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Pollution of the Norwegian Arctic: A review

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Pollution of the Norwegian Arctic:
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Zbigniew Jaworowski
Norsk Polarinstitutt
Rolfstangveien 12
1330 Oslo Lufthavn

ABSTRACT

Studies on the levels of pollutants (acids, heavy metals, radionuclides, organic compounds and gases) in the air, precipitation, soil, plants and animals which were carried out in the Norwegian Arctic are critically reviewed and compared with those performed in other regions. The sensitive parts of Norwegian Arctic, bioindicators and regional effects of the expected climatic warming are also shortly discussed. On the basis of this review research projects are proposed for short- and long-term studies of pollution in the Norwegian Arctic.

There are four types of sources of pollutants in the Arctic environment: (1) local natural, (2) local anthropogenic, (3) remote natural and (4) remote anthropogenic. It appears from the reviewed studies that it is difficult to define what is the contribution of each of these types of sources to the total level of pollution in various compartments of the arctic environment. In a large part of these studies it was tacitly accepted that the arctic environment was originally extremely pure. Therefore currently observed levels of pollutants, which are rather high in the Norwegian Arctic, are in these papers interpreted as caused by the fourth type of sources, i.e., anthropogenic one.

However, the most important argument for this supposition, i.e., an increase in the levels of pollutants in recent precipitation, as compared with the pre-industrial one (preserved as annual ice and firn layers in the arctic and high altitude glaciers) is lacking or indecisive. In the case of heavy metals a vast majority of measurements shows no temporal trend in their concentrations in the annual ice strata, not only in the Arctic but all over the world, except for locations in the immediate vicinity of the industrial emission sources (e.g. in Central Europe).

Some of the long-term studies indicate that the acidity of the arctic precipitation depended on volcanic eruptions in remote regions, and that it was higher in the past than now. In some short-term studies a recent increase in acidity of precipitation was found, but not in the others. In a group of papers showing a recent increase of content of acids in snow and ice, some of the results for recent precipitation reflected probably the influence of the local power installations, rather than global or hemispheric situation.

Studies of the levels of radionuclides, metals and major ions in the contemporary and ancient precipitation, deposited in the arctic glaciers, are of great importance for understanding the changes in the global and Northern Hemisphere environment. It seems that these substances do not presently pose any serious hazard for the biota in the Norwegian Arctic. On the other hand organic compounds, might possibly accumulate in some arctic species to a harmful level. This supposition, however, needs substantiation by further studies.

A climatic warming could have beneficial effects in the arctic biota. On the other hand it might possibly release vast amounts of "greenhouse gases" from the frozen bogs and peats, thus triggering a feedback mechanism, which could enhance the global warming.

Svalbard is a region exposed to a greater atmospheric transport of impurities from distant natural and man-made sources than some other arctic areas. Therefore it is proposed that the Svalbard Environmental Monitoring Station in Ny Ålesund, which is supposed to be little influenced by the local anthropogenic emissions, should attain a status of a global reference station.

The following main projects are proposed for the short- and long-term research in the Norwegian Arctic:

(1) Assessment of deleterious effects of man's activity in the arctic biota.

(2) Historical monitoring of the past and present levels of pollutants. This will enable a prompt estimate of the contribution of anthropogenic emissions to: (a) the total flow of impurities into the global atmosphere, and (b) the contamination of the arctic biota.

(3) Long-term monitoring of pollutants in the air, precipitation, seawater and the terrestrial and marine biota.

(4) Short-term basic research on:

  (a) Migration of impurities in the glacier ice and snow and its implications for the methodology of glacier pollution research.

  (b) Enrichment of metals in the atmosphere and hydrosphere.

  (c) Radioactive contamination of the Arctic by the nuclear explosions at Novaya Zemlya.

  (c) Natural production of the chlorinated hydrocarbons and its impact on the analytical quality of the determinations of synthetic halogenated organic compounds in the arctic environment.

  (d) Emission of volatile chlorinated and brominated hydrocarbons into the atmosphere from natural sources in the Arctic Ocean.

  (e) Transfer of pollutants in the marine and terrestrial arctic foodchain.

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1 INTRODUCTION

The arctic environment is influenced by the man-made and natural contaminants which are from both local and remote sources of pollution. After several decades of research, the quantitative estimates of contributions from these four types of sources to the total contamination of the Arctic are far from being unequivocally resolved.

Probably the easiest to monitor and control are the local anthropogenic sources of pollution, which lead to an acute but spatially limited degradation of arctic ecosystems. These include human settlements, mining and automotive traffic (Låg, 1980; 1987; 1988; Krzyszowska, 1985).

The natural local and regional emissions of pollutants in the Arctic have been virtually unstudied. They are due to such phenomena as volcanic activity, resuspension of heavy metals from weathered rock surfaces, wind transport of sea salts from bursting spray bubbles, and production of volatile organometallic compounds and of chlorinated and brominated hydrocarbons by bacteria and algae in the terrestrial and marine environment. Some of these phenomena may have an important impact at the planetary scale. They may contribute to the natural flow of metals into the atmosphere (Donard and Weber, 1981; Jaworowski et al., 1981). The algae in the Arctic Ocean produce organobromine compounds (Dyrssen and Fogelquist, 1981), which, after entering the atmosphere, may cause cyclic destruction of the stratospheric ozone layer in March and April, through the BrO_x radicals produced from them by photochemical reactions (Barrie et al., 1988, Wever, 1988). It is also possible that an increased exhalation of these species from the seawater after clearing the icepack in the spring may contribute to this phenomenon.

Most of these natural regional and local sources of pollution probably have no adverse effects on the arctic ecosystem, as they were formed long enough ago for adaptation of the biota and for developing a stability of the ecosystem.

By contrast the anthropogenic long-range pollution of the arctic environment could lead to more rapid changes at a large geographical scale. These changes might be of long duration, and their effects may be difficult to mitigate. The arctic ecosystem is characterized by low species diversity, a low level of plant production, and intricate relations between plant cover, energy exchange at the surface, and stability or instability of the underlying frozen ground. This system is regarded as very vulnerable, as it channels all the energy between plants and carnivores through one key link, or through a very few herbivores with different predators (Dunbar, 1973; Larsen, 1985). Fragility of the tundra is directly proportional to the ice content of the permafrost and inversely proportional to the mean annual ground temperature (Ives, 1970). The apparent stability of this system and its survival over longer periods is probably due to the very large spatial scale involved, which provides for the mending of regions of perturbation by repopulation from undisturbed areas. The large geographical scale of man-made pollution from the distant emission source regions in Eurasia and North America may interfere with this basic factor of stability of the arctic ecosystem.

Anthropogenic pollutants from the distant sources of emission may rapidly (i.e., in days or weeks) reach the Norwegian Arctic via the atmosphere and more slowly (up to several years) with the Gulf Stream. The principal atmospheric pollutants which may enter the Arctic are particulates containing a wide spectrum of trace metals from coal, oil and wood combustion, pyrometallurgical production of non-ferrous metals, steel and iron manufacturing, refuse incineration and cement production (Nriagu and Pacyna, 1988), strong acids (H_2SO_4, HNO_3, HCl), toxic organic compounds such as hexachlorobenzene, α- and γ-hexachlorocyclohexane, penta-, hexa-, hepta-, and octachlorobiphenyls, α- and γ-chlordane, hepta-chloroepoxide. 2,4-DDE, 2,4 DDT and 4,4-DDT (Pacyna and Oehme, 1987), soot and other light-absorbing aerosols, radionuclides from nuclear explosions and nuclear accidents, and natural radionuclides from the burning of fossil fuels. The infrared absorbing gases (such as CO_2, O_3, CH_4, N_2O, CCl_3F, CCl_2F_2), which may contribute to climatic warming, are also transported into the Arctic with air masses travelling from distant source regions. Some of these gases, e.g., CH_4 and CO_2 may have their source in the Arctic, large enough to influence the global balance.

The most important marine pollutants reaching the Norwegian Arctic are probably the organic toxic compounds. They reach the arctic environment with the Gulf Stream, into which they enter with river waters in Europe, polluted by industrial effluents and agricultural use of pesticides, and by the air transport of combustion products and resuspended pesticides.

The supposed man-induced global warming, which according to several theoretical models is expected to be more pronounced in the Arctic than in the other parts of the world, may trigger a feed-back mechanism, by emission of methane from the thawing permafrost. The released methane, one of the "greenhouse gases", has been suggested to accelerate global warming (Pearce, 1989). A present methane stored in tundra bogs contributes 1 to 5 million tons per year, i.e. <1 % to the total flow of this gas into the global atmosphere (Iansiti and Niehaus, 1989).

The graphitic carbon found in high concentrations in the arctic haze may also change the reflectivity of the arctic clouds and the ice caps,and add to the "greenhouse effect" (Rahn and Borys, 1977; Heleman, 1983).

Vast amounts of biomass ($\sim 1.3 - 2.5 \times 10^{16}$ $g.y^{-1}$) exposed to natural and man-made burning introduce into the global atmosphere great quantities of "greenhouse gases", i.e. CO_2 ($2 - 4 \times 10^{15}$ $g.y^{-1}$), N_2O (~ 1.3 z 10^{13} $g.y^{-1}$), CH_4 ($\sim 6 \times 10^{13}$ $g.y^{-1}$), and an extremely efficient stratospheric ozone depleting agent CH_3Cl (4.2×10^{11} $g.y^{-1}$) (Crutzen et al., 1979). Heating of the enormous masses of topsoil organic matter and unburned above-ground biomass may also make important contributions to the cycling of these atmospheric trace gases (Crutzen et al., 1979).

The impact of these man-made sources of pollutants may have a cumulative character, such as, e.g., changing the buffering capacity of soils and oligotrophic lakes and increasing the content of heavy metals and other toxic pollutants in the arctic environment to a level that could lead to adverse ecological effects in the tundra biosystems. These biosystems are extremely sensitive to chemical perturbations. Because of this, on the one hand, the changes in the arctic ecosystems may be used as the early indicators of environmental impacts of long range transport of anthropogenic pollutants. On the other hand, one may expect aggravation of the environmental impacts in the Arctic as compared with other regions, with the possibility of transfer of some arctic animals (e.g., predatory birds and polar bear) to the endangered species category. In this respect polychlorinated hydrocarbons (PCH) and especially polychlorinated biphenyls (PCB) seem to be the most dangerous pollutants of the arctic environment.

The existence of a special lichen - reindeer food chain in the Arctic may cause exposure of some of the animals and man to increased levels of radionuclides. Lichens are aerophytic plants that concentrate much more atmospheric pollutants than non-aerophytic ones (Gorham, 1953). Due to this effect, during 1963 - 1966, i.e., after a period of the most intensive nuclear weapons testing in the atmosphere, radioactive cesium and strontium were accumulated in human populations of the Arctic in much higher quantities than in those of other parts of the world (Miettinen et al., 1966; Svensson and Liden, 1965; Hanson, 1986). Concentrations of ^{137}Cs were between hundred and thousand times higher in arctic populations than in those of the middle latitudes (UNSCEAR, 1982). Recently, after the Chernobyl accident, this phenomenon caused economic losses in some regions of Scandinavia and led to much higher contamination of southern Lapps than other parts of the Norwegian population (Baarli, 1988).

The arctic lichen - reindeer food chain also leads to highly increased concentrations of radioactive polonium (^{210}Po) and lead (^{210}Pb) in man (Holtzman, 1966; Kauranen and Miettinen, 1969; Blanchard and Moore, 1970; Persson, 1972; Persson, 1973). It is possible that concentrations of other radionuclides, e.g. ^{239,240}Pu, stable lead and other heavy metals also increase along this pathway. The dependence of the arctic populations on the marine food chain may lead to an increased intake of toxic heavy metals (Grandjean, 1989).

The natural distant sources may contaminate the arctic environment with many of the same types of pollutants as are emitted from man-made sources. This is the case with heavy metals, acids, soot and other particulates and natural radionuclides from uranium and thorium chains and halogenated hydrocarbons. Among the natural sources of these pollutants, transported from large distances to the Arctic, are quiescent volcanic emissions and eruptions, which inject vast amounts of gases and particulates into lower stratospheric altitudes. This material can be dispersed over the whole Earth and may remain in the atmosphere for several years after a volcanic event (Hofmann and Rosen, 1987).

Particulate matter from such natural source regions as Asiatic and African deserts has been found to be transported in the atmosphere over many thousands of kilometers (Prospero et al., 1981; Parrington et al., 1983; Betzer et al., 1988). The natural dust resuspension, forest fires and volatilizations of metals, which leads to their high enrichment in airborne dust, may contribute to the contamination of the Arctic from the Eurasiatic and American continents. Seawater - air interface phenomena and biogenic emissions from the ocean probably add to this contribution as well.

It is of great importance to determine the relative contributions of man-made and natural sources to the total contamination of the Arctic with particulates, acids, heavy metals, radionuclides, "greenhouse" gases and halogenated hydrocarbons.

This information is needed in order to understand the ecological effects that might be expected in the Arctic due to anthropogenic contamination, and in order to take the practical steps needed to prevent and mitigate these effects by such abatement strategies as e.g., implementation of regulations addressing target and critical loads of pollutants.

The same information is also needed for understanding global effects. The Arctic is remote from industrial centers and other regions of anthropogenic emission of pollutants. Therefore the changes in the levels of pollutants in various compartments of the Arctic environment reflect hemispheric and global processes. The temporal changes in these levels, related to the trends in industrial and other activities of man, can be used for quantitative assessment of the anthropogenic contribution to the global cycles of pollutants. As will be seen later, this problem is far from being unequivocally elucidated.

The majority of studies of the pollution of the arctic environment has been carried out in continental parts of Greenland (see e.g. Alderson and Coleman, 1985). These studies were usually limited to measurements of concentrations of dust, heavy metals, SO_2, NO_3, NH_4, Na, Cl, Ca, K, Mg, H+ and radionuclides in glacier ice, precipitation and air. In the past the levels of pollutants in particular compartments of the Norwegian Arctic environment were studied on a rather limited scale. In 1977 systematic and sophisticated studies of air contamination have been undertaken by Norsk Institutt for luftforskning (Ottar et al., 1986 A). Measurements of pollutants in snow, plants and animals were also started in this region.

The air concentrations of heavy metals recorded in the air in Svalbard were higher than in Greenland (Heidam, 1986; Ottar et al., 1986A; Heintzenberg, 1987). The concentrations of heavy metals and major ions in the glacier ice and snow from Svalbard were also much higher than in Greenland and in northern Norway (Svartisen) and similar to those in the temperate zone of Europe (Alps) (Briat, 1978; Fjerdingstad et al., 1978; Jaworowski et al., 1981; Semb et al, 1984; Batifol and Boutron, 1984; Pacyna et al., 1985). This may be caused by the patterns of long-range atmospheric transport of pollutants to the Svalbard region from Eurasia and North America, related to different heat regimes and meteorological conditions over the Greenland ice cap and Spitsbergen, as well as to the distribution of land masses around the Arctic Ocean. Over the eastern Atlantic sector of the Arctic the low pressure systems may pass into the Arctic Ocean without being hampered by large land masses. They may transfer the impurities in air from North American sources to the central Arctic, without passing over central Greenland (Rahn and Show, 1982; Heidam, 1984). The patterns of the average transport of Eurasiatic air masses coming in from southeastern directions expose Svalbard to greater pollution from midlatitude sources than the continental parts of Greenland, as the transport routes rarely pass directly over Greenland (Heidam, 1984). This indicates that the changes in environmental pollution observed in Svalbard are probably more representative for hemispheric processes than those found in Greenland. It is therefore astonishing that, compared with the scale of pollution studies in Greenland, the research in Svalbard was relatively neglected.

It seems that the studies on hemispheric changes to be carried out in Svalbard should receive a greater emphasis and a higher priority in the programs addressing such themes as documenting and predicting global environmental changes, dominant forcing functions, and interactive phenomena in the Earth biospheric system. In this respect the environmental studies in the Norwegian Arctic should concentrate on priority issues recognized in the Brundtland Report: "Our Common Future, A Plan for Sustainable Development", which deals with "key interactions and significant changes on the time scales of decades and centuries, that most affect the biosphere, that are susceptible to human perturbation, and that will most likely lead to practical, predictive capability".

The aim of this review is to identify the areas of environmental studies in the Norwegian Arctic, a further elaboration of which will help in protecting the arctic ecosystems and will enable quantification of the estimates of the global impacts of anthropogenic pollution.

2. LEVELS OF POLLUTANTS

2.1. AIR

2.1.1 Introduction

The studies of the pollution of the arctic atmosphere were started during the International Geophysical Year in 1957 by measurements on Spitsbergen of the concentrations of CO_2 in the air (Jaworowski, 1958; Birkenmajer, 1962).

Since 1960 systematic ground level measurements of fission products and radionuclides of natural origin were initiated at Thule and at other locations in Greenland and in Franz Joseph Land, and also the high altitude aerosol sampling was performed in the Arctic by several American teams (Stebbins, 1961; Aarkrog et al., 1963; Patterson et al., 1964; Vilenskii et al., 1967). The results of these measurements were used for assessment of the global impact of the nuclear test explosions (UNSCEAR, 1962; UNSCEAR, 1982).

The studies of non–radioactive particulate pollutants in the arctic air were started in the seventies. With the notable exception of systematic long–term measurement of SO_2 and SO_4, carried out at Ny Ålesund, Spitsbergen and at Bjørnøya between 1977 and 1984 (Ottar and Pacyna, 1986 A), the measurements of chemical (non–radioactive) components in arctic air have been based on short–term sampling programs. One year observations were performed at six coastal locations in Greenland during 1979 and 1980 (Heidam, 1984). At Spitsbergen the ground level measurements were based on a sampling program limited to 3 winter months in 1957/1958 (Jaworowski, 1958), a few weeks in winter or summer in 1977, 1978, 1979, 1983, 1984 and 1985 (Pacyna and Ottar, 1985; Maenhaut et al., 1989), 20 days in April and May 1979, and 12 days in March 1981 (Heintzenberg, 1987); also 35 samples of aerosols were collected at various oceanic sites between Franz Joseph Land and Greenland (Heintzenberg, 1982). At Jan Mayen cadmium, zinc, nickel and lead concentrations were measured during 12 days in summer 1983, and SO_4 concentrations between July and June 1982 (Pacyna and Ottar, 1985).

The measurements of SO_2 and SO_4 at Ny Ålesund were the longest of all non–radioactive pollutants studies in the Arctic, and they revealed typical seasonal variations, but no long–term trend which might be related to the global annual rate of increase of fossil fuel production in this period of about 3 % (GUS, 1987) (Figure 7).

A meticulously elaborate model was recently constructed by Iversen (1980) to simulate long–range atmospheric transport of sulfur into the Arctic. It takes into account both horizontal and vertical transport through the isentropic surfaces, chemical reactions, and other meteorological parameters. The model enables computing the concentration of SO_2 and SO_4 at 1 m, 40 m, 1000 m, 2000 m, 3000 m, and 5000 m altitudes. The model may become a powerful tool in further studies of the arctic air pollution, and may possibly be modified for other types of pollutants and for higher altitudes.

2.1.2 Radioactivity

One of the important results of the early studies of atmospheric impurities was finding the seasonal variations in the tropospheric content of radioactive pollutants from nuclear explosions, and of natural radionuclides, with winter and spring maxima and summer minima (Peirson et al., 1960: UNSCEAR, 1962; Rangarajan et al., 1976). These variations were found all over the world and were not related to the intensity of nuclear tests. In the Arctic the seasonal variations in fission products fallout were observed systematically at several coastal locations in Greenland from 1962 until recently (Aarkrog et al., 1963; Aarkrog, 1979; Aarkrog et al., 1988).

The same phenomenon was also observed in the case of a natural radionuclide, lead–210, the dominant source of which is the exhalation of its gaseous radon–222 parent from the surface of the lands. This natural radionuclide ascends high in the troposphere, and its concentrations in the lower stratosphere are similar to those at the ground level (Kownacka et al., 1989). In the Arctic, the March–April maximum of the natural lead–210 concentration in the ground level air, an order of magnitude higher than the summer minimum was found at Barrow, Alaska (Daisey et al., 1981). An order of magnitude seasonal variations of lead–210 concentrations in the ground level air, with winter maxima and summer minima, were found both at high and low latitudes (Joshi and Mahadevan, 1968; Nevissi and Schell, 1980). These variations are not related to anthropogenic sources, which contribute only about 0.001 fraction to the global flow of this nuclide (Jaworowski et al., 1981). Nevissi and Schell (1980) supposed that lead–210 accumulates in the upper troposphere and lower stratosphere in summer and early fall, and in the winter and spring it descends into the lower parts of the troposphere. Their model is similar to the reservoir model proposed by Heidam (1984) for non–radioactive pollutants in the Arctic.

The seasonal variations of radionuclides were consistent with the Brewer–Dobson model of circulation of tropospheric air, and with its later modifications (UNSCEAR, 1962), as well as with the Hadley cell circulation and eddy transport across the tropopause (Reiter, 1975) (Figure 1). It was also suggested that the perturbations of the electric field associated with the geomagnetic activity could modulate the winter downdraft in the stratosphere, which may move particles into the tropospheric altitudes (Hale, 1977). In the Arctic the spring maxima of pollutants in the air coincide with the minimum amount of precipitation in April and May (Vowinckel and Orvig, 1970; Steffensen, 1982). This suggests that the decrease in washing out effect may lead to an increase in air concentration of pollutants. Show (1981) suggested that the seasonal variations in the arctic cloudiness may be a factor controlling the content of impurities in the arctic air. Also Hofmann and Rosen (1987) suggested that continued production of sulfate particles may be taking place at stratospheric altitudes in the polar regions, where stratospheric warming may cause aerosol evaporation and recondensation during winter and spring. Strong seasonal variations in content of non–radioactive impurities in Greenland ice from around 1400 A.D. with early spring maxima and summer minima were reported by Langway et al. (1978), which indicates that this is not a new, man–made phenomenon.

The other important observation was that the radioactive fallout at the coastal locations in southern Greenland was only by a factor of about 0.5 to 0.8 lower than in Denmark, and that the population of Greenland was showing a greater sensitivity to environmental contamination with cesium–137, due to a special food–chain. In effect, the average radiation dose of 1.6 mSv accumulated by the members of this population from nuclear weapon tests through 1975 (per caput effective dose equivalent commitment) is the same as in the population of Denmark (Aarkrog, 1979).

From this point of view the coastal parts ofthe Greenland environment cannot be regarded as a typical "remote region". This is also valid in the case of heavy metal pollution of the Inuit population (Grandjean, 1989).

The fission products studies in the Arctic indicated that this region is exposed to contaminations from the distant source regions, such as Nevada, Novaya Zemlya, Semipalatinsk, Lob Nor and even south hemispheric nuclear weapons testing grounds. A part of these radioactive impurities travelled at stratospheric altitudes (UNSCEAR, 1982; Kownacka et al., 1989), with trajectories strikingly similar to low altitude trajectories ascribed for non–radioactive pollutants (Figure 2 and 3). It is astonishing that the early findings in the behaviour of radionuclides in the environment are not being used for interpretation of the more recent observations of the long distance transport of the non–radioactive pollutants in the Arctic. One of the striking examples of this neglect is that they are not used for interpretation of seasonal variations in non–radioactive pollutants in the arctic air, which in the recent literature are usually assigned to the changes in anthropogenic combustion processes (Rosen and Novakov, 1981).

2.1.3 Carbon dioxide

In the early studies carbon dioxide concentrations in the atmosphere were reported to vary between 210 and 500 ppm due to large diurnal and annual variations in which the burning of fossil fuels, photosynthetic processes and soil breathing are invoved (Landsberg, 1975). The use of various analytical techniques and standards also contributed to such a wide scatter of readings. The International Geophysical Year (IGY) stressed the importance of accurate analyzes of atmospheric trace gases important to the global heat balance, and a program to monitor carbon dioxide at locations removed from sources of combustion was organized in 1956 (Fritz et al., 1959).

In the frame of IGY the measurements of CO_2 content in the air were carried out at Hornsund, Spitsbergen between October and December 1957 using a basic titration method (Birkenmajer, 1962). The recorded average concentration at Hornsund was 398 ppm (Jaworowski, 1958), i.e., higher than that measured by infrared analysis at Barrow, Alaska during the same months twenty years later (334 - 340 ppm) (Peterson et al., 1982). Wet chemical techniques, such as the one used at Hornsund, were found to not be adaptable to long–term monitoring and the infrared analysis was found more proper for long–term studies and intercorrelation among several monitoring stations (Kelley, 1974). The high values measured at Horsund were probably due to analytical problems or to local contamination at Hornsund by burning of the heating oil.

The systematic measurements of carbon dioxide concentration in the near surface arctic air were started in 1961 near Barrow, Alaska by Kelley (1974). The concentrations observed in the Arctic between 1973 and 1979 in the Arctic were about 2.5 ppm higher than at Mauna Loa, Hawaii. This was probably due to anthropogenic CO_2 releases in mid–latitudes of the Northern Hemisphere, natural oceanic or terrestrial sources, greater in the northern part of this Hemisphere, or due to greater CO_2 sinks in the southern part, or due to significant filtration of this gas from the ocean through the sea ice to the atmosphere (Petersen et al., 1982). Distinct seasonal variations of carbon dioxide were found at Barrow, with maximums in winter and minimums in summer. The low concentrations in the summer reflect the uptake of carbon dioxide from the atmosphere by vegetation. After 1963 the secular trend of 1.0 ppm was found at Barrow, which is close to the trends noted at Mauna Loa, Hawaii, and the Amundsen–Scott South Pole Station. The annual variarion was much greater in the Arctic, however, than at the two other localities (Kelley, 1974). In Svalbard the systematic measurements of this important "greenhouse gas" have not yet started.

2.1.4 Heavy metals, major ions and organic compounds

More than ten years after monitoring of radioactive substances and of CO_2 was started in the Arctic the first measurements of the non-radioactive particulate pollutants were carried out in 1971 in Greenland by Flyger et al. (1973), who measured the concentrations of 17 elements in the ground level air at a coastal location, and of Aitken nuclei at various tropospheric altitudes (see also Heidam, 1986). The air concentrations of metals and major ions were then determined near Barrow, Alaska, in 1976 by Rahn et al. (1977), in the Arctic haze bands, at an altitude of 3.3 km. In the Norwegian Arctic this monitoring was started in 1977 at Bjørnøya and Ny Ålesund, with particulate sulfate and sulfur dioxide as the main subjects (Larssen and Hanssen, 1980), followed by further studies, also including heavy metals (Rahn et al., 1980; Heintzenberg et al., 1981) and by an expounded Norsk Institutt for Luftforskning program of ground level and aircraft measurements (Ottar et al., 1986 A).

These studies showed that the concentrations of metals in the air in Alaska, Greenland and Jan Mayen was about an order of magnitude lower than on Spitsbergen, and that the Spitsbergen values were similar to those in rural regions of Norway and Sweden (Pacyna and Ottar, 1985; Ottar et al., 1986A; Heintzenberg et al., 1981), (Table 1).

The air concentrations of toxic chlorinated organic compounds (α–HCH, HCB, and γ –HCH), measured during shord periods in the Norwegian Arctic are similar to to those in remote locations in Colorado, South Carolina, Texas, Southern Norway and over the Pacific, Indian, and Antarctic oceans (Ottar et al., 1986 A) This indicates that the pesticides and other organic chemicals of industrial origin, introduced into the environment in tropical and mid–latitude regions are distributed rather evenly in all the global atmosphere, due to meso– and global scale air transport processes. These substances undergo a degradation after exposure to UV radiation. In the Arctic this effect may be expected to be much less important than in the lower latitudes. The low average annual temperatures in the Arctic may lead to a greater deposition of these substances with condensation nuclei and by other mechanisms. The resuspension (due to evaporation) of these substances deposited on the ice–free parts of the Arctic, may be impaired in low temperatures. Therefore one may expect that in the Arctic the airborne toxic organic pollutants may accumulate to a greater extent than in other regions.

Because of a special type of the arctic food–chain this may lead to much more serious ecological consequences than in other regions, including a possibility of extinction of some arctic animals (e.g. predatory birds, polar bear and possibly insects), and to pathological changes in the freshwater and marine biota. The accumulation of organic toxic pollutants might possibly impair the arctic fishery and have a negative health and economic consequences for some local arctic populations. One should note however, that toxic chlorinated organic compounds are produced in great amounts by some marine biota, and that it may be difficult to distinguish them from synthetic ones introduced into environment by man.

Brominated organic species (CH_3Br, CH_2BrCH_2Br, $CHBr_3$, CH_2Br_2), which were found to be more effective than chlorine compounds in the catalytic destruction of stratospheric ozone (Wofsy et al., 1975) were detected in the arctic air at ground level (in Ny Ålesund, Spitsbergen and Barrow, Alaska) in concentrations comparable to those in Bodø, Norway. The mid–tropospheric and stratospheric concentrations were found to be higher than at the ground level (Berg et al., 1984). The concentrations of particulate and gaseous bromine in the arctic troposphere were found to be ten times higher than all previously reported levels in the natural troposphere (Berg et al., 1983). These authors proposed as the most likely sources of airborne arctic bromine (1) the long-range anthropogenic pollution from Eurasia (as a part of haze phenomenon) and (2) biogenic emissions on a large scale by marine organisms. The second proposition is in agreement with later studies (Fogelquist, 1985; Barrie et al., 1988; Wever, 1988).

2.1.5 Arctic haze

Around 1977, there emerged a problem of the arctic haze, interpreted as an effect of anthropogenic pollution. Many years before this, the Artic haze was noticed by pilots flying over the Alaskan Arctic poleward of 70°, and was first studied by Mitchell (1956) in relation to the impairment of visibility. Mitchell supposed that the arctic haze is due to copious local combustion of fossil fuel and the discharge of spent steam from central heating systems in arctic settlements. A later study of arctic cloudiness between 1920 and 1978 was performed, aiming to determine if there has been long term monotonic increase or decrease of cloudiness in the Arctic due to the "continual increases of anthropogenic pollutants" from distant emission sources. This study has revealed no significant trend that might be assigned to increasing air pollution (Raatz, 1981), and certainly not to the trends in the world fossil fuel burning. During this period, the world fossil fuel burning increased more than four times (Möller, 1984), and the world industrial production increased about seven times between 1932 and 1968 (UNDESA, 1957 and 1970). This indicates that the arctic haze is an old phenomenon unrelated to the recent increase in industrial activity.

Other long-term observations, such as conductivity of the air over the South Pacific, turbidity values at Mauna Loa, Hawaii and in Antarctica, and world wide variations in atmospheric transparency based on solar observations, reveal no evidence of any increase in particulate loading of the global atmosphere which might be due to anthropogenic sources (see review in Ellsaesser, 1975).

It was Rahn et al. (1977) who for the first time analyzed the chemical composition of the arctic haze. At Barrow, Alaska they observed between 12 April and 5 May, 1976, at 3.5 km altitude, about one order of magnitude fluctuations of the air concentrations of Al, Mn, V, Na and Ba. They also noticed that the values of an enrichment factor (EF) for the elements Mn, V, Na and Ba (X) relative to Al in the airborne dust as compared with the same in the Earth's crust:

$$EF = \frac{(X/Al)\ \text{aerosol}}{(X/Al)\ \text{Earth's crust}}$$

fluctuated in this period, not consistently with the temporal changes in air concentrations of these elements. The higher EF values they found together with low concentrations of elements in the air in the "non-haze period", and the lower EF in the samples collected from typical haze layers with about ten times higher concentrations of impurities. The higher EF values for vanadium in the aerosols of about 2 and 5 in the "non-haze-period" were assumed by these writers to be pollution derived. But they stated that the haze aerosols (associated with high concentrations of studied elements and with low enrichment factors) were natural, i.e. just the opposite of what they expected. They concluded that the source of the arctic haze must be very strong and that there are natural dust emissions from the Takla Makan and Gobi deserts of eastern Asia. This conclusion was probably correct, as numerous later studies indicate that there exists a vast scale long-range transport of mineral particulates from Asiatic and African continents, some of them having "gigant" area-equivalent diameters (>75 μm), and reaching distances more than 10,000 km (Glaccum and Prospero, 1980; Prospero et al., 1981; Hashimoto et al., 1983; Parrington et al., 1983; Uematsu and Duce, 1985; Betzer et al., 1988).

However, Rahn et al. (1979) and Rahn et al. (1980) refuted this hypothesis, and stated that the aerosols in the arctic haze are "strongly pollution derived", particularly in winter. In the ground level air samples, collected later at Barrow, Alaska they observed a winter maximum of SO_2 air concentrations and an increased enrichment factor for vanadium, reaching values up to 13. The sulfate concentrations were sometimes higher than in the eastern United States, and were not of local origin (Rahn and Shaw, 1982). This was surprising to the authors who, like many others, believed that the pristine arctic environment is not much influenced by chemical processes in the global atmosphere. This assumption on the natural purity of the arctic atmosphere confronted with the high content of impurities found in the arctic air, was the basis of their hypothesis on anthropogenic origin of the arctic haze and its chemical components.

This hypothesis was in conflict with results of earlier studies of atmospheric aerosols in the Arctic, which indicated that the industrial pollution and combustion products are not significant constituents of the arctic air (Junge, 1960; Kumai and Francis, 1962; Fenn and Weickmann, 1959; Fenn, 1960).

Three arguments were offered for support of the anthropogenic origin of the arctic haze aerosols:

(1) Enrichment of heavy metals, especially of vanadium, in the aerosols collected from the arctic haze which is higher than in the Earth's crust;

(2) Very strong seasonal variation in concentration of the arctic haze particles, which are an order of magnitude more abundant in winter than in summer; and

(3) Presence of graphitic carbon and high concentrations of sulfate in the arctic air.

Further studies demonstrated that the origin of the arctic aerosols is mainly the Eurasiatic continent, with much less contribution coming from North America (Heidam, 1984; Barrie, 1986; Ottar et al., 1986; Maenhaut et al., 1989).

Perhaps the most important argument for the anthropogenic origin of the arctic haze is the enrichment of vanadium. Rahn and Shaw (1982) stated that the enrichment factor for vanadium provides an extremely sensitive test of anthropogenic pollution vs. natural character of an aerosol. They assumed that its value below 1.5 may be considered natural and above 1.5 definitely shows influence of pollution. However, this is not in agreement with the results of studies indicating that many heavy metals, including vanadium, are enriched by up to several orders of magnitude in the airborne dust, both over industrialized areas and in remote ones, such as the South Pole, Central Greenland and mid-oceanic localities. It was suggested that this enrichment is due to natural processes (Duce et al., 1977; see also a review in Jaworowski et al., 1981).

Also in the precipitation preserved in annual ice strata in glaciers in Greenland, Antarctica, Norway, the Alps, Africa, the Himalayas, Alaska, the Andes and Spitsbergen the enrichment factors for metals ranged from about ten to tens of thousands. In the Arctic the enrichment factors of metals are almost identical to those observed in Antarctica which are presumed to be of natural origin (Alderton and Coleman, 1985) (Table 2). In the pre-industrial period these enrichment factors in precipitation were as high as at present (Jaworowski et al., 1981; Boutron, 1986; Alderton and Coleman, 1985) .

The content of vanadium in the ice from Spitsbergen (Jaworowski et al., 1981) and from Greenland (Herron et al., 1977 B) did not reveal an increasing temporal trend, and in the average global precipitation its level was found to be lower now than in the pre-industrial period (Jaworowski et al., 1981), probably due to decreased volcanic dust veil in this century (Lamb, 1970). In Greenland ice the values of EF for vanadium were lower in 1971/73 than in the pre-1900 ice samples (Herron et al., 1977). As may be seen in Table 2, EF for vanadium may reach in natural emissions values up to two orders of magnitude above 1.5 assumed by Rahn and Shaw (1982) to be the natural limit.

The presence of sulfate and soot in the arctic haze may be both of natural and industrial origin. About 50% of the gaseous sulfur emissions into the global atmosphere is estimated to be contributed from man-made sources (Schwartz, 1988). It is still an open question how much this contribution changed the sulfur level in the arctic precipitation during the last 100 years. Such historical monitoring is needed to check the current estimates of the relative contributions of man-made and natural sources of sulfur into the global atmosphere.

Still less is known about historical trends in the content of soot in the arctic atmosphere. The glacier ice in the Arctic, Antarctica and in the temperate and tropical zones contains black sooty particulates in the strata reaching far in the pre-industrial period (Murozumi et al., 1969; Jaworowski, 1986). The quantitative assessment of the temporal changes in the level of soot in the glacier ice would be an excellent indicator of the relative contribution of man-made sources to the total atmospheric flow of this pollutant. But this study yet remains to be done. In addition to fossil fuel burning, the biomas burning in the tropical regions may be an important source of soot carbon in the global atmosphere, as indicated by the finding that its air concentration over mid-atlantic sites is comparable to that over the British Channel (Andreae, 1983).

Some estimates indicate that release of carbon into the global atmosphere from terrestrial biomass (1.8 - 4.7 x 10^{15} g y^{-1}) is comparable with those from fossil fuel burning (4.5 - 5.9 x 10^{15} g y^{1}); 80% of carbon release from the biomass is due to deforestation, principally in the tropics and this contribution may reach 9 x 10^{15} g y^{-1} (Woodwell et al., 1983). It has been suggested that this may become a major factor in the process of man-triggered warming of climate.

Until 1960 the annual release of carbon from the biomass exceeded the release from fossil fuels (Woodwell et al., 1983). As the excess (non-marine) potassium to soot carbon ratio was found to be an indicator of the relative contributions to soot carbon aerosols of biomass and fossil fuel burning (Andreae, 1983), measurements of this ratio in the glacier ice strata may be used to estimate the trends in man-made pollution with carbon of the global atmosphere.

This discussion of the arctic haze studies can be summarized as follows:

There is little indication that the enrichment of metals in general and of vanadium in particular could be used to define the anthropogenic origin of aerosols in remote sites; it cannot be used as an argument in support of a hypothesis that the arctic haze is predominantly of anthropogenic origin.

The seasonal variations of content of non-radioactive pollutants in the arctic air are clearly related to patterns of horizontal transport of atmospheric air masses in the northern hemisphere, and probably also to the vertical transfer of aerosols from stratospheric pool, as is the case with nuclear test debris. These meteorological phenomena are responsible for transporting into the arctic region air masses containing heavy metals and major ions of both natural and anthropogenic origin. The precipitation patterns in the Arctic may also contribute to the spring maxima of pollutants in the air. Seasonal variations in their air concentrations, also occur at lower latitudes (Lannefors et al., 1983). They cannot be used as an indicator of the anthropogenic influence on the pollution of the Arctic.

The very presence of soot and sulfur in the arctic aerosols is not a good indicator of the industrial origin of the arctic haze.

The historical monitoring (of glaciers, lacustrine sediments etc.) is probably the best approach to solving the "riddle" of the origin of the arctic haze.

The general impression from reading the "arctic haze literature" is that after finding relatively high levels of metals, sulfate and soot in the arctic air, the initial tacit assumption that the pristine arctic environment is extremely pure and separated from the influences from strong natural sources of atmospheric impurities, lead to a rather premature assignment of an anthropogenic origin of this phenomenon. This hypothesis remains to be documented, but the numerous studies which it stimulated produced valuable information. These data together with the results of future studies aimed at quantitative assessment of relative contributions of man-made and natural sources to the arctic haze phenomenon, will be of great importance for evaluation of the Northern Hemispheric and global impact of man's activity.

Rahn and Shaw (1982) stated that the long-range transport of aerosols contained in the arctic haze, is concentrated at unusually low altitudes, below 3 km and often at or below 1.5 km. This is not in agreement with the observations of the arctic haze covering large areas at an altitude of about 8.5 km (Hileman, 1983) and with the results of measurements of vertical distribution of arctic haze pollutants carried out during 35 sampling flights over Svalbard in 1983 and 1984 by the Norwegian Institute for Air Research (NILU) (Ottar et al., 1986 B; Pacyna and Ottar, 1988). In this study the high concentrations of light scattering particles and of metals and major ions were observed at various altitudes, reaching the upper flight limit of about 5 km. Substantial concentrations of graphitic carbon particles were found throughout the arctic troposphere up to 8 km altitude (Rosen and Hansen, 1984). These authors found that within particular air layers the concentrations were as large as in typical urban air in the United States. Their results indicated significant transport of soot particles at high altitudes above the planetary boundary layer. High concentrations of atmospheric impurities were observed in lower latitudes at high tropospheric and lower stratospheric altitudes over Wyoming, USA, in 1974 and 1975 (Hofmann and Rosen, 1977), and over Central Europe in a period from 1973 to 1987 (Kownacka et al., 1989).

The average vertical cross section of concentrations of impurities in the atmosphere over Central Europe (Figure 4) is similar to that over Wyoming. Such vertical distribution of air impurities is in agreement with Slim's (1982) suggestion that part of the impurities available for the global-scale transport in the atmosphere enters the stratosphere in the mid-latitudes. Then the impurities descend to low altitudes in the arctic air at the polar side of the northern Hadley cell (Figure 5). The stratospheric pool for metallic impurities and sulfuric and nitric acids was inferred from high altitude measurements (Jaworowski and Kownacka, 1976; Heitman and Arnold, 1983; Hofmann and Rosen, 1987; Arnold and Knapp, 1989; Kownacka et al., 1989). All this indicates that the observed levels of impurities in the arctic air are probably an effect of large-scale transport processes, existing in this region since times immemorial. The anthropogenic contribution to these levels remains to be determined.

Relating the concentrations of impurities observed in the arctic air to particular anthropogenic source regions, on the basis of concentration ratios of particular species or on the basis of their distribution in various particle size fractions (Heidam, 1984; Pacyna et al., 1984 A) seems to be at present hypothetical and needs confirmation by future studies.

It seems that the long-range transport of impurities into the arctic atmosphere is due not only to processes in the planetary boundary layer as proposed by Rahn and Shaw (1982), but that it is associated with large-scale global circulation involving air masses in all tropospheric and lower stratospheric altitudes. This is in agreement with the striking similarity of long-term trends of lead concentrations in ground level air at Ny Ålesund, Spitsbergen (Pacyna and Ottar, 1985; Maenhaut et al., 1989) and of its concentrations in the total air column between ground level and 15 km over Poland (Kownacka et al., 1989). This similarity indicates that the temporal fluctuations of the lead concentrations had wide geographical dimensions and that lead in aerosols from so widely dispersed regions was of common origin. This trend was also similar to the trend in aerosol concentrations in the stratosphere over Wyoming, USA (Hofmann and Rosen, 1987). These concentrations were decreasing from maximum values (presumably after the Fuego volcano eruption in 1974) in 1977 to low values about 1985, and then again increasing in 1986 (in coincidence with Nevado del Ruiz eruption in November, 1985) (Figure 6).

2.1.6 Conclusions

(1) The future long-term observations of the vertical distribution of impurities in the arctic atmosphere, including high tropospheric and low stratospheric altitudes, will be needed for checking the predictions by modeling. They will be of great importance for understanding the long-range transport of pollutants in the global atmosphere and will help in identifying their sources. The current studies, limited mainly to the ground level and short-term low altitude sampling of aerosols are not well fit for this task, even with the use of advanced methods of determination of chemical "source signals".

These sampling programs inadvertantly bias the interpretation of the results of these studies in two ways:

(a) the major transport of impurities into the Arctic is often assumed to be limited to the boundary layer, and the air trajectories made for identification of the sources of observed air concentrations of impurities are usually calculated for altitudes between 1.5 and 3 km (Rahn et al., 1977; Barrie et al., 1981; Pacyna and Ottar, 1985), and

(b) the sources of pollutants are usually assumed to be the regional industrial ones, and continental, hemispheric and global contributions are neglected. However, evidence exists that these long-range contributions may also reach the arctic environment via much higher altitudes and that a substantial part of them may be natural.

(2) Except for SO_2 and SO_4^{2-} measurements of other pollutants in the air in the Norwegian Arctic are too scarce to allow for any direct correlation to be made between them and temporal changes in industrial activity in the Northern Hemisphere.

(3) The concentrations of impurities in the arctic air were found to be relatively high. Higher values were usually found at Spitsbergen than in other arctic locations. Concentrations of some of them were similar to those in rural regions of Scandinavia and other parts of Europe, North America, in mid-latitudes in the Southern Hemisphere (see e.g., Nriagu, 1980 A, and Nriagu, 1980 B), and were higher than in continental Antarctica.

(4) The low altitude trajectories indicate that the main sources of impurities in the arctic air are in Eurasia and Northern America. But the current studies do not provide any evidence which unequivocally identifies the respective contributions from anthropogenic and natural emission sources, to the long-range transport of impurities into the arctic atmosphere.

(5) The use of enrichment factors for metals in the arctic aerosols cannot be regarded as an indicator of their anthropogenic origin. Also the ratios of concentrations of particular elements in aerosol samples and their distribution in various size fractions of particles, have only a hypothetical character and are of uncertain value for detailed identification of emission sources of arctic pollutants.

(6) Long-term observations of the temporal trends of contami-nants in the arctic air would provide the best opportunity for a quantitative estimate of the relative contributions of the man-made and natural sources to the arctic pollution with heavy metals, mineral acids and radionuclides. Most of the sampling programs for airborne pollutants in the Norwegian Arctic were until now of too short duration for this aim.

2.2 PRECIPITATION

2.2.1 Introduction

Measurements of contamination of arctic precipitation were started in 1955 by determination of non–radioactive pollutants (Na^+, K^+, Mg^{2+}, Ca^{2+}, HCO_3^-, Cl^- SO_4^{2}+, NO_3^- and pH) in snow at Nordaustlandet, Svalbard by Gorham (1958). It was found that the average concentration of SO_4^{2-}in the samples from strata below 3 m depth was about 1000 µg kg^{-1}. In a sample of old ice from a depth of 14 m Gorham found the highest concentrations of SO_4^{2-} of 2800 µg kg^{-1}. The near surface average value of Gorham was twice as high as 562 µg kg^{-1} found in surface snow 28 years later by Semb et al. (1984) in the same part of Svalbard, and similar to that in continental Norway. Gorham (1958) found lower concentrations of NO_3^- (10 to 15 µg kg^{-1}) and lower acidity (pH 5.5 to 6.7) than those found by later studies in this area.

The level of gross ß–radioactivity in 1957–1958 precipitation in Svalbard at Hornsund (Jaworowski, 1959; Wisniewski, 1959) and Murchison Bay (Lugeon, 1960) was found to be similar to that in Central Europe. In 1961 radioactivity measurements (^{90}Sr and ^{137}Cs) were initiated at Godhavn, Greenland (Aarkrog et al., 1963) and are continued at several coastal locations in Greenland until now (Aarkrog et al., 1988).

In Nordaustlandet Gjessing (1977) measured in 1974–1975 winter snow concentrations of Mg^{2+}, Cl^-, SO_4^{2-}, NO_3^-' NH_4^+, Ca^{2+} and also pH and conductivity. He found total SO_4^{2-} values ranging between 30 and 900 (aver. 370) µg kg^{-1}, NO_3^- between 5 and 85 (aver. 30) µg.kg^{-1} and pH 4.78 to 5.49 (aver.5.20). His average NO_3^- value was lower and pH value higher than those of Semb et al.(1984) of 60 µg kg^{-1} and 4.53, respectively.

In Western Spitsbergen 29 trace elements were analyzed in two surface snow samples by Fjerdingstad et al. (1978) who found the concentrations of lead of 7 and 48 µg kg^{-1}, i.e., up to several orders of magnitude higher than in continental Greenland and similar to those in rural regions of England and the Soviet Union (see reviews in Jaworowski et al., 1981 and Galloway et al., 1982), and at high altitude in the French Alps (Batifol and Boutron, 1984).High concentrations of other metals were also found in these samples, e.g. >3 µg kg^{-1} of Ni, 8 µg kg^{-1} of Cu, >50 µg kg^{-1} of Zn, 10 µg kg^{-1} of Sn, 100 µg kg^{-1} of Ti, <5 µg kg^{-1} of Cr and 1300 µg kg^{-1} of Fe. The high content of metals in the Fjerdingstad et al. (1978) samples could be due to a presence of algae in snow, or due to exposure of the small patches of snow from which the samples were collected to the soil dust from an abandoned mine area.

In contrast, very low concentrations of heavy metals (Cd 0.0003 – 0.0006 µg kg^{-1}; Pb 0.008 – 0.02 µg kg^{-1}; Cu 0.010 – 0.025 µg kg^{-1}; Ni 0.013 – 0.040 µg kg^{-1} and Ci <0.005 µg kg^{-1}) were found by Mart (1983) in the snow collected in 1980 from the ice floes at several sites between North Greenland and Frans–Josef–Land. Several times higher values were found in a few days old snow in the same region and about ten to twenty times higher concentrations were recorded in the snow remaining from the 1979/1980 season.

Concentrations of SO_4^{2-} were determined in snow deposited between September 1982 and April

1983 in 24 sites in Svalbard (Semb et al., 1984). The highest concentration of 562 $\mu g.kg^{-1}$, found at Austfonna, Nordaustlandet, was similar to concentrations recorded from Northern Norway (Ottar et al., 1986 A). Concentrations of H^+, NO_3^-, NH_4^+, Na, Mg and Ca were also measured at two locations (Hornbreen and Gråhuken. The concentrations of SO_4 found in the western part of Spitsbergen were similar to those found in 3 samples of a 1 cm thick surface layer of snow from continental Greenland collected in July and August, 1978, of 120–219 $\mu g\ kg^{-1}$ (Davidson et al., 1981).

In continental Greenland, Boutron (1979) measured concentrations of Na, Mg, Ca, Fe, Al, Mn, Pb, Cd, Cu, Zn, and Ag in twenty samples of surface snow. The concentrations of metals were lower by one or more orders of magnitude than in rural regions in the Northern Hemisphere (Galloway et al., 1982). This is in variance with the geographical distribution of SO_4^{2-} and NO_3^-, which seem to be homogeneously disperssed over the whole planet both now and during pre–industrial periods (see discussion of glacier historical monitoring below). This imply that the magnitude of local natural sources of metals influence more their concentrations in precipitation than is the case with SO_4^{2-} and NO3.

2.2.2 Historical monitoring of acid rain

Existence of "acid rains" in the industrial areas has been known since the middle of the 19th century (Smith, 1852). But it is only recently that it has been recognized that acid rain is a global scale phenomenon, and that it appears in widely dispersed areas remote from industrial centers, such as Mauna Loa, Hawaii, the central Indian Ocean and the South Pacific. In these remote sites fairly high acidity of rainwater was recorded, with pH values of about 4.5 and even as low as 3.7 (Hawaii) (NOAA, 1979; Kerr, 1981).

The majority of measurements of trace metals and major ions in the arctic precipitation was carried out in the ice and snow samples (cores or blocks of firn and ice) collected for studies of historical changes of levels of pollutants in the global atmosphere. The first of such studies were those of Junge (1960) and of Georgii and Weber (1962), who collected relatively shallow firn and ice strata. The first deep core study of pollutants was carried out at four sites in continental Greenland (Site 2, Camp Century, Tuto and Dye 3) by Langway (1967).

Junge (1960) measured SO_4 concentrations in Greenland firn and ice covering the period 1917 – 1957. Analysis of the samples obtained from a 30 m deep pit revealed no increase in SO_4 concentrations over the period studied. Junge explained this by the fact that, industrial SO_2 has only a few days residence time in the troposphere, and it is washed out before reaching the Arctic.

Georgii and Weber (1962) determined Na, Ca, Cl, SO_4^{2-}, NO_3^-, NH_4^+ and pH in snow and ice cores from Axel Heiberg Island, covering the past 40 years. The results showed no significant variations through time of SO_4 concentrations.

Langway's (1967) results from "Site 2"in Greenland indicated that SO_4 concentration in 120 A.D. ice of 40 $\mu g\ kg^{-1}$ was lower than in 930–1920 A.D. ice (420 $\mu g\ kg^{-1}$), but the ice from 1915–1957 had a significantly lower SO_4 concentration (250 $\mu g\ kg^{-1}$) than the 930–1920 ice. At Camp Century site the ice from 1810–1920 had 2.8 times higher content of SO_4 (470 $\mu g\ kg^{-1}$) than the 1963–1964 sample (170 $\mu g\ kg^{-1}$). Langway (1967) concluded that his results indicated that industrial exhausts do not appreciably contribute to the arctic air masses.

This is in variance with his and his coworkers' later statement (Finkel et al., 1986) based on compiling results of analysis of three 20th century core samples from Dye 3 site and one 1450 B.C. sample, together with 15 samples from other studies from the same site (Herron, 1982; Herron and Langway, 1985).

The sulfate concentrations in the 1920–1980 data set from Dye 3 compiled by Finkel et al. (1986) ranged from 52 to 87 μg kg^{-1}, and the 8400 B.C. to 1832 A.D. concentrations ranged from 17 to 30 μg kg^{-1}. Finkel et al. (1986) stated that there was a recent increase in the sulfate and nitrate concentrations, which they attributed to anthropogenic sources. However, they ignored, without explanation, the result of Langway's (1967) sulfate determination in 1963–1964 ice from Dye 3, of 30 μg kg^{-1}, i.e. the same as that from 180 A.D. and lower than in other prehistoric samples. They did neither take into account contributions from major volcanic explosions in sixties and seventies, which might explain the observed variations, nor the local effects of the Dye 3 radar station which was constructed in 1959.

Federer (1970) concluded from his study of an ice core from station Jarl–Joset, Greenland, that no anthropogenic input of sulfate had occurred during the last hundred years, when the global consumption of fossil fuel increased tenfold.

Koide and Goldberg (1971) observed a threefold increase in non–marine sulfate concentration in post–1960 ice from Camp Century, Greenland, as compared with a pre–1930 mean value. The rise of sulfate concentrations in post–1960 samples coincided with the replacement at Camp Century of nuclear power generating equipment by diesel generators in 1964 (Murozumi et al., 1969). One should also note that the post–1960 samples were taken from a site 80 km apart from the place where the pre–1930 samples were collected. More than a 500% difference in sulfate concentration was reported for the samples from the same period but from various locations at the Greenland Ice Cap (Langway, 1967). Therefore one may doubt whether the pre–1930 and post–1960 samples in the Koide and Goldberg (1971) data set are comparable. Moreover, in their conclusion the authors also do not take into account the possible influence of the 1963 Gunung Agung volcanic eruption.

Weiss et al. (1971 A) reported results of sulfur determination in 13 ice samples from Greenland and one from Antarctica. They stated that their data indicate an increase in sulfur in ice since 1960 which they "uniquely" attributed to the fossil fuel combustion at the global scale. But this is not supported by a closer analysis of their results. Weiss et al. (1971 A) did not describe the analytical procedure for sulfur determination and stated that their Greenland samples are from two sites at Camp Century: older samples from an "inclined shaft at Camp Century" and the younger ones (1952–1965) from "open trenches at the virgin site". This was not exactly so. These same samples are described in detail in the Murozumi et al. (1969) paper, and it appears that they are from: (1) Tuto, a place 19 km from an airfield of great American military base at Thule, about 200 km W from Camp Century (a sample from 800 B.C.); (2) Camp Century where the American military base was permanently inhabited by several hundreds persons, with great automotive traffic and power generators (samples from 1815, 1859, 1881, 1892 and 1946); (3) a trench situated 80 km ESE of the Camp Century military base (samples from 1952, 1960 and 1964); and (4) the samples from 1965 are of uncertain origin, as they may be from trenches at 80 km , 45 km or 1 km ESE from the military base. The samples from 800 B.C., 1815, 1859, 1881, 1892 and 1946 were collected from walls of several hundred meters long tunnels excavated by U.S. Army for various military aims. In these tunnels automotive traffic was rather intense and they can hardly be treated as "pristine" or "remote" sites.

As indicated by Dickson (1972) the temporal correlations in this set of Greenland samples, covering a large geographic area, are of doubtful value, due to a great geographical variability in heavy metal concentrations in Greenland. (See also criticism in Alderton and Coleman, 1985, which is also valid for determinations of lead in the same set of samples by Murozumi et al, 1969, discussed below).

In eight samples from the vicinity of the military base at Camp Century, probably collected from sites between 1 to 80 km from the base, covering a period 1946–1965 the average concentration of sulfur of 56.1 $\mu g\ kg^{-1}$ was almost exactly the same as in a 800 B.C. sample from Camp Tuto of 56 3 $\mu g.kg^{-1}$. These values are higher than those in the nineteenth century group of samples (27.3 $\mu g\ kg^{-1}$), and the authors interpreted them as an evidence of a recent input by man. However, these samples might be contaminated by the local sulfur emissions from the Camp Century, Tuto and Thule military bases, and the observed concentrations are not necessarily related to the long-term hemispheric trends but rather to local disturbances.

Cragin et al. (1975) again studied the ice cores from the Dye 3 site in Greenland (where a radar station is located) and found a twofold increase of sulfate concentration in post–1840 ice as compared with pre–1840 samples. They attributed this to be due to long-range transport from man-made sources.

A similar increase in sulfate and nitrate concentrations was reported by Herron (1982) from North Central, Camp Century and Dye 3 sites in Greenland. This author observed a correlation between known old volcanic eruptions and concentration peaks for sulfate in his old ice samples. He also stated that the 1975–1981 peak at Dye 3 site is due to combustion of fossil fuels, primarily coal. However, it may be that the increase was caused by the local contamination effects from the Dye 3 radar station or by effects of the Fuego eruption in 1974 and La Soufriere, Sierra Negra, Mt. St. Helens and other volcanic eruptions in 1979–1981. The 1975–1981 concentration of SO_4^{2-} of 85 $\mu g\ kg^{-1}$ in his Dye 3 ice core may be related to this late volcanic activity, and it is the same as in ice from 1000 A.D. and lower than in a 2000 years sample of >120 $\mu g\ kg^{-1}$.

Mayewski et al.(1986) measured sulfate and nitrate concentrations in an ice core collected near the Dye 3 site in South Greenland (altitude 2615 m above see level), covering the period 1869 – 1984. The concentrations of sulfate and nitrate were measured in small core sections after removing 40 – 50% of the external part of the core. They found a threefold increase in non-marine sulfate concentrations since around 1900 and of nitrate since 1955. However, the highest sulfate concentration in their data set appeared in 1885, i.e., after the Krakatau eruption and the increase in sulfate and nitrate concentrations after 1960 coincide with both the construction of a radar station at Dye 2 and with Gunung Agung, Fuego, Mt. St. Helens and El Chicon eruptions.

Neftel et al.(1985) measured nitrate and sulfate concentrations in the firn cores from the Dye 3 site, 2 km from the radar station, after removing external layers of the cores. They reported a twofold increase in concentration of both species in the recent snow. However, the increased sulfate and nitrate concentrations appeared in their data set after 1960, i.e. after construction of the Dye 3 radar station in 1961, and after the Gunung Agung eruption in 1963. Previous early increases in their data set coincided with the Mt. Pelêe (1902) and Katmai (1912) eruptions, and with the increased Lamb's volcanic dust veil index (v.d.v.i.). The decreases during the 1930s coincided with the low v.d.v.i. values (Lamb, 1970). The highest concentration of nitrate was observed in a 1906 sample. It seems that Neftel et al.'s (1985) conclusion that "human activity drastically changes the atmospheric load of the acid trace species NHO_3 and H_2SO_4" is not sufficiently supported by their data, and that they more correctly state that "more widespread drilling of firn cores in polar regions would allow the historical development of the acid deposition pattern to be established".

The opinion that there was a man-made increase of acidity in the recent ice and snow in Greenland is at variance with the results of the specific conductivity measurements in an ice core from Cretê, Central Greenland (Hammer, 1977 A). This author concluded that little, if any industrial dust reaches this site. He found that the acidity of ice in the period 1920 – 1972 was lower than around 1910 and than in several decades of the 19th and 18th centuries. Hammer et al. (1980) presented the same results as Hammer (1977) for the 20th, 19th and 18th century in addition to the values for much older ice from Cretê, indicating that between the 6th and 16th century the acidity of precipitation was much higher than in the 1920-1972 ice. Extremely high acidity of precipitation was found in the same study in the ice from Camp Century (north Greenland) deposited around 7640 B.C. The authors found that the peak acidity values coincide with large volcanic eruptions. They also found a significant correlation between the warm climatic periods in the Northern Hemisphere and decreasing acidity of precipitation. In their long-term perspective we are now living in a warm period with a rather low acidity of precipitation, similar to that around 1200 AD.

In ice core and pit samples from Agassiz Ice Cap, Ellesmere Island in the Canadian Arctic, covering 30 years between 1954-80 Koemer and Fisher (1982) found average pH value 5.23, and in the samples from a pit (1950-77) average pH 5.28. In the Holocene ice representing the past 5000 years they found average pH 5.48. They interpreted these findings as indicating "a significant trend of increasing acid levels over the past 25 years". But measurements at this pH range in the glacier ice are rather uncertain as such samples are not buffered by organic acids (Lund, 1989). These samples were not completely equilibrated with air and therefore the variable H_2CO_3 level may have affected the results.

Barrie et al. (1985) reported conductivity measurements in two cores from Ellesmere Island. In a core from Oxford Glacier they found an increasing trend in conductivity (interpreted as acidity) between 1945 – 1978. In the Agassiz ice cap the trend was decreasing between 1912 and 1955 and increasing after about 1960. The high 1912 peak of acidity the authors related to the Katmai eruption, but they excluded the interference of other large volcanic events, because of "high background of anthropogenically-affected spring acid-peaks". This means that they assumed that the seasonal variations are due to anthropogenic emissions, which is rather doubtful as was discussed in Chapter 2.1. But their peaks after 1935 coincided with an increased volcanic dust veil after the St. Augustine (Alaska) and Kluchevskoi (Kamchatka) eruptions, after 1958 with the Hekla (Island) eruption, in the sixties with the Gunung Agung eruption in 1963, and the peaks in the seventies with the Fuego eruption in 1974.

Delmas et al. (1985) analyzed sulfates and nitrates in a firn core from Mt. Logan in St. Elias Mtns., Yukon Territory, Canada, covering a period 1953 – 1966, and stated that no clear trend could be related to the increasing atmospheric acid pollution of the Northern Hemisphere.

Holdsworth and Peake (1985) collected two firn cores in the St. Elias Mtns: one from Mt. Logan (5340 m above sea level) covering the years 1880 to 1982, and a shorter one from Elipse (3017 m altitude). These authors concluded that the recent snow in the St. Elias Mtns. is not any more acidic than snow deposited over the last century. The absence of a long–term trend over this period implies that there is no detectable anthropogenic input of sulphate and nitrate to the contemporary precipitation in this region. On the other hand they found a significant correlation between major volcanic eruptions and peaks of acidity in snow.

Evseev et al. (1983) determined chemical composition of an 111 m deep ice core at Fridtjofbreen in western Spitsbergen. They have not recorded any significant changes of pH with depth. The pH values were ranging in particular parts of the core between 5.10 and 6.05, and SO_4^{2-} content between 10 and 2880 µg kg^{1}. In three surface snow samples the SO_4^{2-} concentrations were 190, 580 and 2590 µg kg^{-1}; the highest value was found in the lower part of the glacier. The unusually wide range of SO_4^{2-} concentrations in the ice core samples might be due to thermal drilling method used and due to removing the external part of the core by melting in distilled water.

Davies et al. (1982), in a core from the Folgefonna ice cap near Bergen, Norway, did not find any trend in concentrations of sulfate, nitrate or pH values between 1910 and 1980. However, their results are probably not reliable as they removed mechanically the external layer of the core and then sluiced with deionized water the remaining central sections of the 5 cm thick discs, cut from the core, until about 50% of the original volume remained. Such a procedure, as will be discussed below, may lead to substantial loss of impurities originally contained in the ice samples. This is also indicated by the extremely low concentrations of sulfate, nitrate, Cl, Na and Mg recorded in their samples.

In Antarctica all studies demonstrate no increase of sulfate in the snow and ice during the past few decades, and it was concluded that the effects of fuel combustion and smelting on Antarctic precipitation is negligible in comparison with the natural flux of sulfur (Delmas and Boutron, 1978; 1980; Delmas et al., 1980; Aristarain et al., 1982; Herron, 1982; Alderton and Coleman, 1985).

The sulfate concentrations in Antarctic precipitation changed similarly in time as in the Northern Hemisphere in relation to major volcanic events in both hemispheres (Delmas and Boutron, 1978; Legrand and Delmas, 1984). Delmas and Boutron (1980) stated that anthropogenic pollution as a source of sulfate in Antarctica is negligible and may be excluded from consideration. Legrand et al. (1988), in a Vostok ice core covering the past 160,000 years, found that sulfate concentrations in the recent ice are lower than in past epochs and suggested that emissions of dimethylsulfide from marine biota productivity probably are an important source of atmospheric sulfate, responsible for ancient increases of its content in Antarctic ice.

In this context it is interesting to note that concentrations of sulfate and nitrate found in the recent Antarctic ice and snow, ranging from 35 to 144 µg kg^{-1}, and 10 to 290 µg kg^{-1}, respectively (Delmas and Boutron, 1978; Legrand et al., 1984), are similar to those found in Arctic and subarctic precipitation, ranging from about 20 to 230 µg kg^{-1}, and from 15 to 150 µg kg^{-1}, respectively (Holdsworth and Peake, 1985; Davidson et al., 1985). This indicates that in the Northern Hemisphere sources of these two species are not substantially stronger than in the Southern Hemisphere, and that the global atmosphere is rather homogeneously loaded with sulfate and nitrate. From the Antarctic ice studies it was deduced that stratospheric volcanic sulfate contributes significantly to this load (Delmas et al., 1980), which also may be inferred from the stratospheric measurements by Hofmann and Rosen (1987) in the Northern Hemisphere.

This is in agreement with the opinion of Junge (1960) (based on tropospheric residence times of about 11 hours to 7 days for industrial SO_2 and 40 days for natural sulfur) that the natural sulfur components are found throughout the troposphere, including remote places such as the polar regions, whereas the washout of industrial sulfur is restricted to the mid-latitudes, where major sources are located.

Although the volcanic eruptions contribute probably only several percent to the total natural and man-made emission of sulfur into the global atmosphere (Kellog et al., 1972; Neftel et al., 1985), the long residence time of sulfur in the lower stratosphere of about 1 to 2 years (Kellog et al., 1972) makes the wide geographic range of this contribution. It is therefore detectable in remote places.

2.2.3 Historical monitoring of heavy metals

The levels of metals in the glaciers depend on the distance from the sources of pollution. Some glaciers are situated near the local and regional industrial sources of heavy metals and also near the natural sources, such as volcanic emissions, surface of the land and sea not covered by ice and snow. Snow and ice samples from sites in the clean environments of inland Greenland and Antarctica, reveal much lower concentrations of heavy metal impurities than the glaciers in temperate and tropical zones. This is due to a lack of local and regional emission sources of metals and the very small fraction of global atmospheric flow of metals which reaches Antarctica and inland parts of Greenland. In these parts of Greenland the low content of metals in precipitation may also be due to the fact that the sampling sites are situated at altitudes between about 2000 and 3000 m above sea level. Therefore the impurities at these sites are deposited from much shorter column of air than at low altitude sites. It was found that in the middle latitudes about 50% of impurities contained in the 15 km high air column are concentrated between ground level and 3 km altitude (Kownacka et al., 1989).

The inland glaciers of Greenland and Antarctica may be rightly used for studies of historical changes of levels of metals in the global atmosphere, but they are not suitable for determination of the global flows. For this aim a collection of data from many glaciers in temperate, tropical and polar zones is needed, which could represent various typical geographical regions, both those with high and low natural and man-made emissions.

During the past 25 years the historical changes of stable and radioactive pollutants have been studied in many glaciers in Europe, Asia, Africa, North and South America, Svalbard, Greenland and Antarctica. In polar regions, except for one study on stable lead discussed later (Murozumi et al., 1969), the results of these researches showed no significant increase of metals and metalloids from prehistoric times to the present (see reviews by Alderton and Coleman, 1985 and Boutron, 1986). Small recent increases were reported from sites exposed to local or regional anthropogenic or natural emissions. In the Tatra Mtns., Poland, in ice from a glacier exposed to heavy local automotive traffic and to emissions from the neighboring great Polish and Czechoslovakian industrial centers the lead level increased about 15 times during the last 100 years (Jaworowski, 1968). On Spitsbergen and in southern Norway lead concentrations increased about twofold in contemporary ice as compared with the pre-industrial period (Jaworowski et al., 1981).

In southern Norway a more than fourfold increase in concentration of cadmium was observed in recent ice as compared with pre-industrial samples; in northern Norway, the Alps, Ruwenzori (Africa) and the Peruvian Andes cadmium increased by a factor of about 0.6 to 2, but the mean of summarized results from nine glaciers in both hemispheres (Spitsbergen, northern Norway, Alaska, southern Norway, Alps, Himalayas, Ruwenzori, Andes and King George Island in Antarctica) showed no increase in lead, cadmium, vanadium, mercury, uranium and radium concentrations in the twentieth century (Jaworowski et al., 1981).

In Greenland no recent increase in cadmium content was observed at Camp Century (Weiss et al., 1975) and at the Milcent site (Herron et al., 1977 B), but about sixfold increase was found at the Dye 3 site in post-1960 ice. However, the values from Due 3 site may be influenced by local contaminations from the radar station constructed there in 1961.

Weiss et al. (1975), Herron et al. (1977 B) and Herron (1980) interpreted their results of zinc measurements as indicating a three- to fourfold increase of zinc in Greenland ice and snow during the last century due to anthropogenic emissions. However, a perusal analysis of their set of data showed no significant long-term trend (Boutron, 1986).

An increase in mercury content in Greenland ice was reported by Weiss et al. (1971 B), Weiss et al.(1975), Herron et al. (1978 B) and Weiss et al. (1977) and interpreted as being due to anthropogenic sources. Their results were a compilation of data from several sites (Camp Tuto,Dye 3, Camp Century, and various "virgin trench sites" in its vicinity). This compilation is subject to the same criticism (i.e. poor geographical compatibility and possible local contamination from military bases) as expressed in para. 2.2.2. in the case of acidity studies by the same group of authors in the same set of samples. In a criticism of these mercury studies Dickson (1972) stated that the observed variations in mercury content could be due to heterogeneous distribution of mercury in the ice cap and not to anthropogenic influences.

This suggestion was confirmed by Carr and Wilkniss (1973) who found distinct differences in mercury concentrations between several Greenland sites. The quality of sampling and analytical procedures for mercury determinations by Herron et al. (1977 B) was questioned by Alderton and Coleman (1985).

Appelquist et al. (1978) in the samples from twentieth century ice from Cretê, Greenland, found a mean mercury content of 0.011 μg kg^{-1}, with a peak for that period of 0.015 μg kg^{-1} in a 1913 sample. In the eighteenth century ice the mercury content was in the same order of magnitude, with a peak of 0.014 μg kg^{-1} in 1775-1786 ice. These authors concluded that a lack of temporal trend in their data questions the validity of previous calculations of the global mercury turnover of Weiss et al. (1971).

In the case of other metals studied in the Greenland ice cores (copper, arsenic, selenium, silver, antimony) no clear recent increase has been observed (see review in Boutron, 1986). The lack of increasing trend in concentrations of metals in ice was commented as "surprising" by a reviewer

(Boutron, 1986). The surprise was due to the inconsistency of the lack of temporal trends in Greenland ice concentrations with global inventories of natural and man-made emissions of toxic metals and metalloids to the atmosphere, calculated by Nriagu (1979) and Lantzy and MacKenzie (1979) which indicated that the current anthropogenic emissions of heavy elements to the atmosphere largely exceed natural rates, and have strongly increased in the last century. But Nriagu (1979), Lantzy and MacKenzie (1979) and Nriagu (1989) assessments of natural contributions are based on emissions of particulates for which low concentrations of nuclides were assumed, i.e. such as in the Earth's crust, soil, sea salt, plant ash etc., not taking into account their enrichment in the airborne dust, probably by biogenic and other natural processes. The enrichment factors for some of these nuclides reach very high values, e.g., 50 to 1000 for lead (Zoller et al., 1977; Duce et al., 1975 Herron et al., 1977; Boutron and Lorius, 1979, Jaworowski et al., 1981; Maenhaut et al., 1989), and even higher for other metals (see reviews in Weiss et al., 1978; Jaworowski et al., 1981 and Boutron, 1986, and Table 2).

The assessment of natural flows based on emission of particulates with assumed low concentrations of nuclides in the airborne dust (usually similar to the Earth's crust abundances), which ignore these high enrichments, must lead to an underestimation of global atmospheric flows by at least one order of magnitude. Estimates based on direct measurements of metals deposited with the global precipitation (Jaworowski et al., 1981) and on the long-term measurements of content of metals in the 15 km high air column (Kownacka et al., 1989) indicate that the anthropogenic sources contribute a small fraction to the total flow of metals into the global atmosphere (Table 5).

The first study of historical trends in metal content in precipitation in the Norwegian Arctic was carried out in 1974 by Jaworowski et al. (1981 and 1982) at Hornsundfjorden, Spitsbergen. The concentrations of ^{137}Cs, three natural radionuclides and four heavy metals were determined in 23 contemporary snow and ice samples from Körberbreen, covering a period 1951–1973 and in 2 samples of approximately 100 years old ice from Hansbreen (Table 3).

The mean concentrations found in this study indicate no trend in ^{226}Ra, uranium, vanadium, cadmium and mercury concentrations, and an approximately twofold increase in stable lead concentration. A low level of natural radioactive ^{210}Pb in pre-industrial ice is due to physical decay of this nuclide. The measurable concentrations of ^{137}Cs from nuclear explosions (~10% of 1951–1973 level) in the old ice indicate that it was partially contaminated by the percolating melt water from the recent precipitation. It is interesting to note that the peak values of ^{210}Pb in 1958 and 1965 coincided with the peaks of ^{137}Cs, which indicates that nuclear explosions contributed to a part of ^{210}Pb present in the atmosphere.

As compared with other glaciers in Greenland, Alaska, and Antarctica the Spitsbergen values are higher and similar to those in the middle and tropical latitudes (Table 4). This probably reflects the patterns of long-range transport of impurities to this region or the influence of the local natural sources, such as resuspension of particulates from weathered rock surfaces uncovered with snow and maritime emissions.

The peak concentrations of vanadium, lead and possible radium–226 in 1956 and 1957 may be due to the Bezymjannaja, Kamchatka volcanic eruption in 1956, after which the dust cloud travelled across the Arctic (Lamb, 1970). The peaks of vanadium, lead, and cadmium in 1963, 1964 and 1965 may be related to the Gunung Agung eruption in 1963, after which the abnormal optical phenomena and the stratospheric dust were observed at least to the end of 1966 (Lamb, 1970).

The main drawback of this study was a disproportion between the number of contemporary samples and the pre-industrial ones, and also collecting both types of samples from different (albeit neighboring) glaciers. This makes the assessment of a long-term trend in this locality uncertain.

Evseev et al. (1983) determined Pb, Co, Cr, Fe, Cu, Mn, Zn, Cd and Ni in 16 samples from a 111.2 m deep ice core and in three surface snow samples from Fridtjofbreen in west Spitsbergen. The age of the oldest samples was probably about 220 years. They found very low concentrations of Pb, Co and Cr, below 0.01 $\mu g\ kg^{-1}$. The concentrations of Fe ranged from 43 to 1880 $\mu g\ kg^{-1}$, of Cu from 2.0 to 17.2 $\mu g\ kg^{-1}$, of Mn from 4.8 to 16.0 $\mu g\ kg^{-1}$ of Zn from 15 to 62 $\mu g\ kg^{-1}$, of Cd from 0.1 to 0.48 $\mu g\ kg^{-1}$ and of Ni from 0.01 to 4.1 $\mu g\ kg^{-1}$. Except for Cd their results do not show an increasing trend in metal concentrations in ice. However, as already discussed in para. 2.2.2, one should note that the use of thermal drilling and removal by melting in distilled water the external part of the core might lead to changes in the original composition of the ice core. From the melted samples the authors separated by filtration the particulate matter and determined the metals only in the filtrate. This might influence their results as the majority of some of the metals present in the glacier ice is associated with the particulates (Jaworowski et al., 1983).

2.2.3.1 Stable lead controversy

Since the industrial revolution, vast amounts of various toxic heavy metals have been introduced by man into the atmosphere, but none of them evoke such an interest and plethora of publications as lead. This is astonishing as some of these metals are more toxic than lead and their man-made mobilization was much more dramatic than that of lead. For instance, in the past hundred years the production of lead increased about 100 times, of zinc 1140 times and of cadmium 9400 times (Jaworowski et al., 1985).

It seems that during the last two decades, the impact on man and environment of the industrial emissions of lead was greatly exaggerated in the media and in a part of scientific literature. One of the main arguments used in support of a hypothesis stating that contemporary man and global environment are polluted by lead 500 times above the natural level (Patterson, 1965) are the results of a study on temporal changes of lead concentrations in polar glaciers (Murozumi, Chow and Patterson, 1969). In this paper the authors concluded that in Greenland the lead content in ice recently increased 500 times above the pre-industrial level and 300 times in the Antarctica.

The paper of Murozumi et al. (1969) is the only evidence for such an increase of lead in the global atmosphere. It was not confirmed by several later studies of glaciers in Greenland (Cragin et al., 1973; Herron et al., 1977 A;), Antarctica (Boutron and Lorius, 1979; Boutron et al., 1981; Boutron, 1982; Boutron and Patterson, 1983), which demonstrated no increasing trend in lead content in ice between 1170 A.D. and 1974 in Greenland and during the past 12,000 years in Antarctica. Also no increasing trend of lead was found in a global scale glacier pollution study (Jaworowski et al., 1981).

Murozumi et al.'s (1969) paper was criticized for methodological flaws and improper conclusions (Dickson, 1972; Herron et al., 1977 A; Robinson, 1981; Jaworowski et al., 1981; Jaworowski et al., 1983; Jaworowski, 1986). This criticism was concentrated on the following flaws in this paper: (1) Doubtful validity of the Greenland set of samples compiled from several locations to get the long time scale (see discussion in para. 2.2.1 and 2.2.2). (2) Rejecting the high values from several measurements of 2700 years old ice and using only the minimum values for comparison with maximum values found in several contemporary ice samples from the Camp Century site influenced by local contaminations from a military base. (3) Loss of the majority of particulate lead from old ice samples, by rejecting the "rings of dark, amorphous material" from their sample containers; these rings were found to contain 50 to 100 % of lead present in the ice. (4) Comparing maximum values from a set of several recent samples with a theoretical "natural" lead level in 2700 old ice, calculated from measurements of silica dust in this old ice. They assumed that this silica is the only natural source of lead in the ice and that it has the same lead concentration as the average abundance of lead in the Earth's crust. In this calculation they ignored the phenomenon of enrichment of lead high above the crustal concentrations in the airborne dust; in the polar ice this enrichment reaches values of up to 2400 (Table 2).

The hypothetical "natural" lead concentrations in the old ice of 0.00003 to 0.0004 $\mu g\ kg^{-1}$ were much lower than even the lowest values measured in Murozumi et al. (1969), 2700 years old ice, ranging from 0.001 to 0.007 $\mu g\ kg^{-1}$. They used these hypothetical "natural" concentrations they used for: (a) justification of eliminations of high readings from old ice measurements, and (b) for concluding that the lead concentrations in Greenland snow had increased by a factor of 500 since the year 800 B.C. and 300–fold in Antarctica. A pre–conceived idea of a "lead megaexposure" (Patterson, 1965) evidently biased both the methodological part and conclusions of the Murozumi et al. (1969) work.

Although the poor methodological quality of Murozumi et al.(1969) paper is known since early seventies it was often cited as a proof of "a dramatic increase of lead in the environment" (National Academy of Science, 1972), and as an indication of "a worldwide dispersion of lead from the industrial areas" (British Central Unit of Environmental Pollution, 1974). Recently this paper was used as a demonstration of "how lead levels have risen over the whole of the earth's surface since the start of the industrial revolution, and how twentieth century man is having a profound effect on the amount of lead in his environment" (Royal Commission on Environmental Pollution, 1983).

This is mainly due to a paper by Ng and Patterson (1981), presenting the results of an experiment with the polar ice cores, drilled by other authors 16 years before (Cragin et al., 1975; Herron et al., 1977 A). During such a long period of storage, these cores contracted in size by about 20 % due to sublimation. Ng and Patterson (1981) found that the external parts of these cores contain up to 700,000 times higher concentrations of lead than the central parts. They stated that this is due to contamination of the surface of the cores during drilling and further handling, and that therefore all the previous data on lead levels in the Greenland and Antarctic ice samples, from which the external layers were not removed, are erroneous, except those of Murozumi et al. (1969). However, this last statement is not consistent with this argumentation, as Murozumi et al. (1969) did not remove external parts of their samples before melting.

The Ng and Patterson (1981) paper had two consequences. Numerous scientists (see e.g., Alderton and Coleman, 1985; Boutron, 1986) accepted their claim as proved that in Greenland lead levels in ice increased several hundred times above natural level, due to anthropogenic emissions in the Northern Hemisphere, but not in Antarctica. The lack of an increasing trend in Antarctica was explained by a supposition that equator is a barrier to tropospheric aerosol transport (Boutron, 1980; Boutron, 1986). This assumption is not consistent with (1) the data on acidity of Antarctic precipitation, which is similar to those from Greenland (see para. 2.2.2.), (2) with the results of Boutron (1980) indicating that the volcanic eruptions in the Northern Hemisphere influenced the metal content of the Antarctic ice, and (3) with the well known fact that the radioactive debris from nuclear test explosions in the atmosphere of the Northern Hemisphere was reaching the Southern Hemisphere after a period of about 1.5 year (UNSCEAR, 1972).

The second consequence was that in the new studies of Arctic, Antarctic and other glaciers the external layers of cores and of blocks of ice were regarded a priori as contaminated and discarded, and only the concentrations of metals found in their central parts were taken into account. This lead to a drastic decrease in the levels of metals in precipitation assumed to be natural (e.g. in the case of lead by a factor of up to 1000). This brings into question the interpretation of man's impact on atmospheric pollution, and has far reaching social and economic consequences.

But the high concentrations of metals in the external layers of cores and blocks of ice, observed under extremely clean sampling conditions in later studies (Boutron and Patterson, 1983; Boutron and Patterson, 1986; Boutron et al., 1987; Wolff and Peel, 1985), are probably not caused by the contamination of samples, but by the transfer of metals from the centre of the ice samples to their surface (Jaworowski et al., 1983). This is because there exists a network of capillary veins of liquid in the ice, even at the extremely low temperatures of the Antarctic ice-sheet. This liquid, which contains high concentrations of sulfuric and nitric acids and other impurities, is localized at triple-junctions between the ice grain boundaries. In these triple-junctions up to 100 % of sulfuric acid is localized, and not in the crystal lattice itself (Mulvaney et al., 1988). In an original, undisturbed ice layer in a glacier the impurities contained in a quasi infinite capillary network are in a steady state, and their diffusive migration is in equilibrium. After collecting cores or blocks of ice the vast number of capillaries are cut open and many metals dissolved by the acids will be transported by the capillary effect to the surface of the samples, where they are deposited due to evaporation of the carrier liquid.

Therefore rejection of the external layer of ice samples probably leads to an artificial decrease of the original average concentration of metals by up to three orders of magnitude. It is possible that this has also been the case for some of the acidity determinations in glacier ice. Unfortunately this improper procedure recently became a standard in analysis of glacier ice samples for global pollution research. An experimental study on the capillary transport phenomena in ice at low temperatures should be carried out to check the reliability of the current methods of determination of pollutants in the glacier ice.

Ng and Patterson (1981) stated that the authors who had not found the increasing trend in lead content in the glacier ice in Greeenland (Cragin et al., 1975; Herron et al., 1977) and in Antarctica (Boutron and Lorius, 1979) were unable to measure correctly lead concentrations in the ice samples.

It is interesting that Ng and Patterson (1981) refrain from supporting the hypothesis of an enormous increase of lead in Antarctica, although they disqualified as incorrect the Antarctic lead concentrations of Boutron and Lorius (1979). In this and other publications of Patterson and his co-workers the hypothesis of a 300-fold increase of lead in Antarctica is omitted, and they limit their claim to Greenland ice. This was followed by other authors, the Antarctic part of "lead mega-exposure" from the Murozumi et al. (1969) paper was somehow forgotten, but the Greenland part is still being accepted as valid, although it was coined in the same incorrect way as the Antarctic one.

In Antarctica historical monitoring of lead changes were several times repeated after Ng and Patterson's (1981) paper, and as was discussed above, the results of these studies contradicted the claims of Ng and Patterson (1981) and Murozumi et al., 1969). It is interesting that this has not been done in Greenland or at other arctic sites.

As appears from this discussion the historical studies of lead in the Arctic need a careful revision. This should be done by repeating the measurements of lead concentrations in ice strata covering the last several hundred years. Checking of the reliability of the current methodology of ice sampling should also be done by studying the radial migration of impurities in the ice cores.

2.2.4 Historical monitoring of atmospheric particulates

Kumai (1977) reported results of electron microscopic determination of concentration, size distribution and morphology of dust particles deposited in 2000 to 100,000 years old Greenland ice from Camp Century. The average concentration of particles in 2000 year old ice was 3 x 10^6 g^{-1} (32 x 10^{-9} g g^{-1}), i.e. of the same order of magnitude as that in the present snow. The maximum concentration of silica particles of 830 x 10^6 g^{-1} was found in the 100,000 year old ice. The concentrations of silica dust in contemporary snow was lower than in the ice of Wisconsin age. The size distribution of particles in 2000 year old ice was 0.05 to 8 µm, i.e. similar to that in the contemporary snow. In addition to silica, sea salt particles and possibly also volcanic ash were identified in the ice.

In the same location Thompson (1977) determined the concentration and size distribution of particles in the ice core at various depths between about 82 and 1345 m from the surface. Concentration of 0.65 – 0.82 µm particles ranged from 1.4 x 10^6 g^{-1} at the depth of 523 m to 195 x 10^6 g^{-1} at the depth of 929 m.

Hammer (1977 B) found that the majority of dust in an ice core from Dye 3, Greenland is of non-volcanic origin, although several peak concentrations were related to the Lamb's volcanic dust veil index. In the period 1900–1951 the dust concentrations were of the same order as those at greater depth which corresponded to period of high volcanic activity. This author suggested that the bulk of dust in more recent samples originated from North America, due to its increasing aridity.

Similar dust studies were carried out in recent ice in the Peruvian Andes (Thompson et al., 1979), where the concentrations of particles were found to vary by about two orders of magnitude between 1965 to 1976, probably in relation to seasons.

Petit et al. (1981) reported from Dome C, Antarctica ice core concentrations of particles ranging from 7 x 10^3 g^{-1} at 0 – 381 m depth to 87 x 10^3 g^{-1} at 510–687 m depth. In contemporary snow they found a concentration of 6 x 10^3 g^{-1}. These values are about three orders of magnitude lower than those from Greenland.

Thompson et al. (1988) in two cores from the Peruvian Andes reported low levels of dust in recent samples as compared with those from the years around 600 and 920 A.D. The past concentration peaks the authors related to the changes in agricultural activity.

These studies demonstrate that concentrations of particulates are much higher in both contemporary and prehistoric ice in Greenland than in Antarctica. At both locations the dust contents in ice were ten to hundred times higher in some ice samples from the past epochs than in the contemporary snow.

It is interesting to note that the historical studies on changes of concentrations of the most typical particulates associated with industrial combustion of fossil fuels, i.e., graphitic carbon have not been carried out as yet.

2.2.5 Historical monitoring of organic compounds

These studies were carried out on a limited scale in the Antarctica but not in the Arctic. In Antarctica Risebrough et al. (1976) in a 6 m deep section of snow from Doumer Island found concentrations of DDT of 0.5 to 4 ng kg^{-1} and of PCB's (pentachlorophenyls) of 0.03 to 1.2 ng kg^{-1}; the concentrations increased with depth, indicating a lesser atmospheric transport of these species to Antarctica in the recent years.

At Mizuho station Tanabe et al. (1983) found an opposite temporal trend in BHC and DDT content, i.e., an increase in 1980 snow as compared with 1960, but the PCBs decreased during this period. These data are too scarce to allow for any conclusions on the global situation or even for the Antarctica.

In the Arctic such studies covering a period of the last 40 years should be carried out in view of the possible consequences for the arctic wildlife of both the current atmospheric transport of organic pollutants and of their past accumulation in the arctic ecosystem.

2.2.6. Conclusions

The importance of historical studies of impurities in glacier ice for understanding the impact of man's activity on the global environment is probably second to none. The past research in the polar regions was concentrated mainly on acids, heavy metals and major ions and to a lesser extent on silica, clay and volcanic particulates. Still less known are temporal changes of organic toxic compounds and no such study was carried out in the Arctic. Also no historical study of graphitic carbon was carried out in polar glaciers.

Acids. The results of the current studies of the history of acidity in precipitation are far from unequivocal in indicating a significant role of industrial sources. This is particularly valid for arctic research, as in the Antarctica all studies showed a lack of contribution from anthropogenic sources to the acidity of precipitation. But in the Arctic the results of several studies differ so drastically that a clear cut conclusion cannot be reached as yet.

Uninhabited virgin sites at Austfonna and Olav V Land in Svalbard are the regions exposed to more intensive transport of sulfates, nitrates and other impurities from Eurasiatic and North American continents than other arctic locations. These sites are also situated at much lower elevations (a few hundred meters) than the Greenland sites (2000 to 3000 meters) and therefore are collecting impurities from a much higher column of air. Therefore ice cores from these Svalbard locations could mirror more efficiently long-term changes in the Northem Hemispheric air than the cores from other arctic sampling sites.

The study of historical trends in the acidity of arctic precipitation should be associated with the research on geographical distribution of the temporal changes, by core studies in Norwegian glaciers, situated between $60^{o}00$ N and $79^{o}30$ N (Folgefonna, Storbreen, Okstinden, Seilandsjökelen). Determination of temporal changes of acidity of precipitation, and also of content of other impurities in such a large geographical network will provide a basis for a quantitative estimate of the relative contribution of man-made and natural sources to the pollution of North Hemispheric precipitation. This will also help in estimating the flow of sulfate, nitrate and of other impurities into the global atmosphere.

In Antarctica no increase of acidity in recent snow was observed. In the Arctic several authors claimed two to threefold increase but others have not found any increasing trend. The concentrations of sulfates and nitrates in arctic ice are similar to those found in Antarctica, which indicates that both regions are exposed to global sources of these species of similar strength. A majority of historical studies of acid precipitation in Greenland was carried out at sites influenced by recent local emissions from military bases and therefore probably do not reflect the hemispheric changes. The peak concentrations observed in ice from the past two decades usually coincided with the large volcanic eruptions. From the long-term studies it appears that the current precipitation is less acidic than it was in several past epochs.

Of greatest interest are the changes in acidity of precipitation during the last hundred years, when the global fossil fuel burning (associated with SO_2 emission) increased about ten times, as well as other industrial activities associated with NO_x emission. This problem clearly needs elucidation by future studies.

One may expect that the possible effects of such an increase of the anthropogenic source of atmospheric acids should be reflected in fossil precipitation deposited in glaciers of Svalbard and Scandinavia better than in other glaciers, as this region is particularly exposed to the long-range transport of pollutants from Eurasiatic industrial centers. It seems therefore that the Svalbard and Scandinavian glaciers are best suited for such studies.

Heavy metals. In all glaciers studied heavy metals deposited in ice with the airborne particulates were found to be enriched by up to several orders of magnitude, as compared with the Earth's crust abundances. This was found both in the recent and in old ice deposited ages before man started to emit these metals into the atmosphere. This enrichment is due to such natural processes as chemical fractionation at the air/sea interface, plant emissions or methylation by bacteria and algae in marine and terrestrial ecosystems. These processes need further experimental and field studies, before their contributions to the global flows of metals can be quantitatively estimated. The estimates of inventories of emissions of metals into the global atmosphere which neglect this phenomenon grossly underestimate the natural component.

In several locations an increase of some of the heavy metals was observed in recent ice (by a factor of 0.6 to 15) which was probably mostly due to the local influences. However, the most reliable data from Greenland and Antarctica and from a global scale study do not show any significant increasing trend.

The lack of increase in metal content in the precipitation is not consistent with those estimates of global inventories of emissions, which show the current anthropogenic contributions exceeding largely the natural ones (Nriagu, 1979; Nriagu, 1989; Nriagu and Pacyna, 1988). These estimates did not take into account natural enrichment processes. On the other hand, the lack of an increasing trend agrees with the estimates of emissions of metals into the global atmosphere based on direct measurements of deposition of metals with the precipitation, and on long-term atmospheric measurements. These last estimates demonstrate that man-made sources contribute a small fraction to the total atmospheric load of heavy metals.

The question of the temporal changes of lead in the arctic precipitation since the beginning of the industrial revolution remains open. It should be resolved by detailed study of the temporal changes of heavy metals in remote glaciers in Svalbard and in those exposed on regional influences in Scandinavia.

A study on radial migration of impurities in glacier ice cores is needed for checking the reliability of the currently used ice sampling methods.

Particulates. Only three studies of temporal changes of dust levels in the ice have been carried out in the Arctic. Only in one of them ice samples represented recent precipitation. No increase of particulates was found in a period 1900 - 1951, as compared with much older ice. It seems that particulates need further studies in the Arctic in view of the importance of particulate load of the atmosphere for the climatic changes. Current particulate load causes the global temperature to be about 2 - 3^{o} C cooler than it would be in the absence of aerosols (Harvey, 1988).

The absorption optical depth due to the presence of graphitic carbon in the arctic troposphere is large enough to cause a substantial change in the solar radiation balance over a highly reflecting polar surface (Rosen and Hansen, 1984). It seems that a study on temporal changes of graphitic carbon originated in fossil fuel combustion and biomass burning should receive high priority in research planning for the Arctic.

Organic compounds. The history of changes of toxic organic compounds (DDT, DDE, BHC and PCB's) in glacier ice has been studied in the Antarctic but not in the Arctic. Such research should be included into a program of the future ice core studies in Svalbard and Norway.

2.3. SOIL.

2.3.1. Introduction.

Due to frost segregation structural soils are common in Svalbard. The mechanical mixing of organic and mineral layers of soil caused by frost is of great importance in this region where mixing by soil fauna is not much effective (Sendstad, 1976). The thickness of soil in Svalbard is rather small but generally greater than in continental Norway (Låg, 1986). This is due to a greater amount of mineral components in Svalbard soils (Orvin, 1940). The organic part of soil in Svalbard is mainly of a peat character and has a rather low content of nutrients, except for locations exposed to bird droppings.

An important factor for soil processes in this region is permafrost. Usually a few decimeters thick layer of soil melts during summer and meltwater is not drained down to the deeper frozen layers. Therefore even with low precipitation in many places the soil reveals swampy patterns (Låg, 1986).

Besides cryogenic processes, gleization, peat formation and browning are important soil-forming processes in the periglacial zone. A detailed soil study was carried out at Calypsostranda, Belsund, where a strong skeleton, general occurrence of carbonates, neutral or basic reaction, low content of available phosphorus and potassium, and high content of organic carbon are characteristic features of the soil (Klimowicz and Uziak, 1988). At this locality light and medium, often silty soil formations are prevailing. Poorly developed brown or brownish soils have usually silty sands and gravel composition. Such soils are most common in dry tundra and coastal plains. The heaviest grain-sized composition (clay and silty clay) was found in gley soils of wet tundra and at a pingo.

The soil has a low resistance to crushing, probably due to a greater content of coarse silt than of colloidal clay. Calcium carbonate occurs in most soils and its content increased with depth, reaching 46% in a brown fossil soil. It is lacking in brown soils on roche moutonnée and in peat and shallow peaty soils.

The highest content of organic matter (maximum 61.8%) was found in peat and shallow peaty soils formed in turf hammocks and polygonal soils around fissures. Organic horizons are also the thickest in these soils. In poorly developed brownish soils in dry tundra, a considerable content of organic matter was also found in areas with dense floristic cover. Also in dry and wet tundra more organic carbon was found under dense than under thin vegetation cover. Organic carbon is present in all soil profiles, but most of it occurs in their upper horizons.

Total nitrogen content ranging from 0.01 to 2 % depends largely on the humus content. The lowest nitrogen content was found in the stone polygonal soils and in fossil soils. Phosphorus and potassium contents are low in mineral soils in Svalbard. Only in organic soils with abundant bird and reindeer excrements, a considerable quantity of these elements were found (Klimowicz and Uziak, 1988). Similar patterns of soils were found also in the Hornsund region (Szerszen, 1968, 1974).

2.3.2 Acidity, heavy metals and organic compounds

In Bellsund only the upper horizons of organic (peat and shallow peaty) soils are slightly acidic. Soils under dense vegetation cover in dry and wet tundra have a slightly lower pH than under scarce vegetation. In fissure polygonal soil lower pH values were recorded along the fissures, where more organic matter occurs. Alkaline reaction of most soils on the Calypsostranda is due to occurrence of carbonates in the parent rocks, poor leaching and intensive cryogenic processes which generate permanent vertical movement of the soil material (Klimowicz and Uziak, 1988).

Acidity of unpolluted topsoils at Brøggerhalvøya, near Ny Ålesund, was found to range between pH 6.3 and 7.9 (Sendstad and Sveum, 1985). At another site near Ny-Ålesund acidity of pH 5.1 was found (Låg, 1983). This can be compared with typical acidity of continental Norwegian humus soils of pH 3.6 - 4.5.

In several sites in Svalbard soil was found to be highly contaminated with sulfuric acid by the liquid effluents from coal mine refuse, composed of pyritic minerals. Sulfides contained in these minerals are transformed into sulfuric acid upon contact with oxygen and water. This acid is leached out with rainwater and transferred to the neighboring soil. This caused severe disturbances in the vegetation cover. The acidity of organic parts of soils in several sites in the Longyearbyen region was found to be between pH 2.3 and 3.1 near a new refuse dump. Near 50 to 70 years old dumps deposited up to 1920 and 1937, the acidity of soil was lower (pH 3.4 - 4.2). However, at some of the old dumps the soil acidity was found to be as high as pH 2.4. In the majority of places near old dumps an invasion of new vegetation was observed, as well as near some younger dumps (Låg, 1983 and 1988).

Alkalinized soils (pH 8.2 - 9.6) were found near Mine No. 3 in Bjørndalen, from which the alkaline waste water is pumped out to a refuse dump. In this area all higher vegetation disappeared (Låg, 1983 and 1988). These negative environmental impacts of industrial activity on tundra in Svalbard are limited to the immediate vicinity of mines. It would take several decades to ameliorate this impact by natural processes. Mitigation of this impact by such measures as draining the effluents and the use of calcium and other fertilizers was envisaged (Låg, 1983).

The content of heavy metals in undisturbed soils was determined by Thomas (1986) at three sites in Svalbard. He found concentrations of Zn, Pb, Cu, and Ni of 21 - 43, 0.2 - 2.9, 3.0 - 8.3, and 0.8 - 6.6 $\mu g\ g^{-1}$, respectively. Låg (1983) found in seven samples of soil collected near Longyearbyen concentrations of Zn, Pb, Cu, and Cd of 32 - 120, 4 - 10, 17 - 48, and >0.5 $\mu g\ g^{-1}$, respectively.

The concentrations of Pb in Svalbard are much lower than typical values in Norway, but content of other metals in soil is similar (Steinnes, 1984; Allen and Steinnes, 1985). The Zn and Pb concentrations found in Svalbard soil are also much lower than 84-158 and 62-112 $\mu g.kg^{-1}$, respectively, found in 17 samples from a 98 cm deep virgin Alaskan soil profile. These samples were collected in a region near Mt. McKinley, rich in heavy metal deposits. The vertical distribution of Pb, Zn, Th, U, ^{226}Ra and ^{210}Pb was homogeneous, i.e. no significant increased concentrations were found in the upper 5 cm thick layer as compared with the lower ones. The mean Zn and Pb content in the Alaskan soil profile was slightly higher than soils from industrial regions of Central Europe (Jaworowski and Grzybowska, 1980).

As will be seen in para. 2.4, in contrast to the soils, the concentrations of metals in plants are higher on Spitsbergen than in the continental Norway. This may be due to a greater plant availability of metals in the Svalbard soils, due to intensive weathering processes. This problem deserves a closer study.

At four sites on Spitsbergen Thomas (1986) measured concentrations of polyaromatic hydrocarbons (PAH: fluoranthene and benzo-ghi-perylene) and chlorinated hydrocarbons (α-HCH and -HCH) in soil. He stated that HCHs can originate only from distant source regions, as they are dispersed in the environment as a pesticide lindane, which is obviously not used on Spitsbergen. On the other hand, PAHs may be of local origin, as they are formed by fossil fuel combustion at Svalbard settlements. This is reflected by geographic distribution of these species in Svalbard. Concentrations of PAH (fluoroanthene) were much higher in Adventdalen and Ny-Ålesund, i.e. near the settlements, than in Reinsdyrflya and Krossfjorden in northern Spitsbergen. Concentrations of HCHs were more homogeneously distributed, with the notable exception of Reinsdyrflya where the lowest soil concentrations were found. This might reflect an effect of sea transport of HCHs, as this location is less affected by the northward West Spitsbergen current than the other ones studied. This current transports waterborne pollution from the North Sea to the High Arctic. The North Sea is a main recipient for northern and western Europe (Aarkrog et al., 1983). These authors found that pollution from the North Sea enters coastal waters of Greenland after 3 – 5 years, travelling first to about 80^{o}N latitude, and bringing there the pollutant concentrations to about 1 % of those found in the North Sea.

2.3.3 Conclusions

(1) It seems that in Svalbard the long-range atmospheric transport from industrial sources did not influence significantly the levels of heavy metals in soil. The existing levels are probably dominated by natural contributions from the local weathering processes.

(2) Acidity of precipitation in Svalbard ranging from pH 4.5 to 5.5, and concentrations of sulfates in recent precipitation of 30 to 900 $\mu g\ kg^{-1}$, and nitrates of 60 $\mu g\ kg^{-1}$ are similar to the typical values in Northern Norway (see para. 2.2.1). One may expect that similar to the lakes in northern Norway (Wright et al., 1977) the acidity of precipitation does not increase the acidity of lakes below pH 5.5. The soils in Svalbard have well developed mineral layers. It seems, therefore, that the strong acids from precipitation could be neutralized by cation content of the minerals, so that the fresh water systems should not be acidified by the percolating rainwater. This is supported by the results of acidity measurements in 57 fresh water samples from three areas at Hornsund, in which the average pH values were at the alkaline side (pH 7.2 – 7.6) (Krzyszowska, 1985). This is not expected to occur in places where the summer thawing of soil is limited to the humus layer or where the meltwater and rainwater enter directly into the lakes or streams without passing through the pH buffering system of soil.

(3) Measurements of acidity of freshwater lakes in Svalbard could help to determine the critical loads of strong acids in this region. Two studies of permanent lakes in western Spitsbergen (Foged, 1964; Picinska-Faltynowicz, 1988) indicated that the diatom flora consists mainly of indifferent and alkaliphilous species, typical for waters with pH values of about 7. In the Hornsund region the diatom assamblages are formed by species with low trophic requirements, which suggests that the studied water bodies are unpolluted (Picinska-Faltynowicz, 1988).

A study on diatom populations in the lacustrine sediments may reveal the past history of acidity of lakes on Spitsbergen during the past several thousands years. The results should be compared with similar studies already carried out in Norway (Berge, 1975) and in Sweden (Robertson, 1988).

(4) Other airborne pollutants such as heavy metals, organic compounds and natural radionuclides may be also accumulated in soils. Determination of their vertical distribution in Svalbard soils could be useful for defining the critical loads of these substances in this region.

(5) The ability of organochlorine pesticides to enter the food chain in significant quantities, as a result of their relatively high stability, lipid solubility and propensity for bioaccumulation poses a possible hazard for arctic biota. Accumulation of these organic pollutants in the arctic soils may contribute to this hazard. A study on the accumulation rate of organic pollutants to the soils in Svalbard could help in defining the critical loads of these substances in the Norwegian Arctic.

(6) The content of fission products in soil has not been determined in Svalbard. Such a study could help to elucidate the causes of unusually low content of ^{137}Cs in reindeer meat from Spitsbergen as compared to that from Norway (see para 2.5). Cesium is generally rather strongly fixed in soil. Downward migration and availability to plants are therefore reduced. Its mobility may be somewhat greater in organic soils. Cesium fixation in soil is much less in some areas with wet, peaty and podzolic soils. This allows enhanced transfer to plants and animals, but also a greater leaching out of the soil with melt- and rainwater. On the other hand cesium is more strongly retained in soils containing predominantly micaceous minerals (WHO, 1983), because of cesium's large ionic radius which fits into the large K site in mica.

The low content of ^{137}Cs in Svalbard reindeer may be due to either a higher fixation or greater leachability of this nuclide in soil. A study of vertical distribution of ^{137}Cs in soils in Svalbard, as well as measurements of its amounts deposited in sediments of lakes, which are sinks for the elements leached out from the soil, could help to solve this problem.

Other airborne pollutants such as heavy metals, organic compounds and natural radionuclides may also be accumulated in soils. Determination of their vertical distribution in Svalbard soils and in the lacustrine sediments could be useful for defining the critical loads of these substances in the Arctic.

2.4. PLANTS.

In two studies the heavy metal concentrations have been measured in lichens and mosses from Svalbard in relation to other geographic locations. Jaworowski (1967) measured concentrations of stable and radioactive lead (^{210}Pb) in Cladonia sp. lichens from eighteen locations, spanning 4000 km between 77°18' N (Hornsund, Spitsbergen) and 41°20' N (southern Bulgaria) collected in 1965, along the 20° E meridian. As may be seen in Fig. 8 the geographic distribution of ^{210}Pb (a long-lived daughter of the natural radioactive gas radon-222) and of stable Pb are strikingly different.

On Spitsbergen the stable lead concentrations (aver. 2.6 $\mu g\ g^{-1}$ d.w.) were an order of magnitude lower than in Scandinavia, Poland, Czechoslovakia, Hungary and Bulgaria, where they were influenced by the man-made local emissions of lead. On the other hand, the highest concentrations of radioactive lead-210 were found on Spitsbergen. This was probably due to dispersion of this nuclide in the atmosphere by the intensive nuclear test explosions at Novaya Zemlya in 1958 and 1962. The concentrations of lead-210 increased in lichens collected on Spitsbergen after the tests of nuclear weapons (Table 6), in which this nuclide was used as a component of neutron triggers (Kownacka et al., 1989).

A similar study of geographical distribution of heavy metals (Pb,Cd, Cu and Ni) in three species of mosses (Hypnum cupressiforme, Hyloconium spendens, and Rhacomitrium lanuginosum), spanning 3000 km between Spitsbergen and Oker/Goslar in the northern part of West Germany, was carried out in 1983 (Ottar et al., 1986 A). They found a similar geographic distribution of lead as that of Jaworowski (1967). In Germany they recorded a higher level of lead in the southern than in the northern region of the country. The lead level in mosses decreased in Denmark and again increased in southern Norway high above the northern Danish levels, and then decreased in central and northern Norway. Very low concentrations of Pb were found in moss samples from Spitsbergen. In this study it was found that concentrations of lead in plants on Spitsbergen are about an order of magnitude lower than in the southern parts of Scandinavia and in Central Europe.

This was not the case with Cd, the concentrations of which on Spitsbergen were found to be similar to those in Denmark and in central Norway. In southern Norway the Cd concentrations were twice as high as in Denmark. In the moss from Spitsbergen the concentrations of Cu and Ni were similar to those in the samples from central Norway. The highest concentrations of these two metals were found in northern Norway (Figure 9).

Staaland et al. (1983) and Staaland (1985) determined the concentrations of Fe, Cu, Mn, Zn, Co and Mo in the reindeer forage plants (70 species of Lichenes, Bryophyta, Equisetacea, liliopsids, magnoliopsids and forbs) collected during the winters of 1977 and 1978 and summers of 1976 and 1977 at western Spitsbergen and Nordaustlandet. They found extremely high concentrations of Fe, higher in winter than in summer, which were up to about 30 times higher than in continental Norway, about one to two orders of magnitude higher than the dietary requirement and about 10 times above the toxic level for sheep and cattle. However, no toxic effects were observed in reindeer in Svalbard. Concentrations of Cu were in Svalbard vegetation about 2 and of Co about 8 times higher than in continental Norway. Mn, Zn and Mo concentrations were also higher than in mainland Norway, however, not significantly so. This is similar to the findings in the Soviet Arctic, where higher concentrations of minerals in plants were recorded on arctic islands than in continental tundra (Rodin and Bazilevich, 1967). Concentrations of Mn in Svalbard also surpassed the toxic level for sheep and cattle, but no toxic effects in reindeers were observed. The cause of higher content of metals in plants in Svalbard than at the lower latitudes needs elucidation.

Findings of Thomas (1986) differ from the other studies. He determined Zn, Pb, Cu, and Ni in moss, lichens and an ericaceous shrub (Cassiope tetragona) collected at four sites on Spitsbergen. Concentrations of Zn and Cu in mosses were found to be several times lower than in southern Sweden, of Ni about one order of magnitude lower, and of Pb one to two orders of magnitude lower. Concentrations of Zn and Cu were rather homogeneously distributed in various types of plants.

Ottar et al. (1986 A) also reported the results of determinations of polycyclic aromatic hydrocarbons (benzo-α-pyrene) (PAH), chlorinated hydrocarbons (γ-hexachlorocyclohexane) (γ-HCH), and polychlorinated biphenyls (PCB) in the same moss samples in which the heavy metals were analyzed. As may be seen in Figure 10, concentrations of a polycyclic hydrocarbon (PAH) and of PCB on Spitsbergen are very low compared to Denmark and West Germany. In the case of γ-HCH this difference was smaller and the concentrations of this pesticide in mosses from Spitsbergen were similar to those from central Norway.

Fluoranthene, α-HCH, γ-HCH, and benzo-ghi-perylene were determined by Thomas (1986) in samples of four species of mosses, three species of lichens and one higher plant, collected at four places on Spitsbergen: Reinsdyrflya, Lilliehöökfjorden, Ny-Ålesund and Adventdalen. At the first, most northern of these sites the concentrations of α-HCH were found to be lower than in those from three other sites. But the concentrations of three other compounds were similar in all four sites studied. Concentration of fluoranthene in lichens was much higher than in mosses and a higher plant Cassiope tetragona. This is in contrast to the concentrations of γ-HCH, which were similar in mosses and lichens but several times higher in the Cassiope tetragona. Generally the concentrations of organic pollutants in plants on Spitsbergen are lower than in Scandinavia and in Central Europe.

It is astonishing that radioactive debris from nuclear weapon tests, some components of which accumulated in the arctic foodchain by a factor of 100 to 1000 higher than in other regions, was practically not studied in Svalbard plants. The only available data are on ^{210}Pb (Jaworowski, 1967), ^{137}Cs, plutonium and americium levels in a few samples of lichens (Holm et al., 1983), which are insufficient for drawing radioecological conclusions. For elucidation of the cause of the unusually low content of ^{137}Cs in reindeers in Svalbard (see para 2.5) the measurements of the level of this nuclide in contemporary and museum plant and reindeer tissue specimens from Spitsbergen are needed.

2.5. ANIMALS

2.5.1 Radioactivity

In 1980 Kjos-Hanssen and Rennesund (1981) determined concentrations of ^{137}Cs in six samples of muscles of reindeer and in ten samples of ptarmigans, from Colesdalen on Spitsbergen. In ptarmigans the levels of ^{137}Cs were below detection limit. In reindeer muscles ^{137}Cs concentrations ranged between 0.3 and 2.7 Bq kg^{-1} (aver. 1.3 Bq kg^{-1}). These concentrations were about 100 times lower then those found in the same year in reindeer from continental Norway. The cause of the difference is not clear, as fallout of radioactive cesium from nuclear weapon tests was higher in Spitsbergen than in continental Norway (Jaworowski et al., 1981).

One of the possible explanations of this phenomenon is that lichens, which are an important component of reindeer diet in continental Norway (and a main source of airborne pollutants in their body) contribute but a small part to this diet in Svalbard (Staaland, 1985). But this does not explain the two orders of magnitude difference between the two regions. A possible influence of marine algae and minerals in reindeer diet, and of a strong fertilizing effect of the sea on plants in Svalbard, partially via sea birds activity (Staaland, 1985) might be considered. A higher content of potassium (a cesium alkali metal analog, with high ionic radius, which competes with it in mammalian tissues) in reindeer in Svalbard than in continental Norway found by Kjos-Hanssen and Rennesund (1981), supports this supposition. Elucidation of this problem may have important practical implications as the human populations depending on reindeer in Scandinavia, Siberia, Canada and Alaska receive from this source hundreds to thousands times higher radiation dose than other fractions of populations in the same regions (UNSCEAR, 1982).

Lichens contribute large amounts of other than fission products pollutants to the diet of reindeer, caribou and ultimately to humans. In addition to fission products, this special arctic foodchain is best studied for radioactive lead (^{210}Pb) and polonium (^{210}Po) and stable lead. It would be interesting to find out whether the levels of these nuclides in Svalbard reindeer are also lower than in those from continental Norway. In a seal and two polar bears (tissue type not indicated) Holm et al. (1983) found ^{137}Cs concentrations of 1.1 and 6.0 Bq kg^{-1}, respectively and the concentration of $^{239+240}$Pu in polar bears of 4.4 mBq.kg^{-1}. In the birds (Somateria mollissima, Rissa tridactyla, Pagophila eburnea, Uria lomvia, Fulmarus glacialis) ^{137}Cs concentrations ranged from 1.1 to 3.6 Bq kg^{-1}.

2.5.2 Heavy metals

Heavy metals and Se have been determined in soft tissue samples of sea birds, collected in 1980 off the west coast of Spitsbergen near Nordenskjöld Land (Norheim and Kjos-Hanssen, 1984). Pb, Cd, Zn, and Cu have been determined in livers and kidneys and Hg and Se in livers of glaucous gull (Larus hyperboreus), fulmar (Fulmarus glacialis), Brünnich's guillemot (Uria lomvia), little auk (Alle alle) and eider (Somateria mollissima). The mean concentrations found are given in Table 7. The lead levels were below the detection limit of 0.5 µg g^{-1} The levels of Cu and Zn were similar to those in birds from other parts of the world and the author assumed that his results represent normal physiological level. This is probably a correct statement as, except for eider, the content of these two metals in

Svalbard birds are similar to that in contemporary man, where the levels of Cu and Zn in soft tissues decreased recently as compared with the ancient and 19th century levels (Grandjean et al., 1975, Elinder and Kjellström, 1977).

In eider the content of Cu in liver was about 40 times higher than in other birds from Spitsbergen; no explanation of this phenomenon was offered, but Norheim (1987) stated that similarly high concentrations were found in other species of waterfowl without signs of toxic effects. The high level of Cu in eider may result from feeding not on fish but on mussels and other benthonic mollusks. In this context it is interesting to note that the content of organic pollutants in eider is the lowest among the bird species studied in the Arctic (see below).

With the exception of fulmar the concentrations of Cd in birds were similar to those in contemporary man. This may indicate a pollution effect as Cd level in man increased recently by a factor of about ten above the pre-industrial level (Jaworowski et al., 1985).

Carlberg and Bøler (1985) measured Pb, Cd, Cu and Zn in kidneys and also Hg and Se in livers of eight species of birds from Kongsfjorden, Hornsund and Hopen in Svalbard. They found an order of magnitude lower content of Hg than Norheim (1987), which probably reflects a methodological error in one of these studies. The concentrations of other metals found in both studies were similar. Carlberg and Bøler (1985) also found extremely high levels of Cu in eiders. This might be due to the feeding patterns of these birds. The concentration of Hg in the Brünnich's guillemot and kittiwake eggs was similar to that in the livers of these birds.

Carlberg and Bøler (1985) also measured concentrations of 15 metals in seal livers and in muscles of fish and shrimps. The levels of Se, Hg, Cd, Zn, and Cu were similar or lower than in birds from the same region. In freshwater the toxic levels of metals for fish were found to be (in $\mu g\ l^{-1}$) 100–300 for Al, 1.7 to 3.4 for Cd, 100 to 1200 for Cu, 3 to 24 for Hg, 6 to 119 for Pb and 370 to 2400 for Zn (Kling and Grant, 1984). As may be seen in Table 4, in the case of Pb and Cd the lower limits of these toxic levels are sometimes surpassed in Svalbard precipitation. No data exist on the levels of these metals in lakes and rivers in Svalbard. Measurements of concentrations of metals in freshwater and in precipitation would help to define their critical loads in the Norwegian Arctic.

The concentrations of Cd found in ringed seal were 10 to 100 times lower than those found in Eastern Greenland, but those in bearded seal were similar. In the case of bearded seal Hg concentrations were similar in Svalbard and in Eastern Greenland but much lower than in Canada (Dietz, 1987).

Norheim and Nilssen (1989) measured concentrations of Cd and Zn in 78 samples of liver and 60 samples of kidneys of Svalbard reindeer. The concentrations of Zn found in the liver were similar to those in reindeer from continental Norway and also to those in Svalbard birds and in "reference man". However, in kidneys Zn level was an order of magnitude lower in Svalbard reindeer than in Svalbard birds and in "reference man" (Table 7). Elucidation of the causes of low levels of Cd in the liver and kidneys and of Zn in kidneys of Svalbard reindeer needs further study.

Heavy metals have not been determined in the polar bear from Svalbard. The data compiled by Dietz (1987) indicate that in Eastern Greenland concentrations of Cd were an order of magnitude lower in polar bear than in seals, but Hg concentrations were similar. Norstrom et al. (1986) found in 67 samples of liver of polar bear from the Canadian Arctic a relatively high content of Hg; the levels of

Cd, Zn, and Cu were similar to those in reindeer from contenental Norway, and of Cd and Zn similar to those in the Svalbard reindeer. It is interesting to note that the concentrations of Zn and Pb found in polar bear and reindeer were similar to those in "reference man", the concentrations of Cd were lower, and those of Se, Hg, and Cu much higher in polar mammals than in man (Table 7).

2.5.3 Organic compounds

Polychlorinated biphenyls (PCBs) were determined in liver, muscle and fat in seven species of birds from Bjørnøya and Spitsbergen (Bourne and Bogan, 1972). The highest concentration of 535 μg^{-1} was found in glaucous gull, one to two orders of magnitude higher than in other species.

Concentrations of DDE, PCB, and hexachlorobenzene (HCB) were determined in liver and fat samples of five species of sea birds, collected in 1980 off the west coast of Spitsbergen (Norheim and Kjos-Hanssen, 1984). As was also the case in Bourne and Bogan's (1972) findings, the levels of these pollutants were several times higher in fat than in liver tissue. The levels found in birds from Spitsbergen were much higher than in Antarctic birds. This result indicates that the level of pollution of the aquatic foodchain with persistent chlorinated hydrocarbons is higher in the Arctic than in the Antarctic. The authors suggest that the main source of organic pollutants in the Arctic birds is the transport of these substances by the Gulf Stream.

The mean concentrations of PCB in fat of glaucous gull and eider of 82 and 3 $\mu g\ g^{-1}$ wet weight, respectively, found by Norheim and Kjos-Hanssen (1984) in 1980 are similar to mean concentrations reported for the same species from Svalbard by Edelstam et al. (1987) in the same year. A striking feature of data presented in both papers is the much higher concentration of organic pollutants in glaucous gull and fulmar than in other bird species studied. In Svalbard the concentrations of PCB and DDT decreased between 1971 and 1980 by a factor of about 2 in glaucous gull but not in guillemot (Edelstam et al., 19887). The concentrations of PCBs found in glaucous gull from Bjørnøya (Boume and Bogan, 1972) were seven to ten times higher in 1972 than in 1980 in these gulls caught at Spitsbergen (Edelstam et al., 1987). The concentrations of PCB in glaucous gull from Spitsbergen were found to be similar to those in guillemot from the Baltic, i.e. from a sea much more polluted than the Arctic Ocean, but the levels of PCB in Baltic eider were about ten times lower than in glaucous gull and fulmar on Spitsbergen (Edelstam et al., 1987). It seems therefore that these later two species of arctic sea birds are specially exposed to the impact of toxic organic pollutants.

Norheim (1978) determined concentration of PCBs in 44 arctic foxes caught near Longyearbyen, Spitsbergen. A moderate degree of PCB contamination was recorded (Table 8).

As may be seen in Table 8, the content of organic pollutants in seals from Svalbard is much lower than in the birds and similar to that in the polar bear. However, an extremely high concentration of PCB was found in a young bear in 1980, higher than the values in glaucous gull and fulmar. In polar cod, ascidians and in shrimps the contents of organic pollutants were found to be several orders of magnitude lower than in birds.

High levels of polychlorinated hydrocarbons were recently recorded in fish, seals, polar bear and reindeer from the Canadian Arctic. The high levels of these substances were detected in blood of local human populations depending on the Arctic foodchain, two to three times higher than the limits recommended by WHO (Ottar, 1989). Ottar (1989) reported that in seals from Spitsbergen the concentration of dioxine was similar to that from the Baltic. This is in contrast to an earlier study of other organic pollutants indicating that concentrations of DDT and PCB in ringed seal in 1980 were two orders of magnitude lower in Svalbard than in 1971 in the Baltic (Edelstam, 1987) (Table 8). An increasing trend in organochlorine contaminants between 1969 and 1984 was found in polar bears from Hudson Bay and Baffin Bay (Norstrom et al., 1988).

It was recently suggested that if this increasing trend of the PCB concentration in fat of polar bear will continue, the level of PCB will in the year 2005 reach 50 µg g^{-1}, i.e., a point at which the male of the species will lose the ability to reproduce (Cummings, 1989). This level was already surpassed more than twice in a young bear from Svalbard in 1980 (Table 8).

On the other hand, a recent study recorded a decline in total PCB residues in arctic ringed seal between 1972 and 1984 (Muir et al., 1988). Also, reliable data on population levels gathered the since early 1970s do not suggest a decline in the polar bear population in Canada. In the part of Canada where organochlorine levels were highest, the bear population doubled between the 1960s and 1980. It is therefore unlikely that organochlorines are currently having a significant effect on polar bear reproduction in the Canadian Arctic (Norstrom et al., 1988). Also in the Svalbard region the polar bear population probably doubled between late 1960s and 1983 (Larsen, 1986).

The published information on the levels of organic pollutants in animals from Svalbard are rather limited and do not provide a sufficient basis for realistic estimates of possible ecological impact. A systematic study aimed at an assessment of temporal trends and covering all major components of the Arctic foodchain should receive a high priority in future research activity in Svalbard.

Fennical (1981) indicated that approximately 400 chlorinated and brominated hydrocarbons are produced by various groups of marine organisms such as bacteria (Schizophyta), diatoms (Baccillariophyta), dinoflagellates (Pyrophyta), blue-green algae (Cyanophyta), green algae (Chlorophyta), red algae (Rhodophyta) and invertebrates (Porifera, Mollusca, Cnidaria and Annelida). In some of the species the concentrations of the natural halogenated organic compounds reaches very high levels, averaging 1 to 6 % dry weight. The halogenated metabolites are probably produced as defensive environmental adaptations. Marine organisms that produce substances of this nature clearly benefit by enhanced survival against potential predators. These massive amounts clearly distinguish the toxic natural polychlorinated and brominated organic compounds from the ppb or ppm (i.e. 4 to 7 orders of magnitude lower) quantities of halogenated synthetics encountered in pollution analyzes.

Considering the natural massive production of organohalogens by many species entering the Arctic foodchain, the assessment of halogenated pollutant levels (DDT, PCBs etc.) may present difficulties, due to sample contamination. This possibility of interference of natural substances in determination of synthetic halogenated hydrocarbons in the Arctic biota is usually not taken into account in the recent literature. However, a comparison of several synthetic pesticide standards with extracts of several known halogenating organisms, demonstrated that the mixtures of halogenated natural compounds fall within the limits of detectability of PCBs, DDE, DDD and DDT (Fennical, 1981).

The similar chromatographic behaviour of synthetic contaminants with natural substances present in great amounts in the marine environment shed light upon the possible sources of errors in determination of organic pollutants in the Arctic environment (Fig. 11). Elucidation of this needs further studies.

2.5.4 Conclusions

A majority of studies on the content of pollutants in animals in Svalbard was concentrated on birds and reindeer, much less on polar bear, seals and other marine species. Except for reindeer and fox the number of samples analyzed in these studies was scarce, usually between 2 and 10. The tissues studied were almost exclusively liver, kidneys, muscles and fat. It is astonishing that the bone samples were not analyzed for heavy metals, as this tissue reflects the long-term accumulation of these pollutants in the body. Analysis of the samples of bones and antlers, and in the case of birds also of feathers preserved as museum specimens, could be used for a study of temporal changes in levels of pollutants in the Arctic animals during the past several decades.

In Svalbard the level of ^{137}Cs in reindeer in 1980 was very low. As the biological half-life of ^{137}Cs in soft tissues is short (about 100 days) these low levels represented the current situation at the end of the seventies. On Spitsbergen the fallout of ^{137}Cs from nuclear explosions in 1962 at Novaya Zemlya was several times higher than in Northern Norway, where in 1980 ^{137}Cs concentration in reindeer meat was about 100 times higher than in Svalbard. It is possible that this difference is due to a more rapid elimination of ^{137}Cs from the high arctic ecosystem, or due to different nutritional regimes of two groups of animals. If the latter was the case it is possible that the observed difference is caused by a component of the diet which either eliminates this nuclide from the tissues or decreases its intake from the gastrointestinal tract. Elucidating this may have important practical implications.

The concentrations of heavy metals in soft tissues of birds and reindeer in Svalbard are probably representing the natural levels, with the possible exception of Cd in birds. During the past several decades, Cd concentrations increased in precipitation (Jaworowski et al., 1981), in plants (Kjellström et al., 1975) and in the human body (Drasch, 1983; Jaworowski et al., 1985). The temporal changes of its levels in the arctic sea birds might be revealed by a historical monitoring of Cd in the bone or feather specimens and may demonstrate whether the high content of this metal in the arctic birds is man-made or natural. A striking difference between the low level of Cd in reindeer and high level in the sea birds may be due to: (a) long distance transport of this pollutant by the Golf Stream; (b) natural concentration processes in the ocean; and (c) special patterns of the reindeer diet, discussed earlier.

The level of organic pollutants in tissues of glaucous gull and fulmar from Spitsbergen is much higher than in other birds, seals, fox, old polar bears, fish, ascidians and shrimps. However, the highest content of PCB was found in 1980 in a young bear from this region.

No measurements of organic pollutants in Spitsbergen reindeer were reported.

The levels of organic pollutants found in Svalbard animals were much lower than in the same species from the Baltic. The published results of studies carried out in Svalbard do not permit to state unequivocally whether the existing levels of these substances pose a serious long-term hazard to some of the arctic species.

In the light of the data on massive production of natural polychlorinated hydrocarbons by marine organisms, determinations of synthetic halogenated hydrocarbons that were not taking care of inadvertent natural contaminations may be of doubtful value for estimates of the anthropogenic impact in the polar regions exposed to marine influences.

2.6 MAN

No studies on the content of heavy metals or other pollutants in man have been carried out in Svalbard. In Northern Greenland high concentrations of Hg in blood (8 to 40 $\mu g\ l^{-1}$) were found in Eskimos, i.e., much higher than the average values in Denmark of 1.5 $\mu g\ l^{-1}$. Also the blood concentrations of Cd in Eskimos were found to be appr. twice as high as in Denmark. Similar high values were recorded in the Pacific populations which, as is also the case with Eskimos, depend on marine food supply (Grandjean, 1989).

In six 500 years old skeletons found in frozen state in Western Greenland a low mean concentration of Pb of 0.21 $\mu g\ g^{-1}$ d.w. was recorded (Grandjean, 1989), i.e. similar to that in pre–historic bones from Europe and from other regions, and lower than in contemporary Denmark by a factor of about seven (Grandjean, 1988). This can be compared with Pb levels in medieval populations of Europe reaching skeletal concentrations of Pb 100 to 10,000 times higher (Jaworowski et al., 1985; Grandjean, 1988; Jaworowski, 1989).

Several tens of human skeletons from a period between the 17th and 19th century were collected on Spitsbergen at Smeerenburg, Ytre Norskøya, Amsterdamøya, Russekeila and Hornsund. Some of the remains were preserved in the permafrost in a mummified state (Susan Barr, 1989 – personal communication). They may be regarded as unique material for studying the historical changes of levels of pollutants in man, due to their excellent state of preservation. In contrast to Eskimo bone samples from Western Greenland, the Spitsbergen specimens are representative for contaminated European population which was in the past exposed to ubiquitous sources of metals in household utensils, medicines, beverages etc. Determination of concentrations of Pb, Cd, Zn, Hg, Cu, V and Ag in these bones could provide valuable information on historical changes of levels of toxic heavy metals in Europeans.

3. SENSITIVE PARTS OF THE NORWEGIAN ARCTIC

3.1 SOIL AND BOGS

Heavy metals and acids from distant emission sources probably do not pose a significant hazard neither to the terrestrial nor marine ecosystem in the Norwegian Arctic. The critical load of sulfur for soils poor in carbonates of <3 kg ha^{-1} y^{-1} and >32 kg ha^{-1} y^{-1} for carbonates rich soils was recently proposed (Nilsson and Grennfelt, 1988). Assuming conservatively that in Svalbard the average content of SO_4^{2-} in total precipitation is 562 µg kg^{-1} of H_2O (this is maximum value found recently on Spitsbergen, see para. 2.2.1), and that the average precipitation is 300 kg m^2 y^{-1}, this would correspond to about 1.7 kg of SO_4^{2-} per ha per year, (0.6 kg S per ha per year) i.e. below the range of critical loads for sulfur in soils both poor and rich in carbonates. The critical load for nitrogen in raised bogs is 3 kg N per ha per year (Nilsson and Grennfelt, 1988). The concentration of nitrates in the total precipitation in Svalbard ranged from 5 to 85 µg NO_3 kg of H_2O (para 2.2.1). Assuming that the average value is 50 µg kg this corresponds to about 0.15 kg NO_3 per ha per year (0.04 N per ha per year). This value is much lower than the critical load of nitrogen for raised bogs in lower latitudes. However, the critical loads calculated for lower latitudes might possibly not be valid for Svalbard due to different climatic conditions. More knowledge is required on acid fallout and soil chemistry before more firm conclusions than these tentative ones can be drawn.

3.2 TUNDRA

Pollution from human settlements and industrial activity (acids, fuel spills, gaseous and particulate combustion products, sewage) and mechanical impacts may lead to serious disturbances in the tundra ecosystem (Vik, 1972; Smith. 1977; Krzyszowska, 1986 and 1988). It was found that near the coal mines in Svalbard acidification of soil in limited areas caused total destruction of the plant life. At the abandoned mines, after a period of several decades, the plants started to re-invade the destroyed area. Agricultural measures may enhance this recuperation (see para 2.3.1 and 2.3.2).

Near the human settlements the harm due to mechanical impact was found to affect first the lichens, then the mosses and subsequently vascular plants. This is followed by changes in the ground moisture and by an increase in the fine grained fraction of the soil and by compaction and a greater thaw depth. The chemical effects most dangerous to the tundra environment near the two research stations in Svalbard were petroleum derived spills, because of their spreading through the surface waters (Krzyszowska, 1988). Changes in the soil due to fuel oil spillage and mechanical impact were also reported. The organic pollution (detergents) was limited to the sewage trenches. The effects of fuel combustion on plants in Svalbard remain to be studied.

Moser et al. (1980) reported that $S0_2$ concentrations in air ranging from 0.5 to 1.0 ppm would destroy the photosynthetic capabilities of lichen communities in Alaska. These concentrations are 1000 to 10000 times higher than the levels measured in Svalbard (Ottar et al., 1986 A) and the average thropospheric concentrations (Robinson and Robbins, 1975). On the other hand, low-level SO_2 fumigation was reported to stimulate the growth of plants by relief of sulfur deficiency (Noggle, 1980). Artificial acid mist (pH 2.5 to 4.5) significantly increased the number of leaves and percentage of plants flowering of Bistorta vivipara in Alpine tundra (Funk and Bonde, 1989). In the alpine tundra soil bacterial community, experimentally exposed to 2.5 to 10 times the normal loading rates of H_2SO_4/HNO_3, SO_4^{2-}/NO_3^- and NO_3^- in aqueous solutions, a decrease in the total soil bacterial population, bacterial diversity, populations of nitrifiers and populations expressing lipolytic activity was observed. In contrast, significant increases in the actinomycete population, fungal population and populations expressing proteolytic activity were associated with and increase in acid soil loading rates (Macinelli, 1986).

3.3 BIRDS AND POLAR BEAR

It seems that two species of birds (predatory glaucous gull and fulmar) are most exposed to the effects of chlorinated hydrocarbons, probably entering the Arctic from distant source regions. It was also suggested that the polar bear may also be exposed to dangerous levels of these substances. This problem needs further study (see para 2.5.3 and 6.5.8).

3.4 MARINE BIOTA

Transport of heavy metals and toxic organic compounds into arctic waters with the Gulf Stream from northern Europe and America might lead to increase of their concentrations in the arctic marine biota and to deleterious biological consequences. Concentrations of Cd, Pb, Cu and Ni in the surface and deep waters in the Eastern Arctic Ocean in the Svalbard region were found to be similar to those in the Norwegian Sea and the North Atlantic, and an order of magnitude lower than in the Baltic Sea (Mart and Nürnberg, 1984). However, the available information is too scarce at present to permit one to state unequivocally whether the marine and aeolian transport of impurities could pose a real problem now or in the future.

Radionuclides from industrial effluents in Great Britain and in France were found to be transported with the sea currents to the High Arctic. Bioindicators (algae Fucus vesiculosus, Ascophyllum nodosum and others) were used for a study of this transport. It was found that technetium-99 originating from the British and French plants, reprocessing uranium from nuclear reactors, is guided along the Norwegian coast by the Gulf Stream and reaches the Svalbard and Greenland coasts. Due to dilution in the oceanic masses, the activity concentrations of this nuclide found in fucus at Spitsbergen (8 to 23 Bq.kg^{-1} dry weight) was in 1980 and in 1981 lower than at the coast of continental Norway (5.6 to 240 Bg.kg^{-1} dry weight) (Holm and Rioseco, 1987). The transport time from the British Sellafield reprocessing plant to East Greenland waters was estimated for technetium-99 to be approximately seven years. Although the annual discharge rates of this nuclide from the Sellafield plant decreased by a factor of 50 between 1978 and 1982, its concentration in Fucus vesiculosus and Fucus disticus from Greenland waters increased during that time about four times. The transfer factor to fucus in East Greenland waters is about 150 Bq ^{99}Tc kg^{-1} dry weight per PBq ^{99}Tc discharged annually into the sea from Sellafield (Aarkrog et al., 1987 A).

Similar studies were carried out also in the case of radioactive cesium, plutonium and americium (Aarkrog et al., 1987 B; Holm et al., 1983; Holm et al., 1986). The transfer factor for Greenland coastal seawater contaminated by the radioactive discharges from the Sellafield plant was calculated as 0.1 to 1 Bq m^{-3} per PBq a^{-1}. The transit time for ^{134}Cs and ^{137}Cs from Sellafield to East Greenland, based on seawater measurements, was estimated to be seven years (Aarkrog et al., 1987 B).

It was assumed that 25 % of ^{137}Cs present in the seawater and marine sediments in the region between Northern Greenland, Svalbard and Frans Joseph Land originate from fallout from nuclear weapon tests, 25 % originate from mixing with Atlantic waters from more southern latitudes, and the remaining 50 % were assumed to originate from the British and French nuclear reprocessing plants (Holm et al., 1983).

In seawater samples collected in 1980 near Svalbard the concentration of $^{239+240}$Pu was about 15 mBq m^{-3}, and the ^{241}Am/$^{239+240}$Pu ratio in mixed water layer was 0.13. This activity is believed to originate mainly from the British and French nuclear reprocessing plants (Holm et al., 1983).

The concentrations of plutonium and americium in sea algae (Fucus) from Svalbard were 130 mBq kg^{-1} and 8 mBq kg^{-1}, respectively, i.e. one to four orders of magnitude lower than those of ^{137}Cs.

The concentrations of ^{137}Cs in seal, polar bear, and in five species of beards (tissue types not indicated) ranged between 1.1 and 6.0 Bq kg^{-1}; the highest activity concentration was found in polar bear. In the polar bear the concentration of $^{239+240}$Pu was three orders of magnitude lower than the concentration of radioactive cesium (Holm et al., 1983).

The concentrations of ^{137}Cs and actinides in the arctic marine biota, polar bears and sea birds are very low. They may serve as a tracer for study of migration of these substances, but could now hardly lead to deleterious effects (Aarkrog et al., 1983).

4. EARLY WARNING BIOINDICATORS FOR POLLUTANTS FROM DISTANT SOURCE REGIONS

A bioindicator is an organism whose content of a pollutant (e.g. radionuclide, stable metal or pesticide) is used to indicate the level of contamination in the environment in which the organism is living. The use of bioindicators has several advantages when compared with other types of samples such as air, water or soil: (1) An *integration* of fluctuating levels is made over a period of time; (2) due to *accumulation* sensitivity of detection improves (the limit of detection is lowered); (3) an estimate can be made of *transfer to higher animals and man via* edible organisms; and (4) there is the highest sensitivity for the most *biologically available* substances, i.e. if a pollutant exists in a certain form, which is not accumulated in the bioindicator, it is, ideally, not accumulated in the foodchain, and is thus less important than if it had existed in a highly available form (Aarkrog, 1985). In the Nordic Bioindicator Project blue mussel (*Mytilus edulis*) and a brown alga, bladder wrack (*Fucus vesiculosus*) were proposed as marine bioindicators for radioactive effluents in lower latitudes (Aarkrog, 1985). The results 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. It seems that *Fucus vesiculosus* may be used in the Arctic also for non-radioactive pollutants. *Mytilus edulis* which does not occur in the High Arctic, might be replaced as bioindicator with *Astarte borealis*, *Macoma moesta* or *Mya truncata*.

Mosses, lichens and fungi are excellent terrestrial bioindicators for airborne pollution. They are abundant in Svalbard and the concentrations of heavy metals, radionuclides and organic compounds in them may be used for monitoring long-term trends in arctic pollution. As one may expect significant local variations in the levels of pollutants, depending on the patterns of particular microenvironments, it is necessary to assign permanent collection sites for the long-term terrestrial monitoring.

The adverse effects of the acidic deposition in Svalbard might be aggravated by severe climatic conditions. Long before ion inputs to an ecosystem could develop any toxic effects in some tissues or individuals, they influence the competition between organisms in population and biocoenoses (Ellenberg, 1988). Therefore observation of the changes in composition of plant biocoenoses could be used as an early bioindicator of environmental pollution.

5. REGIONAL EFFECTS OF CLIMATIC CHANGES IN THE ARCTIC

From the point of view of environmental pollution, probably one of the most important regional effect in the Arctic of climatic warming could be release of methane from the peat bogs. Some climatic models predict that the regions of frozen tundras in the Scandinavian, Alaskan, Canadian and Siberian Arctic will warm more than other parts of the planet, by between 6 – 8°C in the coming 50 years. As the frozen bogs melt, they may release huge amounts of methane trapped in the permafrost (Pearce, 1989). However, the temperature recordings in Svalbard indicate that the mean summer temperature has been fairly stable since 1920 and its fluctuactions had a rather increasing trend (Hagen and Liestøl, 1989) (Figure 12).

To the total warming effect of "greenhouse gases" methane contributes about 18 %. Much of the methane generated in the past in the northern swamps is locked in the permafrost. If the permafrost begins to melt, the methane will be released. Probably about 20 % of the methane released into the atmosphere today is ancient, and much of this may come from melting permafrost. Melting the vast frozen peat bogs in the Arctic due to warming of the climate might trigger a powerful feedback mechanism.

Changes in methane releases from the bogs are very sensitive to temperature fluctuations. Several climatic change models indicate that in Svalbard one may expect a higher and earlier warming than in many other regions of the world (Farmer, 1989). Monitoring the methane release over the frozen bogs and peats in Svalbard, some of which reach a substantial thickness (Låg,1988), may provide a signal of cumulative effect of greenhouse warming and confirm or refute the modelling results.

A second source of ancient methane lies at the bottom of the Arctic Ocean in the form of methane hydrates, the stability of which depends on low temperature and high pressure. If warmer waters penetrate to the bottom of the Arctic Ocean, the methane may be released. The total amount of methane tied up as sea bottom clathrates was recently calculated to be 10,000 billion tons of carbon (i.e., more than all the known coal reserves of the world) (McDonald, 1988). If the Arctic part of these deposits were released by global warming, it might contribute to the climatic change.

According to Gordon (1980), recent temperature trends in the Arctic have been characterized by a general cooling between the mid-1950s and the late-1960s followed by a return to warmer conditions in the early 1970s. A marked decrease in summer temperatures of air, and an increase in winter precipitation which occurred after 1963 were reflected by decreased glacier mass loss and increased glaciation in the Canadian Arctic Archipelago and in West Greenland (Gordon, 1980). Since about 1968, the advance of the cirque and small valley glaciers coincided in West Greenland with a period of decreased summer temperatures, and could be seen as a direct response to this climatic deterioration. After 1971, a reversal of this temperature trend occurred, but six out of nine glaciers studied continued to advance at least until 1978. Therefore it seems that the glacier margin fluctuations match only partly the recent temperature trends. The continued advance of the Greenland glaciers probably reflects a lag in their response to increased precipitation during the 1950s and 1960s. It was suggested that this lag might be from at least 9 to as much as 30 years (Gordon, 1980).

Detailed annual observations at two glaciers at West Spitsbergen indicate that a cumulative mass loss between 1966 and 1988 was more than 10 % of the volume in 1967. Zero balance would be obtained if the summer temperature was lowered about one centigrade or if the winter precipitation increased about 50 % (Hagen and Liestøl, 1989). These authors found no sign of climatic warming through increased melting. The trend analysis of their data from the last twenty years shows stable conditions with a slight increase of the winter balance. The net balance is then increasing slightly and thus less negative than twenty years ago.

Lefauconnier and Hagen (1989) also reported a decrease in 1969–1988 of the long-term (1912–1988) net balance deficit at the Brøgger Glacier in West Spitsbergen.

It was suggested that the changes in the ice mass balance in the arctic glaciers were related to great volcanic eruptions (Bradley and England, 1978). These authors supposed that in the period 1750 – 1880, when there were at least 14 eruptions of a magnitude equal to or greater than that of Gunung Agung in 1963, ablation season temperatures in the High Arctic were extremely low, and the glacier mass balance was almost certainly positive, as indicated by stratigraphic studies of the Devon Ice Cap and the Gilman Glacier in the Canadian Arctic. The period 1920 – 1963, when volcanic activity was exceptionally low, was characterized by negative mass balance conditions, observed at the Devon Ice Cap and White Glacier on Axel Heiberg Island. The mass balance conditions in the American Arctic of the post-Agung period may be more typical of colder conditions characteristic of the past century (Bradley and England, 1978). These glacier studies indicate that the retreat of glaciers was recently less dramatic than at the beginning of this century, after the end of the Little Ice Age.

The ocean level could rise up to 1.4 m if glaciers all over the world would melt due to an assumed increase of average surface air temperature of 4^{o}C until the year 2050 (Orheim, 1989). This would certainly have an adverse impact on the low lying terrestrial ecosystems in Svalbard. But it is also possible that the greater humidity of the air, due to increased evaporation from the oceans will result in increasing precipitation in Antarctica, which might overcompensate the effect of melting of the ice and even lower the current sea level.

If, due to climatic changes, the temperature and precipitation rate increase in the Norwegian Arctic, this could lead to substantial changes in terrestrial ecosystems, due to increased productivity of plants and due to their progression to areas now covered by glaciers. Increasing the thickness of the active layer of permafrost would also cause important changes in the soil processes and in plant life.

The past observations of fish populations in the Barents Sea indicate that in warmer periods their biological parameters improve and their feeding area increases dramatically. This influences other populations depending on the marine foodchain (Loeng, 1989). Therefore one might suppose that in Svalbard the climatic warming could have a rather beneficial impact on both marine and terrestrial ecosystems. However, a total melting of sea ice could lead to disappearance of the feeding grounds of the polar bear, and of the breeding grounds for seals. This might possibly cause their extinction.

The possible changes in the arctic ecosystem, due to predicted climatic warming, and the socio-economic consequences of these changes deserve a closer study.

6 PRIORITIES FOR SHORT- AND LONG-TERM RESEARCH ON POLLUTION OF NORWEGIAN ARCTIC ENVIRONMENT

6.1 INTRODUCTION

In this section a list of research objectives for the future studies in the Norwegian Arctic is presented. It may serve as a basis for establishing a priority of research programs, according to available funds, competency, facilities and socio-economic needs. There are four main objectives that should be addressed in future research in the Norwegian Arctic:

(1) Assessment of the deleterious impacts of man's activity on the arctic biota. This is a group of short-term projects, except for the monitoring of dynamics of animal populations which should be continued for a long time.

(2) Historical monitoring of the past and present levels of pollutants for a prompt estimate of the contribution of anthropogenic emissions to:

- (a) the total flow of pollutants into the global atmosphere; and
- (b) contamination of terrestrial and marine biota in the Arctic. This is a short-term project which should be carried out in four to five years, and should produce information for an estimate of the current impact of man's activity on contamination of the Arctic and global environment and a projection of the future impacts.

(3) Long-term monitoring of pollutants in the air, precipitation, fresh and seawater and in the terrestrial and marine biota. This monitoring should be carried out during the next several decades.

(4) Short-term basic research projects on :

- (a) Migration of impurities in the glacier ice and snow and its implications for the methodology of glacier pollution research.
- (b) Natural production of the chlorinated hydrocarbons and its impact on the analytical quality of the determinations of synthetic halogenated organic compounds in the arctic environment.
- (c) Emission of volatile chlorinated and brominated hydrocarbons into the atmosphere from natural sources in the Arctic Ocean.
- (d) Transfer of pollutants in the marine and terrestrial arctic foodchain.
- (e) Enrichment of metals in the atmosphere and hydrosphere.

6.2 ASSESSMENT OF THE DELETERIOUS IMPACTS OF MAN'S ACTIVITY ON THE ARCTIC BIOTA

This study should be based on the observations of the frequency of pathological changes and of population dynamics of the arctic species which are most endangered by man's activity, and on measurements of concentrations of pollutants in their tissues.

6.2.1 Tundra.

The local human activity is now probably the only negative factor influencing the tundra ecosystems. Studies such as those discussed in para. 2.3.1 on the effects of local industrial activity and on the influence of human settlements on the tundra ecosystems should be continued. The expected increase in research activity and tourism in Svalbard may also lead to an increase of the rate and area of tundra degradation above the current level. A monitoring program should cover the phytosociological recording near the settlements, industrial objects, scientific and radio stations and communication routes. The area and degree of mechanical destruction which first affect lichens, then mosses and subsequently vascular plants, should be recorded and the future trends should be monitored during a longer period of time.

The recommendations for minimizing the tundra degradation should be elaborated and implemented. In the areas already affected by tundra degradation due to acidification of the soil, e.g., near the mines, a program of recultivation of the original vegetation should be considered.

A systematic study on the concentration of heavy metals (Pb, Cd, Zn, Cu, and V) and of combustion derived organic pollutants in plants should be carried out, in relation to the distance from human settlements, industrial objects and communication routes. The results should be compared with measurements carried out in plants from parts of Svalbard not affected by local emissions.

6.2.2 Animals.

Two species of sea birds seem to be most exposed to the effects of contamination by organic pollutants, i.e., glaucous gull and fulmar. Recently it was suggested (rather too precisely) that by the year 2005, the polar bear will lose the ability to reproduce and be forced into extinction due to accumulation of PCBs in the body (Cummings, 1989). As was discussed in para. 2.5.3 this seems doubtful in view of the population studies in Canada and Svalbard. However, this problem deserves a careful study. The following objectives of research are proposed:

(1) Determination of synthetic chlorinated hydrocarbons in a representative number of samples of tissues of the major participants of the arctic foodchain in Svalbard: bacteria, blue-green algae, green algae, red algae, invertebrates, fish, seals, birds, polar bears, foxes and reindeer. Care should be taken to resolve the analytical problems indicated in para. 6.5.8.

(2) Monitoring of the pathological changes in livers, kidneys and other tissues of fish, sea birds and polar bears, and in eggs of sea birds.

(3) Monitoring of the population dynamics of sea birds, seals and polar bears.

(4) Determination of the toxic heavy metals (Pb. Cd, Zn, Cu, V, and Hg) in the same set of samples as in (1).

6.3 HISTORICAL MONITORING OF POLLUTANTS

6.3.1 Temporal changes of pollutants in glaciers

The Svalbard area seems to be more exposed to atmospheric transport of impurities from the Eurasiatic and North American continent than other arctic regions. Therefore Svalbard is especially suitable for observations of the long term trends in the concentration of natural and man-made pollutants in the atmosphere of the Northern Hemisphere. These trends are recorded in Svalbard glaciers as concentrations of pollutants in precipitation deposited in annual ice strata. A difference between the average concentrations of impurities in the ice from the pre-industrial period and those in the contemporary ice is a suitable measure of man's contribution to atmospheric pollution.

The temporal trends in the concentration of pollutants in the ice cores from Svalbard glaciers should be compared with those collected in glaciers in continental Norway, which are more exposed to regional sources of anthropogenic pollution. The following glaciers are proposed for this study:

(1) Central part of Austfonna (Nordaustlandet), Svalbard
(2) Sørkappfonna (Sørkappland), Svalbard
(3) Seilandsjökelen (Finnmark
(4) Okstindan (Nordland)
(5) Storbreen (Jotunheimen)
(6) Folgefonna (Hordaland)

This large sampling net spans the territory between 60^{o} N and 80^{o} N and enables one to determine the temporal trends in atmospheric loads of impurities in relation to local, regional and hemispheric sources of emission.

The ice cores and blocks of ice and firn from these glaciers should cover a period of the past several hundred years, which witnessed the greatest increase in man's industrial activity. In these samples the following impurities are proposed to be determined:

(1) organic dust and its morphology, mineral dust and its size distribution and mineral composition
(2) graphitic carbon
(3) SO_4^{2-}, NO_3^{-}, NH_4^{+} and pH
(4) heavy metals (Pb, ^{210}Pb, Cd, Zn, Cu, V, Cr, Sn, Mn, Hg), Al and Se
(5) fission products and natural radionuclides
(6) chlorinated organic compounds
(7) brominated organic compounds

This study will enable estimation of the pre-industrial and contemporary flows of particulates, metals, radionuclides and organic pollutants into the global atmosphere, and also a quantification of man-made contribution to these flows. It will also help to assess the impact of the current long-range atmospheric transport of pollutants from industrial emission sources on the arctic environment. The long term temporal changes in ice strata of graphitic carbon, chlorinated hydrocarbons and brominated hydrocarbons have never been studied in the Arctic. One should note that the glaciers in Svalbard and in continental Norway, are suitable for this study, as they are situated along the main routes of eolian transport of impurities from the Euroasiatic continent into the High Arctic.

The central parts of Austfonna at Nordaustlandet and Sørkappfonna in Sørkappland represent excellent locations for this study due to a lack of permanent settlements, and due to large ice surfaces not exposed to the effects of the local rock dust resuspention. The results of an analysis of chemical composition of contemporary and pre-industrial ice from these locations in comparison with glaciers from continental Norway will bear importantly upon the estimates of the magnitude, time changes and the origin of flux of metals, acids and organics into the global atmosphere from natural and anthropogenic sources.

Information provided by these determinations will also be important for understanding the impact of man-made dust emissions from combustion of fossil fuels on climate, emissions of synthetic halogenated organics on the arctic biota and emissions of natural brominated hydrocarbons on the stratospheric ozone layer.

6.3.2 Lacustrine and marine sediments and peat.

Many metals and organic compounds are adsorbed and immobilized by clay minerals and organic material contained in the sediments. Although they are exposed to physical mixing, resuspension and bioturbation, these sediments are often used for historical monitoring of the past pollution. Some problems are also caused by the sediment-water interface where, due to a major redox boundary, metals and possibly also organic pollutants are not immobilized and can migrate from the sediment. This casts some doubt on the validity of this material for historical monitoring.

Lacustrine and estuarine sediment studies were carried out mainly in lower latitudes and the findings were representative rather to local or regional situations, as the sediments are predominantly influenced by the input of impurities to the particular catchment area. The increasing temporal trends in the levels of heavy metals and polycyclic aromatic hydrocarbons (PAH, DEHPs, PCBs, CB, DDT, DDE, TDE) were demonstrated in sediments in many lakes in the Unites States and Europe. These lakes are exposed to inflow of impurities from the local and regional industrial sources of pollution (Alderton, 1985). Such studies were not performed in the Arctic, where they could be more representative for the global or hemispheric changes.

The sediment cores in Svalbard should be collected from lakes and fjords exposed to local influences of industrial and research activity in the region of Longyearbyen, Ny-Ålesund, Barentsburg and Hornsund and in pristine locations far from human settlements. In the cores the concentrations of heavy metals (Pb, Cd, Zn, Cu, V, and Hg) and of organic pollutants should be determined.

An analysis of ombrotrophic peat bog profiles, used for investigating the temporal trends in pollution in the United States and Europe, has not been performed in the Arctic. Although redistribution of the pollutants due to physicochemical processes could affect or even invalidate the result of temporal trend studies in some locations, in favourable conditions peat cores can record qualitatively these trends (Coleman, 1985). In Svalbard, in addition to other types of historical monitoring, analysis of peat cores may be applied for studying the effects of local man-made emissions of heavy metals using the samples collected near the human settlements, and of long-range emissions in samples collected in remote sites.

6.3.3 Temporal changes of pollutants in biota.

The main pollutants to be determined in this study are heavy metals and chlorinated hydrocarbons, both of which may be of natural and of man-made origin. Observations of the temporal trends in the levels of these substances in the components of the arctic foodchain will help to estimate the impact of man's activity on the contamination of the arctic biota. The results of this study might also provide information on natural levels of these substances in marine and terrestrial organisms.

For this aim the 19th century and beginning of this century samples of plants and teeth, bones, hair, feathers, antlers, egg shells and soft tissues of marine and terrestrial animals housed in museums should be used in comparison with the modern specimens.

The following organisms should be included in this study:
lichens, moss, fish, seals, polar bears, polar foxes and reindeer. Through cooperation with Soviet scientists, it might be possible to include in this list also mammoths, many of which were recently found in an excellent state of preservation in Siberian permafrost. Special care should be taken to check the possibility of inadvertent contamination of the museum samples during long storage and by conservation procedures.

The following substances are proposed to be determined in these samples:

(a) heavy metals (Pb, Cd, Zn, V, Cu, Hg, Mn, and Ag), Se and Al. These elements should be determined in samples covering the period of at least the last 150 years, but where possible also a longer one.

(b) Chlorinated hydrocarbons (DDT, DDD, DDE, HCH, HCB, PCB, dioxine). These compounds should be determined in the samples covering the period of the past 60 years.

6.3.4 Temporal changes of pollutants in ancient inhabitants of Svalbard.

As discussed in para. 2.6 a large number of human remains from the 17th to 19th century have been recovered in Svalbard. These are housed in various museums. Some of these remains were mummified in permafrost where changes due to the fossilization processes were eliminated. Therefore these remains can be regarded as excellent material for a historical study of pollution of Europeans by heavy metals. In the bone, hair, and soft tissue samples from these remains the following heavy metals should be determined: Pb, Cd, Zn, Cu, Ag, V and Hg.

6.4. LONG-TERM MONITORING

6.4.1 Air and precipitation.

Svalbard is a unique remote site, exposed to almost concentric transport of natural and man-made pollutants from the vast land masses of the Northern Hemisphere. It is also exposed to maritime influences, and it is a sink region for the global upper tropospheric and lower stratospheric transport of airborne impurities. The special meteorological and insolation conditions make Svalbard prone to be a region of accumulation of anthropogenic and natural pollutants. As a consequence one may expect that any adverse biological effects of a long-range transport of pollutants would appear there earlier than in other pristine regions (see para. 2.8).

It seems, therefore that localization of a permanent monitoring station in Ny-Ålesund, Svalbard (Svalbard Environmental Monitoring Station) could be of similar importance for understanding the atmospheric chemistry, long-range atmospheric transport processes and temporal trends in pollution of the global atmosphere, as the Mauna Loa observatory in the tropical zone in Hawaii. One should also note that the Mauna Loa observatory is situated by an active volcano, which emits gases and particulates. The effects of rich tropical vegetational activity are also detectable there. On the other hand, the only quaternary volcano in Svalbard (Sverrevulkanen near Woodfjorden) is an inactive one. The Spitsbergen site would serve as a second reference point at the longitudinally opposite side of the Northern Hemisphere representative of the polar deposition region of the Hadley cell circulation system (Figure 1). This site is much more exposed to the influence of emissions from the Northern Hemisphere land masses than the Hawaiian site. This is due to the frequent low pressure systems in the Arctic, transporting the polluted air masses from the lower latitudes. From this point of view the Spitsbergen research and monitoring station may become the most important remote observatory in the eastern part of the Northern Hemisphere.

The following measurements of concentrations of atmospheric impurities are proposed to be carried out at the Svalbard Environmental Monitoring Station on a long-term, secular basis in the ground level air (if not stated otherwise):

(a) Carbon dioxide, methane and dinitrogen oxide

(b) Mineral particulates, including soot carbon

(c) Ozone

(d) Ozone in stratospheric layer by optical methods and ozone sondes

(e) SO_2, sulfates, dimethylsulfide and nitrates

(f) Polychlorinated hydrocarbons

(g) Natural radionuclides (radium-226, lead-210, radon-222 and its short-lived daughters), radioactive fission products and tritium.

(i) Heavy metals (Pb, Cd, Hg, Zn, V, Ti, Mn, Mg, Cu, Ni)

(j) Other metals and ions (Al, Fe, Na, Cl, Ca, K)

(k) Organic bromine

(l) Atmospheric extinction

(m) Systematic high altitude aerosol sampling in winter and summer for determination of sulfates, nitrates, heavy metals, organic compounds, radionuclides and particulates.

The following long-term measurements of impurities in wet and dry fallout are proposed to be carried out in the Svalbard Environmental Monitoring Station:

(a) heavy metals (Pb, Cd, Zn, V, Ti, Mn, Mg, Cu, Ni)
(b) other metals and major ions (Al, Fe, Na, Cl, Ca, sulfates, nitrates, pH)
(c) polychlorinated hydrocarbons
(d) radionuclides
(e) particulates

The following impurities are proposed to be measured in fresh and seawater in Svalbard:

(a) freshwater – Pb, Cd, Zn, Cu, Hg, Al, acidity (pH), sulfates, nitrates;
(b) seawater – radionuclides, polychlorinated hydrocarbons, Pb, Cd, Hg.

For both types of measurements permanent sampling points should be established for the long-term systematic observations.

6.4.2 Terrestrial and marine biota.

The long-term monitoring of the levels of heavy metals and organic toxic compounds should be carried out in fish, seals, sea birds, polar bears, polar foxes and reindeer in Svalbard. After an initial stage described in para. 6.2.1 and 6.2.2 collecting of the samples and measurements of these substances should be systematically carried out at three to five year intervals during the next several decades. The aim of this monitoring is to control the late effects of practices which are: (a) now being carried out, (b) are already discontinued (e.g., production of PCBs), and (c) will be discontinued in the future. This monitoring system could be used for any of the new pollutants which might be introduced into the environment by future practices and which could enter the arctic foodchain.

In addition the typical bioindicators (discussed in para. 4), i.e., lichens, moses, fungi, mussels and Fucus vesiculosus should be collected and analyzed.

A part of the bioindicator samples should be systematically stored for future studies, without being analyzed. The housing for these samples could be arranged at the permafrost temperatures in one of the abandoned mines in Svalbard.

6.4.3 Climatic effects in permafrost, bogs and peats.

A long-term monitoring of the possible effect of climatic warming on the level of the permafrost and on emission of CO_2 and CH_4 from bogs and peat in Svalbard should be envisaged.

6.5 BASIC RESEARCH

6.5.1 Radial migration of impurities in the glacier ice cores.

The aim of this project is to study the capillary transport phenomena in the ice cores at low temperature and to check the reliability of the current methods for determination of pollutants in the glacier ice.

The "Chernobyl layer" formed in May 1986 in Jotunheimen glaciers offers an unique opportunity to carry out this study. After the Chernobyl event the activity of radionuclides deposited on the surface of some Scandinavian glaciers was higher than ever before. In 1986 in the Jotunheimen Mountains the deposition of Cs–137 was more than 100 000 Bq m^{-2}. In contrast, during the past 34 years, all nuclear weapon tests caused a cumulative deposition of ^{137}Cs of only 300 to 600 Bq m^{-2} in Norway (Henriksen and Saxebøl, 1988). The average concentration of ^{137}Cs in Storbreen in Jotunheimen, during the period of the past nuclear test explosions in the atmosphere, was 0.028 Bq kg^{-1} (Jaworowski et al., 1975). One may expect that the activity concentration of the "Chernobyl layer" is about two orders of magnitude higher. The cores of ice collected from this layer, spiked quasi naturally by a long–distance fallout, enable us to study the radial migration of impurities in the ice samples, as the possibility of external contaminations with radioactive species is excluded in this case.

6.5.2 Vertical migration of impurities in the glacier ice strata.

The "Chernobyl layer" in Norwegian temperate glaciers offers an unusual opportunity for studying in situ the vertical migration of impurities in the ice strata due to the percolation of meltwater and intercrystalline migration through the capillary network at low temperatures. This study is needed for validation of the use of temperate glaciers for historical studies of the trends in the global atmospheric pollution. The near zero centigrade temperatures of these glaciers make it possible that the impurities from younger layers probably penetrate into the deeper ones and contaminate the old ice with modern pollutants. The vertical extent of this phenomenon and its rate can now be checked with the Chernobyl trace radionuclides in Norwegian glaciers.

6.5.3 Relation between concentration of impurities in air and in wet and dry precipitation.

The ratio between the level of pollutants in the air and in the surface annual layer of snow at arctic glaciers depends on variations in concentrations of pollutants at various atmospheric altitudes. Pollutants are falling out as dry deposition due to gravitation and eddy diffusion, and are leached by rain and snow. The changing meteorological conditions cause large short–term variations in this ratio, and therefore the validity of ice and snow measurements for defining the changes of atmospheric pollutants is sometimes questioned. In a longer period, e.g., during one year, the variations in deposition are smoothed out by an integration of particular types of deposits in the annual snow layer. But the relationship between the integrated deposit and the concentration of pollutants in the ground level air may be spurious, as the integrated deposit depends not only on the ground level air concentration of pollutants but also on their content at higher altitudes.

In the former studies on the air/precipitation ratio of pollutants were carried out for short periods, not long enough for assessment of the representativeness of snow and glacier ice measurements for the changes of pollutants in the atmospheric air. An inventory of pollutants collected during several years at the same location from the ground level air filters, and with the rain and snow samples, and as a dry fallout, could elucidate this problem. This study should be carried out at the Svalbard Environmental Monitoring Station for several years.

6.5.4 Verification of assessment of radioactive contamination of the Northern Hemisphere by the past nuclear weapon tests in the Arctic.

The United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR) assumed that in the 80^{o}N to 90^{o}N latitude band, the integrated deposition of fission products was only 0.2% of the Northern Hemisphere deposition (UNSCEAR, 1982). This assumption was used for calculation of the radiation dose received by the global population from the nuclear tests performed in the arctic atmosphere. This assumption, however, does not seem to be a realistic one, in view of the location of the testing grounds at Novaya Zemlya close to this latitude band, where a dominant part of fission products was released in the Northern Hemisphere. Also, the few measurements from Svalbard indicate that in 1958–1965, the radioactive fallout in the polar regions was higher than in southern and central Norway and similar to that in the Alps (Jaworowski et al., 1981). This suggests that the assumptions taken by UNSCEAR led to an overestimation of the radiation dose received by the inhabitants of the Northern Hemisphere, as a much greater part than 0.2% of the fission products from Novaya Zemlya explosions fell out at the uninhabited polar regions. Determination of long-lived fission products in the glacier ice strata from the years 1958–1965 from northern and southern Svalbard will help to solve this problem.

6.5.5 Lead–210 from nuclear explosions in the Arctic.

Lead–210 is a main component of the natural radioactive fallout. Due to its polonium–210 daughter, an alpha radiation emitter, it contributes a substantial fraction of internal radiation dose received by man from natural sources. Part of this contribution, however, may be of man-made origin, as it was found that during the periods of peak activity of fission products in the atmosphere, the lead–210 concentrations also increased in the air and precipitation. A plausible explanation is that lead–210, in radioactive equilibrium with polonium–210, was used in less sophisticated nuclear explosives in a neutron source trigger enhancing the chain reaction. Recently disclosed large scale production of polonium 210 in the Windscale military installation supports this supposition (see discussion in Kownacka et al., 1989).

Measurements of the fission products and of lead–210 in ice strata in Svalbard glaciers, deposited between 1958 and 1965, will help to estimate the scale of the injection of ^{210}Pb – ^{210}Bi – ^{210}Po chain members into the atmosphere by nuclear explosions. This will help to reassess the radiation dose received by the population of the Northern Hemisphere from this source.

6.5.6 Measurement of the $^{137}Cs/^{90}Sr$ ratio in glacier ice.

United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR) and many students assumed theoretically that the average value of the $^{137}Cs/^{90}Sr$ concentration ratio in the environment is about 1.6. This value was used for calculations of the radiation dose from nuclear explosions to the global population. However, the results of measurements of these two nuclides in air and precipitation published in many countries (e.g., in the long series of US Health and Safety Laboratory reports) indicate that the value of this ratio was probably largely underestimated, as the measured values were often higher by up to two orders of magnitude. One of the possible causes of this discrepancy is venting of ^{137}Cs and of its gaseous precursors from the underground nuclear explosions, 535 of which were performed until the end of 1978, as compared with 422 of atmospheric ones. This contribution to the radiation dose received by the global population was not taken into account in the UNSCEAR and other assessments, which were based mainly on ^{90}Sr measurements, and therefore this dose was probably underestimated. A revision of the average value of the $^{137}Cs/^{90}Sr$ ratio is needed for reassessment of the absorbed dose commitments to regional and world populations evaluated in the UNSCEAR reports for the General Assembly of United Nations.

Determination of concentrations of ^{137}Cs and ^{90}Sr in precipitation deposited between 1958 and 1980 in Svalbard glaciers, i.e. in a location relatively close to the atmospheric and underground nuclear weapons testing ground at Novaya Zemlya, will help to solve this problem.

6.5.7 Enrichment of metals in the atmosphere and hydrosphere.

Heavy metals in the airborne dust and in precipitation are usually enriched as compared to the Earth's crust abundances by two to four orders of magnitude. The causes for this enrichment are probably natural, as this phenomenon is not related to specific geographical locations, i.e., it is similar in urban, rural and remote regions and also both in the ancient and recent precipitations. One of the possible causes of this enrichment is biological and abiotic methylation of inorganic metals. The other cause may be the physicochemical fractionation at the air–sea interface. These and possibly other natural processes dominate the flow of metals into the global atmosphere (Jaworowski et al., 1981). Volatilization of metals by marine biota was observed in shelf waters and tidal marshes at lower latitudes. This phenomenon probably also occurs in the Arctic, and Svalbard waters with their rich marine life is a suitable object for its study.

Three sub–projects are proposed:

(a) Measurement of enrichment values of Pb, ^{210}Pb, Cd, Zn, Cu, Hg and V relative to Al and Mn in the glacier ice samples collected for the historical monitoring (see para. 6.3.1), and in the airborne dust collected on air filters and in precipitation at the Svalbard Environmental Monitoring Station in Ny–Ålesund (see para. 6.4.1).

(b) Measurements of organometallic compounds in the near–surface air and sea water in fjords and at offshore sites in Svalbard.

(c) Determination of enrichment factors of metals as in (a) in seaspray particles formed by exploding seawater foam bubbles.

6.5.8 Selectivity of gas chromatography in determination of synthetic chlorinated hydrocarbons.

This is a methodological research on the validity of gas chromatographic methods, commonly used for determination of synthetic chlorinated organic compounds in the presence of natural ones. Some of these synthetic compounds were predicted to be of a solely anthropogenic origin (Galbally, 1976), but were found to be produced in vast amounts by marine biota. About 400 marine-derived halogenated compounds are known, and the structures of well over 200 of them have been established. These compounds are produced by several orders and families of algae. It is possible that the past analyses of tissues of species depending on the marine foodchain could have yielded erroneous results due to contamination by natural halogenated hydrocarbons, the concentrations of which in marine organisms were found to be four to seven orders of magnitude higher than those encountered in pollution analyzes of halogenated synthetic compounds. The halogenated synthetic pollutants (DDT, PCB, etc.) have in gas chromatograms similar retention times as natural halogenated metabolites, and fall within the limits of detectability of synthetic pollutants. The similar chromatographic behaviour of man-made and natural contaminants might interfere with DDT, DDE, DDD and PCBs analyzes (Fenical, 1981).

The aim of this study is to check the validity of past analyses and to improve the selectivity of analytical methods for determination of chlorinated organic compounds in environmental samples.

6.5.9 Production of brominated hydrocarbons in the marine ecosystem at Svalbard.

There is an increasing evidence that the arctic marine environment is a potent source of organobromine compounds, and that they may be responsible for cyclic destruction of the stratospheric ozone layer over the Arctic in March and April, through the BrO_x radicals produced from them by photochemical reactions (see para. 1 and 2.1.4). This problem deserves a careful study, and the following research objectives are proposed:

(a) Measurements of concentrations of CH_3Br, CH_2BrCH_2Br, $CHBr_3$ and CH_2Br_2 in phyto- and zooplankton, algae and seawater.

(b) Measurements of exhalation of organobromines from the surface of the seawater in fjord and offshore sites at Svalbard.

(c) Monitoring of seasonal variations of organobromines in ground level air in the Svalbard Environmental Monitoring Station at Ny Ålesund.

6.5.10 Transfer of pollutants in the marine and terrestrial foodchain.

As appears from the discussion in para 2.3, 2.5.4 and 2.5 the transfer of pollutants in the foodchain has virtually not been studied in the Norwegian Arctic. On the basis of the measurements carried out as proposed in para. 6.4.1 and 6.4.2 the ecological concentration processes in this region should be studied and transfer factors determined.

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TABLE 1. Comparisons of air concentrations of metals and major ions (ng m^{-3}).

	Spitsbergen	Rural Norway and Sweden	Greenland	Alaska
Pb	0.1–16[a]	0.1–306[ab]	0.8–5[d] 0.11–0–21[e]	
Cd	0.01–1.0[a]	0.01–0.7[a]		
Mn	0.1–3.2[a]	4.7[b]	0.13–3[d]	0.15–3.4[c]
V	0.2–2.5[a]	2[b]	0.12[b]	0.0--20.4[c]
Al	39–187[a]		1.9–18.2[d]	14.7–240[c]
Ni	0.1–1.4[a]	0.1.2.6[a]	0.08[b]	
Na	130[b]			>12–58[c]
K	1.5–49[a]	56[b]	4.–36.4[d]	
Cl	28–659[a]	140[b]	21.5–1170[d]	
S	40--1170[a]		94–206[d]	
Ca	1–64[a]	58[b]	9.1[b]	
Zn	0.8–19.2[a]	1–30[ab]	0.18[b] <0.85–2.1[e]	
Cu	0.1–1.8[a]	2[b]		

[a] Pacyna and Ottar, 1985 – at ground level; [b]. Heinzenberg et al., 1981 – at ground level; [c] Rahn et al., 1977 – at 3.3 km altitude; [d] Heidam, 1986 – at ground level in maritime locations. [e] Davidson et al., 1981 – at ground level, inland location.

TABLE 2. Enrichment factors for metals in air, snow and contemporary glacier ice in remote locations (After reviews in Jaworowski et al., 1981 and Boutron, 1986).

	Ice and snow	Air
As	500	
Se	3000 - 20,000	
Ag	800 - 7000	
Cu	25 - 100	
Cd	200 - 17,000	730 - 3,000,000
Hg	100 - 53,000	14,000 - 120,000
Pb	5 - 2400	10 - 130,000
V	1 - 2	0.0 - 140

Table 3. Radionuclides (pCi.kg–^{1}) and heavy metals (µg.kg^{-1}) in ice from Körber Glacier (contemporary) and Hans Glacier (pre-industrial) on Spitsbergen. After Jaworowski et al., 1981 and 1982.

Age	^{137}Cs	^{210}Pb	^{226}Ra	U	V	Pb	Cd	Hg
1973	0.55	0.95	0.005	0.011	0.15	5.51	0.95	0.10
1972	0.05	<0.04	0.004	0.015	0.44	2.98	0.38	0.07
1971	0.30	0.30	0.005	0.011	1.59	2.69	0.81	0.14
1970	0.14	0.29	0.008	0.011	0.40	2.98	0.51	<0.11
1969	0.22	0.03	0.013	0.111	0.35	2.03	0.58	0.12
1968	0.12	0.11	0.005	0.025	1.71	2.38	0.45	0.15
1967	0.35	0.31	0.008	0.014	0.33	2.35	0.79	0.05
1966	0.17	0.15	0.003	0.030	0.30	2.42	1.07	0.17
1965	15.41	7.63	0.019	0.117	2.90	9.45	4.77	0.13
1964	5.05	0.28	0.006	0.037	2.18	8.11	1.83	0.11
1963	3.82	0.39	0.011	0.035	<0.15	3.99	2.35	0.30
1962	0.95	0.26	0.006	<0.009	0.38	4.40	1.24	0.10
1961	2.47	0.28	0.007	0.005	<0.15	3.15	0.94	0.08
1960	3.58	0.78	0.005	0.018	0.88	3.77	1.12	0.12
1959	5.57	0.81	0.006	0.022	1.00	5.58	1.08	0.11
1958	18.53	3.56	0.012	0.041	2.88	5.69	2.21	0.13
1957	3.11	0.80	0.008	<0.002	8.98	4.64	1.79	0.11
1956	4.56	2.36	0.027	0.033	2.03	9.18	1.12	0.07
1955	6.70	3.04	0.004	0.010	1.98	4.53	4.07	0.09
1954	1.75	0.55	0.006	0.011	0.45	2.70	2.73	0.04
1953	1.88	0.33	0.017	0.061	<0.21	6.97	2.97	<0.08
1952	1.87	0.50	0.006	0.013	1.12	5.80	8.16	0.14
1951	2.30	0.70	0.003	0.005	0.80	4.69	2.60	0.10
g.m.	1.33	0.47	0,007	0.017	0.75	4.50	1.42	0.10
			PRE-INDUSTRIAL					
	0.18	0.06	0.009	0.025	1.39	2.22	2.55	0.46
	0.12	0.38	0.008	0.037	1.60	1.82	0.68	0.30
								0.24
								0.16
g.m.	0.13	0.15	0.008	0.030	1.49	2.01	1.32	0.27

g.m. is geometric mean

TABLE 4. Comparison of ranges of V, Pb, Cd, and Hg concentrations (μg.g^{-1}) in contemporary glacier ice and snow between Spitsbergen and Antarctica. (After Jaworowski et al., 1981).

	V	Pb	Cd	Hg
78°N, Spitsbergen		7–48		
76°N, Spitsbergen	0.15–8.98	2–10.6	0.38–8.16	0.04–0.3
65–77°N, Greenland	0.008–0.022	0.001–0.42	0.001–1.38	0.002–0.88
68°N, Barrow,Alaska	0.04–0.26		0.02–0.22	0.005–0.026
66°N, Svartisen, Norway	0.1–2.29		0.13–1.2	0.12–0.5
63°N, Kahiltna Gl., Alaska	0.07	0.55–0.02	0.05–0.23	0.04–0.35
61°N, Storbreen, Norway		1.62–20	1.25–35.4	0.1–0.23
58°N, Norway	0.31*	1–14*	0.16–* 3.4	
46°N, Gurgler Gl., Alps	0.06–5.21	1.58–6.1	0.67–18.4	0.1–0.23
45°N, Mt. Blanc, Alps	0.05–0.6	0.04–15.8**	0.02–1.1**	
28°N, Himalaya	0.04–0.32	1.75–7.95	0.22–0.29	
0°, Ruwenzori, Africa	0.69–0.4.1	8.45–32.8	0.08–0.28	
13°S, Andes, Peru	0.06	6.9–29.8	0.45–1.8	0.25–1.96
62°S, King George Island, Antarctica	0.06–1.1	1.27–3.69	0.008–0.35	0.09–0.35
66–80°S, Antarctica		0.001–3.0	0.001–7.5	

* After Galloway et al., 1982; ** after Batifol and Boutron, 1983.

Table 5. Flow of natural radionuclides (kCi) and heavy metals (kt) into the global atmosphere. After Jaworowski et al., 1981 and Kownacka et al., 1989.

BASIS OF ESTIMATION	^{226}Ra	U	Pb	Cd	V	Hg
Contemporary glacier ice	6.6	12	4870	590	180	190
Pre- industrial glacier ice	8.6	23	6650	720	290	238
Content in 0 - 15 km air column	6.2	8.3	3700			
Natural non-enriched particulates	0.62	0.95	54	1.5	88	0.07
Anthropogenic emissions	0.23	1,43	358	0.66	111	11

TABLE 6. Temporal changes of ^{210}Pb concentration in lichens from Hornsundfjorden, Spitsbergen. (After Jaworowski, 1967).

Year	^{210}Pb (pCi.g^{-1} d.w.)	Species
1957	1.4	Cladonia mitis
1957	2.3	Cladonia rangiferina
1958	8.0	Cladonia mitis
1965	6.3	Cladonia mitis
1965	10.4	Cladonia mitis
1965	13.4	Cladonia mitis

TABLE 7. Mean concentration of heavy metals in liver and kidneys of sea birds and reindeer from western Spitsbergen in comparison with reindeer from continental Norway, polar bear from Canada and "reference man" – in $\mu g\ g^{-1}$ wet weight. After Norheim (1987), Norheim and Nilssen (1989), Frøslie (1989), ICRP (1975) and Norstrom et al. (1986).

	LIVER						KIDNEY			
	Se	Hg	Cd	Zn	Cu	Pb	Cd	Zn	Cu	Pb
Glaucous gull	2.2	1.6	3.6	32	7.3	<DL	23	46	5.2	<DL
Fulmar	3.0	2.1	17	73	6.2	<DL	55	50	4.1	<DL
Brünnich's guillemot	1.9	0.6	3.9	35	8.2	<DL	16	39	6.9	<DL
Little auk	2.6	0.5	4.3	37	8.4	<DL	21	40	6.4	<DL
Eider	8.9	1.0	4.3	50	270	<DL	14	33	13	<DL
Reindeer Svalbard			0.7	44			4.3	2.5		
Reindeer N.Norway			0.3	30	65	<0.5				
Polar bear* Canada	8.8	25	<0.4	63	37					
"Standard man"	0.7	0.3	2.2	47	6.6	1.7	31	48	2.9	1

<DL = below detection limit

* calculated from wet weight concentrations, assuming aver. content of water of 64.2%

TABLE 8. Mean concentrations of organic compounds in fat of sea birds, seals, polar bear, fish, ascidians and shrimps from Svalbard and Bjørøya* (µg g^{-1} fat wet weight).

	HCB	DDE	DDT	PCB
Glaucous gull, 1972*[a]				535
1980[b]	1.9	36		82
1980[c]	1.8		15–30	82–74
1971[c]			34	130
Fulmar, 1973*[a]				13
1980b	1.7	22		59
Brünnich's guillemot, 1972*[a]				3.1
1989[b]			6	12
Little auk, 1980[b]	0.4	2.1		7.1
Eider, 1980[b]	0.002	0.011		3.1
1980[c]	0.1		0.6	3
Ringed seal, 1980[c]	0.014		0.8	2
Ringed seal, 1971 (Baltic)[c]			160.0	120
Bearded and ringed seal, 1984[d]		0.7	0.2	
Arctic fox, 1973–74[e]				10
Polar bear, 1971[c]				17
old, 1980[c]	0.06		0.4	4.2
young, 1980[c]				280
Polar cod, 1984[d]		0.02		
Ascidians, 1984[d]		0.01	0.001	
Shrimps, 1984[d]		0.004	<DL	

[a] Bourne and Bogan, 1972; [b] Norheim and Kjos–Hanssen, 1984; [c] Edelstam, 1987; [d] Carlberg and Bøler, 1985; [e] Norheim, 1978

<DL = below detection limit

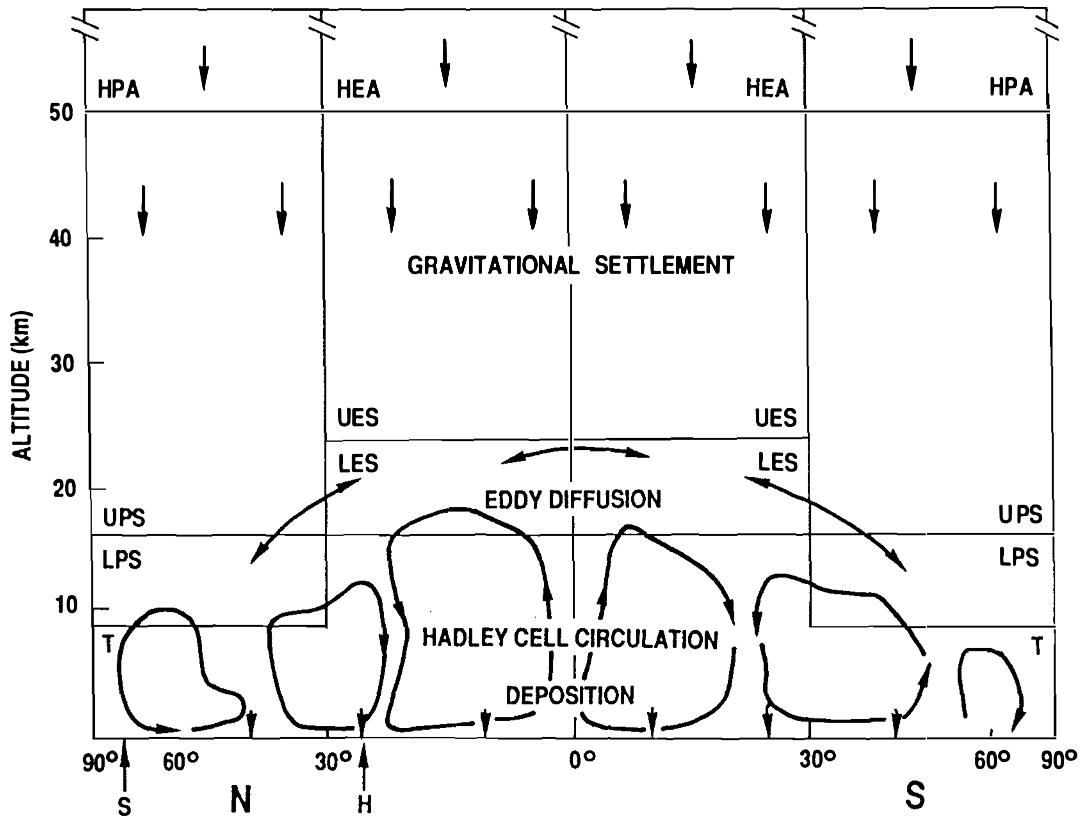

FIGURE 1. Atmospheric regions and predominant atmospheric transport processes. S = Spitsbergen; H = Hawaii; T = troposphere; LPS = lower polar stratosphere; UPS = upper polar stratosphere; HPS = high polar stratosphere; LES = lower equatorial stratosphere; UES = upper equatorial stratosphere; HES = high equatorial stratosphere. Impurities may ascend into the upper troposphere and lower stratosphere at the Equator and in mid-latitudes and descend to low altitudes as indicated by arrows. After UNSCEAR, 1982.

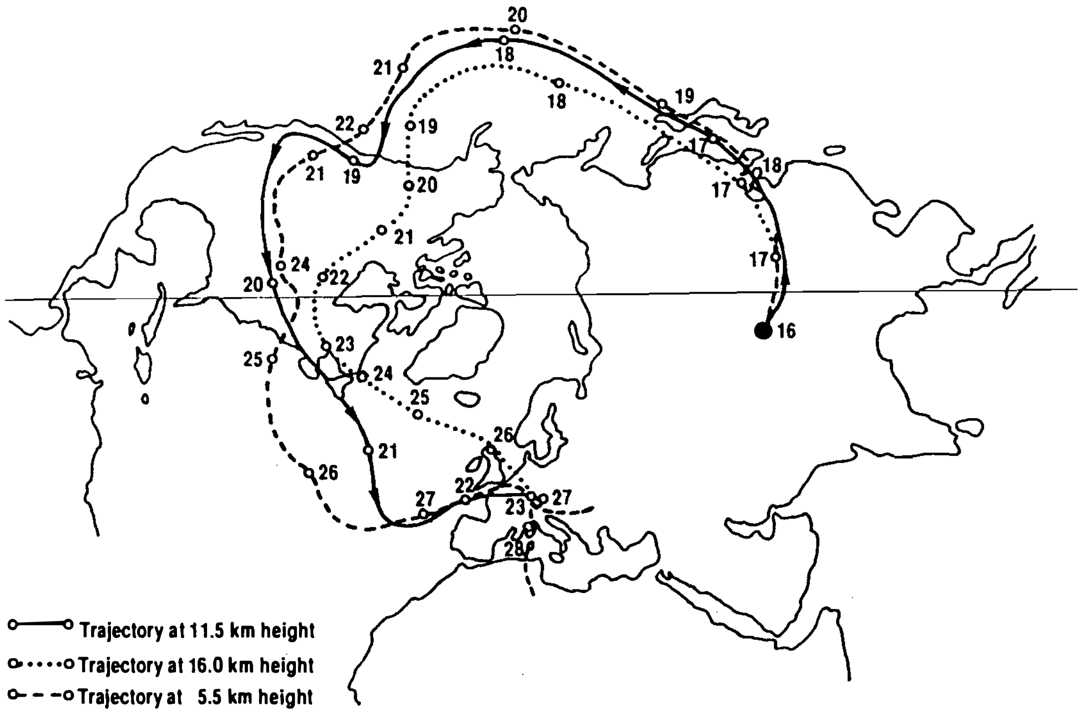

FIGURE 2. Trajectories derived from meteorological data and confirmed by activity measurements in ground-level air of tropospheric fallout from the last (Chinese) atmospheric nuclear explosion of 12 October 1980. After UNSCEAR, 1982.

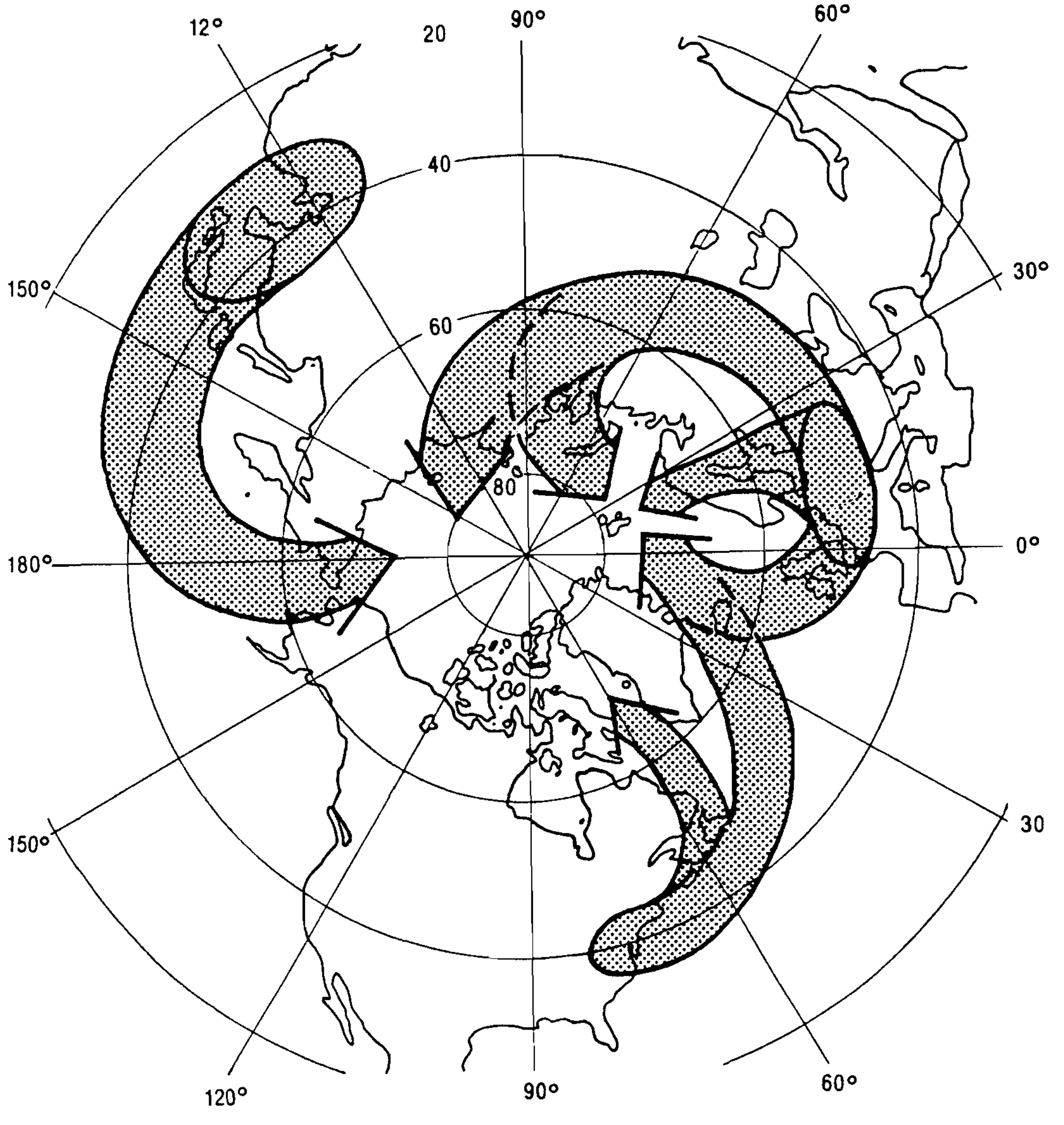

Figure 3. Low altitude pathways for transport of pollution aerosols between mid-latitudes and the Arctic. After Rahn and Shaw, 1982.

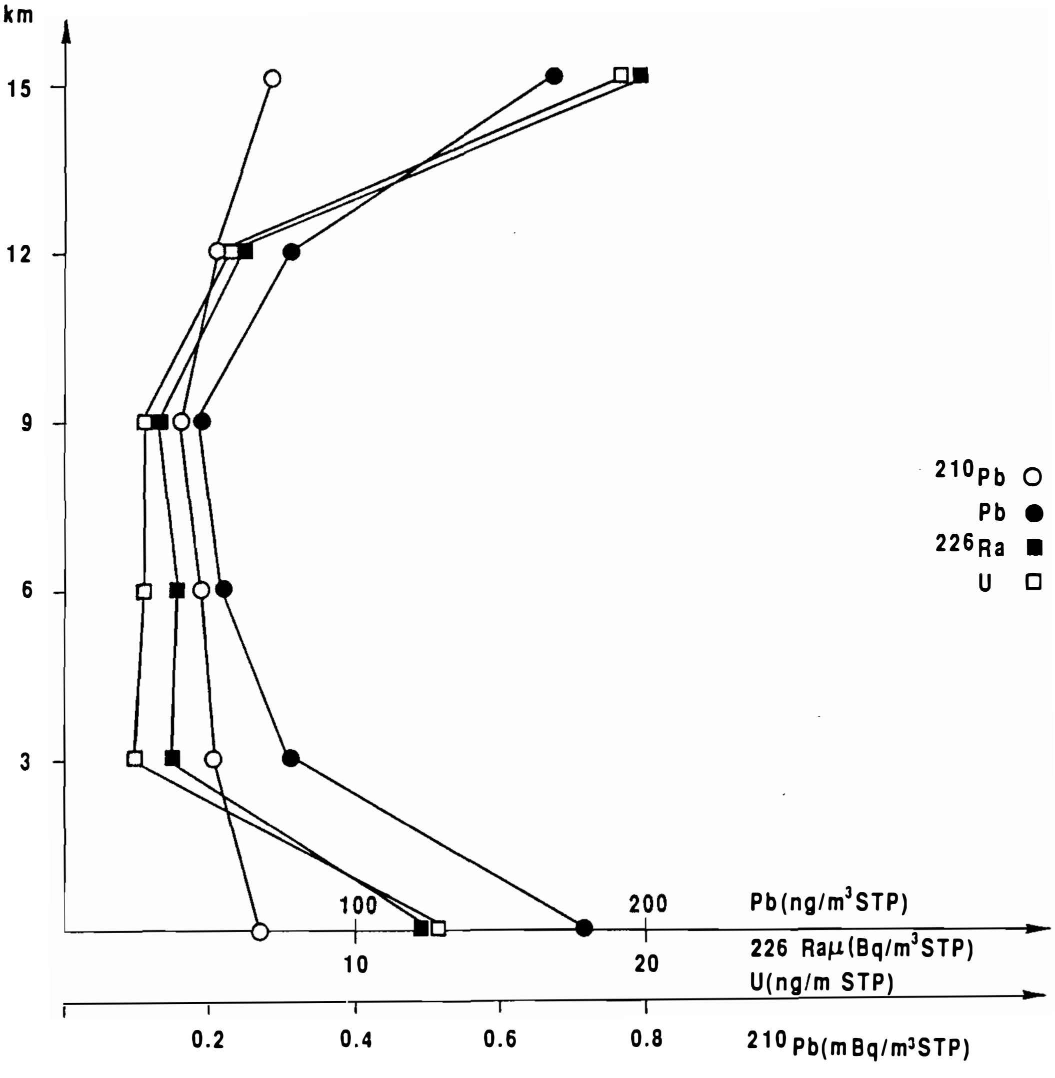

Figure 4. Vertical distribution in the atmosphere over Poland of average concentrations of ^{210}Pb, Pb, ^{226}Ra and U between 1980 and 1981. After Kownacka et al., 1989.

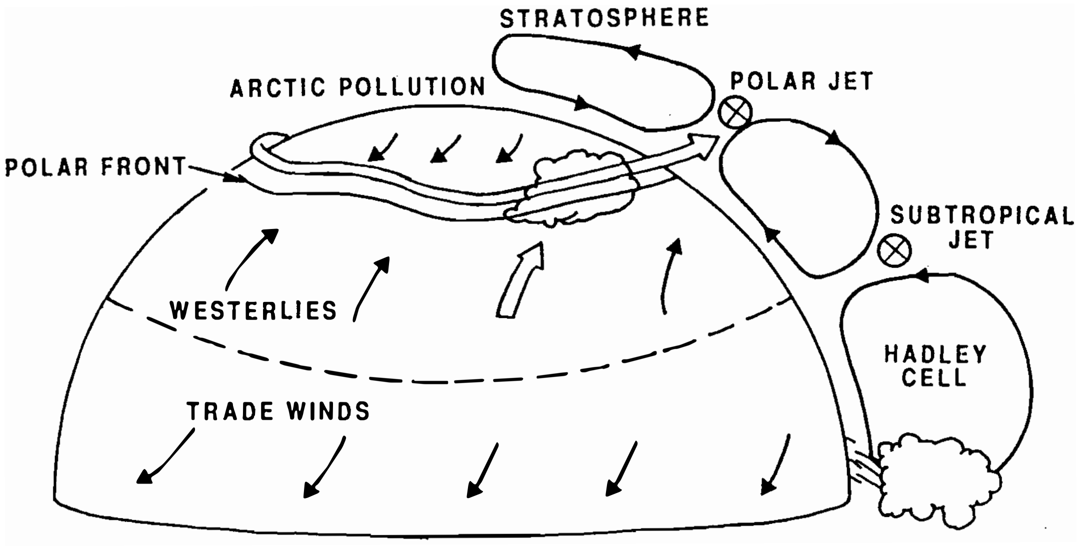

FIGURE 5. Pathways for global-scale pollution of the atmosphere, transporting impurities to the Arctic. After Slim, 1982.

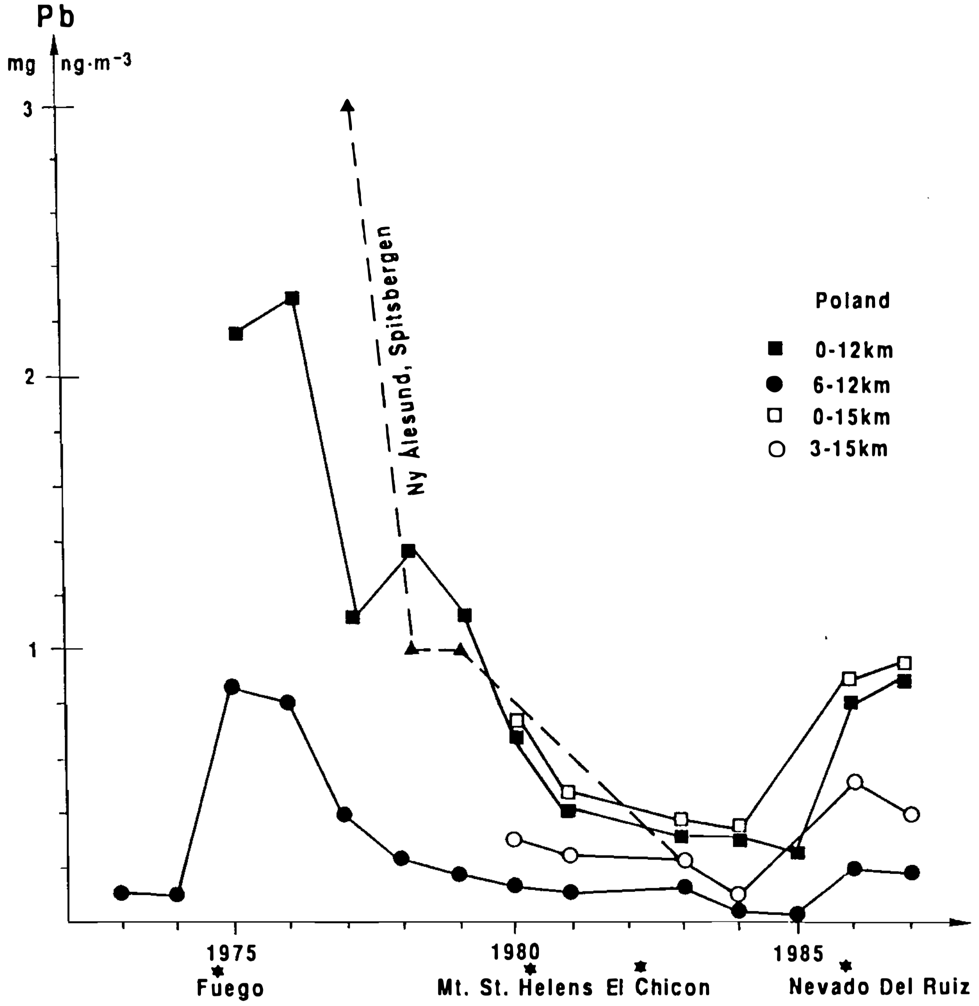

FIGURE 6. Temporal changes of content of Pb in the 1 m^2 column of air between the ground level and 15 km over Poland (Kownacka et al., 1989) and in the ground-level air at Ny–Ålesund, Spitsbergen (after Pacyna and Ottar, 1985 and Maenhaut et al., 1989).

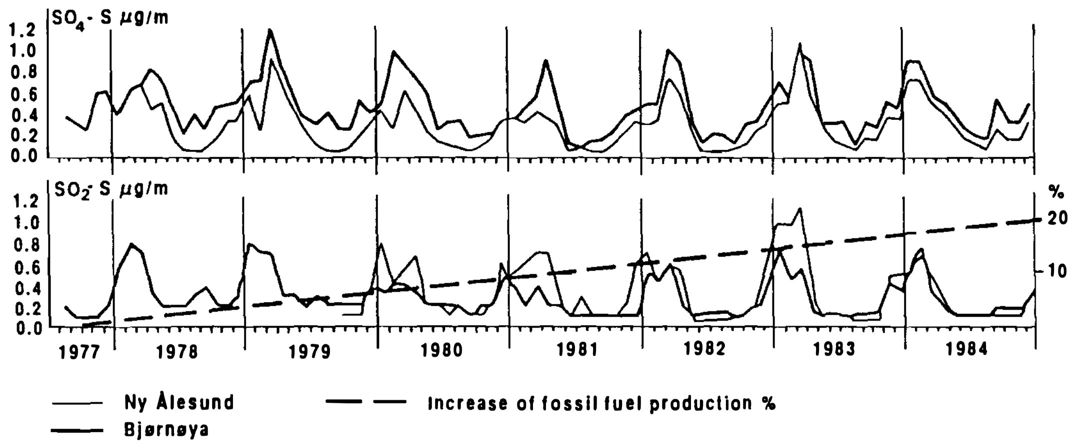

FIGURE 7. Variation of the monthly mean concentrations of SO_2 and SO_4 at the arctic stations Bjørnøya and Ny Ålesund during 1977 and 1984 (after Ottar et al., 1986 A) and increase of global fossil fuel production (after GUS, 1987).

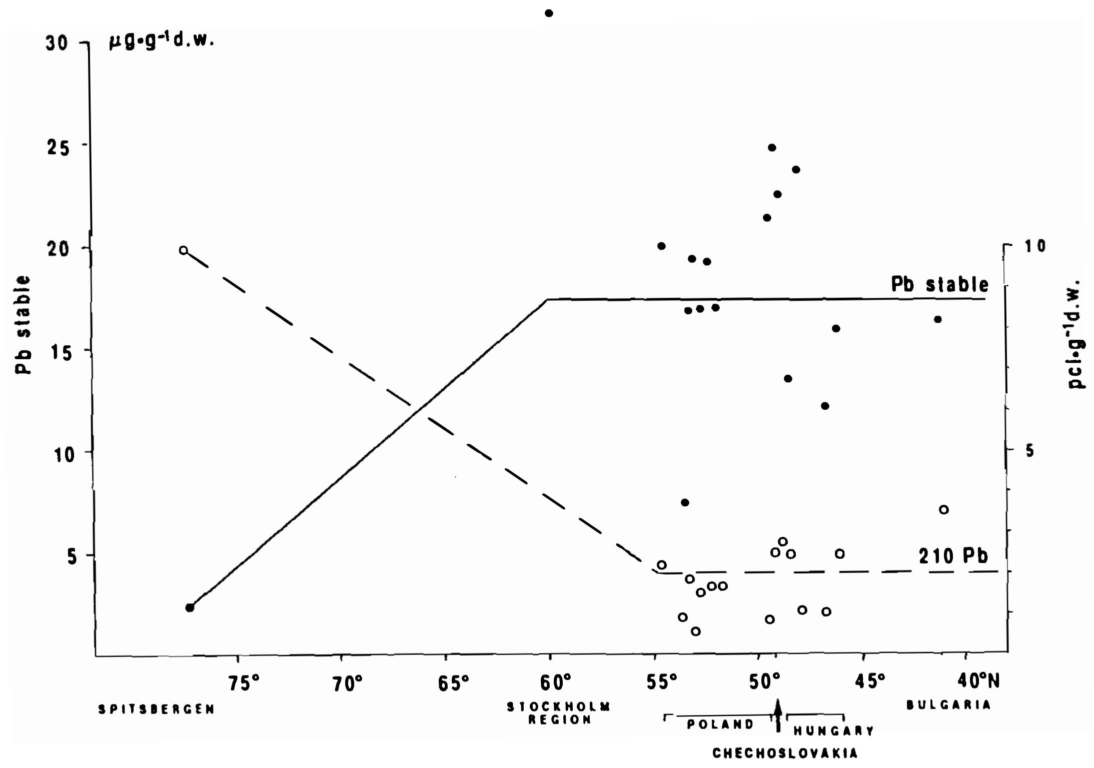

FIGURE 8. Geographical distribution of Pb and ^{210}Pb concentrations in Cladonia sp. lichens between Spitsbergen and southern Bulgaria. After Jaworowski, 1967).

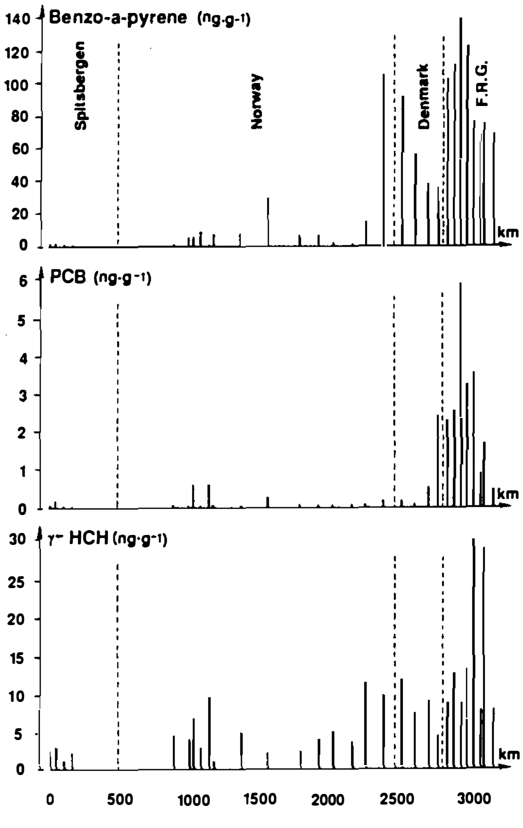

FIGURE 9. Geographical distribution of Cd, Pb, Cu and Ni in mosses between Spitsbergen and West Germany. After Ottar et al., 1986 A.

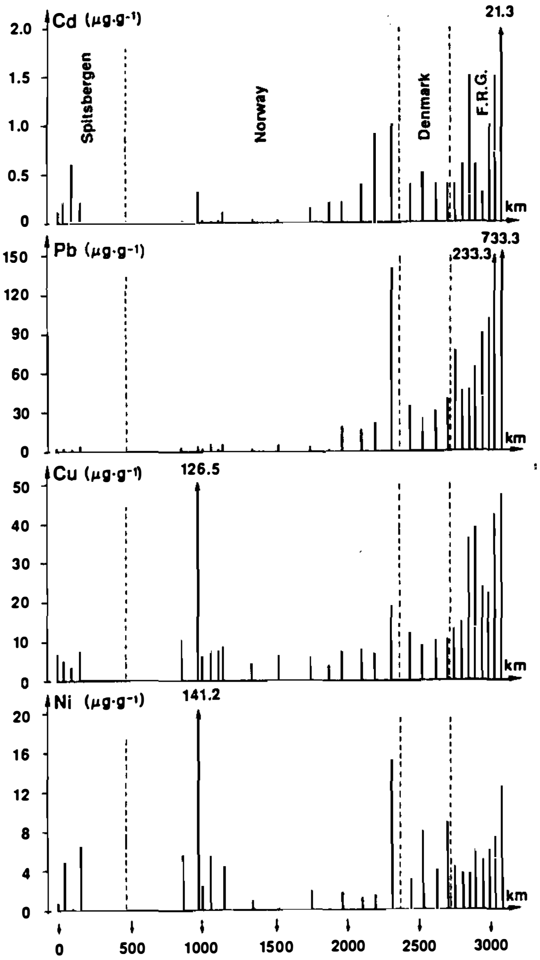

FIGURE 10. Geographical distribution of organic pollutants in mosses between Spitsbergen and West Germany. After Ottar et al., 19886 A.

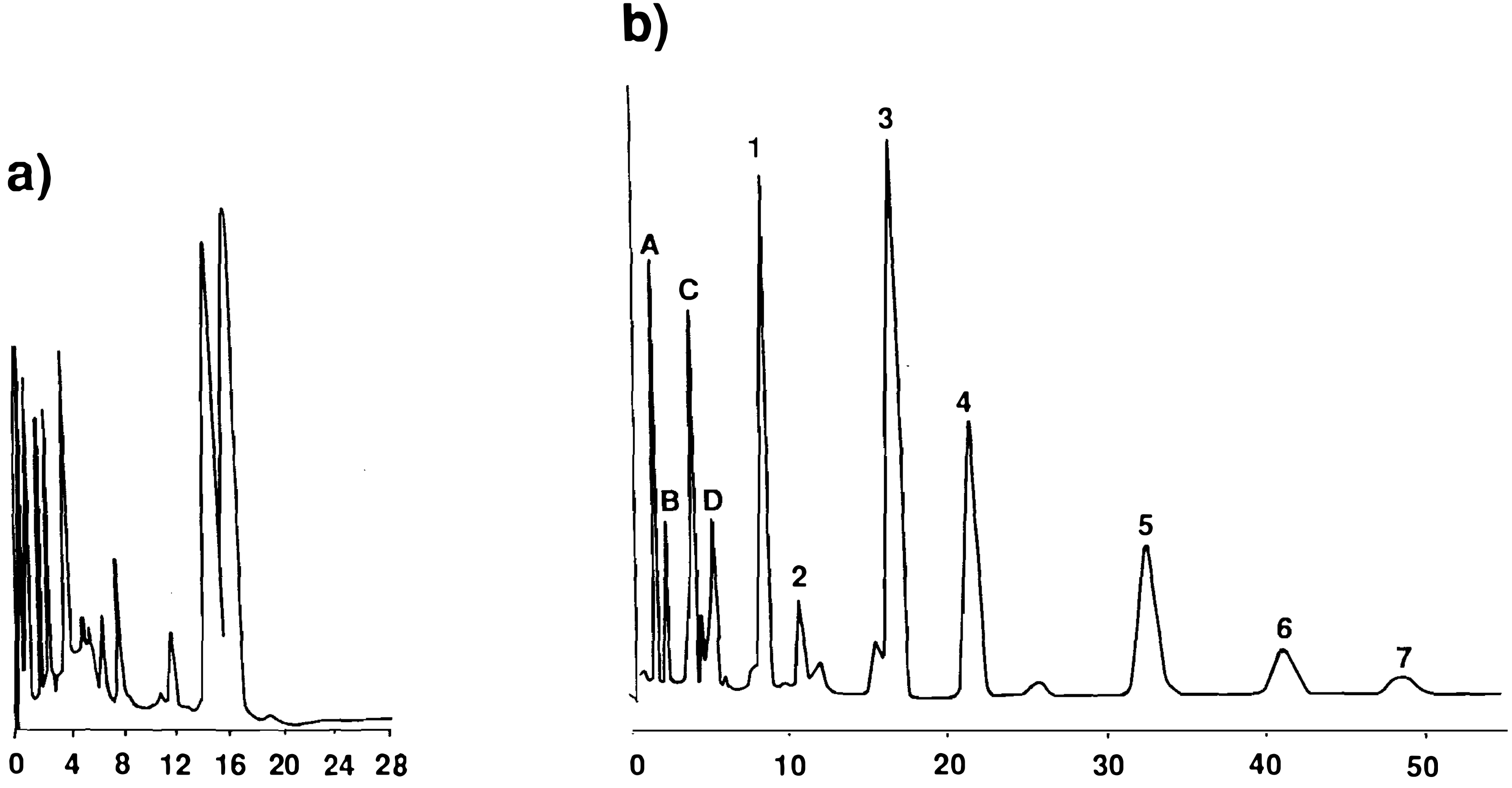

FIGURE 11. Comparison of typical gas chromatograms of:
(a) extract of marine alga Laurentia subopposita containing natural PCBs and (b) extract of liver of an arctic fox Alopex lagopus from Svalbard containing PCBs reported as synthetic.
After Norheim, 1978 and Fennical, 1981.

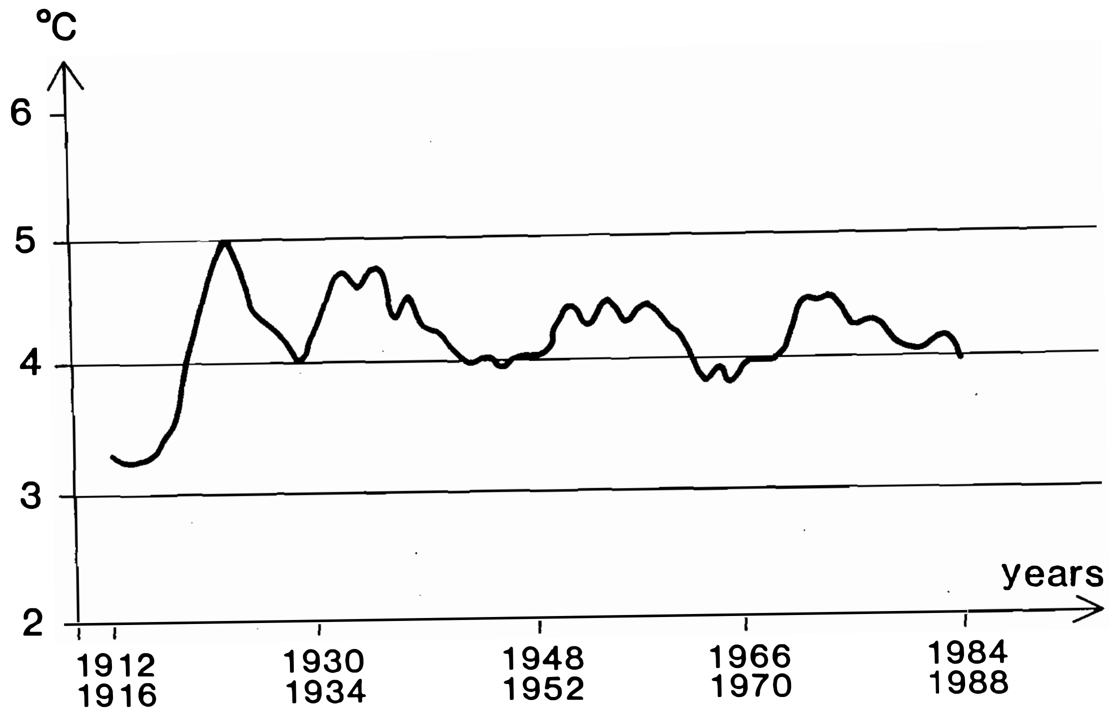

FIGURE 12. Reconstructed running 5–years mean summer temperature for July, August and September in Longyearbyen, Svalbard, 1912 – 1988. After Hagen and Liestøl, 1989.