern lakes in Canada and in Hudson Bay (Lockhart, 1995; Lucotte et al., 1995) suggests that Hg fluxes may have increased by a factor of two or more during the past century.

Both anthropogenically and naturally-derived metals entering the arctic shelves will be processed according to their biogeochemical properties. Pb in the water column is likely to be scavenged by particles and deposited and buried in shelf sediments. Much of the Hg will also enter sediments attached to fine, organically-rich particles. However, Cd will be more likely to follow the nutrient cycle, to be taken up in plankton, released during regeneration, exported from the shelves in brine-enhanced water and, finally, to be found associated with nutrient maxima in the ocean (see Figs 1 and 5a-b; Yeats and Westerlund, 1991; Yeats and Bewers, 1987). Ultimately, Cd is removed to sediments by precipitation in sulphides (Gobel et al., 1987; Pedersen et al., 1989; Rosenthal et al., 1995). A way from the shelves, relatively clean surface ocean water has been observed both for Cd and Pb in the vicinity of Svalbard (M art, 1993) and for Cd in the central Arctic Ocean (M oore, 1981; Yeats and Westerlund, 1991). These observations suggest that contaminant Pb and Cd inputs to the interior ocean water column are either small compared to the natural budgets (likely the case for Cd) or that residence times in the upper ocean are short (likely to be the case for Pb; e.g. Schaule and Patterson, 1981). In contrast to the water, ice has a greater potential to transport over long distances contaminant metals deposited on its surface or picked up from the bottom by suspension freezing over the shelf (Pfirman et al., 1995). The melt-freeze cycle may further concentrate these metals at the ice surface. Metals deposited in ice over the Russian shelves will then be transported predominantly in the ice drift toward the Barents Sea or the Fram Strait to enter the East Greenland Current (Fig. 4). When the ice eventually melts, it will release its contaminants back to the water column.

Numerous studies have investigated the concentrations of Pb, Hg, and Cd in arctic animals (cf. Dietz et al., 1990; Muir et al., 1992 and references therein; Wagemann et al., in press). These studies have been conducted primarily to evaluate sources of the metals or the dietary intake by natives rather than to determine any effects on the animals themselves. The organs of marine mammals from the Canadian Arctic often contain Hg concentrations that exceed human consumption guidelines set for fish (0.5 µg g$^{-1}$; Health Canada) and Cd concentrations that would result in exceeding consumption guidelines (450 µg per week; Health Canada) for a relatively modest diet (for examples of metal content in marine animals see Muir et al., 1992; Kingsley et al., 1993; Wagemann et al., 1995). Hansen et al. (1991) similarly suggest that in sealing districts of Greenland, maximum Cd content in food as recommended by the World Health Organization (WHO) is being exceeded. The detection of high concentrations of these two metals in animals, however, does not necessarily imply contamination (see for example Hansen et al., 1991). It may simply reflect a high natural background. Regional differences in metal burdens in marine mammals observed in the Canadian Arctic for Pb, Cd, and Hg (Wagemann et al., 1994, 1995, in press) imply strongly that metal concentrations in tissues depend to some
Three regions of arctic shelves have experienced or promise of arctic marine biological resources by anthropogenic introductions. Despite this conclusion, it should be noted that Cd is of concern because it is observed at high concentrations in the organs of some animals (i.e. it bioaccumulates; cf. Ray, 1984) and is easily mobilized from marine sediments through small changes in oxygen content (redox conditions) or pH (cf., Gobeli et al., 1987). Our conclusion that anthropogenic loadings of Cd are not presently of concern rests on relatively low contaminant fluxes versus natural fluxes and the fact that Cd does not tend to biomagnify (Guthrie et al., 1979). However, not enough is known about the transfer of Cd into the food chain to predict with confidence how the system would respond to small incremental increases resulting from anthropogenic activities. Hg is of concern because its environmental behaviour parallels that of the organochlorines in two ways – it is volatile, entering easily into long-range atmospheric transport, and it bioaccumulates and biomagnifies in long-lived top predators. Methyl mercury, the more toxic form, is fat soluble, has an affinity for proteins, and therefore biomagnifies more than inorganic Hg (cf. Bloom, 1992). This behaviour, together with the fact that anthropogenic Hg fluxes are comparable to natural fluxes, suggests increased risks over the coming decades for populations heavily dependent on marine food sources. At the very least, Hg could be considered a pollutant if the anthropogenic fluxes are shown to lead to a greater proportion of marine foodstuffs exceeding consumption guidelines (cf. Wagemann et al., in press).

Oil exploration, production, and transport

Three regions of the arctic shelves have experienced or are likely to experience offshore oil production in the foreseeable future: the American Beaufort shelf (cf. Jaffe, 1991); the Canadian Beaufort shelf (cf. Dome et al., 1982; Thomas et al., 1982); and the shelves of the Barents and Kara Seas (cf. Doré, 1995; Klungsøy et al., 1995). Production has occurred already at Prudhoe Bay, Alaska, but only in very shallow water. Exploration on the Canadian Beaufort shelf has identified recoverable quantities of oil in the offshore but it is not likely that this oil will be produced before another decade or two. It appears that the first substantive offshore development will occur in the Russian sector where large gas and oil reservoirs have been found (Doré, 1995; Klungsøy et al., 1995).

Extensive experience in the Canadian Beaufort Sea during the exploration in the 1970s and 1980s has developed techniques to produce oil in seasonally ice-covered waters. Environmental impact assessments and estimates of risk have also been conducted (cf. Dome et al., 1982; Thomas et al., 1982). The major risks are, of course, accidents, such as well blowouts, tanker collisions, and pipeline leakages. An unconstrained blow-out in winter would be difficult, if not impossible, to remediate by drilling relief wells or mounting cleanup efforts until the following spring. Oil would accumulate beneath and within ice to be transported with it and later released into melt ponds, leads, and polynyas. The main arctic populations at risk include migratory birds and animals associated with sea ice (seals and polar bears). It should be emphasized that the marginal seas, which contain the proven oil and gas reserves, are often highly productive for fish, mammals, and birds (cf. Klungsøy et al., 1995).

If large-scale oil production proceeds in the Arctic, both chronic and acute releases of oil will inevitably occur. Spilled oil may travel long distances within the ice cover with little dispersion or degradation; clearly, this is of international concern. In the Beaufort Sea, the oil might transit the Beaufort gyre several times before eventually escaping into the Transpolar Drift (Fig. 4). In the case of the Russian or Norwegian shelves, it is more likely that spilled oil entrained in the ice will transit the Arctic Ocean to arrive in the Barents Sea or in the Fram Strait oil the east coast of Greenland in a period of one to two years (Fig. 4). Driftwood studies provide very clear evidence that oil spilled on the Russian shelves will, in part, end up on beaches in Svalbard (cf. Eggertsson, 1994).

Oil spilled into northern terrestrial drainage basins also poses a risk to the Arctic Ocean as vividly illustrated by the recent Komi spill. Some, as yet unknown, portion of the 1.2 × 10^5 m^3 of oil from leaking pipelines in Uinsk (Komi Republic) will reach the Barents Sea via the Kolva, Uasa, and Pechora rivers. As the oil travels down these rivers, it will be subject to weathering with the lighter fractions, usually associated with acute toxicity, evaporating or dissolving during transit. Upon reaching the estuary, some of the oil will become incorporated into ice forming during winter and some will become attached to particles and sediment on the shelf or disperse and further weather in the water column. Subsequent weathering and microbial attack of the oil will, however, proceed more slowly at the reduced temperatures.

In temperate regions, the major sources of oil to the marine environment are thought to be transportation and chronic municipal and industrial releases that enter the coastal ocean through runoff (National Academy of Science, 1985). Two of the largest rivers, the Lena and the Mackenzie, are apparently still pristine with respect to anthropogenic inputs of petroleum (Peulvé et al., 1995; Yunker and MacDonald, 1995). However, opening the Arctic to oil or other forms of development will result in greater traffic on the Northern Sea Route and through the Canadian Archipelago. Chronic emissions of oil in these transport lanes, parts of which pass...
through sensitive ecozones, will have the potential to cause greater problems than in temperate regions where degradation and dispersal of oil occurs more quickly.

The effectiveness of existing regulatory arrangements

When evaluating the effectiveness of existing regulatory arrangements, it is important that a distinction is made between the provisions of controls and the extent of compliance with them. In the following analysis, attempts are made to draw such distinctions to define whether existing threats or damage exist because of the absence of regulatory controls or a lack of compliance. Another factor that must be considered, when judging whether new controls are warranted, is the amenability of the source to human intervention. In cases where a source is predominantly natural or its nature and distribution makes it intractable to human influence, there is little point in considering the introduction of new control measures.

Where effective regulatory arrangements do not exist for contaminant releases that are amenable to control, the nature of prudent action is relatively obvious - formulate new control measures. However, in cases where the fault lies in the failure to implement the provisions of existing control measures, the solution, while equally obvious, presents a range of administrative difficulties, especially in an international context. It is clearly desirable that all parties to the Arctic Environmental Protection Strategy implement existing control measures in an equally committed manner. The authors of this paper are not able, in all cases, to judge the nature and extent of inequities in national compliance with control measures. Nevertheless, in some cases, it is clear that non-compliance is a significant contributing factor to contemporary anthropogenic effects on the arctic environment.

Persistent organic contaminants

The foregoing discussion has not identified incidents where organic contaminants have produced toxic effects in arctic organisms and, indeed, we would expect the more highly contaminated areas in the northern temperate marine environment to show such effects first. Nevertheless, the mechanism whereby some volatile, toxic compounds are transported rapidly on global scales, together with factors (i.e. long, fat-dominated food chains with long-lived top predators) that lead to bioaccumulation in prey and biomagnification in predators, is disturbing especially since the Arctic is so remote from major production and use regions. Contamination of country foods by these kinds of compounds has already produced anxiety in native northerners about the safety of their traditional foods and of breast feeding (cf. Kinloch et al., 1992).

For regulation, it is important to note that the production and use of many of these contaminants has already been banned or restricted in many developed countries (cf. Pacyna, 1995) and this may be reflected in a decline in burdens of PCBs and DDTs in seals (but see Table 1). Much of the volatile organochlorine flux into the Arctic comes from recycling of old residues between terrestrial, atmospheric, and aquatic media. These recycled components of the contaminants are not amenable to regulatory control. The fundamental lesson, however, is that the release in large quantities of toxic, volatile, long-lived, fat-soluble compounds can leave a long-term legacy that is of global concern. There are three critical steps that need to be taken: (1) the continued use of compounds such as DDT in many developing countries, should be restricted and/or substituted with the use of more environmentally benign compounds; (2) a global reporting and auditing mechanism to account for the production and use of such compounds should be devised and implemented by all nations; and (3) new compounds proposed for use must, as part of their screening process, be evaluated using multi-media models for their potential to cause global harm. The reduction of emissions from incinerators and power plants in northern temperate countries, especially in northern and eastern Europe, would also accomplish reductions in the delivery of chlorinated compounds and PAHs to the Arctic Ocean. The above steps could be achieved through international arrangements at regional and global levels, such as those currently being pursued under the auspices of the Convention on Long-Range Transboundary Air Pollution.

Radionuclides

The foregoing discussion of sources of radionuclides in the Arctic and the associated doses and risks, albeit presented on exemplar grounds, provides little justification for advocating the introduction of new regulatory mechanisms for practices involving the production use, and disposal of radionuclides. Reprocessing operations in Europe, and their associated radionuclide releases, are authorized on the basis of international procedures and standards, largely those established by the International Atomic Energy Agency (IAEA). Such authorization takes account of risks posed to public health and deems them acceptable. Thus, contamination of the Arctic by radionuclides from these sources can only constitute contamination, not pollution, unless some fundamental fault is found within the system of radiological protection on which such authorizations are based or its application to specific practices. The deliberate disposal of radionuclides into the environment through dumping or land discharge, either within the Arctic or elsewhere,
is clearly amenable to control. The basis of control is adequately provided by the IAEA Basic Safety Standards (IAEA, 1995) and the system of radiological protection (ICRP, 1991). However, these regulatory processes must be implemented with uniform rigour by all countries, especially those that are signatories to the Arctic Environmental Protection Strategy. It should be noted that sea dumping of radioactive waste, under the terms of the London Dumping Convention (now the “London Convention 1972”) prior to the ban concluded in 1993, was restricted to deep waters greater than 4000 m between latitudes 50°N and 50°S (IAEA, 1986). Clearly, these restrictions were not observed by the former Soviet Union.

Some steps have already been taken to consider the probability and the consequences of accidents at northern power stations (e.g. Rantalaïinen, 1995; Amosov et al., 1995; Dubkov et al., 1995) and in the servicing and decommissioning of military and civil marine reactors in Russia. It is impossible to guarantee the complete prevention of accidents. Accordingly, regulatory actions must continue to focus on reducing their probability and consequences through the analysis of design, and beyond-design, accident scenarios. These matters are well subsumed within the system of radiological protection; it remains, however, to ensure that all nations apply this system in a consistent manner.

The salient observation is that mechanisms and approaches are already in place to provide a sound basis for the protection of human health and the environment on local, regional, and global scales, including the Arctic. What is needed, therefore, is to have these mechanisms implemented ubiquitously with the same zeal as is presently the case in most countries exploiting nuclear technology. This is not to say that some previous and existing practices are not in need of enhanced reassessment. Specifically, decommissioning and scrapping of Russian nuclear submarines and safe disposal of spent fuel are clearly matters of justifiable contemporary and future concern.

In summary, for radionuclides, no new regulatory controls are needed - existing international control mechanisms need to be rigorously observed and applied in all Arctic states and elsewhere.

**Metals**

Metal levels in the environment may be enhanced locally by releases from mining (Greenland, Canadian Archipelago) or nearby industrial activity (the inner Kara and Laptev Sea shelves). Mining regulations and techniques to reduce metal mobilization from mining are already available; therefore, what is needed is compliance with best practice. Emission controls for industry would reduce the fluxes of metals reaching the Arctic Ocean from eastern Europe and Russia. Recent high-quality data suggest that the largest Russian rivers are not major sources of contaminant heavy metals (Coquery et al., 1995; Dai and Martin, 1995; Martin et al., 1993). It remains to be seen whether there are important contaminant fluxes from smaller rivers resulting from localized anthropogenic activity. A comprehensive survey of contaminants in river transport would provide confidence in the estimates of metal fluxes from rivers to the Arctic Ocean. Should it be determined that smaller rivers represent a significant point of entry of heavy metals (and other contaminants) to the Arctic Ocean, the existing UN ECE Convention on Protection and Use of Transboundary Watercourses and International Lakes (1992) may be considered as an example of the type of instrument with which to address associated concerns.

Although metals are clearly reaching the circumpolar Arctic through atmospheric transport, there is no present evidence that anthropogenically mobilized Cd and Pb are posing a wide-scale biological threat. Hg, however, is probably of greater concern for two reasons. First, there is evidence of significant anthropogenic mobilization of this metal during the past century (Nriagu and Pacyna, 1988; Pacyna and Keeler, 1995). Second, due to bioaccumulation and biomagnification, long-lived marine species in the Arctic have naturally high concentrations of Hg. Therefore, these species, and their consumers, may be particularly vulnerable to added burdens. This problem is not confined to the Arctic; it is global (cf. Nriagu, 1988) and will require global approaches to the reduction of emissions. It is noted that countries of the UN ECE region are considering the possibility of a protocol to control the emissions of certain heavy metals (including Hg, Cd, and Pb) under the Convention on Long Range Transboundary Air Pollution. Further work is urgently needed to verify Hg trends globally and in the Arctic (e.g. measurements in the atmosphere, in sediment cores, and in biota). This would provide a basis to determine the amenability and the nature of any necessary additional control measures.

**Oil**

Regulation and design of safe practices in the oil industry in the context of arctic exploration and development has a long history in the North American sector of the Arctic (Prudhoe Bay, Canadian Beaufort Sea). A decision never to produce oil from the Arctic offshore would seem difficult to achieve and unlikely to be taken. The most stringent practices can be expected only to reduce the probability of accident, not eliminate it. Environmental Impact Assessments (EIA’s), the assessment of risks, and operating practices clearly need to meet international standards and to be tailored specifically to the Arctic. A careful evaluation of the probable effects of increased ship traffic in the Northern Sea Route must
also be included. Emphasis needs to be placed on realistic assessments of risk, compliance with regulation, conformity with acceptable design, the development of proven emergency response and oilspill cleanup technology, and specifications of liability in respect to redressing damage to injured parties. The contemporary example of pipelines on the Komi Republic that have exceeded their design life is instructive. Regulation of the production and transport of oil in northern land areas must meet similarly stringent criteria as for offshore oil production and transport.

Other disturbances

Of primary importance among other disturbances to the marine environment are likely to be habitat destruction in the coastal zone and alteration of hydrology in estuaries resulting from the diversion of water for power generation (cf. Cattle, 1985; Prinsenberg, 1991; Bodaly and Johnston, 1992; Klungsøy et al., 1995). Existing damage to coastal areas through physical destruction of habitats is likely to be significant on local scales in a number of countries, especially from the perspective of indigenous peoples. Furthermore, without the imposition of additional controls to minimize the destruction of habitat and associated biological diversity, continued development of arctic marginal seas is likely to result in further damage. Greater attention should therefore be given by the States that are contracting Parties to the Arctic Environmental Protection Strategy, with respect to small-scale physical impacts on the arctic coastal environment, in order to determine what steps should be taken to protect resources and amenities.

Conclusion

This document analyzes the contemporary situation concerning contaminants that have been identified as priorities within the Arctic. The development of new control measures to protect the arctic marine environment has been advocated only for a single class of contaminants, the organochlorines. This is because their properties (toxicity, longevity, susceptibility to long-range transport, bioaccumulation and biomagnification) lead to global effects, whereas their production and use in many countries continues with inconsistent controls and with little or poor documentation. We strongly advocate the consistent reporting by all countries of production and use statistics for such compounds and that, before new compounds are brought into production, they be evaluated for their potential to cause global harm using multi-media models like that developed by Wania and McKay (1995). In the cases of the other priority contaminants, either: their effects are not of great significance in the arctic marine environment; they give rise to contamination but not pollution (e.g. lead, cadmium, and radionuclides); existing control measures are adequate (e.g. radionuclides) although, in some cases, there has been lack of compliance with them (e.g. disposal of radioactive wastes); or their sources are not directly amenable to control (e.g. PAH's from various combustion sources and organochlorines from secondary environmental reservoirs). The sole exception to the above conclusion among the priority contaminants discussed here may be mercury which is naturally present at high concentrations in long-lived arctic species and for which anthropogenic mobilization rivals the natural flux. Indigenous peoples, especially in Greenland and Canada, depend heavily on marine food from the highest trophic levels, putting them particularly at risk from additional Hg burdens in these species. The properties of mercury in many ways parallel the organochlorines. Therefore, the assessment of temporal trends of Hg in the arctic marine ecosystem should be a priority. Greater attention should also be paid to physical impacts on the arctic coastal environment (see Klungsøy et al., 1995). We note that there exists great concern about the current state of contaminants on the Russian shelves (e.g. Melnikov, 1991; Green, 1993; Pfirman et al., 1993; Luzin et al., 1994) and whether hazardous amounts of these contaminants might be broadcast to other arctic regions by ice (Pfirman et al., 1995a,b). Presently, the database for the Russian shelves is simply not adequate for the task of assessing what action is needed; it should be a priority to develop such a database. Finally, we note that pipelines being used beyond their designed lifetimes clearly pose a risk of future oil spills into northern Russian river drainage basins. In this respect, assistance to prevent future spills by inspecting and upgrading pipelines is required more urgently than new regulations.

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