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FAGRAPPORT

Chemical Pollution in the Arctic and Sub-Arctic Marine Ecosystems: an Overview of Current Knowledge

T. N. Savinova
G. W. Gabrielsen
S. Falk-Petersen

The Joint Norwegian - Russian Commission on Environmental Cooperation
The Seabird Expert Group
Report no. 3: 1994/95

Statlig program for forurensningsovervåking

NINA • NIKU

NINA Norsk institutt for naturforskning

NORWEGIAN - RUSSIAN ENVIRONMENTAL COOPERATION THE SEABIRD EXPERT GROUP

The Agreement on Environmental Cooperation between Norway and USSR was signed in 1988 and later renegotiated between Norway and Russia in 1992. The Commission - the Joint Norwegian-Russian Commission on Environmental Cooperation - is chaired by the Ministry of Environment of the two parties and has annual meetings.

Working groups on different topics have been established in order to contribute to increased collaboration on environmental problems in general, and carry out programmes and projects on different fields (i.e. air pollution, the marine environment, radioactive pollution). The seabird expert group is part of the working group for the marine environment.

The initial aim of the seabird expert group was to establish contact and collaboration between Norwegian and Russian research and management institutions. The expert group aims at contributing to the harmonisation and development of scientific methodology and data bases. Furthermore, mapping of important seabird colonies and the conditions related to seabird habitats, i.e. environmental pollutants and food resources, are important items for the group. Several projects on joint approaches have been initiated within the expert group during the last years.

Annual meetings in the seabird expert group have been arranged since 1989. The delegations from the two countries involve seabird experts from several institutions. On the Norwegian side, the Directorate for Nature Management has the coordinating role in the collaboration and chair the delegations. On the Russian side, VNII Priroda plays the corresponding role. The expert group is chaired by:

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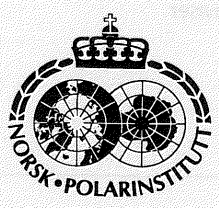

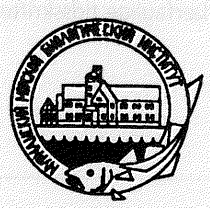

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Preface

The Ministerial Conference on Protection of the Arctic Environment in Rovaniemi 1991 promoted the international co-operation on environmental research in the Arctic. This was the start of the Arctic Monitoring and Assessment Programme (AMAP) which aims to assess the state of the Arctic Environment. This report is a part of the AMAP and is intended to review the levels and types of chlorinated hydrocarbons and heavy metal contaminants in the Arctic and Sub-Arctic marine waters, with reference to the situation in boreal waters.

This report is part of a research project in the framework of the Norwegian-Russian Environmental Cooperation. The project was initiated in 1991 in order to elucidate the present status of environmental contaminants in the arctic marine ecosystem, with special focus on seabirds. New data on contaminants in seabirds will be published in a separate report.

This report has been made possible due to good cooperation with Murmansk Marine Biological Institute (MMBI), directed by Prof. G.G. Matishov. This cooperation has lead to exchange of scientists and the development of research programmes between MMBI, Norwegian Institute for Nature Research (NINA) and Akvaplan-niva.

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1 Introduction

Arctic Environmental Protection Strategy

At a Ministerial meeting in Rovaniemi, Finland, on the 14th. of June 1991, the ministers from the eight Arctic nations (Canada, Denmark, Finland, Iceland, Norway, Russia, Sweden, USA) signed the Declaration on the Protections of the Arctic Environment. In the final document from this meeting the Arctic Environmental Protection Strategy (AEPS) states that "The Arctic is highly sensitive to pollution and much of its human population and culture is directly dependent on the health of the regions ecosystem. Limited sunlight, ice cover that inhibits energy penetration, low mean and extreme temperatures, low species diversity and biological productivity and long-lived organisms with high lipid content contribute to the sensitivity of the Arctic ecosystem and cause it to be easily damaged". As a part of the Declaration the Arctic Monitoring and Assessment Program (AMAP) was adopted to monitor the level of, and assess the effects of, anthropogenic pollutants in all components of the Arctic environment. AMAP has designed a monitoring programme with first priority on persistent organics, heavy metals and radioactivity. The eight Arctic countries have specified national implementation plans to fulfil their obligations. In addition to the eight Arctic countries, Germany, U.K., the Netherlands and Poland act as observers and Germany and the Netherlands have specified projects to be part of AMAP. Approximately 300 projects have been reported as a part of AMAP. The assessment will be based on results from these projects, data already published in international scientific literature, "grey" literature and traditional knowledge from indigenous people. This review summarizes the current state of knowledge on pathways of transport, sources, bioaccumulation and levels of chlorinated hydrocarbons and heavy metals in the Arctic marine ecosystems using published literature available to 1994.

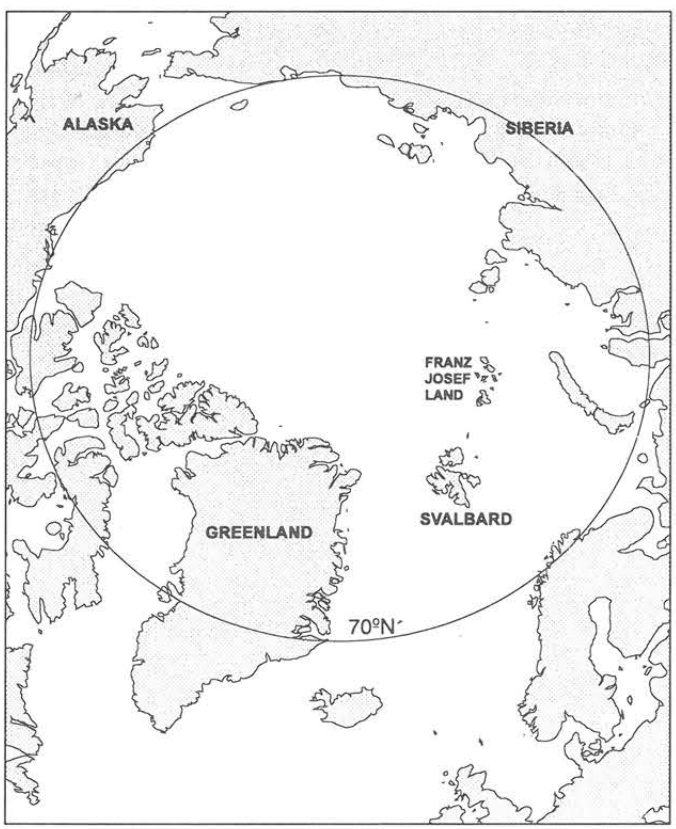

Figure 1
Map of the Arctic, 70°N is indicated.

The Arctic marine environment

The Arctic is defined as the region surrounding the North Pole (Figure 1). AMAP use the Arctic Circle as the guideline to delineate the Arctic and only countries with territory north of the circle are members of the AMAP. The Arctic circle is the mathematical delination of the Arctic. Such a latitudal definition may have no ecological significance. However, light available for the growth of the phytoplankton is the most important factor for the existence of life in the marine environment. After the vernal equinox (21 March), daylength and total global radiation increase from south to north. The peculiar light regime with midnight sun is a key factor influencing the spring bloom in Arctic waters (Falk-Petersen et al., 1990).

The development of phytoplankton in the Arctic Basin is also closely associated with the ice conditions. The bloom coincides with melting of the ice and the penetration of light into the water column. In the adjacent seas to the Pole the bloom lasts no

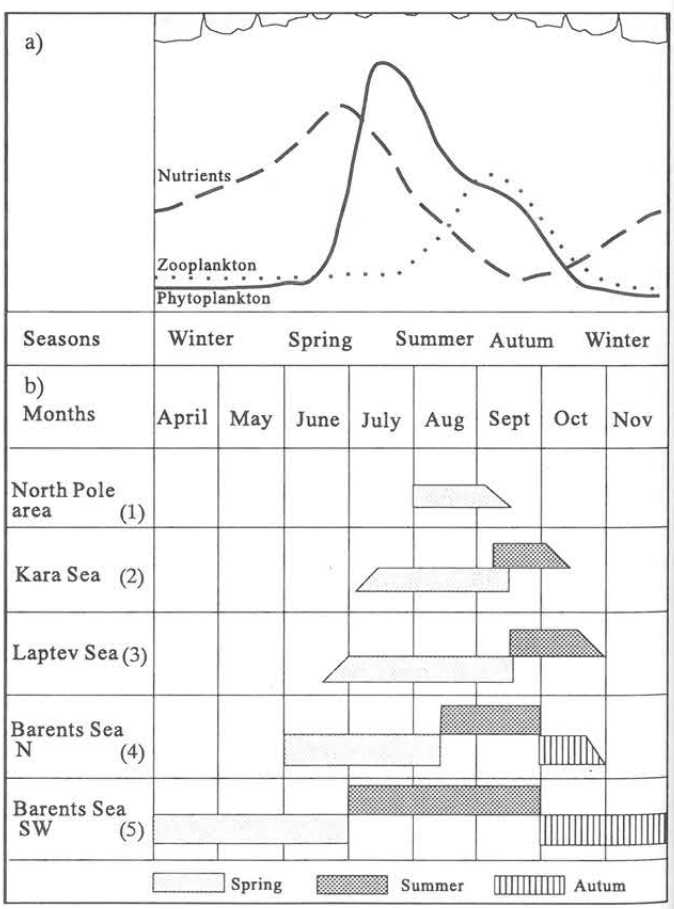

Figure 2
Seasonal progression of plankton in Artic Seas (from Zenkevitch, 1963).

a. General indices.
b. Phytoplankton development.
   1. Circumpolar part of Arctic Ocean
   2. Central region of Kara Sea
   3. Laptev Sea
   4. Northern part of Barents Sea
   5. Southwestern part of Barents Sea

more than a month (August), while in the central part of the Kara Sea the primary production period lasts nearly three months and in the Barents Sea up to eight months (Figure 2.).

As a consequence of the light regime and ice- and hydrological conditions, the primary production in the Arctic is characterized by an intensive and short bloom period (Figure 3.). Herbivorous zooplankton and zoobenthos is thus exposed to a short period of food high availability followed by a long period of food shortage. Marine animals exposed to such variation in food availability have responded to the situation by developing an ability to store large lipid depots as energy reserves to be used during periods of food shortages (Nemoto and Harrison, 1981; Falk-Petersen, 1989).

Zenkevitch (1963) divided the Arctic region into three sub-regions: 1) the Abyssal Arctic sub-region which consists mainly of the Arctic basin and the Greenland abyssal plain; 2) the shallow lower Arctic sub-region which includes the Barents- and White Seas, and 3) the shallow high Artic sub-region which includes all seas on the shelf of the Russian and North American sector (Figure 4). The two latter sub-regions can be separated into two types of marine ecosystems: 1) the pelagic ecosystem which is found in the central and western part of the Barents Sea and have depths from 100-400 m, and 2) the bentho-pelagic ecosystem which is found on the Russian and North American shelf and have depths less than 100 m.

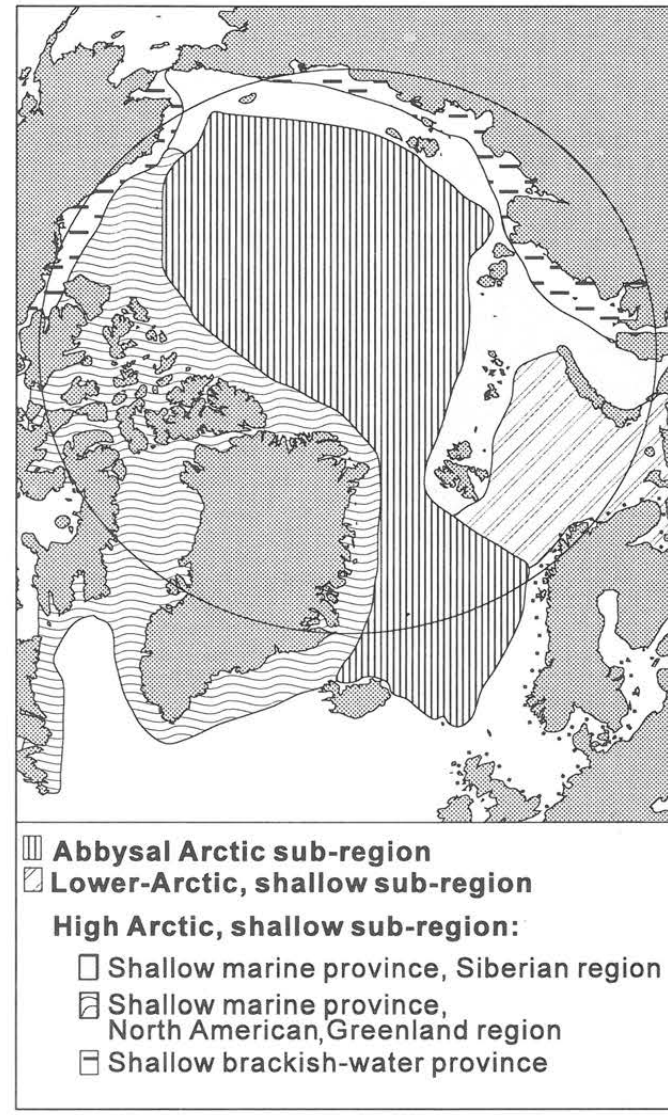

Figure 4
Map of the Arctic Seas. Zoogeographical regions (after Zenkevitch 1963).

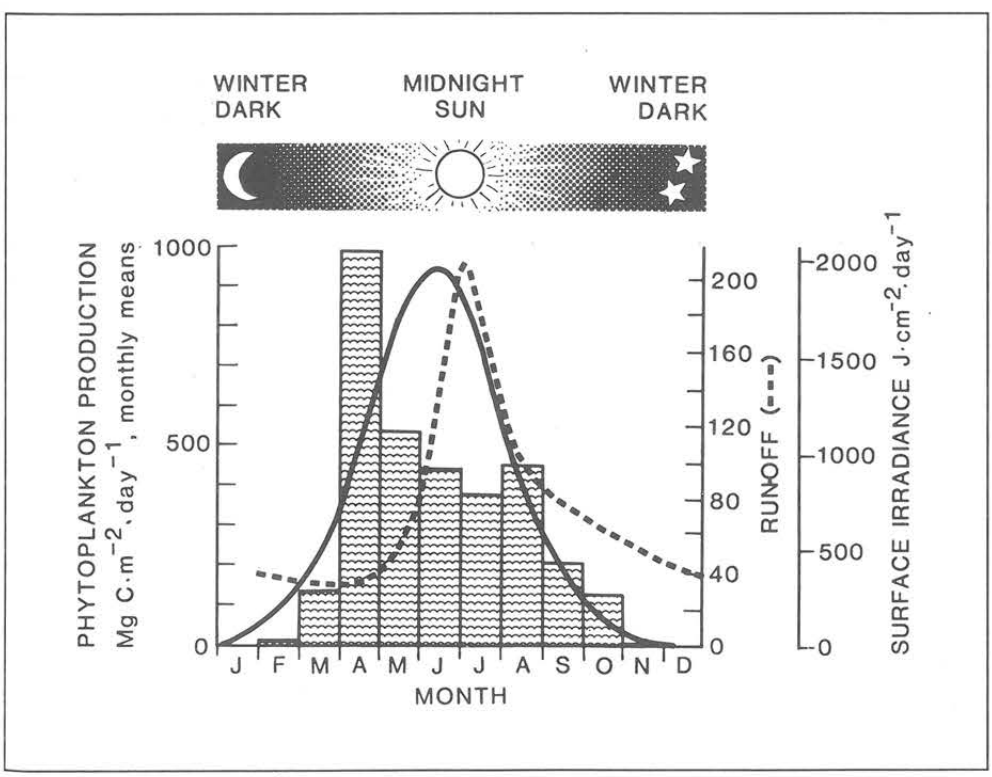

Figure 3
Seasonal cycle of phytoplankton production (histogram), solar radiation (thick, symmetrical curve) at 70°N (from Hopkins et al., 1989)

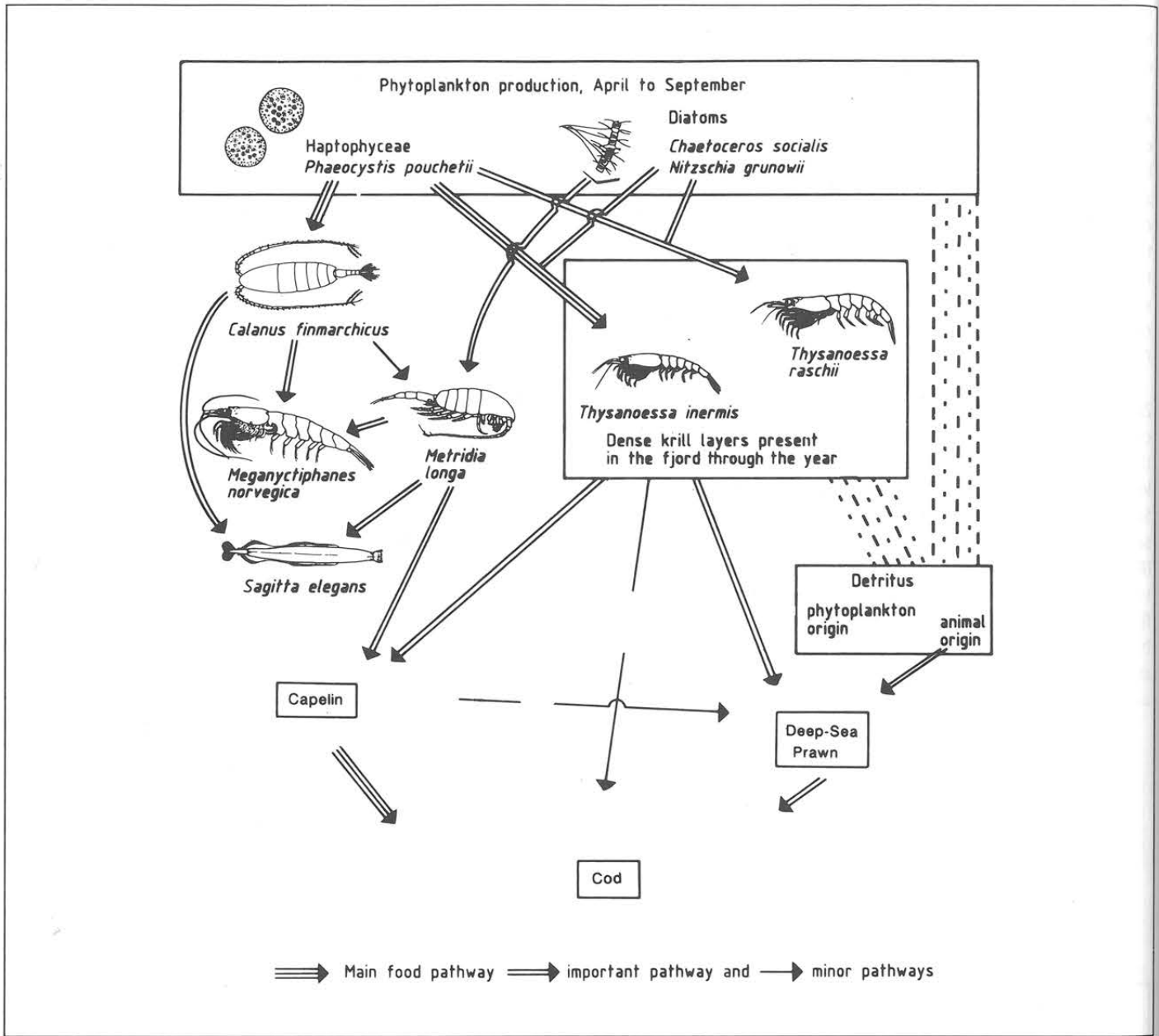

Figure 5
Generalized pelagic food web from the Barents Sea (from Falk-Petersen et al., 1990)

The pelagic ecosystem

The deeper (100-400 m), central and western parts of the Barents Sea are highly productive. The annual primary production varies between 70 and 150 g C/m^2/year (Falk-Petersen et al.,1990). A generalized pelagic food web from the Barents Sea is shown in Figure 5. Primary production is mainly channelled through pelagic food-webs, depending on factors such as depth and match / mismatch between the phytoplankton bloom and the grazing of the zooplankton (Wassmann and Slagstad, 1992). One of the most important food-chains in the Barents Sea is the copepod-krill to capelin-cod chain. The dominant zooplankton species in the pelagic ecosystem are the copepods *Calanus finmarchicus* and *C. glacialis* together with the euphausiids *Thysanoessa inermis*, and *T. raschii* (Figure 5).

Capelin feed mainly on zooplankton as they migrate northwards during the Arctic summer. In the autumn they follow the ice edge as it melts and feed intensively on zooplankton in these highly productive waters. At this time of the year copepods and euphausiids have high lipid contents (60 to 70 % of the dry weight). Capelin and euphausiids are the main food for large populations of cod, seabirds and seals.

The bentho-pelagic type ecosystem

The shelf of the Russian Arctic and the most eastern part of the Barents Sea are shallow with large areas with depths of 20-50 m. Most of the primary production will therefore sediment out from the pelagic water masses and be available as food to benthic animals. In this area there will be a bentho-pelagic ecosystem which consists of fish, seabirds and marine mammals. The Common Eider and walrus feed to a large extent on the benthic community (Figure 6). According to Zenkevitch (1963), the qualities of the littoral, pelagic and benthic fauna and flora is poor

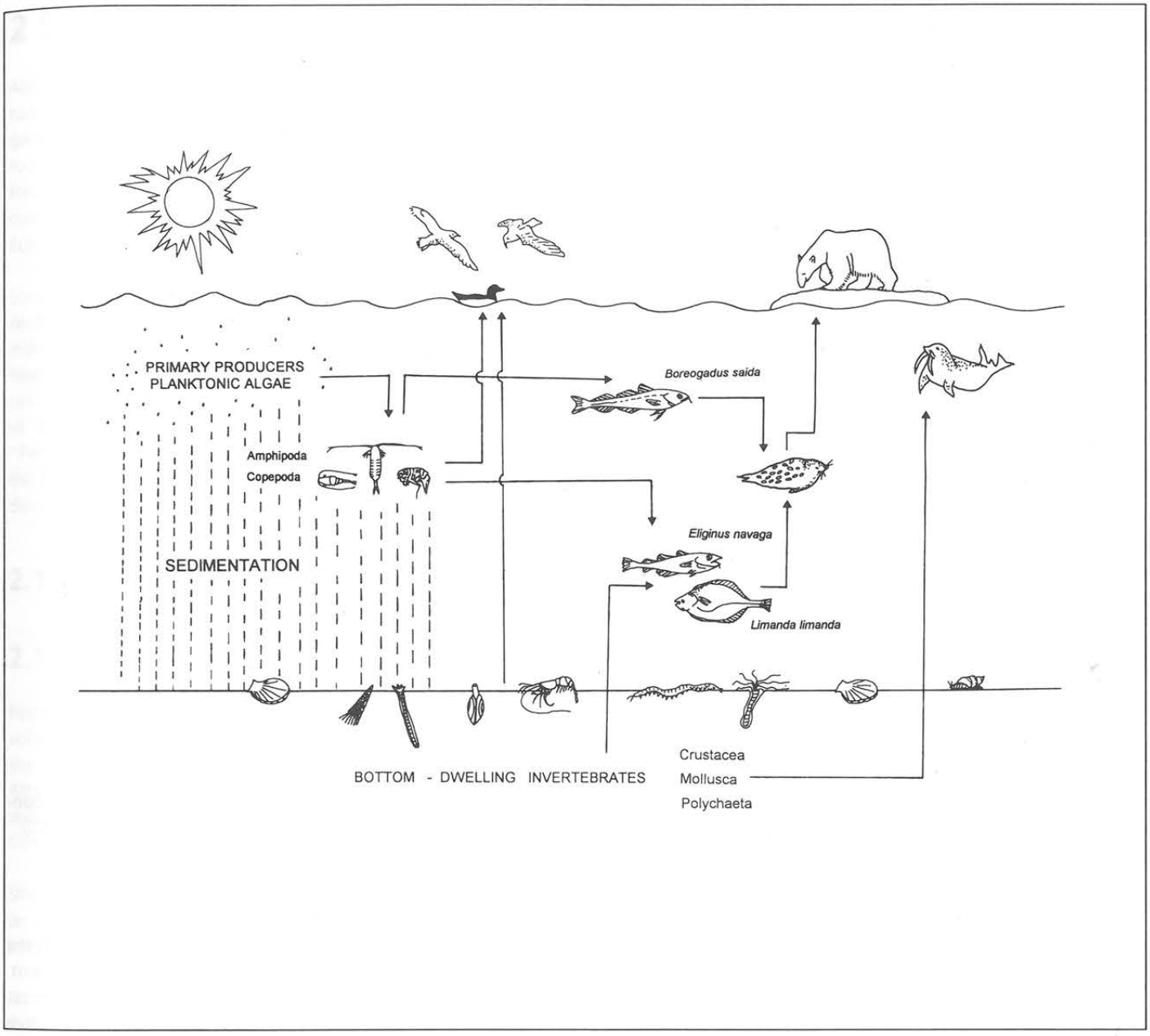

Figure 6
Generalized bentho-pelagic food web from the shallow Arctic shelf.

with reduced biomass and low numbers of species on the Russian Arctic shelf. This may be because of ice-scouring during the short primary production season, but the most important factor is probably the low and variable salinity due to the large input of freshwater from large Russian rivers.

The transfer of lipids in the Arctic Seas

Primary production in Arctic areas is characterized by a short but intensive bloom. This represents an energy peak which is rapidly transferred through the food web from the primary producers to the top carnivores such as marine mammals, seabirds and humans. The base of the pelagic food web in the Barents Sea consists of large calanoid copepods and the euphausiids which utilize the bloom effectively by converting the energy produced directly into lipids. During this period the calanoid copepods grow continuously, and the lipid content increases from 15 to 70 % of their dry weight The euphausiids have a longer growth period and utilize both the spring bloom and the primary production during the summer. During a five month period their lipid content increases from 15 to 50 %. Capelin, feeding on copepods and euphausiids while on their summer feeding grounds near the Polar Front, also lay down large lipid deposits during this period (Falk-Petersen et al., 1990; 1992). Figure 7 illustrate the transfer of lipid from the phytoplankton bloom via calanoid copepods and euphausiids, through the pelagic ecosystem to the top predators. This transfer of lipid can take place within six months.

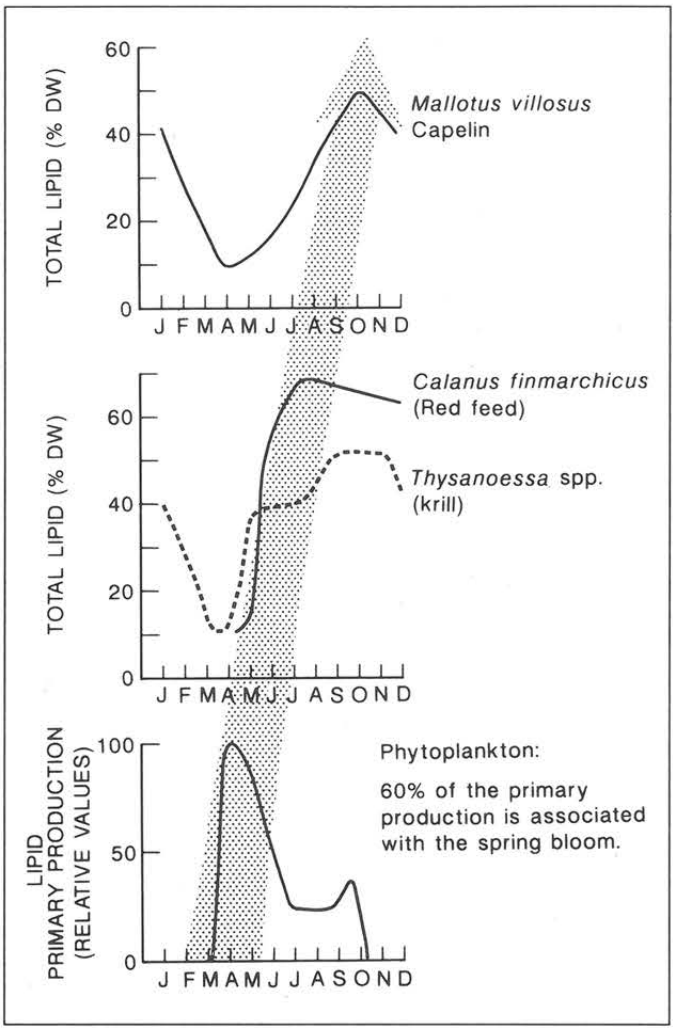

Figure 7
Transfer of lipids from the spring bloom via herbivorous zooplankton to the top predators (from Falk-Petersen et al., 1990)

The contaminants

Among the contaminants entering the ocean as a results of human activity, greatest attention has been paid to chlorinated hydrocarbons. Chlorinated hydrocarbons include the organochlorine pesticides (DDT and its metabolites, the a, b and g-isomers of hexachlorocyclohexane (HCH), aldrin, dieldrin etc.) as well as compounds with physiochemical and chromatographic properties resembling organochlorine pesticides, -polychlorinated biphenyls (PCBs) and polychlorinated camphenes (PCCs, toxaphene). However, other halogenated organics have been detected in arctic biota including chlorinated dioxins/furans, mirex, chlorobenzenes etc (Oehme et al., 1990; Muir et al., 1992c). These toxicants are characterized by biological and chemical persistence in the marine environment and have a high bioaccumulation and biomagnification potential. These substances were, by an international convention, included in the list of the most dangerous chemical substances, and their discharge was prohibited. However, these compounds are still produced in some countries and widely used on a global basis.

Due to their highly lipophilic nature, of most chlorinated organic contaminants become concentrated in the fatty tissues of most species in Arctic food chains. The highest levels of contaminants detected in tissues of top predators such as seabirds and marine mammals (Skaare et al., 1990; Bernhoft and Skaare, 1994; Gabrielsen et al., 1995; Muir et al., 1992 a, b, c; Savinova et al., 1995). As pointed out earlier, the transfer of lipids through the food-chain shows that the Arctic ecosystem is highly efficient. The lipid-soluble nature of the organic contaminants and the rapid transfer of lipids up to the top predators makes therefore bioaccumulation an issue of special concern in Artic regions.

Heavy metals, in contrast to chlorinated hydrocarbons, exist in the marine environment in a wide range of concentrations. In high concentrations heavy metals are toxic to marine fauna and flora, and the toxic effects of heavy metals can be quite different depending on the chemical form of the elements. The levels and fate of chlorinated hydrocarbons and heavy metals in different marine species vary widely depending on their ecology, feeding rates, migration routes, age, physiology and biochemistry. This review summarizes the current state of knowledge on levels, sources, pathways and bioaccumulation of heavy metals and chlorinated hydrocarbons. Data from the Arctic Seas will be compared with data from other regions.

2 Sources and patways

Although the major use of organochlorines has been at mid-latitudes, and the Arctic has few sources of its own, virtually all organochlorines detected at southern latitudes have also been found in the Arctic. These contaminants reach the Arctic environment via long-range transport through air, rivers and ocean currents from industrial centres, particularly in Asia, central Europe and North America.

Long-range transport by air and ocean currents from industrial centres is also important for heavy metals. However, increased industrial production and growing settlements in the Arctic areas have increased the local input of contaminants from local sources such as mining areas, power plants, oil fields, military and civil dumping sites. Examples are mining-metallurgical plants "Pechenganickel" and "Severonickel" at the Kola Peninsula and the Norilsk industrial region in Taimyr, and several large military dumping sites in the eastern Barents Sea (Matishov, 1992).

2.1. Air

2.1.1. Chlorinated hydrocarbons

Pesticides enter the atmosphere from spray drift, evaporation, volatilization and wind erosion. Industrial organochlorines enter the atmosphere through incineration and volatilization. Atmospheric transport on regional and global scales is recognized as a major pathway for the transfer of contaminants to the Ocean.

Since the atmosphere is a major pathway of chemicals to the polar regions, considerable effort has been devoted to measuring toxic organic compounds in the arctic atmosphere.

During the early 1980s, the Norwegian Institute for Air Research (NILU) measured organochlorine compounds at several stations in the European Arctic (Oehme and Mano, 1984; Pacyna and Oehme, 1988; Oehme, 1991). Measurements have been made in the Canadian Arctic (Barrie, 1986; Hargrave et al., 1988a; 1989; Patton et al., 1988; 1991; Bidleman et al. 1989; 1990; Barrie et al., 1992; 1993) and in the Bering and Chukchi Seas (Hinckley et al., 1991).

Between 10 and 70% of pesticides are known to be released to the atmosphere during use (Hurting, 1972). Approximately 50% of ∑DDT enters the atmosphere through evaporation from upper soil layers (Lloyd-Jones, 1971). The contribution of atmospheric origin to the total input of pollutants into the marine environment amounts to 80-90% (Patin, 1977). One estimate suggests that about 374 000 t of a total world production of 1.2 mt PCBs already has been dispersed in to the environment (Tanabe, 1988). Atmospheric inputs contribute to more than 80% of the total input of selected organochlorines and up to 90% for HCH (Warmenhoven et al., 1989).

In the Arctic atmosphere one has detected DDT and its metabolites DDD and DDE, pesticides such as technical chlordane, heptachlor, hexachlorocyclohexanes (HCHs), dieldrin and polychlorinated camphenes (PCCs or toxaphene), industrial chemicals and combustion by-products such as polychlorinated biphenyls and chlorobenzenes (PCBs) and finally polychlorinated dibenzo- p- dioxins and -furans.

The HCH are the generally dominant organochlorine compound in Arctic air with concentrations ranging from 100 to 1000 per m^3 air. Hexachlorobenzene (HCB) is also prevalent with concentrations falling in the range 100 to 200 per m^3 air (Oehme and Ottar, 1984; Pacyna and Oehme, 1988; Hargrave et al., 1988a; 1989; Patton et al., 1988).

PCBs, DDTs and chlordanes in the Arctic atmosphere are generally detected at concentrations one order of magnitude lower than in temperate regions.

Based on concentration data in Arctic and Antarctic air, snow, atmospheric deposition, biological samples, measured in different investigations, Vania and Mackay (1993) suggested that compounds with vapour pressures in a certain low range may preferentially accumulate in polar regions.

2.1.2. Heavy metals

A preliminary review of the atmospheric emissions of various trace elements from anthropogenic sources in Europe was prepared in 1979/1980 (Pacyna, 1983, 1984; Pacyna et al., 1984 a,b). The earlier surveys were concerned with either a single metal (Hutton,1982) or certain types of emission sources (Pacyna, 1982). An estimate of trace element emission from anthropogenic sources in Europe, made by Pacyna (1984), is shown in Table 1.

The most likely sources of aerosols in the Arctic are the atmospheric emissions from Europe and Central North America. The first major studies of the deposition of heavy metals from the atmosphere started in the United Kingdom at the beginning of the 1970's (Cawse, 1974). The total concentrations of some 40 elements were determined in wet and dry deposits at seven non-urban sites in the UK, as part of a survey commenced in 1970-1971 on behalf the Natural Environment Research Council (Cawse, 1980). At the same time, analyses of the deposition of heavy metals were made in Norway (Brekke, 1976). Routine measurements of trace elements in precipitation started at a few stations in Norway in 1978 (Hanssen et al., 1980). Table 2 shows that the precipitation decreases along a south-north gradient.

Moss samples have also been used in studies of the long-range transport of pollutants to the Norwegian Arctic (Ottar et al., 1986). Generally, the concentrations decrease when approaching the Norwegian Arctic, and the low variability observed for the concentrations at different locations in the Arctic, suggest that longrange transport of air pollutants is the major contributor.

Table 1
A comparison of estimated atmospheric emission (tonnes/year) of anthropogenic trace elemenst in western, central and eastern Europe in 1979 (Pacyna, 1984)

Element	Emission (tons/year)		
	W*	Central**	E***
As	1600.0	1320.0	2810.0
Be	10.0	20.0	15.0
Cd	868.0	525.0	816.0
Co	499.0	511.0	631.0
Cr	5580.0	3810.0	7150.0
Cu	3390.0	2370.0	6540.0
Mn	5410.0	3330.0	6870.0
Mo	198.0	267.0	257.0
Ni	3830.0	2920.0	6010.0
Pb	39.0	16.9	43.8
Sb	153.0	104.0	80.0
Se	126.0	111.0	120.0
V	9480.0	5750.0	11.3
Zn	30.1	14.2	21.3
Zr	403.0	615.0	471.0

*Western Europe includes Belgium, Denmark, France, the former West Germany, Ireland, Netherlands, Norway, Switzerland, the United Kingdom, Luxemburg.
** Central Europe includes Austria, Czechoslovakia, Finland, the former German Democratic Republic, Hungary, Poland, Sweden, Romania.
*** Eastern Europe includes the former Soviet Union West off the Urals.

2.2. Rivers and Ocean Currents

Long-range transport of contaminants also occurs via rivers and ocean currents. Transport by ocean currents is, however, a slow process; while transport via atmosphere takes days, the oceanic circulation takes several years. Aarkrog et al. (1987) estimated the water exchange rate between the North Sea and the Arctic Ocean by using the fingerprint of the radionuclide emission from Sellafield, Scotland. About 1% of the North Sea water is transported to the region around Spitsbergen and to West Greenland within 3-5 years, and about 1% ends up in the Baffin Bay after 8-10 years.

Contaminants may enter these pathways though direct run-off from land or through atmospheric deposition. The Arctic has low levels of precipitation, and the largest input of fresh water to the Arctic ocean is from Yenisey River, Lena River and Ob River which account for 65.5% of river drainage into Arctic Ocean. As a comparison the Mackenzie River accounts for 15.9% (MacKay and Loken, 1974). These rivers have large drainage basins and may be responsible for a significant input of organochlorines into the Arctic Ocean. Few data are available on this pathway, although a preliminary study was presented from the Mackenzie River (Jensen, 1991) and from Yenisey River at the AMAP meeting in Tromsø (Norway) in 1994 (Champ, pers. com.).

Research on the distribution of pollutants in waters of the North Atlantic, carried out in the 1970's by scientists of the State Oceanographic Institute (GOIN, Moscow), clarified the role of the basic circular systems and of relatively stagnant zones of oceans and seas in the transport and accumulation of pollutants (Simonov et al., 1974; Simonov, 1978). The greatest concentrations were recorded in coastal zones and in extensive, relatively immobile ocean regions into which the pollutants are carried by the main current systems.

Table 2
Concentrations of trace elements in precipitation (µg/l)

Element	England (Chilton)	Denmark (Hovmand)	Sweden (Velen)	Sweden (Gardsjön)	Norway (Birkenes)
Ca	1900.0	-	250.0	170.0	-
Al	600.0	-	80.0	17.0	-
V	7.0	-	3.0	0.3	-
Cr	3.0	-	1.0	0.2	-
Mn	14.0	15.0	8.0	2.0	7.0
Fe	350.0	150.0	100.0	17.0	-
Ni	11.0	-	2.0	0.4	-
Cu	24.0	2.5	5.0	0.7	-
Zn	75.0	25.0	25.0	28.0	18.0
As	2.0	-	1.0	-	0.8
Cd	-	0.3	0.3	0.2	0.3
Pb	40.0	15.0	20.0	6.0	11.0
Source	Cawse, 1981	Hovmand, 1979	Granat, 1982	Grahn and Rosen, 1983	Hanssen et al.,1980

3 Levels of contaminants in Arctic marine waters

3.1. Chlorinated hydrocarbons

In the early and mid 70's the DDT concentrations in surface waters of the North Atlantic did not exceed 1 ng/l (Harvey et al., 1973; Orlova, 1977; 1978; Brugmann and Luckas, 1979). Jonas and Phaender (1976) studied the distribution of organochlorine pesticides in waters of the North Atlantic (from the surface to a depth of 1000 m) and determined the average concentration in residual quantities of DDE at 3.8 ng/l. In the surface layer, however, the average ∑DDT concentration did not exceed 4 ng/l. The levels of organochlorine pesticides in North Atlantic surface water (36-52°N and 9-53°W) were monitored for a 4 year period by the former Soviet Union State Oceanographic Institute (Orlova, 1985) (Table 3).

Table 3
Changes in the concentrations of organochlorine pesticides (ng/l) in the surface layers of the North Atlantic waters during period 1977-80 (Orlova, 1985)

OCH	1977	1978	1979	1980
DDT	19.2-0(1.2)	42.2-0(0.8)	2410-0(0.9)	17.6-0(0.8)
DDE+DDD	3.8-0(0.6)	3.6-0(0.2)	6.1-0(0.6)	1.8-0(0.3)
HCH	2.8-0(0.5)	5.2-0(0.3)	4.2-0(0.2)	4.9-0(0.1)

This investigation shows that the lowest level of DDT and its metabolites for the entire period of investigation was found in the sub-Arctic region in 1977 with 0.6 ng/l, 0.2 in 1978, 0.6 in 1979 and 0.3 in 1980.

However, the situation for HCH was different, with maximum concentrations found in the sub-Arctic zone. It has been shown (Mikhailov et al., 1982) that the concentration of organochlorine pesticides in the surface microlayer (SML) in the northeastern part of the Atlantic Ocean reaches 100 ng/l. Their distribution in the water is irregular with high concentrations of toxicants found particularly in the SML in shelf zones of Ireland (more than 100 ng/l), Iceland (approximately 80 ng/l) and Norway (up to 80 ng/l). The area through which the North Atlantic current passes is characterized by a change in concentrations of pesticides in the SML from 10 to 30 ng/l. As this water mass moves eastward the concentrations of contaminants decrease to 20 ng/l, which is linked to a significant transformation in this water mass (volatilization losses), to dispersal of these substances in space, and to evaporation. The ∑DDT concentrations in the SML in this region range from 0 to 70 ng/l. A distinctive feature of the ∑DDT distribution in the SML is its insignificant quantities (less than 10 ng/l) in the shelf waters of Ireland and Great Britain. In the Atlantic waters of the Barents Sea, according to data obtained by the Polar Institute for Fishery and Oceanography (PINRO, Murmansk) in 1989, maximum concentrations of beta (b)- and alfa (a)- HCH were 115 ng/l (Kiliezhenko, pers.comm.).

In the early 1970's PCB levels in surface waters of the North Atlantic raged from 25 to 41 ng/l, with maximum concentrations reaching 150 ng/l (Harvey, 1972; Harvey et al., 1973; Ostenberg and Keskes, 1977). But by the summer of 1972, after the use of these substances had ceased, the average PCB level in surface waters in the open part of the Atlantic Ocean was 35 ng/l, and 10 ng/l at a depth of 200 m (Harvey et al., 1973). In 1973, PCB concentrations in surface waters of the North Atlantic declined to 12 ng/l, and in 1974 it reached 0.8 ng/l (Harvey and Steinhauer, 1974a; Ostenberg and Keskes, 1977).

In the ICES area, the distribution pattern of gamma (g)-HCH in 1986 showed a gradient from low concentrations (0.2 ng/l) in the northwestern North Sea (similar to those found in open North Atlantic waters, and resulting primarily from atmospheric inputs) to more than 4 ng/l in the German Bight. In 1990-1991, high concentrations of g-HCH were observed in certain estuaries (5.3 ng/l in the Humber Estuary and 15 ng/l in the Elbe and Weser estuaries). The sum of seven PCB congeners (IUPAC Nos. 28, 52, 101, 118, 138, 153, and 180) ranged from 16 pg/l in French, Belgian, and Dutch offshore areas to 19 pg/l in northern Dutch coastal waters. A high concentration (83 pg/l) was also observed in the central North Sea along a transect traversing the area occupied by, the among others, the Argyll, Auk, Ekofisk, and Fulmar oil fields. The pattern of PCBs in this sample was different from those seen in other samples, but identical to Aroclor 1254 or Clophen A50. This suggests that the higher concentrations observed here may reflect recent inputs, possibly from the oil industry.

In the waters of the Barents Sea the largest concentration of total DDT was detected in Atlantic waters of the Murmansk coastal current. In the open part of the sea and eastern coastal regions the chlorinated hydrocarbons were significantly lower at the background levels (Savinova et al., 1981). In the period from 1983 to 1987 chlorinated hydrocarbons were measured in the Kolsky and Motovsky Bays and Teriberskaya Inlet (Russia). Concentration of DDT were low, while ∑DDT was present at concentrations about 1036 ng/l. The highest concentrations were found in places of industrial and domestic effluent runoff (Kalitovich, 1991).

Measurements of chlorinated hydrocarbons in Arctic marine water and ice carried out by Hargrave et al. (1988 a,b) and Bidleman et al. (1989), are presented in Table 4.

HCH, HCB and toxaphene were the major chlorinated hydrocarbons found in sea water, ice meltwater and snow samples taken during 1986 and 1987 at the Ice Island located at approximately 81°N, 97°W. ∑DDT was the most abundant component in the ice (1321 ng/l). The concentrations and proportions of all major chlorinated hydrocarbons were lower in ice than in sea water suggesting that ice was not the source of these contaminants. HCB, chlordanes, DDT, DDE and PCBs were below levels of reliable quantification (<2 pg/l) in ice. Greater potential volatilization of DDT and PCBs, as well as higher absorption to sediment particles, may explain the low levels of these compounds in Arctic sea waters (Tanabe and Tatsukawa, 1983; Hargrave et al., 1988a).

Table 4
Organochlorines in sea water, ice and suspended particulates (pg/l) at the Ice-Island (81° N, 97°W) (Hargrave et al., 1988a; Bidleman et al., 1989)

Sample type, date	Depth (m)	N	∑DDT	HCB	Dieldrin	HcH	∑PCB
Sea water							
05 1986	10-60	6	4399±591	17±6	15±2	<4-14	7±5
08 1986	10-60	5	5183±545	22±7	14±3	<4	<4-14
Ice							
05 1986		3	1507±342	<2	6±1	<2-12	<23
Suspended particulate							
05 1986	10-60	6	<0.2	<0.1	<0.3-2.7	<0.3-1.0	<1.0-5.9
08 1986	10-60	4	<0.7-1.6	<0.3	<0.3	<0.9	<1.9

Gaul (1992) found concentrations of b- and a-HCH isomers in seawater from two sites in European Arctic waters (73°30'N, 15°55'W and 77°N, 1°33'W) which averaged 2.8 ng/l. Concentrations of ∑DDT in Arctic Ocean waters between the Norwegian coast and Spitsbergen did not decline significantly between 1979 and 1985. Concentrations of HCH isomers were lower in sea ice than in seawater while PCB congeners, DDT-related compounds and HCB were detectable in particulates associated with ice (Gaul, 1992).

Vlasov and Melnikov (1990) found that organochlorine concentrations in surface sea water, collected from 1986 to 1989 in the Kara, Laptev, East Siberian and Chukchi Seas, were similar to those observed in the Norwegian Sea in 1985 by Gaul (1992). Levels of ∑DDT in these two studies were about two-times lower than in seawater, from the Canadian Arctic (Hargrave et al. (1988); Bidleman et al. (1989). The highest concentrations of ∑DDT measured by Vlasov and Melnikov (1990) in waters of Russian Arctic were in samples from the Ob and Gydan River inlets (4.0-4.5 ng/l).

Gaul (1992) observed a gradient of ∑DDT concentrations between the North Sea and Spitsbergen which reflected the importance of European sources of HCH isomers. ∑DDT levels ranged from 4.8 to 6.2 ng/l in the North Sea and 0.9 to 1.5 ng/l in the Norwegian sea south-west of Spitsbergen.

A distinct decline in concentration with depth was observed for HCHs, HCB, and toxaphene in Arctic Ocean water at the Ice Island (Hargrave et al., 1988a). The gradient was less distinct for cyclodiene pesticides and undetectable for DDT and PCBs. Such a gradient is consistent with faster removal of the more hydrophobic compounds via sedimentation on sinking particles.

3.2. Heavy metals

The concentrations of trace metals in ocean waters are extremely low. Since contamination may occur during sampling and analysis, it is difficult to determine trace metals properly. Early published trace metal results for sea water are known to have questionable reliability (Patterson, 1974). As problems with accuracy and reliability of such data became evident, the quality of results improved, and the mean trace metal concentrations reported for the sea water decreased relative to earlier results. The most important toxic metals are Pb, Cd and Hg.

Lead has been investigated most extensively for reliability and accuracy (Patterson et al., 1976). Mart and Nurnberg (1984) reported a lead concentration in the eastern Arctic ocean of 15 ng/l at the surface and 34 ng/l at 1500 to 2000 m; in comparison, surface values of 29 to 41 ng/l in the North Atlantic and the Norwegian Sea, respectively, and 80 to 400 ng/l in surface waters off California (Tatsumoto and Patterson,1963). The lead concentration is higher in surface waters than in deep layers. In coastal waters it can reach 50 ng/l, and can be much higher in heavily polluted waters (Burnett et al., 1980). Probably more than any other metals, lead has been anthropogenically elevated in the northern hemisphere. Contamination by industrial lead is global. The "Black Angel" leadzinc mine at Maarmorilik on the west coast of Greenland, 500 km north of the Arctic Circle, in operation since 1973, has discharged mine tailings (468,000 tons per year) directly into the Affarlikassaa fjord (40 m deep) since its inception. In 1983, the lead concentration in the water of the fjord was still 78,000 ng/l (Sprunk Jansen, 1983), in spite of a reduction in residual lead concentration in the mine tailing discharge since 1979. Concentrations of lead were reported by Thomas et al. (1982) for the waters of Strathcona Sound, a northern Baffin Island fjord receiving both fluvial and aeolian inputs of lead from nearby areas of lead-zinc mineralisation and mining activities at Nanasivik. Lead concentrations were found in the range of 40 to 110 ng/l, or approximately an order of magnitude higher than open ocean background concentrations.

Moore (1981) reported cadmium concentrations of 70 ng/l in the central Arctic Ocean in the upper 100 m, and 20 ng/l to a depth of 2,500 m. Campbell and Yeats (1982) obtained similar results for cadmium in the eastern Arctic Ocean: 13 to 21 ng/l, at depths of 1,500 to 2,000 m, but a lower value, 8 ng/l, at the surface (0.5 m). A comparison with values for the North Atlantic and the Norwegian Sea shows no significant mid-oceanic differences for cadmium (Mart and Nurnberg,1984). No surface de-

pletion of cadmium was found in the Norwegian Sea (22 ng/l in surface water) according to Danielsson et al., 1985. Near the shore or in fjords, the concentration is generally higher in areas of mineralisation and local contamination. A lead-zinc mine (Nanasivik) has been in operation since 1976 on Strathcona Sound. Wagemann et al. (1983) reported under-ice cadmium concentrations of 200-300 ng/l at a depth of 10 to 20 m near shore in the vicinity of the mine and 5,700 ng/l at 2 m depth. In the Affarlikassaa fjord on the west coast of Greenland, which has received mine tailing from Maarmorilik lead-zinc mine for more than a decade, the cadmium in the water was still 1,100 ng/l in 1981, despite improvements in the treatment of mine tailings (Sprunk-Jansen, 1983).

Baseline levels of mercury in sea water have been properly determined only in recent years. Most older published levels are now being recognized as too high. There is very little information available on the concentrations of mercury in Arctic waters. Weiss et al. (1974) reported mean mercury concentrations of 1,122 ng/l (0-400 m) for the southern Beaufort Sea. Thomas (1983) reported background values of 1 to 15 ng/l for total mercury in samples from the Beaufort Sea shelf. For comparison, the average mercury concentration was 4.1 ng/l in the Gulf Stream, 8 ng/l in the Sargasso Sea, and 34 ng/l in the waters around the England (Mukherji and Kester, 1979), off the Swedish west coast, the values ranged from 5 to 12 ng/l (Gustavsson and Edin, 1985).

The heavy metal contents data in seawater in the North sea in 1985-1987 have been summarized in an ICES report (ICES, 1991). Data reported for mercury, indicated that the highest values were found in the north-central and central North Sea (8 ng/l and 5 ng/l); lower values were found in the coastal waters off Scotland, England, the Netherlands, the western Norway; and the lowest values were in the English Channel (0.5 ng/l). Cadmium concentrations declined from 0.03-0.05 ng/l in coastal waters to 0.01-0.02 ng/l on the central and northern parts of the North Sea, and were similar to those reported for ocean waters (0.01 ng/l). The highest concentrations of dissolved lead were found in the central North Sea (0.06 mg/l). Dissolved copper concentrations of 0.1 to 0.4 ng/l were found in offshore areas of the North Sea, with the lowest concentrations occurring in the central part of the northern North Sea. Higher concentrations (mean 0.11ng/l) were observed in some near-coastal areas, such as the German Bight. Data from ICES study showed that the range of dissolved zinc concentrations is fairly narrow (1.6 - 2.2 mg/l) in coastal areas of the North Sea, along the east coast of England, mean concentrations exceeded 7 ng/l. Data on dissolved arsenic were reported from the Strait of Dover, where concentrations ranged from 1.2 to 1.7 mg/l, similar to the background concentrations in Atlantic waters (1.4 ± 0.1 mg/l).

It has been shown by Russian scientists from PINRO (Murmansk), during the spring of 1990, higher concentrations of heavy metals in Nordcape Current waters (the Barents Sea): Pb 0.027, Cd 0.01, Cr 0.002-0.008 mg/l; than in the Norwegian Sea, an average: 0.008 mg/l Cu, 0.08 mg/l Zn, 0.17 mg/l Fe. The maximum concentrations of Cu and Zn in April were at Vesterålen Bank 0.024 mg/l and 0.872 mg/l, respectively and in July at Malangen Bank, were 0.042 mg/l Cu and 0.648 mg/l Zn (Mukhina et al., 1991).

It is clear that more studies on heavy metals level in waters of the Barents sea are needed, especially in connection with oil and gas exploration on the shelf.

Dissolved and particulate trace elements have recently been determined in the Lena River and its mixing zone in the Laptev Sea. Average dissolved concentrations (in nM) in this region were: As - 2.0, Cd - 0.05, Cu - 9.7, Ni - 5.2, Pb - 0.08, Zn - 5.3 and Fe - 410. These concentrations are comparable to and often lower than those observed in large unpolluted world rivers (Martin et al., 1993).

4 Current state of knowledge of contaminants in biota

4.1 Quality of the environmental control data in biotic and abiotic-samples

The acquisition of relevant and reliable data is an essential part of any research or monitoring programme. To obtain such data, representative samples must be collected and stored properly prior to analysis, and all laboratory analytical work must be carried out and validated under a quality assurance (QA) programme.

The contamination levels of different pollutants in biotic and abiotic samples vary widely depending on analytical equipment and methods. The series of external quality assessments of analysis, organised for some European and North American laboratories over the last 20 years by the International Council for the Exploration of the Sea (ICES), have shown that there are large interlaboratory differences in measurements of contaminants i marine samples: trace metals in biota (Topping, 1982; 1986; Berman and Boyko, 1986); trace metals in sea water (Olafsson, 1978; Berman and Boyko, 1988); organochlorine compounds in biological tissue (Topping and Holden, 1978; Uthe and Musial, 1986). Other external quality assessments were organised by the Food and Agricultural Association (Knowles and Burrell, 1981), the International Atomic Energy Agency (IAEA, 1980), and the International Oceanographic Commission (Topping, 1984).

Early PCB analyses by gas chromatography only gave a rough estimation of total PCB concentration based on comparison of combined peak areas between extracts and PCB mixture (usually Aroclor 1254).

The use of Aroclor equivalents overestimates PCB levels in biota because of extensive transformation of congeners with adjacent unsubstituted carbons (Duinker and Hillebrand, 1983; Tanabe et al., 1988). Norstrom et al. (1988) found that PCB levels in polar bears *(Ursus maritimus)* reported by Bowes and Jonkel (1975), as Aroclor 1260 equivalents, were 1.9 times higher than when reanalysed as total PCB. In the case of belugas *(Delphinapterus leucas)* and ringed seals *(Phoca hispida)*, which show less transformation of PCBs than polar bears, Aroclor 1254:1260 (1:1) equivalents were about 1.2-times and 1.8- times higher, respectively, than total PCB (Muir et al., 1988;1992a).

Recent advances in the identification and quantification of chlorinated hydrocarbons through mathematical and computer-assisted techniques will prove useful in data interpretation of metabolic fate studies. Unfortunately, the results of PCB analyses vary widely among co-operating laboratories. An interlaboratory comparison of spiked and unspiked samples of herring *(Clupea harengus)* oil by 23 participating European and North American laboratories has shown that the calculated spike recoveries ranged from 23 to 136% (Musial and Uthe, 1983; Uthe and Musical, 1986).

Not only is the analytical procedure is important for determination of chlorinated hydrocarbons; procedures such as: sampling, extraction, clean-up also need to be correct in order to increase the presition. Organochlorine determinations in seawater have been made by two methods. One is the direct solvent extraction of bulk seawater, occasionally prefiltered through mesh to remove larger organisms. The other is concentration of the samples on resins of Amberlite XAD-2, polyurethane foam, activated carbon or other absorbents (Sericano and Pucci, 1984). The direct extraction approach combines organochlorines in solution with those bound to both microorganisms and inert particles, whereas the sorbent methods allow a proportion of the organochlorines associated with particulates to escape detection.

Cox (1971a) found, after prefiltering seawater through 176 mm mesh to remove large organisms and concentrating particulates by centrifugation and filtration on GF/C filters, that between 2 and 10% of the ∑DDT residues in seawater were associated with the filters. Tanabe and Tatsukawa (1983) reported that 14 to 88% of the ∑DDT, 7 to 58% of the PCBs, and 0.05 to 2.4% of ∑DDT residues in surface waters of the Western Pacific to Antarctic regions were retained on GF-50 filters (0.5 mm).

Due to differences in the methods used, and detection limits, it is very difficult to compare previously obtained data, especially for water samples. The measurements made by the Hydrometrological Service in the former Soviet Union are, according to their standard methods for chlorinated hydrocarbons and heavy metals in seawater, based on samples from two liters of water (Methodicheskie ukazaniya, 1982). Results from investigations using such a small volume of water should therefore be treated with care.

4.2. Plankton

Phytoplankton is one of the main accumulators of contaminant substances. It plays a central role in the distribution of toxins in aquatic ecosystems .

Patin (1977; 1979) suggested that the levels of organochlorines recorded in sea water and plankton may significantly reduce the rates of primary production in the world's oceans. A global reduction of 10% in primary production may result in a corresponding reduction in production at other trophic levels as well.

Measurements of the fluorescence of chlorophyll-a have shown that PCBs accumulate in photosynthetically active centres sensitive to the effect of these compounds (Harding and Phillips, 1978).

Further, the concentrations of DDT and PCBs in phytoplankton attain equilibrium with a concentration in the environment in the course of several seconds or hours, depending on the alga species and the physiochemical properties of the toxicants (Biggs et al., 1980). The chain "diatomzooplankton" may be one of the most important chains in the transport of chlorinated hydrocarbons since the presence of fatty inclusions in diatoms results in a high accumulation of these contaminants. It has been shown experimentally that the coefficient of accumulation of DDT in the

diatom *Cyclotella* sp. may reach 3.7 x 10^4 units, and 3.2 x 10^4 units in *Sceletonema* sp. (Ernst, 1975).

The literature on the contents of chlorinated hydrocarbons in phytoplankton samples are few and the results are difficult to compare. This is mainly because the residues are frequently expressed in different units (i.e., plankton weight or lipid weight, or volume of water) and samples can be contaminated by surface films and floatables during field collection. Table 5 presents data from a small number of analyses, which characterize the accumulation of chlorinated hydrocarbons in phytoplankton and microplankton. The issue of the availability of organochlorines in seawater and zooplankton is unclear because we have only recently attempted to deal separately with the dissolved and suspended components in seawater (Harding, 1986). The availability of organochlorines depends on the quality, quantity and size of living and inert particles, since organochlorines are very hydrophobic and absorb readily to other surfaces (Haque and

Table 5
Residual levels of chlorinated hydrocarbons (ng/g) in phytoplankton and microplankton.

Year, region	Mesh size (µm)	N	∑DDT			∑PCB			Dieldrin			Source
			Wet	Dry	Lipid	Wet	Dry	Lipid	Wet	Dry	Lipid	
1965-1966, Off Northumberland, UK, coastal waters		1	30						20			Robinson et al., 1967
1971-1972, Off Stockholm, Sweden, estuarine and coastal waters	100	25						300-3500				Jensen et al. 1972
1972 , Gulf of St. Lawrence, Canada, coastal waters	73	9				90-3050						Ware & Addison, 1973
1972 North Atlantic	?		4									Williams & Holden 1973
1972 Open part of Atlantic	?		0.2-0.5									Harvey et al., 1973
1972-1973, Turku archipelago Baltic Sea, Finland coastal water	150	134				40-750		4000-77000				Linko et al., 1974
1973 North Sea, off Holland coastal water	50	2	0.23		540	3.5		8.4	0.3		690	Ten Berge & Hillebrand, 1974
1974 Turku archipelago, Baltic Sea, Finland coastal water	150	67				30-500		26000-340000				Linko et al., 1979
1975 Same	150	47				40-720	400-60000					Linko et al., 1979
1976 Gulf of St. Lawrence NG, Canada, coastal water	66-125	9				1.0-6.4 0.6-7.0	6.1-74.4 5.8-60.8	866-2237 104-2899				Harding et al., 1978
1977 Same	25-66 66-125 125-250	4 15 15				0.1-0.9 0.2-3.3 0.2-2.7	1.4-4.7 1.5-31.6 1.9-19.3	173-232 152-440 78-469				Harding, 1986
1976 Bothnia Gulf Baltic Sea,	?					12-170						Kihlström & Berglund, 1989
Finland	?	20					0.675	118000				Delbeke & Joiris, 1988
North Sea 1986-1988 Off Axel, Heiberg and Ellef Ringnes Island, Canada, coastal waters	25-125	4	12-46 (ppDDT) <2.7.16 (ppDDE) <1.9 (opDDE)				12-14			<06-13		Hargrave et al., 1992

Schmedding, 1976, Picer et al.,1977). Several studies have shown that particulate matter whith high organic content accumulates and retains greater concentrations of organochlorines as compared to inorganic particles (e.g Picer et al., 1977; Nau-Ritter et al., 1982).

Chlorinated hydrocarbons are transported to deep water on sinking particles (Knap et al., 1986) and through zooplankton vertical migration and feeding (Harding, 1986). Zooplankton produce large fecal pellets, rich in contaminants, that sink rapidly toward the bottom (Burns et al., 1985). Elder and Fowler (1977) have found that freshly released euphausiid fecal pellets, collected from natural populations, contain relatively high concentrations of PCBs. They proposed that such biogenic particles contribute significanty to the vertical transport of PCBs in the ocean. Concentrations of PCBs in euphausiid bodies, molts, and fecal pellets, as well as in the microplankton upon which they feed, are given in the Table 6 . Fecal pellets contained the highest concentrations of PCBs in all samples examined. The PCB concentration in feces (wet weight) were 3.5 to 21 times higher than those in the food organisms which were egested as feces. Similar concentrating processes have been found to occur for several trace elements and radionuclides from euphausiids (Small et al.,1973).

Several factors seem to have an influence on the accumulation process. The first is the chemical nature of the compounds: the bioaccumulation factor increases with the chlorine content of the PCB isomers. Furthermore, the level of environmental pollution and biological processes, such as feeding, egg laying, overwintering and death influence the accumulation process (Harding et al., 1981). The size of an organism has been recognized to affect the dynamics of organochlorine uptake and release. Cox (1971b) found that small *Euphausia pacifica* of 2 to 3 mg dry weight, initially accumulated about 35 ng/g DDT over a 2 h period as compared to 9 ng/g DDT for 10 mg specimens, in static uptake experiments.

Table 6
Polyclorinated biphenyls in euphausiids (Meganyctiphanes norvegica), and microplankton which serve as euphausiids` food (from Elder and Fowler, 1977)

Sample	Ratio of wet weight to dry weight	∑PCB µg/kg (dry weight)
	November 1974	
Whole animal	4.7	620
Molts	4.6	1,400
Fecal pellets	4.4	16,000
Microplankton*	10.7	4,500
	January 1975	
Whole animal **		260/290
Molts		170
Fecal pellets		4,800
	March 1975	
Whole animal		38
Molts		Not detectable
Fecal pellets**		11,000/38,000
Mikroplankton*		1,800

* Mainly copepods, phytoplankton and detritus.
** Two separate samples.

Sea water temperature affects the metabolism and activity of an organism and should also modify the exchange rates of organochlorines between plankton and the environment. Cox (1971 a, b) found that *Euphausia pacifica* accumulated slightly more DDT from seawater at 15°C than at 5°C, over a 2 h exposure time. Reduced salinity also increases the uptake of organochlorines from water by fish (Murphy, 1970). This would be expected to apply to estuarine copepods as well.

Organochlorine levels in bulk plankton are presented in Tables 7 and 8. According to Orlova (1987) the level of chlorinated hydrocarbons in zooplankton is not a function of the body mass in zooplankton species. However, some species, (e.g. *Themisto abyssorum, Calanus finmarchicus*), accumulated contaminants to higher levels than did others (Table 8). Hargrave et al. (1992) reported that cconcentrations of HCH, chlordane isomers and other cyclodienes in large zooplankton (mainly adult copepods) and ctenophores from the Canadian Arctic, were two four times higher in August than in May.

Bidleman et al. (1989) have obtained data on the level of toxaphenes in zooplankton samples from the Canadian Arctic. They also investigated role of zooplankton in transport of toxaphenes through water masses. Toxaphenes in plankton from the Norwegian and Barents Seas have not been investigated, although Oehme and Stray (1982) showed mass spectral evidence of PCCs in one air sample from Spitsbergen. Use of PCCs has been reported from the former Czechoslovakia, Poland and Hungary (FAO,1986) and by the former Soviet Union (Izmerov, 1983). Since transport of combustion related pollutants to the Arctic is predominantly from Eurasia, the presence of PCCs and other chlorinated hydrocarbons in plankton samples from the Norwegian and Barents Sea calls for further investigations.

Levels of heavy metals in plankton of northern seas has been less investigated than levels of chlorinated hydrocarbons. Numerous investigators have reported on metal concentrations in zooplankton collected from lakes in Finland and Canada (Särkkä et al., 1978; Jackson, 1978; Franzin, 1984; Yan et al., 1989 etc.). In addition to the work of Martin and Knawer (1973), some investigations have been carried out in the Norwegian Sea (Mukhina et al., 1991). Heavy metal contamination levels in plankton from the Barents sea are still unknown. Levels of heavy metals in mesopelagic crustaceans from the NE Atlantic have been reported by White and Rainbow (1987), Rainbow (1989) and Ridout et al. (1989) and from the Canadian Arctic by MacDonald and Sprague (1988).

Table 7
Residues levels of chlorinated hydrocarbons (ng/g) in marine zooplankton.

Year, region	Mesh size (µm)	N	∑DDT			∑PCB			Dieldrin			Source
			Wet	Dry	Lipid	Wet	Dry	Lipid	Wet	Dry	Lipid	
1969 Firth of Clyde, UK, estuarine waters	Bulk plankton >300	1				30						Holden, 1970
1969 NW Atlantic Ocean, oceanic waters	Bulk plankton >239	14		2.5-100	42-7000		71-3000	320-260000				Risebrough et. al. 1972
1970-1971 N.Atlantic Ocean, oceanic waters	Bulk plankton >333	4	<0.01-9.5		120-1250	4-450		925-19300				Grice et al., 1972
1970 N. Atlantic Ocean	Bulk plankton >239	2	<0.01			300-450						Risebrough et al., 1972
1971-1972 Firth of Clyde,UK, coastal waters	Bulk plankton >300	12	21-107		30-460	80-2200		1100-17100	20-230		430-2880	Williams and Holden, 1973
1971-1972 North Channel,UK, shelf waters	Bulk plankton >300	4	<4-50		<90-750	10-920		200-2500	2-15		60-260	Williams and Holden, 1973
1971-1972 N.Atlantic, UK, oceanic waters	Bulk plankton >300	10	2-16		20-8000	10-120		100-5500	1-2		20-1000	Williams and Holden, 1973
1972 Gulf of St.Lawrence coastal waters	Bulk plankton >239	9				tr.-1860						Ware and Addison, 1973
1973 North Sea, off Holland coastal waters	Bulk plankton >300	2	1.3		560	20		10300	1.5		650	Ten Berge and Hillebrand, 1974
1973-1974 Gulf of St. Lawrence, Canada, coastal waters	Bulk plankton >300		270									Sameoto et al., 1975
1976 Gulf of St. Lawrence, NS, Canada, coastal waters	Bulk plankton >230	40				0.2-9.0	2.6-39	40-732				Harding et al., 1978
1977 Gulf of St. Lawrence, NS, Canada, coastal waters	Bulk plankton >250	60				0.1-21	0.5-147	101-820				Harding, 1986
1980 Kattegat	Bulk plankton >250	1	4.3									Savinova, 1982
1980 Skagerrak	Bulk plankton >250	1-9										Savinova, 1982.

continued

Table 7, continued

Year, region	Mesh size (µm)	N	∑DDT			∑PCB			Dieldrin			Source
			Wet	Dry	Lipid	Wet	Dry	Lipid	Wet	Dry	Lipid	
1980 North Sea	Bulk plankton >250		2-9									Savinova, 1982
1980 Norwegian Sea	Bulk plankton >250		2-9									Savinova, 1982
1985 North Sea	>200	20					700	7000				Delbeke and Joiris, 1988
1986 Axel Heiberg Island, Canadian Arctic	zoo-plankton			1.9	3.9		6.6	13				Bidleman et al., 1989
1987 Axel Heiberg Island, Canadian Arctic	zoo-plankton			12	49		27	110				Bidleman et al., 1989
1986-1988 Off Axel Heiberg and Ellef Ringnes Island, Canadian, coastal waters	Bulk plankton 125-509	9		2.7-24 (ppDDT) 1.6-30 (ppDDE) <0.7-5.3 (opDDE)			14-66			2.5-7.5		Hargrave et al., 1992
1986-1988 Off Axel Heiberg and Ellef Ringnes Island, Canada, coastal waters	Bulk plankton >509	17		1.9-50 (ppDDT) <0.5-5.3 (ppDDE) <0.4-0.8 (opDDE)			6-34			1.9-6.8		Hargrave et al., 1992

Table 8
Residue levels of chlorinated hydrocarbons (ng/g dry weight) in different species of Atlantic zooplankton (Orlova, 1987)

Sampling area N	W	Zooplankton species composition	∑DDT	∑PCB
36° 00'	17° 30'	*Euchirella sp., Nanocalanus minor*	134.5	1253.5
36° 00'	40° 30'	*Euphausiacea, Hyperiidae, Calanoida varia*	55.7	354.0
40° 32'	53° 40'	*Euchirella varia, Ostrapoda, Clio sp., Calanoida varia*	47.0	253.0
39° 26'	51° 00'	*Chaetognatha, Nanocalanus minor, Euchaeta marine*	33.0	209.7
42° 19'	53° 37'	*Themisto abyssorum (adult), Calanus finmarchicus*	5.1	88.2
37° 02'	50° 30'	*Pareuchaeta glacialis, Undeuchaeta sp., Euchirella rostrata*	11.2	170.0
41° 52'	49° 30'	*Themisto abyssorum (adult), Calanus finmarchicus*	17.4	143.0
42° 34'	51° 54'	*Themisto abyssorum, Calanus finmarchicus*	12.5	79.3
42° 34'	51° 54'	*Themisto abyssorum (adult)*	14.4	278.5
52° 47'	35° 32'	*Calanus finmarchicus, Thysanoessa sp. (adult), Sagitta elegans*	30.0	125.6
52° 47'	35° 32'	*Cliona limacina (adult), Themisto abyssorum, Calanus finmarchicus*	8.9	67.3
52° 47'	35° 32'	*Themisto abyssorum (adult)*	18.9	426.3
37° 20'	09° 26'	*Meganytiphanes norvegica (adult), Amphipoda*	24.1	168.5

4.3. Benthos

Because of the tendency of chlorinated hydrocarbons and heavy metals to concentrate in suspended (sedimentary) material and to be removed from water by sedimentation, significant concentrations of these substances accumulate in bottom sediment. As a consequence, elevated levels of toxins are to be expected in benthic organisms. This has been confirmed by studies carried out to determine the contamination levels of benthic organisms in the North and Northeast Atlantic (Portmann, 1979; Murray, 1979; 1981; 1982; Bidleman et al., 1989; Hargrave et al., 1992) and in the North and Norwegian Seas (Eisler, 1981; Julshamn, 1981a,b,c,d; De Kock, 1986; Hummel et al., 1990).

Residue levels of chlorinated hydrocarbons and the concentrations of some trace metals in polychaetes from the North Sea are presented in Tables 9 and 10. In a field study, to determine whether or not Cd, Cu, Pb and Zn accumulated in fish, it was found that the concentration of the metal was related to the type of invertebrates ingested. Fish that fed upon polychaetes had the highest concentration of metals (Metayer et al., 1980). Similar dependence has been shown for chlorinated hydrocarbons in fish by Schaefer et al. (1976). The concentration of chlorinated hydrocarbons and heavy metals in polychaetes from the Norwegian and Barents Seas is unknown.

Table 9
Residue levels (µg/kg) of chlorinated hydrocarbons in Polychaeta

Species	Number of pooled animals	PCB	DDE	DDD	DDT	∑DDT	Source
Polynoidae	30	0.10	0.0045	0.0045	0.010	0.020	Schaefer
		2.2	0.094	0.097	0.22	0.42	et al., 1976
Polyphisia	12	0.085	n.d	0.0035	n.d	0.0035	Schaefer
crassa		3.8	n.d	0.16	n.d	0.16	et al., 1976
Aphrodite	6	0.22	0.0034	n.d		0.0078	Schaefer
aculeata		13	0.19	n.d	0.24	0.44	et al., 1976
Arenicola marina			0.8-1.2	3.1-4.5	0.0043		Goerke et al., 1979

Underlined value indicate dry weight, others - lipid weight.

Table 10
Concentrations of trace metals (µg/g dry weight) in Polychaeta from the North Sea (Gibbs et al., 1981)

Species	Cu	Cd	Pb	Zn	Mn	Ag	Fe
Arenicola marina	5-47	0.9-40	8-39	41-400	-	-	-
Melinna palmata	12-3124	-	-	63-258	5-20.7	0.5-5	239-1730

Table 11
Organochlorine residues (ng/g dry weight) in Amphipods from the Canadian Arctic.

Species, year	HCB	PCCs	HCHs	ppDDE	ppDDT	PCBs	Source
Eurythenes grullus, 1986	1-87	2377-9079	1.3-3.6 0.3-208 0.2-49	0.5-4787	1.7-6087	4448-15093	Hargrave et al., 1992
Gammarus wilkitzkii, 1986-1988	2.9	-	73	3.2	53	73	Hargrave et al., 1992
Andaniexis sp. *Anonyx nugax* *Onisimus sp.* *Tmetonyx cicada*	6-40	107-465	11-58 0.1-6.6 0.9-28	34-540	60-320	370-3000	Hargrave et al., 1992
Anonyx sarsi, August, 1986	1.6	12	20	1.2	1.9	6.6	Bidleman et al., 1989
Tmetonyx cicada, June, 1987	2.4	36	6.7	3.2	12	27	Bidleman et al.,1989

Canadian scientist have studied the residue levels of organochlorine pesticides, polychlorinated biphenyls and toxaphene camphenes (PCCs) in amphipods from the Arctic Ocean (Bidleman et al., 1989; Hargrave et al., 1992). HCHs, toxaphene and PCC products were the most abundant chlorinated hydrocarbons in surface (10 m) and deep (270 m) water samples. Levels of PCCs in amphipods were comparable to those of PCBs (Table 11).

As with other organic compounds, the heavier, less water soluble PCCs should preferentially be accumulated in lipids by aquatic organisms. They should also be more susceptible to vertical transport on sinking particles. PCCs may also reach the bottom and be incorporated into the benthic food web through other biological pathways, as evident from the high concentration found in benthic amphipods (Table 11).

Hargrave et al. (1992) reported that large benthic lysianassid amphipods (*Tmetonyx cicada, Anonyx nugax* and *Eurythenes gryllus*) accumulated chlorinated hydrocarbons on dry and lipid weight basis in higher concentrations than did small species (*Onisimus* spp. and *Andaniexis* spp.) or the under-ice gammaridean amphipods *Gammarus wilkitzkii*. Higher ratios of DDT:DDE in zooplankton (2 to 6) than in amphipods (1 to 2) reflect the metabolism of DDT to the more stable DDE isomers in amphipods. The under-ice amphipod *Gammarus wilkitzkii*, which feed on zooplankton as well as ice algae, had two to ten times higher concentrations of the main classes of chlorinated hydrocarbons, than what was found in the under-ice particulates and plankton (Table 12). The scavenging amphipods, trapped just off the bottom over the continental shelf, contained 2 to 60 times higher ∑DDT than plankton. PCBs and PCCs levels found in plankton on a lipid basis are presented in Table 12. HCH and HCB were least concentrated in the shelf amphipods relative to plankton, apparently due to differences in lipid content and composition of these animals. Further studies are required to investigate the dynamics of bioaccumulation of chlorinated hydrocarbons in these organisms, because of their necrophagous diet and their role as scavengers of top predators in the arctic marine food web (see Hargrave et al., 1992 for further discussion)

According to Goldberg (1972), filter feeders such as molluscs, which can rapidly alter concentrations of contaminants in their body in relation to changes in contamination levels in the environment, should be regarded as true indicators of pollution. Bryan (1983) has studied the use of marine snails (*Littorina littirea, L. obstusata*) as indicators for heavy metal pollution. Among the basic criteria imposed on indicator organisms are, that they are massive, cosmopolitan, sedentariness, and secondly, the capacity to accumulate and concentrate pollutants while maintaining vitality and genetic stability at relatively high environmental stress. It is also essential that indicator organisms are easily accessible for collection and they have a long life span so that observations can encompass a number of years.

Molluscs of the class Bivalvia generally satisfy the requirements listed above. They possess one of the highest coefficients of accumulation due to their ability to filter huge quantities of water and concentrate substances dissolved in it. Many researchers feel that species of the genus *Mytilus* are especially promising as indicator organisms (Butler, 1969; Goldberg, 1976; Burdin et al., 1979; Knutzen, 1982).

The International Council for the Exploration of the Sea (ICES) has carried out studies in the North Atlantic, North Sea and Baltic Sea. In these studies the mussel *Mytilus edulis* has been used to monitor levels of mercury and chlorinated hydrocarbons. In the ICES area the average mercury concentration was 0.05 mg/kg wet weight with a range from 0.002 to 0.17 mg/kg w.w. Cadmium concentrations in mussels were generally in the range < 0.06 to 0.94 mg/kg w.w. Within this range, the highest concentrations were found in the Humber area on the east coast of England, along the French coast, and in the Elbe Estuary; mussels from Sørfjord and Hardangerfjord in Norway contained very high concentrations of cadmium (from 2.0 to 7.8 mg/kg w.w.). Temporal trend analyses for cadmium concentrations in mussels from the ICES area showed: 1) an increasing trend in the outer Seine Estuary, and 2) decreasing trends in the Kattegat, the Oslofjord, the Langesunfjord. Reported zinc concentrations were mainly in the range 9.2 to 49 mg/kg w.w., but in Norway (Sørfjord and Oslofjord), the Southern Bight, and the Western Scheldt Estuary, the concentrations ranged from 52 to 89 mg/kg w.w., and at one station a concentration of 127 mg/kg was recorded (ICES, 1988).

There are numerous reports on molluscs in which no quality control of the analytical procedures has been performed (Eisler, 1981). Therefore, the accuracy of data cannot be evaluated. Data reported by Davies et al (1979) show that in clean waters 50 to 100% of the total mercury in *Mytilus edulis* was methylmercury (MeHg), whereas in polluted waters only 20% was MeHg. Results from an international intercalibration study for MeHg in *M.edulis* showed a ratio of 0.31 between methylmercury and total mercury with a total mercury content of 0.18 mg/kg dry weight.

A programme has been going on in the USA since 1976 to monitor the levels of pollution in sea water by analyzing tissues of oysters and mussels from 107 standard stations along the eastern and western shores of the Gulf of Mexico (Bayne, 1979). The NOAA National Status and Trends Mussel Watch Programme has examined chlorinated hydrocarbon residues in bivalves around the US during 1984-1986. Although too early to assess temporal trends, this study has been crucial in identifying persistent hot spots for $\sum$DDT at Hudson Bay, New York, San Pedro Harbor and Palos Verdes, (California), Buzzards Bay, (Massachusetts) Hudson Bay and New York have been identified as source areas of PCBs. (NOAA, 1987).

Table 12
Level of residual chlorinated hydrocarbons (µg/kg wet weight) in bivalve molluscs

Locality,Year	PCBs	∑DDT	Source
	Mytilus edulis		
Baltic Sea 1967-1968	30.0-84.0	20.0-40.0	Jensen et al., 1969
North Sea, coast of the Netherlands	-	21.0	Holden, 1970
North Sea, coast of the Netherlands, 1973	273.0	9.0	Ten Berge and Hillebrand, 1974
North Sea, coast of Great Britain	-	43.0	Holden, 1970
North Sea, coast of Great Britain 1970-1973	65.0	13.0	Portmann, 1979
North Sea, coast of Great Britain, 1974	160.0	20.0	Murray, 1979
North Sea, coast of Great Britain, 1975	80.0	12.0	Murray, 1979
North Sea, coast of Great Britain, 1975	-	4.0-237.0	Butler, 1973
North Sea, coast of Northern Europe	-	20.0	Holden, 1970
North Sea 1965-1968	600.0-1100.0	110.0-250.0	Koeman and Van Genderen, 1972
North Sea, coast of Sweden, 1972	95.0	13.0	ICES, 1974
North Sea, coast of Germany, 1972	90.0	25.0	ICES, 1974
North Sea, coast of Norway, 1972	30.0	25.0	ICES, 1974
Norwegian Sea, (59 10'N:11 00'E)	59.0	5.0 *	Bjerk and Brevik, 1980
Coast of Canada	-	69.0	Holden, 1970
Coast of Canada	-	10.0-170.0	Sprague and Duffy, 1971
Coast of Canada 1970	140.0	20.0	Zitko, 1971
North Sea	(106-362)**	(15-143)**	Franklin, 1987
Irish Sea	(50-1070)**	(92-590)**	Franklin, 1987
	Modiolus modiolus		
English Channel	(380-480)**	(35-112)**	Franklin, 1987
Barents Sea, coast of Eastern Murman	Not detected	Not detected	Savinova et al., 1981
Norwegian Sea	10.0	-	Kveseth et al., 1979

* DDE only.
** Seveval samples

Table 13
Levels of residual hydrocarbons (mg/kg wet weight) in bivalves

Scallop species	Reported area	Type of sample	∑PCB	∑DDT	Source
Placopecten	Coastal waters of Canada	Muscle	-	0.03 (0.01-0.09)	Sprague and Duffy, 1971
P. magellanus	Coastal waters of Canada	Muscle	0.018 (0.005-0.051)	0.03 (0-0.01)	Sims et al., 1977
Chlamys opercularis	Coastal waters of England 1974	Muscle	0.05	0.003	
		Gonad	0.01		
	1975	Muscle	0.01		
Pecten maximus	1975	Muscle	0.01	0.003	Ernst et al., 1976
		Gonad	0.02	0.003	
Chlamys islandica	Barents Sea, 1979	Muscle	-	Not detected	Savinova et al., 1981
		Gonad	-	0.012	

In Great Britain, a programme of pollution monitoring has been performed since 1970 on mussels and other benthic organisms: *Modiolus modiolus, Chlamys opercularis, Pecten maximus, Buccinum undatum, Cancer pagurus, Crangon crangon, Pandalus montague, Pandalus borealis, Eupagurus bernhardus* etc. (Portmann, 1979; Murray,1979; 1981; 1982). Monitoring of fish and shellfish around the British Isles has shown that, while the concentrations of organochlorine contaminants still remain relatively high in some areas along the North Sea coast (e.g. Liverpool Bay), concentrations are generally lower than those previously reported in the seventies (Franklin, 1987).

Results from different studies have shown that it is possible to use gastropod molluscs, especially the genus *Buccinum*, as indicator organisms. No residual chlorinated hydrocarbons were detected in *Buccinum undatum* in coastal waters of the Eastern Murmansk region. However, the muscles in species from the western part of the Barents Sea coast, a more polluted region, had average levels of 0.0447 mg/kg of wet weight (Savinova et al., 1981). This indicates that molluscs can also be good indicator organisms to monitor pollution in the Barents Sea.

The study of heavy metal levels in mussels from the Barents Sea has been carried out infrequently (Burdin et al., 1979). Residual levels of chlorinated hydrocarbons in mussels from the Barents Sea are unknown.

Extensive investigations of the levels of different pollutants in molluscs have been carried out in the Norwegian Sea. Julshamn (1981 a,b,c,d) and Skei et al. (1991) reported distribution of heavy metals in tissues of oyster, common mussel and horse mussel taken from different fjords in Western Norway. Knutzen (1986) reported concentrations of heavy metals in mussels and other species of molluscs. Bjerk and Brevik (1980) and Knutzen and Kirkerud (1984) studied residue levels of chlorinated hydrocarbons in mussels from the Norwegian Sea.

In the Nordic Bioindicator Project (NBP), the blue mussel and the brown alga were proposed as marine bioindicators for radioactive effluents in lower latitudes (Aarkrog, 1985). The result of this project (i.e., the dispersion and uptake models, transfer factors and data base) may be applied with few modifications to the Arctic marine environment (Jaworowski, 1989). It has been shown that *Mytilus edulis,* which does not occur in the High Arctic, might be substituted by *Astarte borealis, Macoma moesta* or *Mya truncata* as a indicators. Residual levels of chlorinated hydrocarbons in bivalves from northern seas are presented in Tables 12 and 13.

In experimental studies on accumulation of chlorinated hydrocarbons in organs and tissues of crabs, it was established that the maximum quantities of toxicants were in the hepatopancreas, brain, thoracic ganglia, hemolymph, gills, and muscles. An almost complete absence of chlorinated hydrocarbons accumulation was noted in the heart and blood. It was concluded that chlorinated hydrocarbons are absorbed by the gills and pass into the hepatopancreas along with the hemolymph (toxicants accumulate in the hemolymph within 5 minutes, and in the hepatopancreas within 15 min after exposure) (Sheridan, 1975).

The contents of persistent, lipophilic chemicals in the abdomen and eggs of the hermit crab *Pagurus* spp. from the North Sea have been subject of extensive research (Knickmeyer and Steinhart, 1988 a,b). According to them, the seasonal bioaccumulation of PCBs is influenced by the uptake of food from the spring plankton bloom and the resuspension of sediments during winter. The patterns of individual PCB congeners depend on the weigth of the abdomen and the origin of the samples. In the offshore samples, the findings indicate a relationship between the weight of the abdomen (i.e. roughly the age) and the bioconcentration of higher chlorinated biphenyls. The results from coastal areas demonstrate that food uptake determines the stable concentration of higher chlorinated congeners in the hermit crab. ∑DDT and lindane are more concentrated in eggs than in abdomen.

It is known that the eggs of aquatic animals contain considerable amounts of phospholipids (Herring and Morris, 1975; Tocher and Sargent, 1984). Other important polar lipids isolated from crustaceans are carotenoids, glycolipids, and lipoproteins (Cheesman and Prebble, 1966; Wallace et al., 1967). Knickmeyer and Steinhart (1988 a, b) proposed that the selective accumulation of ∑DDT and lindane in eggs is a function of the physiochemical equilibrium between water and different types of lipids and pollutants.

Brevik (1978) determinated levels of PCBs, HCB and ∑DDT in crab *(Cancer pagurus)* from the Kristiansand fjord (Norway) in 1975. The difference found between the respective levels of ∑DDT and ∑PCBs in crab caught at the harbour and Flekkerøy, was significant for PCBs, however not significant for ∑DDT. This indicates local contamination of PCBs in the Kristiansand harbour. The level of HCB in the hepatopancreas of crab from the harbour of Kristiansand was about five times higher than the level found in crab from Flekkerøy, indicating a local contamination of HCB near the main harbour.

Isomer-specific determination of polychlorinated dibenzofurans (PCDF) and dibenzodioxins (PCDD) was done for crabs and shrimp from a fjord area contaminated by waste water from a factory (Oehme et al., 1989b). PCDD/PCDF isomers, with chlorine in the position 2,3,7 and 8, belong to the most toxic chlorocompounds known today. It was shown that the 2,3,7,8-TCDD equivalent level decreased by about one order of magnitude from the Frierfjord to the outer region of the Grenlandsfjord. In crab hepatopancreas about the same PCDF concentrations as reported for sites close to a pulp mill (Rappe et al., 1987) were found for the stations inside the fjord system.

Comparable levels of 2,3,7,8TCDF have also been reported for crab samples collected close to wood pulp mills in Canada (Norstrom et al., 1988).

High levels of contaminants were found in shrimps when one considers the low fat content of shrimp and the distance from the source. In comparison, concentrations of only 0.51 pg 2,3,7,8-TCDD equivalents per g were found in shrimp caught near a Nickel refinery, which is also a point source for PCDF/PCDD (Oehme et al., 1989a).

In a collaborative study carried out in the North Sea, the brown shrimp has been monitored for several years. The mercury level ranged from 0.03 to 0.40 mg total Hg/kg (w.w.) (ICES, 1974; 1977). A typical average level would be 0.11 mg Hg/kg (w.w.). Even higher concentrations have been reported for arsenic in crustaceans, with 1 to 50 mg/kg (w.w.) being a representative range for normal values. Data on total arsenic content are available for many species, including shrimp, prawns and lobster (Egaas and Brækkan, 1977; Eisler, 1981). The highest cadmium concentrations in organs of crustaceans were found in the digestive gland, hepatopancreas and kidney and. the lowest were in the muscles (Eisler, 1981; Koli and Whitmore, 1986).

There has been less research on accumulation levels of chlorinated hydrocarbons in echinoderms. On study on the DDT level in gonads of sea urchins of the genera *Eschinus* and *Strongilocentrotus* found 0.05 and 0.005 mg/kg, respectively (Robinson et al., 1967; Risebrough et al., 1967). No residual chlorinated hydrocarbons were detected in the gonads of the sea urchin *Strongelocentrotus droebachiensis* from coastal areas of the Eastern Murman (Barents Sea) in 1979 (Savinova et al., 1981).
A few investigations have been carried out on starfish of the genera *Pisaster, Patria* and *Acanthaster* (Risebrough et al., 1967; McClosky and Deubert, 1973). The DDT levels varied between 10 and 78 mg/kg. Kveseth et al. (1979) determined DDT in *Asteria rubens* from southwestern coast of Norway in 1972 and 1974 in which the concentrations decreased from 45 to 10 ppm during this period. Bjerk and Brevik (1980) reported residue levels for sea star (Asteroidea) from Frierfjord, Finnmark: pentachlorobenzene 0.78, hexachlorobenzene 1.03, octachlorostyrene 0.21 ppm fat weight. Melnikov and Vlasov (pers.com) determined in Urasterias sp., collected in the Barents Sea (near the southern tip of Novaya Zemlya) ∑DDT - 49 , HCB - 1 , ∑DDT - 6 and ∑PCB - <1 ng/g w.w.

4.4 Fish

A large number of studies have been devoted to determining levels of chlorinated hydrocarbons in marine fish. Experiments have shown that chlorinated hydrocarbons may enter the fish both directly from water, by absorption through the body surface and from water through the gills (Murphy, 1971; Hamelink et al., 1971), but mainly via food (Macek et al., 1970; Macek and Korn, 1970).

The accumulation of chlorinated hydrocarbons by fish depends on the sex, season and feeding conditions. The relationship between the level of accumulated DDT and PCBs and the age of marine fish has been determined in several studies (e.g. Cox, 1970; Stenersen and Kvalvag, 1972; Bjerk, 1973; Pertillä et al., 1982; a, b).

There are few studies on the relationship between the concentration of residual chlorinated hydrocarbons in the organs and tissues of fish and the concentration of lipids in them. In the majority of studies authors have found reliable correlations between DDTs, PCBs and lipid concentration and lipid composition (Portmann, 1975; Schaefer et al., 1976; Schneider, 1982; Pertillä et al., 1982 a, b). In some studies no correlation has been found (Henderson et al., 1971; Addison et al., 1972). Earnst and Benville (1971) discovered both a positive and negative correlation between chlorinated hydrocarbon concentration and lipid level in several species of marine fish. This was probably related to the uneven distribution and levels of lipids in fish in relation to sex, age, stage of sexual development and time of the year.

Absorption of DDT directly from water by fish increases with temperature, oxygen consumption (Murphy and Murphy, 1971; Reinert et al., 1974), and salinity of the water (Murphy, 1970).

Stenersen et al. (1977) have established a reliable correlation between concentrations of DDT and PCBs in fish from the North Sea caught near the southwestern coast of Norway. In the liver of plaice from the Barents Sea the coefficient of correlation between the level of DDTs and PCBs turned out to be low, although

a positive correlation was found between the level of lindane and PCBs (r=0.60). This may be tied to certain characteristics of a lindane manufacturing in Russia were PCBs were used as filler to reduce evaporation and improve insecticidal properties (Savinova, 1991).

Contamination levels in marine fish from the ICES area are well investigated. Fish monitoring, which has taken place for nearly two decades in the Baltic Sea region, has demonstrated a significant decrease in the concentrations of DDT residues during the 1970s and 1980s (Andersson et al., 1988). Other pesticides persist, with present lindane concentrations in these fish varying from 0.010 to 0.017 µg/g lipid wt and HCB from 0.013 to 0.019 µg/g . Furthemore, it has been reported that polychlorinated camphene (PCC) concentrations in fish (0.86 µg/g lipid weight) from the west coast of Sweden, the Baltic, and certain areas of the North Atlantic during 1979, were as high or higher than the total DDT concentrations (Andersson et al., 1988). Some Baltic studies have detected a slight increase in DDT in certain areas after 1983, although the origin of these episodic inputs has yet to be explained.

The major chlorinated hydrocarbon residue in Canadian Arctic marine fish is toxaphene. In Arctic cod in three eastern Arctic locations, a major prey of ringed seals and seabirds, toxaphene levels were three to ten times higher than DDTs or PCBs residues (Muir et al., 1987). Levels of toxaphene detected in Arctic cod, herring and char in the Canadian Arctic are about two times lower than the level found in Atlantic cod in the Gulf of St. Lawrence (Musial and Uthe, 1983) and similar to levels in the Antarctic (Zell and Ballschmiter, 1980).

Comparison of PCBs levels in the muscle of Arctic cod and levels in fish from similar trophic levels in other areas indicates that PCB contamination in Arctic marine fish is approximately ten to twenty times lower than the coastal areas of Atlantic Canada, Baltic Sea and the northwest Pacific ocean. However, the Arctic fish PCBs level is thirty times higher than in the Antarctic (Norstrom and Muir, 1988). Based on limited data available levels of a-HCH in Arctic fish are as high or higher than at midlatitudes (Norstrom and Muir, 1988).

Vuorinen et al. (1989) compared concentrations of organochlorines in cod from the Baltic Sea (Gulf of Finland) and the Western Tana Fjord (Barents Sea) (Table 14). The low ratios of the more volatile compounds (chlordane, toxaphene, and HCB) support findings that these compounds are distributed globally through long-range transport and that local pollution is the main contributor to PCB and DDT residue accumulation in fish.

Table 14
Comparison of the levels of chlorinated hydrocarbons (ng/g) in cod, (Gadus morhua), from the Baltic Sea (Gulf of Finland) and Arctic (Western Tana Fjord) (Vuorinen et al. 1989).

Compound	Baltic	Arctic	Baltic/Arctic
∑PCB	5600	570	9.8
∑DDT	1480	147	10.1
Chlordane	168	128	1.31
HCB	61	33	1.81
Toxaphene	640	540	1.19

Although several PCDDs and PCDFs have been detected in the Baltic samples, the 2,3,7,8-TCDF was the only isomer detected in Arctic samples. The level in Arctic samples was 6.3 times lower than in the Baltic. The levels of coplanar PCBs in Baltic cod were one to three orders of magnitude higher than PCDDs and PCDFs. Levels of coplanar PCBs in Arctic cod were about three times lower than in Baltic samples (Vuorinen et al., 1989).

In the south-western part of Norway, during a period of about 25 years, the principal use of DDT was the spraying on commercial fruit orchards 1 to 3 tons per year. The use of DDT in these isolated fruit growing regions is the likely cause of local contamination by DDT and its metabolites in marine animals in the area. (Stenersen and Kvalvag, 1972; Bjerk, 1973; Kveseth and Bjerk, 1976; Stenersen et al., 1977; Kveseth et al., 1977; Brevik, 1978; Brevik et al., 1978; Kveseth et al., 1979).

The time-trend in DDT contamination of the marine environment in this area has been studied for 10 years following the DDT ban (Skaare et al., 1985). The decrease in DDT concentrations illustrates the effectiveness of the 1970 ban on its use in Norway.

Table 15
Levels of residual chlorinated hydrocarbons (mg/kg wet weight) in cod (Gadus morhua) from the Baltic Sea

Sampling area	Type of samples	∑DDT	∑PCB	Source
Coastal waters of Sweden	Muscle	0.063	0.033	Jensen et al., 1969
Coastal waters of Sweden	Liver	4.4-18.0	1.4-9.5	Jensen et al., 1972
Coastal waters of Denmark	Liver	11.3-22.0	2.4-4.9	Westoo & Noren, 1971
Coastal waters of Germany	Liver	7.5-31.7	-	Huschenbeth, 1973
Coastal waters of Germany	Liver	Up to 60.0	-	Priebe, 1978
Gulf of Finland and Gulf of Bothnia	Muscle	0.43-1.4	1.7-5.0	Tervo et al., 1980
Coastal waters of Poland	Liver	7.2	8.8	Falandysz et al., 1980
Northern part of the sea	Liver	0.3-1.0	2.3—5.3	Widestrom et al., 1981
Open part of the sea	Liver	0.29	0.005-0.2	Kullenberg, 1981
Western part of the sea	Liver	0.59	2.87	Schneider, 1982

Table 16

Levels of residual chlorinated hydrocarbons (mg/kg wet weight) in cod (Gadus morhua) from the North Sea.

Sampling area, year	Type of samples	∑DDT	∑PCB	Source
Coastal waters of England	Muscle	0.012	-	Robinson et al., 1967
Coastal waters of Sweden	Muscle	0.005	0.019	Jensen et al.,1969
Coastal waters of Sweden	Muscle	0.005	0.020	Jensen et al.,1972
Coastal waters of Germany	Muscle	30.0	-	Huschenbeth,1973
Open ocean	Muscle	0.055	-	Ten Berge & Hillebrand, 1974
Coastal waters of Germany	Muscle	2.1-4.9	12.0-29.0	Schaefer et al., 1976
	Liver	0.96-5.2	8.9-22.0	
	Whole	4.4	24.0	
Open ocean	Fat	0.42	-	Bruggemann et al., 1976
Central ocean	Muscle	0.006-0.01	0.03-0.07	Ernst et al., 1976
1972	Liver	0.03-1.8	0.3-8.1	
Coastal waters of England	Muscle	0.009	--	ICES, 1977
Southern ocean	Liver	-	39.0	Kerkhoff et al., 1977
1974-1975				
1976-1978	Liver	-	48.0	
Northern ocean	Liver	-	5.0-8.0	Kerkhoff et al., 1977
Open ocean	Liver	-	5.1	Freemann & Uthe, 1979
Coastal waters of England and Wales				Portmann, 1979
winter, 1970	Muscle	0.001-0.007	0.007-0.31	
	Liver	0.025-0.530	0.400-12.0	
summer, 1970	Muscle	0.003-0.008	0.51-0.003	
	Liver	0.46	4.4	
winter, 1971	Muscle	0.006-0.014	0.009	
	Liver	0.25-1.7	0.5-8.3	
summer, 1971	Muscle	0.004-0.005	0.012-0.025	
	Liver	0.77-1.3	2.6-12.0	
winter, 1972	Muscle	0.004-0.010	0.02-0.049	
	Liver	0.42-1.7	1.1-11.0	
summer, 1972	Muscle	0.003-0.006	0.015-0.031	
	Liver	0.7-2.7	2.8-18.0	
winter, 1973	Muscle	0.005-0.006	0.020-0.027	
	Liver	0.64-1.4	2.3-4.6	
summer, 1973	Muscle	0.003-0.004	0.015-0.068	
Coastal waters	Muscles	0.055	0.030	Murray, 1979
of England, 1974	Liver	0.87	5.0	
Open ocean	Muscle	0.004	0.01	Murray, 1979
	Liver	0.393	3.0	
Coastal waters of Norway, 1974	Liver	0.65-11.4	0.78-1.5	Kveseth et al., 1979
Coastal waters of England,	Muscle	0.003	0.02	Murray, 1981
1975	Liver	0.61	8.4	
Coastal waters of Belgium	Muscle	-	0.005	Vandamme & Baeteman, 1982

Table 17
Levels of residual chlorinated hydrocarbons (mg/kg wet weight) in cod (Gadus morhua) from the Norwegian Sea.

Sampling area, year	Type of samples	∑DDT	∑PCB	Source
Norwegian Sea, open part	Muscle	0.02	-	ICES, 1977
Norwegian Sea, open part	Liver	10.0	-	Gerlach, 1981
Sognefjord, 1973	Liver	2.0-33.0	-	Stenersen & Kvalvag, 1973
Fjords of western Norway	Liver	12.0-14.0	-	Bjerk, 1973
Sogndalsfjord, 1977	Liver	1.238	0.684	Skaare et al., 1985
Sogndalsfjord, 1977	Liver	1.202	0.454	Skaare et al., 1985
Dalsfjord, 1977	Liver	2.637	0.486	Skaare et al., 1985
Glomma delta, 1988	Liver	-	1.05	Marthinsen et al., 1991

Table 18
Levels of residual chlorinated hydrocarbons (mg/kg wet weight) in cod (Gadus morhua) from the Barents Sea.

Sampling area, year	Type of samples	∑DDT	∑PCB	Source
Open part	Liver	0.31	1.7	ICES, 1977
	Muscle	0.03	-	
Open part, 1975	Liver	0.009	0.03	Murray, 1981
	Muscle	0.003	0.01	
Southeastern part, 1977	Liver	0.22	-	Savinova
	Muscle	0.0005	-	et al., 1981
Bear Island, 1983	Muscle	0.0005	0.002	Savinova,
	Gonads	0.0016	0.003	1991

The highest 1982 mean level of PCBs residues was found in cod liver (454 µg/kg w.w., range 175-1073 µg/kg) which was about 1/3 the 1972 value. The highest 1982 mean level of ∑DDT was found in the cod liver (1237 µg/kg, wet weight, range 191-3845 µg/kg) and was about 1/3 of the 1972 residue level. The decrease of ∑-DDT was less rapid in the liver of cod and haddock (*Melanogrammus aeglifinus*) (halflife 5.0 years, with a 95% confidence limit of 2.3 years) than in the sea scorpion (*Myoxocephalus scorpius*), cat fish (*Anarhichas lupus*), ballan wrasse (*Labrus bergulta*) and lemon sole (*Microstomus kitt*) (halflife 2.6 years, with a 95% confidence limit of 0.8 years). Cod and haddock have a substantially higher fat content in the liver compared to the other species.

A halflife of 35 years for biologically available DDT residues was suggested to apply generally to the Nordic marine environment. The main conclusion was that levels of organochlorines in fish from Norwegian fjords were generally low and were believed to have little toxicological implications (Skaare et al., 1985).

Tables 15-19 summarize data on the contamination levels of total DDT and PCBs in different fish species from northern seas. These results for total DDT related compounds in the tabulated data are probably the most reliable for temporal, geographic and interspecific comparisons because almost all authors have reported the same three components (4,4'DDE, 4,4'DDD and 4,4'DDT). PCBs have been reported as equivalents of the commercial Aroclor products (generally Aroclor 1254) and most recently as total PCB congeners.

Heavy metal contents in fish from the Arctic and sub-Arctic have been investigated, although studies have been done in the North and Baltic Seas (Falconer et al., 1983; Knutzen, 1984; Skei, 1984; Staveland et al., 1993). The Hvaler archipelago (Southern Norway) was monitored from the 80s to study the extent of mercury and arsenic pollution in fish (Knutzen, 1984; Skei, 1984; Monfelt and Lindestrøm, 1989; Staveland et al., 1993). In 1988 Staveland and co-workers (1993) have determined significantly higher mercury and arsenic concentrations in fillets of flounder than in fillets of cod: 0.15 vs. 0.08 mg/kg and 5.2 vs 4.1 mg/kg w.w., respectively (Staveland et al., 1993). Heavy metal contents in cod from the Barents Sea have also been studied recently by Savinov (1994).

Heavy metal contamination levels in cod from different regions are presented in Table 20.

Table 19
Levels of residual chlorinated hydrocarbons (mg/kg wet weight) in abundant species of fish in northern seas.

Sampling area	Type of samples	∑DDT	∑PCB	Source
		Ocean perch Sebastes marinus.		
North Atlantic	Liver	1.3	1.5	Harvey et al., 1974
Georges Bank	Muscle	0.073	0.190	
Denmark Strait	Muscle	0.032	0.360	Harvey et al., 1974
Barents Sea, SW part, 1977	Liver	0.076	-	Savinova et al.,
	Muscle	0.020	-	1981
		Pollack Pollachius virens		
Barents Sea,	Liver	0.295	-	Savinova et al., 1981
SW part, 1977	Muscle	0.017	-	
North Atlantic	Liver	3.0	45.0	Harvey et al., 1974
Georges Bank	Muscle	0.003	0.037	
		Common catfish Anarhichas lupus		
Barents Sea,	Liver	0.123	-	Savinova et al., 1981
SW part, 1977				
Norwegian Sea, Sogndalsfjord, 1977	Liver	0.711	0.423	Skaare et al., 1985
West coast of Norway, Dalsfjord, 1982	Liver	0.047	0.042	Skaare et al., 1985
		Northern catfish Anarhichas denticulatus		
Barents Sea, waters	Liver	0.06	0.053	Savinova, 1991
off Bear Island, 1983	Muscle	0.019	0.021	
	Gonad	0.078	0.080	
		Spotted catfish Anarhichas minor.		
Barents Sea	Liver	0.123	-	Savinova et al., 1981
SW part, 1977	Muscle	Not detected	-	
Barents Sea, water off Rybachij Peninsula, 1983	Muscle	0.028	0.029	Savinova, 1991
		Canadian Plaice Hippoglossoides		
Barents Sea,	Liver		-	Savinova et al., 1981
SW part, 1977	Muscle	0.003	-	
North Sea, central part	Liver Muscle	0.05-0.1	0.4-0.6	Schaefer et al., 1976
Barents Sea, waters off Rybachij Peninsula, 1983	Liver	0.008	0.02	Savinova, 1991
Barents Sea, waters off Bear Island, 1983	Liver	0.058	0.079	Savinova,1991
		Haddock Melanogrammus aeglifinus.		
North Atlantic	Liver	0.4	8.8	Harvey et al., 1974
Georges Bank	Muscle	0.002	0.030	
Denmark Strait	Muscle	0.003	Not detected	Harvey et al., 1974
Norwegian Sea, Sogndalsfjord, 1977	Liver	0.742	0.559	Skaare et al., 1985
West coast of Norway, Dalsfjord, 1982	Liver	0.687	0.352	Skaare et al., 1985
Barents Sea,	Liver	0.037	-	Savinova et al., 1981
SW part, 1977	Muscle	0.004	-	
Barents Sea, water off	Liver	0.027	0.059	Savinova, 1991
Rybachij Peninsula, 1983	Muscle	0.006	0.008	
Barents Sea, waters	Gonad	0.020	0.001	
off Bear Island, 1983	Liver	0.02-0.03	0.045-0.047	Savinova,1991
		Poutassou Micromestistius poutassou.		
Barents Sea, waters off Bear Island, 1983	Muscle	0.006	0.002	Savinova,1991
		Capelin Mallotus villosus.		
Barents Sea, Kildinskaya Bank, 1983	Whole	0.007	0.009	Savinova, 1991

NOTE: dash indicates no determination

Table 20
Heavy metal concentrations in cod from different regions (mg/kg wet wt)

Region, years	Tissue	Cd	Zn	Cu	Ni	Pb	As	Cr	Se	Source
Coastal waters off England, 1970-1973	muscle	<0.2	3.9	<0.5		<0.1				Portmann, 1979
North Sea,1971-1972	liver	0.06	17.7	2.9	2.5					
	muscle	0.11	6.1	1.0	2.3					
	gill		24.2	0.5	3.3					Wright, 1976
North Sea, 1974	muscle	<0.2	3.7	<0.2-1.0		<0.3		<0.2-0.9		Murray, 1979
Barents Sea, 1975	muscle	<0.05	3.8	0.6		<0.5				ICES, 1977
Norwegian Sea, 1975	muscle	<0.05	4.0	0.8		<0.5				
Waters off W.Spitsbergen, 1975	muscle	<0.05	3.9	0.8		<0.5				
Barents Sea, 1975	muscle	<0.2	5.4	0.4		<0.2-0.4				Murray, 1981
North Sea, 1975	muscle	<0.2	3.6	0.38		<0.2-0.5		<0.2-0.5		
Costal waters off W.England, 1975	muscle	<0.2	3.2	0.5		<0.2-0.4		0.36		
Distant waters W.Greenland, 1975	muscle	<0.2	16	6.0	<0.8	<0.2				
	liver	<0.2	5.0	6.0	<0.2	<0.2				
Baltic Sea, 1979-1981	liver	0.015-0.05	8.0-13.0	4.0-9.5		0.03-0.14				Perttilä *et al.*, 1982 a, b
Baltic Sea, 1982-1986	liver	0.012-0.051	10.0-20.3	2.9-11.8		0.01-0.06				Tervo, 1987
Scottish waters	muscle					0.7-4.6				Falconer *et al.*, 1983
St.Lawrence	liver	0.05-0.26								ICES, 1984
Coastal waters off Norway	muscle	0.02	5	0.5	0.2	0.01	5	0.1	0.5	Knutzen, 1987
	liver	0.5	20	10	0.5	0.1	10	0.5	3.0	
Oslofjord, 1988	muscle						4.2			Staveland *et al.*, 1993
North Sea	liver	0.02-0.66	6.6-43.0	1.7-15.0		0.02-0.08				
Barents Sea, 1992	liver	0.22	24.0	5.1	0.03	0.013	2.2	0.024	0.16	Savinov, 1994
	muscle	0.02	4.0	0.2	0.01	0.008	0.7	0.017	0.08	
	gill	0.05	19.4	0.40	0.044	0.01	1.0	0.29	0.65	

4.5 Marine birds

Although chlorinated hydrocarbons were detected in significant quantities in the organs and tissues of fisheating birds in England in the early 60's (Moore and Tatton, 1965), this did not attract much attention until the death of many thousands Murres and Razorbills on the shores of the Irish Sea in 1969-1970 (Bourne and Mead, 1969; Bourne, 1976).

Persistent chlorinated hydrocarbons and heavy metals have been detected in seabirds from Arctic regions in Canada (Vermeer and Reynolds, 1970; Vermeer and Peakall, 1977; Nettleship and Peakall, 1987; Hart et al., 1991; Elliott et al., 1992) and Norway (Fimreite and Bjerk, 1979; Holt et al., 1979; Norheim and Kjos Hansen, 1984; Ingebrigtsen et al., 1984; Barrett et al., 1985; Norheim, 1987; Norheim and Borch Iohnsen, 1990; Daelemans et al., 1992, Gabrielsen et al., 1995, Savinova et al., 1995). Contamination levels in birds from the Russian Arctic (Savinova, 1991; Savinova, 1992; Savinova and Gabrielsen, 1994), Greenland (Braestrup et al., 1974) and Alaska (Ohlendorf et al., 1982) are less well known.

Levels of chlorinated hydrocarbons in Arctic seabirds vary widely, depending on the species of bird and its ecology, feeding mode, migration routes, sex, age, physiological and biochemical parameters (e.g. rate of normal metabolic processes, quantity and composition of lipids, hepatic microsomal mono-oxygenase activitie, etc.).

Levels of organochlorines have been monitored in sea bird eggs in different Arctic regions from the 1960s. In 1968-1969, concentrations of DDTs in Kittiwake (*Rissa tridactyla*) eggs from the Atlantic Canadian coast were 213 ppm (Vermeer and Reynolds, 1970). In 1968-1973, the level of DDT in Kittiwake eggs from Norway was, on average, 1.2 ppm (Holt et al., 1979). Eggs from the same species collected in the mid 1970s at Prince Leopold Island in the Canadian Arctic archipelago had higher PCB levels (5.2 ppm) than eggs of the northern Fulmar (*Fulmarus glacialis*) (1.93 ppm) or the Thick-billed Murre (*Uria lomvia*) (0.01 ppm). DDE/PCB ratios in eggs and livers of Fulmars and Murres were much lower than in Kittiwakes, reflecting the lower levels of DDE in the latter. Kittiwakes appear to have a greater capacity to metabolize and excrete organochlorines compared with other seabirds (Nettleship and Peakall, 1987). This is probably related to the high metabolic rates of Kittiwakes compared with other species (Gabrielsen et al., 1987). Values in the range 0.05-1.0 ppm DDT and 1.1-2.4 ppm PCB have been reported Kittiwake liver from different Arctic regions (Bourne and Bogan, 1972; Carlberg and Bøler, 1985; Nettleship and Peakall, 1987; Savinova, 1991). The concentrations of DDT and PCB found in Kittiwakes from the east coast of the Kola peninsula (Savinova, 1991) were 35 times lower than in those from David Strait and Bear Island (Bourne, 1976).

Decreases of 55-60% in $\sum$-DDT and 69-86% in PCBs were observed in Thick-billed Murre eggs collected at Prince Leopold Island, Lancaster Sound, between 1976 and 1987. Similar declines in DDT and PCB residues occurred in livers of Kittiwakes and northern Fulmars, but not in those of the Thick-billed Murres from the same region (Nettleship and Peakall, 1987). In livers and muscles of Herring Gulls (*Larus argentatus*) from the Barents Sea, $\sum$DDT content declined 30-40 times during the last decade. However, the decline was not observed for PCBs (Savinova, 1992).

In 1982-83, Puffin (*Fratercula arctica*) eggs from two colonies in Northern Norway contained, on average, about 20% higher residues of PCBs and p,p'DDE than corresponding levels in Alaskan Puffin eggs (Ingebrigtsen et al., 1984; Ohlendorf et al., 1982). By contrast, lower PCB residues were found in eggs and adult brain tissues collected from Puffins in eastern Canadian coastal waters (Pearce et al., 1989).

At present, there are few reports on the levels of total PCBs in seabirds from the Arctic regions. Toxicologically, some congeners are more important than others and may account for most of the toxic effects. Daelemans et al. (1992) report on PCB congeners in Glaucous Gulls (*Larus hyperboreus*) and Black Guillemots (*Cepphus grylle*) from the Svalbard area. Particularly high levels of organochlorines have been recorded in Glaucous Gulls (Bourne, 1976; Norheim, 1987; Gabrielsen et al., 1995). This species partly act as a predator which during the breeding season preys on eggs and chicks of other seabirds. Average PCB concentration in the liver of the Glaucous Gull from Svalbard was 20.9 ppm, i.e. about 160 times higher than that found in the liver of the Black Guillemot (Daelemans et al., 1992). The total concentration of three selected non-orthocongeners represented only 0.18% of total PCB concentration. Congener 126 showed the highest average concentration (0.11% of total PCB) followed by 169 (0.04%) and 77 (0.03%).

Tables 21-29 summarize data on organochlorine residues in different species of seabirds from northern seas.

Heavy metal contents of Arctic seabirds have not been as well investigated as chlorinated hydrocarbons level. Heavy metals have been determined in different birds species collected in the 1980s off the west coast of Spitsbergen (Norheim and Kjos-Hansen, 1984; Carlberg and Bøler, 1985; Norheim, 1987; Norheim and Borch Johnsen, 1990); in Gulls, Fulmars, Long-tailed ducks (*Clangula hyemalis*) from the eastern coast of the Kola peninsula (in 1989) (Savinova, 1992); and in Black Guillemots, Kittiwakes, Common Eiders (*Somateria mollissima*), Glaucous Gulls, Fulmars and Brünnich's Guillemost (*Uria aalge*) from Greenland (Nielsen and Dietz, 1989).

The concentration of trace elements in Arctic seabirds examined in these studies are in good agreement with values reported in the literature for Atlantic Canadian seabirds (Elliot et al., 1992) and represent, in general, normal physiological levels. In contrast, the content of Cu in liver of Common Eider from Spitsbergen was about 40 times higher than in other species (Norheim and Kjos-Hansen, 1984; Norheim, 1987). The high level of Cu in Common Eider may reflect the fact that this species feeds mainly on mussels, snails and crustaceans, which have haemocyanin as their blood pigment. However, there is a scarcity of information on heavy metal bioaccumulation levels in different species of Arctic seabirds. This requires further investigation, especially in connection with oil and gas exploration on the shelf of arctic seas. The heavy metal contents in seabirds are presented in Table 29.

Table 21
Levels of residual chlorinated hydrocarbons (mg/kg wet weight) in Black-legged Kittiwake

Sampling area, year	Type of sample	Number of samples	∑DDT	∑PCB	Source
North Sea, coast 1963	Egg	6	0-0.7	-	Moore & Tatton
1964	Egg	6	0.4-1.2	-	1965
North Sea, coast of England, 1965	Egg	26	0.25	-	Robinson et al., 1967
Pacific Ocean, coast of USA, 1966	Muscle	?	1.3	-	Risebrough et al., 1967
Davis-Strait, 1972	Liver	1	0.13*	3.2	
	Muscle	1	0.08*	1.9	Bourne, 1976
Bear Island, 1972	Liver	1	0.08*	1.6	Bourne, 1976
	Muscle	1	0.18*	3,7	
	Fat	1	-	18.0	
North Scotland, 1971-1975	Liver	9	0.17*	5.2	Bourne, 1976
	Muscle	9	0.08*	2.96	
Northeast coast, England, 1971-1975	Liver	2	0.41*	3.8	Bourne, 1976
	Muscle	2	0.28*	7.2	
Northwest coast, England ,1971-1975	Liver	1**	23.0*	505.0	Bourne, 1976
	Muscle	1	28.0*	61.0	
58 03'N:02 02'W	Liver	1	-	0.3	Bourne & Bogan,
1971	Muscle	1	-	0.4	1972
	Fat	1	-	3.6	
60 22'N:04 31'W	Liver	1	-	4.8	Bourne & Bogan,
	Muscle	1	-	3.3	1972
60 22'N:04 25'W	Liver	1	-	3.5	Bourne & Bogan,
1971	Muscle	1	-	1.3	1972
	Fat	1	-	24.0	
60 31'N:04 52'W	Liver	1	-	2.0	Bourne & Bogan,
1971	Muscle	1	-	3.0	1972
60 08'N:04 00'W	Liver	1	-	2.1	Bourne & Bogan,
1971	Muscle	1	-	0.5	1972
	Fat	1	-	18.0	
60 22'N:04 31'W	Liver	1	-	20.6	Bourne & Bogan,
1971	Muscle	1	-	5.6	1972
58 12'N:00 13'E	Liver	1	-	4.5	Bourne & Bogan,
1971	Muscle	1	-	3.8	1972
57 20'N:00 20'E	Liver	1	-	3.4	Bourne & Bogan,
1971	Muscle	1	-	4.2	1972
57 00'N:00 21'W	Liver	1	-	4.6	Bourne & Bogan,
	Muscle	1	-	4.5	1972
Different parts	Egg	10	0.1*	2.0	Holt et al., 1979
of Norway, 1970-1973					
Prince Leopold	Liver	10	0.05*	1.19	Nettleship &
Island, 1975/1976	Liver	5	0.08*	242.0	Peakall, 1987
	Egg	10	0.38*	5.25	
	Liver of chicks	6	0.05*	2.15	
Eastern Murman,	Muscle	10	0.832	0.992	Savinova, 1991
Podpakhta Inlet,	Liver	10	1.030	1.128	
1979	Brain	10	0.228	0.350	
	Heart	10	0.930	0.996	
Norway, E. Finmark, 1983	Egg	11	0.62*	1.35	Barrett et al., 1985
Norway, W. Finmark, 1983	Egg	10	0.47*	1.59	Barrett et al., 1985
Norway, Lofoten, 1983	Egg	10	0.16*	1.90	
Svalbard, (Hornsund), 1984	Liver	2	0.07*	0.57	Carlberg & Böler,
	Egg	5	0.186*	1.96	1985

NOTE: dash indicates not determinated ; * - DDE only; ** birds found dead

Table 22
Levels of residual chlorinated hydrocarbons (mg/kg wet weight) in Herring Gull

Sampling area, year	Type of sample	Number of samples	∑DDT	∑PCB	Source
Baltic Sea, coast of Sweden, 1968-1970	Muscle	4	-	18.00	Olsson et al., 1973
Faeroe Island., 1971	Liver	1	3.15*	12.60	Bourne & Bogan, 1972
	Muscle	1	3.35*	13.40	Bourne, 1976
Isle of May, Fife,1971	Liver	3	-	0.2;1.2;0.6	Bourne & Bogan, 1972
	Muscle	3	-	0.5;1.2;1.0	Bourne, 1976
	Fat	3			
Eastern Scotland, 1971-1975	Liver	5	0.24*	0.52	Bourne, 1976
	Muscle	5	0.22*	0.64	
Lake Paijanne,	Liver	-	8.26	13.49	Särkkä et al., 1978
Southern Finland,	Muscle	-	7.50	11.27	
1972-1974					
Baltic Sea,	Young				Falandysz, 1980
coast of Poland,	chicks**				
1975-1976	Liver	4	3.90	4.80	
	Muscle	4	6.50	6.60	
	Fat	3	71.00	63.00	
	Adult birds**				
	Liver	4	75.0	100.0	
	Muscle	3	180.0	280.0	
Baltic Sea, coast	Egg	20	7.0	18.1	Lemmetyinen
of Finland, 1978	Muscle of:				et al., 1982
	hatchings,	13	17.20	33.00	
	chicks,	9	0.80	2.10	
	juveniles,	20	1.0-1.1	1.7-2.2	
	adult gulls	25	6.7-6.9	18.7-22.1	
Eastern Murman,	Muscle	6	3.02	2.99	Savinova, 1991
Podpakhta Inlet,	Liver	6	3.47	3.75	
1979	Brain	6	1.34	1.62	
	Heart	6	2.06	1.13	
	Spleen	6	2.92	1.48	
Eastern Murman,	Young				Savinova, 1992
Yarnyshnaya Inlet,	chicks				
1989	Liver	3	1.01	2.03	
	Muscle	3	0.92	1.80	
	Adults				
	Liver	6	1.43	4.06	
	Muscle	6	1.20	3.90	
North Atlantic, coast of Canada, 1968-1969	Egg	7	13.0	30.0	Vermeer & Reynolds, 1970
North Atlantic, coast of Canada, 1971-1973	Egg	43	2.00	4.10	Zitko et al., 1972
St. Lawrence Strait, 1968-1972	Egg	25	6.40	16.00	Vermeer & Peakall, 1977
Lyngør, Aust-Agder, Norway, 1972	Egg	10	0.87*	4.69	Fimreite et al., 1974 1977
Frierfjorden,Telemark , Norway, 1981-1983	Egg	40	0.73*	4.60	Bergstrom & Norheim, 1986

Continued

Table 22, continued

Sampling area, year	Type of sample	Number of samples	∑DDT	∑PCB	Source
Lake Paijanne, South Finland, 1972-1974	Egg	43	3.75*	4.64	Paasivirta et al., 1981
Lake Ontario, Canada, 1977	Egg	10	13.20*	96.0	Norstrom et al., 1981
Appledore Island, 1977	Egg	30	1.94*	7.76	Szaro et al.,1979
Fisherman Island, 1977	Egg	28	1.93*	9.06	Szaro et al., 1979
Different parts of Norway ,1968-1973	Egg	88	1.20*	3.0	Holt et al., 1979
Øsfold , Norway, 1969-1970	Egg	30	1.26*	3.13	Holt et al., 1979
Telemark , Norway, 1969-1970	Egg	10	2.01*	5.0	Holt et al., 1979
Hordaland, Norway, 1969-1970	Egg	10	2.20*	5.0	Holt et al., 1979
Sør-Trøndelag , Norway, 1969-1970	Egg	10	1.35*	1.9	Holt et al., 1979
Nordland, Norway, 1969-1970	Egg	10	1.13*	2.0	Holt et al.,1979
Finnmark , Norway, 1969-1970	Egg	10	1.69*	2.2	Holt et al., 1979
Hornøya, Finnmark, Norway, 1972	Egg	10	1.91*	9.65	Fimreite et al., 1977
Røst, Nordland, Norway, 1972	Egg	4	0.92*	3.00	Fimreite et al., 1977
Runde, Norway, 1972	Egg	14	1.52*	3.14	Fimreite et al., 1977
Østfold, Norway, 1969,	Egg	23	1.20*	4.3	Moksnes &
1979-1981	Egg	23	0.60*	5.3	Norheim, 1986
Telemark, 1969	Egg	21	1.70*	5.5	Moksnes &
1979-1981	Egg	38	0.50*	3.5	Norheim, 1986
Rogland, Norway, 1979-1981	Egg	7	0.20*	1.2	Moksnes & Norheim, 1986
Sør-Trøndelag , Norway, 1969	Egg	10	1.30*	2.7	Moksnes &
1978-1981	Egg	10	0.4*	1.8	Norheim, 1986
Nordland, Norway, 1969	Egg	10	1.80*	1.7	Moksnes &
1978-1981	Egg	9	1.30*	6.7	Norheim, 1986
Finnmark, Norway, 1969	Egg	9	1.60*	4.6	Moksnes &
1978-1981	Egg	25	1.50*	3.8	Norheim, 1986
E.Finmark, Norway, 1983	Egg	9	1.33*	4.1	Barrett et al., 1985
W.Finmark, Norway, 1983	Egg	11	1.29*	3.8	Barrett et al., 1985
N.Nordland, Norway, 1983	Egg	13	0.62*	1.8	Barrett et al., 1985
Lofoten, Norway, 1983	Egg	10	1.05*	7.1	Barrett et al., 1985

NOTE: dash indicates not determinated; * - DDE only
** - birds found dead.

Table 23
Levels of residual chlorinated hydrocarbons (mg/kg wet weight) in Glaucous Gull

Sampling area, year	Type of sample	Number of samples	∑DDT	∑PCB	Source
Svalbard (Hornsund),1984	Liver	1	2.47*	12.4	Carlberg &
	Liver	1	0.61*	2.5	Böler, 1985
Bear Island, 1972	Liver	6	23.30*	72.0 (6.0-311.0)	Bourne & Bogan, 1972;
	Fat	1	-	535.0	Bourne 1976
	Muscle	6	8.4	29.0 (8.0-88.0)	
Western coast of Spitsbergen, 1980	Liver	11	1.9*	6.1	Norheim &
	Fat	6	36.0*	82.0	Kjos-Hansen, 1984
Different parts of Norway, 1970-1974	Egg	37	1.7*	10.0	Holt et al., 1979
London, England	Liver	1	1.4*	7.0	Bourne,1976
	Muscle	1	2.0*	10.0	

NOTE: dash indicates not determined; * - DDE only

Table 24
Levels of residual chlorinated hydrocarbons (mg/kg wet weight) in Great black- backed Gull

Sampling area, year	Type of sample	Number of samples	∑DDT	∑PCB	Source
North Rona, 1971	Liver of adult gull**	1	-	6.8	Bourne & Bogan, 1972
	young gull**	1		0.2	
Different parts of Norway, 1970-1974	Egg	37	1.7*	10.0	Holt et al.,1979
Frierfjorden, Telemark, Norway, 1981-1983	Egg	30	0.41*	0.6	Bergstrom & Norheim, 1986
Lake Paijanne, South Finland, 1972-1974	Egg	45	1.63*	3.36	Paasivirta et al., 1981
Scotland,1971-1975	Liver	6	3.65*	8.5	Bourne, 1976
	Muscle	5	1.28*	7.12	
Massachussets, 1971-1975	Liver	1	3,60*	4.9	Bourne, 1976
Appledore Island ,1977	Egg	28	8.66*	30.96	Szaro et al., 1975
Eastern Murman,	Muscle	6	3.27	1.72	Savinova, 1991
Podpakhta Inlet, 1979	Liver	8	2.27	2.48	
	Brain	7	1.42	0.83	
	Heart	7	5.37	3.95	
	Spleen	5	12.91	3.77	
Frierfjorden, Telemark, Norway, 1981-1983	Egg	30	0.41*	0.6	Bergstrom & Norheim, 1986

NOTE: dash indicates not determinated; * - DDE only
** - birds found dead

Table 25
Levels of residual chlorinated hydrocarbons (mg/kg wet/weight) in Fulmar

Sampling area, year	Type of sample	Number of samples	∑DDT	∑PCB	Source
Coast of California,					Risebrough et al., 1968
1966	Whole	2	0.41;3.4	0.08;0.34	
1967	Whole	1	17.5	6.5	
60 08'N:04 00'W	Liver	1	-	3.2	Bourne & Bogan, 1972
1971	Muscle	1	-	3.1	
	Fat	1	-	57.0	
60 31'N:04 52'W	Liver	1	-	5.0	Bourne & Bogan, 1972
1971	Muscle	1	-	9.4	
	Fat	1	-	103.0	
60 22'N:04 31'W	Liver	1	-	1.2	Bourne & Bogan, 1972
1971	Muscle	1	-	1.7	
	Fat	1	-	32.0	
60 26'N:04 39'W	Liver	1	-	1.5	Bourne & Bogan, 1972
1971	Muscle	1	-	2.3	
	Fat	1	-	42.0	
59 54'N:03 45'W	Liver	1	-	2.1	Bourne & Bogan, 1972
1971	Muscle	1	-	2.8	
	Fat	1	-	40.0	
60 21'N:00 29'W	Liver	1	-	5.0	Bourne & Bogan, 1972
1971	Muscle	1	-	3.9	
	Fat	1	-	55.0	
54 00'N:00 31'W	Liver	1	-	5.9	Bourne & Bogan, 1972
1971	Muscle	1	-	4.1	
Bear Island,	Liver	1	0.3*	1.6	Bourne, 1976
1972	Muscle	1	0.5*	2.6	
	Fat	1	-	12.9	
Fife, 1971	Liver	1	-	17.8	Bourne & Bogan, 1972
	Muscle	1	-	5.2	Bourne, 1976
Davis Strait, 1972	Liver	1	2.1*	4.2	Bourne & Bogan, 1972
	Muscle	1	1.8*	3.7	Bourne, 1976
Different parts of	Liver	8	0.2*	2.0	Holt et al., 1979
Norway, 1972-1973	Brain	7	<0.1*	0.5	
Same location, 1970-1973	Egg	6	0.9*	6.0	Holt et al., 1979
Prince Leopold Isl., 1975	Eggs	10	0.76*	1.93	Nettleship & Peakall, 1987
	Liver of chicks & adult birds	10	0.58*	1.43	
		10	0.25*	0.9	
1976	Liver	10	0.50*	1.79	
Coast of England	Liver	2	2.50*	7.9	Bourne, 1976
	Muscle	2	2.00*	5.2	
West coast of Spitsbergen, 1980	Liver	10	0.63*	1.6	Norheim & Kjos-Hanssen, 1984
	Fat	7	22.00*	59.0	
Svalbard (Hornsund), 1984	Liver	2	0.05*	0.10	Carlberg & Böler, 1985

NOTE: dash indicates determinated; *-DDE only

Table 26
Levels of residual chlorinated hydrocarbons (mg/kg wet weight) in Little Auk

Sampling area, year	Type of sample	Number of samples	∑DDT	∑PCB	Source
Svalbard (Hornsund), 1984	Liver	2	<0.05*	<0.10	Carlberg & Böler, 1985
Bear Island, 1972	Liver	1	0.23*	0.7	Bourne & Bogan, 1972,
	Muscle	1	0.06*	0.2	Bourne, 1976
Aberdeenshire, 1971	Liver	1	-	11.4**	Bourne & Bogan, 1972
	Muscle	1	-	2.1**	
1972	Liver	1	-	3.4**	
	Muscle	1	-	1.3**	
Sutherland, 1972	Liver	1	-	6.0**	Bourne & Bogan, 1972
	Muscle	1	-	1.2**	
West coast of Spitsbergen, 1980	Liver	9	0.067*	0.2	Norheim & Kjos-Hanssen, 1984
	Fat	7	2.10*	7.1	
Massachussets	Liver	2	0.69*	1.82	Bourne, 1976
East England	Liver	4	0.93*	5.02	Bourne, 1976
	Muscle	4	0.23*	1.82	

NOTE: dash indicates not determinated; *-DDE only
**-birds found dead

Table 27
Levels of residual chlorinated hydrocarbons (mg/kg wet weight) in Puffins

Sampling area, year	Type of sample	Number of samples	∑DDT	∑PCB	Source
North Sea, coastal waters, 1963	Egg	1	1.0*	-	Moore & Tatton, 1965
Svalbard (Hornsund), 1984	Liver	2	0.22	0.61	Carlberg & Böler, 1985
Faeroe Island,1971	Liver	5	-	0.34	Bourne & Bogan, 1972
	Muscle	5	-	0.18	
58 48'N:00 16'E	Liver	3	-	0.4	Bourne & Bogan, 1972
1971	Muscle	3	-	0.27	
Spitsbergen,	Liver	2	0.085*	0.5; 0.2	Bourne & Bogan, 1972
1972	Muscle	2	0.035*	0.2; 0.1	Bourne, 1976
	Fat	2	-	5.2; 2.4	
Davis Strait, 1972	Liver	2	0.01*	0.02; 0.03	Bourne & Bogan, 1972
	Muscle	2	0.01*	0.02; 0.05	Bourne, 1976
	Fat	2	-	0.25; 0.5	
East England	Liver	1	0.54*	2.7	Bourne, 1976
	Muscle	1	0.40*	2.0	
South England	Liver	1	2.10*	43.7	Bourne, 1976
	Muscle	1	0.70*	13.8	
E. Finmark, Norway, 1983	Egg	11	0.62*	2.35	Barrett et al., 1985
W. Finmark, Norway, 1983l	Egg	10	0.74*	1.55	Barrett et al., 1985
N. Nordland, Norway, 1983	Egg	10	0.51*	0.56	Barrett et al., 1985
Lofoten, Norway, 1983	Egg	20	0.52*	0.80	Barrett et al., 1985

NOTE: dash indicates not determinated; * - DDE only

Table 28
Levels of residual chlorinated hydrocarbons (mg/kg wet weight) in Brünnichüs Guillemot

Sampling area, year	Type of sample	Number of samples	∑DDT	∑PCB	Source
Svalbard (Hornsund), 1984	Liver	4	0.073*	0.18	Carlberg & Böler, 1985
	Egg	6	0.30*	0.68	
Bear Island, 1972	Liver	2	0.1*	0.3; 0.1	Bourne & Bogan, 1972
	Muscle	2	0.08*	0.2; 0.1	Bourne, 1976
	Fat	2	-	3.1; 3.1	
West coast of Spitsbergen, 1980	Liver	9	0.16*	0.4	Norheim & Kjos-Hanssen, 1984
Prince Leopold Island,					Nettleship & Peakall, 1987
1975	Egg	12	0.31	0.72	
1976	Egg	10	0.44	1.01	
1977	Egg	10	0.39	0.91	
1976	Liver	10	0.17	0.44	
1977	of chicks	9	0.05	0.15	
1975	Liver	10	0.06	0.22	
1976	of	12	0.19	0.53	
1977	adults	11	0.12	0.36	
Coast of Greenland	Fat	5	3.5*	12.9	Clausen & Berg, 1975
Prince Leopold Island,					Wong, 1985
1975	Egg	12	0.29*	1.23	
1975	Liver	10	0.05*	0.36	
1976	Liver	12	0.15*	0.80	
1977	Egg	10	0.37*	0.64	
1977	Liver	10	0.05*	0.30	
Southeast of Maxwell Bay	Fat	2	1.4*	-	Wong, 1985
Lancaster Sound	Fat	8	4.5*	1.25	

NOTE: dash indicates not determinated; * - DDE only ·

Table 29
Levels of residual chlorinated hydrocarbons (mg/kg wet weight) in Common Guillemot

Sampling area, year	Type of sample	Number of samples	∑DDT	∑PCB	Source
Waters off St. Alb. Head, 1963	Egg	4	1.5-4.0**	-	Moore & Tatton, 1965
Waters off Great Saltee Island, 1964	Egg	6	1.3-2.1**	-	Moore &Tatton, 1965
Coastal waters of England and Ireland,					Borlakoglu et al., 1990
1978	Fat	9	-	0.25	
1979	Fat	6	-	0.45	
Faeroe Islands,	Liver	5	-	0.74	Bourne & Bogan, 1972
1971	Muscle	5	-	0.52	
58 00'N:02 22'W	Liver	2	-	0.9; 7.6	Bourne & Bogan, 1972
1971	Muscle	2	-	0.5; 4.6	
58 48'N:00 16'E	Liver	1	-	0.50	Bourne & Bogan, 1972
1971	Muscle	1	-	0.20	
Bear Island,	Liver	1	-	0.80	Bourne & Bogan, 1972
1972	Muscle	1	-	0.80	
	Fat	1		4.70	
57 33'N:01 43'E	Liver	1	-	0.50	Bourne & Bogan, 1972
1971	Muscle adults	1	-	0.30	
	Liver,	1	-	0.04	
	Muscle chicks	1	-	0.03	
Loch Riddon, Argyll	Muscle	1	-	0.10	Bourne & Bogan, 1972
Aberdeenshore,	Liver	1	-	0.10	Bourne & Bogan, 1972
1971	Muscle	1	-	0.10	
Shetland,	Liver	1	-	6.00	Bourne & Bogan, 1972
1971	Muscle	1	-	1.20	
Aberdeenshore,	Liver	1	-	0.60	Bourne & Bogan, 1972
1972	Muscle	1			
	Fat	1	-	11.80	
Prestwick, Ayr.,	Liver	1	-	0.20	Bourne & Bogan, 1972
1970	Muscle	1	-	0.20	
Bear Island	Liver	1	0.20*	0.80	Bourne, 1976
	Muscle	1	0.20*	0.80	
Faeroe Island	Liver	5	0.19*	0.74	Bourne, 1976
	Muscle	5	0.12*	0.76	
East England	Liver	7	0.48*	2.90	Bourne, 1976
	Muscle	7	0.26*	4.56	
Irish Sea and	Liver	12	7.69*	54.90	Bourne, 1976
Firth	Muscle	8	1.24*	9.25	
Coast of Sweden, 1968	Egg	9	40.00	16.00	Jensen et al., 1969

continued

Table 29, continued

Sampling area, year	Type of sample	Number of samples	∑DDT	∑PCB	Source
Gdansk Bay,	Female				Falandysz &
1981	fat	1	15.0	67.0	Szefer, 1984
1981-82,		2	24.0	82.0	
1982,		1	300.0	100.0	
1983,		1	22.0	100.0	
1981-82,	Male	1	36.0	71.0	
1982,	fat	1	19.0	61.0	
1983,		2	45.0	120.0	
	Immature				
1983	fat	1	8.9	25.0	
E. Finmark, Norway, 1983	Egg	10	0.94**	0.64	Barrett et al., 1985
W. Finmark, Norway, 1983	Egg	9	0.69**	0.70	
N. Nordland, Norway, 1983	Egg	7	0.49**	0.36	
Lofoten, Norway, 1983	Egg	8	0.33**	0.79	

NOTE: dash indicates not determinated; * - DDE only; ** - ppDDE only.

The investigations of chlorinated hydrocarbons and heavy metals in different species of marine colonial birds from the Barents sea region, carried out by Norwegian Institute for Nature Research and Murmansk Marine Biological Institute in 1991-1992 have shown some spatial and temporal trends. The chlorinated hydrocarbon levels in seabirds from the Barents Sea region in the 90's are generally lower than in the past decade (Savinova et al., 1995). Heavy metals contamination levels found in seabirds from the Barents Sea indicate that the levels are low compared to other northern seas, i.e. Canada and Greenland (Savinova and Gabrielsen 1994).

The observed contamination levels in Arctic seabirds have been considerably below those expected to produce lethal effects. Althougt there is much evidence that levels are frequently high enough to produce sub-lethal effects. As a group, birds are more resistent to acutely toxic effects of PCBs than are mammals. LD-50s for various species of birds varied from 604 to more than 6,000 mg Aroclor/kg diet (Eisler, 1986).

For all avian species, PCB residues of 310 mg/kg fresh weight or higher in brain were associated with an increased likehood of death from PCB poisoning (Stickel et al., 1984). In the field, PCBs were the probable cause of death of Glaucous gull from Bear Island in 1972 and 1987 (Bourne and Bogan, 1972, Bourne 1976, Gabrielsen et al., 1995), and the mortality of puffins on Lofoten Islands in the 1970s (Walker, 1990). This period was characterized by relatively high levels of PCBs and other organochlorine residues. It was reported in seabirds and it is possible that toxic effects were produced following the rapid mobilisation of such compounds from the storage fat of the birds (Walker, 1990).

Since the beginning of the 1980's scientists from the Norwegian Polar Research Institute have observed Glaucous gulls dead or dying at the southern part of Svalbard and at Bear Island (Bakken, pers.comm.). High concentrations of PCBs were found in brain and liver, and it is possible that PCB and other OCs may be responsible for the death (Gabrielsen et al., 1995).

A number of studies have been concerned with the effects of chlorinated hydrocarbons on reproduction in seabirds, and two major types of effects have been considered - eggshell thinning and direct toxicity to embryo.

Egg-shell thinning associated with DDE and PCBs residues has been reported for several species of Arctic seabirds: gannets (*Sula bassana*) in Canada (Parslow et al., 1973), and falcons in Norway (Nygård, 1983). Eggshell thickness of the Peregrine falcon (*Falco peregrinus*) from Norway declined 85% between 1854 and 1976; eggs collected in 1976, containing dead embryos, had 724 ppm of PCBs in lipids, and up to 110 ppm on a fresh weight basis (Nygård, 1983). In the Canadian study of gannet, it was suggested that DDE-induced egg-shell thinning was responsible for reduced reproductive success and consequent population decline in the 1960s. These effects were associated with DDE levels of 20-30 ppm in the eggs (Parslow et al., 1973). For most avian species, a reduction in eggshell thickness of 15 to 20% is suggested as a critical value beyond which population numbers will decline (Nygård, 1983).

Toxic effects of organochlorines to different species of birds are discussed by Cooke (1973) and Peakall and Fox (1987). Elliot et al. (1988) reported higher levels of dieldrin, heptachlore epoxide, HCH, oxychlordane, trans chlordane, and PCB in failed gannet eggs relative to those in fresh eggs.

Table 30
Mean concentrations of trace metals in seabirds, ppm dry weight (wet weight)*

Sampling area, year	Type of sample	N	Hg	Cd	As	Zn	Cr	Cu	Pb	Se	Ni	Source
Kittiwake (*Rissa tridactyla*)												
60°30'N; 4°30'W 1971	liver	3	3.13	-	-	-	-	-	-	-	-	Dale *et al.*, 1973
Eastern Murman, Barents Sea, 1989	muscle	1	-	-	-	18.2*	-	4.2*	-	-	0.42*	Savinova, 1992
Franz-Josef Land, 1991	liver	5	0.49	9.90	18.1	89.3	0.05	24.0	1.21	6.33	0.40	Savinova & Gabrielsen, 1994
	muscle	5	0.07	0.42	6.94	59.1	0.05	19.2	5.13	1.38	0.34	
Hornøya, Northern	liver	5	0.80	27.8	1.83	99.6	0.06	25.3	0.10	22.4	0.47	
Norway, 1991	muscle	5	0.20	1.06	0.93	43.6	0.10	19.4	0.08	5.92	0.37	
Bear Island, 1991	liver	4	0.58	16.1	0.82	88.6	0.15	21.2	0.08	3.87	0.34	
	muscle	4	0.19	1.47	0.64	63.3	0.06	25.1	35.0	15.1	0.41	
New-Olesund,	liver	5	1.95	48.0	3.12	126.3	0.05	25.1	35.0	15.1	0.41	
Spitsbergen, 1991	muscle	5	0.43	1.93	1.01	49.1	0.07	23.7	16.7	5.94	0.37	
Herring Gull (*Larus argentatus*)												
Faeroe Island, 1971	liver	1	2.5	-	-	-	-	-	-	-	-	Dale *et al.*, 1973
Isle of May, Fife, 1972	liver	2	12.0 8.0	-	-	-	-	-	-	-	-	
Isle of May, Fife, 1976	liver	22	4.08	10.5	-	91.6	-	-	-	7.86	-	Hutton, 1981
	kidney	35	3.87	27.0	-	97.6	-	-	-	14.1	-	
	brain	15	1.42	0.45	-	83.2	-	-	-	-	-	
	muscle		1.29	n.d.	-	70.0	-	-	-	-	-	
	bone	15	-	-	-	-	-	-		37.7	-	
Eastern Murman,	adult											Savinova, 1992
Barents Sea,	liver	2	-	-	-	38.8*	-	5.9*	-	-	0.05*	
1989	muscle	2	-	-	-	20.9*	-	4.4*	-	-	0.21*	
	juvenile											
	liver	2	-	-	-	32.4*	-	5.5*	-	-	n.d.	
	muscle	2	-	-	-	23.6*	-	3.3*	-	-	n.d.	
Canadian Atl. coast	liver	6	1.7	3.6		90.1	-	16.8	0.69	3.21	-	Elliot *et al.*, 1992
Gull Isl., 1988	kidney	6	1.95	34.0	-	-	-	-	-	-	-	
	bone	6	-	-	-	-	-	-	34.0	-	-	
Mana Isl., 1988	liver	6	0.69	1.7		84.7	-	30.6	1.3	3.20	-	
	kidney	6	1.11	13.0	-	-	-	-	-	-	-	
	bone	6	-	-	-	-	-	63.0	-	-	-	
Kent Isl., 1988	liver	6	1.5	5.4		126	12.2	0.24	3.36	1380	-	
	kidney	6	1.6	4.0	-	-	-	-	-	-	-	
	bone	6	-	-	-	-	-	32.0	-	-	-	
Hornøya, Northern	liver	6	0.79	1.48	1.92	137.0	0.05	16.0	0.09	4.34	0.40	Savinova &
Norway, 1991	muscles	6	0.36	0.65	1.99	102.2	0.07	19.3	0.08	2.19	0.39	Gabrielsen, 1994
Glaucous Gull (*Larus hyperboreus*)												
Bear Island, 1972	liver	1	5.2	-	-	-	-	-	-	-	-	Dale *et al.*, 1973
West Spitsbergen	liver	11	1.6*	3.6*	-	32*	-	7.3*	-	2.2*	-	Norheim, 1987
1980	kidney	11	-	23*	-	46*	-	5.2*	-	-	-	
South Greenland,	liver	5	0.68*	5.13*	-	39.06*	-	-	-	2.85*	-	Nielsen & Dietz,
1984-1986	kidney	5	1.37*	39.91*	-	52.97*	-	-	-	0.54*	-	1989
	muscle	5	0.39*	0.20*	-	20.28*	-	-	-	6.75*	-	

continued

Table 30, continued

Sampling area, year	Type of sample	N	Hg	Cd	As	Zn	Cr	Cu	Pb	Se	Ni	Source
Northwest Greenland,	liver	8	2.67*	12.57*	-	42.36*	-	-	-	3.67*	-	
1984-86	kidney	8	2.06*	60.55*	-	54.28*	-	-	-	5.57*	-	
	muscle	8	0.66*	0.35*	-	23.35*	-	-	-	1.07*	-	
Bear Island,	liver	3	1.02	6.06	12.8	111.2	0.05	19.7	0.70	5.25	0.41	Savinova &
1991	muscle	3	0.29	0.45	3.92	50.6	0.04	20.2	2.44	1.41	0.33	Gabrielsen, 1994
New-Olesund,	liver	5	1.48	9.19	2.01	107.2	0.06	23.0	0.22	5.41	0.53	
Spitsbergen, 1991	muscle	5	0.64	0.54	0.91	62.7	0.05	21.7	0.12	2.31	0.41	
Great black-backed Gull(*Larus marinus*)												
Eastern Murman,	liver	1	-	-	-	26.84*	-	88*	-	-	0,18*	Savinova, 1992
Barents Sea,	muscle	1	-	-	-	21.46*	-	2.83*	-	-	n.d.	
1989												
Lesser black-backed Gull (*Larus fuscus*)												
Faeroe Isl., 1971	liver	1	0.7	-	-	-	-	-	-	-	-	Dale *et al.*, 1973
Fulmar (*Fulmarus glacialis*)												
St. Kilda, U.K.	liver	5	29.4	49.0	-	364	-	-	-	-		Osborn *et al.*, 1979
June 1977	kidney	5	13.4	228	-	310	-	-	-	-	-	
	brain	5	<3.33		-	57.6	-	-	-	-	-	
	muscle	5	1.59	3.43	-	57.7	-	-	-	-	-	
	heart	5	2.16	<2.22	-	89.6	-	-	-	-	-	
60°30'N 0°30'W	liver	2	3.8	-	-	-	-	-	-	-	-	Dale *et al.*, 1973
1971			3.6	-	-	-	-	-	-	-	-	
West Spitsbergen,	liver	10	2.1*	17*	-	73*	-	6.2*	-	3.0*	-	Norheim, 1987
1980	kidney	10	-	55*	-	50*	-	4.1*	-	-	-	
Eastern Murman,	liver	1	-	-	-	53.7*	-	9.8*	-	-	n.d.	Savinova, 1992
Barents Sea,	muscle	1	-	-	-	29.9*	-	2.9*	-	-	0.10*	
1989												
Bear Island,	liver	5	1.95	36.6	2.86	141.8	0.06	18.4	0.10	10.2	0.45	Savinova &
1991	muscle	4	0.23	5.10	1.10	56.0	0.07	19.8	0.07	2.63	0.35	Gabrielsen, 1994
New-Olesund,	liver	5	4.88	73.4	3.83	216.4	0.05	22.5	2.59	15.9	0.37	
Spitsbergen, 1991	muscle	4	1.56	44.3	4.73	121.4	0.05	21.6	19.9	11.5	0.48	
Common Guillemot (*Uria aalge*)												
58°N;2°30'W	liver	1	5.3	-		-	-	-	-	-	-	Dale *et al.*, 1973
57°30'N;2°00'W	liver	1	0.7	-	-	-	-	-	-	-	-	
1971												
Aurshire,	liver	1	5.1	-	-	-	-	-	-	-	-	
1970												
(oiled bird)												
Black Guillemot (*Cepphus grylle*)												
Kangatsiaq, South	liver	8	-	2.46*	-	-	-	-	-	-	-	Nielsen & Dietz, 1989
Greenland,	kidney	8	-	21.0*	-	-	-	-	-	-	-	
1984-1986	muscle	8	-	0.17*	-	-	-	-	-	-	-	
Nanortalik	liver	8	-	1.74*	-	-	-	-	-	-	-	
South Greenland	kidney	8	-	15.86*	-	-	-	-	-	-	-	
1984-1986	muscle	8	-	0.075*	-	-	-	-	-	-	-	
South Greenland	liver	20	0.63*	2.0*	-	29.65*	-	-	-	2.38*	-	-
1984-1986	kidney	20	0.50*	14.34*	-	36.82*	-	-	-	5.68*	-	-
	muscle	20	0.20*	0.10*	-	10.33*	-	-	-	0.77*	-	-
Northwest Greenland	liver	9	0.64*	4.07*	-	30.14*	-	-	-	3.41*	-	-
1984-1986	kidney	9	0.44*	26.96*	-	39.89*	-	-	-	5.07*	-	
	muscle	9	0.23*	0.30*	-	12.31*	-	-	-	0.83*	-	

continued

Table 30, continued

Sampling area, year	Type of sample	N	Hg	Cd	As	Zn	Cr	Cu	Pb	Se	Ni	Source
Northeast Greenland	liver	8	0.50*	3.48*	-	36.50*	-	-	-	1.52*	-	
1984-1986	kidney	8	0.35*	19.79*	-	43.83*	-	-	-	4.52*	-	
	muscle	8	0.12*	0.17*	-	36.96*	-	-	-	0.60*	-	
West Greenland	liver	9	0.37*	1.78*	-	-	-	-	-	-	-	
Uummanaq,	kidney	9	0.23*	13.21*	-	-	-	-	-	-	-	
1986	muscle	9	0.08*	0.25*	-	-	-	-	-	-	-	
Dundas Harbour	muscle	5	0.13*	0.59*	12.6*	40.32*	0.87*	15.72*	-	-	-	Wong, 1985
Lancaster Sound	liver	5	0.23*	0.83*	20.38*	84.86*	0.80*	16.05*	-	-	-	
1976	kidney	5	-	-	-	-		-	17.25*	-	-	
						Brünnich's Guillemot *(Uria lomvia)*						
West Spitsbergen,	liver	9	0.6*	3.9*	-	35*	-	8.2*	-	1.9*	-	Norheim, 1987
1980	kidney	9	-	16*	-	39*	-	6.9*	-	-	-	
Northwest Greenland,	liver	12	0.72*	6.66*	-	36.67*	-	-	-	1.84*	-	Nielsen & Dietz, 1989
1984-1986	kidney	12	-	30.89*	-	42.69*	-	-	-	3.69*	-	
	muscle	12	0.19*	0.26*	-	12.14*	-	-	-	0.65*	-	
Northeast Greenland	liver	6	0.95*	8.75*	-	45.91*	-	-	-	2.44*	-	
1984-1986	kidney	6	-	32.12*	-	47.23*	-	-	-	9.01*	-	
	muscle	6	0.24*	0.32*	-	13.75*	-	-	-	0.98*	-	
Hornøya, Northern	liver	5	0.88	5.80	1.86	91.2	0.05	16.9	0.09	7.53	0.44	Savinova &
Norway, 1991	muscle	5	0.29	1.90	0.28	43.5	0.05	20.4	0.11	3.63	0.36	Gabrielsen, 1994
New-Olesund,	liver	5	1.61	14.9	4.61	91.5	0.05	16.1	0.15	5.49	0.59	
Spitsbergen, 1991	muscle	5	0.60	1.23	0.79	40.7	0.05	23.1	0.41	3.36	0.40	
Bear Island,	liver	5	0.33	6.51	3.18	99.0	0.04	18.0	0.07	4.27	0.35	
1991	muscle	5	0.15	0.61	1.09	47.2	0.06	17.7	0.08	1.74	0.43	
						Little Auk *(Alle alle)*						
Aberdeen,	liver	1	5.5	-	-	-	-	-	-	-	-	Dale *et al.*, 1973
1972												
West Spitsbergen,	liver	9	0.5*	4.3*	-	37*	-	8.4*	-	2.6*	-	Norheim, 1987
1980	kidney	9	-	21*	-	40*	-	6.4*	-	-	-	
East Greenland,	liver	13	0.50*	5.42*	-	38.01*	-	-	-	6.14*	-	Nielsen & Dietz,
1984-1986	kidney	11	0.32*	32.21*	-	46.72*	-	-	-	11.53*	-	1989
	muscle	13	0.14*	0.37*	-	15.49*	-	-	-	2.51*	-	
Franz-Josef Land,	liver	2	0.23	6.64	1.05	81.3	0.05	21.4	0.11	4.85	0.44	Savinova &
1991	muscle	10	0.10	0.81	0.42	36.8	0.06	25.3	1.19	2.13	0.43	Gabrielsen, 1994
						King Eider *(Somateria spectabilis)*						
South Greenland	liver	13	0.50*	4.17*	-	-	-	-	-	-	-	Nielsen & Dietz,
Kangatsiaq,	kidney	13	0.28*	15.66*	-	-	-	-	-	-	-	1989
1984-1986	muscle	13	0.12*	0.23*	-	-	-	-	-	-	-	
South and West	liver	21	0.59*	5.81*	-	47.59*	-	-	-	7.71*	-	
Greenland,	kidney	20	0.31*	19.28*	-	35.55*	-	-	-	6.87*	-	
1984-1986	muscle	21	0.13*	0.36*	-	11.34*	-	-	-	0.84*	-	
West Spitsbergen,	liver	9	1.03*	4.3*	-	50*	-	270*	-	8.9*	-	Norheim & Kjos
May 1980	kidney	14	-	14*	-	33*	-	13*	-	-	-	Hansen, 1984; Norheim, 1987
						Common Eider(*Somateria mollissima*)						
New-Olesund, Svalbard,	liver	5	-	12.02*	-	122*	-	146.6*	-	8.48*	-	Norheim & Borch-
July 1986	liver	5	-	10.08*	-	59.8*	-	512*	-	13.9*	-	Iohnsen, 1990
Midlothian, U.K.,	liver	1	40.0	-	-	-	-	-	-	-	-	Dale *et al.*, 1973
January 1971												
E. Lothian,	liver	2	48.0	-	-	-	-	-	-	-	-	
1972		2	4.6									

continued

Table 30, continued

Sampling area, year	Type of sample	N	Hg	Cd	As	Zn	Cr	Cu	Pb	Se	Ni	Source
						Common Eider(*Somateria mollissima*)						
Tay, U.K.	liver		24-48	-	-	-	-	-	-	-	-	Jones *et al.*, 1972
South Greenland	liver	8	0.83*	3.26*	-	-	-	-	-	-	-	Nielsen & Dietz,
Kangatsiaq	kidney	7	0.25*	11.59*	-	-	-	-	-	-	-	1989
1984-1986	muscle	8	0.11*	0.11*	-	-	-	-	-	-	-	
South Greenland	liver	11	0.95*	3.07*	-	-	-	-	-	-	-	
Nanortalik,	kidney	11	0.27*	13.2*	-	-	-	-	-	-	-	
1984-1986	muscle	11	0.19*	0.16*	-	-	-	-	-	-	-	
Franz-Josef Land,	heart	2	0.04	0.06	4.14	100.5	0.04	30.3	0.14	2.77	0.37	Savinova &
1991	liver	5	0.08	0.30	4.36	122.0	0.16	113	0.24	5.14	0.37	Gabrielsen, 1994
	muscle	5	0.04	0.08	2.78	72.8	0.89	18.1	2.24	1.32	0.43	
New Olesund,	liver	6	1.88	25.8	0.37	367.9	0.06	173	0.48	12.3	0.34	
Spitsbergen, 1991	muscle	6	0.40	0.57	0.15	40.6	0.06	20.2	1.46	3.66	0.44	
	heart	1	0.45	1.27	-	77.0	0.06	19.1	0.32	7.29	0.48	
						Puffin *(Fratercula arctica)*						
58°45'N;0°15'W	liver	2	4.1	-	-	-	-	-	-	-	-	Dale *et al.*, 1973
1971			1.8									
St. Kilda, U.K.	liver	10		4.54	19.9	-	-	-	-	-	-	Osborn *et al.*, 1979
1977	kidney	10	5.02	114	-	-	-	-	-	-	-	
	brain	10	<1.93	n.d.	-	-	-	-	-	-	-	
	muscle	10		<1.36	<1.94	-	-	-	-	-	-	
	heart	10	1.24	<1.88	-	-	-	-	-	-	-	
Canadian Atl. coast,	liver	6	2.6	8.9	-	99.5	4.35	15.3	0.17	11.7	-	Elliot *et al.*, 1992
1988: Gull Isl.	kidney	6	2.6	92.0	-	-	-	-	-	-	-	
Isle St. Marie	liver	6	1.4	5.9	-	99.1	1.53	23.3	n.d.	8.29	-	
	bone	6	-	-	-	-	-	-	0.42	-	-	
	kidney	6	1.0	45.0	-	-	-	-	-	-	-	
New-Olesund,	liver	5	1.12	9.77	1.36	100.6	0.08	19.5	0.38	11.9	0.37	Savinova &
Spitsbergen, 1991	muscle	5	0.40	0.77	0.21	43.3	0.08	20.2	0.26	9.9	0.49	Gabrielsen, 1994
Hornøya, Northern	liver	5	1.22	2.60	0.79	84.6	0.06	17.6	0.10	9.18	0.48	
Norway, 1991	muscle	5	0.31	0.29	0.20	40.6	0.04	19.2	0.07	5.32	0.35	
						Long-tailed duck *(Clangula hyemalis)*						
Gudsom Bay,												Wong, 1985
1971	liver	33	1.3*	-	-	-	-	-	-	-	-	
1971	liver	12	0.29*									
1971	eggs	11	0.20*									
						Razorbill *(Alca torda)*						
Faeroe Island,	liver	2	4.0	-	-	-	-	-	-	-	-	Dale *et al.*, 1973
1971			7.5									
						Gannet *(Sula bassana)*						
Shetland,	liver	1	12.9	-	-	-	-	-	-	-	-	Dale *et al.*, 1973
1971												
Fife, U.K.,	liver	1	9.6	-	-	-	-	-	-	-	-	
1972												
Herbides, U.K.,	liver	1	13.3	-	-	-	-	-	-	-	-	
1972												
St. Kilda, U.K.	liver	4	10.2	16.2	-	141	-	-	-	-	-	Osborn *et al.*, 1979
1977	kidney	4	4.67	94.5	-	176	-	-	-	-	-	
	brain	4	<5.8	n.d.	-	70.8	-	-	-	-	-	
	muscle	4	0.9	2.7	-	45.8	-	-	-	-	-	
	heart	4	1.7	1.9	-	118	-	-		-	-	
Hebrides, U.K.,	liver	2	5.7	-	-	-	-	-	-	-	-	Dale *et al.*, 1973
March 1971			4.6									

The induction of enzymes is another sub-lethal effect caused by organochlorines. Many lipophilic pollutants are known to be inducers of hepatic microsomal cytochrome P-450 in vertebrates, including birds (Ronis and Walker, 1989). The induction of cytochrome P-450 forms can be used as an indicator of exposure to pollutants (Stegeman and Kloepper-Sams, 1987).

Other sub-lethal effects on seabirds that may be caused by chlorinated hydrocarbons include behavioural disturbances, immunotoxicity, and inappetence. However, sound evidence is still lacking (Walker, 1990). Mutagenic, carcinogenic, and teratogenic effects of PCBs are not documented for Arctic seabirds.

The existing data on contamination levels in Arctic seabirds will serve as a good base for future toxicological investigations in this region. There are still many unknown aspects concerning the effects of contaminants on Arctic seabirds. However it is clear that effects studies of persistent organochlorines at the biochemical and population levels are needed.

4.6. Marine mammals

The levels of PCBs, organochlorine pesticides and some heavy metals such as mercury and lead, are well documented in Arctic mammals, because of the need for comparisons with biota in more polluted environments and the general interest in the contamination levels of native people´s diets.

There is a large database of chlorinated hydrocarbon levels in Arctic marine mammals (Table 30) beginning with the first reports of DDT and PCBs in Arctic seals by Holden (1972). Ringed seals (*Phoca hispida*) have been used for temporal and spatial trend studies because they are widely distributed (Arctic Ocean, Bering and Baltic Seas) and are a resident species. Ringed seals from the Baltic Sea have high levels of OCs contaminants (Helle et al., 1976a, b). PCB levels in Arctic ringed seals are 10 times lower than those reported for land-locked ringed seals in Finland (Helle et al., 1976a, b; Muir et al., 1992b) and up to 50 times lower than those reported for the same species in the the Baltic Sea during the mid-1970s (Bergman et al., 1981). Ringed seals from seven locations in the Canadian Arctic (Muir et al., 1992b) and the Chukchi Sea (Becker et al., 1989) have very similar levels of total DDT and PCBs in blubber. Levels of total DDT found by Oehme et al. (1988) in ringed seals from Spitsbergen were about twice as high as concentrations in the same species in the Canadian Arctic. PCBs levels were also higher in seals from Spitsbergen whereas HCH levels were higher in seals from Canadian Arctic. Concentrations of 2.3.7.8-TCDF in Canadian ringed seals ranged from less than 2 to 7 ng/kg which was lower than values reported for Spitsbergen ringed seals, range 9 to 13 ng/kg (*Oehme et al.,* 1988; 1990). Concentrations of 2,3,7,8-TCDD in blubber of ringed seals from Spitsbergen ranged from 2 to 8.2 ng/kg (Oehme et al., 1988). Bignert et al.(1989) compared PCDD and PCDF levels in blubber of seals from the Baltic, the Barents Sea and Spitsbergen. Two congeners, 2,3,4,7,8-PnCDF and 1,2,3,7,8-PnCDD, were the predominant PCDD/PCDF residues in the Baltic ringed seals, but were present at 6-10 times lower levels in Arctic ringed seal blubber.

Only recently, Daelemans et al., (1993) reported concentrations of 2,3,7,8-TCDF five to ten times higher than the toxic equivalent (TEQ) values for the PCBs in arctic seal samples from the dibenzodioxin (2,3,7,8-TCDD) of the PCBs in ringed seals from the Svalbard area. (Oehme et al.,1988; 1990). Comparing the three different seal tissues Daelemans et al. (1993) found no significant difference in PCB pattern; the absolute concentrations expressed per unit lipid weight were for the females in the following order: blubber liver > kidney; for males: blubber > liver > kidney.

Residue levels of chlorinated hydrocarbons in blubber of harbour seal (*Phoca vitulina*) from Norwegian waters were analysed by Skaare et al.(1990). About 1000 harbour seals were found sick or dead in the Oslofjord and along the Southern and Northwestern coast of Norway during the disease outbreak caused by a morbilli virus in 1988. PCBs concentrations found were 2-4 times higher than total DDT concentrations. The PCB and DDT concentrations ranged from 0.4-38 and 0.1-8.8 mg/kg w.w., respectively. These concentrations did not canse death in the seal. However, it was possible that the recorded concentrations added additional stress to animals during illness. Recently, concentrations of individual PCBs (including 3 coplanar congeners) have been determined in different tissues of harbour seals from the same region (Bernhoft and Skaare, 1994). The total PCB (sum of 21 congeners) was about 1000-10.000 times higher than the sum of the 3 coplanar PCBs. The PCB concentrations in blubber of harbour seals from eastern and notheastern England that died during the morbilli virus infection in 1988 (Law et al., 1989), were about two times higher than the levels observed by Bernholf and Skaare (1994). From the Baltic to the Norwegian coast, there seems to be a descending gradient of PCB in harbour seal blubber: 59 ppm at Western Sweden (Blomkvist et al., 1992), 23 ppm at Western Denmark (Storr-Hansen and Spliid, 1993), 13 ppm at Southern Norway, 5.8 ppm off the western coast of Norway (Skaare et al., 1990), 3-5 ppm at Northern Norway (Espeland, 1993).

PCB levels in harp seals (*Phoca groenlandica*) observed in the Baffin Bay region were similar to those reported in ringed seals, but total DDT levels were about twice high. PCB levels in harp seals from the Canadian Arctic area ranged from 1 to 8 ppm, with the highest levels in specimens from the Gulf of St.Lawrence where local pollution is greater (Ronald et al., 1984) Harp seals in the Canadian Arctic migrate from the Newfound-land/Gulf of St. Lawrence regions and higher levels in these animals may be due to accumulation in the in the southern areas. Compared to other investigations from the Arctic, female harp seal from the West Ice seem to be less contaminated with organochlorines compared to harp seals from the East Ice population (Espeland 1993).

Fur seals (*Callorhinus ursinus*) also migrate long distances, from the Bering Sea to winter off the coast of Japan and the west coast of Canada and USA. PCBs (as Aroclor 1254) in blubber of fur seals from the Pribilof Islands (Calambokidis & Peard, 1985; Kurtz, 1984; 1987) were about two times higher than in ringed seals at Holman Island in the western Canadian Arctic (Addison et al., 1986). More recent analyses of PCB congeners in fur seal blubber indicate that concentrations are similar to those reported for ringed seals in the Canadian Arctic (Becker et al., 1989).
Walrus (*Odobenus rosmarus*), which feed predominately on molluscs, have much lower blubber levels of PCBs and organochlorine pesticides than fisheating marine mammals (Born et al., 1981;

Taylor et al., 1989). Female Atlantic walrus from Pond Inlet had about five times lower levels of PCBs than ringed seals from the east/central Canadian Arctic (Thomas and Hamilton, 1988).

Geographical trends in organochlorine levels in beluga whales (*Delphinapterus leucas*) provide an additional comparison between Arctic residents and a highly contaminated population from the St. Lawrence River.

Arctic belugas had, on average, about 25-30 times lower levels of PCBs and total DDT, respectively, than did belugas from the St. Lawrence estuary (Muir et al., 1992a). For other organochlorines, differences between Arctic and St. Lawrence belugas were less than tenfold.

Temporal trends in DDT, PCBs and other organochlorine residues in tissues of marine mammals have been examined in a number of recent studies (Addison et al., 1986; Muir et al.,1988;1992 a, b, c; Norstrom et al., 1988).

Addison et al. (1986) concluded that PCBs declined about threefold in ringed seals from Holman Island between 1972 and 1981. This decline was similar to that observed in harp and grey seals on the east coast of Canada during the same time period (Addison et al., 1984). For total DDT, however, the main reason for the decline between 1972 and 1981 could be explained by the thicker blubber of animals collected in 1981. Declines in total DDT were 30-40% in Arctic ringed seals compared with 300% to 500% in the east coast seals. A similar trend in DDT and PCB levels was found by Muir et al. (1988) who compared female ringed seals from 1975/76 with a cohort from 1983, sampled in Admiralty Inlet on north Baffin Island. PCBs showed the largest decline (50%) whereas DDE, total DDT, total Chlordane and HCH levels differed by 15% or less between the two groups. Kurtz (1984; 1987) did not find statistically significant declines in total DDT and PCBs in 3-4 year-old male fur seals collected between 1975 and 1981 from the Pribilof Islands.

Muir et al. (1990) also observed an increase in levels with time when they compared of organochlorines in beluga blubber oil from western Hudson Bay in 1965 with beluga blubber from the same area in 1986.

Canadian scientists have documented heavy metal contamination levels in marine mammal tissues (Wagemann et al., 1983; 1988; 1990; Wagemann, 1989). Wagemann and Muir (1984) summarised published data on heavy metals in tissues of cetaceans and pinnipeds from northern waters. Heavy metal content in marine mammals are presented in table 31.

Mean cadmium concentrations in renal tissue of whales (28-64 mg/kg) were higher than in the same tissue of most seals and porpoises. In both cetaceans and seals cadmium was consistently higher in renal tissues than in hepatic tissue by a factor of 2-5 (Wagemann and Muir, 1984).

Remarkably high concentrations of cadmium were found in kidney of narwhals from the Baffin Bay region (Wagemann et al., 1983) and from western Greenland (Hansen et al., 1990). Wagemann et al. (1983) found a bimodal distribution in the frequency of cadmium concentrations in the kidney of 60 animals without a concominant bimodal distribution for length or age of animals. Approximately one-third of the samples had a higher mean concentration (117 mg/g wet wt) than the rest of the samples. The median cadmium concentration, 63.5 mg/g in narwhal from Pond Inlet, was higher than that found in 93 narwhals (39.1 mg/g) by Hansen et al. (1990). The cadmium concentration in kidney was also high in seals from Greenland, although not as high as in narwhals. Some ringed seals from the vicinity of the Nanisivik mine on Strathona Sound had higher concentrations of heavy metals compared to a reference group located some distance from the mine (Wagemann, 1989). Similarly, belugas from the St. Lawrence estuary had higher lead and mercury concentrations in the kidney and liver tissues than belugas from five Arctic locations (Wagemann et al., 1990)

Much more data are available for mercury than for other trace metals in marine mammals. Much of it has been summarised for marine mammals from northern waters by Wagemann & Muir (1984) and Muir et al (1992a,b). Most cetaceans had mercury concentrations of 0.02-1.6 mg/kg in muscle tissue, 11-2 mg/kg in renal tissue, and 3-20 mg/kg in the liver. Mercury concentrations in whale and seal muscle frequently exceed 0.50 mg/g, particularly in older individuals because of the dependence of mercury concentrations on age. Smith and Armstrong (1978) reported a value of 2-12 mg/g in muscle of 30 year-old ringed seal and means of 0.72 and 27.5 mg/g in muscle and liver, respectively, in 80 ringed seals from the west coast of Victoria Island. Mercury concentrations in livers of belugas from five stocks in the Canadian Arctic ranged from means of 2.21 mg/g (Jones Sound) to 11.82 mg/g in the southern Beaufort Sea (Wagemann et al., 1988). Similar concentrations in beluga from western Greenland were reported by Hansen et al. (1990). By comparison, concentrations in livers of beluga from the St.Lawrence estuary averages 33.6 mg/g (Wagemann et al., 1991).

Contamination levels of marine mammals from the Barents Sea are unknown. During the last decade, studies have revealed that marine mammal populations in this region have declined dramatically (Kondakov, 1991). Future studies of contaminants in the Russian Arctic are necessary, because these substances may be responsible for decrease of marine mammal populations in the area.

The contamination levels of chlorinated hydrocarbons and heavy metals in polar bears (*Ursus maritimus)* from the Arctic were studied in the Svalbard area (Norheim et al., 1992; Kleivane et al., 1994), in the Canadian Arctic (Norstrom et al., 1986; Renzoni and Norstrom, 1990), and in Greenland (Dietz, 1987). Although different methods of quantification have been used in the different studies, there seem to be a tendency toward higher PCB levels in animals from Svalbard.

Table 31
Level of residual chlorinated hydrocarbons in different tissues of marine mammals from northern regions (ppm, wet wt)

Location	Year taken	Tissue, number of animals, sex	Age	∑ DDT	∑ PCB	Source
		Ringed Seal *(Phoca hispida)*				
Western Greenland	1972	blubber n = 5	-	0.12*	0.9	Clausen *et al.*, 1974
Western Greenland, Holm Isl.	1972	blubber n = 13 (M)	10.9	0.61	2.0	Addison & Smith, 1974
		n = 15 (F)		1.31	4.1	
The same	1981	blubber n = 15	-	9.6	0.58	Addison,1985 (quated by Muir, 1985)
		blubber n = 16		8.88	1.28	
W.Greenland, Upernavik	1974-1976	blubber n = 57	-	1.02	0.96	Johansen *et al.*, 1980
W. Greenland, Daneborg	1974	blubber n = 7	-	3.1	3.2	Johansen *et al.*, 1980
W. Finland, Gulf of Bothnia	1968	blubber n = 2	-	63	6.3	Jensen *et al.*, 1969
W. Finland, Bothnian Bay	1973	blubber n = 40(F)	adults	110	69	Helle *et al.*, 1976a
W. Finland, Gulf of Bothnia	1973	blubber n = 33	adults	200	110	Helle *et al.*, 1976a
W. Finland, Simo, Bothnian Bay	1975	blubber n = 37(F)	adults	115	100	Helle *et al.*, 1976b
		blubber n = 24(M)	adults	130	100	Helle *et al.*, 1976b
Finland, Gulf of Bothnia	1976-1982	blubber n=4	0.8	41	33	Pertilla *et al.*, 1986
Arctic Canada	1970	blubber n = 3	adults	2.7	3	Holden, 1972
Arctic Norway	1970	muscle n=2	adults	2.4	-	Holden, 1972
Arctic Norway	1970	blubber n=2	adults	-	1.5	Holden, 1972
W. Arctic Canada, Victoria Isl.	1972	blubber n = 13(F)	10.9	0.61	2.0	Addison & Smith, 1974
	1970-1972	bluber n = 15(M)	14.5	1.31	4.1	Addison & Smith, 1974
	1972	liver	-	0.022	0.041	Bowes & Jonkel, 1975
Sacks Harbour		n = 3(M/F)	-	0.016	0.011	
Hudson Bay, Canada		muscle n = 3(M/F)	-	1.538	0.92	
		blubber n = 5(M/F)				
Lancaster Sound, Canadian Arctic	1972	liver n = 3(F)	-	0.078	0.042	Bowes & Jonkel, 1975
		blubber n = 2(M/F)	-	0.367	0.50	

continued

Table 31, continued

Location	Year taken	Tissue, number of animals, sex	Age	∑ DDT	∑ PCB	Source
		Grey Seal *(Halichoerus grypus)*				
Archipelago of Stockholm	1968	blubber n = 3	-	36	6.1	Jensen *et al.*, 1969
Gulf of Bothnia	1968	blubber n = 2	pups	25	3.9	Jensen *et al.*, 1969
W. England, Wales	1970	blubber n = 3	adults	17.5	212	Holden, 1972
W. England, Cornwall	1970	blubber n = 3	pups	11.5	160	Holden, 1972
N. Scotland, Orkney	1970	blubber n = 8	adults	13.0	18	Holden, 1972
W. Scotland, Hebrides	1970	blubber n = 3	adults	15.1	30	Holden, 1972
W. Scotland, Summer Island	1970	blubber n = 4	adults	9.8	16	Holden, 1972
E. Scotland, Aberdeen-Montrose	1970	blubber n = 16	adults	20.1	38	Holden, 1972
N.E.England, Farne Islands	1970	blubber n = 5	pups	13.1	40	Holden, 1972
E. England, Scroby	1970	blubber n = 2	adults	39.7	123	Holden, 1972
E. Canada, Nova Scotia, Sable Island	1970	blubber n = 5	adults	45.4	27	Holden, 1972
E. Canada, Basque, Gulf of St. Lawewnce	-	blubber n = 2	adults	49.9	32	Holden, 1972
N. Scotland, Outer Hebrides	1972	blubber n = 6	adults	7.4	14.4	Heppleston, 1973
N. Scotland, Shetland	1972	blubber n = 8	adults	8.9	11.1	Heppleston, 1973
N. E. England, Farne Isl.	1972	blubber n = 7	adults	10.71	31.0	Heppleston, 1973
E. England	1972	blubber n = 4	adults	15.53	152	Heppleston, 1973
W. Finland, Gulf of Bothnia	1973	blubber n = 15	adults	210	100	Olsson *et al.*, 1975
Baltic Sea	1973	blubber n = 18	adults	420	140	Olsson *et al.*, 1975
E. Canada, Nova Scotia, Sable Island	1974-1976	blubber n = 11	pups	15.45	2.4	Addison & Brodie, 1977
Sable Island	1974-1976	blubber(F) n = 11	adults	27.1	14.5	Addison & Brodie, 1977
E. England	1975	liver n = 3	-	2.08	14.0 3.0	
		muscle n = 3	-	0.42	4.7 195	
		kidney n = 3	-	0.66		
		blubber n = 6	-	18.5		

continued

Table 31, continued

Location	Year taken	Tissue, number of animals, sex	Age	∑ DDT	∑ PCB	Source
N. Scotland, Orkney	1978	blubber n = 7	-	8.3	26.6	Holden, 1978a
W.Scotland, Shetland	1978	blubber n = 8	-	8.4	11.1	Holden, 1978a
N.E. England, Farne Islands	1975	blubber n = 26(M)	3-27	8.97	48.7	Holden, 1978a
		blubber n = 33(F)	1-36	4.26	35.4	Holden, 1978b
N.E. England, Farne Islands	1972	blubber n = 8(M)	12.9	26.8	44.7	Donkin et al., 1981
		blubber n = 9(F)	19.4	10.1	19.1	
Baltic Sea, Gulf of Bothnia	1976-1982	blubber n=5	0.4	41.0	79.0	Pertillä et al., 1986
Cardigan Bay, West Wales		blubber n=1(M)	3 weeks	1.42	17.0	Morris *et al.*, 1989
		muscle n=1(M)	3 weeks	0.02	0.47	
		liver n=1(M)	3 weeks	0.03	0.47	
		kidney n=1(M)	3 weeks	0.03	0.45	
		liver n=1	12 days	0.02	0.22	
		kidney	12 days	0.01	0.21	
N.E. England, Farne Island	1988*	blubber n=3	adults	4.2	18	Low *et al.*, 1989
Harbour seal *(Phoca vitulina)*						
Netherlands, Wadden sea	1965	blubber n= 2	-	18.5	-	Koeman and van Genderen, 1966
E.England, Sroby	1970	blubber n=3	adult	23.9	131	Holden, 1972
W. Scotland Summer Isl.	1970	blubber n=17	adults	5.3	12	Holden, 1972
N.Scotland, Shetland	1970	blubber n=4	pups	2.6	4	Holden, 1972
E.England, Wash	1971	blubber n=12	pups	6.5	15	Holden, 1972
Alaska, Pribilof Isl.	1971	blubber	-	13.9	-	Anas, 1974a
Netherlands	1972	blubber n=5	14-15	9.5	1470	Koeman *et al.*, 1972
	1972	blubber	Juv.	2.0	240	
Netherlands	1974	blubber n=3(M)	1-24	10.4	151.7	Duinker *et al.*,1979
	1974	blubber n=5(M/F)	1-24	-	16.98	

continued

Table 31, continued

Location	Year taken	Tissue, number of animals, sex	Age	∑ DDT	∑ PCB	Source
Netherlands, Wadden Sea	1979	liver n=5(M/F)	1-24	0.55	-	Duinker *et al.*, 1979
		kidney n=2(F)	1	0.41	16.3	
		blubber n=4(F)	1-23	11.2	217.8	
North Sea, North German coast	1974-1976	liver n=5	-	0.14	0.96	Drecher *et al.*, 1977
		kidney n=4	-	0.15	0.46	
		blubber n=56	-	7.3	131.5	
W.Scotland	1978	blubber n=8	-	22.3	65.8	Holden, 1978 a, b
Netherlands, Wadden Sea	1975	blubber n=6	juv.	29.8	-	Reijnders, 1980
	1976	blubber n=8	adults	47.3	-	
Southern coast of Norway, Oslofjord	1988*	blubber n=35	pups 1.0	3.1	6.6	Skaare *et al.*, 1990
Southern coast		blubber n=25	pups 1.0	3.1	11.0	
Norwegian coast		blubber n=18	pups 1.0	1.7	5.1	
Southern coast of Norway	1988*	blubber n=10(M)		-	15.0	Bernhoft & Skaare, 1994
		liver n=10(M)		-	10.0	
		kidney n=10(M)		-	6.6	
		brain n=10(M)		-	0.96	
E.England	1988*	blubber n=10(M+F)	adults	4.74	23.0	Law *et al.*,1989
		Hooded Seal *(Cystophora cristata)*				
W. Greenland, Disko Bay, Upernavik	1974	blubber n=4	-	3.5	3.9	Johansen *et al.*, 1980
W. Greenland	1972	blubber n=5	-	0.29	2.74	Clausen *et al.*, 1974
		Harp Seal *(Pagophilus groenlandica)*				
W. Greenland, Disko Bay, Upernavik	1972	blubber n=8	-	4.9	19.0	Johansen *et al.*, 1980
	1974	blubber n=3	-	1.5	1.7	Johansen *et al.*, 1980
	1976	blubber n=3	-	2.8	1.6	Johansen *et al.*, 1980
E. Canada, Gulf of St.Lawrence	1973	blubber n=4(F)	adults	4.4	6.05	Jones *et al.*, 1976
		liver n=4(F)	adults	0.03	0.07	
		muscle *n=4*	adults	0.01	0.45	

continued

Table 31, continued

Location	Year taken	Tissue, number of animals, sex	Age	∑ DDT	∑ PCB	Source
E. Canada, Gulf of St. Lawrence	1973	liver n=10(M)	8-14 days	0.129	0.33	Jones *et al.*, 1976
		kidney n=7(M)	8-11 days	0.546	0.49	
		blubber n=10(M)	8-14 days	1.34	1.26	
East of Newfoundland	1970	liver n=9	pups	0.48	0.17	Frank *et al.*, 1973
		muscle n=9	<3wks	0.43	0.21	
		blubber n=9	<3wks	3.46	1.8	
		liver n=3(F)	adults	0.18	0.08	
		muscle n=3(F)	adults	0.11	0.05	
		blubber n=5(F)	adults	8.84	4.50	
E. Canada, Gulf of St. Lawrence, Magdalen Island	1971	liver n=11(F)	adults	0.49	0.30	Frank *et al.*, 1973
		muscle n=11(F)	adults	0.20	0.14	
		blubber n=11(F)	adults	9.05	5.3	
E. Canada, Gulf of St. Lawrence, Escoumins, Quebec	1971	blubber n=4(F)	3.25	6.4	6.8	Addison *et al.*, 1973
		blubber n=14(M)	7.0	10.2	8.6	
E. Canada, Gulf of St.Lawrence	1973	blubber n=20	<2wks	1.98	2.26	Rosewell *et al.*, 1979
		liver n=10	<2wks	0.08	0.09	
	1971	liver n=9(M)	6.5	0.38	0.59	
		muscle n=9(M)	7.5	0.24	0.34	
		kidney n=10(M)	7.7	0.24	0.34	
		blubber n=10(M)	7.7	3.44	3.51	
Greenland Sea	1990	blubber n=10(F)	adult	0.68	0.92	Espeland, 1993
		blubber n=10	pups	0.59	0.62	
Bearded Seal *(Erignathus barbatus)*						
W. Greenland	1972	blubber n=5	-	0.47	1.8	Clausen *et al.*,1974

continued

Table 31, continued

Location	Year taken	Tissue, number of animals, sex	Age	∑ DDT	∑ PCB	Source
		Fur Seal *(Callorhinus ursinus)*				
Alaska, Pribilof Island	1969	liver n=5	pups	2.21	-	Anas & Wilson, 1970
		muscle n=5	pups	2.18	-	
		blubber n=5	pups	15.91	-	
Alaska, St. Paul Island	1972	blubber n=5	pups	39.9	21.9	Kurtz & Kim, 1976
		blubber n=2(F)	adults	5.2	5.8	
		Walrus *(Odobenus rosmarus rosmarus)*				
Alaska	1970	blubber n=4	-	0.08	1.8	Galster & Burns, 1972
Greenland	1975-1977	blubber n=8(M)	0-19	0.09	0.36	Born *et al.*, 1981
		blubber n=20(F)	0-19	0.05	0.18	

Notes: * found dead

Table 32

Concentrations of trace metals in marine mammals from northern regions (mg/kg wet weight).

Location, year	Tissue	N	Sex, age	Hg	Zn	Cu	Pb	Cd	Se	Source
				Ringed seal *(Phoca hispida)*						
Baltic Sea, 1967	liver	3	M	137.8	-	-	-	-	-	Henrikson & Karppanen, 1969
Finnish Bay, 1968	liver	9	F	12.5	-	-	-	-	-	
	muscle	9	F	0.9	-	-	-	-	-	
	kidney	9	F	2.4	-	-	-	-	-	
	liver	7	M	10.96	-	-	-	-	-	
	muscle	7	M	0.99	-	-	-	-	-	
	kidney	7	M	5.2	-	-	-	-	-	
W. Finland,	liver	12		91.0	-	-	-	-	35.0	Kari & Kauranen, 1978
Botnian Bay, 1974	muscle	8		1.1	-	-	-	-		
	kidney	2		4.0	-	-	-	--	-	
S. E. Finland,	liver	3		230	-	-			81.0	
Saimaa, 1974-1975	muscle	3		3.5	-	-	-	-	-	
	kidney	3		7.4	-	-	-	-	-	
	blubber	2		0.3	-	-	-	-	-	
E. Canadian Arctic, Northern Baffin Island, 1975	liver	5		3.27	41.0	8.9	0.04	4.24	-	Fallis (quoted by Wagemann & Muir, 1984)
North Sea, 1975	liver	1	F	0.64	40.0	8.94	0.24	0.31	-	Harms *et al.*, 1978
W. Greenland,	liver	10		2.4	-	-	-	-	-	Johansen *et al.*,
Upernavik, 1973	muscle	10		0.23	-	-	-	-	-	1980;
The same, 1974	liver	7		0.34	-	-	-	-	-	Johansen, 1981
	muscle	7		0.09	5.5	3.2	0.16	0.15	-	
The same, 1976	liver	31		2.1	-	-	-	-	-	
	muscle	31		0.18	-	-	-	-	-	
E. Greenland,	liver	7		2.9	-	8.1	0.03	6.6	-	
Daneborg, 1974	muscle	7		0.42	-	-	-	-	-	
E. Canadian Arctic,	liver	88		19.3	-	-	-	-	16.4	Smith &
Aston Bay, Somerset Island 1975	muscle	89		0.4	-	-	-	-	-	Armstrong, 1978
E. Canadian Arctic,	liver	27		16.2	-	-	-	-	9.4	
Barrow Strait, 1976	muscle	27		0.9	-	-	-	-	-	
The Gulf of Finland,	liver	11		11.9	37.2	3.9	0.11	0.17	8.8	Pertillä *et al.*, 1986
1976-1982	muscle	10		0.64	38.2	1.2	0.08	0.01	0.3	
	kidney	11		2.07	35.6	2.9	0.11	35.6	1.	
	blubber	11		0.04	3.52	0.22	0.05	0.003	8	
									0.13	
The Baltic	liver	10	juv	1.99	36	7.2	0.1	0.11	1.35	Frank *et al.*, 1992
	kidney	10	juv	-	21	3.9	0.08	0.78	-	
The Baltic	liver	3		44	32	4.4	0.16	0.65	19	
	kidney	4		-	26	2.8	0.11	2.12	-	
Vikna, the coast of	liver	7	F(juv)	0.45	-	-	-	-	2.53	Skaare *et al.*, 1993
Norway, 1989-1990	kidney	7	F(juv)	0.28		-	-	-	3.54	
	liver	6	M(juv)	0.39	-	-	-	-	1.81	
	kidney	6	M(juv)	0.21	-	-	-	-	3.19	
				Grey seal *(Halichoerus grypus)*						
E. Canada, Sable Isl.,	liver	6	M/F	94.5	-	-	-	-	-	Sergeant &
1973	kidney	6	M/F	1.2	-	-	-	-	-	Armstrong, 1973
East coast of Scotland,	liver	9	M/F	11.5	-	-	-	0.61	-	McKie *et al.*, 1980
1975	kidney	9	M/F	1.95	-	-	-	1.38	-	
	liver	3	M	36.8	-	-	-	0.49	-	
	kidney	3	M	2.96	-	-	-	1.76	-	

continued

Table 32, continued

Location, year	Tissue	N	Sex, age	Hg	Zn	Cu	Pb	Cd	Se	Source
East coast of Scotland	liver	11	M/F	24.4	-	-	-	-	-	
1976	kidney	11	M/F	2.45	-	-	-	1,28	-	
	liver	5	M	36.6	-	-	-	0.54	-	
	kidney	5	M	3.72	-	-	-	1.38	-	
N.E. England, Farne Isl.	liver	24	M	43.0	-	-	-	-	-	Caines, 1978
	kidney	27	M	4.12	-	-	-	-	-	
	liver	38	F	141.5	-	-	-	-	-	
	kidney	37	F	4.32	-	-	-	-	-	
North Sea, 1976	liver	1	M	19.5	61.0	20.9	0.31	0.02	-	Harms *et al.*, 1978
Baltic Sea, the Gulf	liver	9		99.7	54.8	10.7	0.17	0.04	26.8	Pertillä *et al.*, 1986
of Finland,	muscle	9		1.5	35.0	1.2	0.07	0.004	0.4	
1976-1982	kidney	8		9.5	34.1	2.4	0.1	0.6	2.9	
	blubber	9		0.2	3.9	0.3	0.1	0.002	0.1	
Cardian Bay,	blubber	1	M	0.05	4.7	<0.1	<0.6	<0.06	-	Morris *et al.*, 1989
West Wales, 1988	muscle	1	M	0.78	43	2.5	<0.6	<0.06	<0.06	-
	liver	1	M 3 weeks	1.7	80	26	<0.6		-	
	kidney	1	M 3 weeks	1.1	44	27	<0.6	<0.06	-	
	liver	1	12 days	2.5	88	9.1	<0.7	<0.07	-	
	kidney	1	12 days	1.3	29	3.2	<0.7	0.08	-	
The Baltic	liver	10	juv	15	46	15	0.25	0.1	6.31	Frank *et al.*, 1992
	kidney	10	juv	-	22	2.8	0.15	0.93	-	
The Baltic	liver	9		26	46	14	0.18	0.18	14	
	kidney	9		-	23	2.6	0.11	1.26	-	
Jarfjord, the coast	liver	2	F(juv)	0.78	-	-	-	-	1.19	Skaare *et al.*, 1993
of Norway,	kidney	2	F(juv)	0.12	-	-	-	-	3.18	
1989-1990	liver	18	F(ad)	22.36	-	-	-	-	12.57	
	kidney	18	F(ad)	1.45	-	-	-	-	2.69	
	liver	6	M(juv)	3.55	-	-	-	-	3.01	
	kidney	6	M(juv)	0.49	-	-	-	-	2.70	
	liver	7	M(ad)	9.18	-	-	-	-	5.79	
	kidney	7	M(ad)	1.62	-	-	-	-	2.73	
Vikna, the coast of	liver	3	F(juv)	9.23	-	-	-	-	5.50	
Norway, 1989-1990	kidney	3	F(juv)	1.66	-	-	-	-	2.56	
	liver	2	M(juv)	2.93	-	-	-	-	2.06	
	kidney	2	M(juv)	0.85	-	-	-	-	2.67	
	liver	2	M(ad)	18.16	-	-	-	-	11.35	
	kidney	2	M(ad)	1.96	-	-	-	-	3.40	
Harbour seal *(Phoca vitulina)*										
Alaska, Pribilof Islands,	liver	3	M/F	4.23	-	-	-	-	-	Anas, 1974 a
1971										
E. Canada, Sable Island,	liver	4	M/F	2.91	-	-	-	-	-	Sergeant &
1973	muscle	4	M/F	0.71	-	-	-	-	-	Armstrong, 1973
	blubber	4	M/F	0.04	-	-	-	-	-	
Netherlands, Wadden	liver	2	F	55.5	-	-	-	-	-	
Sea, 1974-1976	kidney	1	F	4.9	-	-	-	-	-	
	blubber	1	F	0.08	-	-	-	-	-	
	liver	1	M	55.0	-	-	-	-	-	
	kidney	1	M	6.3	-	-	-	-	-	
	blubber	1	M	0.07	-	-	-	-	-	

continued

Table 32, continued

Location, year	Tissue	N	Sex, age	Hg	Zn	Cu	Pb	Cd	Se	Source
North Sea,	blubber	3*		-	3-14	0.9-3.0	<00.5-1	<0.01-0.02	-	Duinker *et al.*, 1979
West Wadden sea	placenta	1*			11	2	<0.05	<0.01	-	
	liver	8*		-	16-64	2-20	<0.05-2.3	0.03-0.21	-	
	kidney	2*		-	15-25	4.8-5.1	0.16-0.23	0.15-0.17	-	
	brain	7*		-	8-27	2.5-9.5	<0.05-2	<0.01-0.14	-	
	spleen	2*		-	26-31	3.3-4.4	0.16-0.4	0.04-0.09	-	
	heart	2*		-	31	5.8-8.2	0.29-0.61	0.06-0.47	-	
North Sea,	blubber	5		-	4-13	9-23	10-12	0.9-1.4	-	Holden , 1975
East England	liver	5		-	43-61	-	-	-	-	
	kidney	5		-	28-51	-	-	-	-	
	brain	5		-	19-36	-	-	-	-	
	spleen	5		-	28-35	-	-	-	-	
	heart	5		-	28-32	-	-	-	-	
The North Sea,	liver	57		-	27-56	2.6-17	0.1-0.57	0.01-0.2	-	
East Wadden Sea	kidney	16		-	16-32	2.3-4.0	0.14-0.55	0.06-0.38		
	brain	16		-	11-15	2.4-4.0	0.04-0.20	0.002-0.014		
North Sea,	liver	20		-	-	-	2.31	0.2-0.8	-	Roberts *et al.*, 1976
East England	kidney	9		-	-	-		-	0.1-0.6	
Swedish west coast,	liver	10	juv	3.56	36	9.3	0.12	0.04	2.04	Frank *et al.*, 1992
Skagerrak, 1988	kidney	10	juv	-	19	3.5	0.04	0.21	-	
Kattegat	liver	10	juv	2.42	35	12	0.08	0.04	2.07	
	kidney	10	juv	-	21	3.6	0.04	0.23	-	
Kalmarsund, the Baltic	liver	10	juv	0.44	28	4.0	0.10	0.02	1.02	
	kidney	10	juv	-	21	3.3	0.07	0.10	-	
Skagerrak, 1988	liver	10	juv**	2.84	36	5.2	0.06	0.03	2.35	
	kidney	10	juv**	-	30	6.4	0.04	0.32	-	
Måkläppen, the Baltic,	liver	4	juv**	1.16	45	5.7	0.09	0.03	1.11	
1988	kidney	4	juv**	-	34	5.4	0.04	0.22	-	
Skagerrak	liver	8		26	54	8.6	0.16	0.09	11	
	kidney	8		-	29	4.5	0.05	0.46	-	
S. Norway, Oslofjord, 1988	liver	28*		1.2	47.0	6.1	-	<0.1	1.6	Skaare *et al.*, 1990
Southern coast, 1988	liver	26*		8.8	46.0	5.7	-	<0.1	6.5	
Northwestern coast, 1988	liver	17*		1.5	60.0	11.0	-	<0.1	1.7	
Jarfjord, the coast of	liver	4	F(juv)	0.30	-	-	-	-	1.76	Skaare *et al.*, 1993
Norway, 1989-1990	kidney	4	F(juv)	0.23	-	-	-	-	2.86	
	liver	2	F(ad)	0.83	-	-	-	-	3.73	
	kidney	2	F(ad)	0.19	-	-	-		2.80	
	liver	4	M(juv)	0.49	-	-	-	-	2.13	
	kidney	4	M(juv)	0.21	-	-	-	-	4.45	
	liver	1	M(ad)	0.54	-	-	-	-	1.85	
	kidney	1	M(ad)	0.33	-	-	-	-	2.95	
Vesterålen, the coast	liver	6	F(juv)	6.85	-	-	-	-	4.54	
of northern	kidney	63	F(juv)	0.85	-	-	-	-	5.67	
Norway, 1989-1990	liver	3	F(ad)	1.96	-	-	-	-	2.22	
	kidney	3	F(ad)	0.89	-	-	-	-	3.85	
	liver	5	M(juv)	6.68	-	-	-	-	4.6	
	kidney	5	M(juv)	1.06	-	-	-	-	5.79	
	liver	2	M(ad)	10.48	-	-	-	-	5.60	
	kidney	2	M(ad)	1.72	-	-	-	-	5.94	

continued

Table 32, continued

Location, year	Tissue	N	Sex, age	Hg	Zn	Cu	Pb	Cd	Se	Source
Hooded seal *(Cystophora cristata)*										
E. Canada, Magdalen Isl.,	liver	3	M	37.2	-	-	-	-	-	Johansen *et al.*,
Gulf of St.	muscle	3	M	1.28	-	-	-	-	-	1980
Lawrence, 1971	blubber	3	M	0.08	-	-	-	-	-	
W. Greenland,	liver	4		6.5	-	-	-	-	-	
Upernavik, 1974	muscle	4		0.20	-	-	-	-	-	
1976	liver	10		16.7	-	-	-	-	-	
	muscle	10		0.33	-	-	-	-	-	
Harp seal *(Pagophilus groenlandicus)*										
E. Canada, Escoumins,	liver	2	M/F	1.29	-	-	-	-	-	Sergeant &
Quebec, 1971	muscle	2	M/F	0.24	-	-	-	-	-	Armstrong, 1973
	blubber	2	M/F	0.03	-	-	-	-	-	
W. Greenland,	liver	7		1.2	-	-	-	-	-	Johansen *et al.*,
Unamak, 1972	muscle	12		0.20	-	-	-	-	-	1980
W. Greenland,	liver	11		2.3	-	-	-	-	-	
Upernavik, 1973	muscle	11		0.24	-	-	-	-	-	
1976	liver	4		0.86	-	-	-	-	-	
	muscle	4		0.20	-	-	-	-	-	
N. W. Greenland,	liver	26		8.92	-	-	0.08	4.44	5.99	Ronald *et al.*, 1984
1976-1978	muscle	42		0.31	-	-	0.07	0.15	0.22	
E. Canada,	liver	6	0.5 yrs	0.50	-	-	-	-	-	Botta *et al.*, 1983
Newfoundland, 1980	liver	6	1 yrs	1.67	-	-	-	-	-	
	liver	6	2 yrs	2.42	-	-	-	-	-	
	liver	6	3 yrs	3.96	-	-	-	-	-	
	liver	6	4 years	3.08	-	-	-	-	-	
Jarfjord, the coast	liver	10	F(juv)	0.29	-	-	-	-	2.01	Skaare *et al.*, 1993
of Norway, 1989-1990	kidney	10	F(juv)	0.17	-	-	-	-	3.78	
	liver	8	F(ad)	0.33	-	-	-	-	1.55	
	kidney	8	F(ad)	0.16	-	-	-	-	1.94	
	liver	8	M(juv)	0.39	-	-	-	-	2.28	
	kidney	8	M(juv)	0.16	-	-	-	-	3.93	
	liver	8	M(ad)	0.44	-	-	-	-	2.09	
	kidney	8	M(ad)	0.22	-	-	-	-	2.66	
Walrus *(Odobenus rosmarus)*										
N. W. Greenland,	liver	46		1.78	-	-	-	-		Born *et al.*, 1981
1975-1977	muscles	58		0.08	-	-	-	-		
Beluga Whale *(Delphinapterus leuca)*										
Hudson Bay 1969	muscle	1		0.97	-	-	-	-		Bligh &
	liver	1		8.87	-	-	-	-		Armstrong, 1971
	brain	1		2.44	-	-	-	-		
1971	muscle	43		0.53	-	-	-	-		Lutz, Armstrong,
Kugmallit Bay,	liver	7		6.26	-	-	-	-		(quoted by Muir,
Mackenci River Delta,	muscle	7		0.71	-	-	-	-		1985)
1972										
1977	liver	8		30.62	-	-	-	-	-	Muir, 1985
	muscle	11		2.12	-	-	-	-	-	
	blubber	11		0.08	-	-	-	-	-	
North Sea, 1976	liver			-	32	20.4	0.36	0.9	-	Harms *et al.*, 1978
	kidney			-	29.5	3.1	0.13	1.9	-	
	muscle			-	20.0	1.1	0.08	0.007	-	

Note: * - found dead;
** - epizootic juvenils

5 Conclusions

The Arctic and sub-Arctic marine ecosystems are the least polluted areas in the world, with generally low values of OCs and heavy metals, when compared to temperate and tropical areas. Based on the contamination levels in marine organisms including seabirds and seals, the Barents Sea area is less contaminated by OCs than the North Atlantic, the Baltic and the North seas. However, the analysis of data available on contamination levels in the Arctic and sub-Arctic marine ecosystems show that the systems are contaminated with a whole range of contaminants.

The main pathways of contamination into Arctic and sub-Arctic marine ecosystem are atmospheric transport, ocean currents and rivers. In some Arctic areas dumping and ship accidents are the main input of contaminants to the marine ecosystem.

The analysis of literature data indicate a lack of data (both OCs and heavy metals) from several trophic levels, especially lower levels, in Arctic marine ecosystem. In invertebrates and phytoplankton there is little or no data on OCs and heavy metals from the Arctic and sub-Arctic seas. Geographically there is a lack of comprehensive data from the Russian Arctic, especially east of the Barents Sea.

The analysis of data also show that it is very difficult to compare recent or present data on contamination levels (especially OCs) with previous studies since the demand for quality assurance (both for sampling and analysis) has improved. In order to evaluate future data, one should group the data according to the quality (no or little description is given for sampling and analytical procedure).

The literature analysis indicate that little effort has been made to investigate the biological effects of contaminants at the cellular level, at individual or population levels in the Arctic or sub-Arctic marine ecosystem. There is a need for studies which are related to uptake, biotransformation and excretion of environmental contaminants in Arctic and sub-Arctic animals. Since most animals that live and breed in the Arctic and sub-Arctic areas rely on stored body fat to survive during periods of little or no food availability, most concern should be devoted to the lipid-soluble contaminants. OCs are transferred through the marine ecosystem to top-predators such as Glaucous gulls, seals and polar bears. In the north west part of the Barents Sea these species have been found with very high levels of OCs (10-20 times higher than in the Canadian Arctic). At the present, there is little information on the behaviour, ecology, physiology and biochemistry of the marine species in this area. There is also very limited comparative data on the effects of toxaphene and chlordane on animals. There is a need for a congener-specific PCB analysis to better understand the potential effects of PCB exposure in these areas.

In the future, there is a need for international co-operation both for studies of the contamination levels in the atmosphere and biosphere. In order to handle these data obtained in the Arctic and sub-Arctic areas, the Arctic Monitoring and Assessment Programme (AMAP) program will have an important function to harmonize and evaluate the data obtained. An important function of the AMAP program would also be to indicate gaps in our knowledge and possible future problems in Arctic ecosystems. The following information is still missing, as mentioned on the 5th AMAP Working Group Meeting in Tromsø, 3-5 March 1994:

- There is lack of reliable determinations of levels of organochlorines in sea water and ice. Furthermore, a concern about the quality assurance of existing data has been expressed.

- Little is known about the presence of pollutants incorporated in sea ice as well as in particulate matter transported to coastal Arctic seas.

- Information is missing about the relationship between bioaccumulation of organochlorines, lipid production and type as well as seasonal changes of the lipid content and composition in biota.

- The knowledge of food chain relationships (benthic and pelagic) in marine water environments is limited.

- Little is known about antagonistic and synergistic effects of different contaminants.

- The knowledge is limited about the metabolisation of toxicants in the marine biota.

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