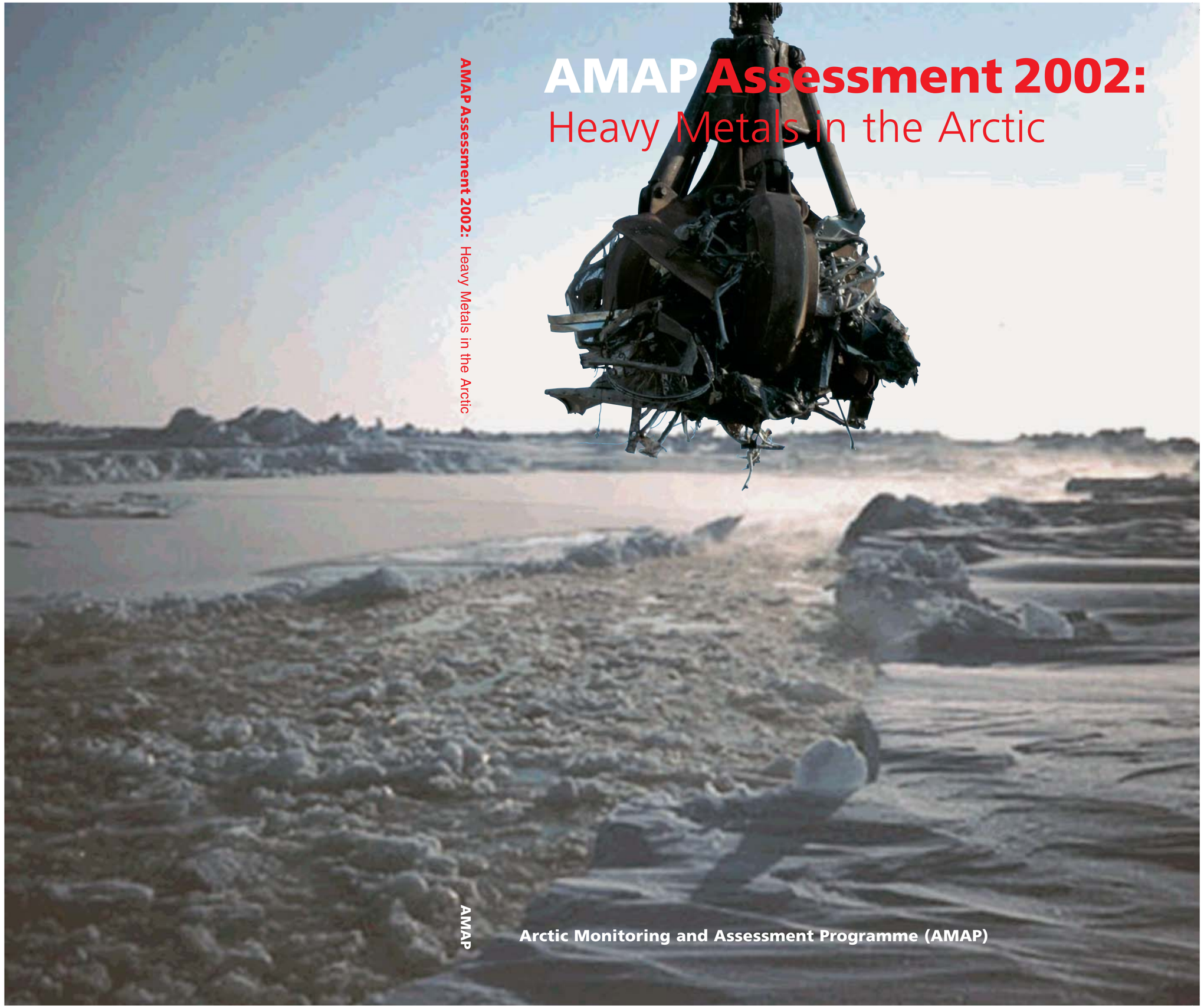

AMAP Assessment 2002:
Heavy Metals in the Arctic

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Arctic Monitoring and Assessment Programme (AMAP)

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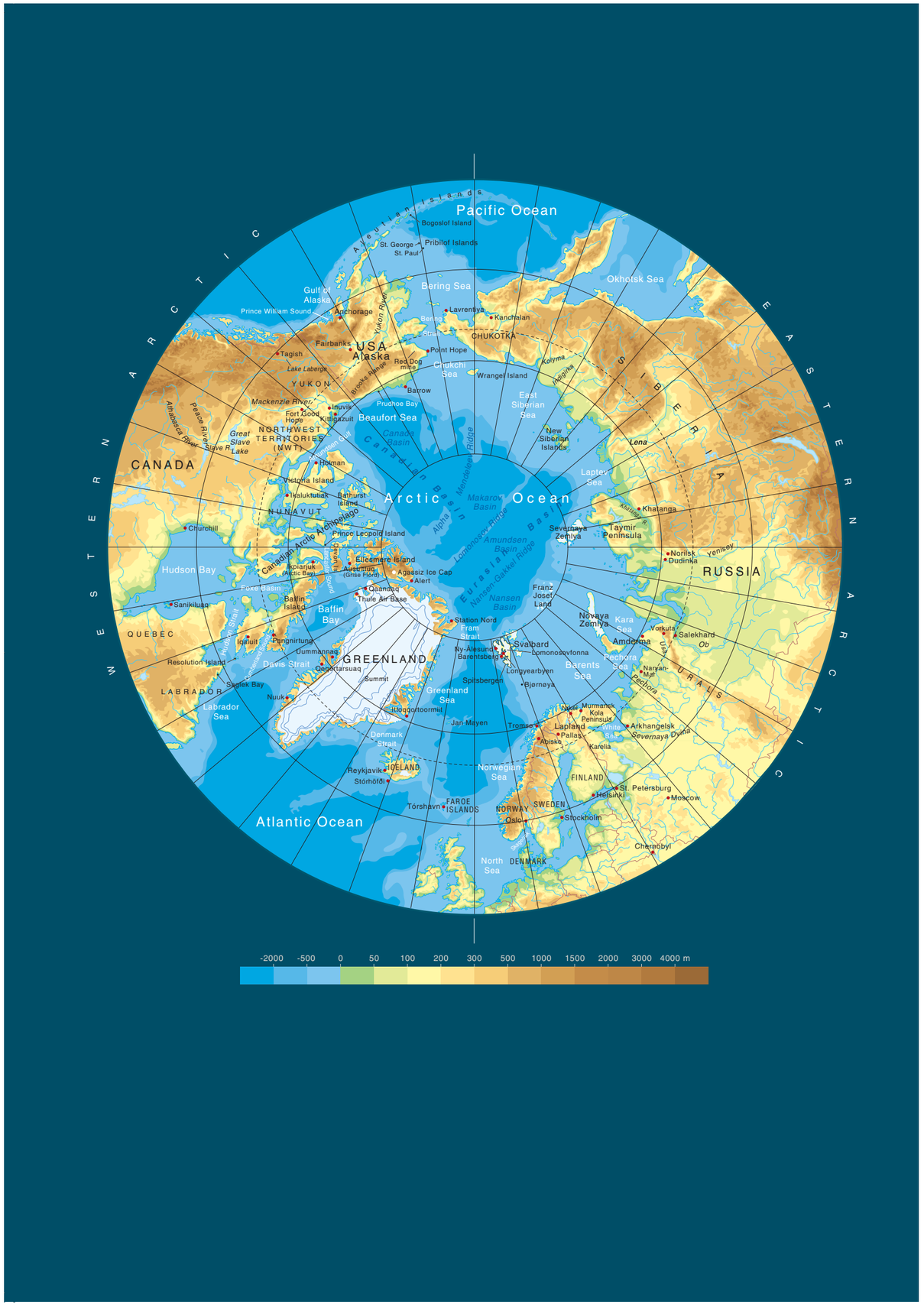
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Nansen Basin
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Production

Scientific, technical and linguistic editing
Carolyn Symon, Simon J. Wilson

Technical production management
Olsen & Olsen, Helstedsvej 10, DK-3480 Fredensborg, Denmark

Lay-out, design and production of computer graphics
Kai Olsen, Olsen & Olsen
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Contents

Preface

This assessment report details the results of the 2002 AMAP assessment of Heavy Metals in the Arctic. It builds upon the previous AMAP heavy metals assessment presented in 'AMAP Assessment Report: Arctic Pollution Issues'[1] that was published in 1998.

The Arctic Monitoring and Assessment Programme (AMAP) is a group working under the Arctic Council.

The Arctic Council Ministers have requested AMAP to:

- *produce integrated assessment reports on the status and trends of the conditions of the Arctic ecosystems;*
- *identify possible causes for the changing conditions;*
- *detect emerging problems, their possible causes, and the potential risk to Arctic ecosystems including indigenous peoples and other Arctic residents; and to*
- *recommend actions required to reduce risks to Arctic ecosystems.*

This report is one of five detailed assessment reports that provide the accessible scientific basis and validation for the statements and recommendations made in the second AMAP State of the Arctic Environment report, 'Arctic Pollution 2002'[2] that was delivered to Arctic Council Ministers at their meeting in Inari, Finland in October 2002. It includes extensive background data and references to the scientific literature, and details the sources for figures reproduced in the 'Arctic Pollution 2002' report. Whereas the 'Arctic Pollution 2002' report contains recommendations that specifically focus on actions aimed at improving the Arctic environment, the conclusions and recommendations presented in this report also cover issues of a more scientific nature, such as proposals for filling gaps in knowledge, and recommendations relevant to future monitoring and research work.

To allow readers of this report to see how AMAP interprets and develops its scientifically-based assessment product in terms of more action-orientated conclusions and recommendations, the 'Executive Summary of the Arctic Pollution 2002 Ministerial Report', which also covers other priority issues (Persistent Organic Pollutants, Radioactivity, Human Health, and Climate Change Effects on Contaminant Pathways), is reproduced in this report on pages xi to xv.

The AMAP assessment is not a formal environmental risk assessment. Rather, it constitutes a compilation of current knowledge about the Arctic region, an evaluation of this information in relation to agreed criteria of environmental quality, and a statement of the prevailing conditions in the area. The assessment presented in this report was prepared in a systematic and uniform manner to provide a comparable knowledge base that builds on earlier work and can be extended through continuing work in the future.

The AMAP scientific assessments are prepared under the direction of the AMAP Assessment Steering Group. The product is the responsibility of the scientific experts involved in the preparation of the assessment. The lead country for the AMAP Heavy Metals Assessment under AMAP phase II was the United States. The assessment is based on work conducted by a large number of scientists and experts from the Arctic countries (Canada, Denmark/Greenland/Faroe Islands, Finland, Iceland, Norway, Russia, Sweden, and the United States), together with contributions from indigenous peoples organizations, from other organizations, and from experts in other countries.

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

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

The AMAP Working Group that was established to oversee this work, and the AMAP heavy metals assessment group are pleased to present its assessment.

John Calder
AMAP Working Group Chair

Suzanne Marcy
AMAP Heavy Metals assessment lead (United States)

Lars-Otto Reiersen
AMAP Executive Secretary

Oslo, December 2005

1 AMAP, 1998. AMAP Assessment Report: Arctic Pollution Issues. Arctic Monitoring and Assessment Programme (AMAP), Oslo, Norway. xii+859 pp.

2 AMAP, 2002. Arctic Pollution 2002: Persistent Organic Pollutants, Heavy Metals, Radioactivity, Human Health, Changing Pathways. Arctic Monitoring and Assessment Programme (AMAP), Oslo, Norway. xii+112 pp.

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The AMAP Working Group would like to thank the following persons for their work in preparing the AMAP 2002 Heavy Metals Assessment.

Assessment Lead/Coordinating Editors:
Suzanne Marcy, U.S. Environmental Protection Agency, Office of Research and Development, Washington, DC, USA
Jesse Ford, Department of Fisheries and Wildlife, Oregon State University, Corvallis, OR, USA

Scientific Secretary:
Lorraine Hetschel, Department of Fisheries and Wildlife, Oregon State University, Corvallis, OR, USA

Contributing Authors:

Chapter 1:
Suzanne Marcy

Chapter 2:
Jozef Pacyna, Norwegian Institute for Air Research, Kjeller, Norway

Chapter 3:
Steve Brooks, NOAA Atmospheric Turbulence and Diffusion Division, Oak Ridge, TN, USA *(section 3.1)*
Jesper Christensen, National Environmental Research Institute, Roskilde, Denmark *(section 3.2)*
Viacheslav Gordeev, P.P.Shirshov Institute of Oceanology, Russian Academy of Sciences, Moscow, Russia *(sections 3.4, 3.5)*
Alexey Gusev, EMEP Meteorological Synthesizing Centre - East, Moscow, Russia *(section 3.2)*
Steve Lindberg, Environmental Sciences Division, Oak Ridge National Laboratory, Oak Ridge, TN, USA *(section 3.1)*
Robie Macdonald, Department of Fisheries and Oceans, Institute of Ocean Sciences, Sidney, British Columbia, Canada *(sections 3.3, 3.7, 3.8)*
Suzanne Marcy *(editor)*
Keith Puckett, Meterological Services of Canada, Environment Canada, Downsview, ON, Canada *(section 3.1)*
Oleg Travnikov, EMEP Meteorological Synthesizing Centre - East, Moscow, Russia *(section 3.2)*
Simon Wilson, Arctic Monitoring and Assessment Programme, Oslo, Norway *(section 3.2)*

Chapter 4:
Hans Borg, Institute of Applied Environmental Research (ITM), Stockholm University, Stockholm, Sweden
Maria Dam, Food and Environmental Agency, Tórshavn, Faroe Islands
Jesse Ford
Frank Riget, Department of Arctic Environment, National Environmental Research Institute, Roskilde, Denmark

Chapter 5:
Anders Bignert, Swedish Natural History Museum, Stockholm, Sweden
Birgit Braune, Canadian Wildlife Service, Environment Canada, Ottawa, ON, Canada
Peter Outridge, Geological Survey of Canada, Ottawa, ON, Canada
Frank Riget
Simon Wilson

Chapter 6:
John Derome, Rovaniemi Research Station, Rovaniemi, Finland
Ann Fairbrother, US Environmental Protection Agency, Corvallis, OR, USA
Suzanne Marcy
Jeffrey Wirtz, Parametrix, Inc., Corvallis, OR, USA
Katie Harding, Department of Fisheries and Wildlife, Oregon State University, Corvallis, OR, USA

Chapter 7:
Anders Bignert
Hans Borg
Birgit Braune
Steve Brooks
Jesper Christensen
Maria Dam
John Derome
Rune Dietz, Department of Arctic Environment, National Environmental Research Institute, Roskilde, Denmark
Ann Fairbrother

Jesse Ford
Viacheslav Gordeev
Alexey Gusev
Steve Lindberg
Robie Macdonald
Suzanne Marcy
Peter Outridge
Jozef Pacyna
Keith Puckett
Frank Riget
Oleg Travnikov
Simon Wilson

Contributors:
Jens-Alfons Hölemann, Alfred Wegener Institute for Polar and Marine Research, Bremerhaven, Germany
Robert Anthony, Department of Fisheries and Wildlife, Oregon State University, Corvallis, OR, USA
Kevin Barrett, Norwegian Institute for Air Research, Kjeller, Norway
Leonard Barrie, World Meteorological Organization, Geneva, Switzerland
Guy Beauchamp, University of Montreal, Quebec, Canada
Paul Becker, National Institute of Standards and Technology, Charleston, SC, USA
Kimberlee Beckmen, Alaska Department of Fish and Game, Fairbanks, AK, USA
Torunn Berg, Norwegian Institute for Air Research, Kjeller, Norway
Kathleen Crane, Department of Geography, Hunter College, NY, USA
Doug Dasher, Institute of Marine Sciences, University of Alaska Fairbanks, Fairbanks, AK, USA
Ashu Dastoor, Meterological Services of Canada, Environment Canada, Dorval, Quebec, Canada
Vladimir Dauvalter, Institute of North Industrial Ecology Problems Kola Science Centre, Russian Academy of Sciences, Apatity, Russia
Larissa Dehn, Institute of Arctic Biology, University of Alaska Fairbanks, Fairbanks, AK, USA
George Divoky, Institute of Arctic Biology, University of Alaska Fairbanks, Fairbanks, AK, USA
Larry Duffy, Department of Chemistry and Biochemistry, University of Alaska Fairbanks, Fairbanks, AK, USA
Brett Elkin, Department of Resources Wildlife and Economic Development, Government of the Northwest Territories, Yellowknife, NT, Canada
Marlene Evans, National Water Research Institute, Environment Canada, Saskatoon, Canada
Bruce Finney, Institute of Marine Science, School of Fisheries and Ocean Sciences, University of Alaska Fairbanks, Fairbanks, AK, USA
Bruce Forbes, Arctic Center, University of Lapland, Rovaniemi, Finland
Nicolas Givelet, Institute of Geological Sciences, University of Berne, Switzerland
Michael Goodsite, Department of Chemistry, University of Southern Denmark, Odense, Denmark
Norman Green, Norwegian Institute of Water Research, Kjelsås, Norway
Katie Harding, Department of Fisheries and Wildlife, Oregon State University, Corvallis, OR, USA
Heikki Henttonen, Finnish Forest Research Institute, Vantaa Research Centre, Vantaa Unit, Finland
Lorraine Hetschel, Department of Fisheries and Wildlife, Oregon State University, Corvallis, OR, USA
Ketil Hylland, Norwegian Institute of Water Research, Kjelsås, Norway
Juha-Pekka Hirvi, Finnish Environment Institute, Finland
Katrin Hoydal, Food and Environmental Agency, Tórshavn, Faroe Islands
Poul Johansen, National Environmental Research Institute, Roskilde, Denmark
Kjell Johansson, Department of Environmental Assessment, Swedish University of Agricultural Science, Uppsala, Sweden
Vitaly Kimstach, Arctic Monitoring and Assessment Programme, Oslo, Norway
Roy Koerner, Geological Survey of Canada, Ottawa, ON, Canada
Alexei Konoplev, SPA Typhoon, Obninsk, Russia
Michael Kwan, Nunavik Research Centre, Nunavik, Quebec, Canada
Pertti Lahermo, Geological Survey of Finland, Espoo, Finland
Dixon Landers, Office of Research and Development, US Environmental Protection Agency, Corvallis, OR, USA
G.G. Lauenstein, U.S. National Oceanic and Atmospheric Administration, Silver Spring, MD, USA
Jerry Lin, Department of Civil Engineering, Lamar University, Beaumont, TX, USA
Göran Lithner, Institute of Applied Environmental Science (ITM), Stockholm University, Stockholm, Sweden
Lyle Lockart, North/South Consultants Inc., Winnipeg, Manitoba, Canada
Nicholas T. Loux, Office of Research and Development, U.S. Environmental Protection Agency, Athens, GA, USA
Colin Macdonald, Northern Environmental Consulting and Analysis, Pinawa, Manitoba, Canada
Jaakko Mannio, Finnish Environment Institute, Helsinki, Finland

Angela Matz, U.S. Fish and Wildlife, Fairbanks, AK, USA
James Meador, NOAA Fisheries, Seattle, WA, USA
Derek Muir, National Water Research Institute, Environment Canada, Burlington, Canada
John Munthe, IVL Swedish Environmental Research Institute, Gothenburg, Sweden
Sathy Naidu, Institute of Marine Sciences, University of Alaska Fairbanks, Fairbanks, AK, USA
Alexy Naumenko, Institute of Northern Ecology Problems, Kola Science Centre, Russian Academy of Science, Russia
Tord Nygård, Norwegian University of Science and Technology (NTNU), Trondheim, Norway
Torgeir Nygård, Norwegian Institute for Nature Research, Trondheim, Norway
Jóhanna Olsen, Food and Environmental Agency, Tórshavn, Faroe Islands
Todd O'Hara, Institute of Arctic Biology, University of Alaska Fairbanks, Fairbanks, AK, USA
Elisabeth Pacyna, Norwegian Institute for Air Research, Kjeller, Norway
Pat Roach, Indian and Northern Affairs, Whitehorse, Canada
Andrew Robertson, U.S. National Oceanic and Atmospheric Administration, Silver Spring, MD, USA
William Schroeder, Meteorological Service of Canada, Environment Canada, Toronto, ON, Canada
Karen Scott, Department of Microbiology, University of Manitoba, Winnipeg, Manitoba, Canada
William Shotyk, Geological Institute, University of Berne, Switzerland
Trond Skotvold, Akvaplan-niva, Polar Environmental Research Centre, Tromsø, Norway
Frits Steenhuisen, Arctic Centre, University of Groningen, Groningen, Netherlands
Alexandra Steffen, Meteorological Service of Canada, Environment Canada, Toronto, ON, Canada
Eiliv Steinnes, University of Trondheim, Trondheim, Norway
Gary Stern, Freshwater Institute, Department of Fisheries and Oceans, Winnipeg, Canada
Yuri Tsaturov, Roshydromet, Moscow, Russia
Vladimir Tsibulski, Institute for Atmospheric Air Protection, Russia
Matti Verta, Finnish Environment Institute, Helsinki, Finland
Bronwen Wang, U.S. Geological Survey, Anchorage, AK, USA
Stig Westerlund, Rogaland Research, Stavanger, Norway
Victoria Woshner, North Slope Borough, Department of Wildlife Management, Barrow, AK, USA
Eva Yngvadóttir, Icelandic Fisheries Laboratories, Reykjavík, Iceland
Christian Zdanowicz, Geological Survey of Canada, Ottawa, Ontario, Canada
Irina Zenkova, Institute of the North Industrial Ecological Problems, Kola Science Centre, Russian Academy of Science, Russia

External Peer Review Panel
W. Nelson Beyer, USGS Patuxent Wildlife Research Center, Laurel, MD, USA *(chapter 6, 2002 draft)*
Bo Elberling, Institute of Geography, University of Copenhagen, Copenhagen, Denmark *(early draft)*
Kelly Falkner, College of Oceanic and Atmospheric Sciences, Oregon State University, Corvallis, OR, USA *(2002 draft)*
Robert Garrett, Geological Survey of Canada, Ottawa, Ontario, Canada *(2002 draft)*
Michael Goodsite, Department of Chemistry, University of Southern Denmark, Odense, Denmark *(early draft)*
Mark Hermanson, Department of Earth and Environmental Science, University of Pennsylvania, Philadelphia, PA, USA *(all chapters)*
David Lean, NSERC Industrial Chair in Ecotoxicology, Department of Biology, University of Ottawa, Canada *(all chapters)*
Nicholas T. Loux, Office of Research and Development, U.S. Environmental Protection Agency, Athens, GA, USA *(all chapters)*
Jerome Nriagu, Center for Human Growth and Development, University of Michigan, Ann Arbor, MI, USA *(all chapters)*
Bruce Rodan, Office of Research and Development, U.S. Environmental Protection Agency, Washington DC, USA *(all chapters)*
William Schroeder, Meteorological Service of Canada, Environment Canada, Toronto, ON, Canada *(chapters 1, 2, 3)*
Alexandra Steffen, Meteorological Service of Canada, Environment Canada, Toronto, ON, Canada *(chapter 3)*
Eiliv Steinnes, University of Trondheim, Trondheim, Norway *(early draft)*
Michael Watson, U.S. Environmental Protection Agency, Seattle, WA, USA *(all chapters)*

Provision of data from AMAP Thematic Data Centres:
Norwegian Institute for Air Research: Torunn Berg
International Council for the Exploration of the Sea: Jørgen Nørrevang Jensen
University of Alaska Fairbanks: Kara Nance, Brian Hay

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Executive Summary to the *Arctic Pollution 2002* Ministerial Report

The Arctic Monitoring and Assessment Programme (AMAP) was established in 1991 to monitor identified pollution risks and their impacts on Arctic ecosystems. In 1997 the first AMAP report, *Arctic Pollution Issues: A State of the Arctic Environment Report** was published.

The assessment showed that the Arctic is closely connected to the rest of the world, receiving contaminants from sources far outside the Arctic region. The report was welcomed by the Arctic Council Ministers, who agreed to increase their efforts to limit and reduce emissions of contaminants into the environment and to promote international cooperation in order to address the serious pollution risks reported by AMAP.

The AMAP information greatly assisted the negotation of the protocols on persistent organic pollutants (POPs) and heavy metals to the United Nations Economic Commission for Europe's Convention on Long-range Transboundary Air Pollution (LRTAP Convention). They also played an important role in establishing the need for a global agreement on POPs, which was concluded in 2001 as the Stockholm Convention. Persistence, long-range transport, and bioaccumulation are screening criteria under both the POPs protocol and the Stockholm Convention, to be applied to proposals to add substances to the agreements. Information from AMAP will be useful in this context in showing whether persistent substances are accumulating in the Arctic and are therefore candidates for control, and also in assessing the effectiveness of the agreements.

The Arctic Council also decided to take cooperative actions to reduce pollution of the Arctic. As a direct follow up of the AMAP reports, the Arctic Council Action Plan to Eliminate Pollution of the Arctic (ACAP) was created to address sources identified through AMAP. ACAP was approved in 2000 and several projects have begun. The AMAP information was also used in establishing priorities for the Arctic Regional Programme of Action to Prevent Pollution from Landbased Sources (RPA), developed by the working group on Protection of the Arctic Marine Environment (PAME), and adopted by the Arctic Council in 1998.

After the first assessment, AMAP was asked to continue its activities and provide an updated assessment on persistent organic pollutants (POPs), heavy metals, radioactivity, human health, and pathways in 2002. Five scientific reports and a plain-language report have been prepared. This Executive Summary provides the main conclusions and recommendations of the 2002 AMAP assessments.

International Agreements and Actions

As described above, the LRTAP Convention protocols and the Stockholm Convention are essential instruments for reducing contamination in the Arctic. However, they cannot have any effect until they are ratified and implemented.

It is therefore recommended that:

- **The UN ECE LRTAP Protocols on Heavy Metals and POPs be ratified and implemented.**
- **The Stockholm Convention on POPs be ratified and implemented.**

Specific recommendations for monitoring activities in support of these agreements are included in subsequent sections.

Persistent Organic Pollutants

The POPs assessment addresses several chemicals of concern, including both substances that have been studied for some time and chemicals that have only recently been found in the environment.

The 1997 AMAP assessment concluded that levels of POPs in the Arctic environment are generally lower than in more temperate regions. However, several biological and physical processes concentrate POPs in some species and at some locations, producing some high levels in the Arctic.

The present AMAP assessment has found that the conclusions and recommendations of the first assessment remain valid. In addition:

It has clearly been established that:

Certain Arctic species, particularly those at the upper end of the marine food chain as well as birds of prey, carry high levels of POPs. Marine mammals, such as polar bear, Arctic fox, long-finned pilot whale, killer whale, harbor porpoise, minke whale, narwhal, beluga, harp seal and northern fur seal, some marine birds including great skua, great black-backed gull and glaucous gull, and birds of prey such as peregrine falcon, tend to carry the highest body burdens.

Most of the total quantity of POPs found in the Arctic environment is derived from distant sources. The POPs are transported to the Arctic by regional and global physical processes, and are then subjected to biological mechanisms that lead to the high levels found in certain species. Several potential source regions have now been identified within and outside of the Arctic. A better understanding of local re-distribution mechanisms has also emphasized the important potential role of local processes and sources in determining observed geographical variability.

There is evidence that:

Adverse effects have been observed in some of the most highly exposed or sensitive species in some areas of the Arctic. Several studies have now been completed on a number of Arctic species, reporting the types of effects that have been associated in non-Arctic species with chronic exposure to POPs, of which there are several examples. Reduced immunological response in polar bears and northern fur seals has led to increased susceptibility

* AMAP, 1997. Arctic Pollution Issues: A State of the Arctic Environment Report. Arctic Monitoring and Assessment Programme (AMAP), Oslo, Norway, xii+188 pp. *and*
AMAP, 1998. AMAP Assessment Report: Arctic Pollution Issues. Arctic Monitoring and Assessment Programme (AMAP), Oslo, Norway, xii+859 pp.

to infection. Immunological, behavioral, and reproductive effects as well as reduced adult survival has been found in glaucous gulls. Peregrine falcons have suffered from eggshell thinning and reproductive effects. Reproductive effects in dogwhelks are associated with exposure to tributyltin.

It is therefore recommended that:

- **AMAP be asked to further enhance studies aimed at detecting effects in Arctic species relating to exposure to high levels of POPs and to integrate this information with an understanding of general population effects and health.** Without this understanding, it will not be possible to assess whether proposed and existing controls can be expected to afford the necessary protection (e.g., under the LRTAP and Stockholm agreements).

There is evidence that:

The levels of some POPs are decreasing in most species and media in the Arctic, but the rates vary in extent, location and media or species being studied. The decreases can be related to reduced release to the environment. For example, declines in alpha-HCH in air closely follow decreases in global usage, but declines in marine biota are much slower due to a huge reservoir of the substance in the global oceans.

For other POPs, declines are minimal and some levels are actually increasing, despite low current emissions. This illustrates the long period that may pass between the introduction of controls and the resulting decrease in levels in biota, as has been observed for PCBs, toxaphene, and beta-HCH.

It is therefore recommended that:

- **AMAP be asked to continue trend monitoring of POPs in key indicator media and biota.** This will enable assessment of whether the measures taken in the LRTAP Protocol and the Stockholm Convention are being effective in driving down POPs levels in the Arctic.

There is evidence that:

POPs substances other than those included in the LRTAP Protocol and Stockholm Convention may be at or approaching levels in the Arctic that could justify regional and global action. For example, levels of the brominated flame retardants such as polybrominated diphenyl ethers (PBDEs), polychlorinated naphthalenes (PCNs), and some current-use pesticides such as endosulfan have been monitored in Arctic air and biota. PBDEs are increasing in the Canadian Arctic.

It is therefore recommended that:

- **AMAP be asked to maintain a capacity to detect current-use POPs in the Arctic.** This will help ensure that Arctic States have an early opportunity to respond to a trend indicating Arctic accumulation, thus allowing a proactive approach to minimize the contamination rather than having to respond to a more serious situation later.

Heavy Metals

The heavy metals assessment focuses on mercury, lead, and cadmium.

It has clearly been established that:

In the Arctic, mercury is removed from the atmosphere and deposits on snow in a form that can become bioavailable. Enhanced deposition occurs in the Arctic. This recently discovered process is linked to polar sunrise, and is unique to high latitude areas. The resulting enhanced deposition may mean that the Arctic plays a previously unrecognized role as an important sink in the global mercury cycle.

There is evidence that:

Some of the deposited mercury is released to the environment at snowmelt, becoming bioavailable at the onset of animal and plant reproduction and rapid growth. Although poorly understood, this process may be the chief mechanism for transferring atmospheric mercury to Arctic food webs.

It is therefore recommended that:

- **The Arctic Council encourage expanded and accelerated research on critical aspects of the mercury cycle and budget in the Arctic.** Such research should include long-range transport, mercury deposition mechanisms, processes leading to biological exposure and effects, and the influence of climate variability and change on these processes.

There is evidence that:

Despite substantial mercury emission reductions in North America and Western Europe during the 1980s, global mercury emissions may, in fact, be increasing. Mercury emissions from waste incineration are likely underestimated. The burning of coal in small-scale power plants and residential heaters, principally in Asia, are major potential sources of current mercury emissions. These emissions are likely to increase significantly due to economic and population growth in this region.

It is therefore recommended that:

- **The Arctic Council promote efforts at global, regional, and national levels to quantify all sources of mercury and report results in a consistent and regular manner to improve emission inventories.** Particular efforts should focus on measuring contributions made by the burning of coal for residential heating and small-scale power plants as well as by waste incineration.

There is strong evidence that:

There is a trend of increasing mercury levels in marine birds and mammals in the Canadian Arctic, and some indications of increases in West Greenland. The effects of these levels are not well understood. However, there are also examples of stable or decreasing levels in other regions, perhaps indicating the importance of local or regional processes.

It is therefore recommended that:

- **AMAP be asked to continue temporal trend monitoring and the assessment of effects of mercury in key indicator media and biota.** This will enable assessment of whether the measures taken in the LRTAP Protocol are being effective in driving down mercury levels in the Arctic.

There is evidence that:
Current mercury exposures pose a health risk to some people and animals in the Arctic. These risks include subtle neurobehavioral effects.

It is therefore recommended that:
- **In view of the fact that reducing exposure to mercury can only be addressed by regional and global action to reduce worldwide emissions, and acknowledging the assessment for global action undertaken by UNEP and its resulting proposals, the Arctic Council take appropriate steps to ensure that Arctic concerns are adequately addressed and to promote the development of regional and global actions.**

It has clearly been established that:
Dramatic reduction in the deposition of atmospheric lead has occurred in Arctic regions where the use of leaded gasoline is banned. Arctic-wide elimination of leaded gasoline use will reduce lead exposure in other regions of the Arctic. Although levels in wildlife and fish have not measurably declined, likely reflecting continued uptake from the large reservoir of lead deposited in soils and sediments, lead levels in the environment are expected to diminish over time if current trends continue.

It is therefore recommended that:
- **The Arctic Council support continued efforts to eliminate the use of leaded gasoline in all Arctic regions.**

It has clearly been established that:
Certain regions of the Arctic contain elevated lead levels in the environment because of past or current use of lead shot by hunters. Even though lead shot is banned in Alaska, for example, lead blood levels in endangered US populations of Steller's eiders are above known avian toxicity thresholds for lead poisoning, which may be responsible for observed reduced breeding success. In Greenland, lead shot appears to be a significant source of human dietary exposure to lead.

It is therefore recommended that:
- **The Arctic Council encourage a complete ban on the use of lead shot in the Arctic, and that enforcement be improved.**

There is evidence that:
Cadmium levels in some seabirds is high enough to cause kidney damage. Monitoring data on cadmium in the abiotic and biotic environment to date provide no conclusive evidence of trends or effects. However, cadmium accumulates in birds and mammals and not enough is known about possible effects.

It is therefore recommended that:
- **The monitoring of cadmium in the Arctic be continued to support human exposure estimates.**

There is evidence that:
Levels of platinum, palladium, and rhodium have increased rapidly in Greenland snow and ice since the 1970s. These elements are used in automobile catalytic converters to reduce hydrocarbon pollution. The toxicity and bioaccumulation potential of these elements are largely unknown, which prevents assessment of their potential impact in the Arctic.

It is therefore recommended that:
- **AMAP be asked to consider the need to monitor trends of platinum, palladium, and rhodium in the Arctic.**

Radioactivity

The radioactivity assessment addresses man-made radionuclides and radiation exposures deriving from human activities.

It has clearly been established that:
In general, levels of anthropogenic radionuclides in the Arctic environment are declining. Most of the radioactive contamination in the Arctic land environment is from the fallout from nuclear weapons testing during the period 1945 to 1980. In some areas, the fallout from the Chernobyl accident in 1986 is a major source. For the Arctic marine environment, a major source of radionuclides is the releases from European reprocessing plants at Sellafield and Cap de la Hague.

However, releases from the reprocessing plants have resulted in increases in levels of some radionuclides in the European Arctic seas during recent years, in particular technetium-99 and iodine-129. The present doses to the population are low but the present levels of technetium in some marine foodstuffs marketed in Europe are above the EU intervention levels for food to infants and are close to the intervention level for adults.

The technetium information adds further weight to the recommendation made by AMAP to the Arctic Council in Barrow in 2000 that:
- **'The Arctic Council encourage the United Kingdom to reduce the releases from Sellafield to the marine environment of technetium, by implementing available technology.'**

There is evidence that:
Radionuclides in sediments are now a source of plutonium and cesium-137 to the Arctic. Earlier releases such as those from Sellafield that have deposited in sediments in the Irish Sea, especially cesium-137 and plutonium, have been observed to remobilize so that these deposits are now acting as sources to the Arctic. Thus, even if operational releases of these radionuclides from reprocessing plants are reduced, releases from environmental sources such as contaminated sediment in the Irish Sea and the Baltic Sea will be observed in the Arctic.

It is therefore recommended that:
- **The Arctic Council support a more detailed study on the remobilization of radionuclides from sediment and its potential effect on the Arctic.**

It is apparent that:
There is continuing uncertainty about the amount of radionuclides present at a number of sources and potential sources in the Arctic. Access to information about civilian and military sources continues to be a problem.

It is therefore recommended that:

- **The Arctic Council promote more openness of restricted information from any sources.**

It has clearly been established that:

Compared with other areas of the world, the Arctic contains large areas of high vulnerability to radionuclides. This is due to the characteristics of vegetation, animals, human diets, and land- and resource-use practices. On land in the AMAP area, there is considerable variation in vulnerability due to differences in these characteristics. In contrast, vulnerability associated with releases of radionuclides to the marine environment is relatively uniform and similar to that for other areas of the world. Maps of vulnerable areas, when combined with deposition maps, can be useful in an accident situation. The information on vulnerability is of importance for emergency planning.

It is therefore recommended that:

- **AMAP be asked to clarify the vulnerability and impact of radioactivity on the Arctic environment and its consequences for emergency preparedness planning.**

It is apparent that:

When performing risk reducing actions, close links to assessment programs are important and interventions should be prioritized in relation to the extent and magnitude of threats posed by nuclear activities, especially in respect to accidents. Interventions themselves can also have negative effects for humans and the environment, and careful judgments have to be made together with environmental impact assessments prior to carrying out a project. It is the view of AMAP that this has not always been done in interventions adopted to date.

It is therefore recommended that:

- **Risk and impact assessment programmes be performed prior to implementation of action to reduce risk.**
- **Risk and impact assessments, including accident scenarios, be performed with regard to the transport of nuclear waste and fuel within the Arctic and nearby areas and with regard to planned storage and reprocessing within the Arctic and nearby areas.**

It is apparent that:

The protection of the environment from the effects of radiation deserves specific attention. The current system of radiological protection is entirely based on the protection of human health. This approach can fail to address environmental damage in areas such as the Arctic that have low human population densities. Recently, an international consensus has emerged that the rapid development of a system and a framework for the protection of the environment needs further effort. The International Union of Radioecology (IUR), with support from AMAP, was one of the first international organizations to promote and present such a system and framework.

It is therefore recommended that:

- **AMAP be asked to take an active part in the continued efforts to address environmental protection, with special responsibility for the Arctic.** This should include the task of adding the need for protection of the environment into monitoring strategies and assessment tools.

It is noted that:

Since the previous AMAP assessment, nuclear safety programmes have been implemented in Russia at some nuclear power plants and other nuclear installations relevant to the Arctic.

It is therefore recommended that:

- **The Arctic Council continue its cooperation with Russia to improve the safety and safeguarding of nuclear installations and waste sites.**

Human Health

The human health assessment considered health risks associated with exposure to contaminants in relation to other lifestyle factors determining health. This assessment has extended geographical coverage and confirmed the conclusions and recommendations from the first assessment.

It has clearly been established that:

The highest Arctic exposures to several POPs and mercury are faced by Inuit populations in Greenland and Canada. These exposures are linked mainly to consumption of marine species as part of traditional diets. Temporal trends of human exposures to POPs have so far not been observed. Exposure to mercury has increased in many Arctic regions while exposure to lead has declined.

It is therefore recommended that:

- **The monitoring of human exposure to mercury, relevant POPs, including dioxins and dioxin-like compounds and other chemicals of concern, be continued in order to help estimate risk, further elaborate geographical trends, and begin to establish time trends of exposure.**

There is evidence that:

Subtle health effects are occurring in certain areas of the Arctic due to exposure to contaminants in traditional food, particularly for mercury and PCBs. The evidence suggests that the greatest concern is for fetal and neonatal development. In the Arctic, human intake of substances with dioxin-like effects is a matter of concern, confirmed by recent results from Greenland. Increasing human exposure to current-use chemicals has been documented, for example for brominated flame retardants. Others such as polychlorinated naphthalenes (PCN) are expected to be found in human tissues. Some of these compounds are expected to add to the total dioxin activity in humans. The AMAP human health monitoring program includes a number of measures of effects, ranging from biomarkers of effects at the molecular level to epidemiological outcomes.

It is therefore recommended that:

- **The human health effects program developed by AMAP be more extensively applied in order to provide a better base for human risk assessment especially concerning pre- and neonatal exposures.**

It has clearly been established that:

In the Arctic, diet is the main source of exposure to most contaminants. Dietary intake of mercury and PCBs ex-

ceeds established national guidelines in a number of communities in some areas of the Arctic, and there is evidence of neurobehavioral effects in children in some areas. In addition, life-style factors have been found to influence the body burden of some contaminants, for example cadmium exposure from smoking. In the Arctic region, a local public health intervention has successfully achieved a reduction of exposure to mercury by providing advice on the mercury content of available traditional foods. The physiological and nutritional benefits of traditional food support the need to base dietary recommendations on risk-benefit analyses. The health benefits of breast-feeding emphasize the importance of local programs that inform mothers how adjustments within their traditional diet can reduce contaminant levels in their milk without compromising the nutritional value of their diet.

It is therefore recommended that:

- **In locations where exposures are high, carefully considered and balanced dietary advice that takes risk and benefits into account be developed for children and men and women of reproductive age.** This advice should be developed by national and regional public health authorities in close consultation with affected communities.
- **Studies of the nutrient and contaminant content of traditional food items be promoted in order to assess their benefits and to estimate exposures as a basis for public health interventions.**
- **Breast-feeding continue to be recognized as a practice that benefits both mother and child.** Nonetheless, if contaminant levels increase or more information indicates increased risk, the potential need for restrictions should continue to be evaluated.

It is noted that:

From the Arctic human health perspective, it is of utmost importance that considerations for global actions against POPs and mercury take into account the concerns for Arctic human health. The Stockholm Convention and the LRTAP protocols should be properly monitored in the Arctic to determine whether their implementation is effective in protecting human health.

It is therefore recommended that:

- **AMAP participate in the global monitoring of human exposure to be established under the Stockholm Convention on POPs.**
- **The Arctic Council monitor proposals for global action on mercury being undertaken by UNEP, and contribute as necessary to ensure that Arctic concerns related to human health are adequately addressed.**

Changing pathways

The assessment of changing pathways provides an introduction to the types of changes on contaminants pathways to, within, and from the Arctic that might be expected as a result of global climate change and variability.

There is evidence that:

The routes and mechanisms by which POPs, heavy metals, and radionuclides are delivered to the Arctic are strongly influenced by climate variability and global climate change. These pathways are complex, interactive systems involving a number of factors, such as temperature, precipitation, winds, ocean currents, and snow and ice cover. Pathways within food webs and the effects on biota may also be modified by changes to climate. Studies using global change scenarios have indicated the potential for substantial changes in atmospheric and oceanographic pathways that carry contaminants to, within, and from the Arctic. These effects mean that climate-related variability in recent decades may be responsible at least in part for some of the trends observed in contaminant levels.

It is therefore recommended that:

- **AMAP be asked to further investigate how climate change and variability may influence the ways in which POPs, heavy metals, and radionuclides move with respect to the Arctic environment and accumulate in and affect biota.** This will enable Arctic States to better undertake strategic planning when considering the potential effectiveness of present and possible future national, regional, and global actions concerning contaminants.

Chapter 1

Assessment of Heavy Metals in the Arctic

Suzanne Marcy

Heavy metals occur naturally in all ecosystems, although concentrations vary according to local geology. These natural levels may be increased or decreased by the redistribution of heavy metals within the environment by anthropogenic activities occurring at local, regional, and global scales. This redistribution can lead to heavy metals occurring at concentrations above naturally occurring levels in locations remote from anthropogenic sources, and in forms available for biotic uptake.

Following its first assessment of the state of the Arctic environment the Arctic Monitoring and Assessment Programme (AMAP) concluded that heavy metals are a potential risk to the ecosystems and biotic communities of the Arctic. At the request of the Arctic Council (AEPS, 1997) AMAP was directed to conduct further investigations and to generate an updated report on heavy metals in the Arctic. This updated information is presented in this report.

1.1. Background

AMAP was established in 1991 to implement components of the Arctic Environmental Protection Strategy (AEPS) as adopted at that time by Ministers of the eight Arctic countries. AMAP was charged with priority actions to monitor the levels and assess the effects of anthropogenic pollutants in the Arctic, and to prepare assessment reports on the state of the Arctic environment. The scope of the work includes the assessment and monitoring of pollutants released from sources within the Arctic, as well as those released from sources at lower latitudes, and to determine the transport pathways for pollutants to, and within the Arctic. Measures of pollutant concentrations in abiotic and biotic environmental compartments are used to identify spatial patterns and temporal trends. In addition, efforts are made to determine the effects of contaminant exposure on humans and other biota. Target pollutants include persistent organic pollutants (POPs), heavy metals, radioactivity, acidifying substances, and petroleum hydrocarbons. The effects of climate change, ozone depletion, and ultraviolet-B radiation are also addressed.

Information compiled during Phase I (AMAP, 1998) increased the understanding of Arctic environmental contamination and human health during the 1990s. This served as the foundation for activities continued under Phase II. Specific assessments for POPs, heavy metals, radioactivity, contaminant pathways, and human health were undertaken between 1998 and 2002.

1.1.1. Major conclusions from Phase I

The major conclusions from the first AMAP assessment include the following.

- Compared to global background levels: 1) atmospheric concentrations of heavy metals in the Arctic were low except near point sources; 2) heavy metals in Arctic soils, lakes, rivers, and oceans generally fell within global ranges; 3) cadmium (Cd) levels were high in some terrestrial birds and mammals, and mercury (Hg) levels were high in some freshwater fish.
- The distribution of heavy metals among the various environmental compartments of the Arctic is dynamic and apparently driven by multiple processes both natural and anthropogenic. Spatial patterns in surface deposition were most apparent in areas around sources of atmospheric emission. Spatial patterns in abiotic concentrations appeared to relate to the geological provinces of the Arctic. In general, spatial patterns in Arctic biota tissue concentrations were not readily apparent.
- Temporal trend data were scarce for Arctic biota and conclusions were considered premature. Preliminary findings included a general decrease in heavy metal concentrations in subarctic air compared to the 1970s and 1980s, although with strong seasonal variations. Mercury concentrations in Arctic sediments appeared to increase over time suggesting a widespread regional process, although further investigations were recommended to understand potential relationships with anthropogenic fluxes.
- Metals taken up by biota reflect both local geology and anthropogenic activities. However, the effects of heavy metal exposure on Arctic biota had not been sufficiently investigated. Mercury and Cd were considered to be the metals of most concern because they were found in some Arctic biota at concentrations that may affect individual health or reproduction.

1.1.2. Gaps in knowledge and recommendations from Phase I

A number of gaps in knowledge and understanding were identified during Phase I. This led to the following guidance and recommendations.

- Heavy metal data collected for AMAP applications should be standardized for consistency in sampling, analysis, and reporting with a more comprehensive quality assurance/quality control (QA/QC) protocol that will enhance intercomparability of datasets.
- Mass balance studies should be undertaken for heavy metals in the Arctic air, rivers, and oceans. Increased source-receptor modeling is required to develop quantitative strategies for formulating policy decisions among nations.
- The data gaps for biota (e.g., sample sizes, important species, geographical regions) should be filled, but priority should be given to metals and organisms for which there are concerns about biological effects.

- Studies on the processes behind geographical differences and trends in the Arctic should be initiated and supported.
- Additional data should be acquired to establish temporal trends in Cd and Hg concentrations in a wider number of Arctic ecosystem compartments. Processes behind potential trends should be studied to resolve whether the changes are natural or man-made.
- Health effects should be studied in Arctic species that have body burdens containing Cd and Hg levels of concern.

The need for expanded and updated emission source inventories was targeted. Regional gaps (e.g., the collection of heavy metal data for the Arctic marine environment in Alaska, the Faroe Islands, Iceland, and Russia) were identified. The importance of understanding the spatial distribution and temporal trends of heavy metals within the High Arctic resulted in recommendations for expanded monitoring studies, a continuation of existing time series, the implementation of new retrospective trend studies (e.g., peat and sediment cores, analyses of banked or preserved specimens), and increased validation of data.

1.1.3. Research plan for Phase II

Ministers at the fourth Ministerial Conference on the Arctic Environmental Protection Strategy (AEPS) held in Alta, Norway in June, 1997 endorsed AMAP's assessment activities and asked that monitoring, data collection, and exchange of data on the pathways, trends, and effects of contaminants continue based on updated priorities. In order to address these requests from Ministers, the AMAP Heavy Metals Expert Group reviewed and revised the assessment program implemented during Phase I. Initial discussions of the program plan for the first AMAP assessment were completed during a meeting in April 1998. A modified and expanded program was finalized in September 1999 (AMAP, 1999).

Priorities under Phase II included the assessment of atmospheric emissions within and outside the Arctic region, and the development of atmospheric transport models to determine source apportionment based on updated atmospheric emissions and measurements of heavy metal concentrations in Arctic air. A detailed trend monitoring program was established for atmospheric, marine, and freshwater abiotic media, as well as for marine, freshwater, and terrestrial biota. Although Phase II includes the assessment of biological effects, it was recognized that current knowledge would limit their identification.

1.2. Phase II

In order to update the AMAP Phase I assessment, this assessment targets specific recommendations by the Arctic Council (AEPS, 1997) to fill data gaps for biota, with priority given to metals and organisms for which there are concerns about biological effects, and to identify spatial patterns, temporal trends and biological effects. It is based mainly on data collected since the end of the first AMAP assessment; however it also includes older data where these were not previously assessed, or where they are relevant to discussions of, for example, temporal trends. Human health issues related to heavy metals are addressed in a separate report (AMAP, 2003).

This assessment is structured so as to address the release, transport and transformation processes that bring heavy metals into the Arctic region (Chapters 2 and 3), the fate of heavy metals in space and time (Chapters 4 and 5), and the potential biological effects of heavy metals (Chapter 6). Conclusions and recommendations arising from the assessment are presented in Chapter 7.

1.2.1. Scope of Phase II

This assessment represents the most comprehensive compilation of information about heavy metals in the Arctic to date. It includes new information on the means by which heavy metals are reaching and re-distributing within the Arctic environment, how levels of heavy metals are changing over time, their distribution within the Arctic abiotic and biotic environments, and the potential for effects in Arctic species from exposure to heavy metals. The report is not intended to be a risk assessment. However, the assessment is organized to facilitate such future synthesis and integration.

This assessment adds to and updates the outcome of Phase I (AMAP, 1998). The results of the first AMAP assessment are not repeated except where necessary for continuity and trend assessment.

For the purposes of this assessment the term 'heavy metal' is broadly defined to encompass all inorganic elements that are potential pollutants in the Arctic. Metals are a class of elements or elemental substances that are typically characterized by opacity, ductility, conductivity, and luster. Depending on the metal and its form, they can be toxic to living organisms in minute amounts. Some elements such as copper (Cu) and zinc (Zn) are essential for health at low levels but toxic in excess. The criteria for defining and prioritizing metals addressed in the assessment, as well as general information on their physical and chemical characteristics and their speciation, were provided during the first AMAP assessment (AMAP, 1998). This assessment focuses on Hg, lead (Pb), and Cd. Other elements were included as data availability permits.

Substantial progress has been made in linking transport pathways to the distribution of heavy metals within the environment, particularly for Hg. Data on the concentrations of heavy metals in air, water, ice, soils, sediments, and plants, and body burdens in animal tissues, were used to identify spatial distributions and the changes in heavy metal concentrations over time. Although the present assessment contains significant advancements relative to the first assessment, there is a clear need for continued monitoring to improve the identification of spatial patterns and temporal trends. In terms of effects, data could only be interpreted within the context of how body burdens of heavy metals may directly affect individuals and populations. The potential for effects was mainly evaluated using laboratory-derived toxicity thresholds since empirical evidence of effects was limited.

With the exception of the effects of smelter emissions reported in Chapter 6, little information is included on the potential interaction of one or more heavy metals with other pollutants or the cumulative impacts of multiple pollutants and other stressors

1.2.2. Data used in Phase II

AMAP assessments are intended to provide scientific information that serves as the foundation for policy decisions and the implementation of activities recommended by the Arctic Council and member nations. As such, careful selection and interpretation of data has been essential.

1.2.2.1. Sources and quality assurance

Data used in this assessment were obtained from the peer-reviewed literature and supplemented by reliable agency reports and recent unpublished data. These include major national and international research programs (e.g., CACARII, 2003; Melnikov *et al.*, 2002). Information on ongoing monitoring and research activities within the Arctic was obtained from those governmental and non-governmental agencies responsible for conducting and/or funding much of the research in the Arctic. When questions arose concerning reported data or the interpretation of data, further information and clarification was obtained from the original researchers. Quality assurance and quality control issues were addressed as material was selected for each chapter. Data provided to AMAP were used with the original authors' interpretation and then integrated with other data in the formulation of conclusions.

The three Thematic Data Centers (TDCs) maintained by AMAP to house and process heavy metal and other contaminant data are the other major sources of information. These include the Atmospheric TDC at the Norwegian Institute of Air Research, the Marine TDC at the International Council for the Exploration of the Sea, and the Terrestrial and Freshwater TDC at the University of Alaska, Fairbanks. These data centers were established to provide for the compilation of and international access to, data from recent monitoring and research activities conducted as a part of the AMAP National Implementation Plans. Original authors have primary responsibility for quality assurance of the data.

Those data not easily accessed through the literature or the TDCs are included in data annexes. These include data on the abiotic and biotic compartments of the terrestrial, freshwater, and marine environments and are referred to in the appropriate chapters. These data annexes include data on air (Table A·1); ice/snow (Table A·2); soil (Table A·3); vegetation (Table A·4); terrestrial birds (Table A·5); terrestrial mammals (Table A·6); lake/river water (Table A·7); freshwater particulates (Table A·8); freshwater sediments (Table A·9); freshwater fish (Table A·10): seawater (Table A·11); marine sediments (Table A·12); marine invertebrates (Table A·13); marine fish (Table A·14); seabirds (Table A·15); and marine mammals (Table A·16).

1.2.2.2. Uncertainties

Data collected for Phase II added significantly to the general understanding of heavy metals in the Arctic. As with any scientific assessment, there are inherent uncertainties in the data and their interpretation. Uncertainties were particularly apparent when identifying spatial patterns, temporal trends, and effects.

The identification of spatial patterns can be problematic depending on the types of data available and how they are reported. Data are generally collected by different investigators, in different regions, using a variety of field and laboratory methods, reporting conventions (e.g., dry and wet weight), and measures of central tendency (e.g., arithmetic mean, geometric mean, and median). Because some heavy metals bioaccumulate, tissue concentrations should be normalized to size or age of an individual. However, these relationships are not routinely reported. In addition, biological processes can alter size and age relationships, and seasonal factors can influence tissue concentrations. These factors directly affect the intercomparability of data sets and make the detection of spatial patterns challenging.

Temporal trends are best detected using long-term data sets collected using consistent parameters and analytical techniques. Such data sets are rare. As with spatial patterns, data are often collected by different investigators using different approaches over relatively short time periods. Due to the inherent challenges of interpreting data from multiple investigators, statistical analyses combining data from several studies were performed in only a few cases for this assessment; for cases in which data were sufficiently comparable over the period of record. Statistical analyses of temporal trend data sets of sufficient length were performed at an AMAP/ICES workshop in 2001 (ICES, 2002). Where trends could be identified, results may be considered relatively robust.

The ability to detect biological responses that constitute effects from heavy metal exposure in wild populations is also limited. Types of effects vary significantly among contaminants and species, and can be subtle. The sensitivity of tissues to metal accumulation varies within an organism, and varies widely between species. Some species are able to tolerate high tissue concentrations of selected metals without apparent ill effects. In some cases, metals may be sequestered or depurated in tissues such that measurable levels in tissues have no relation to effects (e.g., feathers) or can be stored for long periods prior to causing effects (e.g., Cd sequestration in the kidney). In addition, establishing dose-response relationships is fundamental to interpreting data on tissue concentrations. For many heavy metals, this understanding must extend to the more complex role of those elements that are essential at low levels and toxic at higher levels (e.g., Zn and Cu). Identification and correct interpretation of biological effects is best achieved when exposure history (dosage) can be directly linked to tissue concentration, as well as to biological responses. This combination of data, however, is rarely reported as field studies cannot usually estimate dose, and laboratory studies typically do not report tissue concentrations. Therefore, the link between field and laboratory studies is weak. As such, failure to observe effects does not confirm the absence of effects. When effects are observed, they warrant consideration.

1.2.3. Assessments in a changing environment

Conclusions drawn from this assessment have been made in the light of noted uncertainties. Additional uncertainties exist owing to rapid changes ongoing in the Arctic environment in response to global change. As the Arctic responds to rapid change in climate, the challenge of interpreting pathways, patterns, trends, and effects of heavy metals within a highly variable natural system increases. It is yet to be determined how altered atmospheric pressure patterns, modification of ocean circulation and properties, loss of permafrost in broad regions, and other changes are likely to influence heavy metal distribution. However it is more certain that these changes will affect the distribution, transformation, and fate of heavy metals. The nature of recent changes and their potential implications for the distribution of contaminants, including heavy metals, is explored more fully in a complementary AMAP report on contaminant pathways (Macdonald *et al.*, 2003).

Chapter 2

Sources and Emissions

Jozef Pacyna

Atmospheric emissions of heavy metals, particularly within the Northern Hemisphere, encompass the best available data on sources of heavy metals to the Arctic. This chapter reports recent findings concerning the sources, levels, and relative magnitude of anthropogenic and natural emissions of heavy metals to the atmosphere. A comparison of total global emissions of heavy metals from the major anthropogenic sources in 1983 and 1995 is presented in Figure 2·1.

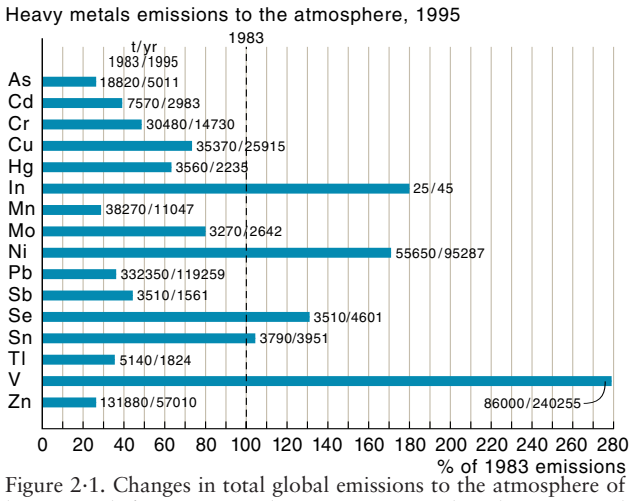


Figure 2·1. Changes in total global emissions to the atmosphere of heavy metals from major anthropogenic sources based on 1983 and 1995 data (Pacyna and Pacyna, 2001).

Although clearly important to the global budget, discharges of heavy metals to terrestrial, freshwater, and marine environments from point and non-point sources are not sufficiently studied to allow detailed evaluations of their contributions to the Arctic. As such they are not addressed in this chapter. However, because heavy metals can be transported to the Arctic via rivers and ocean currents, the role of surface waters is addressed briefly in Section 2.6 and more fully in Section 3.4.

2.1. Anthropogenic sources of emissions to the atmosphere

Although metals occur naturally and have accumulated and cycled throughout the environment over geological time periods, significant quantities of heavy metals are now introduced and redistributed in the global environment from human activities such as industrial processes, agricultural practices, transportation, and waste disposal. Anthropogenic activities now add heavy metals to surficial environmental compartments at significantly greater rates than natural processes, with the possible exception of volcanic sources. Although it was very difficult to assess the relative importance of different heavy metal contributions to the Arctic environment in the first AMAP assessment (AMAP, 1998), it was postulated that anthropogenic emissions from sources within and outside the Arctic region were more significant than natural emissions for many heavy metals. More recent assessments support this proposal.

Improved national and regional emission inventories now provide a better understanding of global heavy metal emissions, and focus mainly on the release of contaminants from anthropogenic sources (Pacyna and Pacyna, 2001). Figure 2·2 (p. 6) summarizes the outcome of an assessment of metals emitted from anthropogenic sources based on 1995 inventories. This represents major progress in assessing the anthropogenic atmospheric emissions of heavy metals for many countries and regions, including Europe, since the first global emission estimate (Nriagu and Pacyna, 1988). Based on existing emission estimates, there are three main anthropogenic sources of heavy metals to the atmosphere: fossil fuel combustion, non-ferrous metal production, and waste incineration.

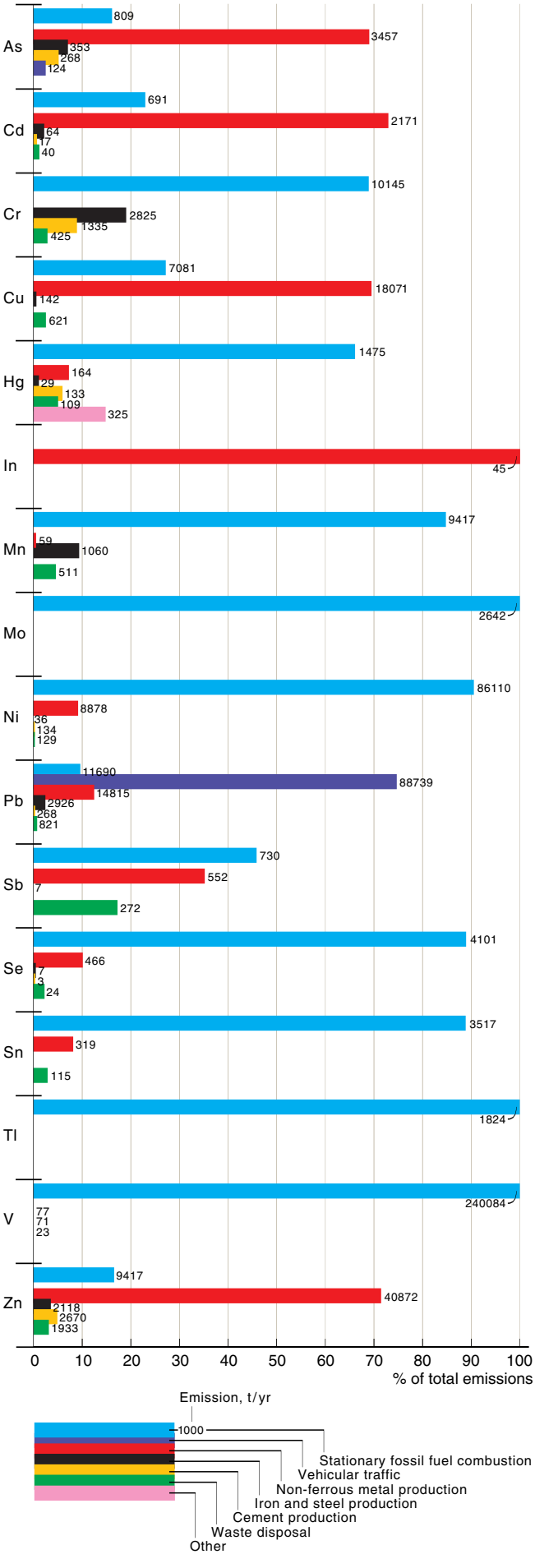
2.1.1. Fossil fuel combustion

At the global scale, stationary fossil fuel combustion continues to be the major source of heavy metal emissions (Pacyna and Pacyna, 2001). Coal combustion contributes chromium (Cr; 69%), mercury (Hg; 66%), manganese (Mn; 85%), antimony (Sb; 47%), selenium (Se; 89%), tin (Sn; 89%), and thallium (Tl; almost 100%) to atmospheric emissions. The dominance of this source persists despite heavy metal emissions from coal combustion in the 1990s having halved since the 1980s due to the installation of abatement technologies, primarily in North America and Western Europe.

Combustion of leaded, low-leaded, and 'unleaded' gasoline continues to be the major source of atmospheric lead (Pb) emissions, contributing 74% to the total anthropogenic emissions of Pb in 1995. Oil combustion was the major source of nickel (Ni; 90%) and vanadium (V; almost 100%). Emissions of Ni and V from oil combustion increased by a factor of two to three between the early 1980s and the mid-1990s. Because most oil-fired power plants are not equipped to remove particles or metals such as Ni and V on these particles, and because it is unlikely that the Ni and V content of crude oil changed significantly over this period, the doubling of Ni and V emissions from oil-fired power plants is probably due to increased production of electricity and heat from this source worldwide. Increases in emissions closely follow a comparable 2.2 factor increase in oil combustion due to rising global electricity and heat demands (UN, 1997).

2.1.2. Non-ferrous metal production

Another major source of heavy metals to the atmosphere is non-ferrous metal production. Emissions during production account for the largest source of atmospheric arsenic (As; 69%), cadmium (Cd; 73%), copper (Cu; 70%), indium (In; 100%), and zinc (Zn; 72%). Heavy metals emitted from non-ferrous metal production decreased by a factor of two to three from the early 1980s



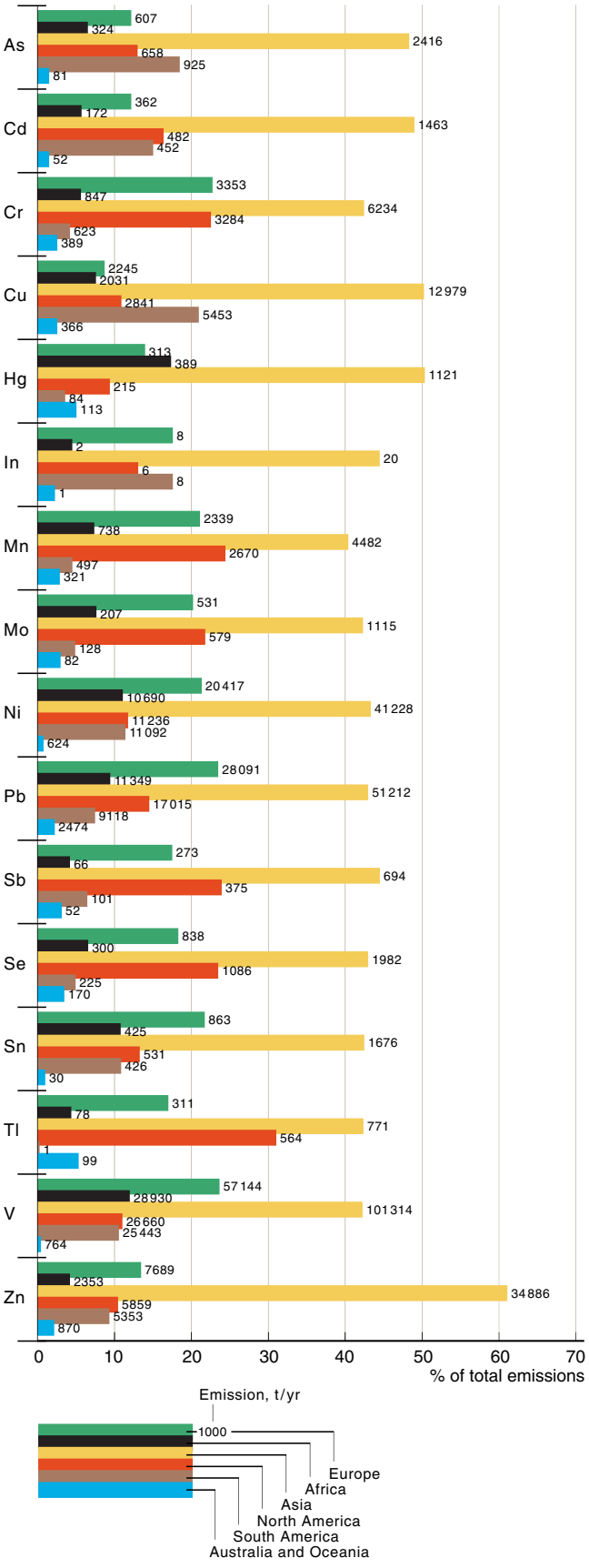


Figure 2·3. Worldwide emission estimates of anthropogenic heavy metals by continent (Pacyna and Pacyna, 2001).

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Figure 2·2. Metals emitted from anthropogenic sources based on 1995 inventories (Pacyna and Pacyna, 2001).

to the mid-1990s. This is largely due to the improvement in emission control efficiency at major smelters in Europe and North America, and is an example of the success of regulatory measures and improved technology.

2.1.3. Waste incineration

Waste incineration is also an important source of heavy metal emissions but estimating emissions from this source remains a challenge. Information is incomplete because a number of countries do not report their emissions from waste incineration, and inaccurate or inconsistent emission factors are used to calculate heavy metal emissions from waste incineration for countries that do report national emissions. In addition, information on the amount of municipal waste and sewage sludge incinerated is incomplete. These limitations probably result in underestimates of heavy metal emissions from waste incinerators, particularly for As, Cd, Cr, Mn, Pb, Sb, Se, Sn, and Zn.

2.2. Anthropogenic atmospheric emission estimates by region

The Arctic environment is affected by emissions in source regions outside the Arctic as well as source regions within the Arctic. Emission estimates for various continents are presented in Figure 2·3.

2.2.1. Arctic emissions

The first AMAP assessment showed that about two-thirds of the heavy metals in winter air over the High Arctic are derived from sources outside the Arctic region; on the Kola Peninsula, the Norilsk region, the Pechora Basin, and the Urals. A recent assessment of heavy metal emissions from the copper-nickel industry on the Kola Peninsula (Boyd *et al.*, 1998), reported emissions for 1994 of 1916 t for Ni, 1097 t for Cu, and 92 t for cobalt (Co; Murmansk Region Committee of Ecology and Natural Resources, 1995). Norilsk plant emissions contributed 1280 t of Ni, 2380 t of Cu, and 67.5 t of Co in 1994 (Surnin *et al.*, 1997).

According to Russian authorities, Norilsk emissions had decreased to 800 t of Ni and 1750 t of Cu by 1999 (S. Gromov, pers. comm., 2002). Thus, the non-ferrous metal industry on the Kola Peninsula and in the Norilsk region is thought to contribute about 3% of Ni and 10% of Cu to current global emissions. The majority of these emissions deposit within the emission region. Therefore, the impact of Ni and Cu emissions from sources on the Kola Peninsula and the Norilsk region is most significant close to the smelter stacks in the Russian Arctic and of less significance in the High Arctic. Based on current information, Asian sources are expected to become more significant for the High Arctic than sources in the Russian Arctic due to anticipated increases in emissions, particularly in China.

2.2.2. Emissions from non-Arctic regions

Asian emissions to the global environment are clearly the largest for all metals based on current estimates, accounting for between 40 and 60% of total emissions depending on the metal. While the majority of these emissions deposit in the emission region, an estimated 5 to 10% are deposited in the High Arctic.

Asian and South American countries are the largest emitters of As and Cd from the non-ferrous metal industry. Asian contributions for As are estimated at 50%, and Cd at 54%, of total global emissions; South America contributes approximately 25% of As and 20% of Cd. As these regions continue to develop, contributions are likely to rise.

While some emissions have increased, Pb emissions have decreased (see also Chapter 3). The average worldwide Pb emissions were estimated at 88 700 t in 1995, compared to 116 700 t in 1989 (Pacyna *et al.*, 1995) and 248 000 t in 1983 (Nriagu and Pacyna, 1988). Decreases in Pb emissions between the early 1980s and mid-1990s correspond well with the implementation of strategies to reduce Pb emissions through the introduction of low-leaded and 'unleaded' (e.g., < 0.013 g/L; Pacyna *et al.*, 1995) gasoline in Europe, the United States, and Canada.

2.3. Natural sources and emissions to the atmosphere

Although natural sources emit significant amounts of heavy metals to the environment, information on these emissions is still difficult to obtain. An assessment by Nriagu (1989) found that biogenic sources can account, on average, for over 50% of the Se, Hg, and Mo, and from 30 to 50% of the As, Cd, Cu, Mn, Pb, and Zn, released annually to the atmosphere from natural sources. Volcanic releases can account for 40 to 50% of the Cd and Hg, and from 20 to 40% of the As, Cr, Cu, Ni, Pb, and Sb. Soil-derived dusts are thought to account for over 50% of the total Cr, Mn, and V emissions, as well as 20 to 30% of the Cu, Mo, Ni, Pb, Sb, and Zn released annually to the atmosphere. Sea salt aerosols are thought to account for less than 10% of atmospheric heavy metals emitted from natural sources. Since the accuracy of emission estimates for natural sources is low, these percentage contributions should be considered with caution.

Figure 2·4 compares the global anthropogenic emission estimates by Pacyna and Pacyna (2001) with the global natural emission estimates by Nriagu (1989).

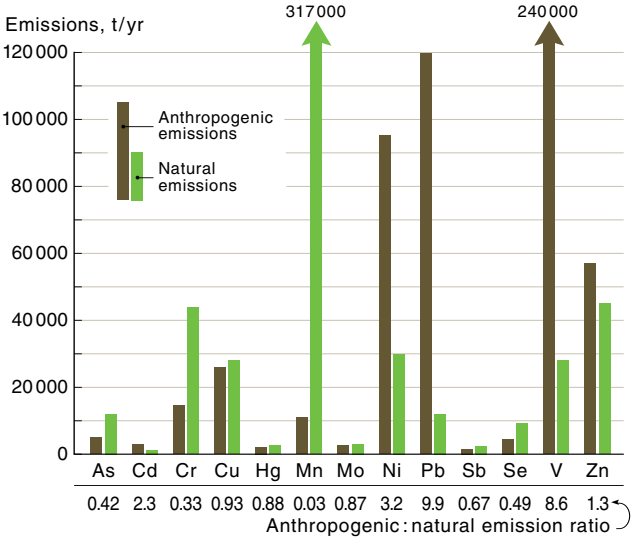


Figure 2·4. Estimated global anthropogenic emissions of heavy metals in the mid-1990s (Pacyna and Pacyna, 2001) compared to estimates from natural sources (Nriagu, 1989).

This suggests that at the global scale anthropogenic emissions of Pb and V are an order of magnitude higher than natural emissions. Anthropogenic emissions of Cd and Ni appear to be higher than natural emissions by a factor of two to three. Conversely, global natural emissions of As, Cr, and Se are two to three times higher than global anthropogenic emissions. Natural sources of Mn are more significant than anthropogenic sources. Natural and anthropogenic emissions appear comparable for Cu, Hg, Mo, Sb, and Zn.

On the basis of this comparison, data appear consistent between the mid-1990s and earlier studies concerning the relative significance of anthropogenic sources of heavy metals to the total budget. The only exception is possibly As, often regarded as very much an anthropogenic heavy metal. Although the As emission estimate from natural sources in Figure 2·4 is quite high, this may represent an overestimate.

There is increasing interest in evaluating natural emissions of heavy metals to the atmosphere. For Cd, Cu, Pb, Ni, and Zn, the entrainment of soil dust particles into the air may be a major source of natural emissions. For Hg, volcanic emissions may predominate (Nriagu and Becker, 2003). Natural sources of Cd, Cu, Hg, Ni, Pb, and Zn may be more significant than previously thought compared to anthropogenic sources of atmospheric emission. Re-emissions of previously deposited Hg derived from anthropogenic sources must be carefully considered in any analysis of natural Hg emissions.

While these relationships may be valid when evaluating the significance of anthropogenic versus natural sources at the global scale, the relationships can be entirely different at the local (e.g., around a waste incinerator) and regional (e.g., volcanic activity, industrial areas) scale, such as in central Europe or the Mediterranean Sea basin. Local and regional scale impacts can be of greater significance than global relationships.

Recent progress in measuring Hg fluxes in the field (e.g., Gustin *et al.*, 1999; Lindberg *et al.*, 1995) has significantly improved the global database for estimates of natural source emissions. New estimates are now being developed and should be incorporated into future AMAP databases (e.g., Ferrera *et al.*, 2000; Gustin and Lindberg, 2000; Gustin *et al.*, 2000; Lindberg and Stratton, 1998).

2.4. Physical and chemical forms of heavy metals emitted to the atmosphere

The physical and chemical forms of heavy metals emitted to the atmosphere influence their behavior, fate, and removal during long-range atmospheric transport. As such, these properties influence how metals are transported to the Arctic environment following their emission in various regions of the globe.

2.4.1. Particulates

Most heavy metals are emitted as aerosols with a wide range of particle size. Once they enter the atmosphere, heavy metals are capable of being transported for distances of up to a few thousand kilometers, depending on particle size and metal solubility. Heavy metals on larger particulates deposit closer to their source and have more local and close regional impacts.

2.4.2. Gaseous phase

Heavy metals emitted in the gaseous phase can be transported long distances from the source and can become globally distributed. Most Hg and some Se is emitted in the gaseous phase.

2.4.3. Physical and chemical forms of mercury

Studies on the chemical forms of mercury in atmospheric emission plumes were recently completed (Pacyna and Pacyna, 2002) and included estimates of gaseous elemental mercury (Hg^0, GEM), gaseous bivalent mercury (also called reactive gaseous mercury), and particulate mercury; the three most important species of Hg emitted from anthropogenic sources. Emission profiles are presented in Figure 2·5. Since the emission source can directly influence chemical speciation and physical characteristics, profiles are presented by source category.

Source type is important in determining the chemical speciation and physical form of Hg emitted to the atmosphere. It is a function of both the chemical status of the Hg in the raw materials (e.g., in coal, ores, or wastes), and the industrial technology employed (e.g., different temperatures and chemical conditions in a coal-fired power plant compared to a waste incinerator). Detailed information is often needed on these industrial technologies; for example temperatures at coal-fired power plants vary depending on whether combustion occurs in a traditional combustion boiler or in a pulverized coal boiler. The chemical composition of the raw materials also affects the formation of Hg species (e.g., higher chlorine levels in coal cause greater formation of gaseous bivalent Hg compared to GEM) although information on halogens in fossil fuels is currently limited (Bettinelli *et al.*,

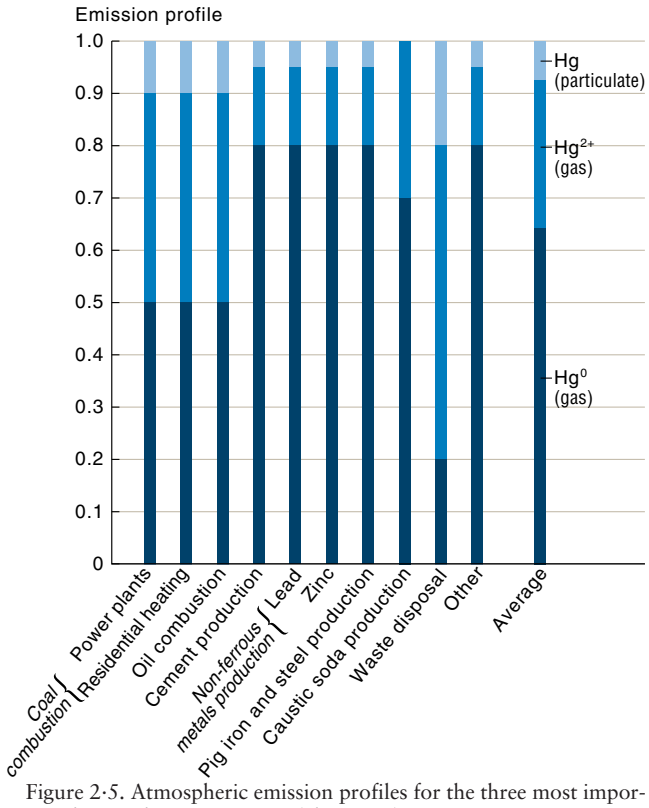


Figure 2·5. Atmospheric emission profiles for the three most important forms of mercury emitted from anthropogenic sources in 1995 (Pacyna and Pacyna, 2002).

2002). Higher concentrations of fine particulates in exhaust gases improve conditions for gas-to-particle conversion, increasing the particulate proportion emitted. The type of emission control device, particularly for flue gas desulfurization, also affects the chemical forms of Hg emitted.

The major chemical form of Hg emitted to the atmosphere is GEM, contributing about 53% to total emissions, followed by gaseous bivalent Hg (37%). Particulate Hg emissions contribute the remaining 10%. Asia contributes about 50% to the total emissions of all individual chemical forms of mercury (see Section 2.2.2). The main source of these emissions is coal combustion for electricity and heat, particularly in China. Chinese emissions from power plants and small industrial and commercial furnaces account for about 500 t/yr, contributing more than 25% to the total global Hg emissions.

The accuracy of the estimates for individual source categories varies, ranging from ±25% for stationary fossil fuel combustion to a factor of up to five for waste disposal. Accuracy estimates for non-ferrous metal production, cement production, and iron and steel production are ±30%.

Emission data for several countries in Europe and North America have been evaluated by national emission experts and are considered more accurate than the emission estimates for other continents. Emission estimates for waste disposal are considered inaccurate owing to a lack of information on Hg emissions for most countries and few reliable data on waste incineration and the Hg content of wastes.

The information on physical and chemical forms of mercury emissions is based on progress since the interim report by Axenfeld *et al.* (1991), and was completed through the European Union projects on Mercury over Europe (Munthe, 2000) and the Mediterranean Atmospheric Mercury Cycle System (Pirrone, 2000). The data are consistent with earlier reports on Hg speciation near point sources in the United States (Lindberg and Stratton, 1998; Stratton and Lindberg, 1995). Note that recent discoveries regarding the dynamic oxidation of GEM in the Arctic atmosphere (Lindberg *et al.*, 2002; Schroeder *et al.*, 1998) suggest that initial estimates of emission speciation from sources will probably change during long-range transport (see Chapter 3 and Loon *et al.*, 2000).

2.5. Spatial distribution of mercury emissions

The 1995 global anthropogenic Hg emission data, including total Hg, GEM, gaseous bivalent Hg, and particulate Hg from primary sources, have been spatially distributed within a 1° by 1° grid system, identical with the Global Emission Inventory Activities (GEIA) grid system. These emission maps are shown in Figures 2·6 and 2·7 and may be used as the basis for models assessing the transport and atmospheric deposition of anthropogenic Hg in the Arctic. Global distribution is also influenced by losses to surface waters and soils, climatic change, revolatilization, and other processes. Such processes have not been accounted for in these emission maps.

2.6. Riverine inputs of heavy metals to the Arctic region

Recent information on riverine inputs of contaminants to the Arctic region, obtained from the Joint Russian-French-Dutch Scientific Program on Arctic and Siberian Aquatic Systems (SPASIBA; see Kimstach *et al.*, 1998), together with data presented in the first AMAP assessment, shows rivers to be a source of heavy metals to the Arctic Ocean. Mercury concentrations measured during SPASIBA, and other Russian projects (Kimstach *et al.*, 1998) provide an estimate of riverine fluxes of dissolved and particulate Hg to the Arctic Ocean. While the importance of rivers as a source of heavy metals to the Arctic should not be minimized, in this assessment rivers are treated primarily as a transport mechanism rather than a source. Anthropogenic discharges to, and the natural heavy metal loads of rivers flowing to the Arctic Ocean must be better documented in order to assess the importance of this source. Section 3.4 discusses rivers as a pathway for heavy metal transport and includes a summary of current knowledge concerning heavy metal loads for rivers discharging into the Arctic Ocean.

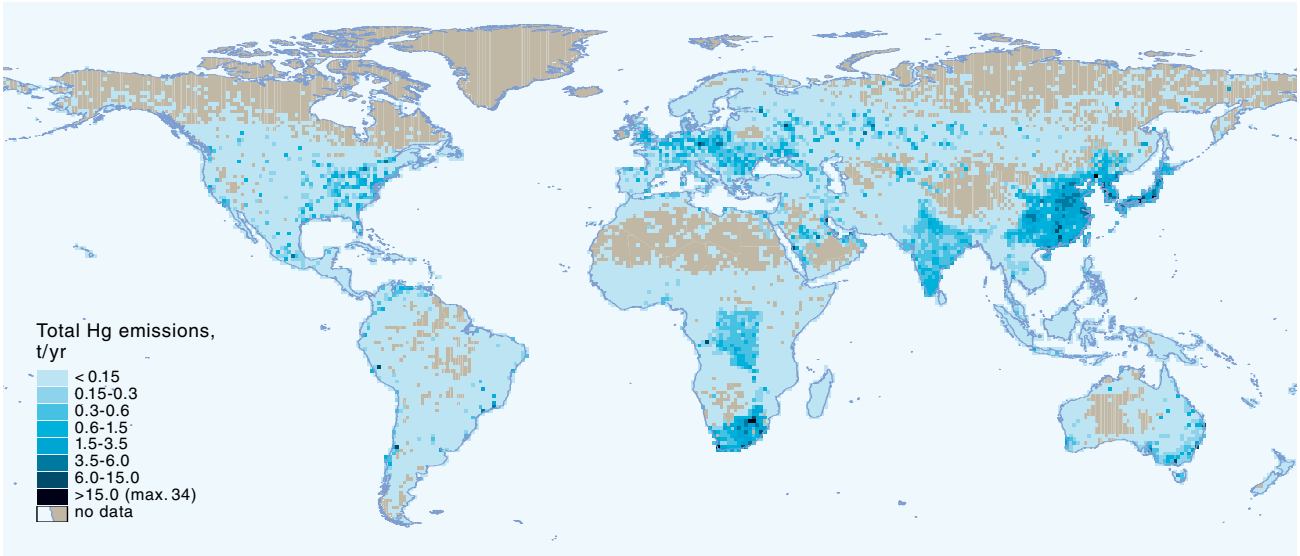
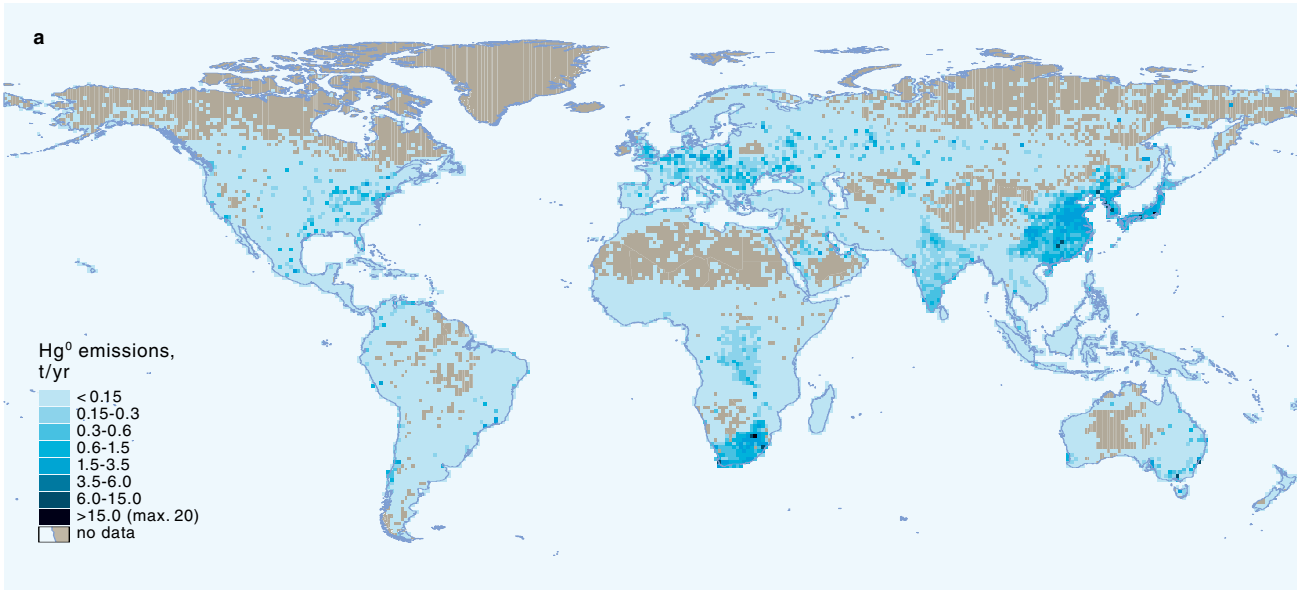
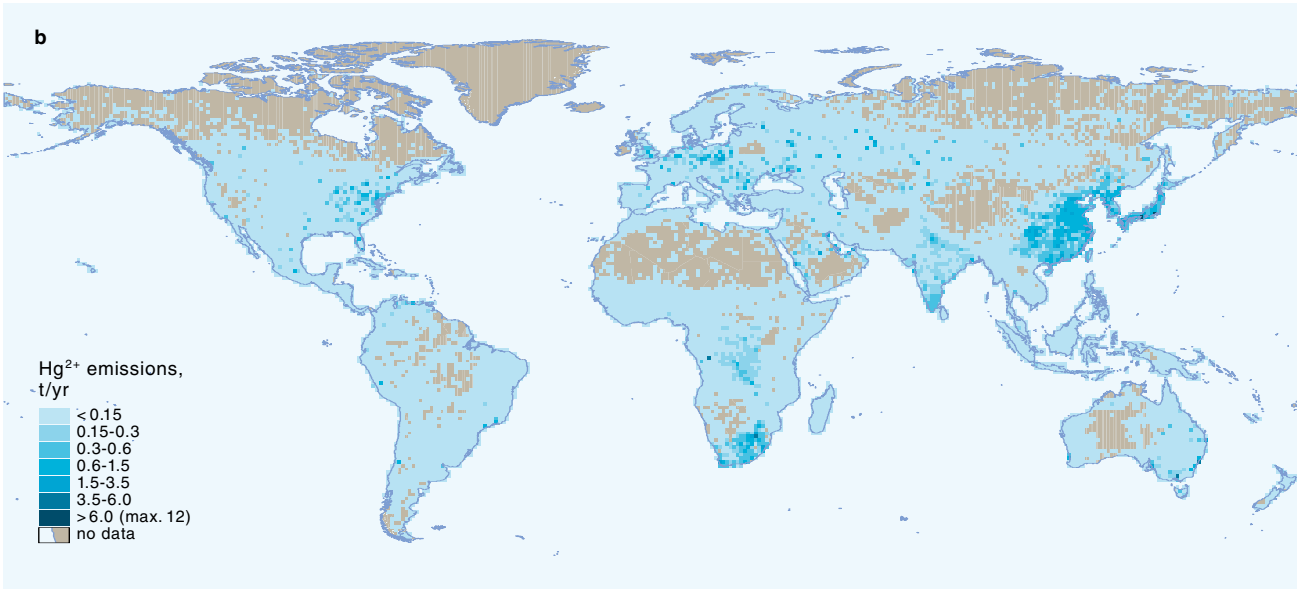


Figure 2·6. Global emissions in 1995 from anthropogenic sources of total mercury (Pacyna and Pacyna, 2002).





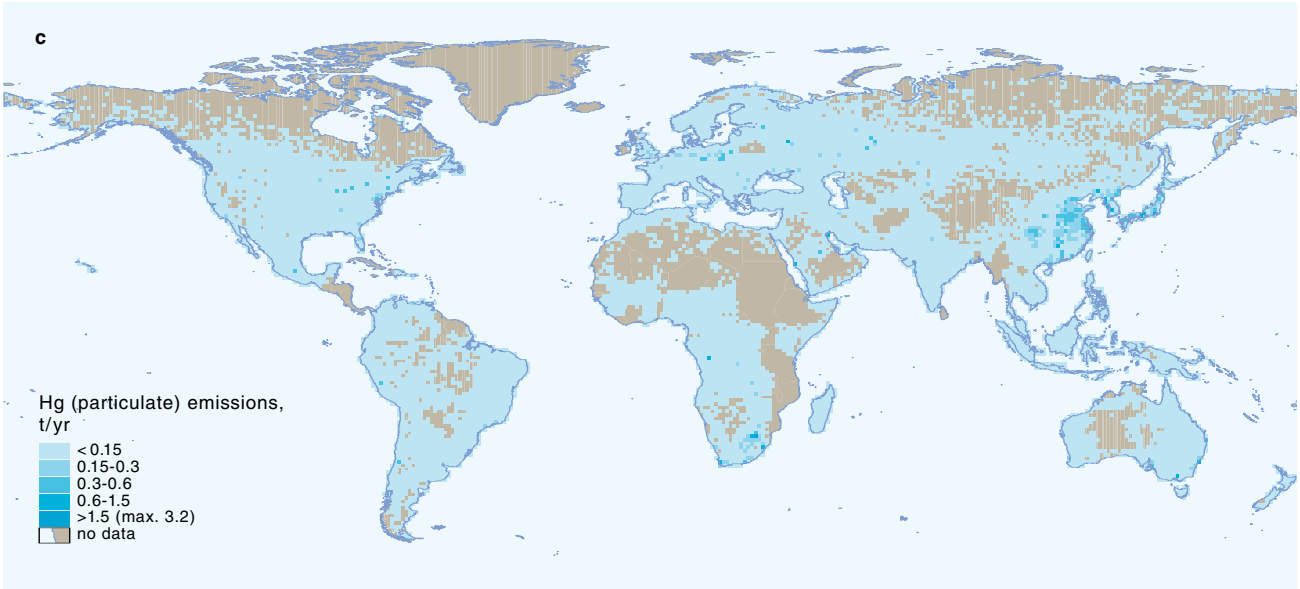


Figure 2·7. Global emissions in 1995 from anthropogenic sources of a) gaseous elemental mercury (Hg^0, GEM), b) gaseous divalent mercury (Hg^{2+}), and c) particulate mercury (Hg-p) (Pacyna and Pacyna, 2002).

Chapter 3

Transport Pathways and Processes Leading to Environmental Exposure

Steve Brooks, Steve Lindberg, Viacheslav Gordeev, Jesper Christensen, Alexey Gusev, Robie Macdonald, Suzanne Marcy, Keith Puckett, Oleg Travnikov, and Simon Wilson

Heavy metals reach and are re-distributed within the Arctic via atmospheric, freshwater, oceanic, ice, sediment, and biotic transport mechanisms. The first AMAP assessment (AMAP, 1998) included an extensive discussion on the main pathways of contaminants to, and within the Arctic. This chapter provides new information on transformation processes and additional information on transport pathways.

The chapter focuses initially on atmospheric pathways for mercury (Hg), lead (Pb), and cadmium (Cd). Section 3.1 describes new discoveries on transformation processes that lead to enhanced deposition of Hg in the Arctic, Section 3.2 provides updates on the results of atmospheric Pb emission reductions, including results from Pb atmospheric transport models using updated model parameters, and Section 3.3 describes new data on Cd transport that results in local distribution patterns similar to those for Hg, although resulting from different mechanisms of transport.

The chapter then focuses on other transport pathways. Section 3.4 presents different aspects of riverine transport, including water, particulates, and ice, and discusses the importance of estuarine marginal filters. Section 3.5 summarizes data on groundwater contaminant levels, and Section 3.6 discusses heavy metals in frozen ground. Marine pathways for Hg, Pb, and Cd are discussed in Section 3.7 and the chapter ends with a discussion in Section 3.8 on the potential impact of global change on marine transport pathways for heavy metals. Further details concerning pathways and global change are available in a companion report (Macdonald *et al.*, 2003).

3.1. Mercury in the Arctic: transport, transformation, and fate

Mercury is emitted to the atmosphere from sources around the world (see Chapter 2). Once emitted, Hg may remain in the atmosphere for up to two years as gaseous elemental mercury (GEM), allowing long-range global transport. As such, atmospheric transport represents a major pathway of Hg to the Arctic environment. Concentrations of Hg have been measured in the atmosphere in many Arctic locations as well as in the Antarctic (see Figure 3·1).

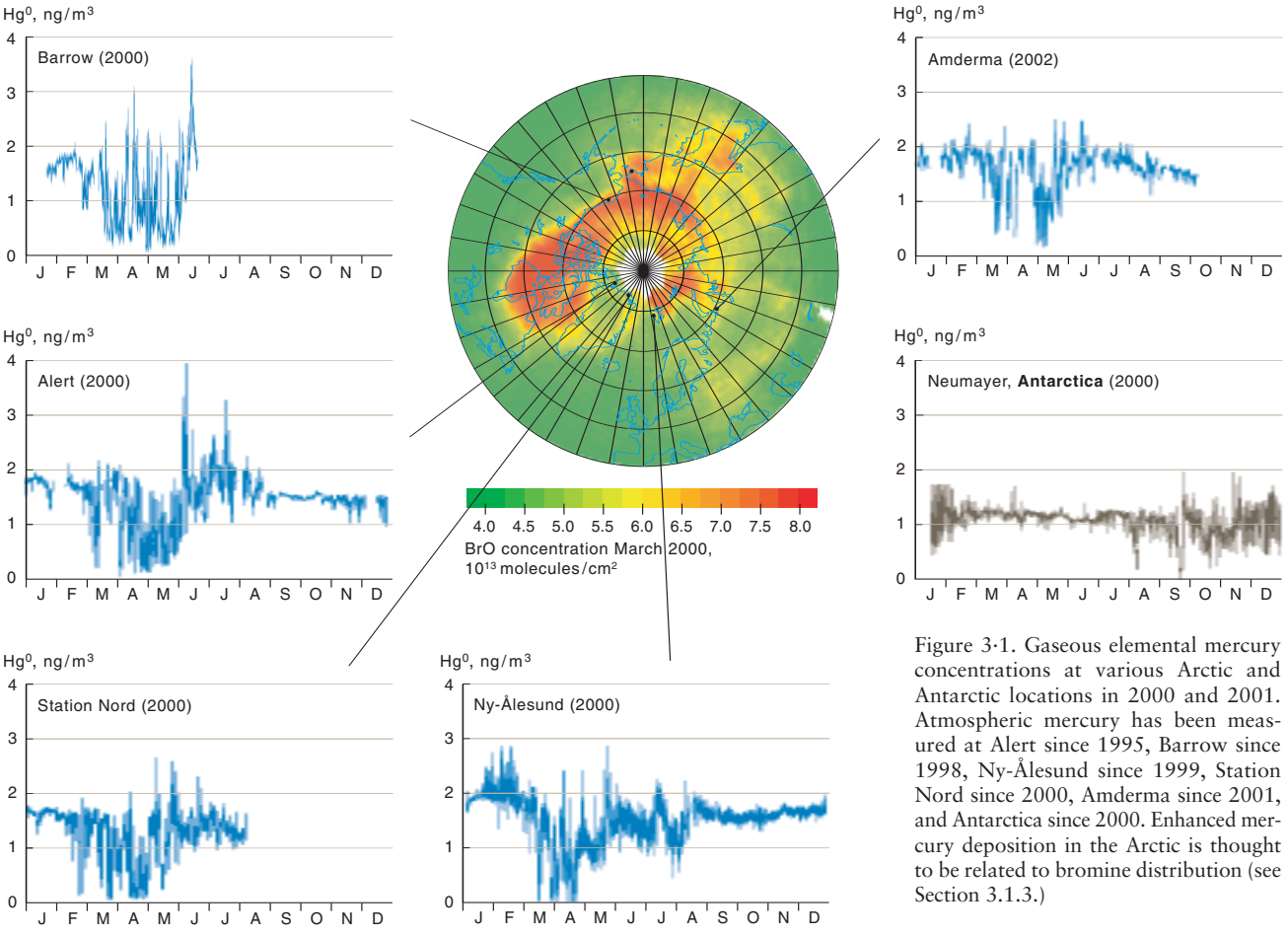


Figure 3·1. Gaseous elemental mercury concentrations at various Arctic and Antarctic locations in 2000 and 2001. Atmospheric mercury has been measured at Alert since 1995, Barrow since 1998, Ny-Ålesund since 1999, Station Nord since 2000, Amderma since 2001, and Antarctica since 2000. Enhanced mercury deposition in the Arctic is thought to be related to bromine distribution (see Section 3.1.3.)

Organic forms of Hg can result in toxicity at very low concentrations particularly in developing embryos and young. Concern about Hg in the Arctic environment has increased following indications that Hg levels may be increasing. For example, contemporary Hg levels in indigenous peoples are higher than historic levels (AMAP, 2003a). The same trend is emerging for some, but not all, animal species studied (see Chapter 5). Mercury levels in marine biota in some areas of the Arctic are increasing, in some cases to levels of concern (see Chapter 6). New research is clarifying how Hg is moving to and within the Arctic environment.

3.1.1. Atmospheric mercury transport: the GRAHM model

Pathways for atmospheric Hg transport to the Arctic have recently been clarified using transport models developed in Canada. The global/regional atmospheric heavy metals (GRAHM) model was developed by the Meteorological Service of Canada to improve understanding of Hg cycling in the Arctic, and to estimate its impact on global Hg budgets. The GRAHM model is a Eulerian multi-scale, comprehensive, horizontal and vertical high resolution Hg model. Grid points occur in three-dimensional space, and model resolution can change from high to low in different regions depending on the application (e.g., low resolution for global applications, high resolution for urban studies). All processes were parameterized using real rather than statistical data. The model includes gas and aqueous-phase Hg chemistry, multiple resistance-based dry deposition, vertical planetary boundary layer diffusion, cloud-chemical interactions using detailed cloud schemes, and wet deposition (Dastoor and Larocque, in press; Petersen *et al.*, 1998).

The model was run using global anthropogenic emissions for 1990 based on their availability from the Global Emissions Inventory Activity (GEIA) at the time of model runs. The model was integrated for 2.5 years, from June 1995 to December 1997, at a resolution of 1° latitude × 1° longitude. Initial conditions were set at no Hg in the atmosphere to represent the absence of anthropogenic contributions. Constant anthropogenic emissions were then introduced during the simulation. Atmospheric Hg reached steady-state conditions after 1.5 years. Model results were calculated for the last model year, 1997.

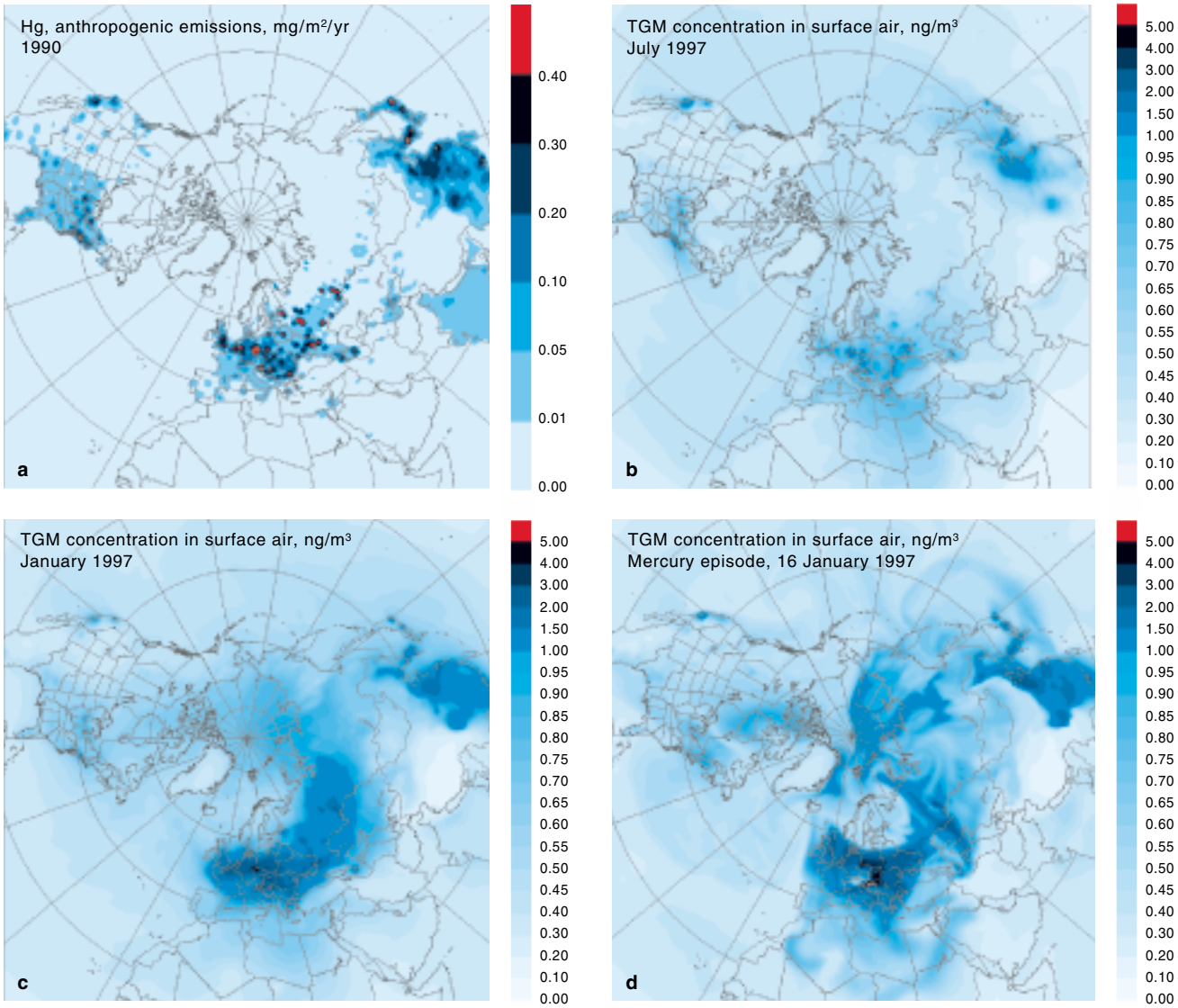


Figure 3·2. The distribution of total gaseous mercury (TGM) in the Northern Hemisphere as simulated by the GRAHM model for 1997. Anthropogenic emissions are based on 1990 data (Meteorological Service of Canada).

3.1.1.1. GRAHM model results: seasonal distribution

Total gaseous Hg emissions and surface air concentrations as simulated by the GRAHM model are shown in Figure 3·2 for the Northern Hemisphere. Three main regions of anthropogenic emissions are evident over Europe, China, and North America (Figure 3·2a). The distribution of total gaseous Hg varies seasonally such that average concentrations are lower in July (Figure 3·2b) and higher and more broadly distributed in January (Figure 3·2c). The model indicated that Arctic Hg pollution was significantly higher in winter from all global sources.

The model indicates that seasonal variations in total gaseous Hg concentration are driven by seasonal differences in patterns of atmospheric circulation, by seasonal cycles in boundary layer height, clouds, and precipitation, and by dry deposition. Figure 3·2d shows typical winter atmospheric circulation around a polar low and illustrates the transport of European pollution to North America via a descending air mass from the Arctic. Model simulations at 1°×1° resolution reveal that episodes of long-range intercontinental Hg exchange contribute significantly to the global background level of atmospheric Hg.

3.1.1.2. GRAHM model results: source regions

To estimate the relative Hg contribution from different global source regions the GRAHM model was integrated at 5°×5° resolution from June 1995 to December 1997 and four different model runs were conducted based on estimated 1990 global anthropogenic Hg emissions of 2144 t. Initial runs included all sources. In subsequent runs, one major source region was removed. The difference between integrations of the first simulation (all emissions), and those in which Europe, North America, or China and Japan were selectively removed, helped to identify the relative contribution from each source region and were used to calculate Hg budgets for Canada, the United States, Europe, and the Arctic (north of 60°N) (Figure 3·3).

There was an average of 29.4 t of GEM in the Canadian atmosphere (troposphere and stratosphere) at the end of the model runs. Of this, 39% was derived from Europe, 30% from China and Japan, 13% from North America, and 18% from all other global sources. Values for the United States were similar with 24.6 t derived from Europe (33%), China and Japan (31%), North America (16%), and other global sources (20%). In Europe, of the 37.3 t in the atmosphere, 52% came from Europe, 22% from China and Japan, 11% from North America, and 15% from all other sources. Finally, it is important to note that in the Arctic there were 73.3 t of Hg in the atmosphere, which was higher than in all other regions. Most GEM was derived from Europe (45%), then China and Japan (27%), North America (11%), and all other global sources (17%).

A general comparison of the global distribution of emissions with their contribution to atmospheric Hg burdens reveals that GEM in the atmosphere is well mixed due to its long atmospheric residence time. Differences between the relative contributions of source regions to different receptor regions arise mainly from: 1) the dominant large-scale atmospheric pathways, 2) circulation in the polar vortex, and 3) patterns of exchange between regions (e.g., a greater European contribution is observed to influence Canada than the United States owing to the influx of European pollution into North America, being mainly via the Arctic (AMAP, 1998)).

3.1.1.3. GRAHM model results: uncertainties

A number of uncertainties were apparent in the GRAHM model. Wet and dry deposition processes were affected by atmospheric non-linearity, making it difficult

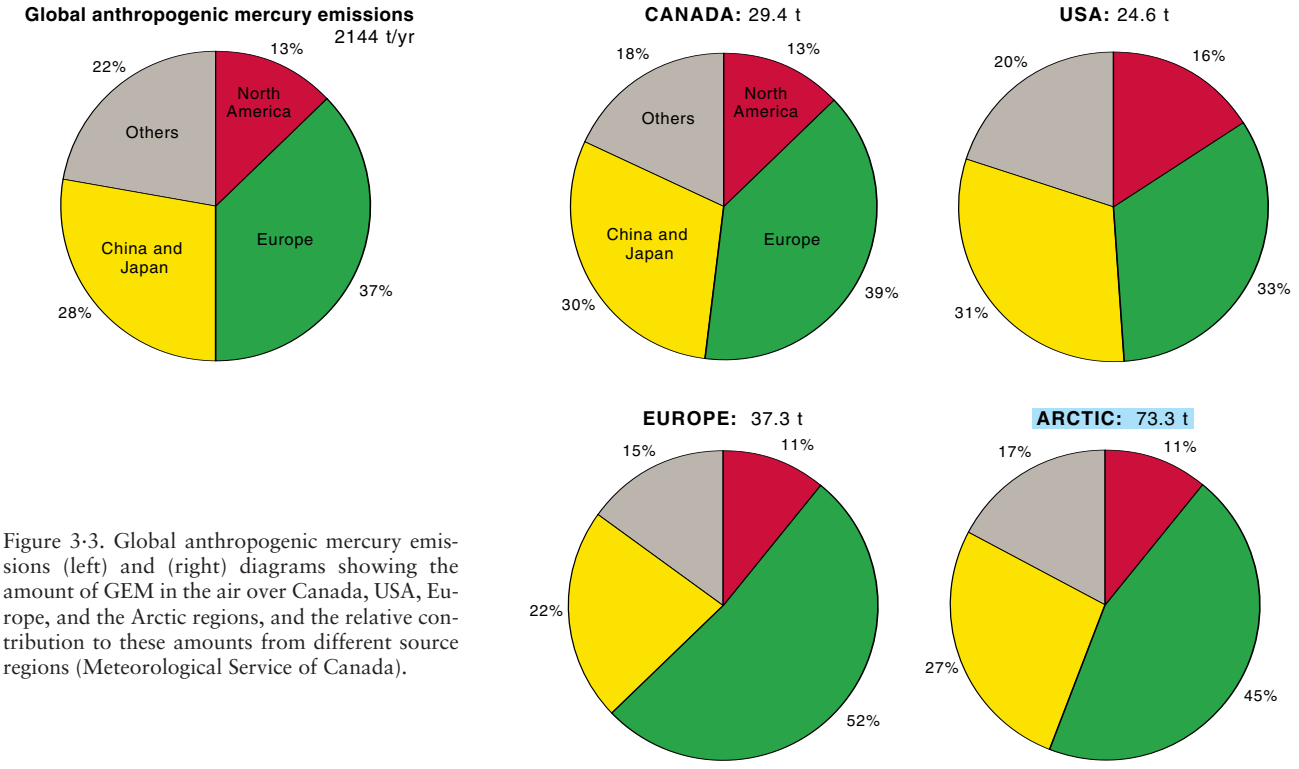


Figure 3·3. Global anthropogenic mercury emissions (left) and (right) diagrams showing the amount of GEM in the air over Canada, USA, Europe, and the Arctic regions, and the relative contribution to these amounts from different source regions (Meteorological Service of Canada).

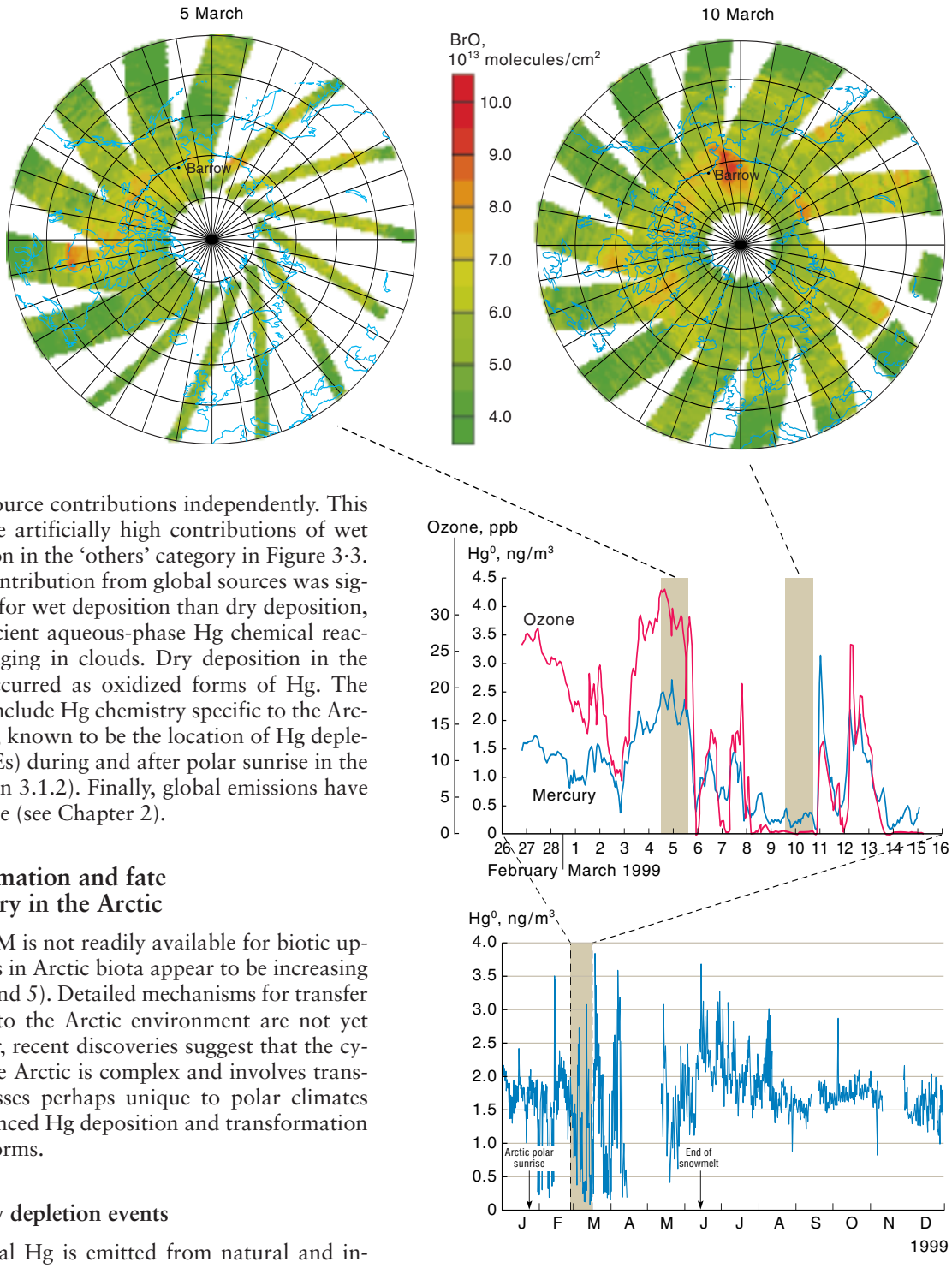


Figure 3·4. Trends in gaseous elemental mercury (GEM) at Barrow during 1998 to 1999, showing similarities between GEM and ozone depletion events, and the development of a broad area of elevated atmospheric BrO concentration near Barrow during a depletion event (Lindberg *et al.*, 2002).

to estimate the source contributions independently. This is reflected in the artificially high contributions of wet and dry deposition in the 'others' category in Figure 3·3. In general, the contribution from global sources was significantly higher for wet deposition than dry deposition, due to more efficient aqueous-phase Hg chemical reactions and scavenging in clouds. Dry deposition in the model mostly occurred as oxidized forms of Hg. The model does not include Hg chemistry specific to the Arctic and Antarctic, known to be the location of Hg depletion events (MDEs) during and after polar sunrise in the spring (see Section 3.1.2). Finally, global emissions have changed over time (see Chapter 2).

3.1.2. Transformation and fate of mercury in the Arctic

Atmospheric GEM is not readily available for biotic uptake yet Hg levels in Arctic biota appear to be increasing (see Chapters 4 and 5). Detailed mechanisms for transfer of global GEM to the Arctic environment are not yet known. However, recent discoveries suggest that the cycling of Hg in the Arctic is complex and involves transformation processes perhaps unique to polar climates that lead to enhanced Hg deposition and transformation to bioavailable forms.

3.1.2.1. Mercury depletion events

Gaseous elemental Hg is emitted from natural and industrial sources and is known to travel long distances around the globe. However, deposition of GEM in the Arctic requires a chemical (oxidation) change, and unlike the concept of global fugacity used to explain condensation and accumulation of persistent organic pollutants (POPs) in Arctic regions (Wania and Mackay, 1993) GEM does not 'condense out' even at –50 °C. However, other forms of airborne Hg, especially oxidized reactive (divalent) gaseous mercury (RGM), may partition from gas to solid phases at low temperatures (Lin and Pehkonen, 1999). While oxidized RGM species are known to be emitted from industrial sources (Fahlke and Bursik, 1995) there are few anthropogenic sources within the Arctic. In addition, RGM compounds have a significantly shorter atmospheric lifespan than GEM (Bullock, 2000; Lindberg and Stratton, 1998) and are not subject to the same global transport as GEM. Although RGM compounds represent only a small percentage of the overall gaseous Hg in typical ambient air, RGM dry deposition velocities (i.e., dry deposition/air concentration) and scavenging ratios (i.e., concentration in rain/concentration in air) exceed those of GEM by more than an order of magnitude (Lindberg and Stratton, 1998). Recent discoveries that RGM exists in the Arctic in the absence of local anthropogenic sources led

to work on the link between polar sunrise and the formation of RGM, fine particulate Hg (FPM), and Hg accumulation in the snowpack.

Ground level ozone and GEM depletions at Alert in the Canadian High Arctic (~82°N) gave the first indication that conditions may exist in the Arctic following polar sunrise that promote depletion of atmospheric GEM (Schroeder *et al.*, 1998). Since then, episodic springtime MDEs have been measured in many Arctic locations; Alert (Canada), Barrow (Alaska), Ny-Ålesund (Svalbard), and Station Nord (Greenland) (Berg *et al.*, 2001; Lindberg *et al.*, 2001; Lu *et al.*, 2001; Schroeder *et al.*, 1998; Skov *et al.*, 2001). At Arctic sites, MDEs begin to appear within a few days of local polar sunrise, are influenced by increasing solar radiation (Figure 3·4, lower panel), and persist until snowmelt (Lindberg *et al.*, 2001). During MDEs, GEM is positively correlated with ozone (Berg *et al.*, 2001; Lindberg *et. al.*, 2002, Schroeder *et al.*, 1998; Skov *et al.*, 2001). At Barrow, for example, $r^2 = 0.76$ (Figure 3·4, middle panel). Unlike ozone, however, atmospheric Hg is neither created nor destroyed, but readily undergoes changes in oxidation state. Depletion of total gaseous Hg from the air mass implies accumulation of Hg elsewhere. Data from Barrow help clarify these processes in the Arctic.

3.1.2.2. **Mercury depletion events in Barrow and Alert**

Recent studies at Barrow and Alert indicate that the Hg deposition mechanism is clearly linked with: 1) the formation of RGM in the near surface air (up to 1.0 ng/m^3), 2) the formation of FPM in air (up to 0.3 ng/m^3), and 3) the deposition to and the accumulation of Hg in the snowpack (>90 ng/L prior to melt; Lindberg *et al.*, 2002).

To illustrate the relationships between RGM, FPM, and solar radiation, Figure 3·5 shows the diel cycle of surface air Hg species and incident solar ultraviolet B (UV-B) radiation for a 24-hour period at Barrow. On 22 March 2001 the levels of GEM were highest in the absence of solar radiation. In the absence of UV-B radiation, RGM and FPM were low and negatively correlated with GEM. Coincident with sunrise (06:00 hours) and increasing UV-B radiation levels, atmospheric GEM concentrations dropped while those for FPM and RGM rose. Fine particulate Hg peaked at 0.027 ng/m^3 at sunrise then decreased to near the detection limit after sunrise. Following sunrise, RGM concentrations increased from near zero to about 0.5 ng/m^3 with simultaneous GEM depletion from normal background levels (~1.7 ng/m^3) to 0.6 ng/m^3. After solar noon, GEM concentrations again rose with mixing from the overlying air mass and RGM concentrations fell, thought to result from deposition onto the snowpack.

During 24-hour sunlight (after ~May 10), there appeared to be no further direct production of FPM. Reactive (divalent) gaseous mercury was primarily generated in the conversion process, supporting a strong role for solar radiation. However, some RGM appears to have been adsorbed onto the existing aerosol. After 24-hour sunlight, RGM and FPM became positively correlated, although RGM greatly exceeded FPM. During peak RGM production, it was not uncommon to see RGM concentrations of >0.5 ng/m^3 and at times as high as 1.0 ng/m^3. By comparison, rural sites in the eastern United States show mean RGM concentrations of 0.06 to 0.10 ng/m^3 (Lindberg and Stratton, 1998).

These observations contribute to mounting evidence that GEM transported long distances to the Arctic, is transformed to RGM and then deposited onto the snowpack. Mercury depletion events coupled with enhanced deposition of RGM are thought to lead to greater Hg accumulation in the Arctic environment.

3.1.2.3. **Enhanced deposition: the DEHM model**

Until recently, enhanced deposition of Hg in the Arctic due to MDEs had not been incorporated into atmospheric models of global Hg transport. To better estimate Hg deposition in the Arctic, the Danish Eulerian Hemispheric Model (DEHM), a three-dimensional air pollution model,

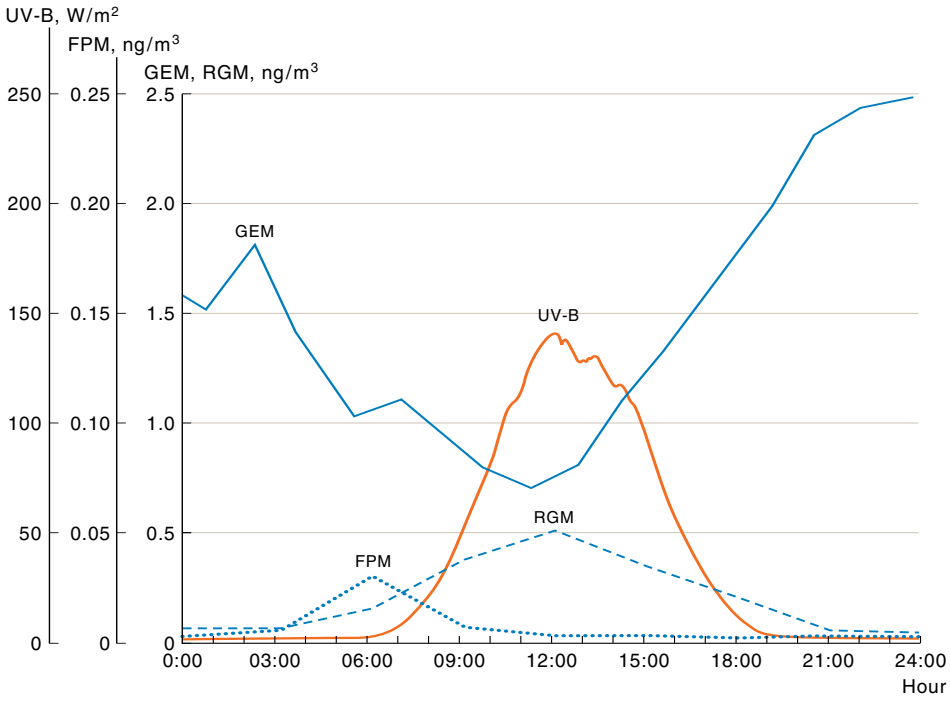


Figure 3·5. Diel cycle of GEM, RGM, FPM, and incident solar UV-B radiation at Barrow, 22 March 2001 (Lindberg *et al.*, 2002).

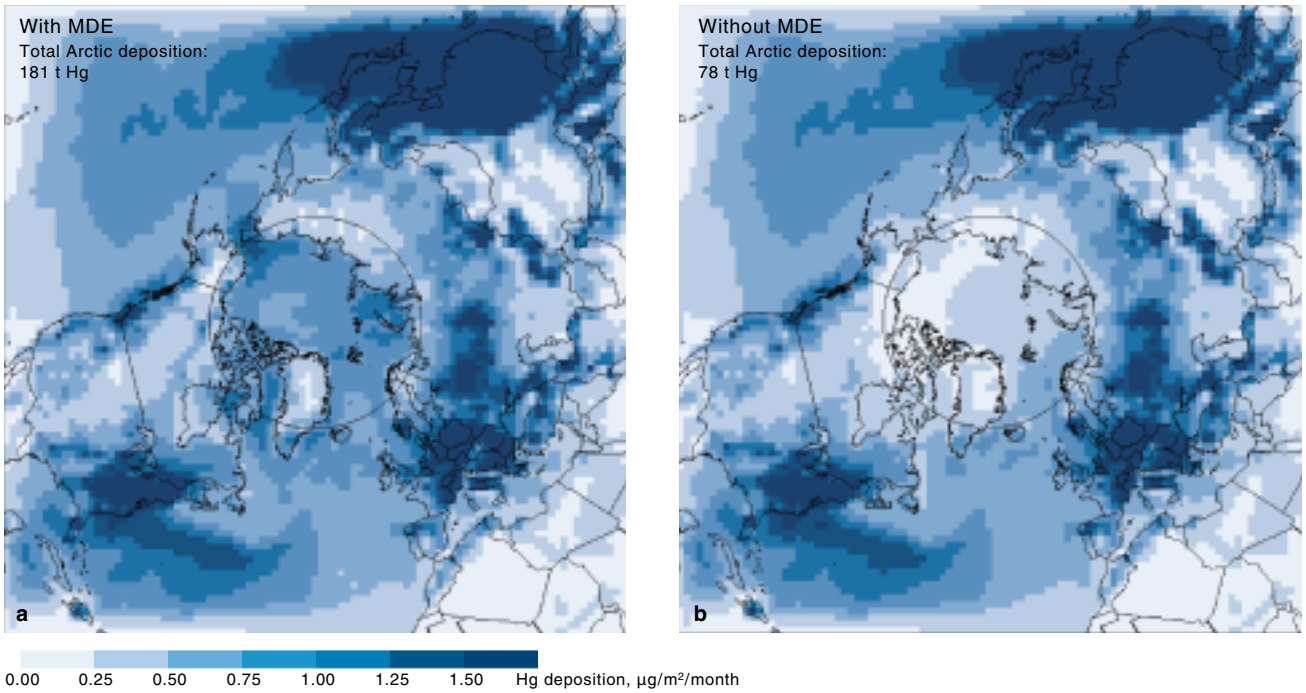


Figure 3·6. DEHM model estimates of total mercury deposition; a) with mercury depletion, and b) without mercury depletion (J. Christensen, unpubl. data, 2002).

was modified to incorporate the speciation of Hg emissions as well as newly discovered chemical transformations at polar sunrise involving GEM, RGM, and FPM.

An earlier version of the DEHM model was used in the first AMAP assessment to model sulfur transport (Kämäri *et al.*, 1998) and is well described elsewhere (e.g., Barrie *et al.*, 2001; Christensen, 1997, 1999). The new model combines the original DEHM model with a meteorological model based on the Pennsylvania State University National Center for Atmospheric Research Mesoscale Model version 5 (Grell *et al.*, 1995). In the present version thirteen Hg species are included; three in the gas-phase (Hg, HgO, and $HgCl_2$), nine in the aqueous-phase, and one in the particulate phase. Emissions of anthropogenic Hg are based on the most recent global mercury emissions inventory (Pacyna and Pacyna, 2002). This inventory, for 1995, includes emissions of GEM, RGM and particulate Hg, and has been distributed to a 1°×1° grid.

Re-emissions of Hg from land and ocean are not incorporated into the model. Since re-emissions are known to occur (see Section 3.1.3.4), a background GEM concentration of 1.5 ng/m^3 was used as the initial concentration and boundary conditions. Assumptions about the chemistry are based on work by GKSS Forschungszentrum (see Petersen *et al.*, 1998). During polar sunrise, the oxidation rate affecting the lifetime of GEM is assumed to increase by a factor of 2000 (decreasing the lifetime from approximately one year to four hours) inside the boundary layer over sea ice during daylight. This ceases when surface temperatures exceed –4°C (S. Brooks, pers. comm., 2001). Dry deposition velocities for RGM were estimated based on the resistance method, using a surface resistance similar to HNO_3 (Bullock and Brehme, 2002). Wet deposition of RGM and FPM was estimated using a simple scavenging coefficient formulation with different in-cloud and below-cloud scavenging coefficients (Christensen, 1997).

The DEHM model run was based on 1995 emissions data (Pacyna and Pacyna, 2002) and atmospheric data from October 1998 to December 2000. After a three-month start-up, data were generated for 1999 and 2000 in two separate runs; one with chemistry for MDEs the other without (Figure 3·6).

Including MDE chemistry in the model more than doubled the amount of Hg predicted to be deposited within the area to the north of the Arctic Circle (181 t/yr, Figure 3·6 a) relative to model results without (78 t/yr, Figure 3·6 b). This suggests that Arctic MDEs have the effect of making the Arctic Basin a regional sink for Hg.

3.1.3. Environmental accumulation of mercury linked to halogen chemistry

Processes leading to the formation of RGM and MDEs are not clear but several associations have been found. Although the chemistry associated with MDEs requires further investigation, current information suggests that bromine (Br) plays a central role in the halogen chemistry of MDEs (see Section 3.1.3.1). Satellite data on the distribution of reactive halogens near Barrow (Richter *et al.*, 1998; see also Figure 3·4 top) indicate that MDEs and ozone depletion events are most often associated with elevated near-surface photolyzable bromine (BrO) (Steffen *et al.*, 2003; Honninger and Platt, 2002; Lindberg *et al.*, 2002; Lu *et al.*, 2001). Several Br and chlorine (Cl) species have the potential to oxidize GEM to RGM (e.g., BrO, Br, Cl, or BrCl may react with GEM to form HgO, $HgBr_2$, or $HgCl_2$; Calvert and Lindberg, 2003; Lindberg *et al.*, 2002). Destructive reactions with ozone are now thought to involve Br and Cl coming from marine open-water sources (e.g., Fan and Jacob, 1992), perhaps in both aerosol and gaseous phases. In addition, many of the halogen compounds formed by reactive Br show a strong diel pattern and enhancement following polar sunrise, reinforcing the importance of solar radiation and photochemical reactions in these processes.

3.1.3.1. Sources and distribution of bromine

Bromine is a common, naturally occurring substance in the marine environment. Photolyzable Br maps (Figure 3·7) suggest that the sources of BrO in the Arctic are sea salts associated with areas of annual (first-year) ice. Sea salts include Cl and Br in large quantities at a Cl : Br molar ratio of roughly 650 : 1.

Photolyzable Br in the Arctic comes primarily from the areas of seasonal ice production which support prime sources for BrO release to the atmosphere, including ice with a relatively salt-rich frozen surface (Barrie and Platt, 1997) and open water or water with light ice cover where pelagic- and ice-algal communities produce halogen-containing gases (i.e., bromoform) during primary production (Sturges *et al.*, 1992). During ice formation, seawater loses approximately 90% of its salt (Br and Cl) content. Ice formed each season, known as annual, or first-year ice, has a salt content of roughly 10 g/kg^3, when first formed, compared to 30 to 35 g/kg^3 in seawater (Tucker *et al.*, 1990; Wadhams and Wilkinson, 1996). As ice ages, the salt content decreases to <1 g/kg^3 at the surface of multi-year ice (in the multi-year icepack, new ice is formed in winter at the bottom of the ice and this migrates slowly upward as ice goes through seasonal freeze-melt cycles eventually to melt at the surface; the definitive test for distinguishing multi-year and annual ice is to measure salt concentration at the ice surface). Annual ice is thinner, transmits sunlight (thus allowing primary production) and mainly occurs in areas with a higher percentage of open water in late summer (i.e., leads, polynyas, marginal seas). 'Frost flowers', which form on the surface of rapidly refreezing leads, were measured roughly 200 km north of Barrow in spring 2003. On average, the salt (ion) concentrations in the newly formed flowers were three to four times higher than the salt (ion) content of the seawater. This is due to the excretion of salt from the upper surface of the rapidly forming sea ice. The salt (ion) content of the frost flowers then decreased rapidly, with a half-life of roughly 48 hours, with an assumed loss to the near surface air (S. Brooks, unpubl. data, 2001). Open-water areas are thought to be responsible for about half the Arctic annual sea ice formed. Owing to a higher salt content and thinness (relative to multi-year ice), annual ice melts more rapidly in spring. In contrast, multi-year ice, which mainly occurs in the interior pack, contains relatively little open-water area (~1%) and blocks sunlight (due to thickness and the accumulation of snow and wind blown sediment) and therefore supports little primary productivity.

3.1.3.2. Bromine transport and mercury accumulation patterns

Once released to the atmosphere, Br remains concentrated in the near surface air due to lack of vertical convection. Reactive Br compounds forming under solar radiation (UV-B) conditions can be moved or advected downwind. Horizontal advection to inland shelf regions is controlled by prevailing winds, and is effectively dammed by mountain ranges (e.g., Brooks, Anadyr, Rockies) and the location of the polar front. The distances Br moves downwind can be quite significant at night but, in general, atmospheric total Br concentrations slowly decrease with distance from the source. However, air over Greenland (Kalaallit Nunaat) and Ellesmere Island is relatively free of BrO enhancements as predominant katabatic (outward flowing) winds off the ice caps block significant inland advection. Coastal locations, such as Station Nord and Alert, remain affected by the local marine environment and experience episodic BrO enhancements and MDEs. BrO enhancements are absent over multi-year ice (e.g., within the Canadian Basin) where open-water area, sea-ice formation, and ice surface salt concentration are all low (see Figure 3·7). Total Br enhancements in the near surface air at Barrow and Alert correlate well with observed MDEs (Barrie and Platt, 1997; Berg *et al.*, 1983; Lindberg *et al.*, 2002) (see Figures 3·4 and 3·7). This is consistent with the demonstration by Lu *et al.* (2001) that

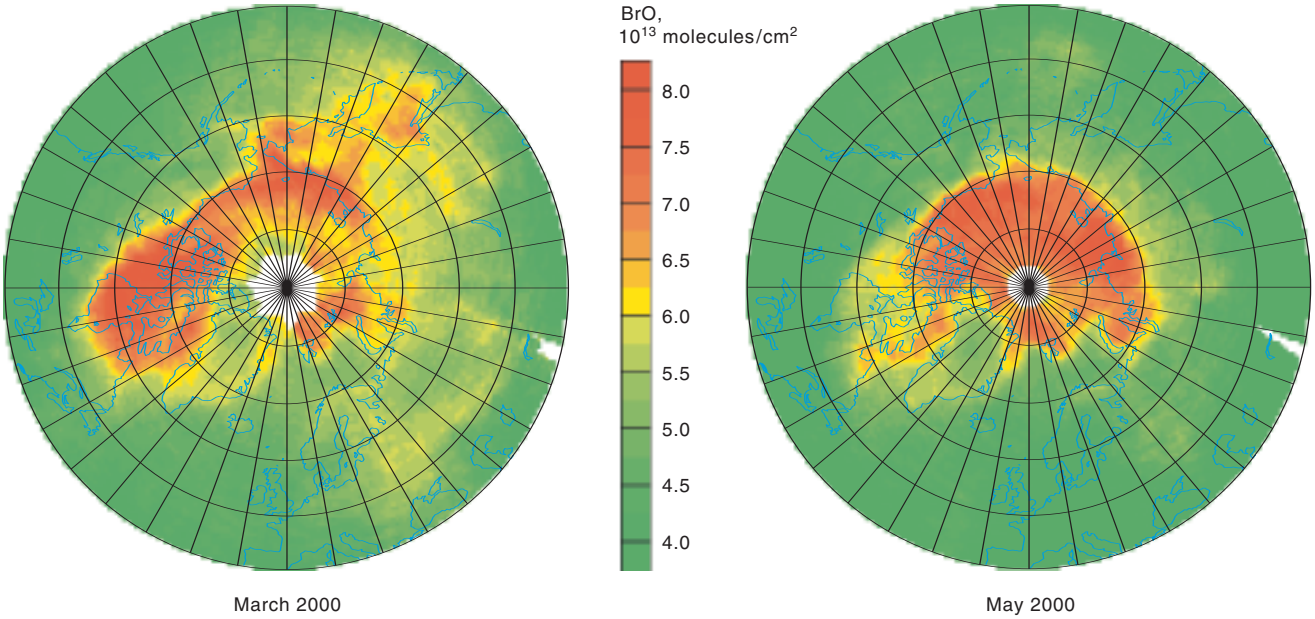


Figure 3·7. Global Ozone Monitoring Experiment (GOME) satellite-derived BrO concentration averages for March and May 2000 (A. Richter pers. comm., 2001).

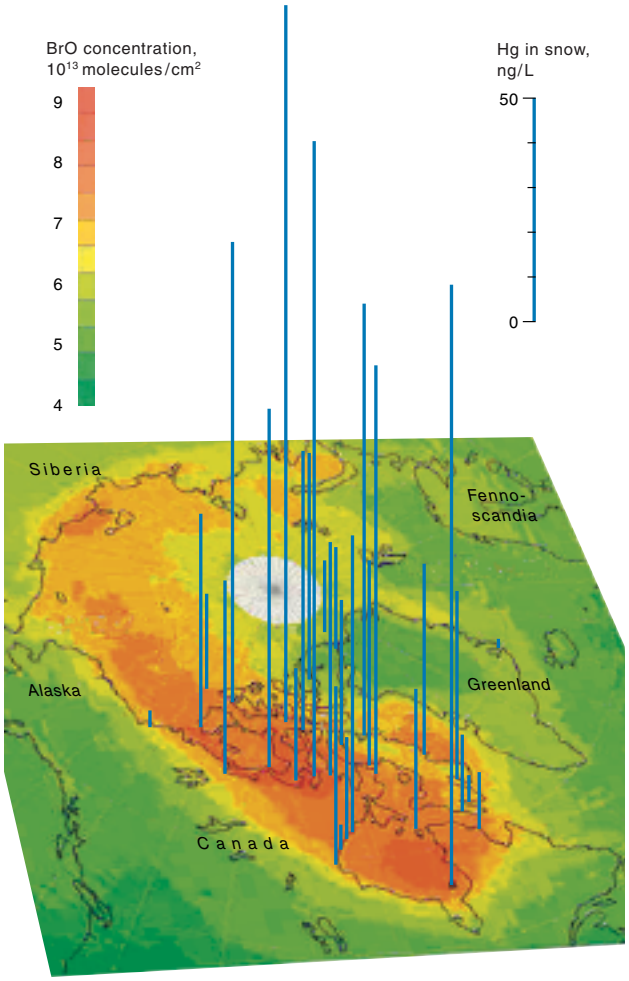


Figure 3·8. Global Ozone Monitoring Experiment (GOME) satellite-derived BrO concentration averages for March to May 1997 and 1998 compared to total mercury concentrations in snow (Lu *et al.*, 2001).

areas of high atmospheric BrO concentration in the Canadian Archipelago correlate well with areas of high total Hg concentration in snow (Figure 3·8). The decrease in conversion of GEM to RGM with increasing distance from marine source regions highlights the importance of marine sources of Br (see the BrO inland gradient in Figure 3·7), and is consistent with earlier observations showing Hg levels in the Arctic Alaskan tundra vegetation and snowpack to be inversely related to distance from the nearest coast (Landers *et al.*, 1995; Snyder-Conn *et al.*, 1997).

3.1.3.3. Proposed halogen chemistry for mercury depletion events

Based on the strong diurnal cycle of measured RGM and its positive correlation with the intensity of solar radiation, RGM is thought to form through rapid, *in situ* oxidation of GEM during MDEs (Calvert and Lindberg, 2003; Lindberg *et al.*, 2002). RGM production is attributed to the same photochemically active Br species involved in surface ozone destruction during Arctic spring, suggesting that the overall process is heterogeneous (i.e., involving multiple chemical phases) and dominated by bromine chemistry (Dickerson *et al.*, 1999; Fan and Jacob, 1992; Vogt *et al.*, 1996). Bromine is oxidized by hydroxyl radicals, and forms hypobromous acid (HOBr), that produces Br and Cl radicals through a heterogeneous photochemical mechanism involving wet sea-salt aerosol. The halogen radicals (Br/Cl) and halogen oxide radicals (BrO/ClO) produced from the destruction of ozone both show strong diurnal patterns with solar radiation and serve as the primary oxidants that produce RGM. These mechanisms explain the strong correlation between GEM and ozone concentrations during MDEs. Some possible physicochemical pathways were proposed

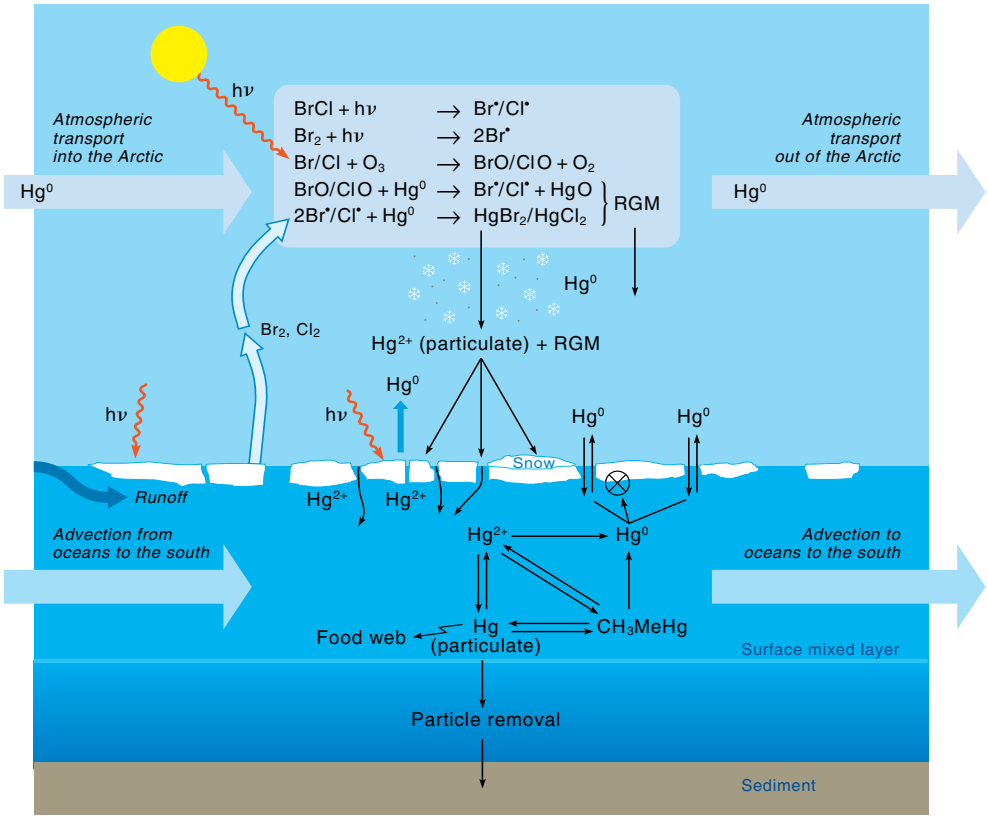
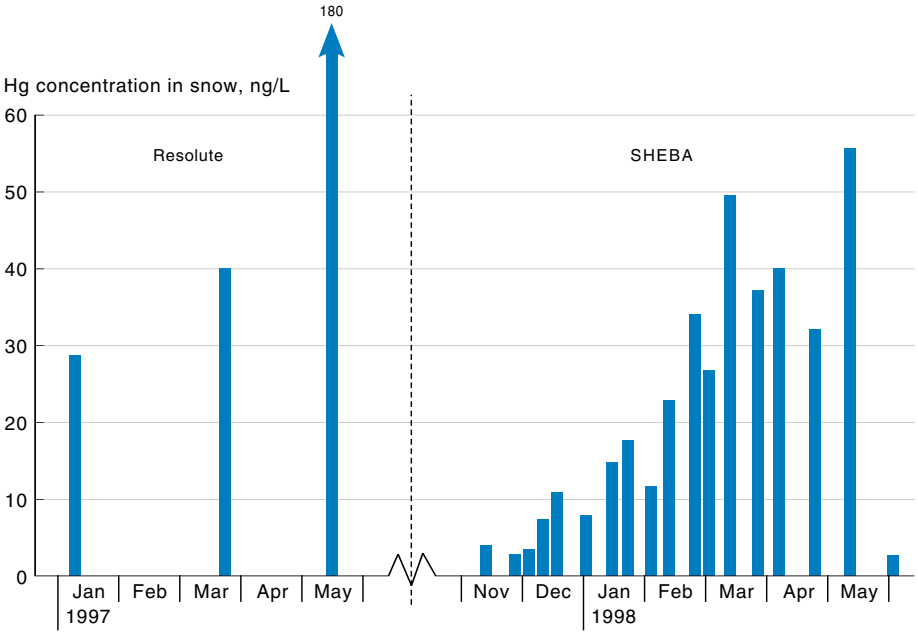


Figure 3·9. Conceptual diagram of proposed Hg^0 (GEM) oxidation reaction sequences in the Arctic at Barrow showing several possible pathways and products. These pathways are the most feasible given current observations (Macdonald *et al.*, 2003; chemistry adapted from Lindberg *et al.*, 2002).



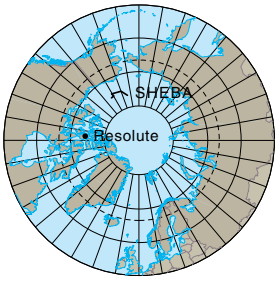


Figure 3·10. Seasonal variation in mercury levels in the surface snowpack of the Canadian Basin during winter 1997 and spring 1998 (Lu *et al.*, 2001).

by Lindberg *et al.* (2002) and later confirmed by detailed modeling by Calvert and Lindberg (2003); some are represented in Figure 3·9. Where BrO enhancements occur, they are associated with MDEs and the accumulation of Hg in surface snow.

3.1.3.4. Photolyzable Br and reactive (divalent) gaseous mercury linked to the rise in total and bioavailable mercury in the snowpack

The first detailed studies of Hg in the snowpack, completed in the Canadian Basin, indicate that Hg levels increase during spring (Figure 3·10). Later measurements of total Hg by Lindberg *et al.* (2002) at Barrow provided supporting evidence of elevated accumulation rates as indicated by a large and rapid increase in Hg levels in surface snow following polar sunrise (Figure 3·11). Total Hg in the surface snow increased from about 1 ng/L in January before sunrise to >90 ng/L in May (Lindberg *et al.*, 2002). In comparison, the total Hg content of snow in the northeastern United States is about 1 to 10 ng/L. Total Hg concentrations in the sea-ice snowpack north of Alaska show that the snowpack predominately downwind of the major leads and polynyas off Point Barrow had late spring Hg concentrations around an order of magnitude more than locations predominately upwind of the open-water areas

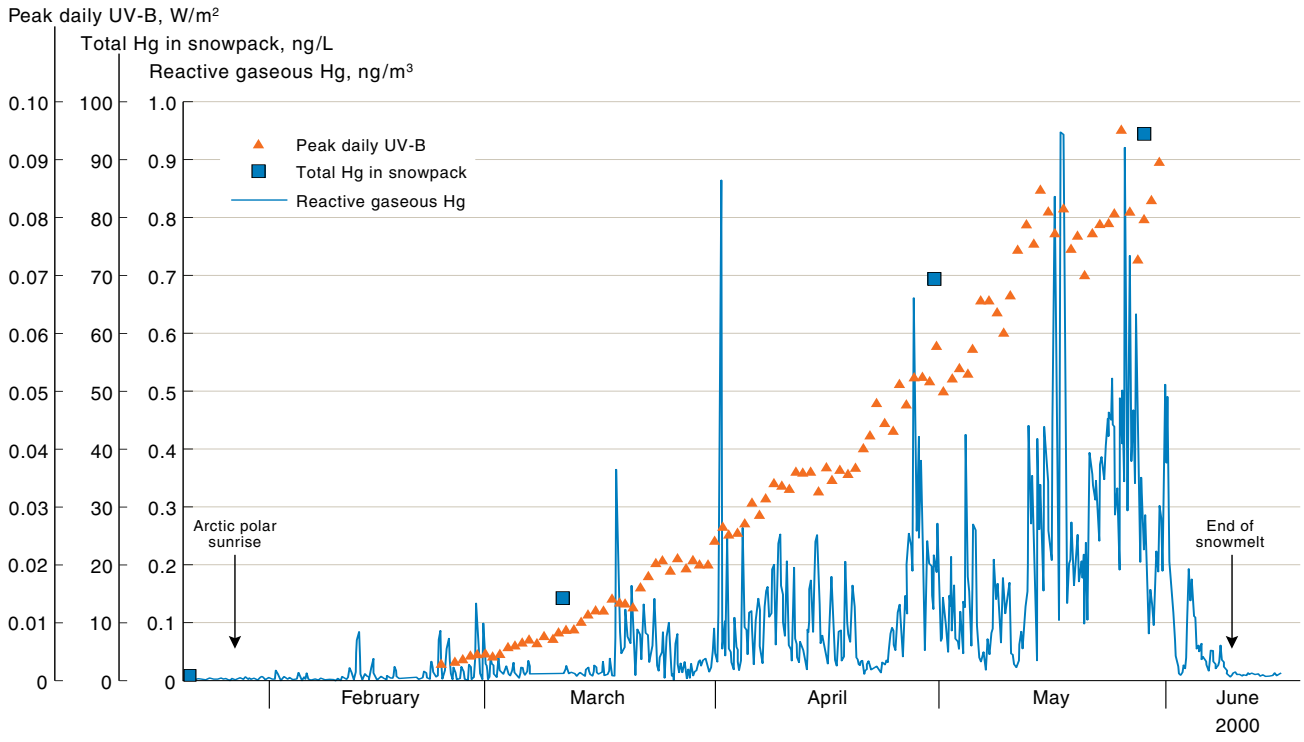


Figure 3·11. Trends in reactive (divalent) gaseous mercury (RGM) at Barrow during spring 2000 showing RGM production starting after sunrise and ending at snowmelt. Trends in total mercury concentration in the surface snowpack and mean daily UV-B radiation are also shown (Lindberg *et al.*, 2002).

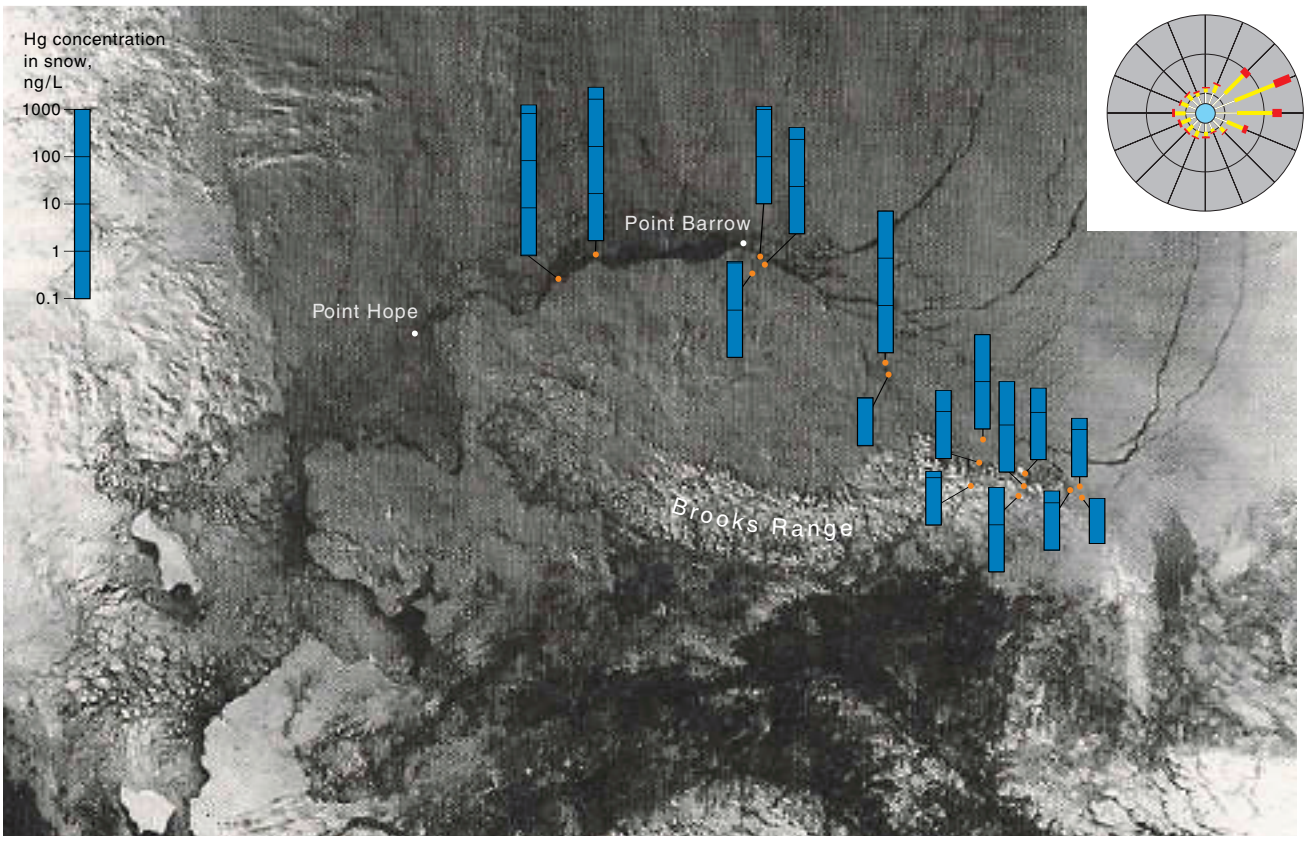


Figure 3·12. NOAA9 satellite image from 9 March 2000 showing polynyas and leads concentrated around Point Barrow to the west, with extensions northward and eastward. Mercury concentrations in snow along the north Alaskan coast around Barrow (Garbarino *et al.*, 2002) are consistent with mercury accumulation downwind of polynyas and leads. The wind rose shows the frequency (%) of wind speeds (blue: calm; white bar: 0.5-5 m/s; yellow bar: 5-10 m/s; red bar: >10m/s); the outer ring indicates a frequency of 20%.

(Figure 3·12). Snowpack Hg levels decreased with distance east and south of Barrow, areas which are upwind and well away from the normally open leads and polynyas. As such, enhanced concentrations of Hg in snow may be linked to high halogen (Br and Cl) levels downwind of open-water leads and polynyas.

Aircraft profile measurements of GEM and RGM concentrations, and flux measurements of the surface-atmosphere exchange of RGM, confirmed the conversion of GEM to RGM under solar radiation within 200 m of the surface where sea-salt aerosols, water vapor, and Br/Cl compounds co-exist. Direct flux rate measurements confirmed that RGM deposits within minutes of formation, where it enriches the most readily available surface, the snowpack (Lindberg *et al.*, 2002).

By itself RGM is not particularly harmful to biota and, when deposited onto the snowpack, causes no immediate harm. However, a significant percentage of Hg(II) originally deposited onto the snowpack as RGM is biologically available to bacteria. This was determined using the *mer-lux* bioreporter that uses genetically engineered bacteria that produce light when bioavailable Hg enters their cells (Selifonova *et al.*, 1993). Bioavailable Hg was below detection limits in the Barrow snowpack on 17 January 2000, five days prior to polar sunrise. The same snowpack had about 9.0 ng/L bioavailable HgII in samples collected on 10 May 2000 (Scott, 2001), the highest value ever measured by the Canadian Fresh Water Institute to date. In comparison, bioavailable Hg was <0.5 ng/L in snow in northwest Ontario (Scott, 2001).

The implications of elevated levels of bioavailable Hg(II) entering the ecosystem are considerable. Although there is considerable uncertainty, an estimated 24 to 36% of deposited Hg in the snowpack and in melting snow and runoff is photoreduced to GEM which can then be lost to the atmosphere by evasion (Schroeder *et al.*, 2003), leading to a moderate summer increase in GEM (Figure 3·1). At Barrow, approximately 20 to 40% of Hg in snow is re-emitted and the remaining 60 to 80% remains at snowmelt (S. Lindberg and S. Brooks, unpubl. data, 2003). Elevated total and bioavailable Hg levels in the snowpack at snowmelt are particularly important to the Arctic Hg cycle because they occur when the seasonal biological activity begins. Once underway, snowmelt at Barrow proceeds rapidly under 24-hr daylight, and large pulses of Hg are delivered to the tundra in percolation and runoff with measured concentrations of roughly 30 ng/L in runoff waters, some of which becomes bioavailable (Lindberg *et al.*, 2002). When bioavailable Hg enters the microbial cell of methylating bacteria, the cell can provide a substrate for methylation of Hg to the highly toxic form – methylmercury (MeHg) – which biomagnifies in aquatic food chains (Kelly *et al.*, 2003). The accumulation of Hg in the Arctic from atmospheric transport and deposition to the snowpack in a bioavailable form is potentially a key step in the pathway for Hg accumulation in Arctic biota.

A pulse of bioavailable Hg entering the Arctic environment in spring during the onset of the breeding season has potentially serious implications for Arctic species. While the effects of exposure to Hg, particularly

during early developmental stages, is known, and tissue concentrations of Hg in some Arctic biota are reaching levels of concern (see Chapter 6), currently there is insufficient information to understand the processes that lead to biotic exposure, or to determine the potential for effects from the enhanced exposure thought to be occurring in the Arctic due to MDEs and snowpack enrichment. However, it is accepted that the Arctic has become an important sink for Hg in the global cycle and that global change is likely to exacerbate risk in the Arctic despite recent improvements in global emission controls (Lindberg *et al.*, 2002).

3.1.4. Mercury and global change

Recent observations indicate that the Arctic has undergone rapid change over the last 30 to 40 years, including decreasing multi-year ice extent with replacement by annual ice cover, later snowfall and earlier snowmelt, increasing ocean temperature, and increasing atmospheric circulation and temperature (Dickson, 1999). Given that the Hg cycle in the Arctic is directly linked to temperature, ice extent, algal production, and other variables influenced by climate, it is important to consider how these changes may change the transport, transformation and deposition of Hg. Questions remain concerning whether Arctic MDEs are recent in origin, perhaps resulting from increased atmospheric transport and production of photooxidants (Br/Cl) in the Arctic, coupled with an increased proportion of annual ice coverage. The future severity and extent of MDEs may well be affected by global change (e.g., alteration of atmospheric transport, ice coverage, primary productivity; Macdonald *et al.*, 2003) as well as potential change in global Hg emissions (see Chapter 2).

3.1.4.1. Influence of multi-year and annual ice

In the Arctic, multi-year permanent-pack sea ice currently covers about eight million km^2. However, multi-year ice in the central Arctic Ocean, as measured by U.S. Navy submarines during the 1980s and 1990s, has shown a remarkable 43% reduction in thickness over two decades (Rothrock *et al.*, 1999). At this rate the Arctic Ocean may become seasonally sea-ice free within 30 to 40 years. A decrease in multi-year ice extent translates directly to an increase in annual ice coverage (in winter the freezing boundary is well south of the northern Alaskan and Siberian coasts) and therefore an increase in salt content to supply the Br/Cl that promote MDEs. Currently, annual sea ice covers an area of about nine million km^2. Although total ice coverage is declining in some locations (Maslanik *et al.*, 1999) as annual (seasonally-forming) ice replaces multi-year ice in the Arctic Ocean, the total area affected by Hg deposition will grow, potentially doubling to about 17 million km^2.

3.1.4.2. Mercury deposition patterns

Mercury deposition appears to occur where the ocean is covered by near annual sea ice. However, the halogens and other potential oxidants (e.g., Br/Cl; see Section 3.1.3) that induce Hg deposition can also be advected by wind and extend over land or multi-year sea ice to a distance of roughly 400 km, consistent with the scale of transport of oxidants within near-surface air at night when they are protected from photodissociation. In late March, 12-hr days and nights coupled with a wind speed of 35 km/hr at about 20 m (e.g., average March wind speed at Barrow measured at 20 m height is 35.1 km/hr) can advect potential oxidants roughly 420 km during a single night. Any increase in synoptic activity and local air mass circulation resulting in stronger winds will increase the inland and multi-year ice areas impacted by Hg deposition. In May, under near 24-hr sunlight, the BrO map shows very little inland advection of potential oxidants, as these products (including BrO) are photodissociated close to their open-water annual ice sources.

3.1.4.3. The Arctic: a global recipient of mercury

Based on current projections of observed changes, it is anticipated that Arctic-wide reductions in multi-year ice coverage and its replacement by annual ice coverage will increase areas of open water, favor the formation of frost flowers, and promote marine primary productivity (ice algal communities), all of which promote the production and release of BrO to near-surface air. As larger surface emissions of BrO encounter air masses containing GEM from southern latitudes under sunlight conditions, greater MDEs will occur, provided that BrO supply is the limiting process – something that remains to be demonstrated. Spring Hg deposition rates and locations are thought to be a function of atmospheric transport, annual sea ice extent, and local air mass circulation. These are influenced by average spring and summer temperatures, the melting of multi-year ice (Maslanik *et al.*, 1996), synoptic activity (frequency of warm and cold fronts moving through the area), and variations in the position of the polar front (Dickson, 1999; Serreze *et al.*, 1993).

Mercury deposition in the Arctic has been estimated through some simplifying assumptions by Lindberg *et al.* (2002), Lu *et al.* (2001), and the DEHM Model (see Section 3.1.2.3) to range from ~50 to 300 t/yr based on Hg data from Barrow and Alert. Clearly, these estimates are subject to considerable uncertainty (e.g., they can only crudely approximate the re-emission of deposited Hg during snowmelt). The range is approximately 6 to 10% of the annual anthropogenic Hg emissions of 2235 t/yr (Pacyna and Pacyna, 2001) depositing on 1 to 2% of the Earth's 510 million km^2 surface.

With projected decreases in ice cover, higher spring Hg deposition rates may be offset by enhanced evasion from open water and Hg re-emission during snowmelt. The major evasion route for Hg from the ocean is by sea-air exchange of gaseous Hg forms. Enhanced evasion may offset the enhanced downward flux of Hg due to MDEs, perhaps reducing the Hg content of some Arctic aquatic environments where loss of ice cover occurs. Re-emission during snowmelt is apparent but the overall balance between deposition and re-emission has not been quantified and remains an important goal for future research.

If Hg deposition in the Arctic is a recent phenomenon (e.g., a result of increased synoptic activity and a trend toward sea-ice conditions favoring MDEs), this

may explain recent increases in Hg accumulation in Arctic marine biota (see Chapter 5) despite efforts to decrease global atmospheric emissions of Hg since around the end of the 1980s (see Chapter 2). The Arctic may be of increasing importance as a 'sink' in the global Hg cycle (Lindberg *et al.*, 2002; Lu *et al.*, 2001). Complex interplay between the Hg cycle and climate change requires further evaluation.

3.2. Atmospheric reduction of lead in the Arctic

Of the trace metals, Pb was early recognized as a valuable natural element as well as an environmental toxicant and has been redistributed and released to the environment by human activities over the last 5000 years (e.g., Patterson, 1971). In the early 1920s, alkyl-Pb was discovered to have special properties that could increase engine performance by preventing self-ignition. Gasoline with Pb additives facilitated the development of high-compression engines (Murozumi *et al.*, 1969). As a result, the consumption of leaded gasoline increased dramatically during the 1970s and became the main source of escalating Pb emissions to the environment. Lead emissions from cars account for a major proportion of the 119259 t of Pb emitted to the atmosphere each year (Pacyna *et al.*, 1995, see also Chapter 2).

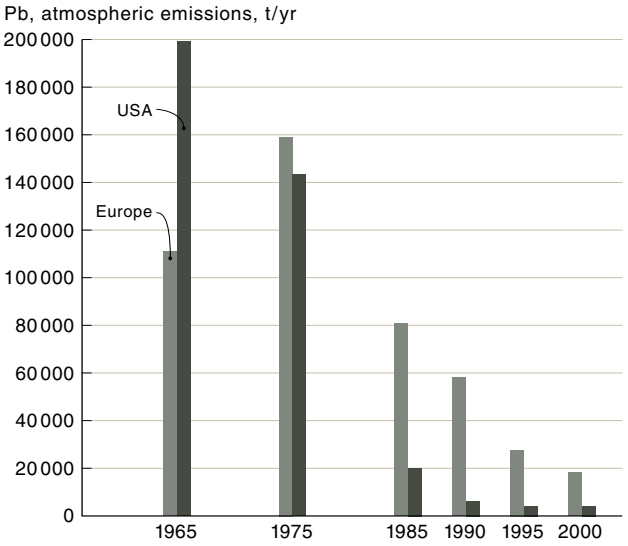


Figure 3·13· Variation in lead emissions to the atmosphere between 1965 and 2000 (Pacyna and Pacyna, 2000; USEPA, 1997).

As early as the 1960s, the role of increasing vehicular traffic in creating severe air pollution problems in high-income countries was recognized. Legislation was passed during the 1970s and 1980s by the United States, Germany, and Scandinavian and other countries to reduce Pb emissions (e.g., Deutscher Bundestag, 1971; Hagner,

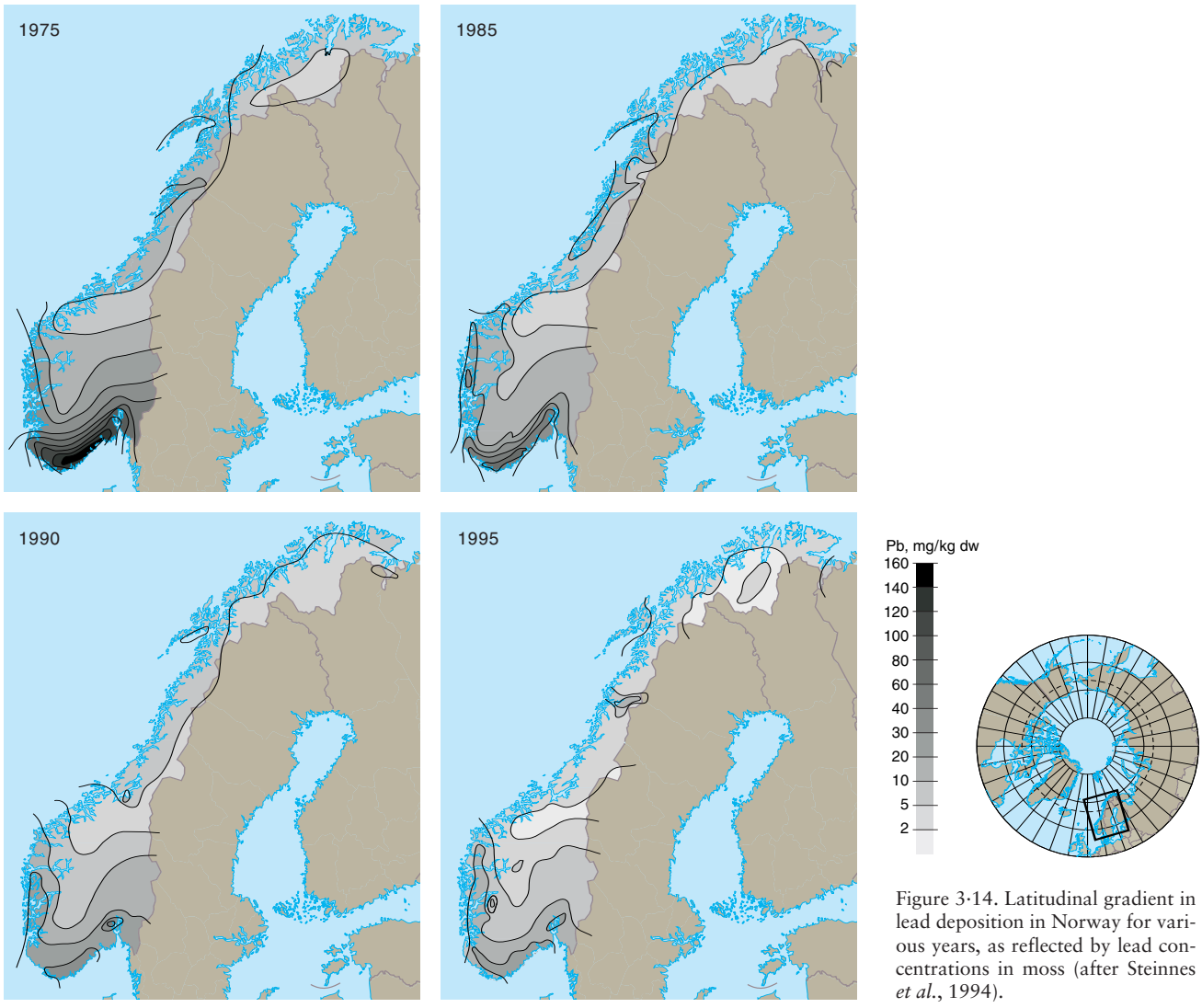


Figure 3·14. Latitudinal gradient in lead deposition in Norway for various years, as reflected by lead concentrations in moss (after Steinnes *et al.*, 1994).

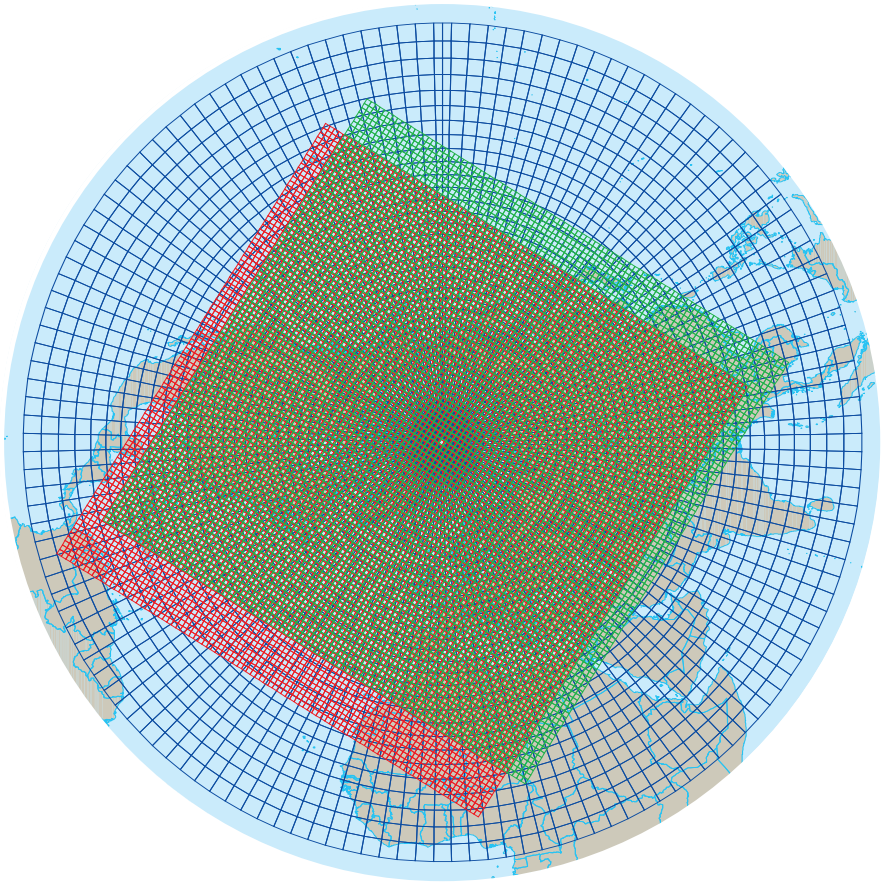


Figure 3·15. Superimposed model domains and grids for the MSC-E model, the DNMI model, and the DEHM model.

2000). The introduction of 'unleaded' gasoline (e.g., a gasoline Pb content below 0.013 g/L) in North America and Europe significantly reduced emissions (Pacyna *et al.*, 1995).

Changes in atmospheric Pb emissions from the United States and Europe from the 1960s to 2000 are shown in Figure 3·13. Sharp decreases in Pb emissions from 1975 to 1985 are evident, particularly in the United States, and are closely correlated with decreases in the use of leaded gasoline. Lead emissions are estimated to have decreased by a factor of three in Europe and a factor of six in the United States between 1985 and 1995. The European emission trend closely parallels trends in estimated global emissions of atmospheric Pb (e.g., >332 000 t in 1983 and 119 000 t in 1995; Pacyna and Pacyna, 2001). These changes are consistent with estimates of Pb emissions from vehicular traffic during this period.

The reduction in Pb emissions can be viewed as a successful environmental outcome based both on empirical data and on transport models that show decreasing atmospheric concentrations and deposition.

3.2.1. Environmental reductions in lead

Atmospheric Pb concentrations in the High Arctic (e.g., stations in Ny Ålesund, Nord, and Alert) in the early 1980s were higher than in the early 1990s, paralleling the reduction in use of leaded gasoline in the Northern Hemisphere (AMAP, 1998). Data support conclusions that atmospheric Pb, known to be transported long distances with air masses (e.g., AMAP, 1998 and Section 3.2.2), is deposited in terrestrial and aquatic environments. When emissions are reduced, environmental concentrations are expected to decrease. Reductions in atmospheric Pb concentration were indicated by moss samples taken in Norway between 1975 and 1995 (Figure 3·14), where deposition in 1975 was four to five times greater than in 1995. These environmental reductions parallel reductions in Pb emissions in Europe, a pattern repeated for Pb and other contaminants in Norway (e.g., Pacyna *et al.*, 1984; Steinnes *et al.*, 1994). Additional assessment is needed to compare concentrations in the late 1990s.

3.2.2. Models of atmospheric transport of lead

Models predicting how Pb moves through the environment can help explain environmental Pb levels and their change over time in response to emission reductions. The first AMAP assessment presented results from the Norwegian Meteorological Institute (DNMI) model simulation of Pb transport to the Arctic (for the reference year 1988), based on an application of the (1996 updated) hemispheric EMEP (Cooperative Programme for the Monitoring and Evaluation of the Long Range Transmission of Air Pollutants in Europe) transport model. Since then, two additional models have been developed for the hemispheric transport of Pb to the Arctic; the DEHM model (Christensen, 1997, 1999) and the Meteorological Synthesizing Centre East (MSC-E) Hemispheric Model (MSCE-HM-Hem; Travnikov, 2001).

Results from the DEHM and MSC-E models are presented in Sections 3.2.2.1 and 3.2.2.2. These show how updated information has altered transport predictions. The model domain and grid for the DEHM and MSC-E models are similar, although not identical to the DNMI Hemispheric Model (Figure 3·15).

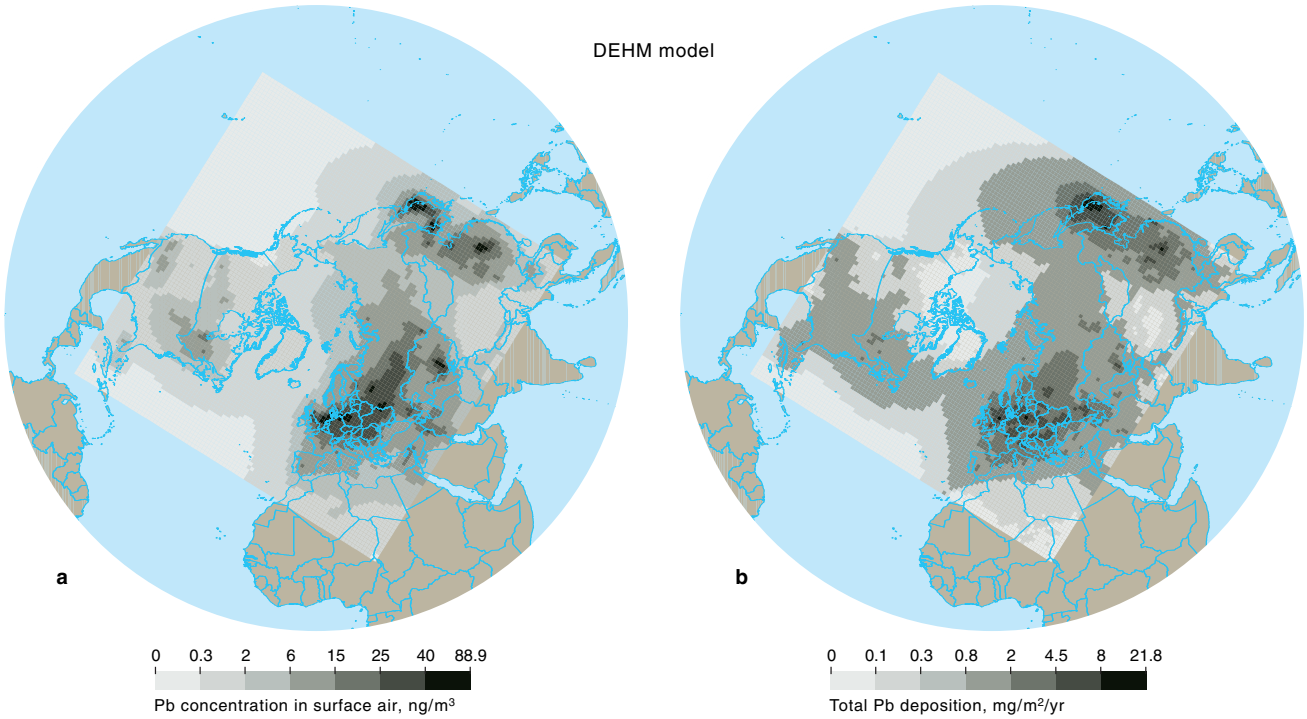


Figure 3·16. a) Mean lead concentrations in surface air and b) total lead deposition in the central part of the DEHM model domain (J. Christensen, unpubl. data, 2002).

3.2.2.1. Danish Eulerian Hemispheric Model

The DEHM is a three-dimensional air pollution model that has been used to calculate the atmospheric transport of Pb in the Arctic and Northern Hemisphere. The DEHM model was run using estimates of Pb emission from 1989 as the source year. The simulation covered a 10-year period, from 1991 to 2000. The model domain covers an area to about 20°N and comprises a 96 × 96 cell grid with a (nominal) spatial resolution of 150 km × 150 km, based on a polar stereographic projection with a true-distance latitude of 60°N.

Figure 3·16 presents estimates of mean Pb concentration in surface air, and total Pb deposition in the central part of the model domain. These results show high concentrations and deposition close to large source regions in North America, Europe, and Asia. Within the Arctic, surface air concentrations vary from 0.3 ng/m^3 in the central Arctic Basin to >15 ng/m^3 near European source

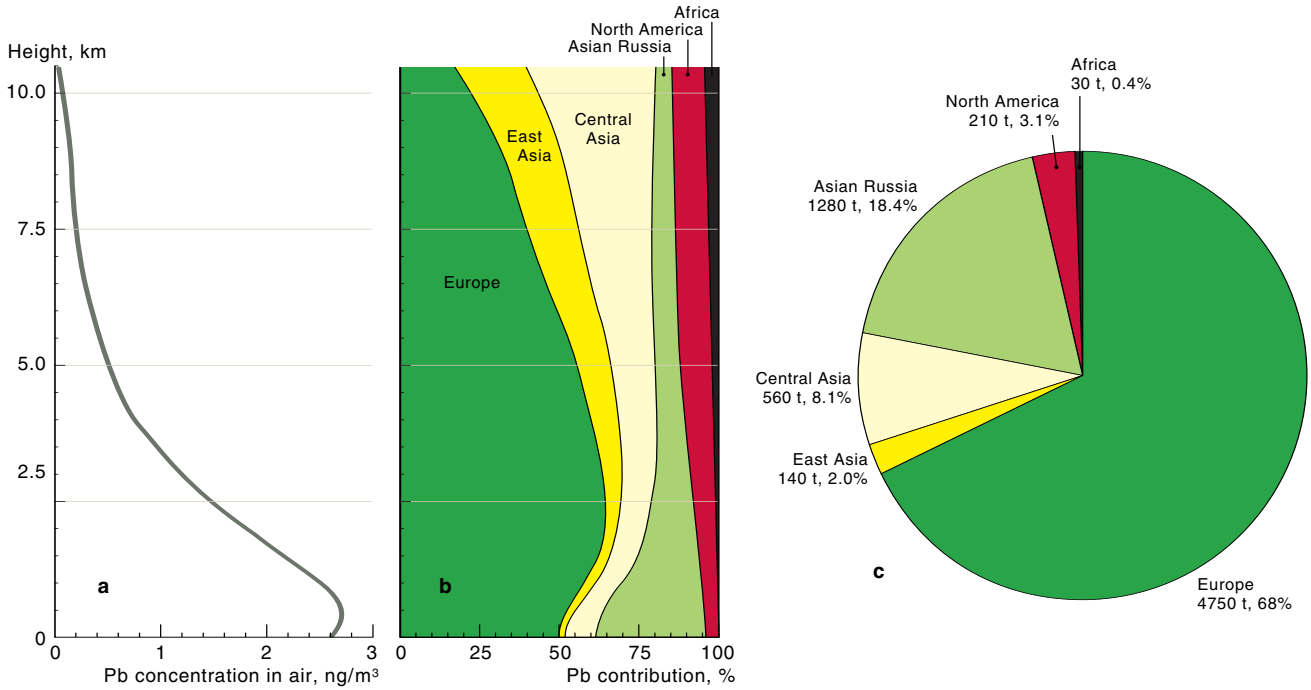


Figure 3·17. a) The vertical distribution of lead in the atmosphere, coupled with b) the percentage contribution of lead emissions from different global regions. Contributions from different source regions to total lead deposition are shown for the area north of the Arctic Circle (J. Christensen, unpubl. data, 2002).

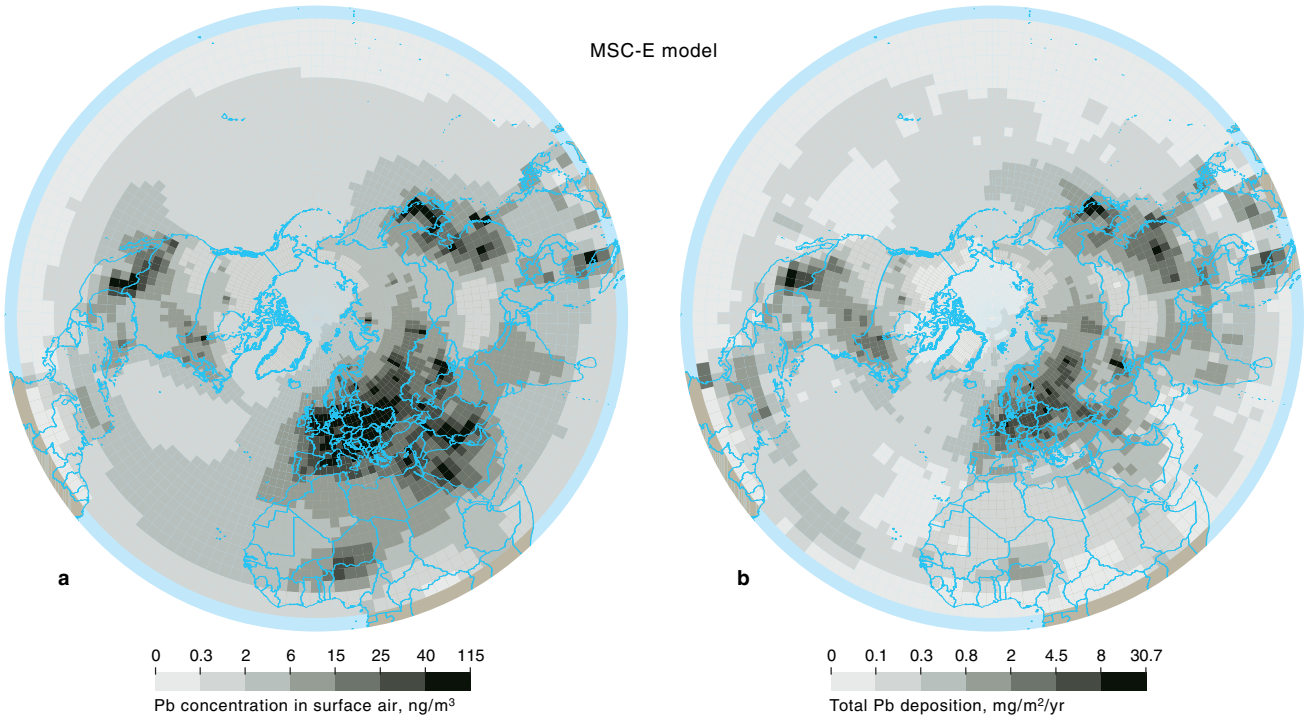


Figure 3·18. a) Mean annual atmospheric lead concentration, and b) total annual deposition of lead to the Arctic in 1990, as calculated by the MSC-E Hemispheric Model (Meteorological Synthesising Centre-East, Moscow).

regions and Norilsk. Deposition varies from <0.1 mg/m^2/yr in the central Arctic to about 2 mg/m^2/yr in the European Arctic. The main transport pathway into the Arctic is from Europe and the Russian part of Asia.

The relative contributions from different source regions to the vertical distribution of the mean air Pb concentrations and to the total deposition for the area north of the Arctic Circle are shown in Figure 3·17. European sources, including the European area of Russia contribute 50 to 60%. Up to 40% of the concentrations in the lower part of the troposphere are from Asian Russia, consistent with vertical atmospheric layers associated with different regions. Around 68% of Pb deposition in the Arctic north of the Arctic Circle is from European sources and about 20% from sources in the Asian part of Russia.

3.2.2.2. MSC-E Hemispheric model

The MSC-E Hemispheric model is related to a family of three-dimensional Eulerian models covering the entire Northern Hemisphere with a spatial resolution of 2.5° latitude by 2.5° longitude. It uses a similar approach to that of the DEHM model, although the models are distinctly different. The MSC-E model was used to calculate the expected atmospheric transport of Pb to the Arctic for 1990.

Model results are presented in Figures 3·18, 3·20, and 3·21. Figure 3·18 shows predicted mean annual air concentrations and total Pb deposition. The overall pattern reflects the influence of major anthropogenic source areas. Maximum Pb concentrations in the surface layer (~ 45 ng/m^3) and the maximum total deposition flux (~5 mg/m^2/yr) within the Arctic Circle are associated with the Norilsk area. The pronounced decrease in both the predicted air concentration and deposition fields from Eurasia toward North America indicate that the major dispersion fluxes into the Arctic are from the Eurasian continent.

The MSC-E model was also used to investigate the relative contributions and transport pathways of airborne Pb from different continents and regions. For this purpose the entire hemispheric emission field was divided into areas representing emission sources in Europe, Africa, North America, South America, and Asia. In addition, Asia was subdivided into: 1) western-, central-, and southern Asia; 2) eastern- and southeastern Asia; and 3) Asian Russia (Figure 3·19).

Figure 3·19. Continents and regions of the Northern Hemisphere used as lead emission sources in computations with the MSC-E model.

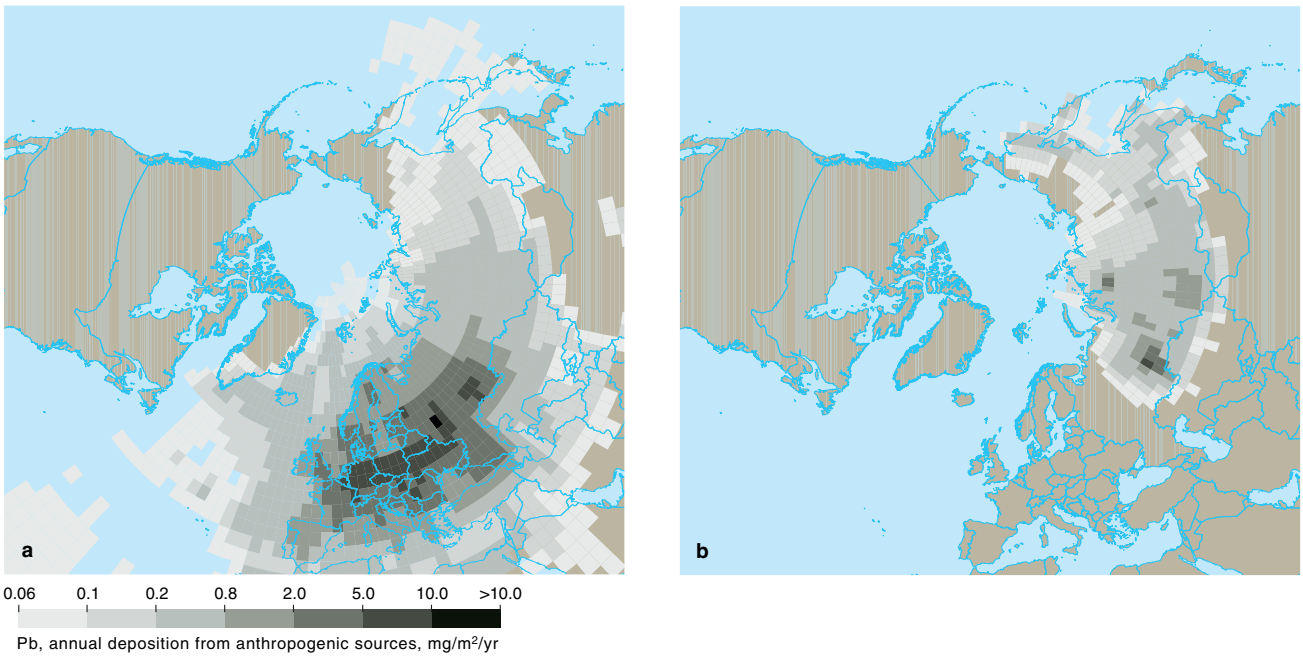


Figure 3·20. Mean annual deposition fields for anthropogenic lead from emission sources in a) Europe, and b) Asian Russia, as calculated by the MSC-E Hemispheric Model (Meteorological Synthesising Centre-East, Moscow).

Figure 3·20 shows the mean annual deposition fields for anthropogenic Pb from the emission sources in Europe (Figure 3·20a) and Asian Russia (Figure 3·20b) that are responsible for most Pb deposited in the Arctic. Although anthropogenic emissions of Pb from Asian Russia are comparatively low, its proximity to the Arctic results in a significant contribution to the Arctic pollution load.

The MSC-E model results provide some evidence of transcontinental transport. Despite the absence of a significant flux of anthropogenic Pb northward from source regions in southeastern Asia and North America, there are indications that Pb from these regions can penetrate into the Arctic over southern Greenland (Kalaallit Nunaat) and to the east of Greenland after crossing the North Atlantic. Arctic air concentrations and deposition associated with these sources, however, is very low. Thus, the main atmospheric pathways of Pb transport to the Arctic, as indicated by the MSC-E model, are from Europe, Siberian Russia, and through the northern Atlantic route.

The relative contribution from sources in different continents to the deposition of Pb within the Arctic is illustrated in Figure 3·21b. According to the MSC-E model, 61% of Pb deposition in the Arctic is from European sources, and 25% from Asian Russia. The combined contributions from other continents, although accounting for some 62.5% of total global Pb emissions in 1989 (see Figure 3·21a), appear to contribute no more than a few percent of the Pb deposited in the Arctic.

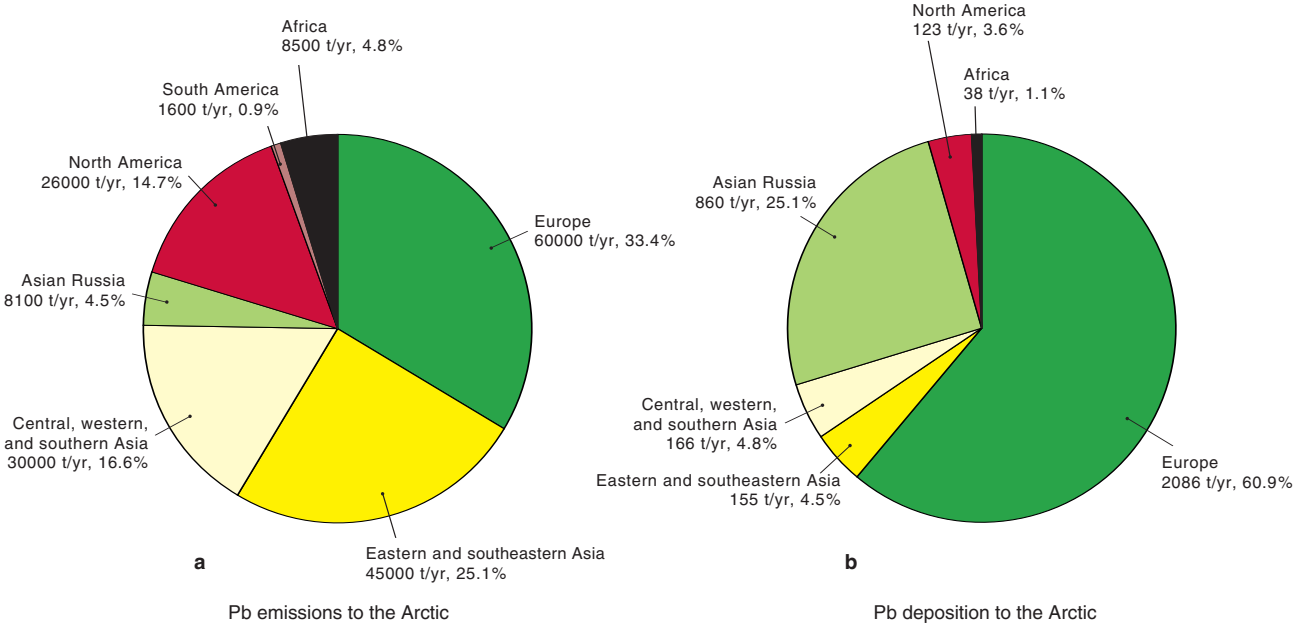
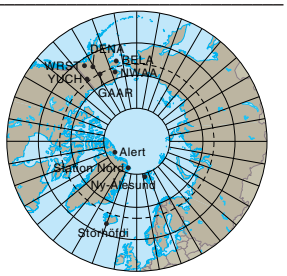


Figure 3·21. Relative contribution of different source regions to a) lead emissions, and b) lead deposition to the Arctic region, as calculated by the MSC-E Hemispheric Model (Meteorological Synthesising Centre-East, Moscow). Source regions are as defined in Figure 3·19.

Table 3·1. Comparison of the modeled annual mean surface air lead concentrations against data from air monitoring sites within or near the Arctic (Berg and Hjellbrekke, 2001). N< = number of values less than detection limit.

		Modeled			Observed										
Station	Coordinates	DEHM, 1990	DNMI, 1988	MSC-E, 1990	Arithm. mean	Arithm. SD	Geomet. mean	Geomet. SD	Minimum	5%	50%	95%	Maximum	N	N<
Alert, 1990	82.5°N, 62.5°W	0.96	0.032	0.799	0.78	0.92	0.38	3.52	0.1	0.1	0.32	2.87	3.1	49	24
Nord, 1994	81.5°N, 17.5°W	1.29	0.058	1.1	0.97	1.21	0.45	4	0	0	0.45	3.48	4.5	52	13
Stórhöfði, 1995	63.4°N, 20.3°W	0.661	0.754	3.27	1	0.39	0.9	1.63	0.2	0.3	1.01	1.47	1.5	22	0
Ny-Ålesund, 1994	78.9°N, 11.9°E	1.97	0.346	2.6	0.83	1.93	0.24	4.65	0	0.1	0.21	2.82	13	52	18
NWAA, 1990	67.9°N, 162.6°W	0.815	0.058	0.958	2.29	3.04	1.6	2.51	0.3		1.64		24	87	16
BELA, 1990	64.8°N, 165.7°W	0.696	0.079	1.046	7.26	10.21	3.17	3.09	0.3		2.91		73	74	5
GAAR, 1990	66.9°N, 151.5°W	0.57	0.049	0.618	1.24	1.38	1	2.29	0.2		1.21		9.8	73	16
DENA, 1987	63.7°N, 149.5°W	0.329	0.167	0.734	1.08	0.92	0.84	1.96	0.3		0.68		5.1	88	37
YUCH, 1990	64.8°N, 141.2°W	0.295	0.036	0.505	1.02	0.74	0.8	2.13	0.3		0.9		3.5	89	32
WRST, 1990	62.0°N, 145.1°W	0.223	0.034	0.883	4	2.85	2.71	2.28	0.7		2.61		9.8	17	0
KATM, 1990	58.7°N, 156.6°W	0.235	0.071	0.791	1.19	0.72	0.89	2.13	0.3		1.16		2.7	45	12

Location of stations
Alert, Canadian High Arctic; Station Nord, Greenland; Stórhöfði, Iceland; Ny-Ålesund, Svalbard; NWAA, Northwest Alaska Areas National Park, Alaska; BELA, Bering Land Bridge National Preserve, Alaska; GAAR, Gates of the Arctic National Park, Alaska; DENA, Denali National Park, Alaska; YUCH, Yukon Charley National Preserve, Alaska; WRST, Wrangell St. Elias National Park, Alaska; KATM, Katmai National Park, Alaska.



3.2.2.3. **Comparison of model results**

The three Pb transport models employed the same basic input data on Pb emissions, derived from the Global Emission Inventory Activity (GEIA) 1989 global anthropogenic Pb emission inventory (Pacyna and Keeler, 1995). The DEHM and MSC-E models were run using the GEIA high estimate emission data set, spatially distributed to a grid suitable for model input. Both employed zero concentrations as the initial and lateral boundary conditions. The DNMI and DEHM models used meteorological data from the European Centre for Medium Range Weather Forecasting (ECMWF). The MSC-E model used meteorological data from the Hydrometeorological Centre of Russia. For the DEHM model, the ECMWF data are used as input to the Mesoscale Model version 5 module that produced the meteorological data for the air pollution model. The DEHM model was run for the 10-year period, 1991 to 2000, using meteorological data for 1989, such that the results effectively represent the mean transport of 1989 emissions. The DNMI model was run based on 1988 meteorological data. The MSC-E model run was based on meteorological data for 1990.

A comparison of the modeled annual mean surface air Pb concentrations using data from air monitoring sites within or close to the Arctic is presented in Table 3·1. The three models all produced results that were generally consistent with the ranges observed at the monitoring sites, although for several locations the concentrations obtained from the DNMI model were an order of magnitude lower than those for the other models. A source of error in these comparisons results from the use of Pb data from air monitoring stations for 1994 or 1995 onward, while the models were run using emission data sets for 1989.

A Geographic Information System (GIS) was used to compare total Pb deposition, as estimated by the models, for different regions around the North Pole (the areas north of 50° N, 60° N, and 70° N, the area north of the Arctic Circle, and the area within the AMAP boundary. Results are presented in Table 3·2. Although not constituting a formal model inter-comparison, the analysis provides insight into the variability in model estimates for total annual Pb deposition to the Arctic in the period around 1990.

The deposition estimates presented in Table 3·2 are based on 1989 emissions estimates. As global anthropogenic Pb emissions decreased from around 330 000 t to 120 000 t between 1983 and 1995, running the models with 1995 emissions data (J. Pacyna, pers. comm., 2002) is a high priority. Although the 1995 emissions inventories have been adapted and used in regional models

Table 3·2. Total lead deposition estimates for emissions and model results for 1990 (S. Wilson and F. Steenhuisen, pers. comm., 2002).

	DEHM	DNMI	MSC-E
In model domain (t/yr)	139835	ca. 150134	151178
North of 50°N (t/yr)	50793	50482	47196
North of 60°N (t/yr)	16188	13371	12282
Within AMAP boundary (t/yr)	11322	7985	6731
North of Arctic Circle (t/yr)	6788	4791	3278
North of 70°N (t/yr)	4363	2637	1884

(e.g., European-scale models), the revised emission inventories have yet to be spatially distributed to a form suitable for input to the hemispheric-scale models. The lack of an updated global anthropogenic emission inventory for Pb for 2000 should also be addressed.

Based on the GIS analysis, considerably higher estimates of total Pb deposition north of the Arctic Circle were obtained using the DEHM model (~6790 t/yr) compared to the other models (DNMI, ~4790 t/yr; MSC-E, 3280 t/yr). Considering air concentration pat-

terns, the MSC-E model shows considerably higher Pb transport across the North Atlantic, and then northward into the Arctic in the Greenland Sea area, than do either of the others. This is evident in Figure 3·18a where the higher air concentrations in the Greenland Sea region and the general area between Iceland and Svalbard are evident. At the location of the Stórhöfði monitoring station in Iceland, the MSC-E model estimates an average air concentration of ~3.3 ng/m^3 versus 0.66 to 0.75 ng/m^3 for the other two models. Note that this is more than the maximum observed value at Stórhöfði. However, the Stórhöfði monitoring data are for 1995 and the model was run for 1990; these are not directly comparable. Conversely, the DEHM model appears to transport more Pb into the Arctic in the region north of central Siberia, resulting in higher air concentrations over the Eurasian Basin of the Arctic Ocean. Higher deposition over this part of the Arctic Ocean, and also over the Barents and Kara Sea areas (Figure 3·16), is apparently responsible for the higher total deposition estimates obtained from the DEHM model. The use of different meteorological data in the DEHM and MSC-E models may explain some of these differences. However, the difference in total deposition is also apparent when comparing the outcome of the DEHM and DNMI models, for which comparable meteorological data were used. Surface air concentrations over the central Arctic are generally higher for both the DEHM and MSC-E model relative to the DNMI model.

Significant atmospheric changes were occurring during the data years used for modeling, which probably influenced model outcomes. The 1990-based MSC-E model would have more intense circulation out of eastern North America and up into the Nordic Seas under a high Arctic Oscillation or North Atlantic Oscillation index and more intense precipitation in the Nordic Seas and Eastern Arctic. Consideration of these processes under global change scenarios would be an appropriate next step for future modeling. With the exception of the differences noted (particularly different meteorological data inputs to the models) the broad features of surface air concentration and deposition patterns obtained from the models were consistent. The DEHM and MSC-E models both confirm that the major sources of Pb transported to the Arctic are Europe and the Asian part of Russia. The associated estimates (DEHM: 68% from Europe and 18% from Asian Russia, and MSC-E: 61% from Europe and 25% from Asian Russia) are comparable. However, in terms of the total Pb deposition flux to the Arctic, the models yield distinctly different results. The total Pb deposition north of the Arctic Circle as estimated by the DEHM model (6790 t/yr) is approximately twice that estimated by the MSC-E model (3280 t/yr). The reasons for these differences should be further examined.

3.2.3. Cause and effect: elimination of lead gasoline additives and environmental reductions

Model results for surface air concentrations and deposition patterns, coupled with data on changes in emissions and environmental levels of Pb over time, suggest a causal link between atmospheric emissions from vehicles and measured environmental Pb levels. Rapid increases in atmospheric Pb emissions and environmental Pb levels between the 1920s and 1970s are consistent with: 1) introduction of leaded gasoline, 2) attendant development of high performance gasoline engines for vehicles, and 3) significant increases in use of vehicles throughout North America and Europe. Rapid decreases in atmospheric Pb emissions are consistent with reduction and elimination of Pb additives in gasoline used for vehicles.

Since vehicles represented one of the largest and most pervasive combustion sources of leaded gasoline prior to implementation of environmental regulations, and vehicle use has not decreased, the reduction and elimination of Pb additives in gasoline by environmental regulation is considered to be directly responsible for decreasing Pb emission levels in the United States, Europe, and worldwide. Decreasing emissions of Pb to the atmosphere has been matched by a consistent decrease in deposition to the terrestrial and aquatic ecosystems, both in high emission areas and remote regions such as the Arctic. Thus, while it is premature to quantify chemical recovery of ecosystems as Pb inputs decrease, and even more difficult to assess biological recovery particularly in remote locations such as the Arctic, environmental policies on removing Pb additives in gasoline worldwide can be considered a significant success. The phase-out of Pb additives from gasoline has been environmentally efficient and beneficial as well as cost-effective (Hagner, 2000).

3.2.4. Potential risk linked to elimination of leaded gasoline

The invention of catalytic converters to reduce CO and NO_x emissions was a key driver resulting in the phase out of leaded gasoline. While this resulted in a significant reduction in global Pb emissions, there may be a risk associated with this substitution that requires assessment. Greenland (Kalaallit Nunaat) ice cores show increases in platinum group elements (platinum, palladium, rhodium) that are probably related to the escalating use of catalytic converters. The magnitude of the increase in rhodium is second only to Pb in these ice core records (Barbante *et al.*, 2001). Global atmospheric increases in these elements may be an emerging environmental issue, although there is little information from which to assess likely implications for plants, soils, wildlife, and humans. In addition, use of other additives (e.g., methylcyclopentadienyl manganese tricarbonyl (MMT), a Mn-organic compound) by gasoline producers as an alkyl-Pb replacement requires evaluation owing to the volume of emissions from vehicles.

3.3. Cadmium: new questions about pathways

Recent work at Barrow and in the surrounding area on the distribution of Cd has raised questions about how Cd moves through the Arctic environment. Sections 3.3.1 and 3.3.2 contain a set of linkages believed to explain similarities between the deposition of Cd and Hg, despite significant differences between these metals, which warrant additional assessment.

3.3.1. Local transport of cadmium and mercury

Following emission to the atmosphere, metals may be deposited in the vicinity of the emission source or transported long distances in air masses. The proportion of particles containing heavy metals deposited locally or transported away from the source is influenced by a number of factors, such as particle size, temperature and speed of exhaust fumes, height of release, and prevailing winds and precipitation conditions. However, measurements of Cd in the late spring snowpack near Barrow show a pattern of Cd enhancement similar to that of Hg (Garbarino *et al.*, 2002). Samples obtained in the vicinity of the Alaskan coast were highest for Hg and Cd southwest of Point Barrow in the region downwind of recurring open leads and polynyas. Garbarino *et al.* (2002) speculated that these snow enhancements indicated mechanisms of deposition from the atmosphere over marine waters. Analysis of Arctic mosses showed that the Enrichment Factor (EF; derived by normalizing target elements to a conservative soil element) for Cd and Hg were consistently >25 and often in the hundreds, suggesting that atmospheric deposition is important for both heavy metals (i.e., the higher the EF, the higher the proportion of the target element in the plant sample derived from sources other than soils) (J. Ford, pers. comm., 2002).

Based on current data, it seems unlikely that Hg, a gas-phase pollutant, and Cd, a particulate pollutant with different anthropogenic and natural sources, would share a common input pathway to the Arctic environment. However, they both show declines in snowpack concentration with distance inland (Snyder-Conn *et al.*, 1997) and significant atmospheric deposition in tundra vegetation (J. Ford, pers. comm., 2002). While this deposition pattern is consistent for the characteristics of Hg (see Section 3.1.3), conditions during most of the Arctic winter and spring do not promote deposition of particulate pollutants.

The transport of fine particulates (mostly sulfate and soot) to the Arctic atmosphere from anthropogenic sources outside the Arctic can result in the development of the phenomenon known as Arctic haze. The most severe haze episodes are associated with the stable, calm weather conditions that can prevail during winter. Several metals can be associated with Arctic haze particles. Their concentrations thus increase throughout the winter and early spring, as Arctic haze intensifies, and peak in late spring (e.g., Heintzenberg, 1989; Shaw and Khalil, 1989).

Exceptions to typical Arctic winter and spring conditions can occur in those areas downwind of open leads and polynyas within the annual ice zone (Figure 3·22). Fog droplets and sea-salt aerosols, with which Cd and other particle-associated contaminants have an affinity, concentrate in the near surface air downwind of these open-water areas. It is possible that scavenging of Cd by fog droplets hastens Cd deposition.

3.3.2. Mechanisms of transport and deposition

While open-water leads and polynyas appear to drive Hg and Cd deposition, the mechanisms for deposition are different. As described in Section 3.1.3.1, Hg deposition downwind of open water relies on Br and Cl derived from sea salts, followed by sunlight dissociation, and a rapid (within one to two hours) conversion of near surface GEM to RGM. Subsequent deposition of RGM to the snowpack occurs with an average deposition velocity of ~2.0 cm/s (see Section 3.1.3). Conversely, Cd deposition relies on particulate aggregation with fog droplet and sea-salt aerosols downwind of the open-water sources (Gibson and Shullinger, 1998). The aggregate particles have a deposition velocity an order of magnitude less than for RGM (i.e., ~ 0.2 cm/s for aggregate Cd particles). Given the vastly different deposition velocities in the downwind areas of open-water sources, it is difficult at first to explain the similarities in the distribution and correlation of Cd and Hg concentrations in the snow and vegetation. Based on deposition velocities, Hg should deposit close to open water, and Cd should deposit farther downwind. However, the influence of UV-B in sunlight may explain this apparent inconsistency. Cadmium deposition occurs regardless of light regime. Whereas photolyzable Br/Cl derived from the open ocean dissociates only under UV-B radiation. As a result Hg deposition occurs primarily during daylight hours.

Figure 3·22. The ice-edge ecosystem in the Chukchi and Bering Seas (Gibson and Schullinger, 1998).

Based on current information, best estimates for the rates of Cd and Hg deposition are derived as follows. The atmospheric boundary layer in the Arctic during spring is roughly 75 to 200 m thick. Using 100 m as an average depth, a particulate deposition velocity of 0.2 cm/s, and a mean wind speed of 8 m/s, the Cd particulate deposition should extend to 400 km (8m/s ×100 m/0.2 cm/s) from open-water sources. Using the same numbers for Hg, BrO emissions at night under late March conditions (12 hours of darkness) should advect

to approximately 346 km (12 hours × 8 m/s) from open-water sources. Using 2.0 cm/s as the deposition velocity of the newly formed RGM, the deposition extent once RGM is formed is estimated at 40 km (8 m/s × 100 m/2.0 cm/s). The total of 386 km (346 km + 40 km) for Hg is comparable to Cd (400 km), and consistent with the BrO maps in Section 3.1.3. The advection distance of photolyzable Br/Cl at night delays the production of the faster depositing RGM such that it approximates the deposition range of more slowly depositing Cd aggregated particles. Under low wind conditions, both Hg and Cd are expected to deposit closer to open-water sources. High winds should extend deposition.

The observed correlation of Hg and Cd may be influenced by environmental variables that must also be considered. Arctic mosses show inland trends of EFs that are consistent with the proposed deposition patterns. However, these patterns may also be related to environmental variables such as soil acidity. Mercury and Cd mobilization within soils is highly affected by pH (Lodenius and Autio, 1989; Lodenius and Malm, 1990). Mercury and Cd uptake rates are also dependent on plant type and tissue distribution (root/rhizoid mass versus leaf mass). Further investigation is warranted.

Another interesting characteristic of Cd is that it appears to deposit more readily in the Arctic than other particulate elements having similar deposition mechanisms. This appears to be related to the location of anthropogenic releases and characteristics of air mass movement to the Arctic. Air masses in the Arctic during winter and spring are stratified according to source region. Warmer wetter temperate air masses typically rise in altitude with increasing latitude (Carlson, 1981) and override cold dry Arctic or subarctic surface air masses. Since the Cd sources in the Arctic are primarily the ocean and the non-ferrous metal industries in Siberia (to where the Arctic air mass extends during winter), air masses enriched with Cd tend to be confined close to the Earth's surface as they move to higher latitudes. Harris and Kahl (1994) confirmed that particulate pollutants in Arctic haze transported in cold dry air masses from Siberia are found near the surface, while warmer wetter north European emissions travel at higher altitudes.

Pacyna *et al.* (1985) reported that, of 15 major elements in Arctic haze particles, Cd was the only one found exclusively at low (below 2000 m) altitude. Since low altitude is a prerequisite for deposition from the atmospheric boundary layer, this may explain increased deposition of Cd from Arctic haze. Particulate elements evenly distributed throughout the Arctic air column are more likely to experience transport out of the Arctic, or persist until summer when rainout over broad areas may occur.

3.4. Freshwater pathways and fluxes

Freshwater transport is an important pathway for contaminants reaching the Arctic Ocean (AMAP, 1998). Rivers, groundwater discharge, and river ice are all considered freshwater pathways (Kimstach, 1997; Skotvold and Wartena, 1997). The following sections focus on new information about the volume of freshwater flow from rivers, and discuss the possible transport of contaminants in ice. In addition, information is provided about the sources and inputs of contaminants in the major rivers and other freshwater pathways entering the Arctic Ocean. Groundwater transport is discussed in Section 3.5.

3.4.1. River discharges to the Arctic Ocean

Recent investigations provide relatively consistent estimates of total river inflow to the Arctic Ocean at 4270 km^3/yr (Vuglinsky, 1997) and 4280 km^3/yr (Shiklomanov *et al.*, 2000). Statistical analyses of multi-annual data on water discharge showed no significant variations in annual river discharge to the Arctic Ocean from 1926 to 1965, 1966 to 1976, and 1977 to 1996 (4275, 4370, and 4318 km^3/yr respectively; Shiklomanov *et al.*, 2000). The difference between maximum (4870 km^3, 1974) and minimum (3820 km^3, 1953) discharges between 1926 and 1996 is about 30%.

Analysis of Russian river discharge dynamics as influenced by human activity has shown that due to low population densities and minor economic development of the Siberian territory, anthropogenic influence is relatively minor and unlikely to produce noticeable effects on water discharge to the Arctic Ocean, even in the near future. As an example, Figure 3·23 shows monthly discharges in the Yenisey basin, the largest of the Arctic rivers with a multi-annual average discharge of 630 km^3.

The Yenisey also has the greatest potential for anthropogenic impact based on future projections. In 1995, the rate of total water withdrawal for public services,

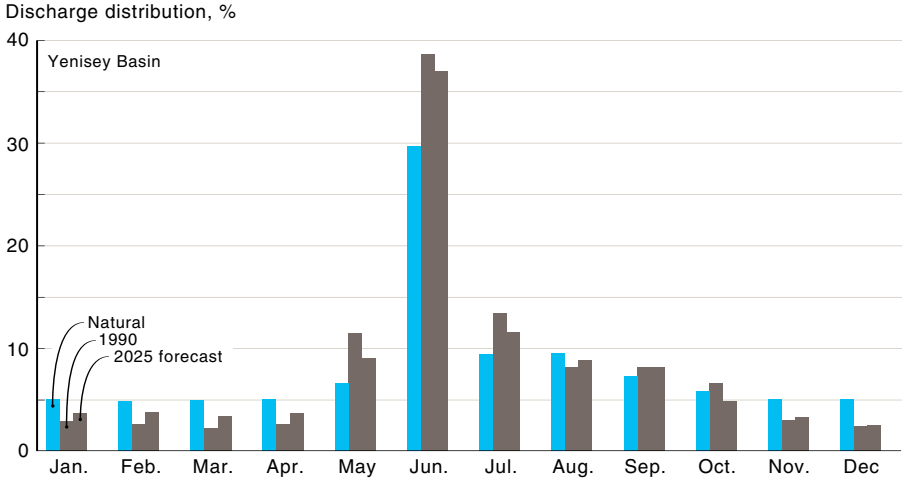


Figure 3·23. Monthly discharge distribution for different levels of human activity in the Yenisey River Basin (Shiklomanov *et al.*, 2000).

agriculture, and industry, together with water loss by evaporation from reservoirs was 8.7 km^3/yr (total storage of operating and constructed reservoirs in the basin was 482 km^3, operating at 174 km^3). Water consumption was 4.9 km^3/yr. By 2025, these values are expected to grow to 12 and 7 km^3/yr, based on an expected 25 to 40% increase in human use, or 1.2 to 2.0% of the annual river runoff at the mouth (Shiklomanov *et al.*, 2000). This represents a small change from current riverine discharge to the ocean. The current total freshwater withdrawal from the Arctic basin is 0.9 to 1.4% of the total water discharged to the ocean.

3.4.2. Riverine contaminant fluxes

Most recent work on heavy metals transported to the Arctic Ocean by rivers focused on the Ob, Yenisey, and Lena in Russia. Other major inputs to the Arctic include the Mackenzie (Canada) and Yukon (United States) rivers (the Yukon discharges into the Bering Sea). Most estimates of heavy metal input to the Arctic from rivers have used summer flow estimates, possibly overlooking the most important transport period – spring freshets – that are driven by terrestrial snowmelt and river ice break-up.

Spring freshets represent about 35% of the annual flow for the Lena River. In the Lena River, levels of suspended matter (see Section 3.4.3) increased from 2 mg/L in summer to 37 mg/L during spring flows (Hölemann, 2001). Although the concentration of metal contaminants on suspended matter was relatively constant during spring freshets, dissolved metals (manganese (Mn), iron (Fe), nickel (Ni), copper (Cu), zinc (Zn), and Pb) in the Lena River, and in the freshwater layer under the fast ice in the Laptev Sea, were about two to five times higher. Previously published metal budgets for the Lena River, and potentially other Arctic rivers, that do not include spring high flow periods are likely to have considerably underestimated the fluxes of dissolved trace metals from large rivers discharging to the Arctic Ocean.

Ecological-geochemical investigations, including the measurement of heavy metal concentrations in water and suspended matter in the middle reaches of the Ob River were recently completed (Leonova *et al.*, 2000; Schvartsev *et al.*, 1999; Sorokovikova *et al.*, 1999). Schvartsev *et al.* (1999) analyzed ten metals (Hg, Cd, Pb, Cu, Zn, Mn, chromium (Cr), molybdenum (Mo), silver (Ag), lithium (Li)) in 20 samples from the Ob River, taken from the confluence of the Ob and Tom Rivers, along the Ob main stream to the town of Nizhnevartovsk in July 1999 (Figure 3·24). Over this distance (approximately 1000 km) total dissolved minerals in water decreased from 150 mg/L to 90-110 mg/L and pH decreased from 8.3-8.5 to 6.5-7.5, caused by inflows of relatively less mineralized and more acidic waters from major tributaries. Concentrations of dissolved trace metals varied insignificantly: Hg, 0.09; Cd, 0.08; Pb, 0.50; Cu, 1.0; Cr, 8.0; and Mo, 0.4 µg/L. The Zn and Mn concentrations, however, increased from south to north: Zn, 0.5-1.0 to 30-40; Mn 5-6 to 15-18 µg/L. These increases are thought to be due to elevated concentrations of dissolved organic acids originating from wetlands in the northern part of the watershed (Schvartsev *et al.*, 1999).

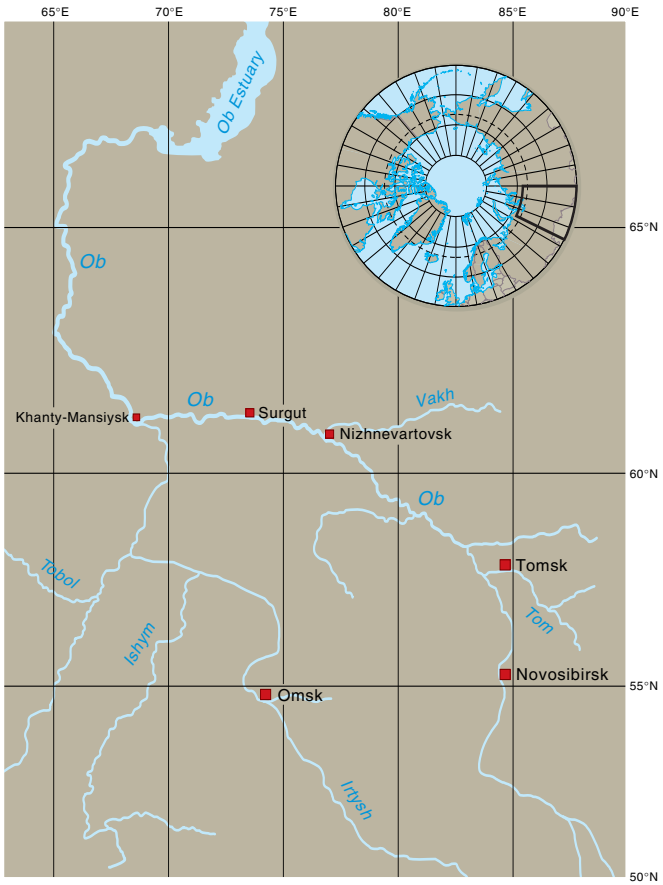


Figure 3·24. Sites for assessing heavy metal concentrations in water and suspended matter along the middle reaches of the Ob River.

Fifty-three samples of Tom River water from Seversk downstream to the confluence with the Ob River, and of Ob River water downstream to Surgut were collected in summer 1999 and analyzed for heavy metals (Sorokovikova *et al.*, 1999). High concentrations of Mo, Ni, cobalt (Co), and beryllium (Be) were detected in the Tom River near Seversk. At the same location, elevated concentrations of sulfate, chloride, and mineral nitrogen indicate contamination due to industrial and domestic wastes. Along the mainstem of the Ob River, and in contrast to the results by Schvartsev *et al.* (1999), heavy metal concentrations varied widely and with no systematic pattern of variability (e.g., Ni, 1.1-2.3; Zn, 0.02-15.2; Mo, 0.1-3.0; Cd, 0.11-1.0; Pb, 0.04-0.5 µg/L).

In September 2000, Leonova *et al.* (2000) studied a section of the Ob River from the Novosibirsk dam downstream to the Tom River confluence and the Tom River from its mouth upstream to the town of Tomsk. Analyses show that the lower reach of the Tom River is subject to more intensive anthropogenic impact than the Ob in its middle course (e.g., near Seversk on the Tom River). Leonova *et al.* (2000) measured up to: Hg, 1.5; Cu, 4.4; Zn, 2.5; Mn, 9.9; Pb, <0.5; and Cd, <0.05 µg/L. In the Ob River these authors found: Hg, <0.02; Cu, 2.7-4.6; Zn, <1-3.2; Mn, 2.0-8.0; Pb, <0.5; Cd, <0.05; Co, <1.0; and arsenic (As), <5.0 µg/L.

Gordeev (2001) reviewed data collected over the previous ten years on heavy metals in Russian Arctic rivers and found the data obtained during expeditions to the Ob, Yenisey, and Lena rivers and their estuaries and deltas under the Russian-French-Netherlands 'Scientific

Table 3·3. Dissolved concentrations (mean and range) of heavy metals and iron in the three largest Russian Arctic rivers in comparison with the global average (Cossa *et al.*, 1996; Dai and Martin, 1995; Gordeev, 2001; Guieu *et al.*, 1996; Kravtsov *et al.*, 1994; Martin and Gordeev, 1986; Martin *et al.*, 1993).

	Cu, µg/L	Zn, µg/L	Ni, µg/L	Pb, µg/L	Fe, µg/L	Hg, ng/L	Cd, ng/L
Ob	2.12 (1.85-2.43)	0.30 (<0.1-0.43)	1.32 (1.24-1.41)	0.014 (0.011-0.017)	25.8	0.56 (0.48-0.64)	0.70 (0.56-0.90)
Yenisey	1.62 (1.41-1.85)	1.32 (0.51-2.0)	0.54 (0.52-0.55)	0.0055 (0.005-0.006)	14.3	0.30 (0.16-0.42)	1.5 (1.2-1.8)
Lena	0.81 (0.47-1.0)	0.42 (0.08-1.37)	0.30 (0.23-0.38)	0.035 (0.015-0.083)	32.6 (21.6-48.5)	1.0 (0.9-1.08)	6.0 (2.2-12.0)
Eurasian Arctic rivers (mean)	1.48	0.74	0.67	0.018	23.5	0.60	2.8
Global mean	1.5	0.6	0.6	0.03	40	0.68	10

Program on Arctic and Siberian Aquatorium' to be reliable (Table 3·3). The results differed little from the global river water averages for Cu, Zn, Ni, Pb, and Hg. Cadmium was lower by a factor of about three and Fe by a factor of about two in Arctic Eurasian river water. There were no such studies on heavy metal concentrations in river waters of the North American Arctic.

3.4.3. Riverine transport of suspended particulate matter

Many contaminants, and particularly heavy metals, are concentrated on fine particulate matter and are transported by rivers as suspended sediment. A recent pan-Arctic synthesis addressing sediment flux from large rivers to the Arctic Ocean and coastal seas prepared by Holmes *et al.* (2002) includes critical analyses of published data and some unpublished material. Results provide estimates of contemporary average annual suspended matter flux in the eight largest Arctic rivers (Table 3·4). Stein and Macdonald (2003) have also compiled information about the sources of particulates and organic carbon to and within the Arctic Ocean. They conclude that, although rivers are important conduits, for many of the shelves coastal erosion rivals and sometimes exceeds the riverine supply.

The rivers of the Norwegian and Barents Seas (up to the border between Norway and Russia) deliver little suspended sediment to the Arctic Ocean – 0.26 thousand t/yr (Table 3·9 in AMAP, 1998). Gordeev (2000) and Gordeev *et al.* (1996) estimated the total sediment flux in Russian Arctic rivers to be 115 million t/yr; after corrections by Holmes *et al.* (2002) this is slightly reduced to 102.2 million t/yr. Sediments entering the Arctic Ocean from Canada are mainly from the Mackenzie River: 124 million t/yr (Table 3·4; Holmes *et al.*, 2002; Macdonald *et al.*, 1998). The Mackenzie River alone transports almost the same mass of sediments as the other Eurasian rivers together. Other rivers in Canada, including the Coppermine, Back, Hayes, Churchill, Nelson, Moose, and Nottaway, together deliver an additional 0.025 million t/yr. The total sediment discharge of all rivers flowing into the Arctic Ocean for which there are data is estimated at 227.3 million t/yr. The supply due to coastal erosion has not been quantified, but is likely to be significant.

Table 3·4. Estimates of total sediment flux for Arctic rivers. Values are accompanied by a qualitative assessment of confidence in the estimates as well as their source (Holmes *et al.*, 2002).

	Best estimate of sediment flux, 10^6 t/yr	Confidence in estimate	Source/reference
Sev. Dvina	4.1	Fair	This work
Pechora	9.4	Poor	This work
Ob	15.8	Good	This work
Yenisey	4.7	Fair	This work
Lena	20.7	Good	This work
Kolyma	10.1	Fair	Ivanov and Piskun, 1999
Mackenzie	124	Good	This work
Yukon	54	Fair	Brabets *et al.*, 2000

3.4.4. Heavy metals on particulates in rivers

Over the past few years, there have been several publications on heavy metals in suspended matter from the largest Arctic rivers (e.g., Rachold, 1999; Rachold *et al.*, 1996). The discharge-weighted average heavy metal concentrations in suspended matter were calculated on the basis of these studies (Gordeev, 2001; Table 3·5). Arctic river heavy metal concentrations were slightly lower but similar to other global river concentrations for Cu, Zn, Ni, Pb, Mo, antimony (Sb), and vanadium (V). Levels of As, Bi, and Sb in suspended sediments in the Yana River are thought to derive from the granitic intrusions forming the source of the Au and Sn ore deposits in the Yana basin. Levels of Co, Cu, Ni, V, and Fe in suspended matter of the Khatanga River are from basaltic rocks of the Siberian trap rocks in its basin (Rachold, 1999).

Average concentrations of heavy metals in suspended matter in the Lena River do not differ significantly from the global mean (Table 3·5). However, heavy metal distributions along the course of the river (Figure 3·25) peak in two places; near the Aldan river confluence (below Yakutsk) and in the lower reaches of the river (below Kjusjur), with concentrations as high as 338 mg/kg for Cu, 1020 mg/kg for Zn, 6.65 mg/kg for Cd, 1539 mg/kg for Pb, and 1083 mg/kg for tin (Sn). Rachold (1999) shows a positive correlation between carbon content and Cu, Zn, and Cd concentration (Figure 3·25), concluding that concentrations of these metals in the particulate material were linked to absorption by organic material. High concentrations of Pb, Sn, bismuth (Bi), and Sb, however, could not be fully explained and may be related to local contamination from towns. High

Table 3·5. Suspended particulate heavy metal and iron concentrations in the biggest Russian Arctic rivers in comparison with global mean values (mg/kg, Fe in %).

	No of samples	Cu	Zn	Ni	Pb	Hg*	Cd	As	Bi	Co	Mo	Sb	Sn	V	Fe	Reference
Lena	2	28	143	31	23	0.12										1
	6	28	160	34	36		0.25			13					3.32	2
	5	42	185	42	42		0.96								4.84	3
	31	35	141	53	24		0.65	9.1	0.24	18	1.23	0.57	1.9	97	3.80	4
Ob	6	50	104	38	16	0.05	0.53			19					6.0	5
Yenisey	6	144	220	77	30	0.05	2.2			23					5.8	5
Khatanga	12	82	104	84	12		0.22	9.3	0.13	35	0.88	0.43	1.6	349	6.12	4
Yana	7	30	130	39	23		0.32	26.7	0.23	17	0.96	2.1	1.5	110	3.80	4
Eurasian Arctic rivers (mean)	75	55	138	47	22	0.07	0.63	11.8	0.23	19	1.1	0.8	1.8	118	5.0	6
Global mean		80	250	84	35	0.06	0.7	5	–	20	3.0	2.0	–	130	5.1	7

* all mercury data from Cossa *et al.*, 1996.

References

1. Martin *et al.*, 1993; 2. Gordeev and Shevchenko, 1995; 3. Nolting *et al.*, 1996; 4. Rachold, 1999; 5. Gordeev *et al.*, 1995; 6. Gordeev, 2001 (with corrections); 7. Martin and Gordeev, 1986 (with corrections).

local concentrations of heavy metals are quickly suppressed by the huge natural flux of suspended matter in the river (Rachold, 1999).

These new data on heavy metal concentrations in water and suspended sediment from the lower courses of the rivers do not directly indicate the existence of any significant anthropogenic impact at the regional scale. However, the reasons for high concentrations in the Lena River near the Aldan River confluence remain unresolved.

3.4.5. Estuarine marginal filters

The role of estuaries as marginal filters (MFs) for riverine transport of heavy metals was discussed in the first AMAP assessment (AMAP, 1998). Estuaries were found to play a special role in the fate of riverine transported sediments, organic matter, heavy metals, and other contaminants. Estuaries are more than mixing zones for fresh and marine waters. They are ecosystems of high productivity that trap and accumulate up to 90% of material transported by rivers and provide an environment for an interplay of physical, chemical, and biological processes that can lead to dramatic transformation of riverine material en route to the deep ocean (Lisitzyn, 1995, 1999; Lisitzyn *et al.*, 1994).

3.4.5.1. Features of Arctic marginal filters

Marginal filters in Arctic estuaries are influenced by the geographical, climatic, and geological conditions of the region. Watersheds of the huge Siberian rivers flowing to the Arctic Ocean are characterized by a thin weathering crust which results in relatively low discharge of suspended sediments and low wash of sedimentary material from associated tundra and taiga surfaces. In addition, sharp seasonal variability in sediment fluxes occurs between the winter regime (200 to 300 days in the year),

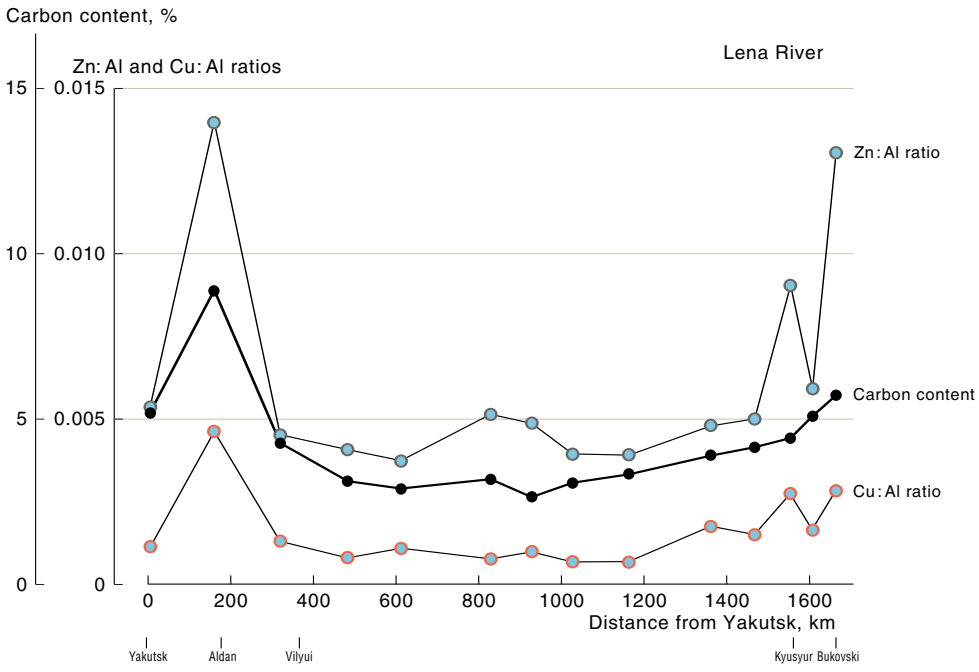


Figure 3·25. Aluminum-normalized copper and zinc concentrations, and carbon content of particulate material in the Lena River. The Aldan and Viljuy rivers are the main tributaries (Rachold *et al.*, 1996).

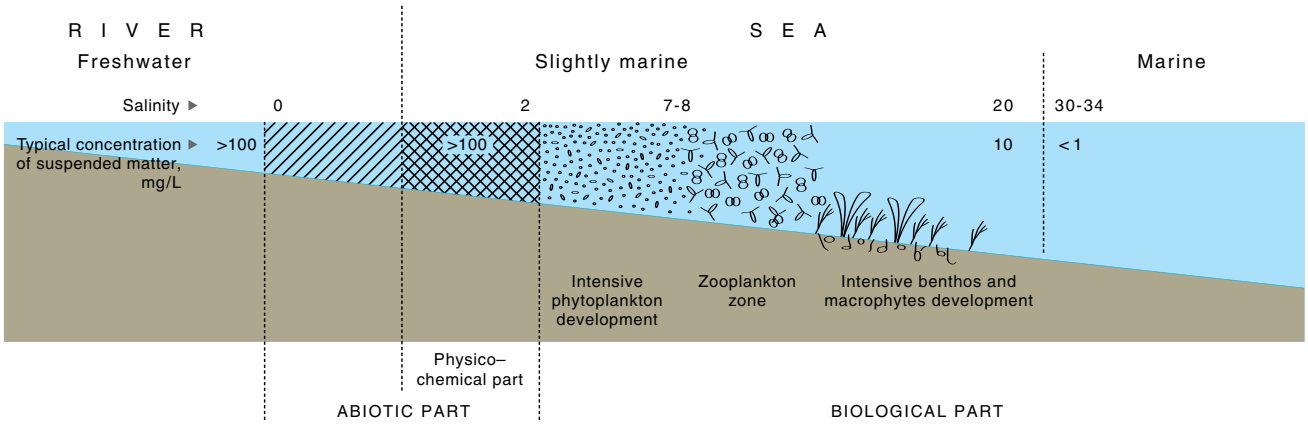


Figure 3·26. Schematic illustration of an estuarine marginal filter.

and the much shorter summer regime when the MF is most intense. During winter, ice cover protects the surface water layer from wind and allows freshwater to penetrate under the ice for long distances. As such, the MF winter regime is characterized by low river runoff, low concentration of suspended material, long distance spreading of the surface layer of freshwater, and inactivity of the biotic MF. However, during spring snowmelt, a pulse of sediment is transported by rivers to the estuarine MF where biotic and abiotic processes influence sediment distribution (see Section 3.4.5.2). Then, as winter begins and the seawater starts to freeze, heavy saline waters form and flow along deep bottom channels spreading into the open ocean. This facilitates the movement of sediments and contaminants to the deep ocean. Thus, surface ice, which forms in estuarine zones, provides an effective mechanism for transporting sediment and contaminants to the North Atlantic and other parts of the Arctic and subarctic (Lisitzyn, 1999). These processes differentially influence the distribution of Hg, Pb, and Cd.

3.4.5.2. Influence of the marginal filter on mercury, lead, and cadmium

The distribution of dissolved and particulate Hg in estuarine mixing zones is similar even in very different watersheds (e.g., the Ob, Yenisey, and Lena rivers in Russia). In the Lena Estuary, Hg concentrations increase from five pM to more than 13 pM for salinities of 7 to 20. Mercury concentrations in particulate matter are lower in river waters and higher in the mixing zones. For Cd, trends in the Ob and Yenisey estuaries appear complicated and scattered. A well recognized desorption of particulate Cd is observed for salinities of 0 to 10, which then declines until a salinity of 25. This increases again in high-salinity bottom water samples, perhaps associated with an additional benthic source of Cd, or desorption from re-suspended sediments. In the Lena River mixing zone, Cd concentrations reach a maximum of about 0.2 nM for salinities of 15 to 20. Dissolved Pb is also removed in the Ob and Yenisey estuaries. The proportion of colloidal Pb in the dissolved fraction decreases along the salinity gradient. Distribution of dissolved Pb in the Lena mixing zone is scattered, probably due to local contamination. Guieu *et al.* (1996) found no significant variations in Pb concentration.

3.4.5.3. The abiotic and biotic components of estuarine marginal filters

A conceptual model of the estuarine MF is shown in Figure 3·26 and comprises abiotic (terrigenous) processes close to the river mouth and biotic processes further seaward.

The abiotic processes of the MF include three sequential steps: 1) coarse-grain processes, 2) fine-grain processes, and 3) physico-chemical processes (i.e., the colloidal pump). Coarse-grain processes occur at the river mouth and adjacent parts of the estuary. Inert grains coated with adsorbed material, retain some dissolved elements, including heavy metals, and represent suspended matter that settles to the bottom. Arctic MFs deposit about 50% of the total particulate fluvial discharge through this process (Lisitzyn, 1999). The fine-grain processes largely concern clay minerals with extremely high surface areas and a high capacity for adsorption due to a coating of organic substances and Fe hydroxides. Under the effect of an electrolyte (e.g., seawater) the finest particles coagulate. With extensive coagulation a 'silt plug' is formed which contains higher concentrations of suspended minerals than the river water. For example, in the outer parts of the Ob and Yenisey estuaries and in the adjacent Kara Sea, vertical daily fluxes of suspended matter were 1 to 20 mg/m^2, while daily fluxes near silt plugs ranged from 1300 mg/m^2 in the Ob River to 22 100 mg/m^2 in the Yenisey River, which was several orders of magnitude greater than on the adjacent shelf (Lisitzyn *et al.*, 1994). Physico-chemical processes occur within a salinity range of 3 to 20. Dissolved humic substances and iron hydroxides form flocs that are powerful sorbents for many dissolved elements, including heavy metals. Organo-mineral colloids strongly influence the fate of heavy metals. For example, in the Ob and Yenisey rivers, colloidal forms were detected in up to 60 to 80% of dissolved organic carbon, 80 to 97% of Fe, 50 to 70% of Ni, Cd, Zn, Mn, and 20 to 50% of Pb (Dai and Martin, 1995). It is the colloidal pump that transfers elements from solution into particulate form. In this respect it is important to note that the sorbents are not released from terrigenous material but arise from solution in the MF. Also, that dissolved organic carbon is essential as it is involved in the formation of organo-metallic compounds (including helates) where the chemical proper-

Table 3·6. Gross river flux (10^3 t/yr) of dissolved and particulate heavy metals and iron to the Eurasian Arctic (Gordeev, 2001).

	Cu	Zn	Ni	Pb	Hg	Cd	Fe
Ob							
dissolved	0.85	0.13	0.56	0.006	0.0002	0.0003	11
particulate	0.84	1.71	0.63	0.26	0.0008	0.0087	990
total	1.69	1.84	1.19	0.266	0.001	0.009	1001
Yenisey							
dissolved	1.0	0.82	0.34	0.003	0.0002	0.0009	9
particulate	0.85	1.3	0.45	0.177	0.0003	0.013	342
total	1.85	2.12	0.79	0.18	0.0005	0.0139	351
Lena							
dissolved	0.47	0.22	0.16	0.018	0.0005	0.003	17
particulate	0.6	2.58	0.84	0.485	0.0021	0.011	680
total	1.07	2.80	1.0	0.503	0.0026	0.014	697
Whole Eurasian Arctic							
dissolved	4.40	2.2	1.99	0.052	0.0018	0.0081	69
particulate	6.37	15.85	5.48	2.6	0.0084	0.078	5607
total	10.77	18.05	7.47	2.652	0.0102	0.0861	5676

ties of metals are not active until released from the compound.

The biotic processes include two components: 1) phytoplankton, and 2) zooplankton. The role of phytoplankton in the MF is to assimilate and transform dissolved elements into suspended particles (i.e., the bodies and carapace/frustules of bacterio- and phytoplankton). This is known as 'the phytoplankton pump'. The phytoplankton pump captures only dissolved elements and the remains of colloidal forms of elements, decreasing the role of mineral riverine suspended and colloidal particulate matter. This represents the primary difference between the abiotic and biotic zones of the MF. The zooplankton filtration mechanism depends on biomass. The filtration is not selective, both biotic and terrigenous particles are collected from the water and transferred to bottom sediments. In the Kara Sea MF, the zone comprises copepods (50 to 80% of total biomass), ostracods, appendicularians, rotifers, and larger crustaceans and euphausids. These filter almost the entire range of suspended particles, from coarse (biotic) to fine particles (<1.0 μm). To meet food requirements, zooplankton in the Ob and Yenisey estuaries appear to filter the entire water volume and to remove the majority of the phytoplankton and suspended matter within only one to two days.

Figure 3·26 highlights the important processes occurring in the transition zone between river and sea. These result in a massive transfer of sedimentary matter (suspended particulate, colloidal, and dissolved) to bottom sediments.

3.4.6. Gross and net fluxes: rivers and marginal filters

Recent data on heavy metal concentrations in Russian Arctic rivers were used to calculate new gross (Table 3·6) and net river fluxes. The gross river flux to sea basins is defined as the amount of substance in dissolved and suspended form transported by the river to the sea boundary. The net river flux is defined as the amount of substance transported across the river–sea boundary (GESAMP, 1987). This boundary is defined as the river cross-section where flow direction is unidirectional and there is no marine influence.

Although the results in Table 3·6 must be considered preliminary (owing to the small number of water samples analyzed and insufficient data on seasonal variations), the table shows suspended matter to be the most prevalent form for all metals, especially for Pb and Fe (with >98% in particulate form).

The behavior of dissolved heavy metals in the mixing zones of the Ob, Yenisey, and Lena rivers was used to calculate net fluxes (Cossa *et al.*, 1996; Dai and Martin, 1995; Guieu *et al.*, 1996; Martin *et al.*, 1993). The net fluxes of dissolved Cu and Ni are slightly higher than the gross fluxes. For dissolved Fe, the net flux is about 25% of the gross flux. Net and gross fluxes of Pb and Hg are roughly equal.

The relative contribution of rivers to the input of heavy metals to the Arctic Ocean varies by metal. Akeredolu *et al.* (1994) estimated atmospheric deposition of selected heavy metals to the Arctic region from emissions in the Eurasian basin, a region contributing >90% of the atmospheric deposition of heavy metals from emissions in the Arctic. Table 3·7 compares their estimates with those for riverine inputs by Gordeev (2001).

Table 3·7. Comparison of heavy metal fluxes (t/yr) to the Arctic via atmospheric deposition (Akeredolu *et al.*, 1994) and three Russian rivers (Gordeev, 2001).

	Hg	Cd	Pb	Zn
Atmospheric deposition	40	47	2400	1350
Riverine transport	10	66	2450	17800
Air/water	4	0.7	0.98	0.08

Based on current results, atmospheric transport is the main pathway to the Arctic for Hg, while rivers provide the main transport mechanism for Zn. Atmospheric and riverine transport are equally important for Pb and Cd. However, these conclusions require further verification and must be evaluated in comparison with marine transport mechanisms (see Section 3.7).

3.4.7. Transport by ice

River ice is thought to be an important source of contaminants to the Arctic seas (Melnikov, 1991). However, there are almost no data on heavy metal concentrations

Table 3·8. Dissolved (< 0.4 µm) heavy metal concentrations (nmol/kg) in newly formed ice and water of the Lena river. Ice concentrations are median values (Hölemann *et al.*, 1999).

	Fe	Mn	Cd	Zn	Pb	Cu	Ni
River water (Martin *et al.*, 1993)	410	-	0.037- 0.07	5.4	0.08	9.4	5.1
River water (Transdrift III, Station 41H)	1494	99.3	-	15.4	0.51	11.9	5.9
Water column at Station 291 (north of the Tumatskaya branch; 2 m, salinity 29.1)	428	52.0	-	9.7	0.23	6.5	8.2
Ice (stations 291 and 292)	10 855 5497 (n = 9)	706 317 (n = 9)	1.09 0.67 (n = 7)	479.1 353 (n = 9)	3.08 1.91 (n = 9)	17.03 11.3 (n = 9)	10.94 7.3 (n = 9)
Ice (all other stations)	281 143 (n = 10)	51.9 23 (n = 10)	0.44 0.08 (n = 6)	47.7 28 (n = 10)	0.20 0.06 (n = 10)	4.93 1.22 (n = 10)	4.56 2.38 (n = 10)

in river ice. More is known about sea-ice entrainment and contaminant transport (Hölemann, 2001; Hölemann *et al.*, 1997; Pfirman *et al.*, 1995), although data are still sparse (Olsson, 1999).

The 'Laptev Sea System' Russian-German Program investigations of newly formed ice provided the first estimates of off-shelf transport of ice-rafted sediments (Hölemann *et al.*, 1999). It was estimated that 4 million t/yr of sediment were entrained in the sea-ice-pack (Eicken *et al.*, 1996) representing about 20% of the total sediment flux to the Laptev Sea by the Lena River (20.7 million t/yr). It is important to note that sediment exported by ice entrainment in suspension freezing is probably not the same as the river export. Sediments may be mined over the central shelf and shipped to the Arctic interior or the Greenland Sea, while river input and coastal erosion support sediment accumulation on the inner shelf (Eicken *et al.*, 2000). Analyses of dissolved metals in newly formed ice at two marine sites near the Lena delta, and other stations in the Laptev Sea indicate that new ice formed north of the Lena delta (Tumatskaya branch) was highly enriched in Mn, Fe, Zn, Cd, and Pb compared to average concentrations in Lena River water (see Table 3·8). Dissolved metal concentrations in young ice are thought to result from re-mobilization of metals from the particulate phase in ice. The median concentrations of metals in ice-bound particles were comparable, however, with those in unpolluted river and marine sediments.

Hölemann (2001) highlights the importance of near-shore sediments becoming entrained in or on newly formed sea ice. Off-shelf transport of ice-rafted sediments depends on the degree of sediment entrainment when ice forms in shallow water during autumn freeze-up, on ice growth over the course of the winter, and on the volume of ice exported beyond the limit of the minimum summer ice extent. Results suggest that near-shore new and young ice containing riverine particulate trace elements should have a moderate effect on the river-to-sea transport of heavy metals within the Arctic.

3.5. Groundwater transport

Groundwater is often a recipient of contaminants entering terrestrial and aquatic environments. While little is known about this transport pathway, sections 3.5.1 and 3.5.2 report the outcome of recent investigations on transport processes and heavy metal levels.

3.5.1. Transport processes

During the mid-1980s, investigations were undertaken into regional submarine inflow into the inner seas of the former U.S.S.R. and the world's oceans, complementing previous work by Kudelin (1975) and Zektser *et al.* (1984). Global groundwater discharge (2500 km^3/yr) was found to be about 8% of global river discharge. However, dissolved salts discharged by groundwater represented more than 50% of the global discharge of dissolved salts in river runoff.

Gordeev *et al.* (1999) assessed the relationship between river and groundwater sources. Groundwater discharge was estimated using the groundwater discharge module from Kudelin (1975). Watershed areas were adjusted based on gauging station locations. Calculations show that the average ratio between groundwater and river discharge to the Russian Arctic seas is about 1:10 (see Table 3·9).

In the absence of heavy metal data, data on the input of inorganic nutrients via groundwater provides an insight into transport processes for this pathway (Gordeev *et al.*, 1999). Results indicate that groundwater contributions decreased from 15 to 7% from west to east, and that nutrient species attributable to groundwater also decreased from west to east: NH_4: 20 to 5%; NO_3: 22 to 3%; and PO_4: 15 to 6%. There was a concurrent decrease in nutrient concentrations from west to east in river waters. This gradient is thought to be linked to regional climatic conditions and the high numbers of wetlands in the western Kara Sea watershed compared to wide expanses of permafrost in the east, and the higher

Table 3·9. Groundwater and river discharges (km^3/yr) for rivers and their watersheds discharging into the Arctic Ocean, and groundwater discharge as a percent of river water discharge.

	Groundwater	River	%
Severnaya Dvina	27.8	110	25.3
Pechora	5	130	4
White and Barents Seas	68	463	14.7
Ob	76	429	17.7
Yenisey	39	620	6.3
Kara Sea	152	1478	10.3
Lena	37	525	7
Laptev Sea	57	745	7.7
Indigirka	4.6	61	7.5
Kolyma	8.5	132	6.4
East-Siberian Sea	16.9	250	6.8
Eurasian Arctic (excluding Chukchi Sea)	294	2936	10

anthropogenic impact in the western part of the Russian Arctic. It would be reasonable to expect heavy metals in groundwater to show a similar spatial distribution but confirmation will require new data.

3.5.2. Contaminants in groundwater

No information is available on heavy metal concentrations in groundwaters flowing into Arctic rivers or the Arctic Ocean. Finnish data on the levels of dissolved constituents and heavy metals in wells and shallow groundwater discharged to the surface via seepage, springs, and headwater streams are used as a frame of reference only.

In Finland, typical concentrations of total dissolved solids range from 20 to 100 mg/L depending on the granulometric and lithological composition of aquifer material and overlying bedrock (Lahermo *et al.*, 1990, 1996; Tarvainen *et al.*, 2001). Shallow groundwater pumped from dug wells and wells drilled in bedrock generally contains higher levels (e.g., from 50 to 200 mg/L). Levels of most trace metals (Zn, Cu, Co, Ni, Cr, V, Mo, Cd, U, Th, Hg, Ag) and other trace elements (Sr, Ba, Li, Rb, B, As, Sb, Bi, Tl, Se) are also very low in well and spring waters. Lead concentrations are < 0.15 µg/L. In the few cases where elevated concentrations do occur (0.5 to 5.0 µg/L) this probably reflects contamination from well pumps, water pipes, and other fittings (Tarvainen *et al.*, 2001). Cadmium also occurs at very low concentrations in natural waters with low mineralization; concentrations in spring and well waters are mostly < 0.02 µg/L with few cases exceeding 0.5 µg/L. There is little information on Hg levels in intact shallow groundwater. Recent studies show that almost all Hg in well waters is < 0.01 g/L. In a region with Hg-bearing black shale, concentrations in stream water ranged from 0.8 to 4 ng/L, levels probably related to geochemically associated shallow groundwater discharged from springs (Loukola-Ruskeeniemi *et al.*, 2003).

There are seasonal fluctuations in spring and well water quality depending on the groundwater regime, depth, type of aquifer, and hydrometeorological conditions. This is particularly the case for heavy metals and other trace elements occurring naturally at low concentrations (Backman *et al.*, 1998). Variation amplitudes exceeding an order of magnitude are common.

3.6. Contaminants in frozen ground

Specific sites throughout the Arctic have been contaminated by the planned, accidental, and haphazard disposal of waste materials by military, municipal, and other bodies. Waste disposal sites vary, for example, unlabeled barrels, rusted and leaking along the shore of an unnamed lake; abandoned military installations; and active landfill operations that may or may not include containment. However, a common feature of most waste sites is their location on permanent or semi-permanent frozen ground or permafrost (Macdonald *et al.*, 2003). Contaminants that leak from these sites percolate into the overlying soils and contaminate groundwater to the depth of the permafrost. Drainage follows typical soil pathways, and may eventually enter surface systems including streams, lakes, rivers, and the ocean. Vertical movement of contaminants in groundwater is constrained by frozen ground, but facilitated by macropores and fractures in the surface soil and permafrost. There are no summary estimates of contaminant flux from terrestrial dumpsites to drainage systems. Only a few specific sites have been fully examined as part of remediation activities.

It is often assumed that permafrost and long periods of freezing temperature combine to limit the mobility of contaminants deposited on seasonally unfrozen ground. This situation may change rapidly as global change affects Arctic temperatures and the extent and depth of permafrost. Under warming scenarios, contaminants in groundwater and permafrost will disperse throughout the three-dimensional soil environment, with the area and speed of dispersal increasing as permafrost melts. Permafrost melt is likely to affect river hydrology with potential impacts on contaminant transport to lakes and rivers, and perhaps eventually the Arctic Ocean.

3.7. Marine pathways

Few studies have been conducted on marine pathways in the Arctic since the first AMAP assessment. However, there is increasing recognition that the marine environment is an important transport pathway, particularly for some heavy metals. Available data are based on research evaluating the broad features of the marine geochemistry of Hg, Pb, and Cd, sufficient to understand their sources, sinks, and cycling within world oceans (Boyle *et al.*, 1976; Lamborg *et al.*, 2002; Patterson and Settle, 1987). Geochemical profiles for these metals in the Arctic Ocean are limited and mostly based on old data (Campbell and Yeats, 1982; Coquery *et al.*, 1995; Mart, 1983; Moore, 1981; Pohl *et al.*, 1993; Yeats, 1988; Yeats and Westerlund, 1991). Despite limitations in data availability, some important relationships for Hg, Pb, and Cd can be identified and may provide a focus for future studies.

3.7.1. Mercury

The marine environment provides a significant reservoir of global Hg. For example, Lamborg *et al.* (2002) estimate that the top 100 m of the Earth's oceans contains twice as much Hg (11000 t) as the atmosphere. Thus, any global Hg cycle must include processes that enrich and remove Hg from oceanic waters, as well as mechanisms that transport Hg around the globe, including to the Arctic. The removal of Hg from the ocean mainly occurs through gaseous evasion of GEM to the atmosphere, and vertical particle flux. Both are restricted in Arctic marine waters by the permanent ice cap. This may result in higher Hg concentrations in the Arctic compared to oceans not covered by ice. In addition, unique processes at polar sunrise in the Arctic atmosphere help to facilitate Hg deposition to Arctic surfaces (e.g., sea ice) in spring (see Section 3.1.3). Currently, little is known about how MDEs might influence oceanic pathways for Hg. Studies of the Hg cycle in the Arctic Ocean are needed to determine how Hg is accumulating in marine waters. Additional work is needed to establish the links between atmospheric processes, upper ocean enrichment, oceanic transport, and exposure of marine biota to Hg.

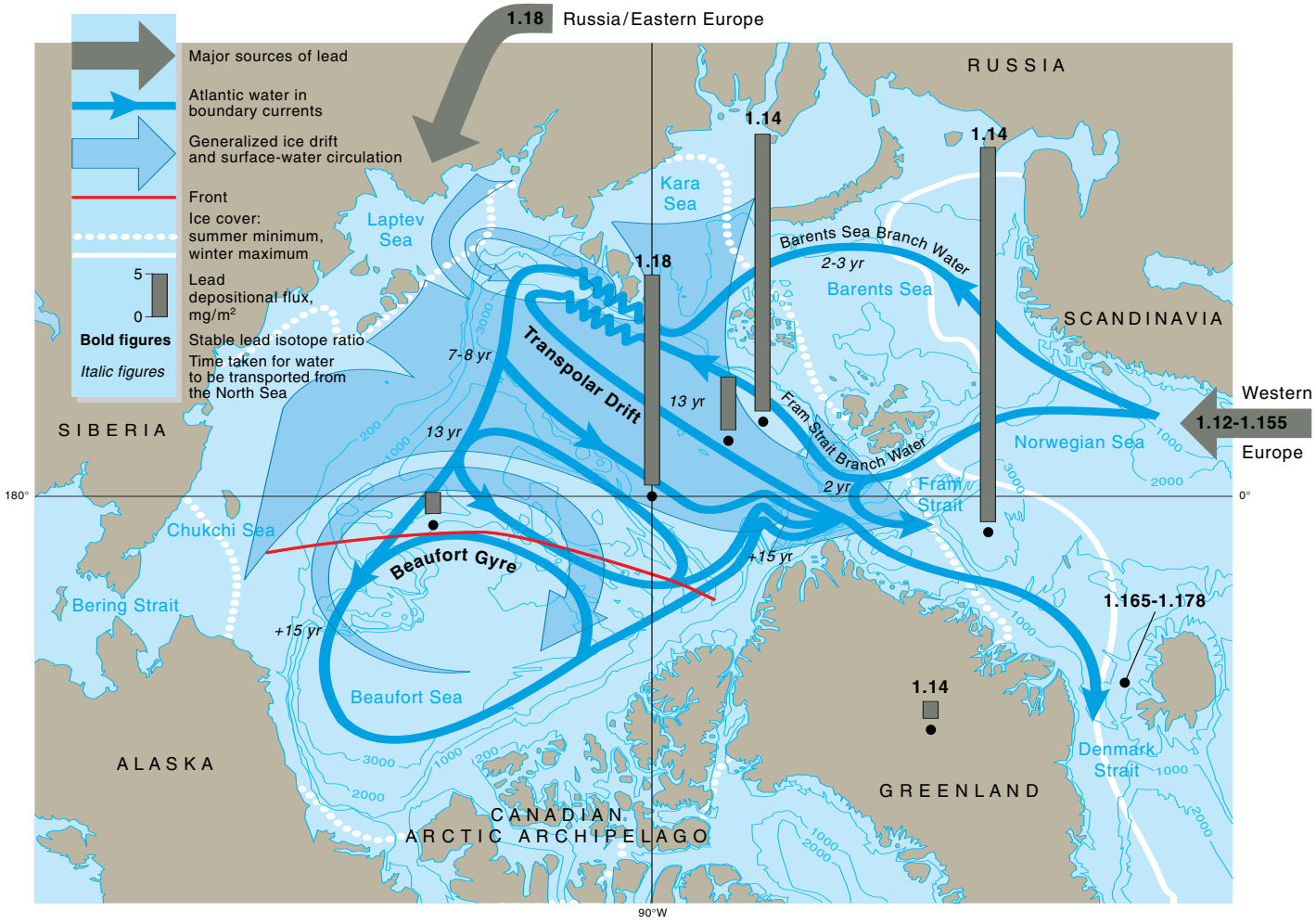


Figure 3·27. The major sources of contaminant lead transported to the Arctic are indicated by the stable lead isotope composition of sediment cores as being either Western Europe (indicated by a $^{206}Pb : ^{207}Pb$ ratio of 1.14) or Eurasia (indicated by a $^{206}Pb : ^{207}Pb$ ratio of 1.18) (adapted from Gobeil *et al.*, 2001).

3.7.2. Lead

Anthropogenic Pb is rapidly transported to the Arctic via the atmosphere (Boutron *et al.*, 1991; Rosman *et al.*, 1994, 1997; Sirois and Barrie, 1999). However, new work suggests that the ocean may provide a central role for Pb transport and distribution along the Eurasian Basin boundaries. Gobeil *et al.* (2001) examined contaminant Pb inventories and their isotopic composition in sediment cores collected from all major Arctic Ocean basins, in areas of minimum and maximum ice cover and in relation to riverine inputs and atmospheric transport pathways (Figure 3·27). High sediment inventories were found in the Greenland Sea and along the southern boundary of the Eurasian Basin. The isotopic composition of the contaminant Pb ($^{206}Pb : ^{207}Pb$ = 1.14) suggested a western European source. Gobeil *et al.* (2001) proposed that Pb may enter marine waters off Europe via rivers and the atmosphere, following the same transport pathways as radionuclides (^{137}Cs, ^{129}I) released by European reprocessing plants (AMAP, 2003b).

Lead is believed to be scavenged from ocean waters during transport northward to the Greenland Sea, Barents Sea, and along the Eurasian Basin margin that constrains inflowing water from the East Spitzbergen current (e.g., Rudels *et al.*, 1999). A relatively large contaminant Pb inventory in sediments at the North Pole (along the Eurasian flank of the Lomonosov Ridge) also suggests that boundary currents or ice and water moving with the Transpolar Drift are transporting Pb, but the stable Pb isotopic composition ($^{206}Pb : ^{207}Pb$ = 1.18) indicates that Pb from eastern European or Russian sources is probably entering the Arctic Ocean via the Laptev Sea. The isotopic composition of Pb in waters off Iceland (Véron *et al.*, 1999) suggests that some of the Eurasian Pb is transported across the Arctic Ocean and exported into the Greenland Sea, consistent with the relatively short time scale involved (~2 yr). Most of the contaminant Pb in Arctic Ocean basin sediments, estimated at 9000 to 48 000 t, appears to have been deposited into the Eurasian Basin. Boundary current structure and speed, and the residence time of Pb in the upper ocean, appear to restrict the entry of contaminant Pb into the Canadian Basin.

3.7.3. Cadmium

Cadmium in the upper ocean undergoes biological cycling mirroring the phosphorus cycle. Cadmium depletion occurs near the surface where primary production extracts nutrients and incorporates Cd. Cadmium enrichment occurs below the surface mixed layer where regeneration of nutrients and associated Cd occurs (Bruland and Franks, 1983; de Baar *et al.*, 1994; Moore,

1981). As a result, the residence time of Cd in the ocean (~15 000 yr; Boyle *et al.*, 1976; Yeats and Bewers, 1987) is considerably longer than the 1000 to 2000 years required for the Global Thermohaline Circulation to transport water from the North Atlantic to the North Pacific Ocean. The natural oceanic geochemistry of Cd results in higher Cd concentrations in the North Pacific, by a factor of about five, compared to the North Atlantic (Bruland and Franks, 1983). Macdonald *et al.* (2000) constructed a Cd budget for the Arctic Ocean that reflects the importance of natural ocean processes. Their budget is revised in this assessment to include improved estimates of riverine input and atmospheric deposition (see Figure 3·28). Results suggest that most Cd in biota is probably derived from natural oceanic geochemical processes.

The nutrient-rich water of the Pacific entering the upper Arctic Ocean provides the most important source of Cd in this budget. However, distribution of these waters to the Arctic Ocean differs by region, potentially influencing environmental enrichment. In the Canadian Archipelago and shelf margins along the East Siberian, Chukchi, and Beaufort Seas, upwelling and mixing from nutrient-rich waters at depths of 100 to 200 m may expose biota to maximum levels of Cd, supporting a food chain with higher Cd burdens. However, surface waters under the permanent ice cap of Canada Basin, although derived from the Pacific, are relatively nutrient-depleted and exhibit recycled production. These waters may support a food chain with low Cd burdens. Further research is needed to verify these relationships.

Atmospheric and riverine inputs are approximately the same, and small compared to other components of the budget. The minute component contributed by anthropogenic sources to the Cd budget implies that human activities only affect the ocean in regions close to industrial sources.

3.8. Potential impact of global change on marine pathways in the Arctic

The twentieth century has been the warmest in the Arctic for the past 400 years, reflecting a regionally integrated increase approaching 1°C. This trend is expected to continue and to amplify, resulting in an additional 1.4 to 5.8°C temperature increase by 2100 (IPCC, 1995, 1996; Showstack, 2001). Uncertainty about the physics of the atmosphere, the behavior of Arctic ecosystems and effects of contaminants, as well as their interactions, makes it challenging to predict how global change may influence future contaminant pathways and trends. However, transport pathways in the Arctic are likely to undergo significant alterations as a consequence and, if so, heavy metal transport will be affected. Section 3.8.1 provides highlights from the more comprehensive discussion by Macdonald *et al.* (2003) of the subtle connections between global and regional pathways that put the Arctic at risk from contaminants. These relationships are based on available information, but much remains unknown.

3.8.1. Arctic oscillation and environmental change

A major feature of environmental change in the Arctic is linked to the positive and negative phases of the Arctic Oscillation (AO) (Dickson *et al.*, 2000; Morison *et al.*, 2000; Serreze *et al.*, 2000). This is a mode of atmospheric variability that is potentially active over a broad

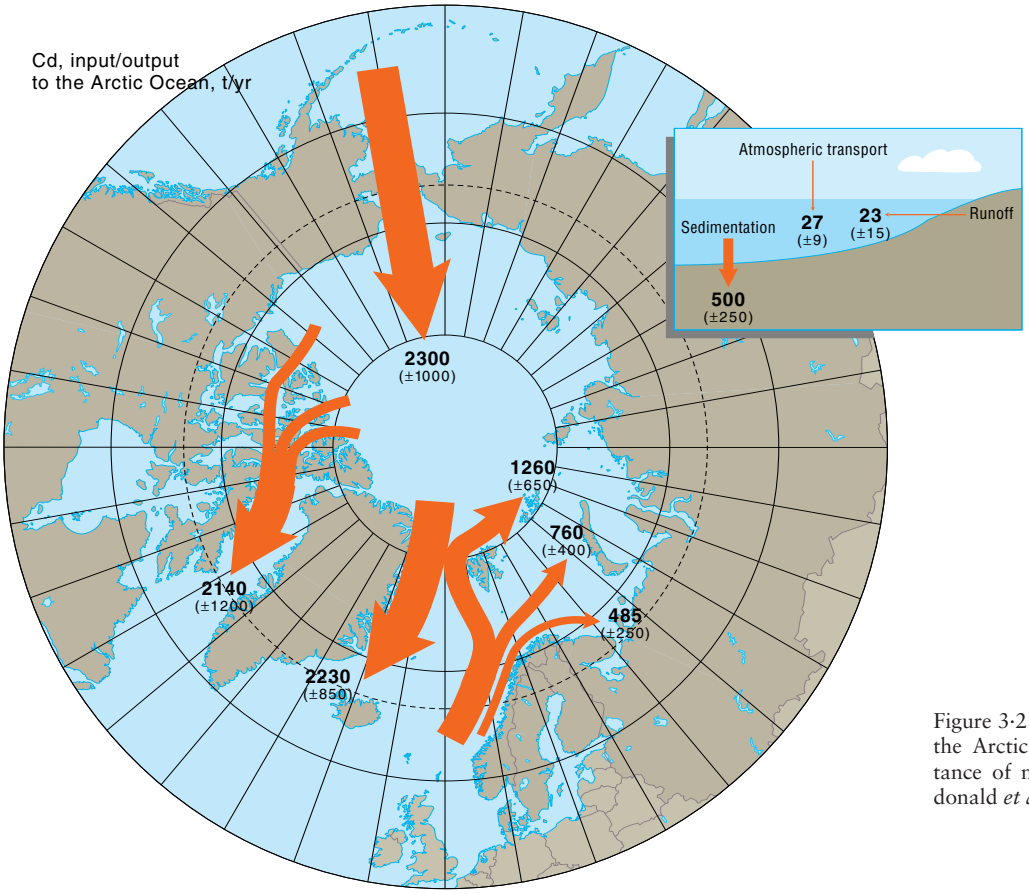


Figure 3·28. A cadmium budget (t/yr) for the Arctic Ocean, reflecting the importance of natural ocean processes (Macdonald *et al.*, 2000).

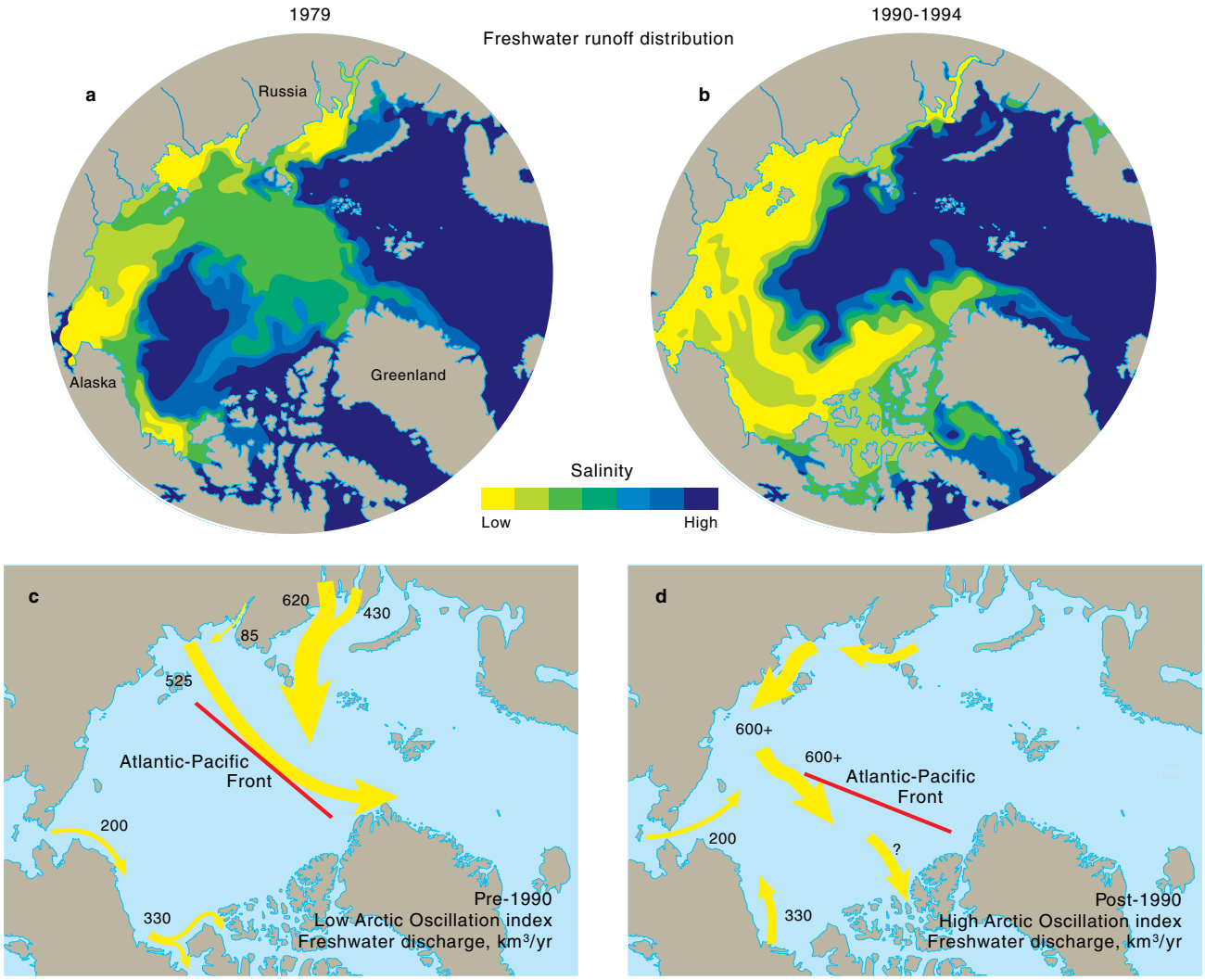


Figure 3·29. Transport of freshwater runoff across the Arctic Ocean. This figure illustrates a) freshwater pathways during pronounced AO⁻ conditions (1979); b) freshwater pathways during pronounced AO⁺ conditions (1990-94) (both a and b are based on model results by W. Maslowski reproduced in Dickson, 1999); c) the amounts and changes in pathways for freshwater inflows during AO⁻ conditions; and d) the amounts and changes in pathways for freshwater inflows during AO⁺ conditions.

range of time scales, including climatic time scales, and involves changes in the strength of the atmospheric polar vortex. This correlates strongly (85 to 95%) with the North Atlantic Oscillation. The AO directly influences wind speed and direction, precipitation, runoff, ice drift, ice cover, and ocean currents in the Arctic, all of which influence heavy metal transport pathways.

During the 1990s, under a high AO, climate changes influenced winter storms, increasing their strength and frequency. Poleward transport of heat and southerly air masses increased the atmospheric connection of the Arctic to industrial emissions in Europe, Asia, and North America. Increased precipitation provided a larger vector for contaminant delivery to the Arctic. Arctic ice cover and multi-year ice volume decreased during the 1990s. Accelerated permafrost declines observed during the 1990s were probably due, in part, to the high AO that moved warm air into the Russian Arctic. A strong increase in the AO, starting around 1989, resulted in an intensified flow of ocean currents from the Atlantic to the Arctic through Fram Strait and the Barents Sea. The temperature of Atlantic water entering the Arctic Ocean increased by 2.3 °C. Surface water inflow to the Russian shelves from major Russian rivers, historically flowing westward after entering the Kara and Laptev Seas, was diverted eastward and into the Canadian Basin. Eventually, this river discharge may leave the Arctic through the Canadian Archipelago (Figure 3·29). Changes in hydrology, currents, ice, winds, and temperature linked to the AO will influence the distribution of heavy metals within the upper ocean and the flux of heavy metals to and within the Arctic.

3.8.2. Consequences of global change for heavy metal pathways

Observed and predicted changes in forcing and transfer functions influencing heavy metal pathways, indicate the likelihood of significant change in biogeochemical cycles, food webs, and human activities in the Arctic. Together they will influence physical, biological, and human-mediated pathways.

Under warming conditions, natural processes may re-mobilize existing contamination sequestered in various Arctic 'sinks' (e.g., sea ice, glaciers, permafrost, and coastal sediments in oceans and lakes). Glaciers act as repositories for historic contaminant inputs and may release their load during episodes of accelerated melt back

(since about 1960, glacial melt back in the Canadian Arctic has been estimated at 800 km^3, possibly half the glacial melt back of the entire Arctic; Dyurgereov and Meier, 1997). Sea ice modulates the exchange of properties between atmosphere and ocean for all contaminants, and in some cases, air-sea exchange is a significant component of regional budgets. Warmer temperatures will affect permafrost depth and extent. This in turn, affects the hydrological cycle including particle and organic carbon cycles. Ocean and air currents, and patterns of precipitation, may shift dramatically, directly influencing pathways and the availability of heavy metals for transport.

Heavy metals entering biological systems could redistribute regionally. Warming will probably enhance the conversion of Hg to the highly toxic form MeHg, resulting in increased biotic uptake. Also, release of organic carbon from drainage basins due to permafrost destruction is likely to be accompanied by increased Hg inputs to lakes. Climate change may precipitate changes in the complexity of food webs, adding or subtracting trophic levels or altering the age-distribution of species. The effect of such changes on Hg may be comparable to, or even exceed the effect of change to physical pathways owing to the power of biomagnification.

Global change is likely to influence human activities in the Arctic and may encourage new activities that affect heavy metal contamination. Increased risk of storm damage to coastal infrastructure (e.g., pipelines, mines, communities) could enhance the release of heavy metals from erosion and flooding. Increases in shipping, tourism, commercial fishing, agriculture, industry, and oil exploration could introduce local and regional sources of contamination. Erosion, development, and commerce are likely to increase activities such as dredging, waste discharge, and sea disposal, adding to and remobilizing contaminants.

3.8.3. Potential change in heavy metal transport

Inherent differences in the characteristics of heavy metals and their transport pathways will result in different responses to forcing functions influenced by global change. Aerosol metals (predominantly Pb, but also Cd and Zn) are most likely to be influenced by changes in wet and dry removal processes (Section 3.3.2). Presently, the Arctic tends to be a poor trap for atmospheric contaminants, sequestering <10% of Pb emissions and <15% of Cd and Zn emissions (Akeredolu *et al.*, 1994; Pacyna and Keeler, 1995). There is potential for enhanced deposition of airborne contaminants through changes in precipitation or in the balance between rain and snow. Under high AO conditions, more efficient transfer of metals from Western Europe to the Arctic will probably occur via rainout in the air transport corridor to northwestern Europe or via coastal discharges to the North or Baltic Seas. The atmospheric corridor from eastern North America and Western Europe will become a more efficient trap for particulates, raining them out in the Nordic Seas and the southern Eurasian Basin.

Mercury is particularly important under global change scenarios as it is volatile, undergoes biological transformations, and bioaccumulates, potentially increasing risk to wildlife and humans. The natural Hg cycle has been enhanced by human activities and recent data suggest that two to three times as much Hg is now cycling through the atmosphere and upper oceans compared to pre-industrial times (Mason *et al.*, 1994; Mason and Fitzgerald, 1996; Pacyna and Keeler, 1995). However, lake sediments show considerable spatial variation concerning the increase in Hg fluxes to the Arctic during the past century or two, ranging from –0.3-fold to +7-fold increases (Landers *et al.*, 1998; Lockhart *et al.*, 1995); a range that may also reflect inaccuracies in the analysis of atmospheric Hg in sediments. In several cases of well-constrained, local Hg contamination, Lockhart *et al.* (2000c) have shown that Hg profiles in dated sediments faithfully record the contamination history. However, the case of global enhancement of the Hg cycle as proposed by Mason *et al.* (1994) may not be as reliably recorded in sediments. Post-industrial increases in Hg concentration in lake sediments could be due entirely to enhanced delivery from the atmospheric inputs from human activities, or may reflect, in whole or in part, alterations in Hg transfer functions in response to climate change (e.g., altered ice cover and lake coupling with the atmosphere, enhanced MDEs, drainage basin changes, and change in lake productivity or scavenging).

The relatively long residence time of gaseous Hg in the atmosphere (~2 yr) increases the probability of Hg transport to the Arctic. The polar sunrise phenomenon (Section 3.1) probably increases the vulnerability of the Arctic to Hg deposition. Revolatilization after deposition in snow and evasion of reduced forms of Hg also complicate the Hg cycle. The vulnerability of the Arctic to global Hg emissions in the future lies in the degree of balance between invasion and evasion processes in aquatic systems and the impact of global change on these pathways.

Chapter 4

Spatial Patterns

Jesse Ford, Hans Borg, Maria Dam and Frank Riget

The spatial distribution of heavy metals across the Arctic is related to local geology, natural processes, and anthropogenic activities. Factors influencing their distribution include the location, quantity, and timing of release from anthropogenic and natural sources, transport pathways, and the characteristics of the receptor compartments. The spatial distribution of heavy metals leading to biotic exposure directly influences their potential for environmental harm, and is an important issue for policy makers.

This chapter builds on the conclusions of the first AMAP assessment (listed in Section 4.1). The change in terminology from 'trends' to 'patterns' in this assessment acknowledges that contaminants distribute in space in non-linear gradients to form patterns of accumulation. The identification of areas of high concentration is a key step toward linking the sources of anthropogenic release of heavy metals, distribution pathways, and observed accumulation trends over time. The data reported here represent a building block for future assessments, providing an important compendium of information on the concentrations of heavy metals in an array of Arctic environmental compartments.

The inherent challenges associated with identifying spatial patterns are instructive and should direct future research (see Section 4.3). One of the key conclusions of this assessment is that the greatest opportunity for identifying spatial patterns in heavy metal accumulation in the Arctic may be by using plants and other media that are less mobile (e.g., ice, soils, sediments). Although there are limitations in the use of every environmental compartment, animal data are highly variable for a variety of reasons and patterns are difficult, at best, to detect. Nevertheless, animal data are critical to understanding how, when, and where different species are being impacted, and are particularly important for subsistence species.

Data are presented on the concentrations of mercury (Hg), lead (Pb), cadmium (Cd), and other heavy metals in terrestrial, freshwater, and marine environments. Where spatial patterns can be identified, they are discussed. Where spatial patterns cannot be detected, data are presented to inform future assessments. As more high quality data become available, the task of identifying spatial patterns in heavy metal accumulation will become more achievable.

4.1. Conclusions from the first AMAP assessment

The main conclusions of the first AMAP assessment (AMAP, 1998) on the spatial distribution of heavy metals are as follows.

Concentrations of heavy metals within environmental media were fairly uniform (within a factor of two or three) across the High Arctic, with exceptions in terrestrial regions near emission sources such as those on the Kola Peninsula.

No spatial patterns were found in fish species or in benthic flora or fauna. For stationary marine species, seasonal and local differences for metals such as Cd were often greater than regional differences in baseline data. Spatial relationships were principally found in marine ecosystems. Spatial trends were found for Cd in seabirds (highest levels in northwest Greenland and especially the Lancaster Sound area) and in ringed seal (*Phoca hispida*), beluga (*Delphinapterus leucas*), and polar bear (*Ursus maritimus*) (highest levels in northwest Greenland and the eastern Canadian Arctic). In contrast, Hg concentrations in these mammals were highest in the western Canadian Arctic, decreasing to the south and east. Geology, food resources, and growth processes linked to temperature were possible explanations for these trends.

Concentrations of trace elements in marine sediments were dependent on local geology, particle size, organic matter content, and anthropogenic influence. The background spatial distributions of Pb, Cd, Hg, and copper (Cu) in marine sediments were related to the geological provinces of the Arctic.

Based on these conclusions, it was recommended that data gaps for biota be filled, with priority given to Hg, Pb, and Cd and organisms for which there were concerns about biological effects. Information for marine mammals in Russian waters was stressed as a data gap that needed to be addressed. Studies on the processes underlying spatial differences were also encouraged.

4.2. Data used in the second AMAP assessment

Most data presented in this assessment were collected after the end of the first AMAP assessment in 1997; although earlier data are discussed when relevant observations were not previously reported. Data are primarily reported for Hg, Cd, and Pb with limited attention given to other elements (e.g., selenium (Se), copper (Cu), nickel (Ni), and vanadium (V)). The assessment focuses on Hg and Cd because these elements are found in concentrations that may raise concerns for animal health, and potentially for human health. The limited data for Se are discussed with Hg, as concentrations tend to vary together and Se can ameliorate biological effects of Hg. In this chapter, Hg refers to total Hg unless otherwise specified. For marine biota, only Hg and Cd are discussed, as the other elements of anthropogenic origin are not generally considered of regional significance to marine ecosystems.

Because elements partition differently in the environment, the discussion is organized first by element, and then by compartment. This allows a consideration of the spatial distribution of each element across environmental compartments and so increases the likelihood of identifying spatial patterns. However, because most studies present data on more than one element and many address

more than one environmental medium, results from individual studies sometimes appear in several sections. To avoid redundancy, descriptions of individual studies are given in the section in which it is first used (usually, but not always, the Hg section). Also, comments on the strengths and limitations of different environmental compartments (e.g., precipitation, soils, marine mammals) for identifying spatial patterns appear in the Hg section. Other data limitations are discussed in Section 4.3.

4.3. Limitations of data for detecting spatial patterns

Detecting spatial patterns depends on the spatial coverage of the region, the quantity and diversity of the data collected in specific environmental compartments, and the quality and consistency of the data reported. In many cases identifying spatial patterns is only possible from the impression given by aggregated mean values. Difficulties in interpretation typically arise owing to differences in the field and laboratory methods used as well as a lack of standardized reporting formats. These differences can obscure spatial patterns and hinder the integration and interpretation of data from different investigators. Standardized reporting procedures will increase the possibility of identifying spatial patterns.

For this assessment, data are reported using the original measures of central tendency and variance (e.g., arithmetic versus geometric means, standard deviations versus standard errors). Where reporting formats for tissue concentrations differed, conversions between dry weight (dw) and wet weight (ww) were made and the assumptions used were specified. Because some heavy metals bioaccumulate, concentrations in animals are generally adjusted for size and/or age to allow meaningful comparison. However, in this assessment such adjustments were made only where population-specific information was available, as different populations require different adjustments.

4.4. Mercury

Mercury is a naturally occurring element with a high vapor pressure. This leads to the release of gaseous elemental Hg (GEM) to the atmosphere, where it is easily transported. Long-range atmospheric transport results in the distribution of Hg to regions and environments distant from original emission sites. Mercury occurs throughout the Arctic, in the atmosphere, and in biotic and abiotic compartments of terrestrial, freshwater, and marine environments.

4.4.1. Atmospheric mercury

Monitoring GEM has been a recent priority in many countries, although spatial coverage is still sparse (Figure 4·1). Since 1996, automated observations of GEM have been made within the context of various national programs at Alert (Canada), Barrow (Alaska), Zeppelin mountain (Svalbard), Station Nord and Nuuk (Greenland), and Tórshavn (Faroe Islands) In addition, Hg measurements are being made at Pallas (Finland) using manual traps that probably include a small admixture of particulates (T. Berg, pers. comm. 2003). In June 2001, a collaborative Canadian-Russian station was added at Amderma (Russia) to support Canadian commitments to the AMAP programme (B. Schroeder and A. Steffen, pers. comm., 2002).

Data on vapor-phase Hg have been reported to the Norwegian Institute of Air Research (NILU) from most of these sites. The AMAP Thematic Data Center (TDC)

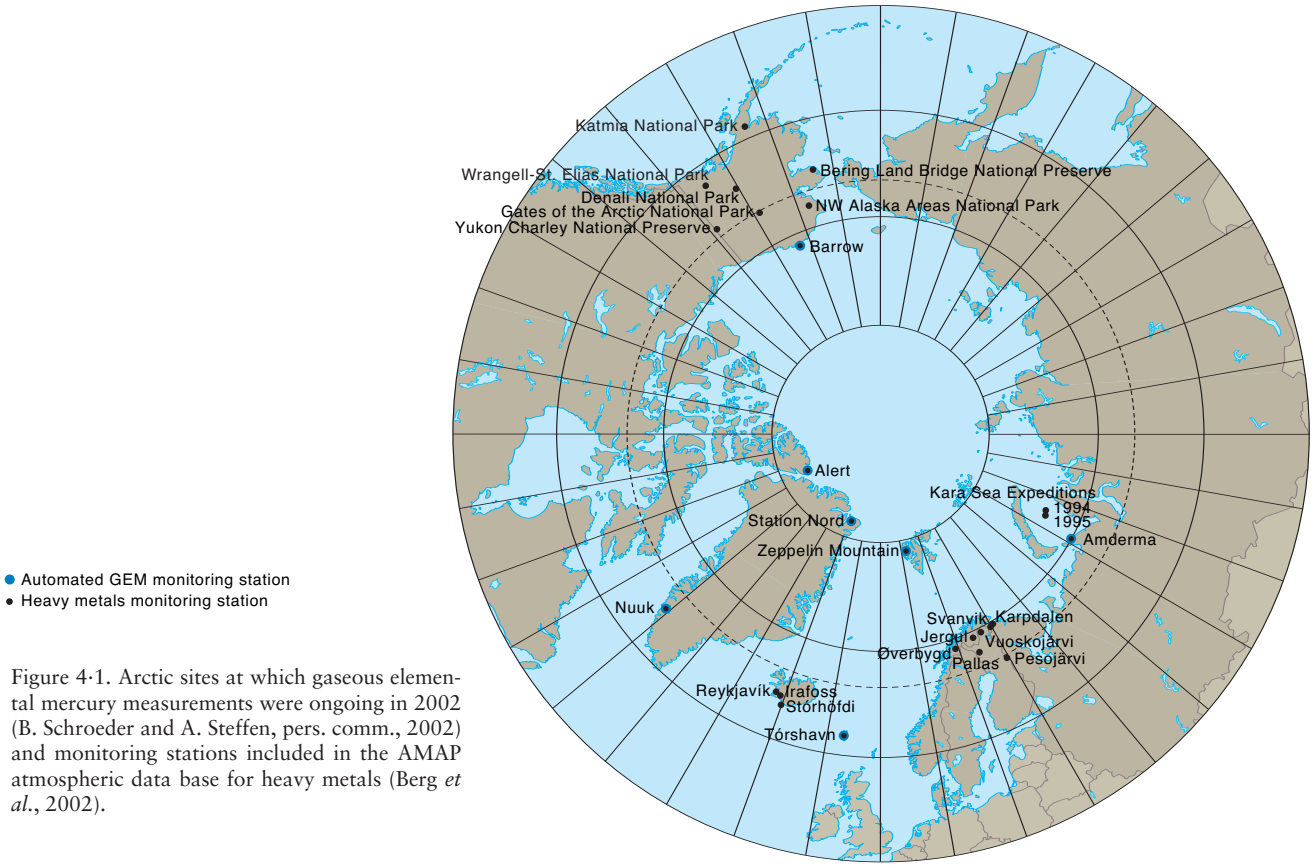


Figure 4·1. Arctic sites at which gaseous elemental mercury measurements were ongoing in 2002 (B. Schroeder and A. Steffen, pers. comm., 2002) and monitoring stations included in the AMAP atmospheric data base for heavy metals (Berg *et al.*, 2002).

at NILU is the largest single repository of data on the Arctic atmosphere; data on metals have been submitted for 24 sites from seven countries (Figure 4·1) as part of individual National Implementation Plans. Methodologies may differ among laboratories, and quality assurance is primarily the responsibility of the data originators although semi-quantitative evaluations are performed at NILU to help eliminate obvious errors.

Data suggest that annual mean concentrations of atmospheric Hg are about 1.5 ng/m^3 and are spatially relatively uniform. Wängberg and Munthe (2001) report similar findings for six European monitoring stations during 1999. Against this background, however, levels are highly variable and 'spikes' observed throughout the NILU datasets are the subject of current research (B. Schroeder and A. Steffen, pers. comm., 2002). Significantly, Hg depletion events (MDEs) have now been observed at many Arctic stations using high resolution (5 to 30 minute) automated instrumentation; earlier work based on manual collection did not reveal these patterns (Berg, 2001; 2003). The 'spikes' and especially the MDEs are of particular interest, as they appear to be essential in characterizing the biogeochemical cycle of Hg in the Arctic. These features are discussed in detail in Section 3.1.

4.4.2. Mercury in precipitation

Total Hg concentrations in precipitation have been reported to the NILU data center from Pallas (Finland) only. During March 1996 to December 1997, concentrations were between 4 and 52 ng/L, while occasional observations at Ny-Ålesund (Svalbard) were between 4 and 31 ng/L. These levels are comparable with those at remote sites in many parts of Europe, but lower than those at background sites in Germany (Berg *et al.*, 2001).

Snowpacks are subject to dynamic changes in Hg concentration at polar sunrise (Section 3.1.3). For example, Lu *et al.* (2001) reported the first large-scale systematic measurement of seasonal Hg variations in fresh surface snow in the Arctic using a time series obtained from an icebreaker frozen in Canada Basin through one full snow season. Mercury in the snow increased from 1 ng/L during darkness to about 60 ng/L in sunlight as atmospheric photochemistry at the air-ice interface increased (see Figure 3·10). Similar results have been found at Barrow.

In snowpack surveys of the eastern Canadian Arctic, Hudson Bay, and Greenland, higher Hg concentrations were found at coastal sites (25 to 160 ng/L) than inland sites, although concentrations were highly variable (Lu *et al.*, 2001). Similar patterns were found in northern Alaska, where concentrations were high (~100 ng/L) at the westernmost (Chukchi Sea) sites, an order of magnitude lower at sea-ice sites east of Barrow, and a further order of magnitude lower at inland sites sampled in 1994 near Prudhoe Bay (Garbarino *et al.*, 2002; Snyder-Conn *et al.*, 1997). The one inland exception was a coastal Prudhoe Bay site immediately northwest of what was then the largest gas-handling facility in the world, where spring Hg concentrations were about 80 ng/L (Snyder-Conn *et al.*, 1997). Mercury enrichment calculated against manganese (Mn) as a normalizing element $((Hg/Mn)_{snow}/(Hg/Mn)_{crust})$ was extremely high, particularly over the Chukchi Sea (440 to 1400 times). Collectively, the high sulfates, strong correlation of Hg with non-seasalt sulfates, and low Hg concentrations in seawater suggest that high Hg concentrations on the Chukchi Sea side were related to scavenging of particulate Hg (Hg II(s)) and/or vapor-phase mercury, possibly following long-range atmospheric transport resulting from waste incineration or fossil fuel combustion (Garbarino *et al.*, 2002). Given that the snow collections were made in late April, another possible explanation for the high Hg concentrations involves a recent halide-driven MDE, although Br was not measured.

To better understand the constraints on seasonal Hg dynamics in the Arctic atmosphere, there is a need for systematic studies on fresh snowfall and the spring snowpack. Ideally, these would use standardized methods and reporting conventions. As a minimum, intercalibration studies are encouraged to help link different studies in a quantitatively rigorous fashion. Starting such studies at existing air monitoring stations at inland and coastal sites linked to offshore sites on pack ice would be particularly valuable.

Data on total Hg in rainwater are available for several Scandinavian sites. Iverfeldt (1991) reported a strong decreasing south-to-north gradient in Hg in precipitation, with much of the Hg associated with particulates. For southern stations, a strong correlation between Hg and sulfate (as well as pH) suggests an anthropogenic connection. Wet deposition of Hg decreased substantially between 1987-89 and 1990-92 (from 27 to 10 µg/m^2/yr), consistent with political and economic change in Eastern Europe (Iverfeldt *et al.*, 1995). More recently, Wängberg and Munthe (2001) report a general south-to-north gradient in wet deposition from southern Sweden to northern Finland. Between 1995 and 1999 the south-to-north gradient weakened, as wet deposition decreased at all but the Pallas station. Wet deposition at Pallas over this period was about 2 µg/m^2/yr, but varied between 2 to 10 µg/m^2/yr at southern and central Swedish stations over the five-year period.

With respect to methylmercury (MeHg) in rainwater, Wängberg and Munthe (2001) report significant interannual variability in wet deposition between 1995 and 1999 at southern Swedish stations, with lower variability (0.04 to 0.06 µg/m^2/yr) at Pallas, Finland.

Deposition of total particulate Hg averaged 1.4 ng/m^3 at Pallas. Deposition was higher and more variable at five more southerly Scandinavian stations (7.6 to 34.6 ng/m^3; Wängberg and Munthe, 2001). Based on the absence of spikes in the Pallas data set, and using back-trajectory calculations, these authors concluded that Pallas is generally outside the zone of influence (central Europe) that affects the deposition of total particulate Hg (TPM) at more southerly stations.

Wängberg and Munthe (2001) analyzed different Hg phases at the Pallas station during 1995 to 1999 and compared them to those at the Rörvik station on the west coast of southern Sweden over the same period. Total particulate Hg was generally six times higher (8.7 versus 1.4 to 1.7 ng/m^3), and wet deposition three times higher (4.25 to 8.97 versus 1.91 to 2.52 µg/m^2/yr), in the

south. Differences in wet deposition of MeHg were somewhat less (0.029 to 0.143 versus 0.05 to 0.063 µg/m^2/yr). This suggests that Pallas is much less affected by industrial emissions than the Rörvik site, which has been demonstrated to be influenced by TPM from central Europe (Wängberg *et al.*, 2003). In contrast, GEM was generally similar at the two sites (1.39 to 1.53 versus 1.3 to 1.50 ng/m^3).

4.4.3. Mercury in the terrestrial environment

4.4.3.1. Soils and peats

Background concentrations of Hg in soils have been estimated at < 0.01 to 0.03 mg/kg based on Greenland soils (AMAP, 1998). This is lower than previous estimates of background from Fennoscandian forest soils north of the Arctic Circle (0.7 mg/kg; Anderson *et al.*, 1991), which probably reflect anthropogenic contributions from industrial sources close to the Fennoscandian border with Russia (E. Steinnes, pers. comm., 2002).

Three nationwide surveys of heavy metals in the surface humus (organic) layers of forest podzol soils were undertaken in Norway. Each covered about 500 sites, of which about a third were north of the Arctic Circle. The first survey took place in 1977 (Allen and Steinnes, 1980; Steinnes *et al.*, 1997), the second in 1985 (Bølviken and Steinnes, 1987; Njåstad *et al.*, 1994; Page and Steinnes, 1990), and the third in 1995 (Nygård, 2000; Steinnes and Rühling, 2002). Differences in Hg concentration from south to north were relatively small, despite a strong south-to-north decreasing gradient in the wet deposition of Hg observed in Scandinavia (Iverfeldt, 1991). The lack of a similar gradient in surface soil may reflect the higher retention of dry deposited GEM in surface soils of colder northern climates (Steinnes and Andersson, 1991). Surface peat from a large number of ombrotrophic bogs in Norway gave similar results (Steinnes, 1997).

Collaborative studies on northern peats have focused on cores from ombrogenic and minerogenic peat deposits in the Canadian High Arctic (Bathurst Island, Nunavut), southwest Greenland, the Faroe Islands, Finland, Denmark, and Iceland as well as several more southerly regions (W.M. Shotyk *et al.*, pers. comm., 2001). Highest Hg concentrations (up to 700 µg/kg) occurred in the Faroe Islands, with lower concentrations in non-AMAP areas (Shetland Islands → Scotland → Switzerland). The cumulative long-term rates of atmospheric Hg accumulation averaged over the past one to five millennia are 6.1 to 7.8, 7.9 to 9.7, 5.1 to 6.1, and 2.0 to 2.3 µg/m^2/yr, respectively. Elevated Hg concentrations and net accumulation rates at deeper peat layers in Scotland, the Shetland Islands, and the Faroe Islands relative to the ombrogenic bog in Switzerland suggest a significant natural source of Hg in the North Atlantic region. According to Roos-Barraclough *et al.* (2002) possible sources include volcanic emissions and/or degassing of the earth's mantle in Iceland. However, their contribution to local pathways is unclear. For example, Hg concentrations in landlocked Arctic char (*Salvelinus alpinus*) from Iceland are lower than in landlocked char from other Arctic regions discussed in the current assessment (Section 4.4.4.3), which would be surprising if degassing in Iceland were a major source of Hg.

4.4.3.2. Mosses and lichens

Mosses and lichens generally lack vascular systems, which minimizes the possibility for uptake from substrates and internal translocation. Nutrients and pollutants are accumulated primarily from the atmosphere. Together with their broad geographical range and ecological amplitude, these factors make mosses and lichens useful biomonitors of atmospheric deposition (Steinnes, 1995). Studies of moss chemistry have been undertaken at fine-scale spatial coverage to assess international patterns of atmospheric deposition of heavy metals and trace elements, especially in Europe and Scandinavia (e.g., Rühling and Steinnes, 1998). Most large-scale studies focus on the feather mosses *Pleurozium schreberi* and *Hylocomium splendens*. *H. splendens* is of particular interest because it develops identifiable annual increments, thus permitting analysis of known exposure intervals. Its broad distribution allows it to serve as a spatially extensive real-time deposition proxy, complementing the temporal archives of atmospheric deposition found in radiometrically-dated ice, peat, and sediment cores. Detailed studies on *H. splendens* have focused on issues such as laboratory rates of foliar uptake (Rühling and Tyler, 1970), environmental factors influencing element concentrations (Ford *et al.*, 1995; Økland *et al.*, 1999), plant architecture and growth rates (Callaghan *et al.*, 1997; Økland *et al.*, 1997), and comparability between observed tissue concentrations and measured rates of atmospheric deposition (Berg *et al.*, 1995; Berg and Steinnes, 1997; Ross, 1990).

Regular monitoring of heavy metals and trace elements in feather mosses has been undertaken at five-year intervals since 1970 in Sweden and 1975 in Norway. The two most recent moss surveys (1995, 2000) covered most of Europe, including the Kola Peninsula (Figure 4·2). Quality control has been an important part of this integrated program since the outset, including the ana-

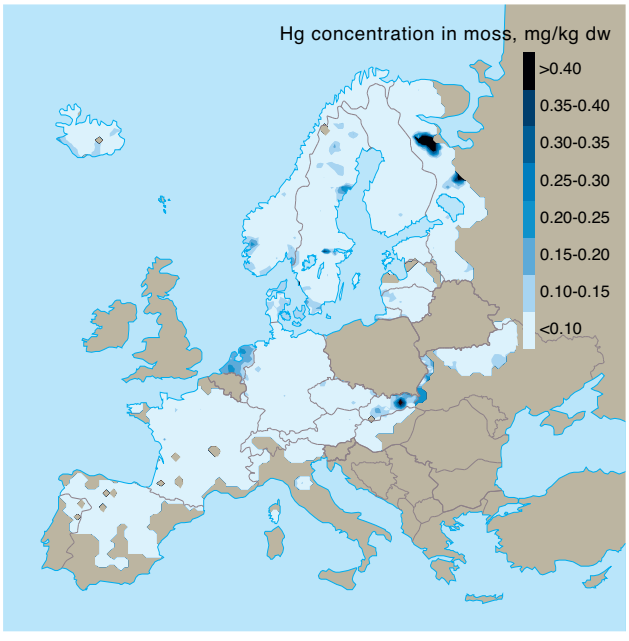


Figure 4·2. Mercury concentrations in European feather mosses, 1995 (Rühling and Steinnes, 1998).

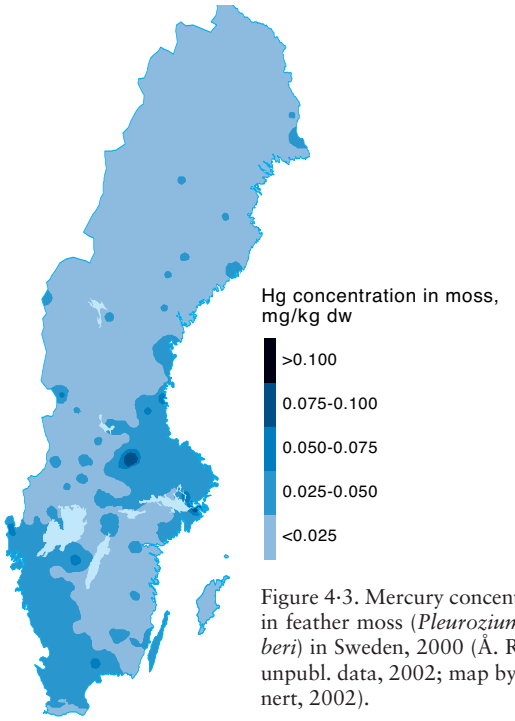


Figure 4·3. Mercury concentrations in feather moss (*Pleurozium schreberi*) in Sweden, 2000 (Å. Rühling, unpubl. data, 2002; map by A. Bignert, 2002).

lysis of intercalibration samples by all participating countries. Reports from the program discuss spatial patterns by element, and various publications (e.g., Rühling and Steinnes, 1998) detail situations in which differences in sample handling and analysis may have influenced results.

The moss survey of 2000 continued to show south-to-north decreasing Hg gradients in Sweden (Figure 4·3) that are statistically significant (Figure 4·4). A similar pattern is seen in Finland. Mercury concentrations are generally between 0.02 and 0.04 mg/kg dw in the northern (AMAP) regions of Sweden, Norway, and Finland (Å. Rühling, pers. comm., 2002; E. Steinnes, pers. comm., 2002; Mähönen, 2002), and only slightly higher in the Faroe Islands (0.02 to 0.07; Annex Table A4). However, concentrations are about 2.5 times higher and much more variable (up to 0.1 mg/kg) in the southern (non-AMAP) regions of Sweden and Finland (up to 0.14 mg/kg dw) (Mähönen, 2002; Metinfo, 2003).

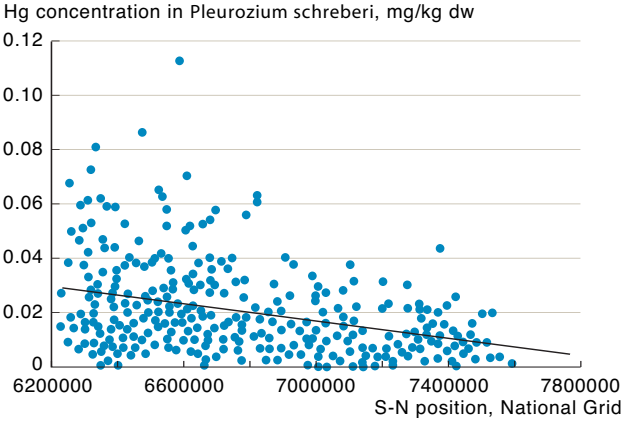


Figure 4·4. Relationship between mercury concentration in *Pleurozium schreberi* and latitude in Sweden, 2000 (significance of the correlation: $p < 0.0001$; slope of the regression line: $p < 0.0001$) (Å. Rühling, unpubl. data, 2002).

In contrast, strong south-to-north Hg gradients are not found in Norwegian mosses. This is consistent with similar observations on Norwegian surface humus soils. Steinnes (2001) hypothesizes that the difference may be due to a higher supply of GEM in the colder northern mountains of Norway, the retention of which would add to the general Hg burden from atmospheric deposition in the form of precipitation. Differences in precipitation as well as overall atmospheric deposition patterns from local and regional sources in Norway, Sweden, and Finland are also likely to influence these spatial patterns.

In Arctic Alaska, Hg concentrations in *Hylocomium splendens* are higher than in the Nordic countries, ranging from 0.02 to 0.112 mg/kg dw (Ford *et al.*, 1995). Concentrations are yet higher on the Taymir Peninsula, Russia (0.12 to 0.55 mg/kg dw; Allen-Gil *et al.*, 2003; Ford *et al.*, 1997). Again, these spatial patterns are consistent with the notion that Hg may be more strongly retained in colder climates, although there is no independent evidence to this effect.

Despite significant smelter activity on the Kola Peninsula around Monchegorsk, Nikel, and Zapoljarnij, spatially extensive studies on *Hylocomium splendens* performed as part of the Kola Ecogeochemistry Project (www.ngu.no/Kola) found Hg concentrations spanning only one order of magnitude (0.02 to 0.16 mg/kg dw), despite the dense sampling grid around industrial centers (Reimann *et al.*, 1997b). This contrasts with the findings for other elements (e.g., Cd) for which concentrations over two orders of magnitude were common, with an epicenter at industrial centers. For Hg, the Monchegorsk and Nikel industrial complexes were each marked by a single high concentration accompanied, in the case of Monchegorsk, by a small halo of elevated concentrations to the south. However, elevated concentrations also occurred in samples along the Norwegian coast and the Varanger Peninsula. The source of elevated Hg at these sites is attributed to sea spray (Reimann *et al.*, 1997b), although the more recently discovered Br-mediated coastal MDEs may also be involved.

Using a different moss species (*Racomitrium lanuginosum*) Riget *et al.* (2000b) found Hg concentrations of 0.059 to 0.196 mg/kg dw in Greenland, similar to concentrations in the Faroe Islands (0.16 mg/kg dw; see Annex Table A4), and slightly higher than concentrations in Arctic Alaska (0.016 to 0.112 mg/kg dw; Ford *et al.*, 1997). Concentrations in the Greenland lichen *Cetraria nivalis* were lower (0.033 to 0.089 mg/kg dw; Riget *et al.*, 2000b), and similar to concentrations in Alaskan lichens (*C. cucullata* and *Masonhalea richardsonii*; 0.015 to 0.085 mg/kg dw; Ford *et al.*, 1997).

Many factors influence concentrations of heavy metals in moss in addition to atmospheric deposition (e.g., marine spray, and foliar leaching from overstory vascular plants) (Steinnes, 2001). The most potentially troublesome confounding factors are windblown dust, and dissolved and/or particulate metals in snowmelt. The effect of windblown dust is especially important in areas with sparse vegetation, such as Arctic tundra (Riget *et al.*, 2000b; Steinnes, 1995). Soil dust particles must be accounted for when assessing anthropogenic contributions of elements such as Cu, Ni, Pb, and V (Steinnes *et*

al., 2001), and can also be important for minor elements (E. Steinnes, pers. comm., 2002). The most common way to address this concern is by using Enrichment Factors (EFs), in which plant concentrations are normalized to soil concentrations by taking the ratio of the concentration of the target element to the concentration of a conservative soil element in plant tissues relative to the same ratio in soils (Nash and Gries, 1995; Puckett and Finegan, 1980). Reimann and de Caritat (2000) criticized the use of generic crustal material as a reference. More recent synoptic studies in Alaska and Russia have addressed this concern by using local crustal material as a reference (Ford and Hasselbach, 2001, Ford *et al.*, unpubl.).

Enrichment factors have been used to normalize *Hylocomium splendens* moss concentrations to local inorganic soil parent material (total organic carbon <10%) (Ford and Hasselbach, 2001; Ford *et al.*, unpubl.). Unlike the situation with elements such as Pb, Cu, and V, uniformly high EFs are found for Hg (generally >35) for Arctic Alaska and the Taymir Peninsula, Russia (see figure 4·20). This suggests that moss Hg concentrations are in excess of what would be expected based solely on local soil parent material. Similar results are found for the lichen *Cetraria cucullata*, which has a broader distribution in the circumpolar Arctic than *H. splendens*.

4.4.3.3. **Terrestrial birds**

Ptarmigan are key monitoring organisms for this assessment because they are widespread and resident. Willow and rock ptarmigan (*Lagopus lagopus* and *L. mutus*) are considered together in this assessment.

Recent data from Canada, Greenland, and Russia indicate that Hg concentrations in ptarmigan muscle were generally below detection limits (< 0.001 to < 0.013 mg/kg ww; see Annex Table A5, Champoux *et al.*, 1999; Melnikov *et al.*, 2002). Concentrations were higher in liver. Concentrations of Hg in ptarmigan liver and kidney collected in 1999 from two locations in West Greenland (Kitaa) were 0.022 to 0.036 mg/kg ww and 0.037 to 0.046 mg/kg ww, respectively, generally similar to levels previously reported for Arctic Canada but higher than levels previously reported in northern Scandinavia (AMAP, 1998). In Scandinavia, variability was high and Hg showed no distinct regional pattern, while in Canada higher values were observed in central and western regions. Mean values of Hg in ptarmigan liver from Russia were 0.006 to 0.025 mg/kg ww, generally lower than those from Greenland (0.020 to 0.040 mg/kg ww). Within Russia, the highest liver Hg concentrations occurred in the Dudinka area of the Taymir Peninsula (Melnikov *et al.*, 2002), with the exception of a set of samples from the Pechora Basin (mean 0.093 mg/kg ww; see Annex Table A5).

Contaminants in northern Canadian waterfowl and game birds were studied in the 1990s owing to their importance to subsistence diets (Braune *et al.*, 1999a, 1999b). Again, concentrations in breast muscle were generally quite low (<0.4 mg/kg ww) with concentrations in fish-eating birds higher (up to 2 mg/kg ww). For waterfowl, median values of liver Hg ranged from 0.1 to 0.7 mg/kg ww. The lowest concentrations were recorded in northern pintail (*Anas acuta*), a surface feeding duck, while high values (up to 3.8 mg/kg ww) were found in diving ducks at Fort Good Hope (Radili Ko) (e.g., surf scoter, *Melanitta perspicillata*). Relatively high values (>1 mg/kg ww) were not uncommon in other species (e.g., bufflehead (*Bucephala albeola*), goldeneye (*Bucephala* spp.), king eider (*Somateria spectabilis*), common eider (*S. mollissima*), and long-tailed duck (also known as oldsquaw, *Clangula hyemalis*); Braune *et al.*, 1999b). Similar results were found by Champoux *et al.* (1999) for northern Quebec. Specimens obtained in the 1980s from that region had even higher levels (> 2.5 mg/kg ww) for the three species of mergansers reported, while concentrations in raptors (peregrine falcon (*Falco peregrinus*) and osprey (*Pandion haliaetus*)) muscle, liver, and kidney were < 0.7 mg/kg ww (Champoux *et al.*, 1999).

As these species are migratory, body burdens of metals may also be influenced by feeding behavior and pollution loads in the overwintering grounds, in addition to potential influences from local lithology. Thus, it is difficult to evaluate the implication of spatial patterns in tissue concentrations in Arctic waterfowl and game birds.

4.4.3.4. **Terrestrial mammals**

Contaminant concentrations in animals are studied for a variety of reasons. Long-lived ungulates are favored for regional comparisons owing to their importance in subsistence diets and because they can accumulate significant levels of trace metals (Frøslie *et al.*, 1986; Scanlon *et al.*, 1988). Moose (*Alces alces*) is the species of choice for monitoring environmental contaminants in Swedish forest areas (Odsjö *et al.*, 2001), where diets consist mainly of twigs and leaves of trees and shrubs (Cederlund *et al.*, 1980). In contrast, reindeer/caribou (*Rangifer tarandus*) are more common in the circumpolar Arctic, where forested habitat is limited. As such, they are more widely studied in the Arctic. Summer diets include grasses, sedges, twigs, leaves, and mushrooms; winter diets mainly comprise lichens (Kelsall, 1968; Parker, 1978) that accumulate nutrients and contaminants from the air.

Comparisons of metal concentrations in ungulates from different regions can be complicated by differences in age distribution, local diet, and/or sampling season. The reindeer/caribou data discussed here are generally for three- to six-year old animals, so the influence of age should be minimal. Seasonality is a demonstrateds issue in studies from Greenland, where tissue collected in late winter reindeer/caribou had higher metal levels than tissue from early winter animals, especially at sites rich in lichen (Aastrup *et al.*, 2000). The availability of lichen as winter forage seems to be a key factor influencing metal concentrations in reindeer/caribou and should be controlled for, or at least considered, when interpreting monitoring data (Aastrup *et al.*, 2000).

Mercury concentrations in reindeer/caribou tend to decrease in the order kidney → liver → muscle. Spatial patterns in reindeer/caribou liver Hg are summarized in Figure 4·5. Data are discussed on a wet weight basis; where data were originally reported on a dry weight basis, wet weight concentrations were estimated using a conversion based on percentage water content, with the ww : dw relationship generally about 1 : 0.28.

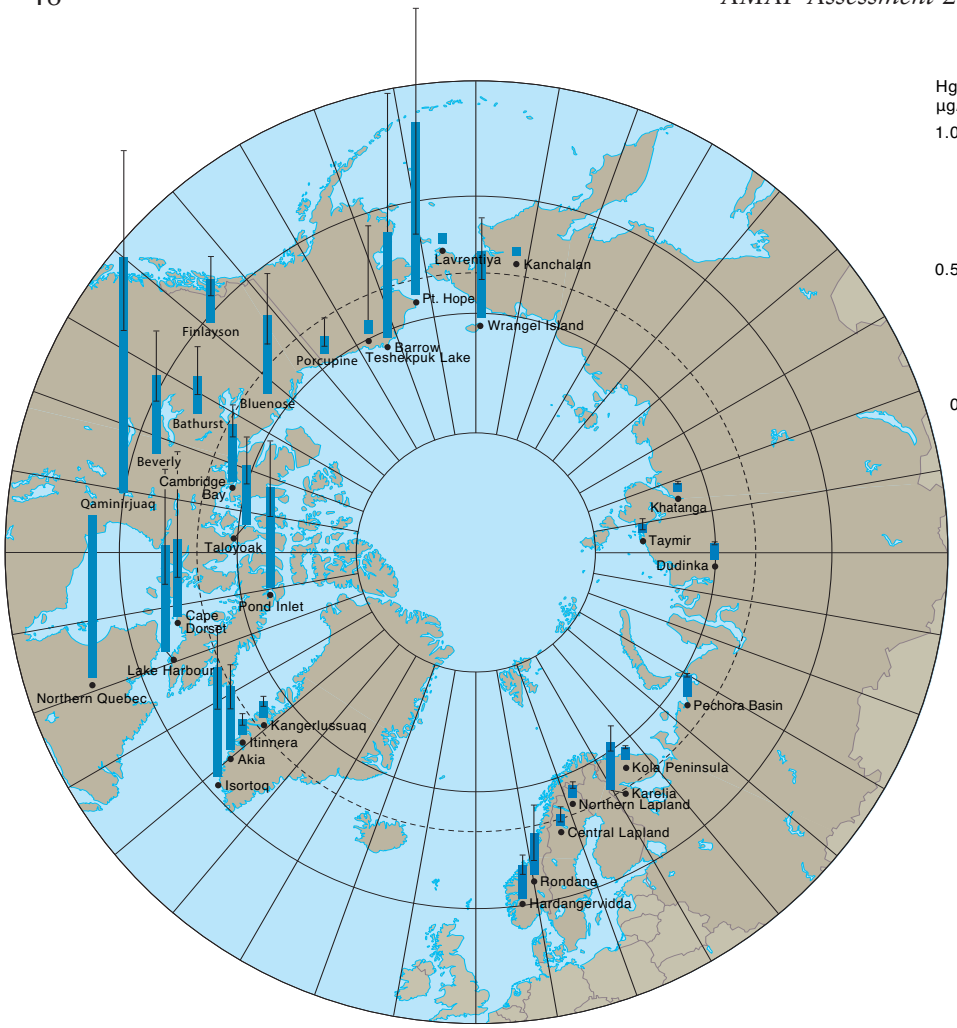


Figure 4·5. Mean mercury concentrations in reindeer/caribou liver (Aastrup *et al.*, 2000; Elkin, 2001; Espelien *et al.*, 1999; Macdonald *et al.*, 2002; Melnikov *et al.*, 2002; Odsjö, 2003; O'Hara, unpubl. data, 2002; Robillard *et al.*, 2002; F. Riget, unpubl. data, 2002).

Mean values of Hg in reindeer liver from Greenland were 0.06 to 0.3 mg/kg ww, with highest values in Akia in the southwest (Aastrup *et al.*, 2000, see also Annex Table A6). Concentrations in northern Sweden were at the lower end of this range, with mean values around 0.04 mg/kg ww at each of two sites (Odsjö, 2003). Sites in southern Norway had slightly higher concentrations (means 0.13 to 0.16 mg/kg ww; Espelien *et al.*, 1999). In the first AMAP assessment, highest values were found in Canada. This was also the case in this assessment, with particularly high Hg levels in caribou liver from two regions in northern Quebec (means of 0.38 ± 0.15 and 0.70 ± 0.41 mg/kg ww; Robillard *et al.*, 2002), with an overall geometric mean of 0.59 mg/kg ww (G. Beauchamp, pers. comm., 2002) and Qaminirjuaq, NWT. New data from Alaska show similar concentrations (0.6 mg/kg ww at Point Hope and 0.4 mg/kg ww at Barrow) with high local variability (O'Hara unpublished data 2002). In contrast, Canadian values for herds in the Yukon, Northwest Territories, and Nunavut sampled between 1992 and 1999 are lower, with geometric means of 0.13 to about 0.4 mg/kg ww in liver (Macdonald *et al.*, 2002, see also Annex Table A6), with the high end generally lower than values for northern Quebec and Akia in southern Greenland, but with high variability (Elkin, 2001; Macdonald *et al.*, 2002, see also Annex Table A6 and Figure 4·5).

Concentrations of Hg in reindeer/caribou liver sampled at six sites in Arctic Russia in 2001 were generally lower, ranging from 0.013 to 0.138 mg/kg ww with the highest levels in the Pechora Basin and on the Taymir Peninsula near Dudinka (Melnikov *et al.*, 2002). A study from the early 1990s found higher liver Hg concentrations in reindeer/caribou from Wrangell Island (0.24 mg/kg ww), on the Chukotka Peninsula (0.17 mg/kg ww), and in northern Karelia (0.19 mg/kg ww) (Espelien *et al.*, 1999). However, no overlap samples were available with which to compare the earlier and later data.

In summary, the highest Hg concentrations in reindeer/caribou liver occurred in northern Quebec, southwest Greenland, and Alaska. Variability was also highest in those regions. Scandinavia and Russia had mostly lower Hg concentrations in reindeer/caribou liver.

Data on Hg concentrations in other species are limited. Concentrations in lamb (*Ovis* spp.) liver from the Faroe Islands were studied in connection with the October 1997 and 1999 slaughters (Larsen and Dam, 1999; Olsen *et al.*, 2003). In both cases, Hg concentrations were below a detection limit of 0.02 mg/kg ww. Similar results were found for sheep and muskox (*Ovibos moschatus*) in Greenland where concentrations in muscle were either low (0.002 mg/kg ww) or undetectable (see Annex Table A6). Values were only slightly higher in liver (means of 0.005 and 0.023 mg/kg ww, respectively) and kidney (means of 0.012 and 0.072 mg/kg ww, respectively).

Data on Hg in hare liver are available for northern Canada (Quebec), Greenland (Qeqertarsuaq region), the

Faroe Islands, and Russia (Kola, Pechora, Taymir, and Chukotka). In samples of mountain hare (*Lepus timidus*) from 1991 and later, Russian hare had the lowest and Faroe Islands the highest Hg concentrations (Melnikov *et al.*, 2002; Olsen *et al.*, 2003), with concentrations in Faroe Island hare about ten times higher than in Russia. Arctic hare (*L. arcticus*) from Greenland and snowshoe hare (*L. americanus*) from northern Quebec had intermediate values (~ 0.024 mg/kg ww; see Annex Table A6).

Selenium was also analyzed in muscle and liver from snowshoe hare in northern Quebec. Concentrations in these tissues were quite similar (~ 0.30 mg/kg ww). Liver Se was generally higher in Faroe Island hare than Greenland hare (means of 0.6 and 0.1 mg/kg ww, respectively). Concentrations were high in kidney from Greenland hare (up to 0.84 mg/kg ww). There is a lack of kidney data for other areas.

A study of bank vole (*Clethrionomys glareolus*) liver in 1981 showed Hg concentrations three times higher in the south than the north of Sweden (Nyholm and Rühling, 2001), as would be expected from the gradients in humus soils and mosses at that time. Frøslie *et al.* (1984) found a similar gradient for moose, and suggested a relationship with atmospheric deposition. By 1996-97, Hg concentrations in bank vole liver from southern Sweden had declined significantly (0.0025 to 0.049 mg/kg ww versus 0.0045 to 0.21 mg/kg ww; Nyholm and Rühling, 2001). Unfortunately, northern Sweden was not resampled. In Arctic Alaska, liver of Arctic ground squirrel (*Spermophilus parryii*) sampled between 1991 and 1993 had similarly low concentrations of total Hg, ranging from 0.01 to 0.08 mg/kg ww, of which only a very small percentage (< 0.4%) occurred as MeHg (Allen-Gil *et al.*, 1997b).

4.4.4. Mercury in the freshwater environment

4.4.4.1. River water and sediments

Few studies address Hg concentrations in Arctic river water and sediments, and those that do are concerned with potentially elevated levels due to local geology or land-use (e.g., reservoirs, placer mining).

The Kuskokwim River in southwestern Alaska drains a largely undeveloped basin of about 130000 km^2 rich in gold (Au) and Hg. Mining is common but large-scale industrial activities are not. Several studies have focused on Hg distribution in the smaller drainage basins of the Kuskokwim region (Bailey and Gray, 1997; Gray *et al.*, 1991, 1994; Nelson *et al.*, 1977). More recently, the chemical composition of the mainstem river has been studied in relation to both lithology and mining activity (Wang, 1999). Total Hg concentrations in the mainstem were below U.S. EPA drinking water criteria at all sites, but exceeded concentrations that might affect aquatic life in one low-discharge tributary creek. Dissolved Hg was 24 to 32% of total Hg. Methylmercury was detected at two sites only, and both also had high concentrations of sedimentary MeHg.

Bed sediments from the mainstem Kuskokwim occasionally exceeded the Canadian Interim Sediment Quality Guidelines for total Hg (Wang, 2001). Elevated Hg and antimony (Sb) levels were attributed to the combined influence of lithology and past mining activity. Background concentrations varied with size fraction and ranged from 0.09 to 0.15 mg/kg dw for total Hg and 1.6 to 2.1 mg/kg dw for Sb. Concentrations of MeHg were three orders of magnitude lower, ranging from 0.08 to 0.25 µg/kg dw, with maxima of 1.75 and 3.12 µg/kg dw at high total-Hg sites (Wang, 2001).

The Kemijoki River is the largest river in Finland with a mean discharge of 540 m^3/s and a drainage basin of 50000 km^2, most of which is north of the Arctic Circle (Porvari and Verta, 1998). The river basin is regulated for hydroelectric power production, and includes 15 power plants and two major lakes constructed in 1967 and 1970. In 1994, total Hg and MeHg were measured in June, August, and December at 14 sites upstream and downstream of reservoirs and power plants, and in the reservoirs themselves. Total Hg concentrations ranged from 0.44 to 8.43 ng/L upstream, 0.98 to 3.94 ng/L downstream, and 0.72 to 2.29 ng/L in the reservoirs, with MeHg comprising from 1.3 to 14.8% of the total Hg. Slight increases in MeHg concentration (and MeHg as a percentage of total Hg) were found in the reservoirs in late summer.

The Pechora River Basin in western Siberia is considered one of the most polluted areas in northwestern Russia due to intensive development of mineral deposits, including oil. The watershed of the Usa River in particular is exposed to severe Hg pollution from Vorkuta coal and Usa oil-extracting industrial processes. River sediments were analyzed for Hg at more than 20 sites in different parts of the basin (Dauvalter, 2002). Deep (20 to 25 cm) sediments had a wide range in Hg concentration (0.010 ± 0.047 mg/kg dw to 0.140 ± 0.033 mg/kg dw); even so, surficial Hg concentrations were elevated at several sites relative to bottom sediments (e.g., 0.11-0.12 versus 0.01-0.04 mg/kg dw in the Usa River, and 0.07 versus 0.01 mg/kg dw in the Pechora delta; Dauvalter, 2002).

4.4.4.2. Lake sediments

Freshwater lake sediments are commonly used to track temporal changes in the deposition of trace metals (see Section 5.3.4). However, several national studies also use enrichment factors (EFs, i.e. the ratio of element concentrations in surface versus downcore (presumably pre-industrial) sediments, with or without adjustments for differences in organic matter and/or aluminum) to look at spatial patterns in recent enrichment. A more refined version of this technique is the flux ratio (FR), in which raw concentrations are adjusted for changing rates of sediment accumulation as determined by radiometric dating of the sediment column. More commonly, however, cores are not dated and raw concentrations only are used.

A recent study of 210 Norwegian lakes showed that Hg was one of the elements with the highest EFs (Rognerud and Fjeld, 2001). A decreasing south-to-north gradient in surface enrichment was found after correcting for differences in organic matter. This is similar to the south-to-north gradients in surface enrichment observed in Sweden (e.g., Bindler *et al.*, 2001b) and Finland (Verta, 1990), and agrees with similar spatial patterns in other media, including atmospheric deposition in Norway (Iverfeldt, 1991).

Mercury EFs are generally 1.5 to 3.0 throughout the

Arctic, including northern Norway, northern Sweden, northern Finland, Siberia, Arctic Canada, and southwest Greenland (Bindler *et al.*, 2001a, 2001b; Cheam *et al.*, 2001; Lockhart *et al.*, 2001b; Mannio *et al.*, 1997; Rognerud *et al.*, 1998; SLU, 2003). However, using radiometrically dated cores Landers *et al.* (1998) demonstrated a generally decreasing south-to-north gradient in FRs in Scandinavia corresponding to gradients in other environmental compartments. In North America FRs were at a maximum in the Canadian High Arctic, decreasing to the west, south, and east (Landers *et al.*, 1998).

In Eurasia, elevated Hg EFs have been observed close to regions with major industrial/urban emissions. For example, in their study of 66 lakes across northern Norway and the Siberian coast, Rognerud *et al.* (1998) found elevated Hg EFs in western and lower latitude areas. Similarly, a concentric pattern for several elements, including Hg, has been reported for surficial sediments from 100 lakes in the Murmansk region (Dauvalter, 1994; Dauvalter and Rognerud, 2002). In the latter, sedimentary Hg was negatively correlated with distance from the smelter ($p = 0.05$) suggesting that sources may include atmospheric emissions as well as wastewater from tailing dams and mines (Dauvalter, 1994; Dauvalter and Rognerud, 2002). These patterns are consistent with the outcome of the spatially extensive Kola Ecogeochemistry Project moss surveys (see Section 4.4.3.2) although the gradient is steeper with lake sediments. Whether this difference is due to differences in internal processing in the two compartments (mosses and lake sediments) or to additional sources to the lakes (such as wastewater) is currently not known.

One study in Greenland examined the variation in Hg EFs over a transect of lakes from the coast inland to the Greenland ice sheet (Bindler *et al.*, 2001a). The MDE hypothesis requires marine Br, and slightly higher Hg EFs were in fact found in the coastal lakes than most of the inland lakes (three-fold versus two-fold increases). However, contrary to expectations based on the MDE hypothesis, much higher EFs (4 to 11) were found in the two lakes adjacent to the inland ice sheet and one lake on a nunatak within the ice sheet. The authors speculate that meteorological conditions near the ice margin, particularly strong summer katabatic flows from the ice

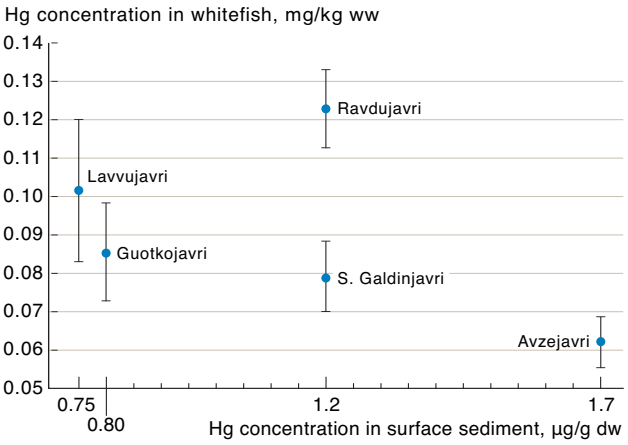


Figure 4·6. Total mercury concentrations in common whitefish (also known as European whitefish, *Coregonus lavaretus*) versus mercury concentrations in surficial sediments (0 to 1 cm) (Skotvold *et al.*, 1997).

down over the adjacent tundra, may increase snow scavenging and/or dry deposition, or inhibit re-emission of Hg (Bindler *et al.*, 2001a). For three of the 21 lakes studied (including Nunatak Lake), a full stratigraphic sequence was dated and analyzed; FRs based on dated sediments gave similar results.

Three Arctic studies examined the relationship between sedimentary Hg and fish tissue Hg and failed to show a systematic relationship between the two. The Tundra North West 99 expedition studied small landlocked Arctic char from seven Arctic Canadian lakes (Borg *et al.*, 2001). One had highly (20- to 40-fold) elevated levels of sedimentary Hg relative to other lakes in the data set, but low concentrations of Hg in char (overall median 0.065 mg/kg ww). Similarly, in Greenland, a three-fold difference in sedimentary Hg between two lakes was not reflected in char Hg concentrations (Riget *et al.*, 2000a). For whitefish (*Coregonus* spp. and *Prosopium* spp.), a study by Skotvold *et al.* (1997) showed an inverse relationship between sedimentary Hg and fish Hg (Figure 4·6) in Norwegian lakes. They proposed that low fish Hg levels are driven by high sedimentary concentrations of Se (0.9 to 3.9 mg/kg dw) that co-occur with high sedimentary Hg (0.06 to 0.17 mg/kg dw), which is consistent with laboratory studies showing that Se additions to water decrease fish tissue Hg (Håkansson *et al.*, 1990). However, this relationship is based on a small number of sites, and the exception to the general trend (Ravdujarvi) demonstrates that sedimentary Hg is not a reliable proxy for fish Hg.

4.4.4.3. **Freshwater fish**

Fish are key monitors for contaminants owing to their importance in natural food webs and their contribution to the subsistence harvest. Arctic char was designated a key freshwater monitoring species for the first AMAP assessment together with brown trout (*Salmo trutta*), burbot (*Lota lota*), northern pike (*Esox lucius*), and whitefish (*Coregonus* spp. and *Prosopium* spp.). Element concentrations were reported for char, whitefish, and burbot, and a time series was reported for Hg in pike from northern Sweden.

Arctic char is a key species in the present assessment owing to its circumpolar distribution (Figure 4·7) and its importance to subsistence diets. For example, Arctic char and threespine stickleback (*Gasterosteus aculeatus aculeatus*) are the only freshwater species found in Greenland. Unlike the first AMAP assessment, for which data on freshwater fish were sparse, contributions for this assessment cover almost two dozen species, with particularly good spatial coverage for whitefish (considered as a group) and char.

Many factors affect contaminant concentrations in fish and should be taken into account in spatial comparisons. For example, water chemistry, size and age of fish, growth rates, trophic position, and diet are all likely to be important. The ultimate consequence of such factors on monitoring network design is that (temporal) trend detection as well as the identification of spatial patterns becomes more difficult. Variability can be minimized in several ways (e.g., by selecting only smaller, benthic-feeding individuals for analysis, or by increasing the total size of the sample). Establishing the influence of

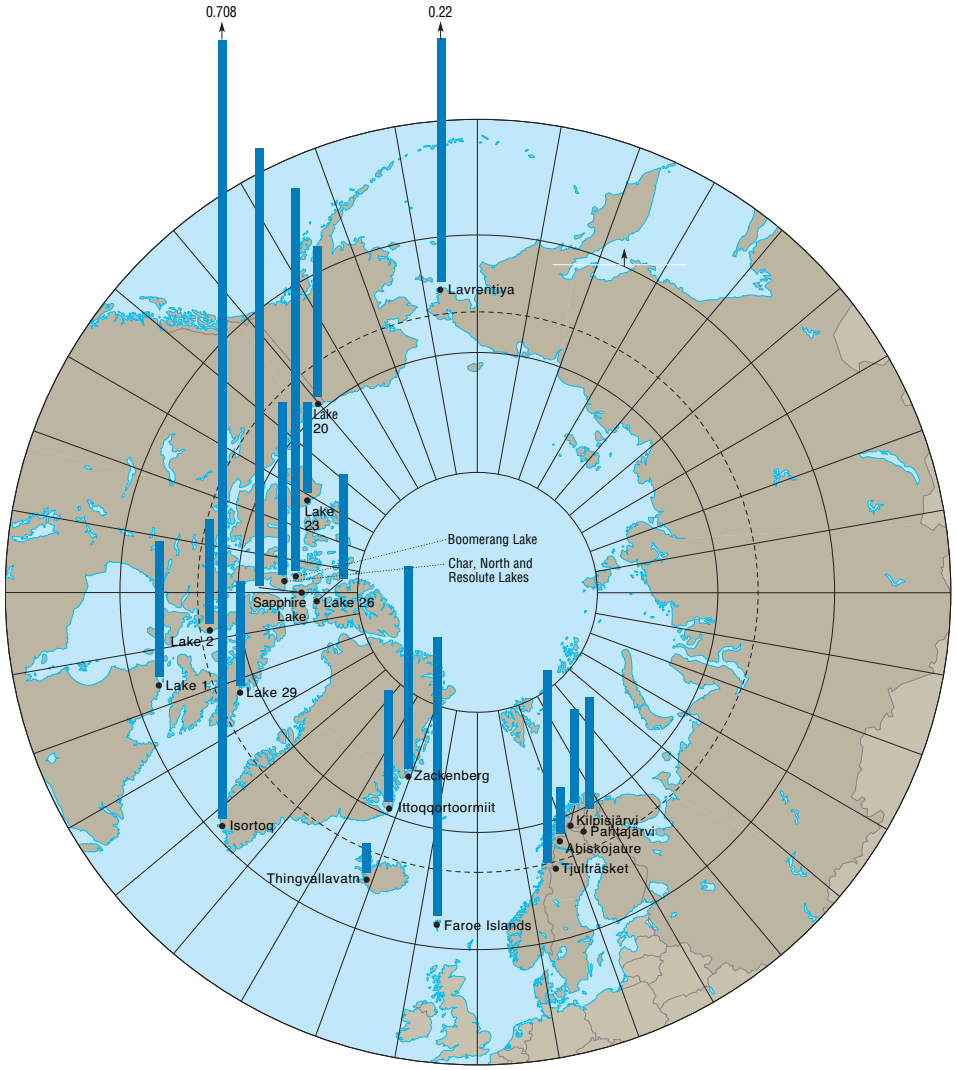


Figure 4·7. Concentrations of total mercury in muscle of Arctic char from lakes in various Arctic and subarctic regions (Bignert, 2001; Borg *et al.*, 2001; Mannio, 2001; Muir *et al.*, 2000, 2001b; Olsen *et al.*, 2003; Snorrason and Jónsson, 2000).

such variables on contaminant burdens is critical to a proper interpretation of spatial patterns.

Mercury concentrations in fish tissue are of particular interest, owing to the occasional posting of fish consumption advisories based on Hg as well as persistent organic pollutants (POPs). Thresholds for such advisories vary between countries and even within countries (for example, guidelines for subsistence consumption and commercial purposes may differ). The AMAP assessment on human health discusses such issues in relation to tolerable daily intake (AMAP, 2003a).

Factors affecting tissue concentrations of mercury in fish

In contrast to terrestrial and marine mammals, spatial comparisons of Hg in fish are usually done on the basis of muscle, rather than liver, because muscle is commonly eaten and because fish muscle contains a high proportion of MeHg. Also, while concentrations of most metals are generally higher in fish liver than fish muscle (Amundsen *et al.*, 1997; Zhou *et al.*, 1998) this difference is less pronounced for Hg.

Water chemistry is important. Higher concentrations of humic acids favor the transport and retention of Hg in the water phase as organic complexes, which may increase the availability of Hg for uptake and bioaccumulation in the food web. In the Mackenzie River Basin, higher fish Hg is observed in streams with higher humic acid content (Evans and Lockhart, 2001). Early studies of acidified lakes in Sweden and Norway suggested that acidification increased Hg accumulation in fish, especially where the food web had changed in response to acidification (Andersen *et al.*, 1986; Björklund *et al.*, 1984). This may be pertinent to Arctic regions subject to local anthropogenic acidification. As discussed in Section 4.4.4.2, sedimentary Hg is not a reliable predictor of fish Hg. Characterizing the bioavailable fraction of the sedimentary Hg may help to identify the controls over transfers between fish and sediment.

Mercury concentrations are sometimes related to the size and age of the fish (Hermanson and Brozowski, 1993; Lockhart *et al.*, 2000a; Muir *et al.*, 2000; Riget *et al.*, 2000a), although when growth rates are high this relationship may diminish or disappear due to biomass dilution (e.g., Bernatchez *et al.*, 1996). It is important to recognize that, generally speaking, larger fish do not necessarily contain higher levels of Hg than smaller fish of the same species either within individual lakes or among lakes in a region (Amundsen *et al.*, 1997; Evans and Lockhart, 2001; Skotvold *et al.*, 1997). While it is generally desirable to normalize contaminant data to fish length and age, this should only be done if the growth curve of the specific population under study is known.

Body burdens of Hg in fish generally increase with trophic level (Bruce and Spencer, 1979; Kidd *et al.*,

1995). Therefore, piscivorous fish such as burbot, pike, walleye (*Stizostedion vitreum*), and perch (*Perca fluviatilis*) tend to have higher Hg concentrations than species such as grayling and whitefish that generally feed lower in the food chain (Lockhart *et al.*, 2001a; Mueller *et al.*, 1995). However, in some species, shifts in feeding strategies over the lifetime of an individual can result in significant variation in element concentrations. For example, both Arctic char and some species of whitefish can have a bimodal size distribution within a single lake, usually driven by differences in feeding behavior. In these cases, higher concentrations of Hg are observed in piscivorous forms.

Arctic char

Mercury concentrations can be three to fifteen times higher in landlocked than sea-run Arctic char (see Section 4.4.5.3). For this reason, only landlocked Arctic char are discussed in this section.

Mercury concentrations are generally low (< 0.1 mg/kg ww) in char from Canada, Iceland, northern Sweden, and Finland (Figure 4·7). Particularly low concentrations (0.013 mg/kg ww) occurred in Lake Thingvallavatn, Iceland, among morphotypes feeding on benthos, with concentrations only slightly higher in the piscivorous morph from the same lake (0.09 mg/kg ww; Snorrason and Jónsson, 2000). In lakes in the Troms area (northern Norway) Hg concentrations in char muscle were about 0.03 mg/kg ww, similar to those in Abiskojaure (northern Sweden), while char from the northernmost lake sampled in Finmark had mean concentrations of 0.075 mg/kg ww (Fjeld *et al.*, 2001; Annex Table A10). Higher levels (0.15 to 0.8 mg/kg ww) were found in char from about a dozen lakes in Greenland, Chukotka-Lavrentiya (Russia), and the Faroe Islands, and lakes in the Canadian High Arctic (Lockhart *et al.*, 2001a; Muir *et al.*, 2001b, see also Annex Table A10). In Greenland, the Hg levels in char were higher in southwestern and West Greenland (Kitaa) lakes where uplands supported vigorous dwarf shrub heath (and higher soil and sedimentary Hg), than in northwestern and East Greenland (Tuna) lakes surrounded by sparsely vegetated fell-fields (means of 0.526 to 0.666 versus 0.133 to 0.260 mg/kg ww, respectively; Riget *et al.*, 2000a). This pattern is similar to that in lake sediments and humus soils from the same catchments, and highlights the importance of understanding local watershed conditions when interpreting coarse-resolution spatial data.

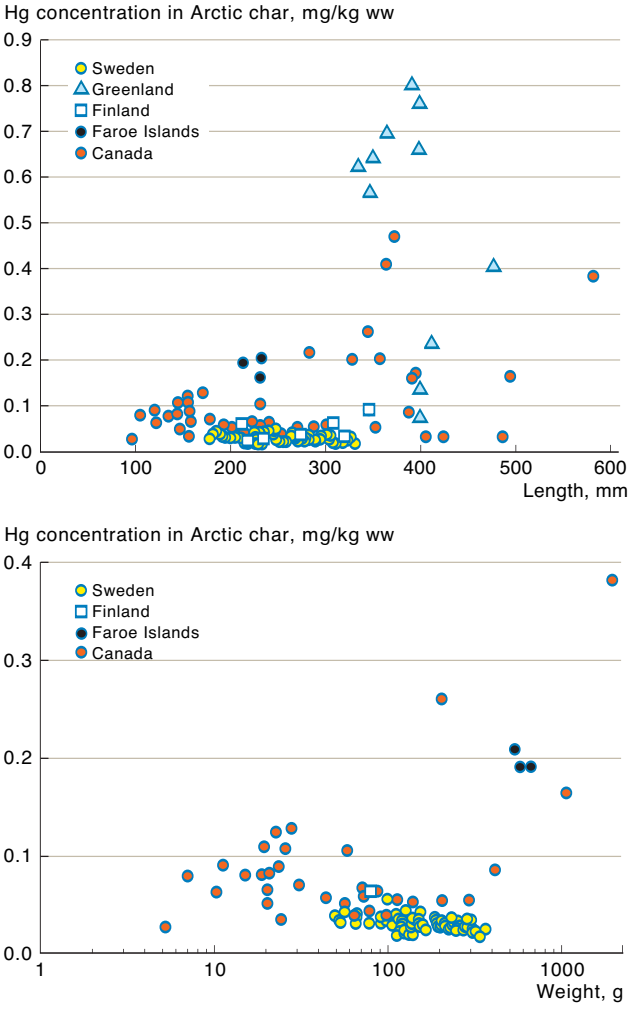


Figure 4.8. Relationship between total mercury concentration and length and weight of Arctic char from different Arctic and subarctic regions (Bignert, 2001; Borg *et al.*, 2001; Mannio, 2001; Muir *et al.*, 2000, 2001b; Olsen *et al.*, 2003; Riget *et al.*, 2000c).

Regional relationships between char size and Hg concentration are shown in Figure 4·8. The relationship between fish length and Hg concentration varies for different regions, and in some areas there appears to be no correlation. For example, the median Hg level in Abiskojaure (Àbeskojávre; Sweden: 68°18'N) is 0.03 mg/kg ww with low variability and little relationship to size. While char < 35 cm long generally contain < 0.25 mg/kg ww, longer char may have much higher Hg concentrations (Figure 4·8), perhaps owing to a shift from a benthic to a piscivorous habit (Hammar, 1998; Muir *et al.*, 2001b). Bimodal populations of Arctic char are common (Hammar, 2000; Hammar and Filipsson, 1988; Riget *et al.*, 2000d), and a single lake can have several different forms or morphs if the habitat is sufficiently complex (e.g., Sandlund *et al.*, 1992). For example, in northern Svalbard a pattern of ontogenetic niche shift in char was found at 10 to 15 years and a length of 20 to 30 cm, at which point the individual switched to a fast-growing, cannibalistic mode (Hammar, 2000). By contrast, in Greenland, a significant relationship between Hg concentration and char length was found that did not vary much for different populations (Riget *et al.*, 2000a). A similar slope was found for Arctic char in Myranar (Faroe Islands) (Olsen *et al.*, 2003).

Other freshwater fish

Whitefish (*Coregonus* spp. and *Prosopium cylindraceum*) are broadly distributed across the circumpolar Arctic. In this assessment, the several species of whitefish are treated together. Both Canada and Russia have substantial new contributions of data, and new data are also available for Finland, Norway, and the United States (Alaska). Spatial coverage is, therefore, better for whitefish than for the other species discussed in this section.

Most of the data reported on whitefish Hg from Arctic countries are between 0.05 and 0.20 mg/kg ww, with concentrations > 0.2 mg/kg ww only reported for lake whitefish (*C. clupeaformis*) and cisco from a few Canadian lakes. In at least one Canadian case (Lake Giaque),

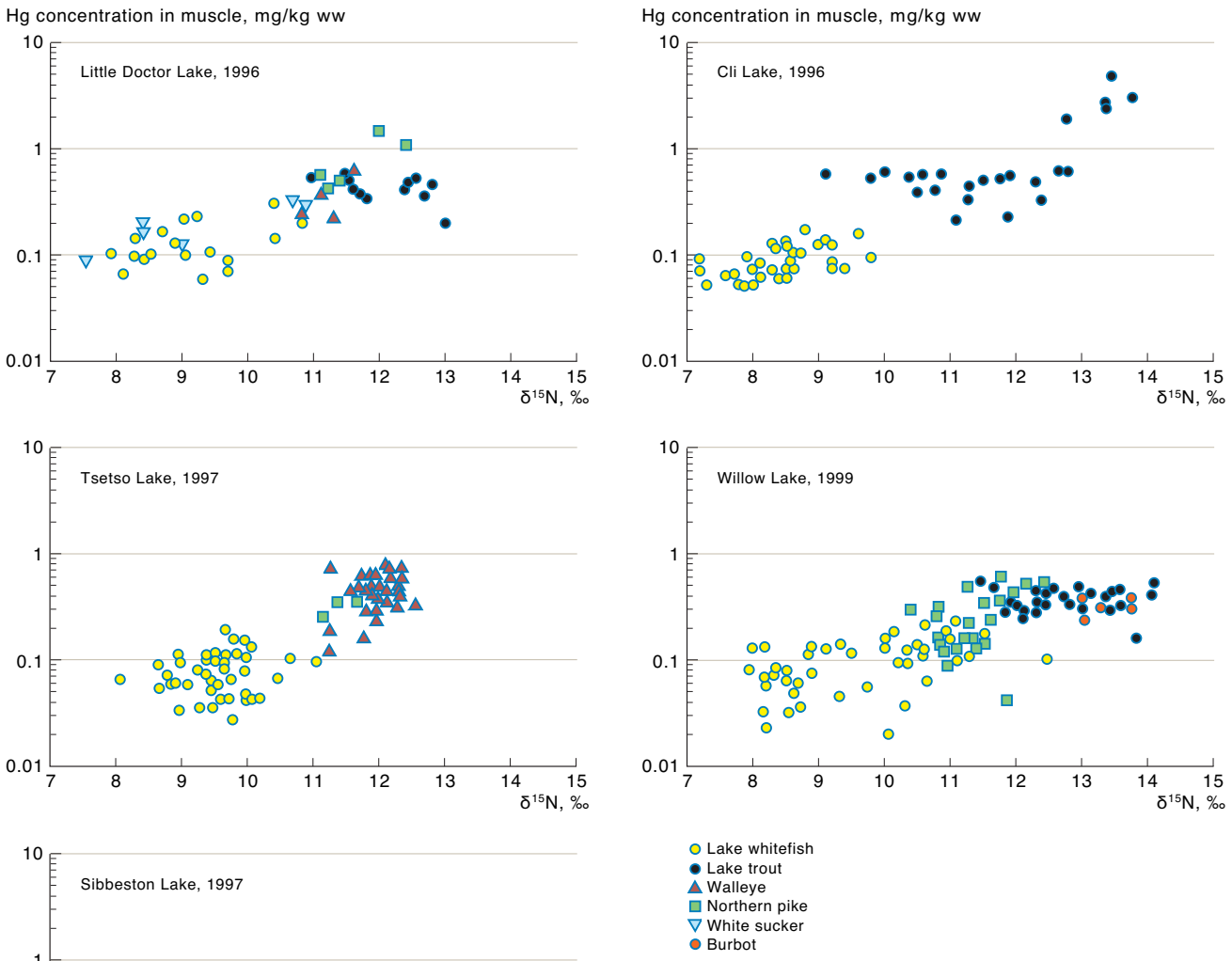


Figure 4·9. Total mercury concentrations in muscle versus trophic level (as reflected by $\delta^{15}N$) for fish from lakes in the Mackenzie River Basin (Evans and Lockhart, 2001).

high values (0.88 mg/kg ww) were probably linked to gold mining within the basin. Muscle concentrations < 0.05 mg/kg ww occurred in Alaska and on the Taymir Peninsula. The median Hg concentration in whitefish from northern to far eastern Russia was 0.08 mg/kg ww (see Annex Table A10), similar to the overall whitefish median for the current assessment (0.09 mg/kg ww). The species most commonly sampled in Russia was European whitefish, for which the lowest levels occurred in the Pechora Basin, and the highest levels on the Kola Peninsula and in one set of samples from the Taymir Peninsula (Khatanga). Mean Hg levels in broad whitefish (*C. nasus*) from Chukotka and the Taymir were 0.11 and 0.08 mg/kg ww, respectively.

Fish Hg concentrations are generally related to trophic status, as reflected by stable isotope analysis. The heavier isotope of nitrogen, ^{15}N, is progressively enriched from prey species to their predators, and thus provides a continuous, quantitative measure of trophic position within a food web. A shift in the ratio of ^{15}N to ^{14}N ($\delta^{15}N$) of about 3 to 5 per mill (‰) is generally thought to be associated with a change in trophic level (Hobson *et al.*, 1997a; Peterson and Fry, 1987). Significant within-lake relationships between log [Hg] and $\delta^{15}N$ were found across a range of Canadian freshwater fish species covering a broad range of trophic levels in northwestern Ontario and the Mackenzie River Basin (e.g. Evans and Lockhart, 2001; Kidd *et al.*, 1995; Figure 4·9).

In the Northwest Territories and Nunavut (Canada), char and whitefish Hg concentrations are usually <0.2 mg/kg ww, but piscivorous fish generally exceed this value and not infrequently exceed 0.5 mg/kg ww (Evans and Lockhart, 2001; Lockhart *et al.*, 2000a, 2001a; Snowshoe, 2001; Stoddart, 2001). Such species include walleye, northern pike, lake trout (*Salvelinus namaycush*) and inconnu (*Stenodus leucichthys*). Mercury concentrations are generally lower in the Yukon than the Northwest Territories for northern pike and lake trout (< 0.2 mg/kg ww for about half the Yukon lakes studied, see Annex Table A10).

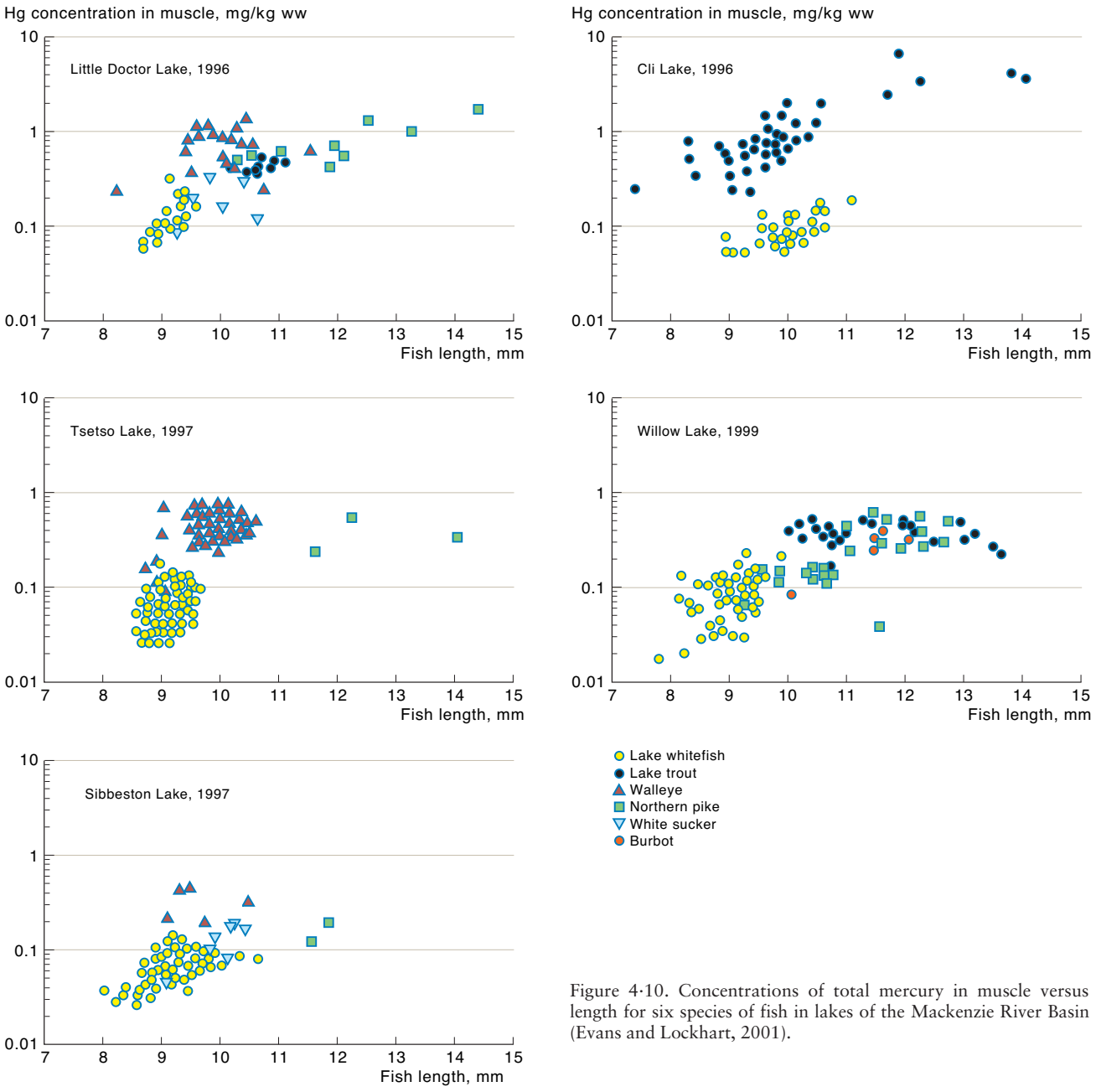


Figure 4·10. Concentrations of total mercury in muscle versus length for six species of fish in lakes of the Mackenzie River Basin (Evans and Lockhart, 2001).

In the Mackenzie River Basin, piscivorous fish approach 0.5 mg/kg ww at about 10 to 12 years, when their diet begins to include more small forage fish (Evans and Lockhart, 2001). For reasons that are not yet fully understood, spatial variability is high (Jensen *et al.*, 1997; Lockhart *et al.*, 2001a; Stoddart, 2001). No relationships have yet been found between fish Hg and factors such as lake size, drainage basin area, and wetland area, possibly because the lake sample is too small or too homogeneous to reveal such relationships (Evans and Lockhart, 2001). Detailed studies on several species from five lakes in the Mackenzie Basin also show that although relationships between fish length and Hg concentration sometimes occur within or among species, in other cases only weak or even no relationships are found (Figure 4·10). Using age instead of length improves the relationship with Hg concentration in some but not all cases (Evans and Lockhart, 2001).

New data on Hg in burbot are available for Canada, the Kemijoki River in Finland (Porvari and Verta, 1998), and the Kola and Taymir Peninsulas (see Annex Table A10). The overall median for all burbot muscle Hg studies is 0.2 mg/kg ww, with lowest mean concentrations in Russia (0.12 mg/kg ww), intermediate concentrations in Canada (0.23 mg/kg ww), and highest concentrations in Finland (0.43 mg/kg ww after excluding high outliers (ca. 1 mg/kg ww) from four particularly old fish). The Canadian burbot data are from two regions, with higher concentrations in the Fort Good Hope region than in Great Slave Lake (0.2 to 0.4 and 0.1 mg/kg ww, respectively). Ranges in Hg concentration were narrower in Russia, decreasing from 0.16 to 0.09 mg/kg ww from the Kola Peninsula to the Taymir Peninsula, which parallels the pattern for whitefish. Studies in the Dogrib region of the Northwest Territories found higher Hg concentrations in lake trout and lower concentrations in burbot and whitefish (Snowshoe, 2001; Stoddart, 2001), which is interesting as burbot is generally considered to feed quite high in the food chain.

In studies at the Kanuti National Wildlife Refuge (Alaska) on northern pike and Arctic grayling (*Thymallus arcticus*), no relationship was seen between length and Hg concentration (Mueller *et al.*, 1995). However, in four rivers of southwestern Alaska, studies on both species found increasing MeHg concentration with length, although the slope differed between the Yukon and Kuskokwim drainage basins (Naidu *et al.*, 2001a). Lithological sources may drive this difference. For any given size, Yukon River pike had significantly higher muscle concentrations of total Hg and MeHg than pike from the Kuskokwim River (mean MeHg of 1.56 versus 0.58 mg/kg ww). Similar patterns were found for Yukon versus Kuskokwim River grayling (mean MeHg of 0.25 versus 0.08 mg/kg ww, respectively). Whitefish were analyzed from the Kuskokwim River only and were found to have mean MeHg body burdens of 0.03 mg/kg ww, about one-third that of similarly sized grayling, probably due to differences in diet (ADF&G, 1994).

In the dammed Kemijoki River system in Finland (see Section 4.4.4.1) studies on four fish species have been in place since the early 1980s (Porvari, 1998). In 1994, none of the 58 whitefish but 35% of the 130 pike, 27% of the 76 burbot, and 15% of the 104 perch exceeded 0.5 mg/kg ww Hg in muscle. Mean Hg concentrations in the four species were 0.16, 0.47, 0.43, and 0.32 mg/kg ww, respectively. A minor but statistically significant increase in concentration was found below reservoir-affected areas for all species that paralleled differences in water chemistry (see Section 4.4.4.1). Older individuals occasionally contained high Hg concentrations (>1 mg/kg ww) even upstream of the reservoir (see Annex Table A10). Perch from lakes above the reservoir showed a marked biomass dilution effect, with Hg concentrations inversely related to growth rates.

In Sweden, studies of Hg concentration in piscivorous fish have been of particular interest due to regionally extensive national blacklisting of Swedish lakes re-

Table 4·1. Mean concentrations (plus standard deviations) of trace metals (mg/kg dw) and iron (weight percent) in organic-rich surficial sediments from selected Arctic shelf regions, together with levels reported by Long *et al.* (1995) at which adverse effects may be seen (modified from Naidu *et al.*, 1998, 2001b).

	n	V	Cr	Cu	Ni	Zn	As	Reference
Bering Sea	28		59±23	11±9	17.8±8.7	56±26	14.8±5.2	NOAA (unpubl. data)
Chukchi Sea	12	116±30	82±21	22±6	27±6	79±18		Naidu *et al.*, 1997
Beaufort Sea	23	152±26	89±14	33±9	47±11	98±18		Sweeney, 1984; Naidu, 1982
	13	115±17	78±10	24±4		93±13		Crecelius *et al.*, 1991
	62	117±24	63±13	27±7	34±7	96±18	16±7	Naidu *et al.*, 2001b
	7		87±12	29±8	37±9	115±22	27±16	Valette-Silver *et al.*, 1997
Pechora Sea	40	175±46	110±15	21±2	43±9	84±9	33±55	Loring *et al.*, 1995
	109	63±43	101±69	17±10	37±22			Ivanov *et al.*, 1999, 1997
Kara Sea	36	147±27	110±25	20±6	42±10			Loring *et al.*, 1995
	16		97±12	27±14	44±17	80±22		Esnough, 1996
Svalbard	15	248±11	153±5		50±1	107±3		Loring *et al.*, 1995
E. Greenland	10	162±65	118±45	46±32	59±29	89±20		Loring and Asmund, 1996; Loring *et al.*, 1995
W. Greenland	22	129±70	163±154	49±40	82±96	77±19		Loring and Asmund, 1996
W. Baffin Bay	12	91±32	63±19	29±8	22±9	61±14		Loring, 1984
E. Siberian Sea	24		68±14	16±6	26±8	85±25	18±6	Esnough, 1996
Laptev Sea	11		78±12	19±7	30±7	92±15	15±6	Esnough, 1996
	10			15±4	28±7	98±22		Nolting *et al.*, 1996
Effects range, median			370	270	51.6	410	70	Long *et al.*, 1995
low			81	34	20.9	150	8.2	Long *et al.*, 1995

	n	Cd	Pb	Fe	Mn	Co	Hg	
Bering Sea	28	0.09±0.03	8.5±2.8	3.35±1.84	682±436		0.025±0.023	NOAA (unpubl. data)
Chukchi Sea	12			3.46±0.64	295±37	26±5		Naidu *et al.*, 1997
Beaufort Sea	23			3.36±1.12	410±174	89±14		Sweeney, 1984; Naidu, 1982
	13	0.19±0.06	15±4					Crecelius *et al.*, 1991
	62	0.27±0.11	15±6	3.36±0.89	540±337		0.017±0.007	Naidu *et al.*, 2001b
	7	0.18±0.13	17±3	4.1±0.50	3747±5212		0.074±0.034	Valette-Silver *et al.*, 1997
Pechora Sea	40	0.11±0.05						Loring *et al.*, 1995
	109		24±18		301±268	8±6		Ivanov *et al.*, 1999, 1997
Kara Sea	36	0.11±0.07						Loring *et al.*, 1995
	16	0.09±0.03	14±3	4.39±1.41			0.028±0.009	Esnough, 1996
Svalbard	15	0.22±0.03						Loring *et al.*, 1995
E. Greenland	10	0.11±0.05	19±7					Loring and Asmund, 1996; Loring *et al.*, 1995
W. Greenland	22	0.15±0.16	18±8					Loring and Asmund, 1996
W. Baffin Bay	12							Loring, 1984
E. Siberian Sea	24	0.14±0.08	18±5	3.49±0.95			0.037±0.019	Esnough, 1996
Laptev Sea	11	0.11±0.06	20±5	3.97±0.92			0.037±0.016	Esnough, 1996
	10	0.06±0.02	18±3	4.06±1.00	206±175			Nolting *et al.*, 1996
Effects range, median		9.6	218				0.71	Long *et al.*, 1995
low		1.2	46.7				0.15	Long *et al.*, 1995

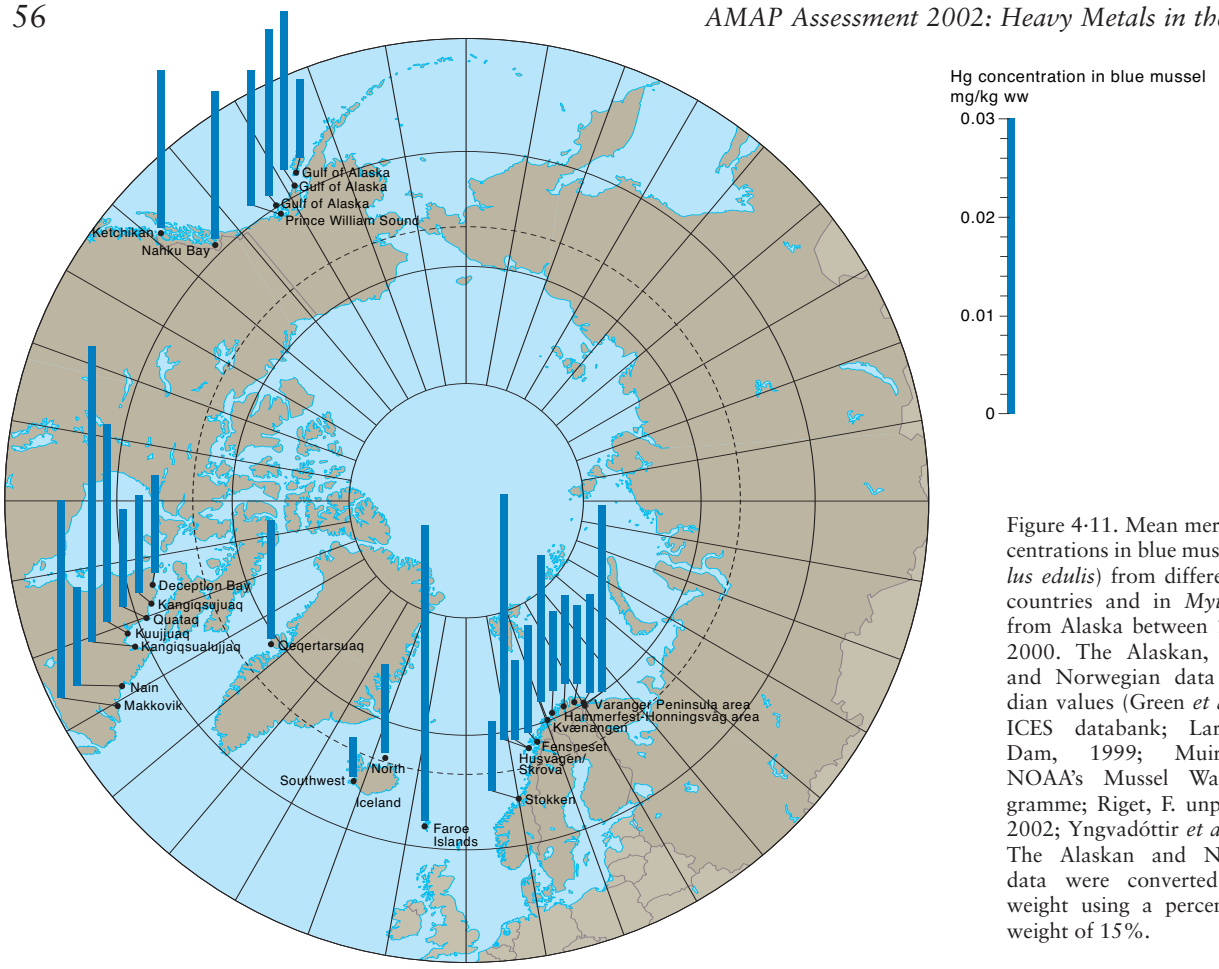


Figure 4·11. Mean mercury concentrations in blue mussel (*Mytilus edulis*) from different Arctic countries and in *Mytilus* spp. from Alaska between 1995 and 2000. The Alaskan, Icelandic and Norwegian data are median values (Green *et al.*, 2001; ICES databank; Larsen and Dam, 1999; Muir 2000; NOAA's Mussel Watch Programme; Riget, F. unpubl. data 2002; Yngvadóttir *et al.*, 2002). The Alaskan and Norwegian data were converted to wet weight using a percentage dry weight of 15%.

sulting from concerns about Hg (e.g., Johansson *et al.*, 1991, 2001). Piscivorous fish from lakes in southern Sweden have been especially well-studied, because the relatively high modern Hg concentrations (>0.5 mg/kg ww) are believed to be about five times higher than those 100 years ago (Johansson *et al.*, 1991). Recently, Johansson *et al.* (2001) reported significant declines in Hg in pike from southern lakes over the period 1988 to 1995 versus 1981 to 1987. This suggests that decreases in Hg deposition associated with national emission controls have now propagated through to the freshwater fisheries of the region. Although similar studies have not been done in northern Sweden, the general finding has great potential relevance for Arctic fisheries.

4.4.5. Mercury in the marine environment

In the first AMAP assessment, limited data were available for marine areas of Alaska, the Faroe Islands, Iceland, and Russia. Although Alaska, the Faroe Islands, and Iceland are better represented in this assessment, there are still relatively few data for Russia.

4.4.5.1. Marine sediments

In the first AMAP assessment, data on metal concentrations in surficial marine sediments were reported for Davis Strait and Baffin Bay between Canada and West Greenland (Kitaa), from Davis Strait between East Greenland (Tuna) and Iceland, and from the Barents Sea and north of Russia. There were few data for the East Siberian and Chukchi Seas, or most of the central Arctic Ocean and Canada Basin. Data are now available for the Chukchi Sea (Naidu *et al.*, 1997), Beaufort Sea (Crecelius *et al.*, 1991; Naidu, 1982; Naidu *et al.*, 2001b; Sweeney, 1984; Valette-Silver *et al.*, 1997), Kara Sea (Esnough, 1996), East Siberian Sea (Esnough, 1996), and Laptev Sea (Esnough, 1996; Nolting *et al.*, 1996). Sediment data are excluded from the following overview if they are known to be influenced by local point sources.

Naidu *et al.* (1998, 2001b) compared mean concentrations of trace metals in organic-rich sediments from selected Arctic shelves. Concentrations of Hg were fairly uniform (within a factor of three to four), with the possible exception of lower concentrations in the coarser (sandy) Bering Sea sediments. No Hg concentrations exceeded levels above which adverse effects are observed (Table 4·1).

Concentrations of trace metals in marine sediments depend on grain size, organic matter content, regional and local geology, and proximity to local (including natural) inputs. Surficial concentrations can also be affected by bioturbation and geochemical processes as well as ocean currents. For these reasons, Macdonald *et al.* (2000) concluded that marine sediments were not particularly good indicators of regional contamination.

4.4.5.2. Marine invertebrates

Marine invertebrates have been widely used as indicators of heavy metal pollution (Phillips, 1976). Mussels (*Mytilus* spp.) are a particularly common biomonitor in national programs, where ongoing monitoring occurs at selected locations. Examples include: NOAA's Mussel Watch Program in the United States (Cantillo *et al.*, 1999), the Norwegian State Pollution Monitoring Programme (Green *et al.*, 2001), and Iceland's ICES/OSPAR program (e.g., Yngvadóttir and Halldórsdóttir, 1998,

1999). In Greenland, blue mussel (*Mytilus edulis*) is used as a pollution indicator in mining areas, using both natural populations and transplants from clean to highly polluted sites (Johansen and Asmund, 2001; Riget *et al.*, 1997b).

In the first AMAP assessment, data were available mainly for bivalves, amphipods, and decapods. Blue mussel data are now available for several new Arctic locations, including Arctic components of national monitoring programs. Recent data on blue mussel are also available for Labrador and Nunavik, Canada (Muir *et al.*, 2000), the Faroe Islands (Larsen and Dam, 1999), and Greenland (Annex Table A13). No circumpolar trend in mussel Hg concentration is apparent (Figure 4·11), partly because within-region variation is generally greater than variation between regions. However, element accumulation is known to be influenced by sampling season (Phillips, 1976), location within the intertidal zone (Phillips, 1976), and mussel size (Riget *et al.*, 1996, 2000e). Using mussels as indicators thus requires a high degree of standardization in sampling procedures. As common standardization procedures were not used in the various studies, a rigorous spatial comparison is not possible. Thus, firm conclusions cannot be drawn at this time regarding spatial patterns of Hg concentration in mussels.

In scallops from Labrador and Nunavik, and in queen scallop (*Chlamys opercularis*) from the Faroe Islands, Hg concentrations were generally <0.05 mg/kg ww, in the lower end of the range previously observed in Arctic bivalves (see Table 7·A12 in AMAP, 1998).

4.4.5.3. **Marine fish**

Most data on heavy metals in marine fish in the first AMAP assessment were from Greenland, Canada, and to lesser extent Norway. Few data on marine fish from Alaska and Russia were available. Some of these data gaps have now been filled, and new data on one or more species are available for Canada, Iceland, Norway, Greenland, and the Faroe Islands (Green *et al.*, 2001; Larsen and Dam, 1999; Muir *et al.*, 1999a, 2000; Olsen *et al.*, 2003; Savinov *et al.*, 1998; Tsibulski *et al.*, 2001; Table 4·2).

Mercury concentrations in liver of shorthorn sculpin (*Myoxocephalus scorpius*) from the Faroe Islands increased with size, similar to findings for sculpin and several other fish species in Greenland (Riget *et al.*, 1997a). Concentrations in sculpin liver from the Faroe Islands were higher than those from Greenland (means 0.04 to 1.38 mg/kg ww versus 0.008 to 0.061 mg/kg ww; Annex Table A14), possibly due to the influence of local sources at particular locations (Larsen and Dam, 1999; Olsen *et al.*, 2003). Patterns for Se were similar. Concentrations of Se in sculpin liver from Greenland in 1999 and 2000 were 0.88 to 1.17 mg/kg ww, within the range previously observed (AMAP, 1998), while Se concentrations in sculpin liver from the Faroe Island were slightly higher (1.15 to 1.54 mg/kg ww; Olsen *et al.*, 2003).

Mercury levels in muscle of common dab (*Limanda limanda*) from Iceland, the Faroe Islands, and Norway were low (0.02 to 0.07 mg/kg ww), similar to those found previously (see Table 7·A13 in AMAP, 1998).

Data on Atlantic cod (*Gadus morhua*) are available for the Faroe Islands (Larsen and Dam, 1999), Iceland (ICES databank, Auðunsson *et al.*, 1997; Yngvadóttir and Halldórsdóttir, 1998, 1999), and Norway (Green *et al.*, 2001). Muscle Hg is in the range 0.011 to 0.128 mg/kg ww. Concentrations in Arctic cod (also known as polar cod, *Boreogadus saida*) from the Pechora Sea, Russia (Tsibulski *et al.*, 2001), were similar to concentrations previously reported for Arctic Canada and Greenland (0.02 mg/kg ww; AMAP, 1998) but higher than those in cod from the Barents Sea (AMAP, 1998). Mean Se concentrations in Pechora Sea cod were 0.23 mg/kg ww, generally within the range previously observed.

Muir *et al.* (1999a, 1999b, 2000) report generally low Hg levels in muscle of sea-run Arctic char in 1998 and 1999 from Labrador and northern Quebec. Means ranged from 0.03 to 0.07 mg/kg ww with no clear spatial patterns. Mercury concentrations in muscle of Canadian sea-run char were about a third those of landlocked populations (Muir *et al.*, 1999a, 2000), confirming earlier observations by Bruce and Spencer (1979). In southwestern Greenland the corresponding reduction for sea-run char is a factor of ten to fifteen times less (Riget *et al.*, 2000a).

Mercury concentrations in liver of flathead sole (*Hippoglossoides elassodon*) and fourhorn sculpin (*Myoxocephalus quadricornis*) collected between 1984 and 1986 from coastal Alaska were about 0.015 to 0.11 mg/kg ww and 0.1 to 0.12 mg/kg ww, respectively, similar to concentrations previously reported for marine fish

Table 4·2. New marine fish data sets.

Location	Species (tissue)	Reference
Southern Alaska, USA	Flathead sole (liver)	Meador *et al.*, 1994
Northern Alaska	Fourhorn sculpin (liver and stomach)	Meador *et al.*, 1994
Alaska	Arctic flounder (stomach contents)	Meador *et al.*, 1994
Labrador and northern Quebec, Canada	Sea-run Arctic char (muscle)	Muir *et al.*, 1999b; 2000
Greenland	Shorthorn sculpin (liver)	F. Riget, unpubl. data, 2002
Greenland	Greenland cod (muscle)	F. Riget, unpubl. data, 2002
Greenland	Atlantic salmon (muscle and liver)	F. Riget, unpubl. data, 2002
Greenland	Greenland halibut (muscle and liver)	F. Riget, unpubl. data, 2002
Greenland	Capelin (muscle and whole fish)	F. Riget, unpubl. data, 2002
Faroe Islands	Shorthorn sculpin (liver)	Olsen *et al.*, 2003
Faroe Islands	Atlantic cod (liver)	Larsen and Dam, 1999
Faroe Islands	Common dab (liver)	Dam, 2000
Iceland	Atlantic cod and Common dab (liver)	Audunsson *et al.*, 1997; Yngvadóttir and Halldórsdóttir, 1998; 1999; Yngvadóttir *et al.*, 2002
Norway	Atlantic cod (liver)	Green *et al.*, 2001
Norway	Common dab (whole fish)	Green *et al.*, 2001
Pechora Sea	Arctic cod and navaga (muscle and liver)	Tsibulski *et al.*, 2001

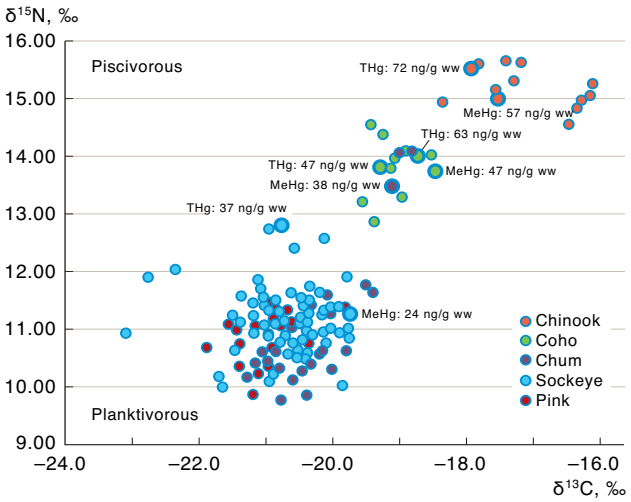


Figure 4·12. Trophic positioning of five species of Pacific salmon using $\delta^{13}C$ and $\delta^{15}N$ in adult muscle tissue (Satterfield and Finney, 2002) also showing total Hg (THg) and methyl-Hg (MeHg) values (marked symbols).

from other Arctic areas (AMAP, 1998). As expected, spatial patterns in Hg concentration were observed because some locations were specifically chosen to reflect the influence of local sources (Annex Table A14). In contrast, Se concentrations in sculpin were relatively high (about 0.9 to 1.6 mg/kg ww) compared to those reported previously for Arctic fish (Meador *et al.*, 1994; AMAP, 1998).

Pacific salmon (*Oncorhynchus* spp.) are common in southwestern Alaska. As in other salmon-bearing regions, all species are heavily used in local subsistence activities. Salmon put on most of their biomass (and thus accumulate most of their contaminant burdens) during their years at sea, returning to freshwater only to spawn and die. Total Hg and MeHg were analyzed in several salmon species returning to spawn in four Alaskan rivers (Naidu *et al.*, 2001a; Zhang *et al.*, 2001). Mean muscle total Hg concentrations were low, ranging from 0.034 to 0.096 mg/kg ww of which about 80% was MeHg. Mean liver concentrations were slightly higher (0.069 to 0.112 mg/kg ww). Chinook (king) salmon (*O. tshawytscha*) were the largest and oldest salmon sampled. The $\delta^{15}N$ signature indicated that chinook salmon fed at the highest trophic level (Satterfield and Finney, 2002) and generally had the highest levels of total Hg in muscle, which also correlated well with size (length) (Naidu *et al.*, 2001a; Zhang *et al.*, 2001). The relationships between Hg, $\delta^{15}N$, and $\delta^{13}C$ (Satterfield and Finney, 2002) suggest that the higher mean concentrations of Hg in chinook salmon are probably related to the higher trophic level at which they feed (Naidu *et al.*, 2001a; Figure 4(·12). Generally speaking, Chinook salmon and coho salmon (*O. kisutch*) had higher levels of total Hg in muscle than sockeye salmon (*O. nerka*), chum salmon (*O. keta*) and pink salmon (*O. gorbischa*). No clear differences were observed between species, trophic level, or size using liver tissue. Eggs had extremely low Hg concentrations in all species (total Hg < 0.016 mg/kg ww, MeHg < detection limit; Zhang *et al.*, 2001). No correlations were found between total Hg or MeHg and Se, lipids, and omega-3 fatty acids.

4.4.5.4. Seabirds

Seabirds (like waterfowl, game birds and raptors) pose special problems for the interpretation of spatial patterns, as most species migrate over long distances. Thus, body burdens reflect an unknown contribution from natural geochemistry and anthropogenic contamination of overwintering grounds, which can vary both within and between species by year. The influence of local geology also varies. Finally, changes in feeding behavior will affect body burdens (e.g., Kim *et al.*, 1996; Savinov *et al.*, 2003), particularly if there are dietary shifts between fish (which are generally lower in Cd and higher in Hg) and invertebrates (especially copepods and amphipods, which are generally higher in Cd and lower in Hg) (Dietz *et al.*, 1996).

In the first AMAP assessment, data were available for Greenland, Canada, and Norway, with only limited data for Arctic Russia and Iceland and no data for Alaska and the Faroe Islands. These gaps have been addressed to some extent in this assessment (Table 4·3). Data are also available for several new species; e.g., Arctic tern (*Sterna paradisaea*), northern pintail, long-tailed jaeger (*Stercorarius longicaudus*), parasitic jaeger (*S. parasiticus*), Arctic loon (*Gavia arctica*), spectacled eider (*Somateria fischeri*), Steller's eider (*Polysticta stelleri* (= *S. stelleri*)), bald eagle (*Haliaeetus leucocephalus*) and white-tailed sea eagle (*H. albicilla*). This extends the Hg data to 28 seabird species and the Se data to 19 seabird species. Information on Hg and Se in eggs has also been added.

In general, Hg concentrations in seabirds from the

Table 4·3. New seabird data sets.

Location	Species	Reference
Barents Sea	13 species	Savinov *et al.*, 2003
Chaun, northeast Siberia	11 species	Kim *et al.*, 1996
Alaska, Arctic Russia	4 eider species	Stout *et al.*, 2002
Arctic Canada	2 eider species	Wayland *et al.*, 1999a, 2001
Western Alaska	Spectacled eider	Trust *et al.*, 2000
Arctic Canada	4 species (eggs)	Braune *et al.*, 2002
Great Slave Lake, Canada	Herring gull (eggs)	Wayland *et al.*, 1999b
Faroe Islands	Black guillemot (liver/eggs)	Olsen *et al.*, 2003
Greenland	Black guillemot (liver)	F. Riget, pers. comm., 2002
Faroe Islands	Fulmar	Larsen and Dam, 2003
Barents Sea (northern Norway, Svalbard, northwest Russia)	10 species (eggs)	Barrett *et al.*, 1996
Northern Norway	4 species	T. Nygård, pers. comm., 2002; Wenzel and Gabrielsen, 1995
Aleutian Islands, Alaska, USA	Bald eagle (eggs)	Anthony *et al.*, 1999

Barents Sea (Savinov *et al.*, 2003) were lower than in Greenland, Canada, and northeast Siberia. Highest liver concentrations (1.7 to 3.03 mg/kg dw) were found in northern fulmar (*Fulmarus glacialis*) and razorbill (*Alca torda*) and lowest levels in little auk (also known as dovekie, *Alle alle*). Within the Barents Sea, spatial differences in Hg concentration were found for fulmar, black-legged kittiwake (*Rissa tridactyla*), and thick-billed murre (also known as Brünich's guillemot, *Uria lomvia*). Highest Hg concentrations were found in birds from Ny-Ålesund, Svalbard (Savinov *et al.*, 2003). Selenium levels also varied by species, with highest concentrations in common guillemot (also known as common murre; *U. aalge*) and razorbill, and lowest levels in little auk (Savinov *et al.*, 2003). Spatial differences in Se levels within the Barents Sea were found for several species (Savinov *et al.*, 2003).

Mean Hg concentrations in tissues of four species of eider (*Somateria* spp. and *Polysticta* spp.) from Alaska and Arctic Russia were 1.18 to 4.27 mg/kg dw (liver) and 0.09 to 1.3 mg/kg dw (kidney) (Stout *et al.*, 2002), within the range observed for corresponding species in Greenland, the Barents Sea, and northeast Siberia. Liver Hg concentrations in common eider from Nunavut, Canada were generally similar (1.46 to 3.32 mg/kg dw) (Wayland *et al.*, 2001). In contrast, Se concentrations in four species of eider from Alaska and Arctic Russia ranged from 7.85 to 124 mg/kg dw (liver) and 7.34 to 68.9 mg/kg dw (kidney) (Stout *et al.*, 2002; Trust *et al.*, 2000), higher than those observed in Canada (Wayland *et al.*, 1999a), Greenland (AMAP, 1998), and the Barents Sea (Savinov *et al.*, 2003).

Within the Aleutian chain of islands (Alaska), spatial patterns in Hg concentrations in unhatched bald eagle eggs were similar to those for *p,p'*-DDE, mirex, oxychlordane, and *trans*-nonachlor, with elevated levels associated with decreased reproductive success in the westernmost islands (Anthony *et al.*, 1999; R. Anthony, pers. comm., 2003; Annex Table A15). Of the several hypotheses suggested to account for this pattern, the authors discounted migration as populations are thought to be resident on Adak and Amchitka Islands. A direct or indirect Asiatic source has been suggested, owing to the increasing gradient in concentration westward. The diet of the westernmost birds was rich in marine birds such as fulmars and glaucous-winged gulls (*Larus glaucescens*) that themselves may carry elevated levels of these contaminants.

Similar data for white-tailed sea eagle are not available, although Norwegian feather Hg data are available for 1968 to 1995 (Nygård, 1997). A comparison with concentrations in archived samples suggests that the ban on agricultural use of Hg for seed dressings, coupled with the phasing out of Hg discharges from pulp mills and chlor-alkali plants, has resulted in decreased feather Hg concentrations in this species. Whether contamination from old, now discontinued, point sources are still affecting concentrations in this species is unknown. Comparable data for other regions would be useful.

Seabird liver and kidney from Chaun, northeastern Siberia, had particularly high Hg concentrations compared to Greenland and eastern Canada (Kim *et al.*, 1996), especially for herring gull (*Larus argentatus*; liver 4.01 mg/kg dw) and long-tailed duck (liver 27.1 mg/kg dw). This may be due in part to the fact that northeast

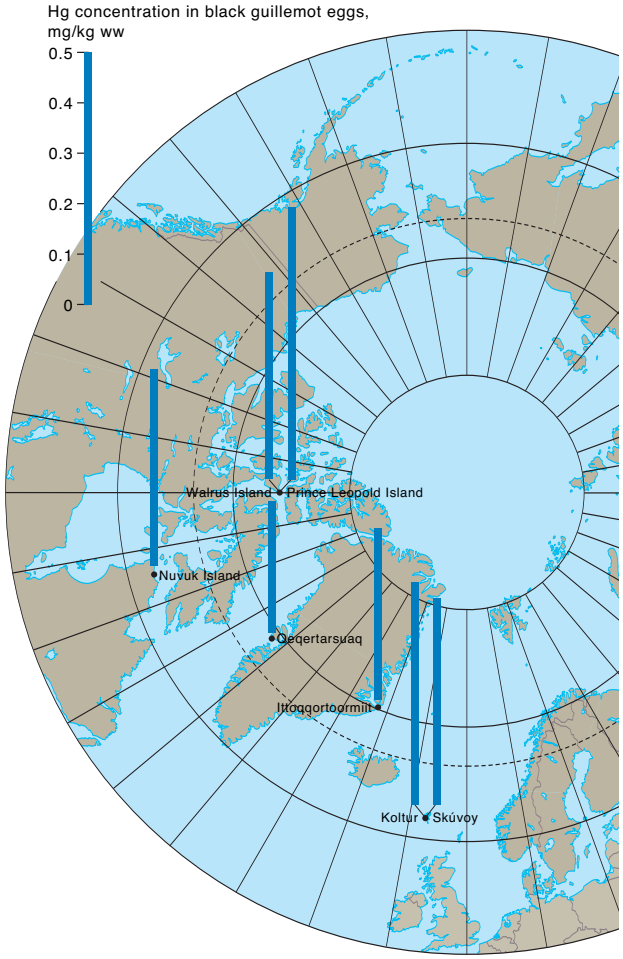


Figure 4·13. Arithmetic mean total mercury concentrations in black guillemot eggs collected in 1993 in Canada and in 1999 and 2000 in Greenland and the Faroe Islands. For the Canadian data, a percentage moisture content of 73 to 74% was used to convert from dry weight to wet weight (Braune *et al.*, 2002; F. Riget, unpubl. data 2002; Olsen *et al.*, 2001).

Siberian seabirds overwinter in southeast Asia while those from Greenland and eastern Canada overwinter in North America (Kim *et al.*, 1996). However, Hg deposits large enough for commercial exploitation are numerous in northeastern Siberia and may contribute to the high levels observed in Chaun (Titov, 1967).

Braune *et al.* (2002) studied Hg levels in eggs of glaucous gull (*Larus hyperboreus*), black-legged kittiwake, thick-billed murre, and black guillemot (*Cepphus grylle*) collected in 1993 in the Canadian Arctic. Increasing Hg concentrations were found at higher latitudes in the Canadian Arctic. Variability was higher for Se than Hg. Mercury concentrations in kittiwake and thick-billed murre eggs were 0.4 to 1.5 mg/kg dw, similar to concentrations in eggs from the Barents Sea (Barrett *et al.*, 1996). For guillemot, both Hg and Se concentrations were higher than in eggs from West and East Greenland (F. Riget, unpubl. data, 2002), although Hg concentrations were similar to those in the Faroe Islands in 1999 and 2000 (Olsen *et al.*, 2003) (Figure 4·13). Stable isotope analysis suggests that diet is unlikely to explain the increasing south-to-north gradient in Hg concentration. A similar study of Hg in eggs of ten seabird species from the Barents Sea in 1993, showed no clear regional differences (Barrett *et al.*, 1996).

Mean Hg concentrations of 2.66 mg/kg ww were re-

Table 4·4. New marine mammal data sets.

Location	Species	Reference
Alaska	Ringed seal, bowhead whale, beluga whale, polar bear	Woshner *et al.*, 2001a,b
Eastern Arctic Canada	Ringed seal	D. Muir, pers. comm., 2002
Labrador and northern Quebec	Walrus	Muir *et al.*, 2000
Arctic Canada	Beluga	Lockhart, 1999
Faroe Islands	Grey seal, pilot whale, white-sided dolphin	Dam, 2001; Dam and Bloch, 2000; Larsen and Dam, 1999; Olsen *et al.*, 2003
Greenland	Ringed seal, minke whale, narwhal, polar bear	F. Riget, pers.comm., 2002
Svalbard	Ringed seal	Fant *et al.*, 2001
Chukotka, eastern Arctic Russia	Ringed seal, bearded seal, spotted seal, grey whale	Melnikov *et al.*, 2002

ported for fulmar liver from the Faroe Islands (Larsen and Dam, 1999), similar to concentrations previously observed in Greenland and Canada (AMAP, 1998). Concentrations in black guillemot liver were 0.33 to 1.73 mg/kg ww, at the higher end of the range observed in the same species in Canada (AMAP, 1998) and Greenland (Annex Table A15).

4.4.5.5. Marine mammals

The importance of monitoring contaminant concentrations in marine mammals cannot be overstated, as they are key dietary routes for human exposure to Hg (as well as to POPs) in many Arctic communities. However, by themselves, marine mammals are not ideal environmental monitors of ambient conditions. Species are long-lived, and therefore bioaccumulate (and depurate) contaminants throughout their lives. Reported concentrations need to be accompanied by information on sex and age (which can be difficult to obtain), because both substantially affect interpretation. Subsistence communities that rely on marine mammals may have a local interest in supporting contaminant studies. In such cases, more thorough collection of ancillary information important to data interpretation can be achieved.

Because marine mammals often carry substantial concentrations of Hg, Cd, and POPs, there is a risk that these animals may be subject to toxic effects. Monitoring studies can increase the general understanding of the status of these populations with respect to potential biological effects, as well as providing baseline information relevant to pollution abatement.

Most data on metal levels in marine mammals in the first AMAP assessment were for Canada and Greenland. Few data were available for marine areas of Alaska and Norway and almost none for Russia. These gaps have been addressed to some extent in this assessment (Table 4·4). Mercury data from nine seal species, ten whale species, and polar bear *(Ursus maritimus)* are now available.

Mercury concentrations in marine mammals generally decrease in the order liver → kidney → muscle. The exception is polar bear, for which highest concentrations occur in the kidney. The proportion of total Hg as MeHg in marine mammals varies by tissue, species, and location (Wagemann *et al.*, 1998). Mercury concentrations are given as total Hg except where specified, because concentrations of the more toxic MeHg are reported in only a few cases (Wagemann *et al.*, 1998).

Liver tissue from ringed seal *(Phoca hispida)* was analyzed for animals collected in the late 1990s from Barrow (Alaska), Labrador and the eastern Canadian Arctic, Greenland, Svalbard, and Chukotka (Russia) (Fant *et al.*, 2001; Melnikov *et al.*, 2002; D. Muir, pers. comm., 2002; Muir *et al.*, 2000; F. Riget, unpubl. data, 2002; Woshner *et al.*, 2001a). Figure 4·14 shows Hg concentrations in ringed seal liver for various locations mean-adjusted to five-year old animals for most locations. Contrary to the generally declining west-to-east pattern across Arctic North America for marine mammals, Hg levels in ringed seal liver were lower in Alaska than in the eastern Canadian Arctic. Some eastern locations had particularly high concentrations (e.g., 10 to 20 mg/kg ww at Grise Fjord, Ungava Bay, and Kangirsuk,

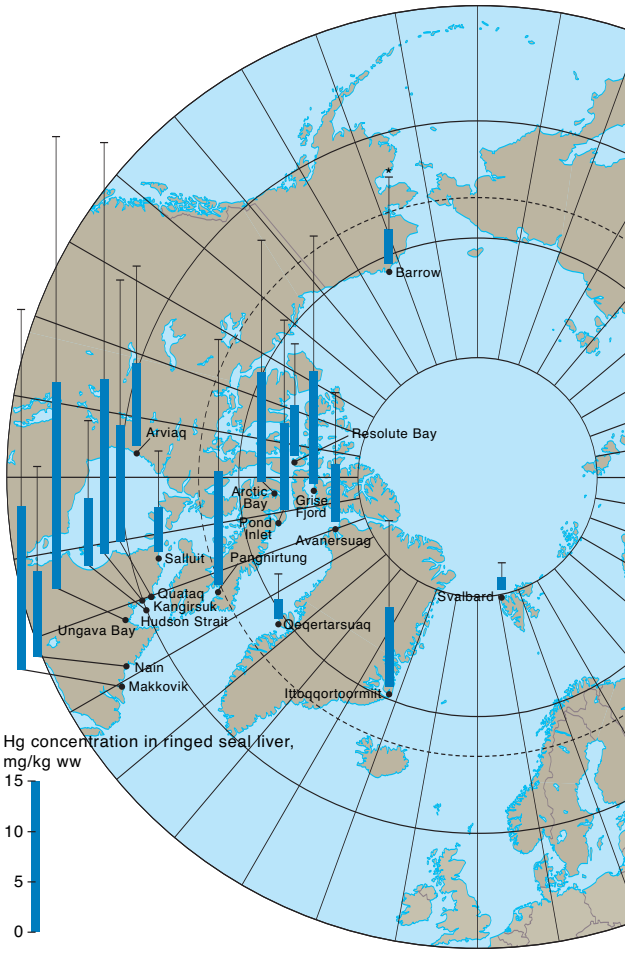


Figure 4·14. Mean concentrations of total mercury in ringed seal liver. Data for Greenland and Canada were collected during 1999 to 2000, and for Alaska during 1995 to 1997. Data are mean-adjusted to five-year old animals unless marked by an asterisk indicating unadjusted data (recalculated from Muir *et al.*, 1999b; Muir, D. pers. comm., 2002; Riget, F. unpubl. data, 2002; Woshner *et al.*, 2001b).

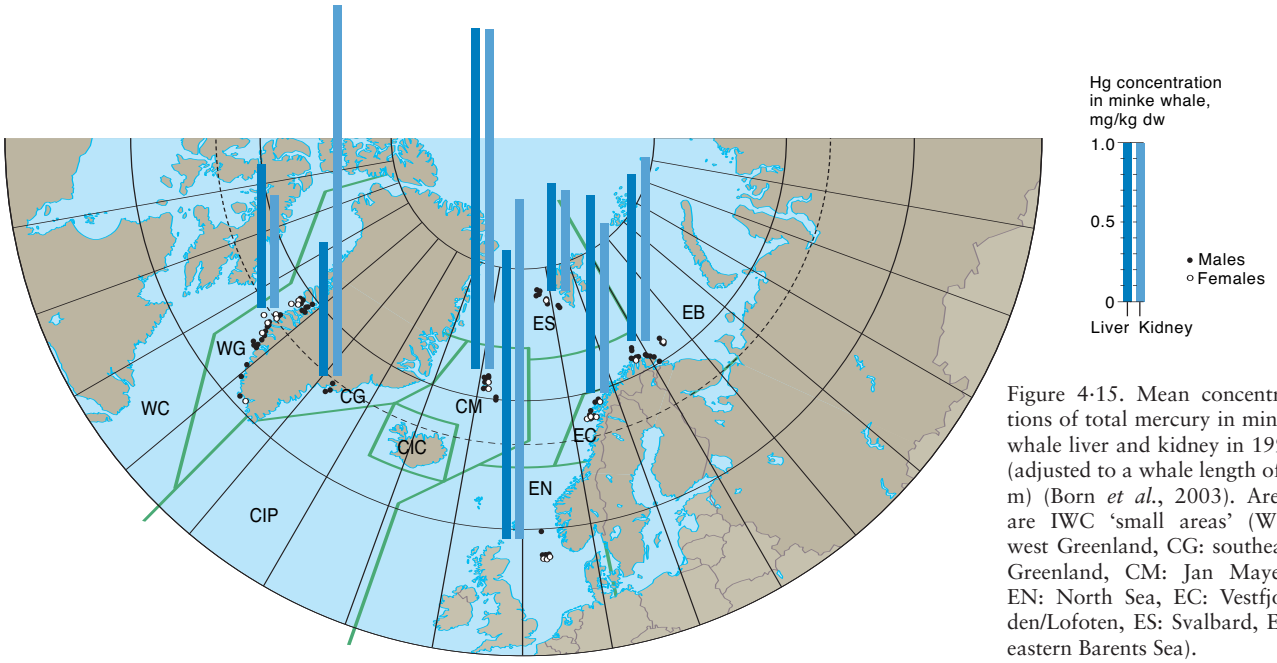


Figure 4·15. Mean concentrations of total mercury in minke whale liver and kidney in 1998 (adjusted to a whale length of 7 m) (Born *et al.*, 2003). Areas are IWC 'small areas' (WG: west Greenland, CG: southeast Greenland, CM: Jan Mayen, EN: North Sea, EC: Vestfjorden/Lofoten, ES: Svalbard, EB: eastern Barents Sea).

Labrador). Mercury concentrations in seals collected in northern, northeastern, and especially central western Greenland appeared lower than in the eastern Canadian Arctic. Ringed seal from Svalbard had the lowest Hg levels. No spatial pattern in Se was observed in ringed seal tissues in a 1998 study covering seven sites in Labrador and northern Quebec (Muir *et al.*, 2000). Selenium concentrations in ringed seal from central western and East Greenland (Tuna) show the same spatial pattern as Hg with higher concentrations in East Greenland (Annex Table A16).

For grey seal (*Halichoerus grypus*), Hg concentrations in muscle (0.37 to 2.31 mg/kg ww) and liver (11.6 to 155 mg/kg ww) from the Faroe Islands (Larsen and Dam, 1999) were similar to those in grey seal at Sable Island, Canada (Sergeant and Armstrong, 1973), and were higher than those from Jarfjord, Norway (Skaare, 1994). Mean Hg concentrations in liver of bearded seal (*Erignathus barbatus*) from Chukotka (Russia) were 29.4 mg/kg ww (Melnikov *et al.*, 2002), within the range of the few data previously reported for bearded seal from other locations (AMAP, 1998). Mean Hg concentrations in liver of spotted seal (*Phoca largha*) from the same area were lower (21.1 mg/kg ww; Melnikov *et al.*, 2002), although still higher than typically observed in ringed seal liver (e.g., Figure 4·14).

Beckmen *et al.* (2002) reported total Hg concentrations in fur of depleted populations of northern fur seals (*Callorhinus ursinus*) from the Pribilof Islands. Comparisons with declining and thriving populations of Steller sea lions (*Eumetopias jubatus*; Prince William Sound, Alaska and southeast Alaska, respectively) showed higher concentrations in northern fur seal pups (3.15 to 8.14 versus 0.9 to 3.14 mg/kg ww).

Mercury concentrations in walrus (*Odobenus rosmarus*) from Nunavik (Labrador) in 1999 were 2.02 mg/kg ww in muscle, 2.64 mg/kg ww in liver, and 0.31 mg/kg ww in kidney (Muir *et al.*, 2000), comparable to concentrations previously reported by Wagemann *et al.* (1996) for the same population.

Recent data on Hg concentrations in bowhead whale (*Balaena mysticetus*) reported by Krone *et al.* (1999) and Woshner *et al.* (2001b) indicate levels comparable to those for other baleen whale species reported in the first AMAP assessment (e.g., liver 3.8 to 6.4 mg/kg dw). Similarly, Hg in the liver of one grey whale (*Eschrichtius robustus* (= *E. gibbosus*)) reported from Chukotka (Russia) was 0.50 mg/kg ww (Melnikov *et al.*, 2002), also within the range previously reported for other baleen whales (AMAP, 1998).

Mercury levels in liver of minke whale (*Balaenoptera acutorostrata*) collected in 1998 from the North Atlantic area (West Greenland, East Greenland, Jan Mayen, Svalbard, the Norwegian coast, the North Sea, the Barents Sea) were 0.59 to 2.04 mg/kg dw (Born *et al.*, 2003) similar to those reported by Dietz *et al.* (1997) and Hansen *et al.* (1990). Selenium levels in muscle, liver, and kidney of minke whale were also similar (Born *et al.*, 2003; Dietz *et al.*, 1997; Hansen *et al.*, 1990). Mercury concentrations in whales from the area around Jan Mayen and the North Sea were generally high for all tissues, but low in whales from West Greenland and Svalbard (Figure 4·15). Selenium concentrations were generally highest in whales from the North Sea (e.g., kidney 9.23 mg/kg ww) and lowest in whales from West and southeastern Greenland (e.g., kidney 5.59 and 6.06 mg/kg ww, respectively), with intermediate values in whales from Svalbard, Jan Mayen, the Norwegian coast, and the Barents Sea.

Recent data on Hg concentrations in beluga whale *(Delphinapterus leucas)* have been reported for Alaska (1992 to 1996), the Mackenzie Delta, Canada (1993 to 2001), and three sites in eastern Canada (Lockhart, 1999; Lockhart *et al.*, 2001b; D. Muir, pers. comm., 2002; Woshner *et al.*, 2001b). Mean Hg concentrations in Alaskan beluga were 1.16 mg/kg ww in muscle, 12.4 mg/kg ww in liver, and 4.58 mg/kg ww in kidney, similar to levels previously observed in western Canada. The Mackenzie Delta data fit well with the generally decreasing west-to-east gradient for Hg and MeHg in beluga from Canada (Muir *et al.*, 1999a; Wagemann *et al.*, 1996; 1998). Relatively high concentrations were also found in whales from southern Hudson Bay (Lockhart *et al.*, 2001b). Both the Beaufort coast and the southern

Hudson Bay area are influenced by large freshwater drainage. This suggests a link with freshwater input (Lockhart *et al.*, 2001b). Spatial patterns may be further complicated by the existence of several distinct stocks of beluga in eastern Canada (Lockhart *et al.*, 2001b).

Mercury and Se concentrations in muscle, liver, and kidney of 53 narwhal (*Monodon monoceros*) from central West Greenland sampled in 1993 were not significantly different from those in 88 narwhal from northwest Greenland sampled in 1984 to 1985 (Riget *et al.*, 2002). For example, Hg levels in West Greenland narwhal muscle were 0.28 to 1.15 mg/kg ww, similar to concentrations previously reported for narwhal from Arctic Canada (AMAP, 1998).

Mercury concentrations in muscle of adult pilot whale (*Globicephala melas* (= *G. melaena*)) collected in 1997 and 1999 from the Faroe Islands were between 1.63 and 3.46 mg/kg ww (Dam, 2001; Dam and Bloch, 2000; Larsen and Dam, 1999; Olsen *et al.*, 2003), about the same as concentrations observed in the late 1970s (Anon, 1984/1985; Juelshamn *et al.*, 1987) but higher than for Newfoundland (Muir *et al.*, 1988). Selenium concentrations were between 0.55 and 0.81 mg/kg ww for adults, somewhat higher than for the Faroe Islands (Juelshamn *et al.*, 1987) but similar to Canadian levels (Muir *et al.*, 1988). Liver concentrations were considerably higher. For example, Caurant *et al.* (1994) report average liver levels in adult females 20 years and older to be as high as 200 mg/kg ww.

In Alaska, concentrations of divalent Hg and MeHg were 0.09 mg/kg ww in muscle of polar bear from Barrow (Woshner *et al.*, 2001a) and 0.4 mg/kg ww in muscle of adult male bears from northern and western Alaska (see Annex Table A16). These levels are lower than for western Canada (AMAP, 1998) but higher than observed in bears from the eastern Chukchi Sea (Lentfer and Galster, 1987). Mercury concentrations in muscle of polar bear from East Greenland collected in 1999 to 2000 were 0.09 to 0.13 mg/kg ww for muscle, confirming the relatively low levels previously reported (AMAP, 1998). Concentrations in liver (15.6 vs. 3.79 mg/kg ww) were much higher than in muscle.

Selenium data are usually available if Hg is reported because Se is considered an antagonist to Hg. Selenium is believed to play an important role in detoxification of Hg via the formation of mercuric selenide complexes (Björkman *et al.*, 1995; Wagemann *et al.*, 1998). Interpreting Hg data is thus strengthened if concentrations of both analytes are known. In marine mammals, Se accumulates with size and age, especially in liver (relative to muscle and kidney), but there is no indication that Se levels increase with increasing trophic level (Dietz *et al.*, 1996). In general, seals have approximately equal Se concentrations in kidney and liver, concentrations in both being greater than in muscle. In whales the order is liver → kidney → muscle, and in polar bear kidney → liver → muscle. In most Arctic marine mammal samples, Se is present in a substantial surplus compared to Hg on a molar basis (Dietz *et al.*, 1997). However, in marine mammals from Greenland with high Hg concentrations (above about 10 nmol/g; 2 mg/kg), a 1:1 Se:Hg molar ratio was found (Dietz *et al.*, 2000). In liver of pilot whales from the Faroe Islands, lower ratios were found (0.65 to 0.89). Whales from schools with higher mean liver Hg (84 versus 56 mg/ kg ww) typically had lower Se:Hg ratios (Caurant *et al.*, 1994). Overall, Se and Hg were correlated both to each other and to age in both liver and kidney (Caurant *et al.*, 1994). Molar ratios of Se to Hg were lowest in lactating females, with a minimum of 0.36 (Caurant *et al.*, 1996). If anthropogenic release of Hg to the environment continues to increase, species with high Hg burdens should be monitored to determine whether the protection conferred by Se is maintained in target tissues.

Recent data on Se concentrations have been reported for liver, kidney, muscle, and blubber in Alaskan ringed seal and polar bear and for the same tissues plus epidermis in Alaskan bowhead and beluga whale (Woshner *et al.*, 2001a). Concentrations are generally within ranges previously reported for marine mammals (Woshner *et al.*, 2001a; 2001b). In these Alaskan data, hepatic Se:Hg molar ratios vary for different species but are generally greater than unity.

The new data provide evidence that Hg concentrations in marine biota exhibit broad regional patterns with high sub-regional variability. Several causes or combinations of causes have been proposed to explain this variation. In addition to local sources of contamination, spatial differences in sedimentary geology are strongly suspected to drive observed Hg patterns across the Canadian Arctic for polar bear, ringed seal, and beluga (Muir *et al.*, 1999a). In Greenland, an increasing south-to-north trend in Hg was found in seabirds and marine mammals (Dietz *et al.*, 1996), despite the lack of corresponding differences in sediment concentration (Dietz *et al.*, 1996). This trend was not as evident in the 1994 to 1995 AMAP data (Riget *et al.*, 2000e). The proportion of total Hg held in the form of bioavailable MeHg may help explain some of the differences, although there is currently insufficient information on MeHg to evaluate spatial patterns. Feeding behavior has also been proposed as a factor influencing spatial patterns (e.g., Muir *et al.*, 1995). Stable isotope analysis of tissues provides a tool for evaluating trophic position and food source (Hobson *et al.*, 1994, 1997b) and has been included in several studies to help interpret contaminant levels (e.g., Braune *et al.*, 2002; Muir *et al.*, 1995; Riget *et al.*, 2002).

4.5. Cadmium

4.5.1. Atmospheric cadmium

Only two stations have reported more than one year of data on Cd in aerosol to the NILU data center. Observed concentrations vary within a relatively narrow range (0.01 to 0.2 ng/m^3) with the Icelandic site reporting higher and more variable concentrations than the site on Svalbard.

Arctic air samples from continuous flow air filtration systems have been analyzed in several programs, although differences exist between collection years, filtration duration, flow rates, filter sizes, and analytical methods, making direct comparisons difficult. For example, in Finland, weekly samples were taken between 1996 and 2000 on 3 μm Teflon filters using an air flow of about 50 L/min and analyzed by ICP-MS (Leppänen, 2002). The mean Cd content of particulates averaged 0.038 ng/m^3, with a maximum of 0.357 ng/m^3. A ship-

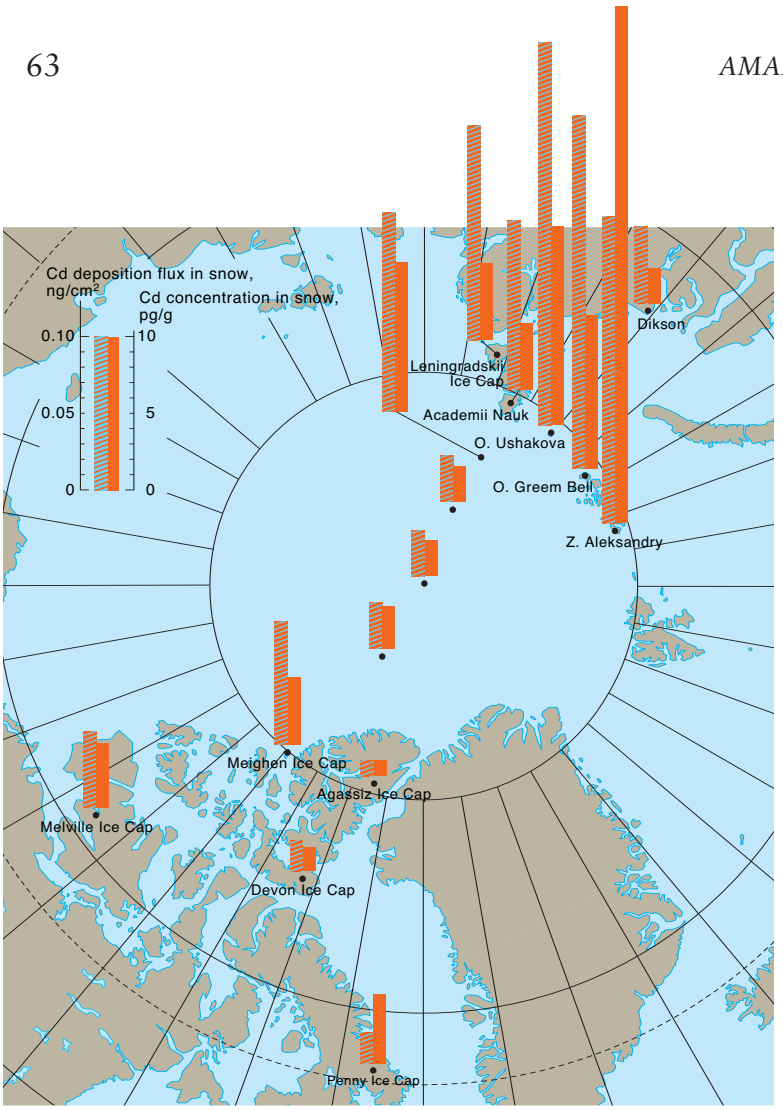


Figure 4·16. Cadmium concentrations (weighted averages for a snow accumulation period of about nine months; mean, see Annex Table A2) and deposition calculated over the period of snow accumulation (i.e., the nine months from the end of the previous summer to late spring of the sampling year) in surface snow across the Russian and Canadian Arctic Islands and the Arctic Ocean. Where samples were collected repeatedly over several years (e.g., the Agassiz ice cap) concentrations were averaged between individual years. (Koerner *et al.*, pers. comm., 2002).

based air sampling program in 2001 in the Barents and White Seas analyzed FPA filters exposed for 47 to 72 hours to an airflow of up to 60 m^3/hr by flameless AA. Cadmium concentrations averaged 0.02 ng/m^3, with a maximum of 0.038 ng/m^3 (Golubeva *et al.*, 2002). Such data are useful baseline information, but are not appropriate for determining spatial patterns.

4.5.2. Cadmium in precipitation

A spatially extensive snow-sampling program covering the Canadian and Russian Arctic islands and the Arctic Ocean was undertaken between 1993 and 1996. Most snow samples were from the interior regions of the polar ice caps, collected between late March and early May. Surface snow was considered a better indicator of spatial patterns in contaminants from long-range atmospheric transport than sub-surface snow, which can retain a fingerprint from soil or loess deposited in summer from local, seasonally snow-free, areas. Cadmium concentrations were generally <5 pg/g on the Canadian ice caps and sea-ice transect, reaching a maximum of 38.6 pg/g at Aleksandry Island (Figure 4·16; Koerner *et al.*, 2002). Total Cd deposition showed a similar pattern.

Snow collections from four northwestern Arctic Alaskan estuaries were also made during 1995 to 1996 (Garbarino *et al.*, 2002). Cadmium, like Hg, was typically present in higher concentrations in marine snow, relative to snow from Arctic terrestrial environments south of the Beaufort Sea. Unlike Hg, however, Cd concentrations were higher on the Beaufort (versus Chukchi) Sea. Enrichment factor analysis using Mn as the normalizing element indicated significant Cd enrichment (14 to 45 times) in these samples. Also, Cd was elevated relative to strontium (Sr), suggesting a non sea-salt (i.e., possible atmospheric) contribution for this element. Unfortunately, comparisons cannot be made between this study and that of Koerner *et al.* (2002) due to differences in methods and reporting conventions.

Studies of precipitation have been made for eight catchments in Arctic Finland, Norway, and Russia as part of the Kola Ecogeochemistry Project (see Section 4.4.3.2; Reimann *et al.*, 1997c). Five of the study catchments were located in Russia and Norway near heavily industrialized areas of the Kola Peninsula, with the other three in Finland further from point sources on the Kola Peninsula. The chemical composition of rainwater, snowmelt water, and snow filter residue (but not dry deposition) was studied in triplicate at five locations in each catchment, and volume-weighted annual budgets were constructed. For the two Finnish catchments considered background for the region, annual Cd deposition was 0.05 to 0.4 kg/km^2, with comparable contributions from snowmelt water and rainwater (Chekushin *et al.*, 1998).

In the NILU dataset, precipitation-weighted mean annual concentrations of Cd are almost all <0.2 µg/L, and generally about half that. Slightly higher values near the border of the Kola Peninsula at Svanvik, Norway may reflect Kola emissions. Generally speaking, data from Svanvik were about two to ten times higher than

those of the Kola Ecogeochemistry Project, probably because the latter used filtered rainwater samples (Reimann *et al.*, 1997c). The Kola Ecogeochemistry project indicated concentrations of <0.02 to 0.05 µg/L just west of Svanvik, which contrasts with the findings at Monchegorsk of 0.32 to 5.11 µg/L (Reimann *et al.*, 1997c). Substantially higher mean annual values were seen at Irafoss, Iceland, in 1994 and 1995 (0.69 and 11.31 µg/L, respectively) (Berg and Hjellbrekke, 2001), possibly related to volcanic emissions.

4.5.3. Cadmium in the terrestrial environment

4.5.3.1. Soils and peats

Although Cd is easily leached from soils (Borg and Johansson, 1989; Niskavaara *et al.*, 1997), Steinnes *et al.* (1997) found a strong relationship between Cd and organic matter in Norwegian humus soils, possibly due to immobilization in fungal biomass. Based on samples taken north of the Arctic Circle (Andersson *et al.*, 1991), background concentrations of Cd in surface humus were estimated at 0.17 mg/kg dw, somewhat higher than concentrations reported from remote locations in Greenland (0.04 to 0.10 mg/kg dw). Relative to these background concentrations, Cd was only moderately elevated (two to four times) in northern Sweden, whereas concentrations in southern Sweden were five to ten times higher than background (Andersson *et al.*, 1991).

In contrast to Hg (see Section 4.4.3.1), all three Norwegian surveys showed decreasing Cd concentrations in the surface humus of forest podzol soils from south to north (1977, 1985, 1995). This has been attributed to higher loadings from long-range atmospheric transport in southern Norway (Johansson *et al.*, 2001; Steinnes *et al.*, 1997). Parts of northern Norway closest to the Russian border may also be influenced by Russian smelter emissions to the east. For example, the Kola Ecogeochemistry Project found Cd concentrations of 0.05 to 1.16 mg/kg dw in O-horizon soils from the Norwegian catchment closest to the Russian border, although the median was low (0.24 mg/kg dw) and within-catchment variability was high (Reimann *et al.*, 1997a, 1998). Similarly, the range in the Finnish catchment closest to the Russian border was 0.09 to 1.16 mg/kg dw with the same median. In contrast, two Finnish catchments further west had the same medians but lower maxima (0.65 and 0.7 mg/kg dw).

In Alaska, a sampling program with different objectives measured contaminant concentrations in pooled vertical increments of ten soil cores from watersheds in three regions; the western Arctic Coastal Plain, the western Arctic Foothills, and the eastern Arctic Foothills (Ford *et al.*, 1997). The 0 to 5 cm increments included surface vegetation. Cadmium concentrations ranged from 0.21 to 0.80 mg/kg dw in the different catchments. Corresponding samples from five catchments on the Taymir Peninsula (far from Norilsk) ranged from 0.12 to 0.53 mg/kg dw (Ford *et al.*, 1997; Allen-Gil *et al.*, 2003).

4.5.3.2. Mosses and lichens

Cadmium concentrations in feather mosses from northern Scandinavia have continued to decline. In 2000, Cd concentrations in *Pleurozium schreberi* from northern

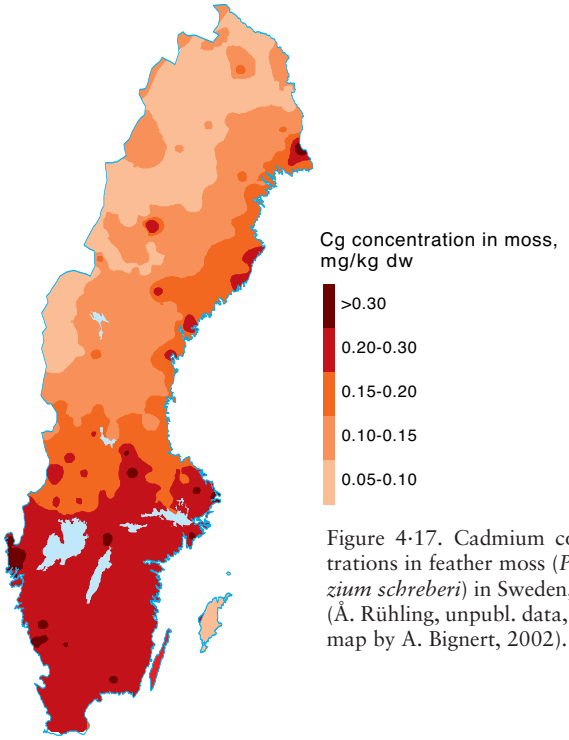


Figure 4·17. Cadmium concentrations in feather moss (*Pleurozium schreberi*) in Sweden, 2000 (Å. Rühling, unpubl. data, 2002; map by A. Bignert, 2002).

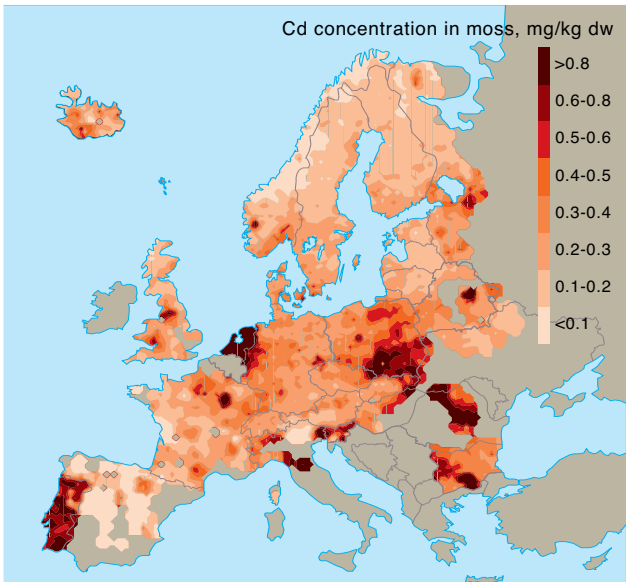


Figure 4·18. Cadmium concentrations in European feather mosses, 1995 (Rühling and Steinnes, 1998).

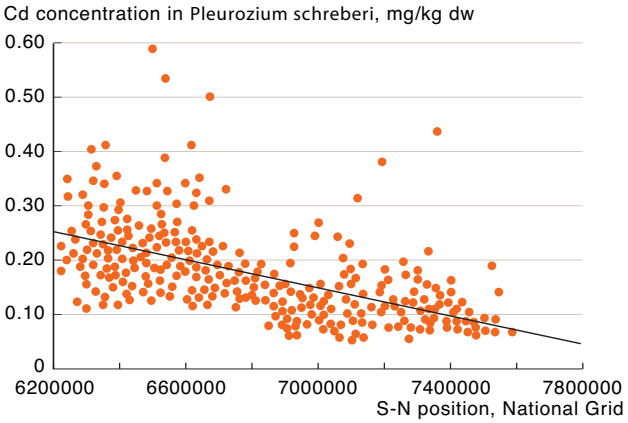


Figure 4·19. Relationship between cadmium concentration in *Pleurozium schreberi* and latitude in Sweden, 2000 (significance of correlation: $p < 0.0001$; significance of regression: $p < 0.00001$) (Å. Rühling, unpubl. data, 2002).

Table 4·5. Metal concentrations (mg/kg dw) in reindeer lichen *(Cladonia* spp.*)* on the Kola Peninsula, 1996 (Tsibulski *et al.*, 2001) and the Faroe Islands, 1996 (Larsen and Dam, 1999).

	Cu	Zn	Mn	Fe	Cd	Pb	Ni	Co	Cr	As	V	Hg	Al
Kola Peninsula													
Southeast of Monchegorsk	170	30	60	147	0.7	8.8	96	9.6					
Near Imandra slope of Khibini	11	27	59	221	0.2	11	26	1.8	0.6				
East part of Lapland nature reserve	261	40	39	170	0.1	6.6	94	4.5	0.7				
Background regions	1.8	17	84	47	0.05	1.4	0.7	0.2	0.3				
Faroe Islands (basalt, volcanic bedrock)	17	20		251	0.08	2.4	2.2		2.4	0.1	0.8	0.14	212

Sweden were generally < 0.2 mg/kg dw (Figure 4·17), with concentrations about three times higher in southern Sweden. Despite improvements, however, decreasing south-to-north Cd gradients were still visible in Fennoscandia in the 1995 and 2000 moss surveys (Figures 4·18 and 4·17). This is particularly true for Norway where the gradient was still relatively steep, owing to hotspots in southern Norway and to generally lower (< 0.1 mg/kg dw) concentrations over much of the north. However, latitudinal gradients even in Sweden are still significant in the 2000 survey (Figure 4·19). Exceptions to the general latitudinal trend occur around large point sources (e.g., smelters along the Swedish coast of Bothnian Bay; Figure 4·17).

The Kola Ecogeochemistry Project found Cd gradients in *Hylocomium splendens* to decline by two orders of magnitude (1.23 to 0.023 mg/kg dw) away from Monchegorsk, Nikel, and Zapoljarnij, presumably associated with short-range transport from smelter emissions (Reimann *et al.*, 1997b). Tsibulski *et al.* (2001) report similar findings for 1996 lichen samples (*Cladonia* spp. and *Cetraria* spp.) on the Kola Peninsula, with background concentrations of 0.05 to 0.1 mg/kg dw and concentrations near Monchegorsk of about 0.7 mg/kg dw (Table 4·5). The Barents Ecogeochemistry Project (Reimann *et al.*, 2001b) extends the work of the Kola Ecogeochemistry Project to eight new stations extending as far east as Vorkuta. At these sites there is a smaller range in Cd concentration in feather mosses, with maxima < 50% of those in the Kola Ecogeochemistry Project.

Hylocomium splendens has also been analyzed on the Taymir Peninsula, which extends north and east from one of the world's largest heavy metal smelting complexes at Norilsk, Russia (Allen-Gil *et al.*, 2003; Ford *et al.*, 1997). Collections were made in 1993 from 13 sites ranging from 80 to 300 km from Norilsk. Concentrations of Cd were 0.102 to 0.231 mg/kg dw, generally similar to the cleaner areas of the Nordic/European moss monitoring network at that time (Rühling and Steinnes, 1998) and lower than concentrations in Arctic Alaska (0.02 to 0.98 mg/kg dw; Ford *et al.*, 1995). These results support the findings of Blais *et al.* (1999) who suggest that impacts from the large Norilsk smelting complex on the tundra ecosystems of the Taymir Peninsula do not extend northward beyond about 100 km.

Riget *et al.* (2000b) found Cd concentrations in the moss *Racomitrium lanuginosum* from Greenland to be similar to those in *R. lanuginosum* from Arctic Alaska (Ford *et al.*, 1997). For lichens, Cd concentrations in Canadian *Cladina stellaris* (Chiarenzelli *et al.*, 1997, 2001) were similar to those in Nuuk (southwestern Greenland) (F. Riget, pers. comm., 2002) and from background regions on the Kola Peninsula.

Enrichment factors for the lichen *Cetraria cucullata*

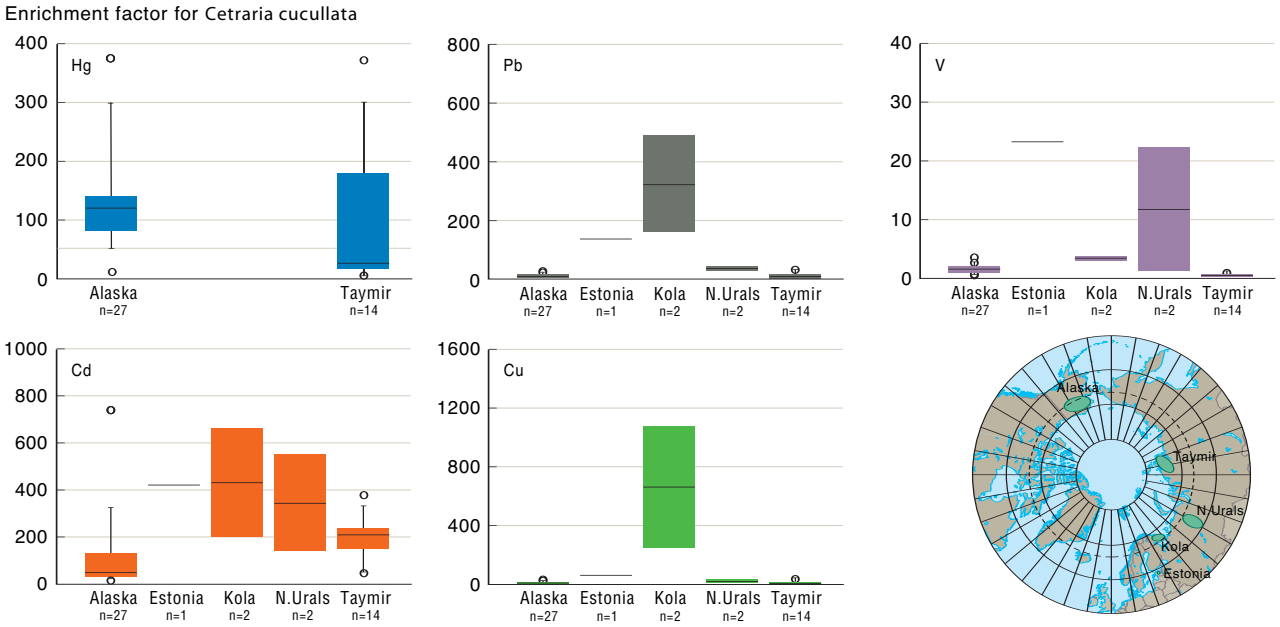


Figure 4·20. Enrichment factors for the lichen *Cetraria cucullata* from Arctic Alaska, Estonia, the Kola Peninsula, the northern Urals, and the Taymir Peninsula (Ford, unpubl. data, 2002). Sample sites in Alaska were in remote (unroaded) areas except for sites accessed on foot from roads in Prudhoe Bay and Barrow, and adjacent to the Dalton Haul Road. Sample sites in Estonia, the Kola Peninsula, and the northern Urals were in countryside, accessed on foot from roads or railroads and about 40 to 80 km from urban areas. Samples from the Taymir Peninsula were taken about 80 to 350 km upwind of Norilsk in remote (unroaded) areas.

relative to local lithology (see Section 4.4.3.2) have been calculated for Arctic Alaska, the Taymir Peninsula, and five sites in Estonia, the Kola Peninsula, and the northern Urals (Ford *et al.*, unpubl.). As expected, higher EFs occur in industrialized regions relative to Arctic Alaska and the Taymir Peninsula (Figure 4·20). The lower values for the Arctic Alaskan samples are of interest as they include samples from within the Prudhoe Bay oil fields (Ford *et al.*, 1997). Similar EF patterns were found in *Hylocomium splendens* where this species co-occurred (Ford *et al.*, unpubl.).

4.5.3.3. **Birds**

Concentrations of Cd in willow ptarmigan are thought to reflect local metal loads in vegetation (Wren *et al.*, 1994), which in turn reflect the underlying geochemical environment. Willows (*Salix* spp.) are known Cd accumulators, thus spatial patterns in tissues of ptarmigan using willow habitat require careful interpretation. Local variation in type, distribution, or use of food resources will influence the apparent distribution of Cd in animals, complicating an evaluation of spatial patterns due to atmospheric deposition.

Cadmium concentrations in liver and kidney of willow ptarmigan from Canada and Scandinavia were reported in the first AMAP assessment to be variable. Highest levels occurred in the Yukon and are probably related to the highly mineralized geochemical environment of that province. More recent reports support this interpretation. For example, Braune *et al.* (1999a) reported Cd concentrations in Yukon ptarmigan of 38.8 mg/kg dw in liver and 143 mg/kg dw in kidney, with some individuals having kidney concentrations of up to 1200 mg/kg dw. Limited data for Quebec (1991 to 1997) on liver and kidney of rock and willow ptarmigan are lower (Champoux *et al.*, 1999) and similar to two locations in West Greenland (7.6 to 20.8 mg/kg dw in liver and 62.8 to 86.4 mg/kg dw in kidney, Annex Table A5). Similarly, Pedersen and Fossøy (2000) report median Canadian levels of 121 mg/kg dw in kidney. This contrasts with Eurasian data. Two sites in eastern Siberia had ptarmigan liver levels similar to low Scandinavian values (7 to 9 mg/kg dw); median kidney levels were 50 mg/kg dw in Russia and 74 mg/kg dw in Norway (Pederson and Fossøy, 2000). More recent data from five regions of Russia show even lower means in ptarmigan liver (1.3 to 3.5 mg/kg dw) (Annex Table A5).

Cadmium concentrations have also been reported for a range of waterfowl in northern Quebec, primarily for muscle. Concentrations range from < 0.06 to 0.18 mg/kg ww. Limited data for osprey are at the low end of the range (Champoux *et al.*, 1999). Braune *et al.* (1999a) found no clear spatial patterns in Cd concentration in waterfowl from the eastern and western Canadian Arctic, perhaps owing to the influence of winter diet on body burdens in these migratory species.

4.5.3.4. **Mammals**

Cadmium concentrations in mammals are generally higher in kidney than liver. In Scandinavian and Russian reindeer, kidney Cd was generally five to ten times higher than in liver, which in turn was ten to 100 times

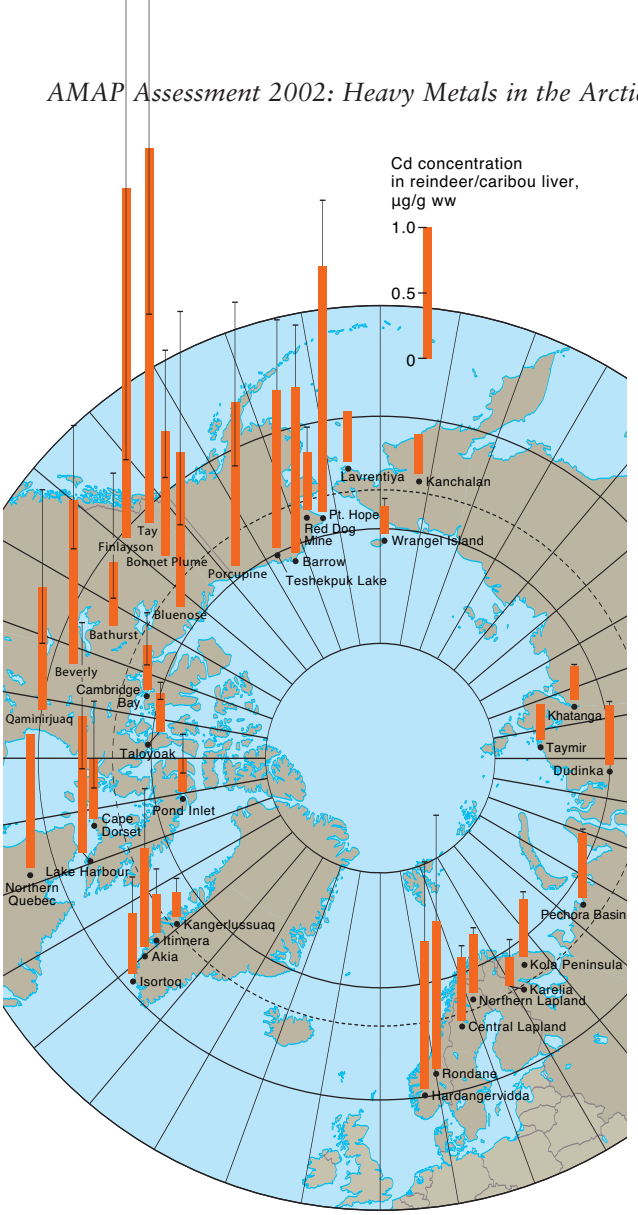


Figure 4·21. Mean cadmium concentrations in reindeer/caribou liver (Aastrup *et al.*, 2000; Elkin, 2001; Espelien *et al.*, 1999; Macdonald *et al.*, 2002; Melnikov *et al.*, 2002; Odsjö, 2003; O'Hara unpubl. data, 2002, F. Riget unpubl. data, 2002).

higher than in muscle (Elkin, 2001; Melnikov *et al.*, 2002). Similar patterns have been observed in moose in Sweden, where liver concentrations were significantly lower (mean 0.28 mg/kg ww) than in kidney (mean 0.91 mg/kg ww) (Odsjo *et al.*, 2001).

In Sweden, a large-scale pattern of decreasing metal levels from south to north has been found for some terrestrial mammal species, paralleling patterns for mosses and soils. However, for Cd in moose liver there was no significant difference between concentrations in southern and northern Sweden. Decreasing south-to-north gradients for Cd were reported for moose and other cervids in Norway (Frank and Petersen, 1984; Frøslie *et al.*, 1984; Selinus *et al.*, 1996 cited in Nyholm and Rühling, 2001). Other studies found elevated liver Cd concentrations in hare and moose in southern Norway (Kålås and Lierhagen, 1992). Collectively, and in the absence of south-to-north Cd gradients in underlying geology, these results suggest that body burdens in Scandinavian cervids are related to atmospheric deposition.

In Canada, moose liver and kidneys from the Yukon had higher Cd concentrations than moose from Manitoba, similar to the situation for willow ptarmigan. Con-

centrations in moose from some parts of Ontario and New Brunswick were also high (Braune *et al.*, 1999a), although parallel data for other species of flora and fauna were not available.

Good spatial coverage exists for Cd in reindeer/caribou liver (Figure 4·21). Geometric mean Cd concentrations in western Canadian herds ranged from 0.22 to 3.8 mg/kg ww (Macdonald *et al.*, 2002), compared to means of 0.94 to 1.18 mg/kg ww for northern Quebec herds (Robillard, *et al.*, 2002). Greenland herds had means of 0.12 to 0.70 mg/kg ww, with lowest concentrations in the Kangerllussuaq area and the highest concentrations in Akia (Aastrup *et al.*, 2000; Annex Table A6). Higher concentrations were found in caribou liver from Alaska, where mean liver Cd ranged from 0.4 (Red Dog Mine) to 1.9 mg/kg ww (Point Hope; O'Hara *et al.*, 2003). Swedish reindeer from northern and central Lapland had intermediate Cd concentrations in liver (0.4 mg/kg ww; Odsjö, 2001), similar to Finnish reindeer liver (range 0.2 to 0.7 mg/kg ww; Anon, 2001). Reindeer tissue from six sites in Arctic Russia showed mean liver Cd levels of 0.3 to 0.5 mg/kg ww, similar to those in Fennoscandia, with the highest levels in the Pechora Basin and Dudinka region of the Taymir Peninsula (Melnikov *et al.*, 2002). In an earlier study of samples from 1991 to 1993, concentrations of up to 1.7 mg/kg ww were found on the Chukotka Peninsula (eastern Siberia) and up to 0.7 mg/kg ww in Lovozero on the Kola Peninsula (Espelien *et al.*, 1999). Elevated levels also occurred at two sites in southern Norway; Hardangervidda and Rondane (means 1.1 mg/kg ww; Espelien *et al.*, 1999).

Similar to the situation for birds, the large-scale Cd pattern showed highest mean values and regional variations in Arctic Alaska and the Canadian Yukon and NWT provinces. In the southeastern Yukon, occasional extreme levels probably reflect naturally elevated Cd levels in soils and plants of the foraging areas (Braune *et al.*, 1999). For example, the geometric mean kidney concentration for 32 animals from the Yukon Finlayson herd was >30 mg/kg ww in 1992, with individual values of up to 70 mg/kg ww (Macdonald *et al.*, 2002). Concentrations in the Tay herd were comparable, but significantly lower in the more northern Porcupine and Bonnet Plume herds (geometric means 4.5 to 7.5 mg/kg ww). In the Northwest Territories and Nunavut, geometric mean kidney concentrations during the 1990s ranged from around 2 to 7 mg/kg ww (Macdonald *et al.*, 2002). The underlying geochemical environments particularly affect the Cd tissue concentrations of herbivorous mammals feeding on Cd accumulators such as willow.

Studies of liver and kidney in bank vole for 1981 showed ten-fold higher Cd concentrations in southern than northern Sweden (Nyholm and Rühling, 2001). By 1995, concentrations at the southern sample sites had decreased to 81% of 1981 levels, presumably in relation to decreased atmospheric deposition. Because comparable data for northern populations were not obtained in the more recent study, spatial patterns could not be assessed (Nyholm and Rühling, 2001).

Shrews eat soil-containing invertebrates such as earthworms, and are of special interest when organic soils have relatively high concentrations of contaminants, such as in southern Sweden. Cadmium concentrations in the liver and kidney of common shrew (*Sorex araneus*) in 1986 were two to five times higher in the south compared to northern Sweden (Lithner *et al.*, 1995). Studies at Pallas (northern Finland) in 1999 to 2000 found kidney concentrations of 0.3 to 0.7 mg/kg ww in immature shrews, and 0.5 to 1.5 mg/kg ww in adults (J.P. Hirvi and H. Henttonen, pers. comm., 2002).

New data on Cd in hare have been reported for Russia, the Faroe Islands, Greenland, and Canada (Annex Table A6), with most data available for liver. As expected, highest concentrations occur in kidney. For example, the Russian data set contains values for kidney, liver, and muscle, with concentrations generally decreasing by an order of magnitude at each step (ratios of about 1.0 : 0.13 : 0.01; Melnikov *et al.*, 2002). For liver, the concentration range is narrow (0.13 to 0.53 mg/kg ww) relative to that for Hg. No overall spatial patterns are evident, although within Russia the ranking among regions is similar to that for Pb, with highest values in the Pechora Basin and Dudinka region of the Taymir Peninsula and lowest values in the Khatanga region of the Taymir Peninsula and Chukotka.

4.5.4. Cadmium in the freshwater environment

4.5.4.1. River water and sediments

Spatial patterns in Cd concentration in river water and sediments unaffected by point sources are not apparent in the Arctic. Variations in heavy metal concentrations are generally related to local geology coupled with the organic content of the sediments. In the Kola Ecogeochemistry Project, however, Cd concentrations were found to be strongly affected by deposition from industrial sources (De Caritat *et al.*, 1996). Concentrations in the catchments furthest from industrialized centers were generally below detection limits (< 0.02 µg/L), while maximum concentrations were found in the vicinity of Monchegorsk. A summary of new data on Cd in river water and sediment is provided in Section 3.4.

Table 4·6. Water chemistry and trace metals in lakes in Arctic Canada, Tundra North West 99 expedition (Borg *et al.*, 2001).

Location	Coordinates	pH	Alk, meq/L	Ca, meq/L	Mg, meq/L	Cd, µg/L	Cu, µg/L	As, µg/L
Ivvavik National Park	69°26'N, 139°36'W	7.4	0.4	0.988	0.27	0.086	1.2	0.22
Cape Bathurst	70°46'N, 127°45'W	8.1	3.7	1.98	3.2	0.009	0.6	2.34
Banks Island N	73°40'N, 116°12'W	7.8	0.88	0.616	0.392	0.002	0.3	0.09
Melville Island S	75°03'N, 107°51'W	8.1	1.8	0.93	5.36	0.002	0.6	1.28
Ellef Ringnes Island	78°49'N, 103°40'W	7.0	0.12	0.206	0.177	0.01	0.9	0.08
Ellesmere Island	76°28'N, 86°50'W	8.0	1.3	1.11	0.322	0.004	0.1	0.07
Devon Island S	74°36'N, 82°24'W	8.1	3.1	1.89	1.97	0.002	0.2	0.23
Baffin Island S	68°26'N, 66°50'W	6.5	0.039	0.034	0.159	0.004	0.3	0.03
Baffin Island S	68°26'N, 66°50'W	6.4	0.021	0.034	0.178	0.006	0.6	0.02

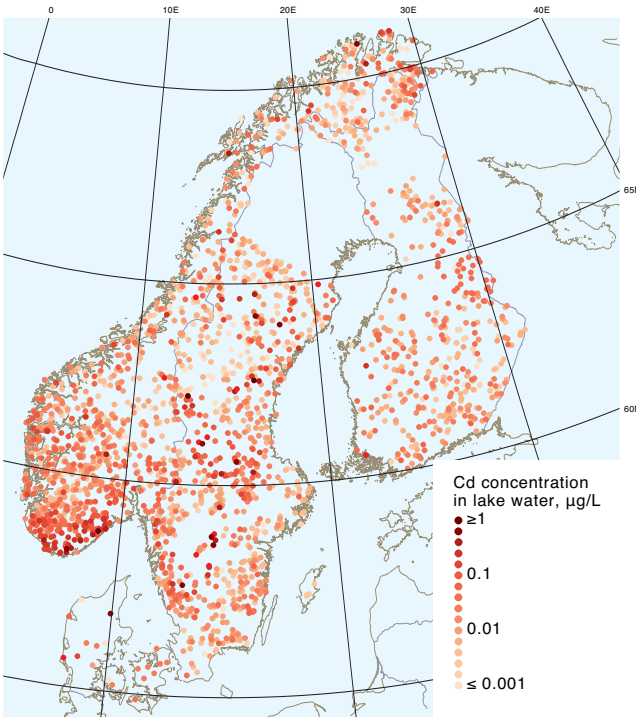


Figure 4·22. Cadmium concentrations in lake water in Scandinavia, 1995 (Skjelkvåle *et al.*, 2001).

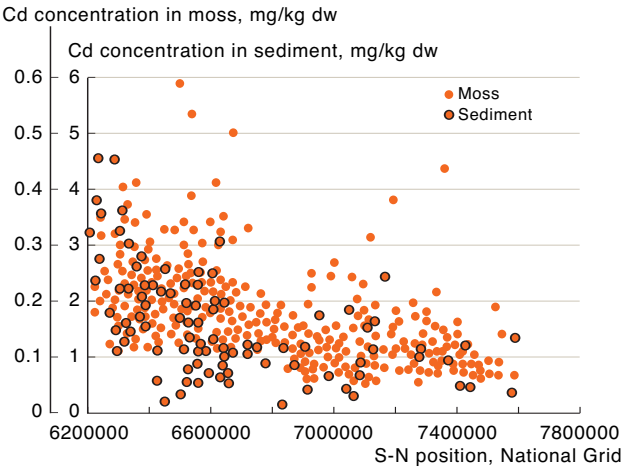


Figure 4.23. Latitudinal trends in cadmium concentration in moss samples in Sweden in 2000 (Å. Rühling, pers. comm., 2002) and lake surface sediments in reference lakes from the national monitoring program (Swedish national monitoring data base, IMA, SLU).

4.5.4.2. Lake water

National surveys of trace element water chemistry were performed in 1995 for nearly 3000 lakes in Scandinavia and on the Kola Peninsula (Figure 4·22). Aqueous metal concentrations in the northernmost region were characterized as low or very low in the majority of lakes, relative to Swedish guidelines for freshwater (Skjelkvåle *et al.*, 2001). However, large-scale patterns were evident with decreasing concentrations from south to north, consistent with similar gradients in moss and humus layers of forest soils. Mean Cd levels in lake water from southern Norway were around 0.1 µg/L compared to <0.01 µg/L in the north. Variations in local geology appear to influence lake Cd concentrations, but only on a local scale (Skjelkvåle *et al.*, 2001).

In Canada, lake water Cd concentrations were also low in the majority of lakes (Table 4·6). The influence of local geology was reflected in elevated Cd concentrations at Ivvavik in the Yukon Territory (Borg *et al.*, 2001).

4.5.4.3. Lake sediments

In the Swedish national lake sediment survey between 1998 and 2000 (SLU, 2003) highest Cd concentrations occurred in surface sediments of lakes in southern Sweden, similar to the pattern in the moss, soil, and lake water studies. The mean concentration in recent sediments was about 4 mg/kg dw in southern lakes and between 0.5 and 1.0 mg/kg dw in northern lakes. Core-top/core-bottom gradients also show a decreasing south-to-north Cd gradient (Johansson, 1989). Collectively, these data indicate a strong influence of anthropogenically-derived long-range atmospheric transport.

Figure 4·23 compares spatial patterns in Cd concentration in surface sediments from the lake survey with spatial patterns in moss from the moss survey in 2000. Despite differences in uptake processes and temporal resolution, latitudinal patterns in the two matrices are very similar. In both cases, mean concentrations in southern Sweden are about three times higher than in northern Sweden. The sediment data show some particularly high values in the southernmost parts of the country. This may be due to the strongly acidified soils in the catchments of this region, which can lead to elevated Cd export to lakes. Excluding these lakes improves the similarity of the two spatial patterns.

Similar patterns are evident in Norway. As expected from the moss and soil studies, concentrations of sedimentary Cd were highest in southeastern Norway, attributed to long-range atmospheric transport of Cd from anthropogenic sources (Rognerud and Fjeld, 2001). For the 1996 to 1997 Norwegian national lake survey, Rognerud and Fjeld (2001) report Cd to be one of the elements most enriched in surface sediments relative to sediments at depth, again with a pronounced south-to-north gradient in the overall level of enrichment.

A study of sediment cores from some 100 lakes in the border area between Norway and Russia found a concentric pattern of enrichment for several elements, including Cd, around the Murmansk smelters (Dauvalter, 1994; Dauvalter and Rognerud, 2002). Cadmium, like Hg, was negatively correlated with distance from the smelter ($p = 0.05$). A sediment core from the eastern Taymir Peninsula, much further east along the Siberian Arctic Coast, showed no recent Cd enrichment (Allen-Gil *et al.*, 2003).

4.5.4.4. Freshwater fish

In freshwater fish, Cd tissue concentrations tend to increase in species feeding at lower trophic levels (Amundsen *et al.*, 1997) because invertebrates typically contain higher Cd concentrations than most fish. This is the opposite of the situation for Hg.

Cadmium concentrations are not widely reported for freshwater fish, although existing data show concentrations in muscle to be around < 0.005 mg/kg ww. Muir *et al.* (1999a, 1999b, 2000) found Cd in muscle for various

species of char from lakes in northern Labrador and Nunavik to approach very low detection limits (< 0.001 to <0.006 mg/kg ww). Concentrations for char from Chukotka are higher (0.1 to 0.15 mg/kg ww; Annex Table A10).

Cadmium concentrations are higher in char liver, ranging from < 0.5 mg/kg dw (equivalent to < 0.12 mg/kg ww) in Iceland, to 2.0 to 2.5 mg/kg dw in Chukotka (Annex Table A10). Most of the char collected in Arctic Canada during the Tundra North West Expedition had liver concentrations of <1 mg/kg dw, except for char from one lake which had a mean concentration of 8.0 mg/kg dw. This elevated value was approximately ten times higher than concentrations in the other lakes in the study as well as in northern Sweden (Borg *et al.*, 2001). Elevated Cd levels were also found in lake sediments and water at this site, and high Hg concentrations were found in sediments but not fish (Borg *et al.*, 2001).

4.5.5. Cadmium in the marine environment

4.5.5.1. Marine sediments

Concentrations of Cd are fairly similar in different areas, ranging from 0.06 to 0.11 mg/kg dw in the Laptev Sea and 0.18 to 0.27 mg/kg dw in the Beaufort Sea (Table 4·1). Macdonald *et al.* (2000) note that although Cd concentrations in marine sediments can be up to four to five times higher than the Canadian marine sediment quality guideline of 0.7 mg/kg dw, the higher concentrations are generally found beneath a Mn-enriched surface layer, suggesting that Cd distribution in sediments, like Mn, is generally controlled by redox factors. Therefore, marine sediments may be inappropriate for examining spatial patterns in Cd distribution.

4.5.5.2. Marine invertebrates

A study of Cd concentrations in ten species of bivalve, four species of amphipod, three species of polychaete, and one species of siphunculoid and nemertine from the Pechora Sea (Savinov *et al.*, 1998) revealed highest Cd levels in epifaunal suspension feeders such as Iceland scallop (*Chlamys islandica*; 2.92 to 4.30 mg/kg ww) and *Musculus niger* (2.18 mg/kg ww). Cadmium levels in invertebrates from the Pechora Sea were consistent with those in the literature for the same species from other Arctic regions (Savinov *et al.*, 1998). Scallops from Labrador and Nunavik (Canada) and queen scallop from the Faroe Islands had Cd concentrations at the lower end of the range previously reported (AMAP, 1998) for Arctic bivalves (0.95 mg/kg ww in Canadian scallop muscle (Muir *et al.*, 2001a) and 0.42 mg/kg ww in soft tissue of Faroese queen scallop (Larsen and Dam, 1999)).

Cadmium concentrations in Alaskan blue mussel were higher than in blue mussel from Labrador (Canada), Norway, and the Pechora Sea (Figure 4·24). The relatively low Cd concentrations in Canada are consistent with concentrations previously reported by Doidge *et al.* (1993) for Hudson Bay, Hudson Strait, and Ungava Bay, and by Muir *et al.* (2000) for Labrador and Nunavik. Mussel concentrations in Qeqertarsuaq, Greenland were high compared to other Arctic locations (1.1 to 2.3 mg/kg ww), probably due to local geological conditions. High Cd levels in this area relative to other Greenland locations have been reported previously for both blue mussel and ringed seal (Riget *et al.*, 2000e). Concentrations in blue mussel from the Faroe Islands and southeast Iceland were in the lower range of concentrations observed in the Arctic (Annex Table A13). Caution in the interpretation of these results is warranted, as Cd concentrations (like those for Hg) are dependent on mussel size (Riget *et al.*, 1996, 2000e).

4.5.5.3. Marine fish

A summary of new data on marine fish is provided in Section 4.4.5.3. Additional Cd data are also available for liver and muscle of Arctic cod and navaga (*Eleginus navaga*) (Savinov *et al.*, 1998; Tsibulski *et al.*, 2001).

Flathead sole were collected from seven sites along the southern coast of Alaska (Boca de Quadra, Dutch Harbor, Kamishak Bay, Lutak Inlet, Port Moller, Port Valdez, and Skagway). Liver showed high spatial variability in Cd concentration (0.03 to 2.22 mg/kg dw) although concentrations were within the range reported in the first AMAP assessment for other fish species (Meador *et al.*, 1994). Highest concentrations were found in Lutak Inlet, originally selected as a reference site as it had no known point sources. Liver from flathead sole collected at Dutch Harbor and Port Valdez, originally selected as urban sites, had lower concentrations (0.03 and 2.01 mg/kg dw, respectively).

Cadmium concentrations in muscle of sea-run Arctic char from Labrador collected in 1999 were low (0.001 to 0.002 mg/kg ww; Annex Table A14), similar to limited older data from West Greenland. Cadmium levels in shorthorn sculpin liver from Greenland were relatively high, especially in central West Greenland. Concentrations were 0.66 to 1.70 mg/kg ww, generally higher than in sculpins from the Faroe Islands (Annex Table A14). Generally speaking, Cd levels in biota (fish, mussels, and ringed seal) from central West Greenland are higher than in neighboring areas, probably due to local geological conditions.

Cadmium levels in liver of Atlantic cod from northwest Iceland were higher than in cod from the Faroe Islands and Arctic Norway (Figure 4·25). Similarly, Icelandic dab had higher liver Cd concentrations than Norwegian dab (0.14 to 0.88 versus 0.18 to 0.23 mg/kg ww, see Annex Table A14 and data reported by the Environmental and Food Agency of Iceland at www.hollver.is). Cadmium concentrations in Arctic cod from the Pechora Sea were 0.006 mg/kg ww in muscle and 0.49 mg/kg ww in liver, higher than in cod from the Barents Sea but within the range previously reported for the eastern Canadian Arctic, Jan Mayen, and West and East Greenland (AMAP, 1998).

4.5.5.4. Seabirds

New data are available on Cd concentrations in several seabird species since the first AMAP assessment: Arctic tern, common guillemot (also known as common murre), northern pintail, long-tailed jaeger, parasitic jaeger, Arctic loon, spectacled eider, and Steller's eider. In total, Cd data are available for 25 seabird species. The spatial distribution of new seabird data sets is given in Table 4·3.

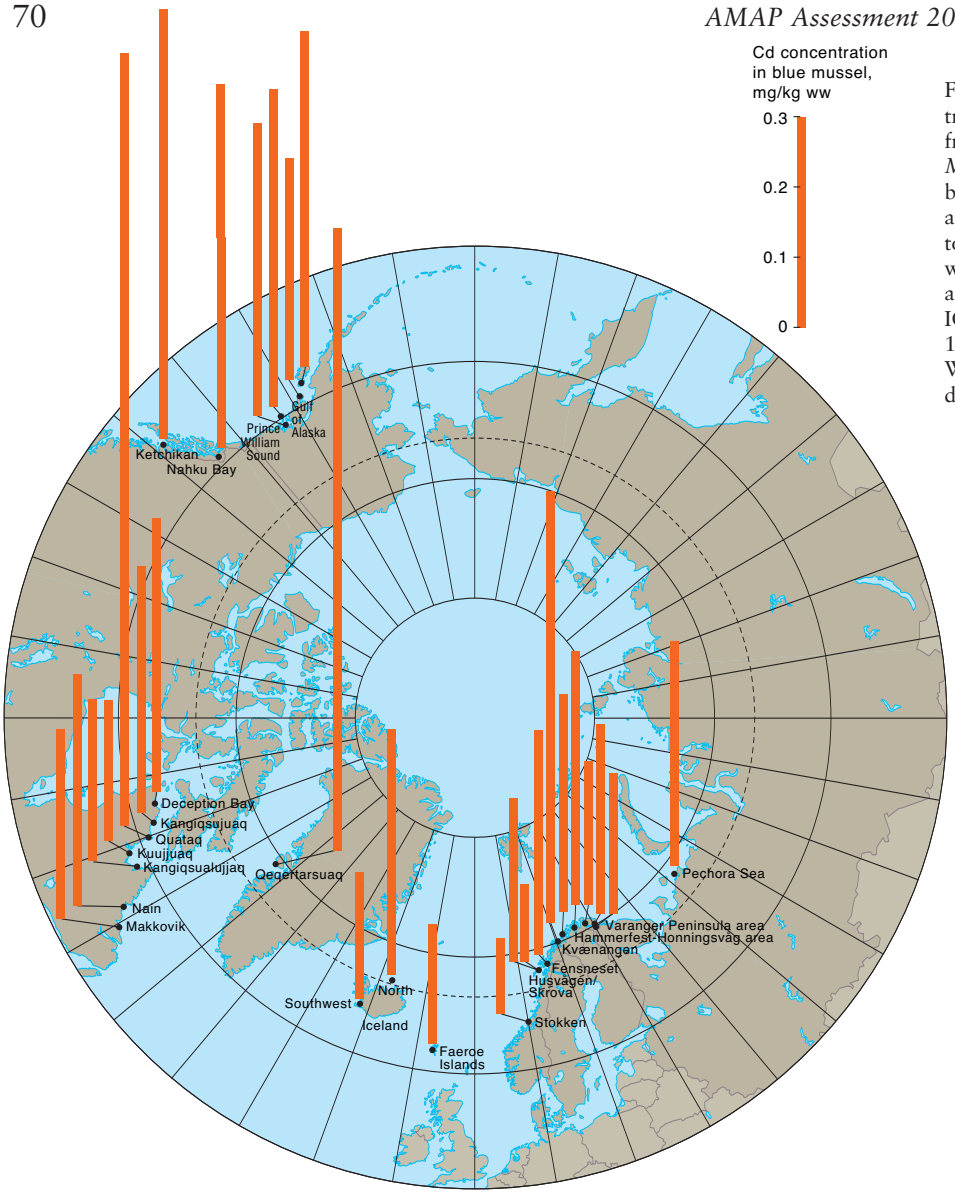


Figure 4·24. Mean cadmium concentrations in blue mussel (*Mytilus edulis*) from different Arctic countries, and in *Mytilus* spp. from Alaska, collected between 1995 and 2000. The Alaskan and Norwegian data were converted to wet weight using a percentage dry weight of 15%. The Norwegian data are median values (Green *et al.*, 2001; ICES databank; Larsen and Dam, 1999; Muir 2000; NOAA's Mussel Watch Programme; Riget, F. unpubl. data 2002; Yngvadóttir *et al.*, 2002).

Cadmium concentrations in seabirds from the Barents Sea were generally lower than those for Greenland, Canada, and northeastern Siberia (Savinov *et al.*, 2003). Highest Cd concentrations in the Barents Sea were found in fulmar, kittiwake, Arctic tern, and common eider, and lowest in common guillemot. Spatial differences in liver Cd levels within the Barents Sea were found for several species, with highest concentrations found in birds from Ny-Ålesund, Svalbard (Savinov *et al.*, 2003).

Of four eider species from Alaska and Arctic Russia the highest liver Cd levels were found in spectacled eider (18.2 to 37.0 mg/kg dw; Stout *et al.*, 2002). Trust *et al.* (2000) also reported elevated liver Cd in spectacled eider (33.8 mg/kg dw) compared to other marine birds. Cadmium concentrations in spectacled eider liver were higher than in common eider from Norway but comparable to eiders from Greenland (Stout *et al.*, 2002). Cadmium concentrations in common eider from Nunavut, Canada, were 71 to 165 mg/kg dw in kidney, similar to or slightly higher than those for Europe, Greenland, and Svalbard (Wayland *et al.*, 2001).

Cadmium levels in liver of long-tailed duck (also known as oldsquaw) and herring gull from Chaun, northeastern Siberia were 78 to 88 and 159 mg/kg dw, respectively, higher than for Greenland and eastern

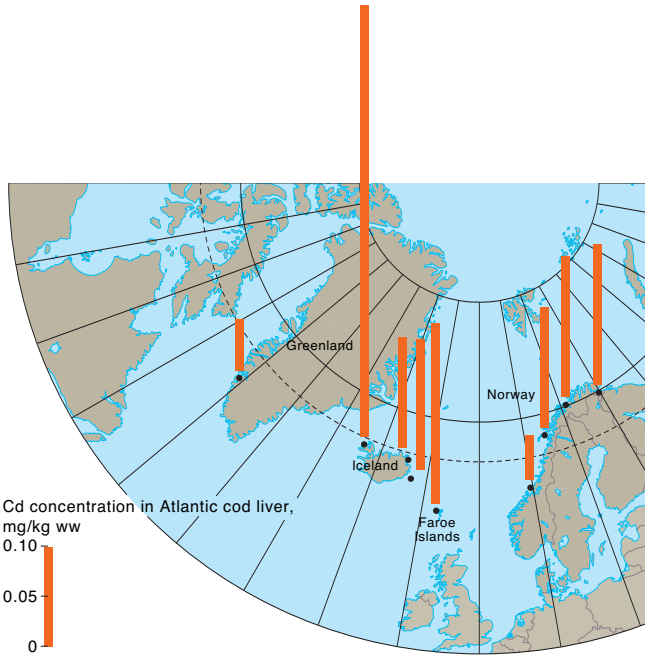


Figure 4·25. Cadmium concentrations in Atlantic cod liver for samples collected between 1995 and 1999. The data for Greenland, the Faroe Islands and Iceland are mean values, the data for Norway are median values.

Canada (Kim *et al.*, 1996). Generally, breeding birds in northeastern Siberia overwinter in eastern and southeastern Asia, while breeding birds in Greenland and eastern Canada overwinter in North America (Kim *et al.*, 1996). The relative influence of exposure on the breeding versus overwintering grounds is not known.

Cadmium levels in liver of northern fulmar and black guillemot from the Faroe Islands (Larsen and Dam, 1999) were 8.55 mg/kg ww and 1.27 to 1.36 mg/kg ww, respectively, similar to concentrations previously reported for Greenland, Canada, and the Barents Sea (AMAP, 1998). Cadmium concentrations in liver and kidney from adult black guillemot from Greenland in 1999 were 1.85 and 15.0 mg/kg ww (F. Riget, unpubl. data, 2002), similar to those previously reported for this area (Nielsen and Dietz, 1989).

Spatial differences in Cd concentration in species from higher trophic levels, including seabirds, have been explained by differences in available food items (e.g., Kim *et al.*, 1996; Savinov *et al.*, 2003). Cadmium levels are known to be relatively high (and Hg levels relatively low) in Arctic marine copepods and amphipods, with the reverse true for marine fish (Dietz *et al.*, 1996). Because Cd bioaccumulates, food web differences may lead to apparent spatial differences if populations are feeding differently in different regions.

4.5.5.5. **Marine mammals**

Data on Cd concentrations in marine mammal tissues are available for eight species of seal, seven species of whale, and polar bear. Data on grey seal are newly available. In general, Cd bioaccumulates strongly in marine mammals and concentrations increase with size and age, although Caurant *et al.* (1994) found this trend to level off in liver and kidney of pilot whale after about age ten. These factors must be considered when evaluating spatial patterns.

Liver tissue was analyzed for ringed seal collected in the late 1990s from Barrow (Alaska), Labrador and the eastern Canadian Arctic, Greenland, Svalbard (Norway) and Chukotka (Russia) (Fant *et al.*, 2001; Muir *et al.*, 2000; D. Muir, pers. comm., 2002; F. Riget, unpubl. data, 2002; Woshner *et al.*, 2001a) (Figure 4·26). As with Hg, concentrations were standardized to five-year old seals wherever possible assuming the same log[concentration]:age relationship for all areas. Concentrations in ringed seal from Barrow and Chukotka were lower than in those from northeastern Canada, although the Russian seals were younger (one- to three-year olds). The lowest Cd levels were found in ringed seal from Labrador and Svalbard, with higher concentrations in ringed seal from Salluit, Hudson Strait (Muir *et al.*, 2000). Concentrations were particularly high in Greenland ringed seal, especially those from Qeqertarsuaq. The relatively high levels in seal, blue mussel, and shorthorn sculpin from Qeqertarsuaq are consistent with earlier (1994 to 1995) data and may reflect local geological conditions (Riget *et al.*, 2000e).

Data are now available on Cd levels in 45 grey seal livers collected in the Faroe Islands between 1993 and 1995 (Larsen and Dam, 1999). The mean Cd concentration in liver from 20 adult females was 14.6 mg/kg ww, at the higher end of the range observed in other seal species. However, the mean concentration in liver from four adult males was much lower (1.85 mg/kg ww).

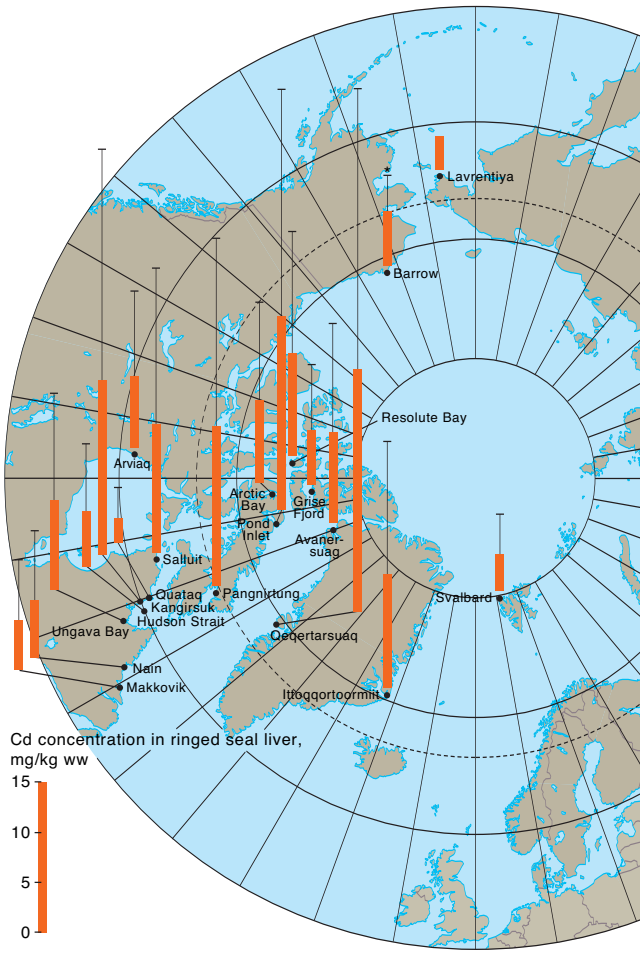


Figure 4·26. Mean cadmium concentrations in ringed seal liver. Data for Greenland and Canada were collected during 1999 to 2000, and for Alaska during 1995 to 1997. Data are mean-adjusted to five-year old animals unless marked by an asterisk indicating unadjusted data (recalculated from Muir *et al.*, 1999b; Muir, D. pers. comm., 2002; Riget, F. unpubl. data, 2002; Woshner *et al.*, 2001b).

Data for bearded seal are scarce, but concentrations in liver for two animals from Chukotka averaged 2.24 mg/kg ww (Melnikov *et al.*, 2002), in the same range as for other seal species (AMAP, 1998). Data on spotted seal are even scarcer. Concentrations in spotted seal liver for Chukotka averaged 3.01 mg/kg ww (Melnikov *et al.*, 2002), higher than for bearded seal but lower than for ringed seal (cf. Figure 4·26).

Cadmium concentrations in walrus from Nunavik, Labrador collected in 1999 were 3.32 mg/kg ww in liver and 16.5 mg/kg ww in kidney (Muir *et al.*, 2000), comparable to concentrations previously reported by Wagemann *et al.* (1996) for the same population.

Grey whale was not included in the first AMAP assessment, but Cd concentrations in one individual from Chukotka (Melnikov *et al.*, 2002) were similar to those for other baleen whale species (AMAP, 1998).

In the first AMAP assessment few data were available on heavy metals in bowhead whale. More data are now available for the Alaskan Arctic. Krone *et al.* (1999) report concentrations in liver for 20 individuals taken in subsistence hunts near Barrow during 1992 to 1994. In addition, Woshner *et al.* (2001b) report data

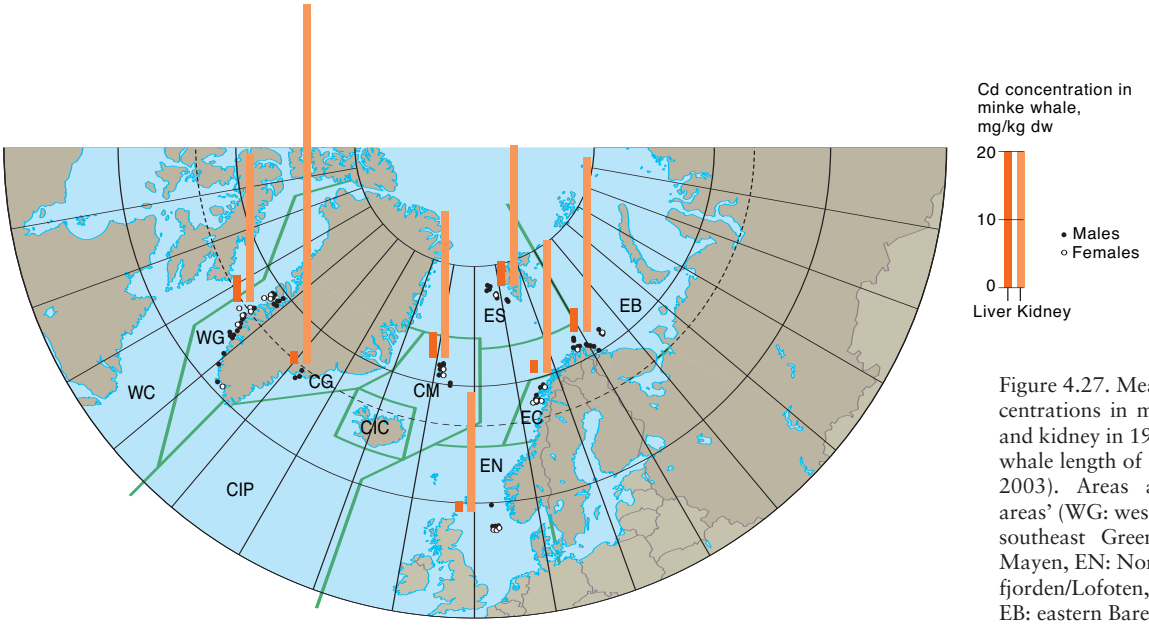


Figure 4.27. Mean cadmium concentrations in minke whale liver and kidney in 1998 (adjusted to a whale length of 7 m) (Born *et al.*, 2003). Areas are IWC 'small areas' (WG: west Greenland, CG: southeast Greenland, CM: Jan Mayen, EN: North Sea, EC: Vestfjorden/Lofoten, ES: Svalbard, EB: eastern Barents Sea).

for 20 bowhead whales harvested between 1995 and 1997 and compare their results with data for 41 whales harvested between 1983 and 1990 (Bratton *et al.*, 1997). In general, the mean Cd levels in liver (9.63 mg/kg ww) and kidney (20.0 mg/kg ww) were within ranges previously reported for Arctic marine mammals (Woshner *et al.*, 2001b). Becker (2000) includes bowhead whale in a list of Alaskan Arctic marine mammals reported to have elevated Cd concentrations. High levels of hepatic Cd are probably due to extensive feeding on krill.

Data on metal concentrations in minke whale were available for the first AMAP assessment for the West Greenland area only. In 1998, a study was undertaken to clarify the population sub-structure of North Atlantic minke whales using PCB signatures (Hobbs *et al.*, 2003), genetic analyses (Andersen *et al.*, 2003), fatty acid profiles (Møller *et al.*, 2003), and element and stable isotope signatures (Born *et al.*, 2003). In total, 159 samples of liver, kidney, muscle, and baleen tissue were collected from the North Atlantic area, including West and East Greenland, Jan Mayen, Svalbard, the Norwegian coast, the North Sea, and the Barents Sea (Born *et al.*, 2003). Cadmium levels in muscle, liver, and kidney were similar to those reported previously by Dietz *et al.* (1997) and Hansen *et al.* (1990). Concentrations in liver and kidney of whales caught in the North Sea and along the Norwegian coast were generally lower than in whales from the Arctic (Figure 4·27). This supports the outcome of stable isotope analyses which indicate that populations of North Sea whales are feeding at higher trophic levels than West Greenland whales, and is consistent with the Cd concentration differences in fish relative to crustaceans. The relatively high Cd levels in minke whale kidney from East Greenland are based on two individuals and cannot be considered representative for that population.

Additional data on Cd levels in liver, kidney, muscle, blubber, and epidermis of beluga whale from Arctic Alaska were collected by Woshner *et al.* (2001b). Mean concentrations were 3.8 mg/kg ww in liver and 12.2 mg/kg ww in kidney, and were consistent with the trend of increasing concentrations from west to east in the Canadian High Arctic previously reported by Wagemann *et al.* (1996).

Cadmium concentrations in muscle, liver, and kidney for 53 narwhal from central West Greenland in 1993 were not significantly different from those for 88 narwhal from northwestern Greenland collected in 1984 and 1985 (Riget *et al.*, 2002). Furthermore, significantly higher Cd levels were found in females than males for all three tissues. In the West Greenland area, concentrations are low for liver (geometric means of 2.4 to 19.5 mg/kg ww) and kidney (geometric means of 33 to 76 mg/kg ww) relative to previously reported data from the eastern Canadian Arctic (AMAP, 1998).

Cadmium concentrations in muscle of adult pilot whale from the Faroe Islands collected in 1997 and 1999 were 0.11 to 0.26 mg/kg ww (Larsen and Dam, 1999; Olsen *et al.*, 2003), about the same level as reported in the late 1970s by Juelshamn *et al.* (1987) but higher than more recent findings for Newfoundland, Canada (Muir *et al.*, 1988). Mean Cd concentrations in kidney for two schools sampled in the Faroe Islands in 1999 and 2000 were 125 and 122 mg/kg ww, which are higher than concentrations reported for three schools in 1986 to 1987 (86, 93, and 55 mg/kg ww, Caurant *et al.*, 1994) and two schools reported in 1978 (73 and 6 mg/kg ww, Juelshamn *et al.*, 1987). These concentrations are high compared to those for other marine mammals. Studies to date have not identified toxicity problems (Caurant and Amiard-Triquet, 1995), although because Se-dependent detoxification processes may be limited in lactating long-finned pilot whale (Caurant *et al.*, 1996) studies of possible kidney histopathology are ongoing.

Woshner *et al.* (2001a) reported Cd concentrations in the liver, kidney, muscle, and blubber of polar bear harvested between 1995 and 1997 near Barrow. Tissue levels were 0.47 and 8.69 mg/kg ww in liver and kidney, respectively, generally within previously reported ranges. However, concentrations were not adjusted for size or age, which limits the interpretation of the data. Data for 18 to 20 adult male bears taken during 1993 to 1999 from northern and western Arctic Alaska were about 1.05 mg/kg ww in liver and about 6.3 mg/kg ww in kid-

ney, similar to levels in western Canada (Annex Table A16). Cadmium concentrations in polar bear from East Greenland collected in 1999 to 2000 had geometric mean concentrations of 0.65 to 1.54 and 17.6 to 32 mg/kg ww for liver and kidney, respectively, confirming the relatively high levels previously reported for this area (AMAP, 1998; and Annex Table A16).

As with Hg, several causes or combinations of causes have been proposed to explain spatial differences in observed Cd levels, including sedimentary geology, trophic position, food source, and local inputs.

4.6. Lead

4.6.1. Atmospheric lead

Weekly mean concentrations of 18 elements in aerosols were collected on filters at Alert (Ellesmere Island) between 1980 and 1995 (Sirois and Barrie, 1999). Seasonal variation in Pb concentration in the Arctic aerosol peaked in winter and was at a minimum in summer. This is typical of metals from anthropogenic sources. Associations between the various metals yielded further information on trends and sources. Applying the positive matrix factorization approach showed that Pb, V, Mn, and Zn were mainly linked with an anthropogenic aerosol factor that peaked in January to March. This factor, which is interpreted as anthropogenic combustion aerosols originating from Eurasia, was less pronounced in 1995 than in 1980. The analyses confirm that the major source region influencing concentrations at Alert is primarily Eurasia, although further studies are needed to determine how changes in Arctic Oscillation state may influence transport patterns.

The stable Pb isotopic composition of weekly aerosol samples at Alert was characterized by Mercier *et al.* (1999) using high-precision analytical techniques. Their studies showed seasonal trends in Pb isotope composition that reflect air mass flows from different source regions, confirming earlier reports (e.g., Sturges and Barrie, 1989). Autumn aerosols contained the lowest Pb concentrations and were relatively radiogenic (i.e., high ^{204}Pb-based ratios). Back-trajectory calculations show these to originate from natural sources in the Canadian Arctic Archipelago and coastal West Greenland. Relatively non-radiogenic Pb isotopic composition during late autumn to winter reflects the dominance of western European emissions, while an industrial Russian and eastern European Pb contribution is evident during late-spring and early-summer. Higher elevation regions adjacent to northern Canada (e.g., the Greenland Plateau) have a different aerosol mix and may be more influenced by emissions from North America (Sherrell *et al.*, 2000; Sirois and Barrie, 1999).

Several stations have reported many years of data to NILU. Currently, concentrations of Pb in aerosols are at or below 1 ng/m^3 for data reported during the late 1990s. Higher values (generally 1 to 10 ng/m^3) occurred during the period 1985 to 1993 at several Alaskan sites (notably Bering Land Bridge and Northwest Areas National Parks on the Chukchi Sea, as well as Wrangell-St.Elias National Park on the Canadian border) but the stations ceased reporting data in 1994. Temporal trends in Pb in aerosols are discussed in Section 5.1.

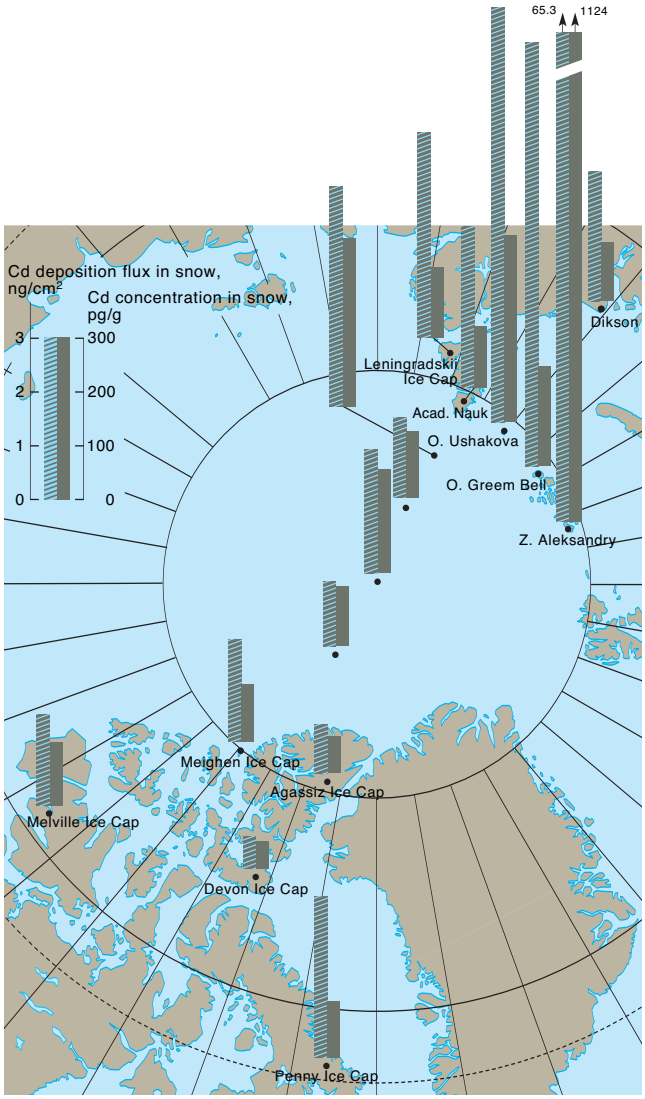


Figure 4·28. Lead concentrations (weighted averages for a snow accumulation period of about nine months; mean, see Annex Table A2) and deposition calculated over the period of snow accumulation, i.e., the nine months from the end of the previous summer to late spring of the sampling year) in surface snow across the Russian and Canadian Arctic Islands and the Arctic Ocean. Where samples were collected repeatedly over several years (e.g., the Agassiz ice cap) concentrations were averaged between individual years. (Koerner *et al.*, pers. comm., 2002).

4.6.2. Lead in precipitation

The spatially extensive snow sampling program spanning the Canadian and Russian Arctic islands and the Arctic Ocean (Section 4.4.2) found spatial patterns in Pb concentration in surface snow (Figure 4·28) similar to those for Cd (Figure 4·16) (Koerner *et al.*, 2002; Annex Table A2). Concentrations increase from the easternmost sites (Academii Nauk, Leningradskii) to the western islands (Ushakova, Greem Bell, Aleksandry). The relationship between surface and subsurface snow concentrations probably reflects a seasonal dynamic in which the Severnaya Zemlya sites have local dust inputs or very small contributions from Norilsk during the summer and/or autumn, while concentrations at the western islands are generally near background. In the late winter and spring, however, Severnaya Zemlya appears to receive a smaller contribution from polluted southern air masses than the western islands. Because Pb deposition (Figure 4·28) depends both on concentration in precipi-

tation and the total quantity of precipitation, the gradient in increasing Pb deposition from Severnaya Zemylya to Franz Josef Islands is consistent with increasing snow accumulation. Koerner *et al.* (2002) point out that while the spatial pattern in the snow concentration data does not indicate Norilsk as a primary source for the Russian sites, it does not rule out Norilsk as a source for the Arctic Ocean and Canadian snow collection sites.

Comparison of these snow data with the Pb transport and deposition models discussed in Section 3.2.2.3. suggests a closer agreement with the MSC-E model than either the DNMI or the DEHM models (e.g., Figures 3·18, 3·16, and 3·15), although the years covered by the models (1989 and 1990) are different to those covered by the snow collections (1993 to 1996). Model validation to date is based on air concentrations; it would be useful to attempt validation based on other compartments with longer residence times and slower fluxes out of the Arctic systems.

In the study by Garbarino *et al.* (2002) of element concentrations in snow samples from the Chukchi and Beaufort Seas, Pb behaved differently than Hg and Cd. Lead concentrations were generally low (0.0032 to 0.08 µg/L) relative to other snow studies (particularly for Greenland snow) and Pb was not enriched relative to lithology (Mn-normalized EFs of 0.06 to 2.1).

The stations reporting Pb precipitation data to the AMAP data center differ from those reporting aerosol data on Pb, and so spatial comparisons are based on a different configuration of sites. This is also the case for Cd. It would be useful to establish relationships between aerosol and precipitation data, although in practice this may be unrealistic as many factors are involved. In the AMAP data set, annual mean concentrations of Pb in precipitation are currently < 1 µg/L at most AMAP stations, with occasional excursions as high as 2 µg/L at Svanvik in the late 1990s. The latter may reflect emissions from the Kola Peninsula. However, Äyräs *et al.* (1997) believe the Pb signal at Svanvik to be more related to vehicular traffic than to the Kola smelter emissions, as Pb emissions from the Kola smelters are typically small and the border area has many roads. Precipitation data have not been reported for North American stations.

Lead concentrations estimated during the Kola Ecogeochemistry Project at the site closest to Svanvik (see Section 4.4.2) had a median Pb concentration of 2.42 µg/L in summer 1994 (range: 0.53 to 6.8 µg/L), which is similar to the annual mean concentrations at Svanvik reported to AMAP for 1993 (~ 3 µg/L) and 1994 (~1.5 µg/L). The samples from the Kola Ecogeochemistry Project were filtered, the Svanvik samples were not (Reimann *et al.*, 1997c). In contrast, the median concentration at Monchegorsk was 6.3 µg/L (range: 2.1 to 40.5 µg/L). Chekushin *et al.* (1998) estimate an annual Pb deposition of 0.562 to 0.599 kg/km^2/yr at the two Finnish catchments of the Kola Ecogeochemistry Project considered background for the region.

4.6.3. Lead in the terrestrial environment

4.6.3.1. Soils and peats

The three Norwegian surveys of heavy metals in surface humus (1977, 1985, 1995: see Section 4.4.3.1) showed distinct decreasing south-to-north trends in Pb concentration, with highest concentrations in the south attributed to long-range atmospheric transport (Johansson *et al.*, 2001). Analyses of surface peat from a large number of ombrotrophic bogs throughout Norway gave similar results (Steinnes, 1997).

Background concentrations of Pb in the humus layer of forest soils were estimated at 8 mg/kg dw based on Fennoscandian samples from sites north of the Arctic Circle (Andersson *et al.*, 1991). In the two Finnish catchments of the Kola Ecogeochemistry Project considered background for the region, Pb concentrations in O-horizon soils had median concentrations of 9 to 13 mg/kg dw and a range of 4 to 52 mg/kg dw (Reimann *et al.*, 1997a, 1998).

4.6.3.2. Mosses and lichens

A generally decreasing gradient across relatively homogenous ecosystems from south to north in Scandinavia has been demonstrated for Pb as well as for many other elements, with exceptions around large point sources (e.g., smelters along Bothnian Bay; Rühling *et al.*, 1996). In 1995, decreasing south-to-north gradients were slight in most Scandinavian countries (Figure 4·29; Rühling and Steinnes, 1998), probably due to decreasing inputs in the south from long-range atmospheric transport. Despite continuing improvements in moss concentrations (e.g., Rühling and Tyler, 2001), the moss surveys in 2000 show slight but still significant ($p < 0.01$) south-to-north decreasing gradients for Pb in Sweden (Figures 4·30 and 4·31). Lead levels in southernmost Norway also remain relatively high.

Current Pb concentrations in northern Scandinavia are relatively uniform, although slightly lower in northern Norway than northern Sweden. Concentrations in southern Scandinavia are about 2.5 to 4 times higher. Recent data for Finland show moss Pb concentrations to be slightly higher in the northeast, although in the Norwegian moss survey of 2000, Pb concentrations do not appear to be elevated in the Varanger area of northern Norway (Steinnes, 2001). The relative influence of vehicular emissions versus industrial emission sources on the Kola Peninsula is unclear although the Kola Ecogeochemistry Project established that smelter emissions on the Kola Peninsula around Monchegorsk, Nikel, and Zapoljarnij cause only weak Pb gradients in moss (Äyräs *et al.*, 1997; Reimann *et al.*, 1997b) and O-horizon soils (Reimann *et al.*, 1997b).

On the Taymir Peninsula, concentrations of Pb in *Hylocomium splendens* collected in 1993 ranged from 1.41 to 3.27 mg/kg dw (Allen-Gil *et al.*, 2003; Ford *et al.*, 1997), similar to or slightly less than concentrations in the cleaner areas of the Nordic/European moss monitoring network at that time. However, these concentrations were generally higher than the very low concentrations measured in *H. splendens* from Arctic Alaska in 1990 to 1993 (range 0.35 to 2.33 mg/kg dw, median 0.62 mg/kg dw; Ford *et al.*, 1995).

Using a different moss species (*Racomitrium lanuginosum*) Riget *et al.* (2000b) found elevated Pb concentrations in Greenland relative to Arctic Alaskan moss.

Enrichment factors suggest that moss Pb derives primarily from soils in both Arctic Alaska and the Taymir

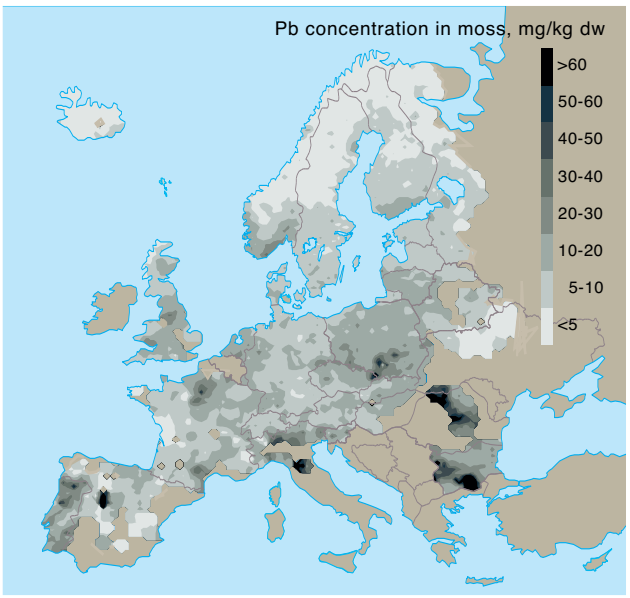


Figure 4·29. Lead concentrations in European feather mosses, 1995 (Rühling and Steinnes, 1998).

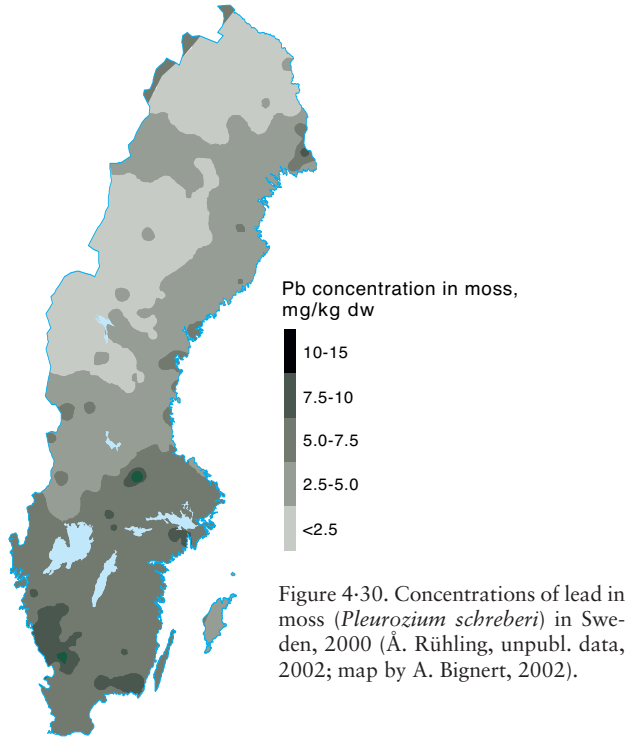


Figure 4·30. Concentrations of lead in moss (*Pleurozium schreberi*) in Sweden, 2000 (Å. Rühling, unpubl. data, 2002; map by A. Bignert, 2002).

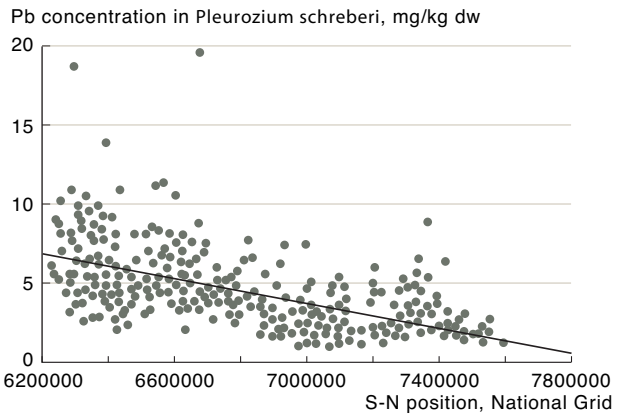


Figure 4·31. Relationship between lead concentration in *Pleurozium schreberi* and latitude in Sweden, 2000 (significance of the correlation: $p < 0.0001$; slope of the regression line: $p < 0.0001$) (Å. Rühling, unpubl. data, 2002).

Peninsula (EFs generally <10; Ford *et al.*, unpubl.). Similar results are found for the lichen *Cetraria cucullata*. As expected, higher Pb EFs are found for *C. cucullata* in industrialized regions (Figure 4·20). The potential impact of local point sources is reflected by much higher Pb concentrations and Pb EFs (10 to 100) adjacent to the haul road for a large Pb-Zn mining operation in northwestern Alaska (Ford and Hasselbach, 2001).

4.6.3.3. **Higher plants**

Steinnes *et al.* (2000) studied element concentrations in downy birch (*Betula pubescens*), bilberry (*Vaccinium myrtillus*), and four other higher plants on a south-to-north transect centered on the Pechenganickel smelter on the western Kola Peninsula. Despite variability probably due to hyperaccumulation by some species, concentrations of many elements, including Pb, were elevated close to the smelters. Regional data on higher vegetation is scarce. Due to root uptake and species-specific redistribution, vascular plants are not considered suitable biomonitors for most elements.

4.6.3.4. **Birds**

In the first AMAP assessment, data on willow and rock ptarmigan were reported for Canada, Alaska, and Scandinavia. More recently, Pedersen and Fossøy (2000) reported that Pb, like Cd, was generally higher in western than central Canada, but reached much higher levels in southern Scandinavia (up to 5 mg/kg dw), decreasing northward. At two Russian sites in eastern Siberia, Pb concentrations in ptarmigan liver were 1.6 to 2.3 mg/kg dw, in the same range as the lower Scandinavian values (Pedersen and Fossøy, 2000). The single values for Pb concentrations in willow and rock ptarmigan liver in northern Quebec are about 1 mg/kg dw, with highly variable muscle concentrations (< 0.08 to an extreme outlier of about 42 mg/kg dw; Champoux *et al.*, 1999). Similarly, Pb concentrations in muscle of waterfowl range from < 0.08 mg/kg dw to one extreme outlier of about 48 mg/kg dw in surf scoter (Champoux *et al.*, 1999). Lead poisoning as a result of ingesting lead shot has been implicated in decreased survival rates for spectacled eider on the Yukon-Kuskokwim Delta of southwestern Alaska (Grand *et al.*, 1998).

4.6.3.5. **Mammals**

Mean Pb concentrations have been reported for reindeer/caribou liver for a variety of locations (Figure 4·32). Concentrations in Greenland were 0.1 to 0.9 mg/kg ww, with highest values in Akia and Isortoq in the southwest (Aastrup *et al.*, 2000). Concentrations in Canadian caribou were highest in the Lake Harbour Herd at southern Baffin Island (mean 2.3 mg/kg ww), followed by Northern Quebec caribou (0.8 mg/kg ww) (Macdonald *et al.*, 2002, Annex Table A6). Liver Pb concentrations from Arctic Alaska were in the same range, with highest levels at Barrow (1.1 mg/kg ww), except for slightly higher concentrations near a large Pb-Zn mine in northwestern Alaska (mean 1.7 mg/kg ww; O'Hara *et al.*, 2003). Swedish liver samples showed a general mean of 0.1 mg/kg ww (Odsjö, 2003). Most

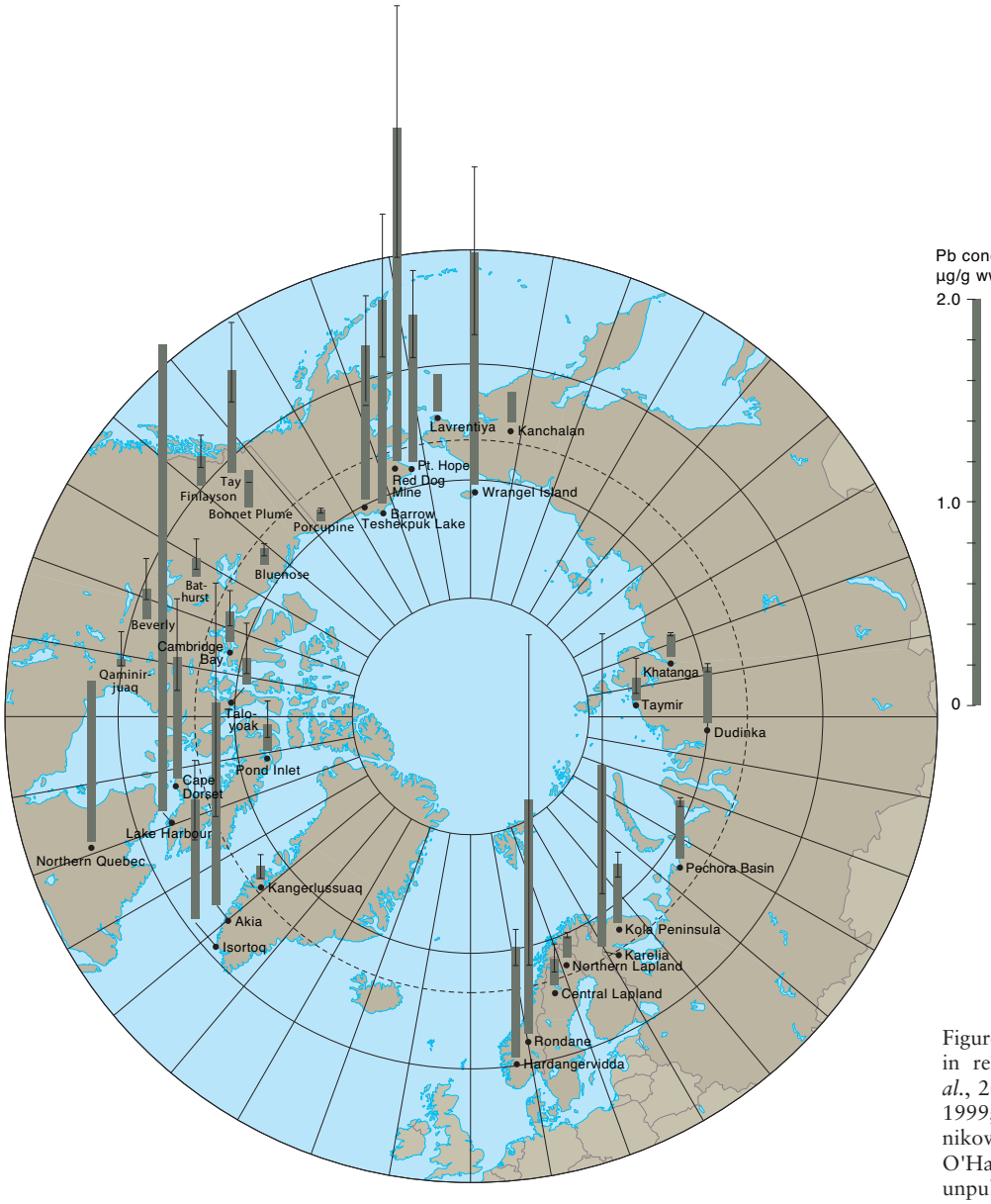


Figure 4.32. Mean lead concentrations in reindeer/caribou liver (Aastrup *et al.*, 2000; Elkin, 2001; Espelien *et al.*, 1999; Macdonald *et al.*, 2002; Melnikov *et al.*, 2002; Odsjö, 2003; O'Hara unpub. data, 2002; F. Riget, unpubl. data, 2002).

samples in the recent Russian survey had Pb concentrations in reindeer liver similar to those from Sweden (0.1 to 0.3 mg/kg ww), with maxima in the Pechora Basin, Taymir, and the Kola Peninsula (Melnikov *et al.*, 2002). Earlier data from the 1990s showed elevated levels at Wrangell Island (1.15 mg/kg ww), Karelia and Kola Peninsula (0.9 mg/kg ww), and Chukotka (0.5 mg/kg ww), similar to the high levels found at two sites in southern Norway (Hardangervidda and Rondane, 0.6 and 1.2 mg/kg ww, respectively; Espelien *et al.*, 1999).

The large-scale spatial distribution of Pb concentrations in reindeer/caribou showed some similarities to that for Cd, with high levels in Alaska and the Yukon (Canada). For Pb, concentrations were also high in southern Baffin Island. The distribution pattern for Greenland was similar to that for Cd and Hg with highest levels in the southwest. Lead patterns in Russia were also similar to those for Cd and Hg, with the highest levels in Pechora Basin and at Dudinka on the Taymir Peninsula. The Swedish sites had the lowest concentrations; two sites in southern Norway had higher values, comparable to Arctic Alaska.

Older studies documented large-scale spatial patterns for several mammals in Scandinavia, with decreasing metal levels from south to north that parallel those for moss, soils, and lake sediments. For example, in 1981 Pb concentrations in liver and kidney of bank vole were an order of magnitude higher in southern than northern Sweden (Nyholm and Rühling, 2001). In 1986, Pb concentrations in liver of the common shrew were two to five times higher in southern than northern Sweden (Lithner *et al.*, 1995). In Norway, elevated Pb concentrations were found in liver of hare and moose in the south (Kålås and Lierhagen, 1992). Similar patterns occurred in lamb and other cervids (Frank and Petersen, 1984; Frøslie *et al.*, 1984; Selinus *et al.*, 1996; Steinnes, 2001). All these studies are consistent with Pb body burdens reflecting atmospheric deposition in this region. Decreases have occurred over time. For example, by 1995, Pb concentrations in kidney and liver of voles from southern Sweden had decreased to 30% of the 1981 levels. This is also consistent with observed changes in atmospheric deposition.

New data on Pb concentrations in hare have been

reported for Russia, Norway, and northern Quebec (Annex Table A6). The Russian data show concentration ratios of 1 : 2.5 : 0.2 for kidney, liver, and muscle. Thus, in Russian hare Pb primarily accumulates in liver, whereas for Hg and Cd accumulation primarily occurs in the kidney. Russian hares had relatively low liver Pb. Lowest concentrations occurred in Chukotka and Khatanga, and the highest in the Pechora Basin and the Dudinka region, similar to the situation for Cd. The Norwegian hare data show Pb concentrations in liver to increase with age; hares two to eight months old (n = 71) have a mean Pb level of 0.14 mg/kg ww; hares over 14 months old (n = 49) have a mean Pb level of 0.29 mg/kg ww (assuming a 75% moisture content). The younger hares have concentrations more typical of the AMAP Phase II data set, while the older hares have concentrations second only to those from northern Quebec (mean 2.07 mg/kg ww).

4.6.4. Lead in the freshwater environment

4.6.4.1. River water and sediments

A summary of new data on Pb in river water and sediments is provided in Section 3.4.

4.6.4.2. Lake water

Large-scale spatial patterns exist in the Pb concentrations of lake waters for the 3000 lakes studied in the Scandinavian/Kola Peninsula study (see Section 4.4.4.2; Figure 4·33). Decreasing concentrations northward parallel the trend for mosses and the humus layer of forest soils and provide independent evidence for the regional influence of long-range atmospheric contaminants.

4.6.4.3. Lake sediments

The Swedish national lake survey of 1998 to 2000 (see Section 4.4.4.3) showed highest Pb concentrations in surface sediments of lakes in southern Sweden (150 to 200 mg/kg dw), similar to the trend in the moss, soil, and lake water studies. A sediment core study of 210 lakes during the Norwegian national lake survey in 1996 to 1997 showed Pb to have one of the highest EFs for all the elements studied (Rognerud and Fjeld, 2001). The decreasing south-to-north Pb gradient is similar to that for Cd.

In the border area between Norway and Russia, a sediment core study of about 100 lakes showed a concentric pattern for several pollutants (e.g., Cd and Hg) around the Murmansk smelters (Dauvalter, 1994; Dauvalter and Rognerud, 2002), but not for Pb. Lead enrichment increased westward, possibly indicating source contributions by vehicle emissions from northern Finland and northeastern Norway (Dauvalter, 1994). This is consistent with reports indicating that Pb is not a major component of the smelter emissions in the Kola region (e.g., Pacyna *et al.*, 1993).

Bindler *et al.* (2001a) found higher Pb EFs in lakes closest to the coast in Greenland, decreasing along a 150 km transect to the inland ice sheet. This pattern is the inverse of that for Hg (see Section 4.3.4.2), and may reflect local sources of Pb close to the coast.

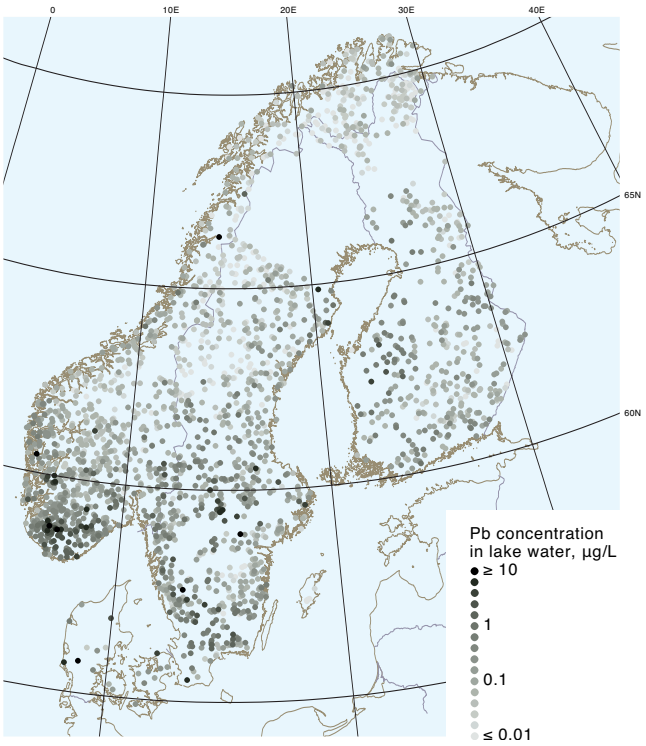


Figure 4·33. Lead concentrations in lake waters in Scandinavia, 1995 (Skjelkvåle *et al.*, 2001).

A lake sediment core from the eastern Taymir Peninsula in central Siberia indicated no recent upcore increases in Pb (Allen-Gil *et al.*, 2003), suggesting that emissions from anthropogenic activities at Norilsk, Dikson, and Khatanga do not reach this remote region.

4.6.4.4. Freshwater fish

Muir *et al.* (2001b) studied Pb concentrations in muscle of landlocked Arctic char from Nunavut, reporting mean concentrations of 0.007 to 0.03 mg/kg ww except for one highly variable location (n = 17, 0.129 ± 0.486 mg/kg ww). Limited older data from West Greenland show similarly low levels in sea-run Arctic char (F. Riget, pers. comm., 2001) and slightly lower concentrations (0.001 to 0.005 mg/kg ww) in Labrador (Muir *et al.*, 2000).

Comparable data for other parts of the circumpolar Arctic are scarce. Mean Pb concentrations in Arctic char liver from the Canadian High Arctic are 0.006 to 0.064 mg/kg dw (Annex Table A10), generally bracketing mean values for Pb in Arctic char from Swedish Abiskojaure (Àbiskojávre) between 1981 and 2000. Relative to Hg and Cd, Pb is of limited concern in fish tissue.

4.6.5. Lead in the marine environment

Naidu *et al.* (1998, 2001b) analyzed Pb concentrations in sediments from coastal shelves throughout the Arctic. Table 4·1 compares observed concentrations against levels estimated to induce biological effects (Long *et al.*, 1995). In no case did Pb concentrations approach levels of concern.

4.7. Other elements

4.7.1. Atmospheric concentrations of other elements

The associations between various aerosol constituents present at Alert between 1980 and 1995 (see Sections 4.6.1 and 5.1) showed that several elements (V, Mn, Pb, and Zn) were mainly linked with an anthropogenic aerosol factor that peaked in January to March and included ammonium as well as sulfate ions (Siriois and Barrie, 1999). This factor is interpreted as anthropogenic combustion aerosols entering the Arctic from Eurasia. Another factor, dominated by Cu as well as Zn, is associated with smelter emissions. The smelter factor does not include a contribution from sulfate, because most of the sulfur associated with Arctic smelting operations occurs as sulphur dioxide gas, which in this factor analysis is reflected in the first (primary) component.

These analyses confirm that the major source region influencing the concentrations at Alert is primarily Eurasia. However, other regions adjacent to northern Canada but at higher elevations have a different aerosol mix and may be more influenced by emissions from North America (Sirois and Barrie, 1999).

4.7.2. Other elements in the terrestrial environment

4.7.2.1. Soils and peats

Three Norwegian surveys (1977, 1985, and 1995; see Section 4.4.3.1) of heavy metals in the organic humus layers of forest podzol soils found decreasing south-to-north trends for Zn, arsenic (As), Sb, and other elements. Analyses of surface peat from a large number of Norwegian ombrotrophic bogs showed similar results (Steinnes, 1997).

Parts of northern Norway, such as the Sør-Varanger area, are influenced by Russian smelter emissions to the east (see Sections 4.4.3.2 and 4.5.2). For example, during the Kola Ecogeochemistry Project extremely high Cu and Ni concentrations were found in Russian O-horizon soils near industrial centers. These decreased sharply to the west, although elevated concentrations of Cu and Ni were still found in O-horizon soils in Finland and northern Norway (Reimann *et al.*, 1997a, 1998). A large proportion of the phytoavailable Ni occurred in the O-horizon, with smaller amounts in the B-horizon, and almost none in the C-horizon (Räisänen *et al.*, 1997). Similar features were reported for Cu in the O-horizon (Räisänen *et al.*, 1997; Steinnes *et al.*, 1997).

4.7.2.2. Mosses and lichens

Results of recent moss surveys (1995, 2000) covering most of Europe, including the Kola Peninsula, have shown moderately elevated metal concentrations in mosses relative to natural background concentrations, although local exceptions occur (Figure 4·34). For example, some locally elevated concentrations of Cu and As (as well as Pb and Cd) are evident along the Swedish coast of Bothnian Bay, probably related to smelter emissions (cf. Figures 4·34, 4·18, 4·29). Copper concentrations in *Hylocomium splendens* collected between 1990 and 1993 in Arctic Alaska were similar to those from the cleaner areas of the Nordic/European moss monitoring network (Ford *et al.*, 1995). The Kola Ecogeochemistry Project data showed smelter emissions on the Kola Peninsula around Monchegorsk, Nikel, and Zapoljarnij to cause regionally elevated levels of Cu and Ni in monitoring mosses as far away as northern Norway and Finland (Äyräs *et al.*, 1997), similar to those in O-horizon soils. In the Norwegian moss survey of 2000, concentrations of Ni, Cu, and cobalt (Co) were still elevated in the Varanger area of northern Norway as a result of Kola emissions (Steinnes *et al.*, 2001). Of the more than 50 additional elements in the survey, V, Zn, As, Sb, tin (Sn), molybdenum (Mo), silver (Ag), thallium (Tl), and bismuth (Bi) in that region of Norway also appear to originate at least in part from long-range airborne pollution (Steinnes *et al.*, 2001). On the Russian Taymir Peninsula 80 to 300 km north of Norilsk, concentrations of heavy metals and trace elements in *H. splendens* were generally similar to those in the cleaner areas of the Nordic/European moss monitoring network (Allen-Gil *et al.*, 2003; Ford *et al.*, 1997). These results are consistent with those of Blais *et al.* (1999) in suggesting that the impact on tundra ecosystems of the large Norilsk smelting complex is localized, and does not extend northward beyond about 100 km.

Using *Racomitrium lanuginosum* Riget *et al.* (2000b) found that concentrations of Zn, Ni, Al, and Cd in Greenland were similar to those in the same species from Arctic Alaska between 1990 and 1993. In contrast, concentrations of Cu, iron (Fe), V, Pb, and Hg tended to be higher in Greenland.

Element concentrations in moss are affected by a variety of factors in addition to atmospheric deposition of pollutants. For example, marine aerosols are a source of Se and Mg. Concentrations of Mn, Mg, Ca, rubidium (Rb), cesium (Cs), Zn, and possibly to some extent Cu and Cd, can be elevated by foliar leaching from overstory vascular plants that take up and translocate certain elements (Steinnes, 2001). Zinc is an example of an element for which the moss monitoring technique is known not to work. In the Kola Ecogeochemistry Project, large regional processes do not influence moss concentrations of Zn, and local variability is as high as regional variability (Äyräs *et al.*, 1997). Some of this local variability in Zn may be due to hyperaccumulation, as well as to leaching from overstory plants.

The effect of windblown dust may be especially important for Co, chromium (Cr), Cu, Fe, Ni, Pb, and V in areas with sparse vegetation, such as Arctic tundra (Riget *et al.*, 2000b; Steinnes, 1995) and must be accounted for when assessing anthropogenic contributions of elements (Steinnes *et al.*, 2001). Soil dust particles can also be important for minor elements such as lithium (Li), yttrium (Y), thorium (Th), uranium (U), beryllium (Be), titanium (Ti), zirconium (Zr), niobium (Nb), hafnium (Hf), tantalum (Ta), tungsten (W), and the lanthanide elements. Enrichment factors that normalize concentrations in vegetation to local soil parent material (see Section 4.4.3.2) have been used for Arctic Alaska and the Taymir Peninsula (Ford and Hasselbach, 2001; Ford *et al.* unpubl.). In both regions, moss Cu, V, and Pb derive primarily from soils (EFs generally <10; Ford *et al.*, unpubl.).

Strong gradients in metal concentration have been reported in Sphagnum moss (*Sphagnum* spp.) as well as for lichens (*Cladonia* spp. and *Cetraria* spp.) on the Kola

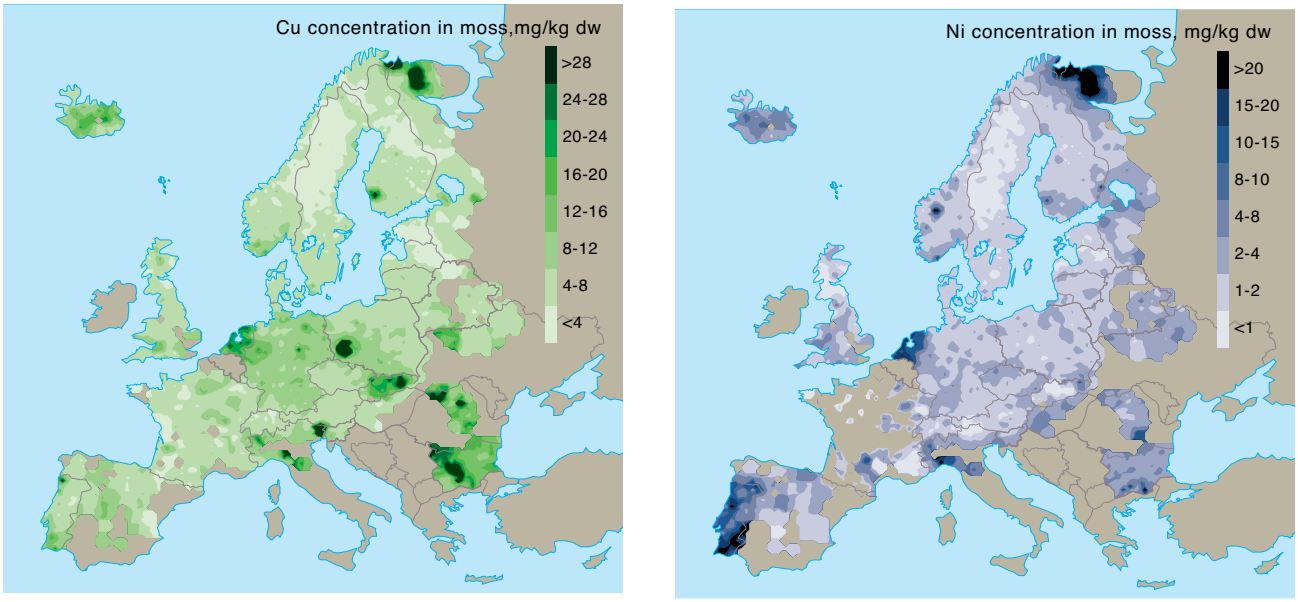


Figure 4.34. Copper and nickel concentrations in European feather mosses, 1995 (Rühling and Steinnes, 1998).

Peninsula (Tsibulski *et al.*, 2001), with Cu concentrations at 30 times background, Ni at 105 times background, and Co at 50 times background. Although high concentrations near point sources diminish to background levels with distance, generalizations are difficult due to differences in factors including local geology.

Enrichment factors for the lichen *Cetraria cucullata* from five sites near industrialized areas were much higher for Cu and in some cases V (Figure 4·20). The elevated EFs for V are unique in the data set and are probably related to oil combustion (see Section 2.1.1).

4.7.2.3. **Higher plants**

Studies of vascular plants in the Sör-Varanger area in northern Norway reflect inputs from smelter emissions on the Kola Peninsula. Leaves of bilberry and crowberry (*Empetrum nigrum* ssp. *hermaphroditum* (= *E. hermaphroditum*)) showed increased Cu and Ni concentrations at polluted versus background sites (Table 4·7). Nickel levels in leaves were positively correlated with the phytoavailable fraction of Ni in the soil (Uhlig and Junttila, 2001). Crowberry leaves generally accumulate more Ni than bilberry leaves and at several sites had Ni concentrations above reported levels for plant toxicity (Uhlig and Junttila, 2001). Further, discolored leaves had much higher concentrations of both elements than green leaves, suggesting a link between metal uptake and physiological function. These results suggest possible regional impacts on crowberry from the Sør-Varanger area due to smelter emissions (Uhlig and Junttila, 2001). Toxicity levels are species-specific and further work is required on this matter.

Steinnes *et al.* (2000) studied element concentrations in downy birch, bilberry, and four other higher plants on a north-to-south transect centered on the Pechenganickel smelter in the western Kola Peninsula. Despite variability, many elements, including Cu, Ni, and Co were elevated close to smelters.

Ford *et al.* (1995) found that Cu concentrations were high in blueberry (*Vaccinium uliginosum*) relative to the monitoring moss *Hylocomium splendens*. This was different than the situation for Pb and Hg, which occurred in low concentrations in blueberry relative to the monitoring moss, and Cd, which was present at similar concentrations.

4.7.2.4. **Birds**

Champoux *et al.* (1999) found low (< 0.25 mg/kg ww) concentrations of Se in muscle of rock ptarmigan, with As concentrations <0.05 mg/kg ww. Selenium concentrations in ptarmigan liver from Greenland were <0.25 mg/kg ww, with levels about twice that in kidney (Annex Table A5).

In Canada, concentrations of Se showed more variability among waterfowl species than Hg, with highest levels in liver of common eider and long-tailed duck (23 mg/kg ww), and median values for other species in the range 2 to 8 mg/kg ww (Braune *et al.*, 1999a). In northern Quebec, Se concentrations in waterfowl liver exceeded 1 mg/kg ww in liver and/or muscle of scoters and eiders, common loon (*Gavia immer*), and osprey (Champoux *et al.*, 1999). Arsenic concentrations were generally low (<0.1 mg/kg ww) in muscle of geese and surface-feeding ducks, but higher in muscle of diving and piscivorous ducks, particularly scoters and loons (Champoux *et al.*, 1999).

Pedersen and Fossøy (2000) reported on several elements in ptarmigan from Canada, Scandinavia, and Russia. Copper and Zn in liver were fairly uniformly distributed but with higher levels in Canada. The difference was more pronounced for kidney, where Cu in particular was higher in Canada than in Scandinavia and Russia.

4.7.2.5. **Mammals**

Moose is the species chosen to monitor environmental contaminants in Swedish forest areas (Odsjö *et al.*, 2001). Metals in moose liver and kidney from the Swedish monitoring program do not show significant differences between southern and northern Sweden (Odsjö *et al.*, 2001). In contrast, decreasing south-to-north gradients occurred in liver of moose and other

cervids in Norway for As and Se (Frank and Petersen, 1984; Frøslie *et al.*, 1984; Selinus *et al.*, 1996), consistent with the spatial patterns for Hg, Cd, and Pb, and probably linked to atmospheric deposition.

In 1999 and 2000, studies on common shrew in northern Finland found significant differences between immature shrews and adults for Zn and Rb. Trace elements occurred in the order Zn > Rb > Cu > Mn > Al > Cd > Pb > Cr > Mo > As > V > Ag > platinum (Pt) (J.P. Hirvi and H. Henttonen, pers. comm., 2002).

In Greenland, the spatial pattern of Se in reindeer was similar to that for Hg. Highest liver concentrations occurred at Isortoq in southern Greenland (1 mg/kg ww in 1994 and 0.26 mg/kg ww in 1999) and lowest at Kangerlussuaq (0.09 mg/kg ww) (Aastrup *et al.*, 2000; Annex Table A6) with intermediate concentrations in the Nuuk area (0.3 mg/kg ww). Selenium in caribou liver from the Northwest Territories was about 0.3 mg/kg ww (Elkin, 2001), similar to the higher end of the Greenland samples.

4.7.3. Other elements in the freshwater environment

4.7.3.1. River water

The major Russian rivers draining into the Arctic Ocean were monitored between 1990 and 1996 for Fe, Cu, Zn, and in some cases Ni (Alexeeva *et al.*, 2001). Concentrations of Cu averaged 4 to 8 µg/L, slightly higher than in the Mackenzie River and other smaller rivers in Canada (3.1 µg/L and 1 µg/L, respectively), and much higher than smaller rivers in northern Norway (0.6 µg/L) (Alexeeva *et al.*, 2001). Similarly, Zn concentrations in the Russian rivers were 10 to 30 µg/L, compared to 1.5 to 15 µg/L in Canada and 0.8 µg/L in northern Norway. Concentrations of Fe in the Russian rivers were lower than in the Mackenzie River (200 to 1000 µg/L versus 2200 µg/L, respectively).

The Ob, Yenisey, and Lena rivers dominated metal fluxes to the Arctic Ocean. For example, the Ob delivered 39% of the total Fe flux, followed by the Yenisey (14%), and the Lena (9%). Copper was more equally transported, with the Ob contributing 27% of the total, the Yenisey 34%, and the Lena 17%. Zinc was discharged primarily by the Yenisey (46%), followed by the Ob (21%), and the Lena (12%). The basin receiving the largest metal loads was the Kara Sea. Annual fluxes of Fe, Zn, and Cu to this basin, were 855000, 42000, and 10000 t/yr, respectively (Alexeeva *et al.*, 2001).

The export of elements in river sediments to the Arctic Ocean is discussed in Section 3.4.

4.7.3.2. Lake water

In 1995, national lake surveys in Scandinavia and on the Kola Peninsula revealed large-scale spatial patterns for Zn (similar to Cd and Pb and to a certain degree Co). Again, concentrations decreased from south to north, similar to patterns in moss and humus layers of forest soils. Total organic carbon was associated with increased Fe, Mn, and to some extent As, Cr, and V. Bedrock geology was the primary controlling factor for Cu (Figure 4·35) and Ni, except near the smelters on the Kola Peninsula. Similarly, smelter emissions in northern Sweden influenced As distribution.

The 21 lakes of the Tundra North West 99 expedition (Canadian Arctic) were mostly hard-water lakes, as reflected by high concentrations of Ca and Mg as well as relatively high pH and alkalinities (Table 4·6). Soft-water exceptions were on Ellef Ringnes Island and at Cape Hooper, southeastern Baffin Island. In general, the trace metal concentrations in lake water were low and can be considered natural background concentrations (Borg *et al.*, 2001). The influence of catchment lithology was reflected in elevated concentrations of Cu and Cd in Ivvavik National Park, northwestern Yukon Territory, and on Ellef Rignes Island. A shallow lake on the Cape Bathurst coastal plain had elevated concentrations of As (Borg *et al.*, 2001).

4.7.3.3. Lake sediments

Sediment cores were taken from 210 lakes during the 1996 to 1997 Norwegian national lake survey (Rognerud and Fjeld, 2001). Elements with the highest EFs included Sb, Bi, As, Hg, Pb, and Cd. A spatial distribution similar to that for moss and the humus layer of forest soils was found with higher concentrations of Sb and Bi in the south; Zn, like Cd, in the southeast; and As in the southwest. Concentrations decreased with increasing latitude and altitude. One exception of elevated concen-

Table 4·7. Metal concentrations mg/kg dw in bilberry (*Vaccinium myrtillus*) and crowberry (*Empetrum hermaphroditum*) leaves in Sør-Varanger, northern Norway (Uhlig and Junttila, 2001).

Sample site	Copper			Nickel		
	Bilberry (green leaves)	Crowberry (green leaves)	Crowberry (discolored leaves)	Bilberry (green leaves)	Crowberry (green leaves)	Crowberry (discolored leaves)
Polluted sites						
1	11	12	58	14	21	116
2	10	10	31	11	14	49
3	11	8	29	11	11	46
4		6	17		13	42
5	11	8	22	11	12	41
Background sites						
19	7	6	8	2	4	10
20		4	4		4	6
21		4	4		4	8
22		5	5		4	8

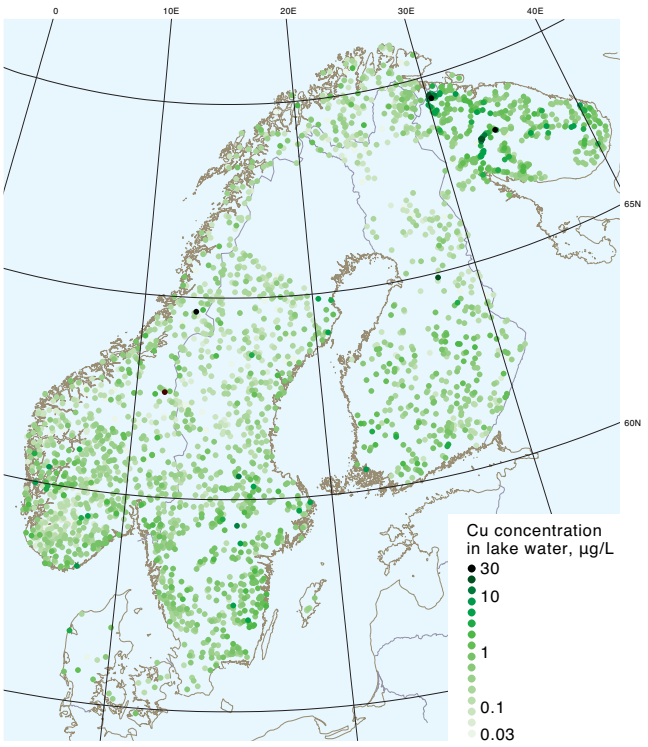


Figure 4.35. Copper concentrations in lake waters in Scandinavia and on the Kola Peninsula, 1995 (Skjelkvåle *et al.*, 2001).

trations of Ni and Cu occurred in northeastern Norway close to the Russian border. Regional distribution patterns for Ni and Cu are probably due to long-range atmospheric deposition from Russian smelters to the east (Rognerud and Fjeld, 2001).

Although the concentric pattern of several elements around the Murmansk smelters on the Kola Peninsula (Dauvalter, 1994) showed significant negative correlations with distance from the smelter ($p = 0.05$), Ni and Zn did not show the same relationship ($p = 0.137$ and 0.391, respectively). In fact, the proportion of soluble (toxic) to total (soluble + particulate) Ni increased with distance from both the Severonickel and the Pechenganickel smelters, with about 90% occurring as soluble Ni at 150 km from the sources. Elevated Zn was found only in a few lakes within 10 km of the smelters.

A lake sediment core from the eastern Taymir Peninsula indicated no recent enrichment of Cu or Ni (Allen-Gil *et al.*, 2003), suggesting that heavy metal emissions from the large Norilsk smelting complex some 300 km to the southwest are not a regional issue with respect to Cu and Ni deposition.

4.7.3.4. Freshwater fish

Data have been reported for several elements in Canadian Arctic freshwater fish (e.g., Muir *et al.*, 2000, 2001b; Stern *et al.*, 2000, 2001a). Selenium is of particular interest because of its ameliorating influence on Hg. The highest concentrations of Se in freshwater fish were found in Arctic char from Lake Thingvallavatn, Iceland (4 mg/kg ww; Snorrason and Jónsson, 2000). In Canada, significantly lower Se levels were found in char from Tasuijaq (Nunavik, Canada) than Labrador and other locations (Muir *et al.*, 2000). A study of landlocked and anadromous Arctic char populations in northern Labrador and Nunavik, showed higher Se levels in landlocked char from Kangiqsujuaq than in sea-run char, associated with three-fold higher levels of Hg (Muir *et al.*, 2000). A weak tendency for liver Se to decline with increasing fish length was reported for burbot at Fort Good Hope, Northwest Territories (Stern *et al.*, 2000).

4.7.4. Other elements in the marine environment

Concentrations of several elements have been analyzed by Naidu *et al.* (1998, 2001b) in marine sediments from coastal shelves throughout the Arctic. Table 4·1 compares concentrations to estimated levels for biological effects (Long *et al.*, 1995). Values greater than the ER-L (effect range-low) level reflect concentrations of potential concern. Many coastal sediments had concentrations above the ER-L level for Cr and Ni. In East and West Greenland, mean values of Ni in marine sediments were above the ER-M (effect range-medium) level (Naidu *et al.*, 1998; 2001b). Regionally high concentrations of many trace metals probably derive from the erosion of volcanic rocks of the Mesozoic-Tertiary Provinces of Greenland (Loring and Asmund, 1996).

4.8. Assessment of spatial patterns

Spatial coverage in contaminant concentrations has improved significantly since the first AMAP assessment. Progress in obtaining measurements for different media in the same region is also being made. In general, there is a continued need for standardizing sampling and analytical protocols and reporting conventions. Continued progress is important for the clear delineation of spatial patterns.

4.8.1. Atmosphere and precipitation

Atmospheric Hg concentrations at five Arctic stations were relatively homogenous at about 1.5 ng/m^3. Comparable data for ambient precipitation are sparse, and were only available for one of the five stations. Mercury concentrations appear higher in marine than inland environments. Analysis of spatial patterns in Hg concentration from late spring snowpacks can be confounded by MDEs.

Significant work continues on the measurement of Pb aerosols in Arctic air. Some stations have begun to include stable isotope characterization, which can assist in identifying source regions and evaluating back-trajectories. At Alert, recent studies have confirmed earlier reports of seasonally variable sources, with late autumn and early winter aerosols dominated by western European sources, late spring and summer aerosols dominated by eastern European and Russian sources, and autumn aerosols dominated (at much lower concentrations) by sources in West Greenland and the Arctic Canadian Archipelago.

Arctic air is not homogeneous, at least with respect to Pb, although part of the evidence for this derives from now-discontinued Alaskan stations. Stations at different locations and elevations can be expected to sample different aerosol mixes, which in turn will propagate in

characteristic ways throughout receptor ecosystems as a function of, e.g., precipitation, continental influences, geology, vegetative cover, and relief. Thus, for example, the atmospheric monitoring record at Alert is not a particularly good tool for interpreting the high elevation ice core record at Summit, Greenland, to the east. Spatial patterns in snowpack chemistry reflect these complexities in time and space. For example, the snow sampling program spanning the Arctic Ocean from the Arctic Canadian Archipelago to the Russian Arctic islands found distinct spatial patterns in Pb and Cd, with general maxima at the Franz Josef Islands, probably due to anthropogenic contributions in late winter/spring.

4.8.2. Terrestrial and freshwater environments

Spatial patterns in moss concentrations of several heavy metals and trace elements appear to provide a robust record of current (3 to 5 yr) atmospheric deposition. Clear spatial patterns appear for many elements in moss monitoring programs with sufficiently dense coverage, and these have changed over time with changing emissions. For example, decreased Pb deposition in response to bans on leaded gasoline is apparent throughout Scandinavia. Associated decreases in Cd may be related to increasing controls on industrial processes. However, comparable information is not available for vast sections of the continental Arctic, including Canada, much of Russia, and to some extent Greenland. As moss monitoring extends to these regions, it will be important to consider the potential influence of soil dust, especially in areas with sparse vegetation.

Concentrations in surficial soils and lake sediments have been widely interpreted to reflect atmospheric deposition patterns over long time periods, particularly when concentrations are normalized to pre-industrial deposits to eliminate the influence of local geology (the enrichment factor – EF – approach). Lake sediments can be dated using radiometric methods, permitting correction for changes in sediment accumulation rates (i.e., permitting a more sophisticated comparison of recent to ancient flux rates). As with other matrices, matrix-specific factors must be accounted for to enable the approach to achieve its full potential. For lake sediments, caution is particularly warranted if there are marked differences in organic carbon levels. However, even the EF approach is often sufficient to demonstrate spatial patterns if strong gradients are present. This is the case on the Kola Peninsula for Hg and Cd, and in Scandinavia where there are well-characterized decreasing south-to-north gradients. It is important to recognize that both sediments and soils represent a legacy, or bank, of contaminants that can continue to be tapped by organisms.

Large-scale spatial patterns in element concentrations in tissues of terrestrial and freshwater animals are difficult to observe, and may not necessarily reflect spatial patterns in anthropogenic inputs except in cases of very heavy deposition, such as near industrial sites. This is especially true for Hg and Cd, for which bioaccumulation depends in complex ways on factors such as catchment geology, water chemistry, forage vegetation, and trophic position etc. Spatial patterns in animal tissue may simply reflect geochemical environments, as is often the case for Cd, for example, Cd in Canadian caribou. However, where strong anthropogenic gradients do exist, either due to point sources (e.g. the Kola Peninsula and Norilsk) or to long-range atmospheric transport (Fennoscandia), biota have been shown to reflect that gradient quite closely.

Mercury concentrations in reindeer/caribou liver show high mean concentrations in southwestern Greenland and eastern Canada (means 0.6 to 1.0 mg/kg ww) as well as western Alaska. This is consistent with the temporally increasing Hg concentrations found in marine mammals at high trophic levels in these regions over recent decades (see Section 5.4.3.4). Older data from the Kola Peninsula and Karelia also reflect elevated concentrations, as do early data from Norway and the Wrangell Islands. On the Kola Peninsula, where more recent data are available, lower concentrations are now reported.

Lowest concentrations of Hg in hare liver are reported for northern Norway, Russia, and Greenland, with concentrations about an order of magnitude higher in the Faroe Islands.

Mercury concentrations are higher in landlocked Arctic char than in sea-run Arctic char. For landlocked char, concentrations are generally lower in Finland, northern Sweden, and Iceland than in Greenland, Chukotka-Lavrentiya (Russia), and the Faroe Islands. Lowest concentrations were found in Lake Thingvallavatn, Iceland. The range in concentration in Canadian lakes spans almost the full range of concentrations encountered in the entire AMAP region. Highest mean concentrations occurred in Greenland (0.49 mg/kg ww), but the range is wide with highest concentrations in western and southwestern Greenland paralleling similar relationships for lake sediments and the soils from the catchments.

Mercury concentrations are generally lower in whitefish (*Coregonus* spp. and *Prosopium* spp.) than Arctic char, except for instances involving point-source pollution. Whitefish concentrations are usually below 0.2 mg/kg ww. As with char, Canadian samples span a wide range in concentration, in this case having the highest mean Hg concentration (0.13 mg/kg ww; but note that whitefish do not occur in Greenland). Finnish, Russian, and Norwegian whitefish means are lower, and the very lowest mean values (0.03 mg/kg ww) were reported for whitefish in Alaskan rivers. Within Russia, Hg concentrations were lowest in European whitefish from the Pechora Basin, with intermediate values on the Taymir Peninsula, and highest concentrations on the Kola Peninsula.

Spatial coverage for burbot is not as extensive. The highest mean Hg concentration was found in the Kemijoki River, Finland, and the lowest in Great Slave Lake and on the Taymir Peninsula. In some parts of Canada, however, burbot Hg approaches that of the Kemijoki River (0.2 to 0.4 mg/kg ww).

In general, Hg concentrations in animal tissue are beginning to indicate maxima in eastern Canada and West Greenland as well as western Alaska. The Faroe Islands and parts of Chukotka also seem to have particularly high Hg concentrations in many compartments. Spatially extensive data for other environmental compartments such as plants, soils, and lake sediments would be helpful to better evaluate these spatial patterns.

4.8.3. **Marine environment**

Heavy metal concentrations in surface marine sediments within the High Arctic are relatively uniformly distributed. Regional and local geology, particle size, the amount of organic matter, and anthropogenic influence explain observed differences. Vertical movement of metals by bioturbation and geochemical processes add to the difficulties of evaluating spatial patterns in metal concentration in relation to large-scale contamination.

The blue mussel is widely used to monitor contaminants. Heavy metal data exist for Alaska, the eastern Canadian Arctic, West Greenland, the Faroe Islands, Iceland, and Norway. Cadmium concentrations appear higher in Alaska than in other Arctic areas, except for central West Greenland, which is known to have locally high Cd levels in biota. Mercury concentrations in blue mussel show no spatial pattern.

The circumpolar coverage of heavy metals in marine fish species is poor. Some regional differences in Cd concentration have been observed in shorthorn sculpin (higher levels in Greenland and the Faroe Islands) and Atlantic cod (higher levels in northwestern Iceland than the Faroe Islands and Norway). However, in all cases the cause is probably natural rather than anthropogenic. No significant regional differences were observed for Hg or Se in fish.

Since the first AMAP assessment, additional heavy metal data on seabirds have become available for Norway and Russia, which has improved circumpolar coverage for several species. Spatial patterns for some seabirds are partly explained by differences in the overwintering areas. In general, the highest Cd levels in seabirds occur in northeastern Siberia and the lowest in the Barents Sea, with intermediate levels in Arctic Canada and Greenland. Spatial patterns in Cd concentration in seabirds were also observed within the Barents Sea. Mercury levels in seabirds were generally lower in the Barents Sea than in Greenland, Canada, and northwestern Siberia. However, four eider species had similar Hg concentrations in Arctic Russia, Greenland, northwest Siberia, and the Barents Sea. Furthermore, long-tailed duck and herring gull from northeastern Siberia had higher Hg levels than in Greenland and eastern Canada. Spatial patterns were observed within the Barents Sea for several species. Mercury levels in seabird eggs in Canada showed spatial variations and were generally higher at higher latitudes. Selenium levels in four eider species were higher in Alaska and Arctic Russia than Canada, Greenland, and the Barents Sea. Spatial patterns in Se concentration occurred in seabirds within the Barents Sea, and in seabird eggs in Arctic Canada. However, these spatial patterns were less pronounced than for Hg.

The highest Hg concentrations in ringed seal, beluga whale, and polar bear were previously reported for western Arctic Canada, decreasing to the south and east. New data on Hg levels in beluga from western Canada support this trend, and beluga from Alaska show similarly high levels consistent with this pattern. However, Hg levels in ringed seal from the eastern Canadian Arctic, while highly variable, were generally greater than those observed in Alaska, West and East Greenland and Svalbard. Mercury concentrations in polar bear from Alaska appear lower than previously observed in western Arctic Canada, and new samples from East Greenland confirm the relative low Hg levels there. In the North Atlantic, Hg concentrations in minke whale from West Greenland and Svalbard were lower than in whales from the North Sea and around Jan Mayen.

The highest concentrations of Cd in ringed seal, beluga, and polar bear were previously reported for the eastern Canadian Arctic and northwestern Greenland. That spatial pattern is partly confirmed by this assessment. Cadmium concentrations in ringed seal from the eastern Canadian Arctic as well as West and East Greenland were higher than in Alaska and Svalbard. Furthermore, Cd levels in polar bear from Alaska were similar to those observed previously in western Canada. In the North Atlantic, Cd concentrations in minke whale were lowest in the most southerly areas such as the North Sea and the Norwegian coast.

Chapter 5

Temporal Trends

Birgit Braune, Peter Outridge, Simon Wilson, Anders Bignert and Frank Riget

Assessments of temporal trend data for heavy metals provide essential information for decision-makers concerned with science-based policy decisions about contaminants in the Arctic environment. 'Long-term' data sets, comparing recent with historical (i.e., pre-industrial) metal concentrations, may be used to estimate the relative importance of natural versus anthropogenic inputs to the Arctic environment. 'Short-term' data sets, covering the previous one to three decades, illustrate how metal concentrations have changed in recent times and may indicate probable trends in the near future. This chapter features assessments of long-term change and short-term trends.

The first AMAP assessment (AMAP, 1998) included information from the few studies on temporal trends in heavy metals in the Arctic environment available at the time. The main conclusions were as follows:

- All heavy metals in air showed strong seasonal variation in the High Arctic. Concentrations of some heavy metals in subarctic air had decreased over the preceding two decades.
- Mercury concentrations in Arctic sediments had increased over time, indicating that a widespread regional process was responsible. However, because anthropogenic emissions did not show the same pattern, but rather, have exhibited large decreases, firm conclusions could not be drawn from the sediment data without further investigations.
- Time-series data sets for Arctic biota were scarce. Some evidence was reported of a two- to three-fold increase in mercury (Hg) concentration in liver, and in some cases kidney, of some marine mammals over the preceding two decades. However, it was unclear whether these observations reflected a real increase or were due to interannual variation. Comparison of Hg concentrations in human and seal hair from Greenland (Kalaallit Nunaat) from the fifteenth century with concentrations in present day hair indicated a two- to three-fold increase since the pre-industrial period.

Information on temporal trends was therefore highlighted as an important gap in knowledge, particularly for the High Arctic. Such studies were given high priority for the second AMAP assessment, both in terms of continuing existing time series as well as new retrospective trend studies (e.g., using peat and sediment cores, and analyses of banked and preserved specimens).

Data are now available for several studies on temporal trends in metal concentrations in abiotic media. These are discussed in Sections 5.1 to 5.3.

An AMAP workshop was convened in 2001 to analyze in a consistent and statistically rigorous manner the newly available time-series data on metals in biological tissues (ICES, 2002). Analyses were confined to short-term time series containing at least four years of data; in some cases the data spanned longer time periods with gaps of varying lengths between data years.

The outcome of the workshop is summarized in Table 5·1. Detailed results are given in Appendix Table 5·1, while other biological data sets not covered by the workshop are summarized in Appendix Table 5·2. The

Table 5·1. Summary of statistical analyses of biological time series data sets covering the last 30 years.

Data type	Number of datasets	Trends, Hg	Trends, Cd	Trends, Pb	Time interval spanned by data, yr	Years to detect 5% annual change*	Adequacy of data median (range)
Canada							
Terrestrial mammals	6 (2 species, 1 at 5 sites, 1 at 1 site)	6 (/)	6 (/)	6 (/)	4-9	3-38	0.24 (0.10-0.73)
Marine birds	3 (3 species at 1 site)	2 (+); 1 (/)	n.a.	n.a.	24 (5-7)	9-16	0.64 (0.31-0.67)
Greenland							
Marine mammals	4 (1 species at 1 site, 4 tissues)	4 (/)	3 (/)	n.a.	16 (5-6)	12-38	0.22 (0.13-0.60)
Iceland							
Marine invertebrates	8 (1 species at 8 sites)	8 (/)	1 (+); 1(–); 6 (/)	n.a.	5-10	9-30	0.43 (0.20-1.00)
Marine fish	6 (2 species at 3 sites)	2 (–); 4 (/)	6 (/)	n.a.	10	11-38	0.50 (0.13-0.69)
Norway							
Marine invertebrates	8 (1 species at 8 sites)	2 (–); 6 (/)	8 (/)	8 (/)	4-13	6-27	0.40 (0.17-1.00)
Marine fish	2 (1 species at 2 sites)	2*	2*	n.a.	7-9	14-32	0.47 (0.28-0.53)
Sweden							
Freshwater fish	2 (2 species at 2 sites)	2 (/)	2 (/)	2 (/)	19-29	10-20	1.70 (0.90-2.64)
Terrestrial mammals	4 (2 species at 2 sites)	2 (/)	1(+); 3 (/)	1(–); 1(/)	5-16	11-31	0.61 (0.19-1.55)
United States							
Marine invertebrates	2 (1 species at 2 sites)	1(+); 1(/)	2 (/)	2 (/)	9-10	8-18	0.80 (0.56-1.25)

* power = 80%; significance level = 5%.

n.a.: data not available; (/): no significant change; (+): significant increasing trend; (–): significant decreasing trend. Adequacy of time series = ratio of number of years of data and number of years of data required to detect an average trend of 5% change per year. 1.0 = adequate statistical power to detect a trend of 5% change per year; <1.0 = inadequate power; >1.0 = more than adequate. For details, see Appendix Table 5.1.

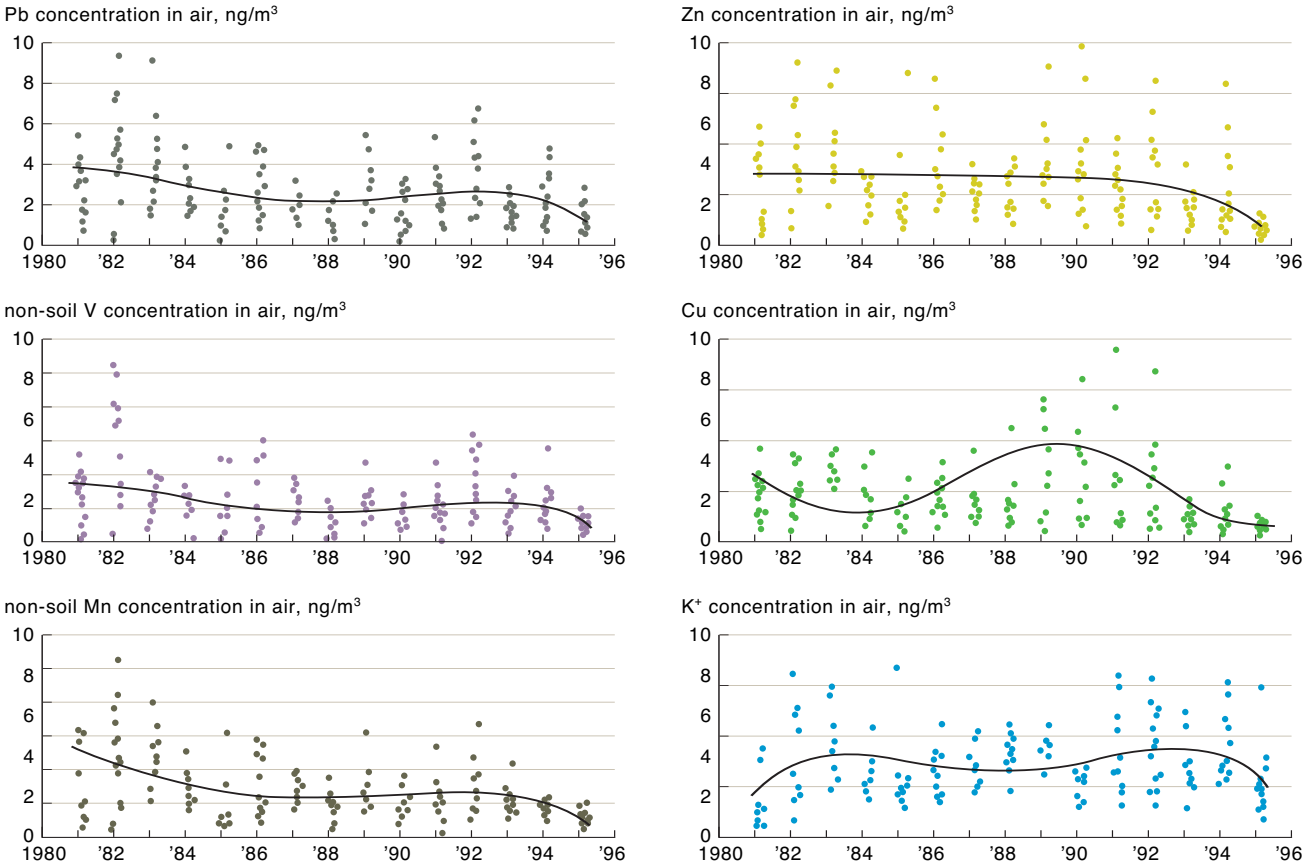


Figure 5·1. Trends in aerosol metal concentrations at Alert, Ellesmere Island, 1981 to 1995 (Sirois and Barrie, 1999).

analyses conducted at the workshop, were based on the objective of detecting annual average changes of 5% in metal concentrations with a significance level (i.e., a Type I error rate) of 5% and at a power level of 80% (i.e., a Type II error rate of < 20%). These parameter values are conventionally used in statistical power calculations (ICES, 2002).

Power analysis of the Greenland data in the first AMAP assessment indicated that, for a variety of contaminants in various species, a minimum of about ten years of temporal trend data was required to detect a statistically significant change in concentration of 10% per year (Riget *et al.*, 2000c). This estimate has been revised in the light of the new data.

5.1. Air and aerosols

The AMAP Thematic Data Center at the Norwegian Institute of Air Research (NILU) is the single largest repository of data on the Arctic atmosphere and stores data on metals in atmospheric media from 22 Arctic monitoring stations in seven countries. Of these, six constitute the main atmospheric background monitoring network in the Arctic.

Trends in the weekly mean concentration of 18 elements in aerosol collected on filters at Alert (Ellesmere Island, Canada) over a 15-year period (1981 to 1995) were reported by Sirois and Barrie (1999). Seasonal variation was evident, with concentrations of the anthropogenically-derived metals lead (Pb), zinc (Zn), and copper (Cu), and the non-soil components of vanadium (V) and manganese (Mn), at their maximum during winter and their minimum during summer. No long-term trends were detected in the annual average concentrations of most elements, with the exception of Cu which increased from 1980 to 1987 and then declined. However, trend analyses for the winter period only showed consistent decreases in the concentrations of Pb, Zn, and the non-soil components of V and Mn since 1980. The decline in V at Alert occurred against a global trend of increasing V emissions for this period (see Section 2.2.1).

Associations between metals help explain trends and sources of the metals (Sirois and Barrie, 1999). Applying the positive matrix factorization approach showed that Pb, V, Mn, and Zn were linked to a factor that peaked in January to March and included non-acidic salts of sulfate, with ammonium as the dominant cation. This factor, which generally reflects anthropogenic combustion sources, gradually declined over the 15-year period (Figure 5·1). Another factor was dominated by Zn and Cu. This factor did not include a contribution from sulfate (which mostly forms in the Arctic atmosphere); in this case the associated sulfur reflects primary emissions from smelters in the Arctic airshed (Sirois and Barrie, 1999). The long-term trend in this factor showed a slight increase in the 1980s, and then in 1991 a sharp decrease which continued until measurements ceased in 1995. As this decline occurred in parallel with the break-up of the Soviet Union and the resulting decline in industrial activity at the smelters, Sirois and Barrie (1999) conclude that the decrease reflects the reduced smelter emissions at that time. These analyses confirm that the major source region influencing airborne concentrations at Alert is Eurasia. Regions close to Alert, but at higher elevations,

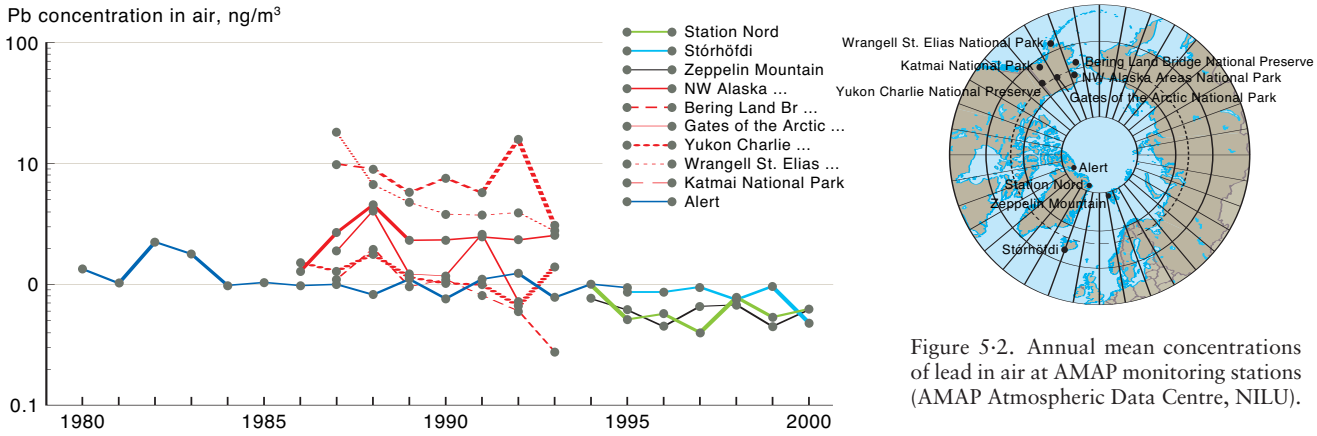


Figure 5·2. Annual mean concentrations of lead in air at AMAP monitoring stations (AMAP Atmospheric Data Centre, NILU).

such as the Greenland Summit, have a different aerosol mix and may be more influenced by emissions from North America (Sirois and Barrie, 1999).

Macdonald *et al.* (2003) propose a different explanation for the decline in aerosol metal concentrations at Alert since 1991. They postulate that rather than resulting from reduced emissions following the collapse of industry after the break-up of the former Soviet Union the decline could at least in part be explained by the effects of (natural) shifts in climate regime on contaminant pathways. Around 1989, the Arctic Oscillation (AO) Index shifted from a predominantly negative state to an extreme positive state and this persisted throughout much of the 1990s. Positive AO conditions are associated with increased precipitation over the Nordic Seas and southern Eurasian Basins, which can enhance particle scavenging from the atmosphere, and so reduce the pollution load at receptor sites such as Alert.

Annual mean concentrations of Pb in air/aerosols at Arctic monitoring sites are shown in Figure 5·2. Concentrations are generally higher at the six Alaskan monitoring sites than at those in Greenland and Europe, however, the recording periods vary (1986/87 to 1993 in Alaska and 1994/95 to 1999 in Greenland/Europe); only at Alert does the time series extend throughout the 1980s and 1990s. The North American data sets generally show weak decreasing trends over the periods of record, although reporting from the six Alaskan sites ceased in the early-1990s. Statistically significant ($p < 0.05$) trends for Pb in air were found at two of the six Alaskan stations, and also at Alert (Table 5·2). The spread in estimates of the slope of these trends (–0.024/yr at 82.5°N to –0.894/yr at 62°N) suggests important subregional differences. None of the three stations in Greenland and Europe, for which post 1994-95 data are available, show significant trends for Pb in air (Table 5·2), possibly because any regional changes in atmospheric Pb concentration, for example following the introduction of unleaded gasoline, had already occurred by then.

Recent characterization of the stable Pb isotope composition of weekly aerosol samples at Alert using high-precision analytical techniques (Mercier *et al.*, 1999) has largely confirmed the earlier work of Barrie and colleagues, which was based on less precise measurements. Seasonal trends in aerosol Pb isotope ratios reflect air mass flows from different source regions. In the autumn, aerosols contained the lowest Pb concentrations and were relatively radiogenic (i.e., high ^{204}Pb isotope ratios). Back-trajectory calculations indicated that these originated from natural sources (dust, etc.) in the Canadian Arctic Archipelago and coastal West Greenland. During autumn/winter, aerosols had relatively high Pb concentrations and considerably lower isotope ratios than in summer, reflecting an anthropogenic input from western and northwestern Europe and Eurasia. An Asian industrial Pb contribution was also detected during late spring and early summer, which had not been detected in earlier studies. This may reflect the relatively poor precision of earlier analytical instrumentation and the recent industrialization and dramatically increased use of Pb in Asia, particularly China (Sangster *et al.*, 2000).

Time series of cadmium (Cd) concentrations in air/aerosols are available for only two stations (Zeppelin, Svalbard for 1996 to 2000; and Stórhöfdi, Iceland for 1995 to 2000). Higher and more variable concentrations were observed at the Icelandic site relative to Svalbard; however, at present the data available are insufficient for establishing temporal trends.

Gaseous elemental mercury (GEM) is measured on a routine basis at several Arctic sites, including Amderma in northern Russia (see Figure 4·1 and Section 4.4.1). Owing to its long atmospheric residence time (one to two years), GEM emissions are globally dispersed, with the result that concentrations at remote locations, such as the Arctic sites, mainly reflect variability in the global background concentration. This is supported by annual mean concentrations of GEM at the six Arctic sites of around 1.5 ng/m^3. For most stations, there are insufficient data to examine interannual trends in a statistically robust fashion. Consistent within-year trends associated with mercury depletion events in spring have, however, been observed at all sites (see Section 3.1.3). The longest instrumental record of GEM is for Alert (Figure 5·3), for which Schroeder (2001) reported no significant trends in

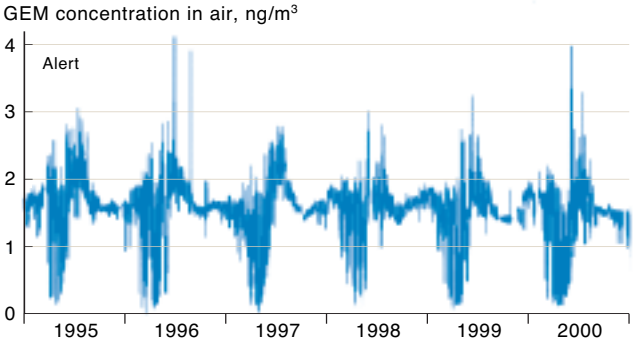


Figure 5·3. Concentrations of gaseous elemental mercury at Alert, Ellesmere Island (Meteorological Service of Canada).

Table 5·2. Metal concentrations in atmospheric media at AMAP monitoring stations (Berg *et al.*, 2002).

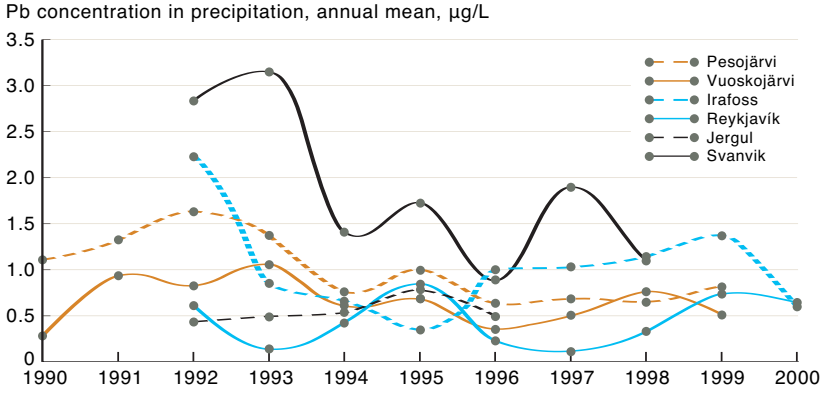
	Metal	Media	Start	End	Trend/ significance	+90%	Best estimate (coefficient of slope)	−90%
Canada								
Alert, Ellesmere Island	Hg	Air/aerosol	1992	2000	<0.05	−0.009	−0.029	−0.042
" " "	Pb	Air/aerosol	1980	1995	<0.05	−0.002	−0.024	−0.057
Greenland								
Station Nord	Pb	Air/aerosol	1994	2000	No trend			
Iceland								
Stórhöfdi	Pb	Air/aerosol	1995	2000	No trend			
Irafoss	Pb	Precipitation	1992	2000	No trend			
"	Cd	Precipitation	1992	2000	<0.10	−0.003	−0.018	−0.038
Reykjavík	Pb	Precipitation	1992	2000	No trend			
"	Cd	Precipitation	1992	2000	No trend			
Svalbard								
Zeppelin	Pb	Air/aerosol	1994	2000	No trend			
"	Hg	Air/aerosol	1994	2000	<0.10	−0.022	−0.082	−0.16
Norway								
Jergul	Pb	Precipitation	1992	1996	No trend			
"	Cd	Precipitation	1992	1996	<0.10	−0.000	−0.007	−0.018
Svanvik	Pb	Precipitation	1992	1998	No trend			
"	Cd	Precipitation	1992	1998	No trend			
Finland								
Pallas	Hg	Air/aerosol	1996	2000	No trend			
"	Hg	Precipitation	1996	2000	No trend			
Pesosjärvi	Pb	Precipitation	1990	1999	<0.10	−0.009	−0.083	−0.154
"	Cd	Precipitation	1990	1999	No trend			
Vuoskojärvi	Pb	Precipitation	1990	1999	No trend			
"	Cd	Precipitation	1990	1999	No trend			
Alaska								
Barrow	Hg	Air/aerosol	1998	2003	No trend			
Northwest Alaska Areas National Park	Pb	Air/aerosol	1986	1993	No trend			
Bering Land Bridge National Preserve	Pb	Air/aerosol	1987	1993	No trend			
Gates of the Arctic National Park	Pb	Air/aerosol	1987	1993	No trend			
Yukon Charlie National Preserve	Pb	Air/aerosol	1986	1993	No trend			
Wrangell St. Elias National Park	Pb	Air/aerosol	1987	1993	<0.05	−0.373	−0.894	−2.753
Katmai National Park	Pb	Air/aerosol	1987	1993	<0.05	−0.042	−0.240	−0.319

average annual concentrations between 1995 and 2000. In contrast, at the Zeppelin mountain station on Svalbard, a slight decreasing trend is observed between 1994 and 2000 (Table 5·2).

5.2. Precipitation

Data on heavy metal concentrations in precipitation (Table 5·2) are also stored at the AMAP Thematic Data Center at NILU, mostly collected as part of the United Nations Economic Commission for Europe (UN ECE) European Monitoring and Evaluation Programme (EMEP). Time series covering the 1990s are available for six subarctic stations; two in northern Norway, two in northern Finland, and two in Iceland. Figure 5·4 shows annual mean Pb concentrations in precipitation at these sites. Only one (Pesosjärvi, Finland) showed a decrease in concentration (slope = −0.083, $p < 0.10$). No trends were evident at the other sites.

Ukonmaanaho *et al.* (1998) report declines of 10 to 50% in Pb and Zn concentrations in bulk precipitation (wet + dry) at several remote Finnish sites between 1990 and 1996, with the greatest declines at southern sites near urban/industrial areas. Data for the four monitoring stations in northern Scandinavia (i.e., the two in northern Norway and the two in northern Finland) are



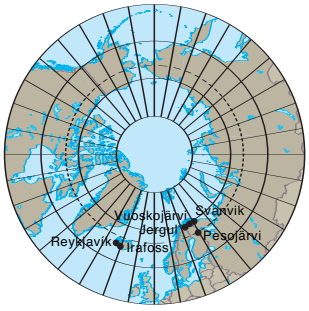


Figure 5·4. Annual mean concentrations of lead in precipitation at AMAP monitoring sites (AMAP Atmospheric Data Centre, NILU).

not consistent with these observations and indicate that Pb levels, if they decreased at all over the 1990s, did so to a much lesser extent in subarctic Scandinavia than southern Finland.

Cadmium concentrations in precipitation are relatively uniform. Apart from an anomalous 1994 value at Irafoss (IS0002), nearly all the concentration data are < 0.2 mg/L, and most about half that. Although statistical analyses reveal a slight declining trend at Jergul, Norway, and at Irafoss, Iceland if the anomalous 1994 value is excluded, these are marginal; at Jergul, the range in estimates of the trend includes a zero trend. As with Pb, the slightly higher concentrations at Svanvik may relate to emissions from the Kola smelters.

Only one station (Pallas, Finland) reported data for Hg in precipitation. Higher concentrations were reported in the first two years (1996 and 1997) than in subsequent years, however, the short time series and the apparent variability in the data preclude reliable conclusions on trends.

There is a need to continue to improve the standardization of sampling procedures and measurement techniques for metals in precipitation and air/aerosols at AMAP atmospheric monitoring stations. This should be accompanied by intercalibration and intercomparison exercises to document the degree of consistency in different measurement techniques. Although some exercises have been run in the past, generally in connection with the EMEP program, specific programs are required that focus on the particular problems associated with atmospheric monitoring in Arctic conditions.

5.3. Environmental archives

5.3.1. Ice cores

The largest single dataset concerning ice core-based reconstruction of Arctic metal deposition is from the Greenland Summit deep-drilling program (Boutron, 1995). Since the Industrial Revolution (i.e., after 1800 AD), ice cores have exhibited maximum increases in deposition fluxes of 12-fold for Pb, 8-fold for Cd, 5-fold for arsenic (As), and 4-fold for Cu, with maximum concentrations occurring during the 1960s and 1970s. However, because there was already a significant proto-industrial Pb contribution by 1800, the maximum Pb deposition rate in the 1970s resulted in a 200-fold increase in concentration above the natural background level (Figure 5·5). Since the 1970s, there has been a steady decline in the deposition of all four metals, with the rate of Pb deposition declining by 6.5-fold up to the early-1990s. More recent data are not available for this site. Trends in stable Pb isotope ratios indicate that the Greenland Summit site was influenced by North American and Eurasian air masses, with the North American Pb input declining rapidly after the introduction of unleaded gasoline. By the late-1980s, the Eurasian input accounted for 75% of the total Pb deposition.

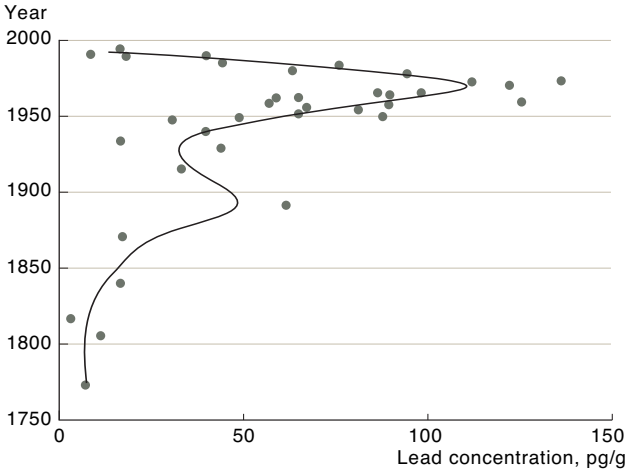


Figure 5·5. Lead concentrations in a Greenland ice core showing increases during the industrial period, but decreases since the early-1970s when unleaded gasoline was introduced in North America (Boutron, 1995).

Recent data reported by Boutron *et al.* (1998) for Hg in the Greenland Summit ice indicate that the post-1970 declines in the deposition of Pb and other metals may have been accompanied by a decline in Hg, with a 75% reduction from maxima in the late-1950s to the late 1980s. However, this should be treated with caution, as the data set comprised two separate time sequences (1949 to 1975 and 1967 to 1989) from different cores sampled with different equipment. In fact, taken alone, the more recent core indicates a steady increase of about 20% between 1967 and 1989. Further work of this type is a priority, and studies by at least two research groups began at Station Nord in spring 2002 (M. Goodsite, pers. comm., 2002).

It is important to note that the Greenland Summit site is 3200 m above sea level and reflects the net result of Hg transport and deposition to this high altitude sink. However, the ice core data cannot be linked with contemporary data on reactive gas phase or particulate Hg in the atmosphere, snowfall, or surface snowpack, because such measurements have not been made at this altitude. Thus, the relationship between photochemically-mediated atmospheric Hg deposition and Hg in the Greenland Summit ice is uncertain (see Section 3.1).

Recent analytical improvements have made it possible to determine the low level concentrations of platinum (Pt) group elements (PGEs; i.e., Pt, palladium (Pd), and rhodium (Rh)) in environmental media. The largest single use of PGEs is in automobile catalytic converters. Barbante *et al.* (2001) found that Pt, Pd, and Rh concentrations in Greenland snow in the late-1990s were 40- to 120-fold higher than in ancient ice (formed 5000 to 7000 years ago) indicating that almost all the recent deposition of PGEs in the Greenland Arctic is from industrial sources. There is no indication of deposition rates stabilizing or decreasing in the most recent snow

Box 5·1. Reconstructing temporal trends from cores

The use of media such as ice cores, sediments, and ombrotrophic peat bogs to reconstruct temporal trends in atmospheric metal deposition is widespread and long-standing. Recently, the issue of chemical diagenesis (alteration of metal profiles) in sediments and, to a lesser extent, peat bogs has become a topic of increased research interest and discussion. Diagenesis appears to be of greatest concern with respect to marine sediment cores, particularly for Hg profiles. The extent to which diagenesis of metal profiles occurs in lake sediment and peat bog cores is a matter of continuing debate. Caution should be exercised in interpreting metal deposition trends from these media until the effects and implications of diagenesis are better understood.

layers. The largest increase, 120-fold for Rh, was second only to that for Pb. The toxicity and bioaccumulation potential of these elements are largely unknown.

5.3.2. Peat bogs

Information on temporal trends in atmospheric Hg in the Arctic has recently become available from peat bog cores. Cores have been taken from a number of peat bogs across the Arctic and subarctic, including Bathurst Island, Canada, southwest Greenland, and the Faroe Islands, and were dated using ^{210}Pb and ^{14}C. At all sites, peat samples dating from the post-industrial period were enriched in Hg relative to pre-industrial samples (W. Shotyk, pers. comm., 2001). Similar trends have been found in cores from ombrogeneous bogs in central and northern Norway (Steinnes, 2001). Post-industrial Hg increases were between 7- and 17-fold higher than pre-industrial levels in Bathurst Island, and up to 14-fold higher in southwestern Greenland (W. Shotyk, pers. comm., 2001). These increases are several times those reported for lake sediments across the Arctic. A better comparison between lake sediment and peat core data will be possible when Hg flux data, corrected for changes in peat accumulation/sedimentation rates, are available for peat cores.

Background levels and natural variability of Hg in peat is an active area of research. Possible evidence of significant natural Hg inputs has been found in pre-industrial peat samples from Scotland, the Shetland Islands, and the Faroe Islands, possibly from volcanic emissions and mantle degassing in Iceland (Roos-Barraclough *et al.*, 2002).

5.3.3. Marine sediments

The crucial factor governing surface Hg enrichment in Arctic basin sediments is low sedimentation rates. Physical and biological sediment mixing tends to obscure stratigraphic patterns in these slowly accreting, low temporal resolution sediments. The sedimentation rate in deep-water marine cores is often so low that temporal resolution for the recent past (100 to 200 years) is not possible.

Total Hg was measured in seven sediment cores from the major basins of the Arctic Ocean in 1994 (Gobeil *et al.*, 1999). The Hg content of all seven decreased from the surface to a depth of 10 cm and strong similarities were observed between Hg and reactive iron (Fe) profiles, implying that a significant quantity of the total Hg deposited is recycled with Fe during redox changes.

Sediment cores from the Voronin Trough, Kara Sea, in 1965 had marked increases in Hg and As at the surface that may reflect industrial inputs from Siberian sources (Siegel *et al.*, 2001). Diagenesis was suggested as an additional contributor for the high surficial As but not for Hg.

Twenty marine sediment cores from near Greenland were analyzed for Hg and dated using ^{210}Pb (Asmund and Nielsen, 2000). Mercury concentrations and fluxes were generally higher in recent sediments with the apparent Hg flux having approximately doubled over the past 100 years. This was interpreted to reflect changes in anthropogenic inputs. In four cores, Mn concentrations also increased in the upper layers, indicating post-depositional diagenetic movement. However, in the remaining 16, higher Hg concentrations were not accompanied by higher Mn concentrations. The Hg flux to the sediment surface was generally proportional to the ^{210}Pb flux, suggesting that the Hg was mainly from atmospheric deposition. However, high variability indicates that other processes are also important.

5.3.4. Lake sediments

Diagenetic effects on Hg and other metal profiles in lake sediments are the subject of on-going research (e.g., Boyle, 2001). Lockhart *et al.* (2000c) showed that in three Canadian lakes, including one in the Yukon, Hg profiles closely matched well-documented fluvial releases of Hg, and therefore concluded that negligible post-depositional remobilization of Hg had occurred. In a varved (annually-laminated) lake on Devon Island, Lockhart *et al.* (2000b) reported about a two-fold enrichment of Hg in recently deposited sediment layers compared to pre-industrial sediments, similar in magnitude to that observed in other Arctic lakes. Although the varved nature of the sediment precluded physical mixing as a mechanism of diagenesis, geochemical and mineralogical studies provided evidence that physico-chemical conditions in the sediment (especially low sediment accumulation rates, low organic matter content, and lack of free sulfides for metal binding) had promoted the upward diffusion of many trace metals including Cd, Cu, Zn, nickel (Ni), and cobalt (Co) (Outridge *et al.*, 2005). For Hg, increased phytoplankton productivity related to the global warming trend over the past century may have increased the rate of Hg scavenging from the water column to the sediments, thereby increasing Hg concentrations in upper sediments (Figure 5·6). Mercury and diatom concentrations are significantly correlated for the past 400 years ($F = 89.3$, $r^2 = 0.84$, $p < 0.0001$, $df = 18$;

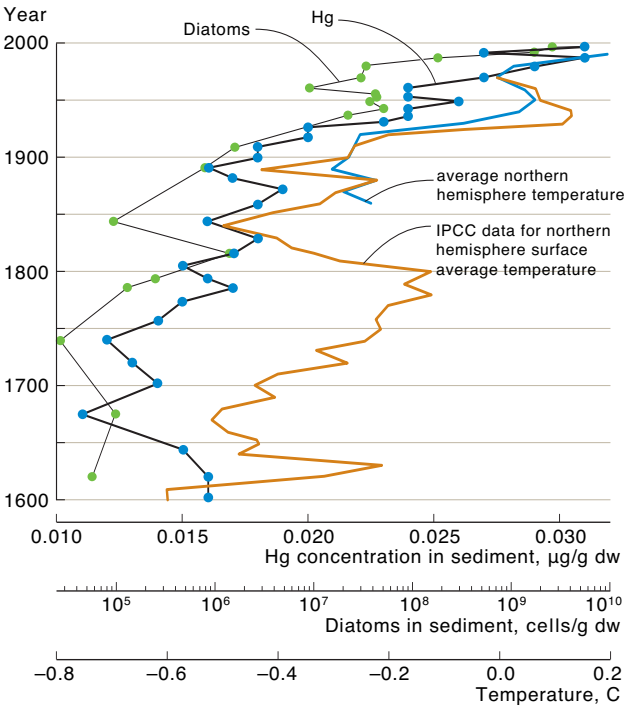


Figure 5·6. Correlation between mercury concentration, diatom concentration, and temperature change in a varved lake sediment from Devon Island, Canada (Outridge *et al.*, 2005).

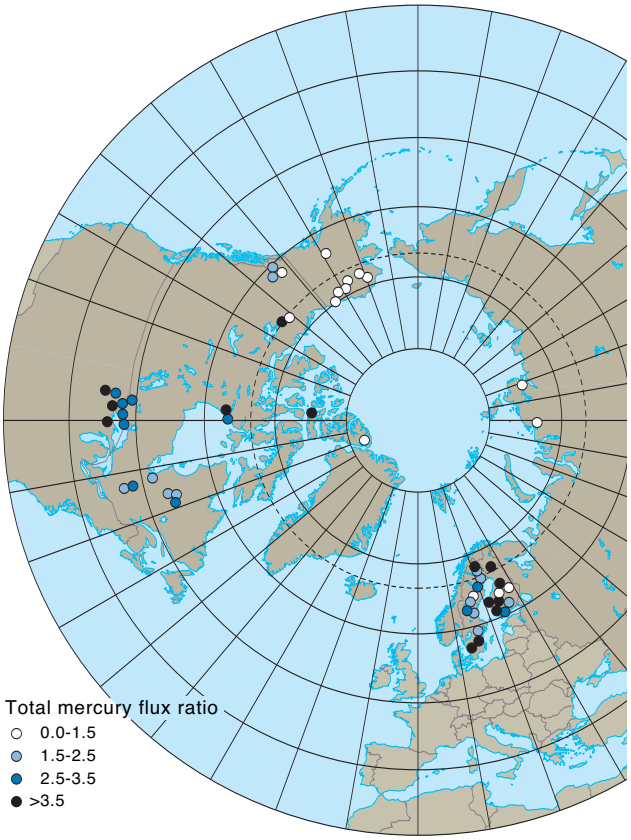


Figure 5·7. The ratio of post-industrial to pre-industrial flux of mercury to northern lake sediments (adapted from Bindler *et al.*, 2001a; Landers *et al.*, 1998).

Outridge *et al.*, 2005). More research is required on the role of geochemical diagenesis and on the influence of global warming in affecting the observed increases in Hg in sediment cores.

Lockhart *et al.* (1998) analyzed Hg concentrations in radiometrically-dated sediment cores from lakes across the Canadian Arctic and subarctic. Mercury fluxes were often higher in modern sediments than in pre-industrial sediments, which the authors took to reflect recent (i.e., anthropogenic) increases in atmospheric Hg deposition. Modern fluxes were generally low (0.3 to 2.1 μg/m^2/yr) in High Arctic lakes relative to those in lakes further south (1.5 to 18 μg/m^2/yr; Cheam *et al.*, 2001; Hermanson, 1998; Lockhart *et al.*, 1998, 2000c; Stern *et al.*, 2001b; Swain *et al.*, 1992). Recent declines in Hg concentration in southern lakes (e.g., Lake Winnipeg) were not observed in Arctic lakes. This suggests that Hg deposition rates had not declined recently in the Arctic, in contrast to the situation at lower latitudes. Modern deposition rates of Hg, Cd, Pb, and several other metals in lakes on Ellesmere Island and the North Alaskan coastal plain are similar to those of pre-industrial times (Allen-Gil *et al.*, 1997a; Landers *et al.*, 1995; Lockhart *et al.*, 1998), suggesting an absence of anthropogenic Hg in these regions.

In a series of lakes in the ice-free margin along the West Greenland coast, recent Hg concentrations in sediments were on average three times higher than in pre-industrial sediments (Bindler *et al.*, 2001a). Higher apparent Hg deposition rates were measured in lakes closer to the ice margin than in lakes closer to the coast, suggesting a localized atmospheric deposition phenomenon close to permanent ice.

Bindler *et al.* (2001b) analyzed Hg in sediment cores from a 1100 km south-to-north transect of Swedish lakes in 1998. Eight of the nine lake sediments showed subsurface Hg maxima, with decreasing concentrations toward the surface. Loss-on-ignition varied from 25 to 60%, suggesting ample organic matter to bind Hg. These results differ from those of a 1979 Swedish study in which Hg maxima were found in the uppermost sediment layers, but are consistent with observed reductions in atmospheric Hg deposition reported by Iverfeldt *et al.* (1995). In contrast, no recent decrease in Hg concentration was observed in the 100-lake survey carried out in 2000 as part of the National Swedish Monitoring Programme (SLU, 2003). Possible explanations for the lack of agreement between Hg deposition and the sediment record in the 100-lake survey include physical and biological mixing processes; low temporal resolution of surficial slices; continuing Hg inputs via runoff from catchments containing accumulated Hg as a result of high deposition during the 1900s; and diagenetic changes in sedimentary Hg profiles.

Sufficient information on lake sediment Hg profiles has now been accumulated to compare anthropogenic Hg inputs at continental scales. Sediment flux ratios (FR: ratio of the last 10 to 40 years of Hg deposition and average pre-industrial Hg deposition) were derived from lake sediment profiles throughout the Arctic and subarctic (Canada, Finland, Sweden, Russia, United States) to estimate temporal Hg fluxes to watersheds (Landers *et al.*, 1998). Flux ratios varied widely (Figure 5·7). The highest values occurred in southern regions associated with strong regional sources of atmospheric Hg emission (eastern North America, Central Europe). There were large differences in the background Hg flux depending on watershed geology, with some lakes in Alaska exhibiting relatively high concentrations of sedimentary Hg without evidence of further anthropogenic enrichment. In Arctic and subarctic North America, there is a tendency for FRs to increase to the southeast toward the areas of highest emissions, consistent with the gradient in decreasing lake water pH in Canadian surveys (R. Garrett, unpubl. data, 2001).

In another study of metals in surface versus pre-industrial sediments, Rognerud *et al.* (1998) compared 66 lakes across northern Norway and along the Siberian coast. Significantly higher Hg and Pb concentrations occurred in surface sediments, with the greatest increases in western and lower latitude areas close to regions of major industrial/urban emissions. The most distant areas sampled, including Wrangell Island off the northeast Siberian coast, had recent Hg increases taking concentrations to twice those of pre-industrial sediments but with no associated increase in the concentrations of Cu, Ni, Zn, or selenium (Se).

Latitudinal differences in inputs of recent anthropogenic and natural Pb were also observed in North American lake sediment cores (Outridge *et al.*, 2002a). In the recent sediments of four High Arctic Canadian lakes, Pb concentrations and stable Pb isotope ratios were similar to those of pre-industrial sediments; anthropogenic Pb accounting for no more than 19% of the acid-leachable Pb in recent sediments. In contrast, for two lakes in the Hudson Bay lowlands, the significance of anthropogenic sources increased during the nine-

teenth century; accounting for 70 to 90% of the acid-leachable Pb by the 1980s or 1990s. Trends in Pb isotope ratios, together with meteorological data, indicate that Eurasia contributed most of the anthropogenic Pb in northwestern Hudson Bay, with Canadian/U.S. sources affecting southeastern Hudson Bay. Similar latitudinal differences were reported by Cheam *et al.* (2001), with cores from four High Arctic lakes showing no recent increase in Pb concentration, while a more southerly lake in northern Quebec (at 54° N) showed evidence of recent anthropogenic Pb enrichment. Reduced atmospheric Pb deposition at higher latitudes may be associated with generally lower precipitation rates. However, over Hudson Bay, the polar front (an area of meteorological disturbance) may play a role in increasing both precipitation and Pb deposition (Outridge *et al.*, 2002a).

Similar to the Hudson Bay lakes, Pb concentrations and Pb isotope ratios in lake sediments from southwestern Greenland showed that significant increases in Pb concentration had occurred during the eighteenth and nineteenth centuries, with maxima around 1970 (Bindler *et al.*, 2001c). Concentrations in recent sediments were about 2.5-fold higher than background, with slightly greater increases toward the coast where precipitation was also higher. The $^{206}Pb:^{207}Pb$ ratio (1.14 to 1.15) in recent sediments was interpreted as reflecting a recent increase in the influence of western European sources on Pb deposition to Greenland, in contrast to the previous dominance of Russian sources (with a typical $^{206}Pb:^{207}Pb$ ratio of 1.15 to 1.16).

A lake sediment core from the eastern Taymir Peninsula in central Siberia showed no recent Pb enrichment (Allen-Gil *et al.*, 2003), suggesting that emissions from industrialized areas, including automobile emissions, were not being deposited in large quantities in this remote region. The Norilsk smelting complex 300 km to the southwest does not appear to be a significant regional source of Pb, which is consistent with the findings of a lake sediment study by Blais *et al.* (1999). Both studies indicate that the main deposition to the north of Norilsk is confined to an area within 80 to 100 km of the smelters.

5.4. Biological compartments

Biological availability of heavy metals deposited to land, water, ice, and snow is governed by the location, concentration, and chemical form of the metal. With the possible exception of mosses, which receive their metal burden directly from the atmosphere, biota in the terrestrial, freshwater, and marine environments show trends that are not always consistent with those observed in abiotic media such as ice, peat, and sediments.

5.4.1. Terrestrial biota

In the first AMAP assessment, few data were available with which to investigate temporal trends in terrestrial biota. This assessment contains information on trends in Norwegian mosses using an extended data set, and new moss data for Sweden and Russia. Time series for metals in Swedish reindeer (*Rangifer tarandus*) are also extended, and there are new short-term trend data for peregrine falcon (*Falco peregrinus*) and moose (*Alces alces*).

5.4.1.1. Mosses and lichens

With the exception of metals in mosses from Norway and Sweden, there are no long-term monitoring programs in place to monitor metals in Arctic terrestrial plants. Mosses, as well as lichens, are considered excellent biomonitors of atmospheric metal deposition because they lack root systems and almost exclusively extract metals from air and precipitation rather than the surface on which they grow (Steinnes, 1995). Regular monitoring of heavy metals in feather mosses has been conducted at five-year intervals in Scandinavia, since 1970 in Sweden and 1975 in Norway, and the programme was expanded to include Finland and the Kola Peninsula in 1995.

The substantial decrease in Pb concentration in Scandinavian mosses since the early-1980s is consistent with bans on leaded gasoline (AMAP, 1998), as are more recent data for the Faroe Islands. A comparison of results from the 1970, 1985, and 1995 Swedish moss surveys (Rühling and Tyler, 2001), shows that decreases in Pb and Cd were greater in southern than northern Sweden (Steinnes *et al.*, 2001). Significant reductions in Pb and Zn concentrations were observed in forest mosses between the late-1980s and mid-1990s at remote sites in Finland (Ukonmaanaho *et al.*, 1998), matching reductions in the bulk precipitation of Pb and Zn. However, concentrations in humus and leaf litter in Finnish forest catchments did not change significantly over that period (Ukonmaanaho *et al.*, 1998).

The moss survey in 2000 showed that Pb and Cd levels in Sweden continued to decrease, even in northern areas (Figures 4·21 and 4·35), with Pb levels in northern Sweden now generally <3 mg/kg. Similarly, the Pb levels in northern Norway are currently around 2 mg/kg dw; a further decrease from the levels observed in 1995 (Steinnes *et al.*, 2001, see Section 4.5.3.2). The 1995 survey indicated decreases relative to earlier surveys in V, Zn, As, antimony (Sb), tin (Sn), molybdenum (Mo), silver (Ag), thallium (Tl), and bismuth (Bi), all of which have at least some anthropogenic component (Steinnes *et al.*, 2001). Levels of Fe, which was previously thought to originate only from soil dust, also decreased in Norway and Sweden over the 25-year period. This suggests that there may also be anthropogenic sources of Fe, probably associated with coal combustion and the metallurgical industry (Rühling and Tyler, 2001; Steinnes *et al.*, 2001).

The only temporal data available for lichens are from Nuuk, Greenland, where samples of *Cetraria nivalis* were collected in 1994 and 1999 (F. Riget, unpubl. data, 2001). There was no significant change in concentrations of Hg or Cd between these dates, while Se concentrations decreased significantly (see Appendix Table 5·2).

5.4.1.2. Birds

The peregrine falcon is the only Arctic terrestrial bird for which time series data on heavy metals are available. Metals in American (*Falco peregrinus anatum*) and Arctic (*Falco peregrinus tundrius*) peregrine falcon eggs from Alaska were measured from 1988 to 1995 (Ambrose *et al.*, 2000). Copper, Fe, Mg, Hg, and Zn were detected in most samples and consistently measured. Owing to low sample sizes in some years, data were

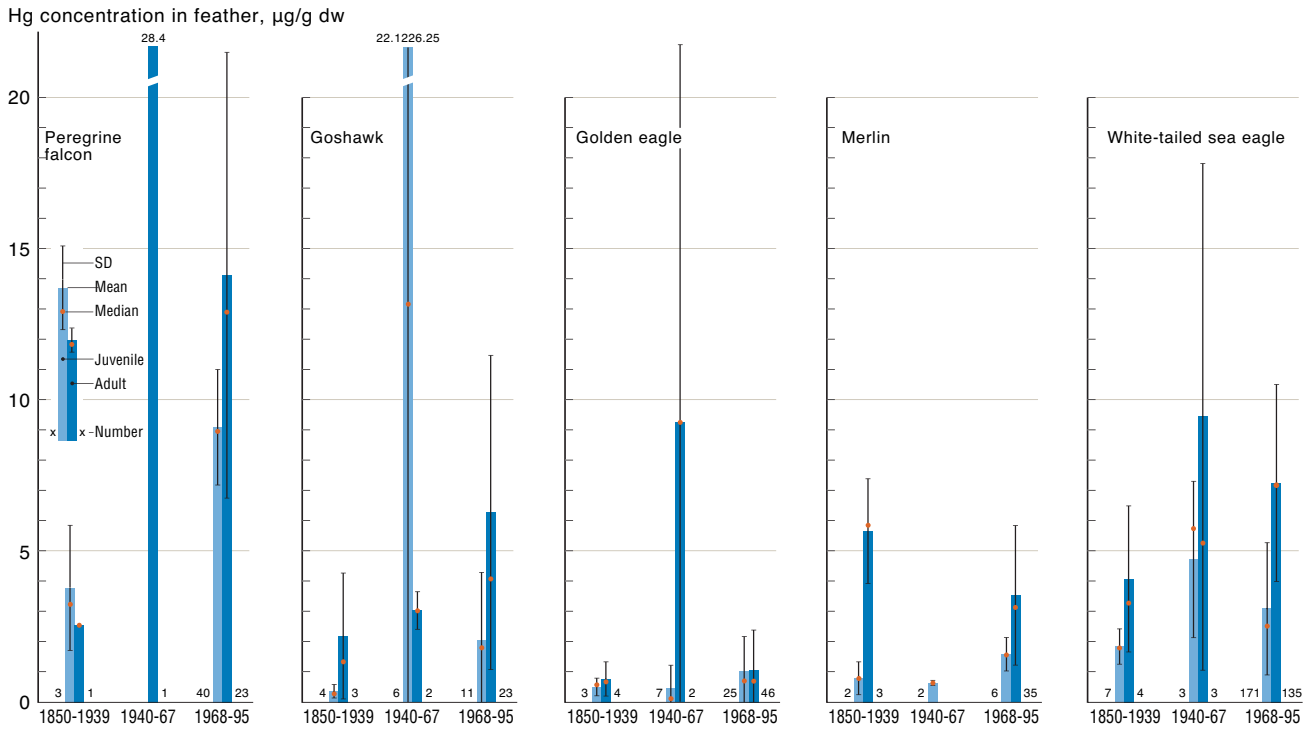


Figure 5·8. Mercury in archived feathers of Norwegian birds of prey from north of 62°N, 1850 to 1995 (Nygård, 1997).

grouped into year groups. Copper, Fe, and Zn were significantly lower in 1991 to 1995 compared with 1988 to 1990 (Annex Table A5). Mercury and Mg were not significantly different between year groups, although Hg may be increasing, at least in American peregrine falcons (see Annex Table A5). Both Arctic and American peregrine falcons in Alaska are highly migratory and overwinter from the southern United States to Argentina (Ambrose and Riddle, 1988; Britten, 1998; R.E. Ambrose and T. Swem, unpubl. data, 2000). Although the migration routes and overwintering areas of the two subspecies overlap, they are separated on their Alaskan breeding range, with American peregrine falcons nesting south of the Brooks Range and Arctic peregrine falcons nesting north. Both subspecies are top predators consuming mainly migratory avian prey, although dietary studies have shown that the boreal-dwelling American peregrine falcon feeds more on waterfowl and less on shorebirds than the tundra-dwelling Arctic peregrine falcon (Cade *et al.*, 1968; White and Cade, 1971). Females arrive at the Alaskan breeding grounds up to a month before laying their first egg (Cade, 1960; R.E. Ambrose and T. Swem, unpubl. data, 2000), but as both predator and prey are migratory, contaminant trends exhibited by peregrine falcons may not be exclusively indicative of the Arctic environment.

Nygård (1997) measured Hg levels in archived feathers of Norwegian birds of prey including golden eagle (*Aquila chrysaetos*), goshawk (*Accipiter gentilis*), merlin (*Falco columbarius*), and peregrine falcon (Figure 5·8), which all feed in the terrestrial environment. This study used the innermost primary feather, which is shed in spring and reflects Hg accumulated primarily during the winter. Concentrations increased in the 1950s to 1960s, and then declined following the introduction of restrictions in Norway on the use of Hg in seed dressings (by 1968), and the phasing-out of Hg discharges from pulp mills and chlor-alkali plants during the late-1960s to early-1970s. In the 1950s, increases in Hg concentration in adult peregrine falcon were greater than in the juveniles, probably because adults took up pollutants on their overwintering grounds in southwestern Europe, while concentrations in juveniles more closely reflected Norwegian conditions. Goshawk showed the opposite, probably owing to differences in prey fed to the young (easily caught seed-eating birds such as pigeons, crows, and sparrows from nearby fields) relative to the cleaner prey (grouse, hare (*Lepus* spp.) and red squirrel (*Tamiasciurus* spp.)) consumed by adults over winter (Nygård, 1997). Modern levels seem to have stabilized at higher levels than the pre-1950s background level.

5.4.1.3. Terrestrial mammals

There are no long-term data sets for metals in Arctic terrestrial mammals except for data on Hg in human teeth and hair, which span several centuries. Teeth and hair are reliable indicators of Hg body burden in mammals (Eide and Wesenberg, 1993; Eide *et al.*, 1995). Geometric mean Hg concentrations in human deciduous teeth (without fillings) in Norway in the 1970s were 13-fold higher than in similar teeth from the twelfth century (Eide *et al.*, 1993), although recent studies suggest that Hg concentrations have declined substantially over the past 20 years (Tvinnereim *et al.*, 1997). The hair of modern Greenlanders contains significantly (2.5-fold) more Hg than samples from fifteenth century Inuit mummies (Hansen *et al.*, 1991). Similarly, Wheatley and Wheatley (1988) report that modern Hg levels in human hair from the Canadian Arctic were several times higher than in pre-industrial samples.

There are several short-term data sets for Arctic terrestrial mammals, particularly moose and reindeer/caribou. The reindeer of northern Europe and Asia and the

caribou of North America are considered to be a single species, *Rangifer tarandus*, although there are a number of well-marked geographical subspecies. Long-lived ungulates such as moose and reindeer/caribou can accumulate significant levels of some heavy metals, such as Cd (Frøslie *et al.*, 1986; Scanlon *et al.*, 1988), especially in their livers and kidneys. Moose, whose diet mainly comprises twigs and leaves of trees and shrubs (Cederlund *et al.*, 1980), was the species chosen to monitor environmental contaminants in Swedish forest areas (Odsjö *et al.*, 2001). At the most northerly monitoring site (in Norbotten County), no significant changes in renal or hepatic chromium (Cr), hepatic Cu, or renal Cd, occurred in moose between 1996 and 2000. In Canada, Yukon moose sampled annually between 1993 and 1998 displayed no significant changes in renal Hg, Cd, or Pb (Table 5·1 and Appendix Table 5·1). However, an indication of some decline in Hg levels was observed in two areas (Dawson and Ross River) (P. Roach, unpubl. data, 2001).

Reindeer/caribou have a summer/autumn diet that includes grasses, sedges, twigs, leaves, and mushrooms, and a winter diet that consists mainly of lichens (Kelsall, 1968; Parker, 1978), which are known to accumulate contaminants from the air. As part of the Swedish Monitoring Programme, metals have been analyzed in muscle and liver of (3-year old, male) reindeer collected annually from northern Sweden (Abisko) since 1983 (Odsjö, 2003; A. Bignert, unpubl. data, 2001). Cadmium concentrations in liver showed a significant increase between 1983 and 1999 (Figure 5·9) while muscle concentrations showed no significant change (Appendix Table 5·1). Over the same period, Pb concentrations decreased significantly in liver but showed no change in muscle. Mercury concentrations in liver and muscle showed no change between 1983 and 2000. Although this time series is one of the longest for metals in mammals, it failed to meet minimum requirements for statistical power to detect trends (corresponding to an annual change of 5%) for all element-tissue combinations. Based on the variance inherent in the data, the number of years required to detect an annual change of 5% was estimated at 11 and 28 for Cd, 20 and 16 for Hg, and 21 and 31 for Pb in liver and muscle, respectively.

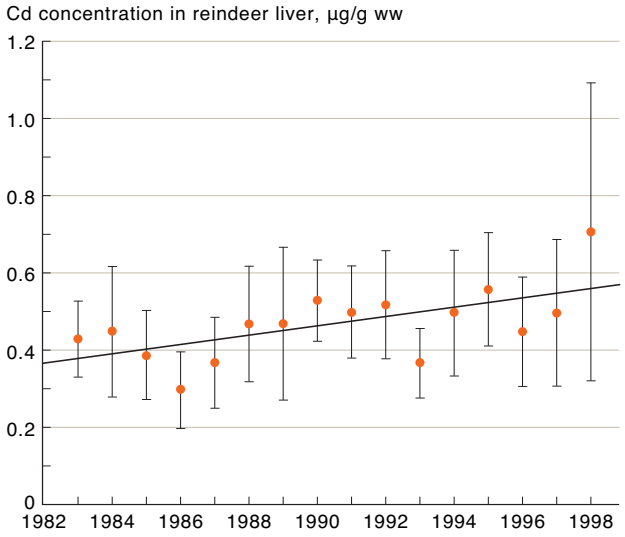


Figure 5·9. Cadmium concentrations in liver of reindeer collected annually from northwestern Sweden (Norbotten County) between 1983 and 1998 (Odsjö, 2003; A. Bignert, unpubl. data, 2001).

Liver and kidney of barren-ground caribou (*Rangifer tarandus groenlandicus*) from several herds have been sampled across the Canadian Arctic over the past ten years (see Annex Table A6). Cadmium concentrations in kidney from the Lake Harbour Herd were about 30 mg/kg dw for both the 1992 collection and the 1999 collection (Elkin, 2000), although pairs of collections covering a similar time period showed substantial decreases in the Bluenose and Tay Herds, and a substantial increase in the Beverly Herd (Macdonald *et al.*, 2002). However, comparisons between single years spread over an interval of several years may be misleading, as interannual variability appears high and age and sex can be important confounding factors. Yukon caribou in the Porcupine Herd, sampled annually between 1991 and 1999 (P. Roach, unpubl. data, 2001), showed no statistically significant changes in renal Cd (Appendix Table 5·1). Similarly, no significant change was found in renal Hg or Pb in the Porcupine Herd (Appendix Table 5·1; Macdonald *et al.*, 2002) despite differences between pairs of collections in the Bluenose, Tay, Beverly, and Lake Harbour Herds (Macdonald *et al.*, 2002).

Cadmium concentrations in liver of reindeer from Isortoq, Greenland, showed no significant differences between samples collected in 1994 and 1999, whereas Hg and Se concentrations were significantly lower in 1999 than 1994 (F. Riget, unpubl. data, 2001; Appendix Table 5·2). Lower Hg levels also occurred in another West Greenland herd between 1995-7 and 1999 (F. Riget, unpubl. data, 2001).

5.4.1.4. Assessment of temporal trends in terrestrial biota

The long-term data sets for human teeth and hair show that Hg concentrations increased between the pre-industrial era and modern times (1970s) although there is evidence to suggest that Hg concentrations in Norwegians have decreased substantially since then. Analyses of the ten most reliable short-term data sets for Hg, Cd, and Pb in terrestrial mammals (Table 5·1; ICES, 2002) showed a significant increase in Cd and a significant decrease in Pb in Swedish reindeer. The other terrestrial mammal data, including all the Hg data, showed no significant short-term trends.

Other short-term data sets, which were not analyzed by the AMAP workshop on temporal trends (Appendix Table 5·2) owing to their limited number of data records, suggest that Hg concentrations in Alaskan peregrine falcon eggs may be increasing, and that Hg and Se in Greenland reindeer and Se in Greenland lichen may be decreasing. However, more years of data are required to confirm these trends.

At present, the longest and most complete data sets available for evaluating temporal trends in metal concentrations in Arctic terrestrial biota are those for moss from Norway and Sweden, and for reindeer from Abisko, Sweden. Based on an analysis of the statistical power of the existing terrestrial mammal data series, at least 10 to 20 more years of (annual) monitoring may be required in many cases before reliable conclusions can be drawn about temporal trends.

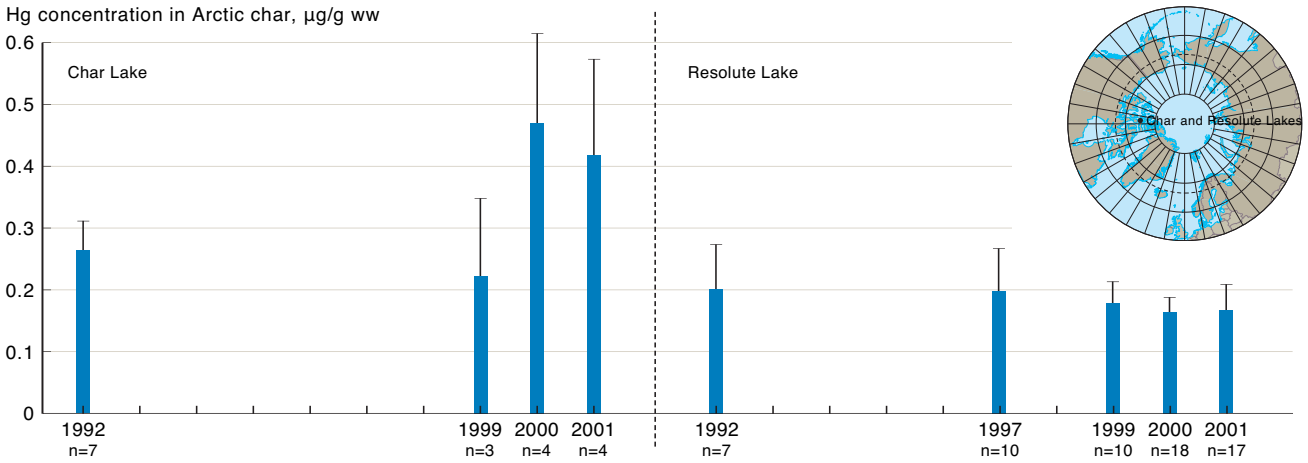


Figure 5·10. Mean mercury concentrations in Arctic char from Char and Resolute Lakes (Muir *et al.*, 2001; D. Muir, pers. comm., 2001).

5.4.2. Freshwater biota

Freshwater organisms are exposed to atmospheric contaminants deposited onto snow, deposited directly onto lake surfaces, or deposited onto catchment vegetation and soils. Anthropogenic contaminants also enter lakes and rivers via direct discharges and in catchment runoff such as from agricultural areas. As for terrestrial biota, few data were available for identifying trends in freshwater biota in the first AMAP assessment. Short-term data sets are now available for a variety of fish species, in addition to the Swedish northern pike (*Esox lucius*) data set used in the first AMAP assessment. There are still no short- or long-term data sets available for freshwater invertebrates or plants.

5.4.2.1. Freshwater fish

The value of extended annual sampling for determining temporal trends in contaminants is well illustrated by Swedish fish studies. In the first AMAP assessment, the only time series for metals in freshwater fish was for Hg in muscle of pike collected between 1968 and 1996 from Lake Storvindeln, a forest lake in northern Sweden. Additional data points for that time series are now available. Arctic char (*Salvelinus alpinus*) have also been collected in Sweden since 1981 in Abiskojaure (Ábeskojávre), a lake 200 km north of the Arctic Circle. For pike and char, annual samples are selected for consistency in sex, age, size, and sampling season (A. Bignert pers. com., 2001). Each sample comprises 10 to 25 individual fish, allowing estimates of within-year variation. Some limited data for landlocked char, lake trout (*Salvelinus namaycush*), and burbot (*Lota lota*) are also available for other countries.

No significant trends were observed for (length-adjusted) muscle Hg concentrations, or for hepatic Cd, Cu, or Pb concentrations in pike from Lake Storvindeln between 1968 and 1999; nor in Arctic char from Abiskojaure between 1981 and 2000, although non-length-adjusted Hg in char did decline significantly (Appendix Table 5·1).

Power analyses show that the longevity of the two fish data sets from Sweden are sufficient to detect annual changes in Hg, Pb, or Cd of 5% (ICES, 2002). The failure of the fish time series to reflect the decreasing Pb emissions that have resulted in declining Pb levels in Swedish moss (Section 5.4.1.1) and some Baltic marine biota (A. Bignert pers. com., 2001) is worth noting. A possible explanation is that, because Arctic Sweden appears to have been less exposed to Pb contamination than more southerly parts of the country, small decreases are not easily seen against the background of normal interannual variability.

Landlocked char have been collected over a number of years from lakes near Qausuittuq (Resolute) in the Canadian High Arctic. There was no significant change in Hg concentrations in Arctic char from Resolute Lake between 1992 and 2001 (Appendix Table 5·2). However, mean concentrations in char from 2000/2001 were about 20% lower than in 1992 and 1997 (Figure 5·10). In nearby Char Lake, Hg levels were higher in 2000 than in 1999 but the fish collected in 2000 were larger and may have been feeding at a higher trophic level. Concentrations of Hg and other metallic elements in char from three other lakes in the region (North, Sapphire and Boomerang) were similar to those in Char and Resolute Lakes (Muir *et al.*, 2001b). Mercury was significantly correlated with $\delta^{15}N$ in char from Resolute Lake in 1997 and 1999, suggesting that biomagnification of Hg was occurring within the population due to the presence of piscivorous char (Muir *et al.*, 2001b). These results demonstrate the importance of ensuring that time series studies are based on a well designed and structured sampling scheme that takes into account factors such as size and food web structure.

Mercury data for Arctic char muscle are also available for 1994, 1995, and 1999 from Isortoq, on the southwest tip of Greenland (Appendix Table 5·2). Sex was not a significant factor and the length-normalized data show no significant trends over time (F. Riget, pers. comm., 2001).

Burbot is a predatory, bottom-feeding species. Burbot were collected in various years between 1985 and 2000 from Fort Good Hope (Radei Ko) in the Northwest Territories, Canada. Mean Hg concentrations in muscle from male burbot increased by 36% between 1985 and 2000 (Figure 5·11). No significant correlation between length and Hg concentration was found in muscle or liver for either male or female fish. Arsenic levels in male burbot liver in 1988 were significantly higher than in 1993, 1999, and 2000.

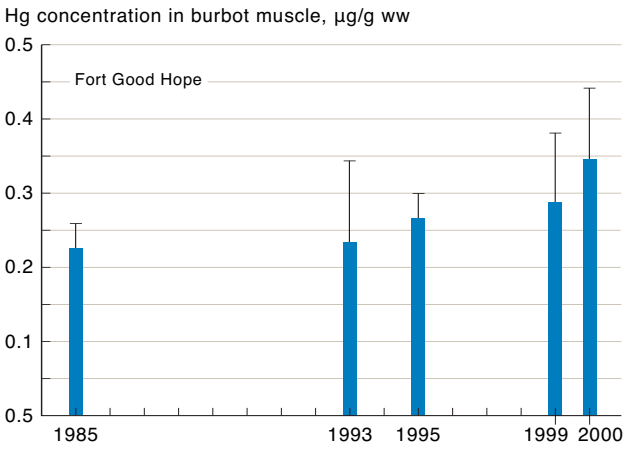


Figure 5·11. Mean mercury concentrations in male burbot from Fort Good Hope (Stern *et al.*, 2001a).

Lake trout and burbot were also collected from Lake Laberge in the Canadian Yukon at intervals between 1993 and 1999 (Appendix Table 5·2). There was a 30% decline in length-adjusted Hg concentrations in muscle of lake trout between 1993 and 1996 and no change between 1996 and 1998 (Stern *et al.*, 2001b). Similarly, no change in Hg concentration was observed in liver of burbot from Lake Laberge between 1996 and 1999. No significant change in Hg levels occurred in muscle of lake trout from another Yukon lake, Quiet Lake, between 1992 and 1999 (Stern *et al.*, 2001b).

5.4.2.2. Assessment of temporal trends in freshwater biota

In the freshwater environment, biological time-series data sets were available for fish only, with the longest data sets for consistently monitored species being those for landlocked char and northern pike from northern Swedish lakes. Apart from the Swedish data sets, the data sets were all of relatively short duration. The only significant trend observed was an increase in Hg concentration in burbot from Fort Good Hope on the Mackenzie River in Canada. Other data sets showed no clear change in Hg concentration or the concentration of any other element analyzed. Based on the Swedish extended data sets for pike and Arctic char, 10 to 20 years of monitoring data are required to provide sufficient data to detect average trends of 5% per year.

That Hg concentrations in an Arctic lake sediment in Canada were strongly correlated with diatom concentrations (see Section 5.3.4) raises the possibility that climate warming may be increasing the Hg trapping efficiency of phytoplankton populations in Arctic lakes, with attendant consequences for Hg biomagnification in lake food webs. The possible role of climate warming in driving Hg increases in freshwater food webs is an issue for future research.

5.4.3. Marine biota

The first AMAP assessment concluded that Hg concentrations in seals and toothed whales from Canada and Greenland were increasing, but found moderate to no increases in seabirds. Data are now also available for marine invertebrates and fish. Some of these data have been collected annually or at specific intervals, while other data sets allow comparison between specific time periods spread over several decades. The effects of age, sex, season, diet, and possible changes in sampling and analytical protocols must be considered.

5.4.3.1. Marine invertebrates

Time-series data for blue mussel (*Mytilus edulis*) ranging from a few years to a long-term data set spanning several millennia are now available. Outridge *et al.* (2000) reported significant north-to-south variations in the relative inputs of anthropogenic and natural Hg and Pb as reflected in bivalve species from sites across the Canadian Arctic. At two high-latitude sites, Expedition Fjord (Axel Heiberg Island) and Resolute Bay (Cornwallis Island), long-term sequences of bivalve shells collected from Holocene (9000 to 1000 BP) beach ridges contained similar concentrations of Hg, Pb, and Cd to modern (1980s to 1990s) collections, suggesting an absence of industrial inputs to those areas. However, at a low-latitude site in southeast Hudson Bay, concentrations of Hg and Pb were significantly higher in modern (1970s) blue mussel shells, and the temporal trend in stable Pb isotopes was consistent with the recent introduction of Eurasian or Canadian anthropogenic Pb to the area. At all sites, the concentrations of Cd, Zn, and other trace metals were similar or lower in modern samples than in pre-industrial samples.

More recent time-series data for blue mussels are available from coastal sites in Alaska, Greenland, Iceland, and Norway. As part of the Mussel Watch Project, blue mussels were collected annually between 1986 and 1999 from two stations (Siwash Bay and Mineral Creek Flats) in Prince William Sound on the southern coast of Alaska (Cantillo *et al.*, 1999). Statistical analyses of data from 1986 to 1996 showed no significant trends in Ni, Cu, Zn, As, Se, Ag, Cd, Hg, and Pb, although the annual medians did suggest a decrease in As in mussels at Siwash Bay and an increase in Hg at Mineral Creek Flats (Cantillo *et al.*, 1999). Data from 1997 and 1999 appear to continue these trends (A. Robertson, unpubl. data, 2001).

Blue mussels collected annually from coastal sites in northern Norway from 1992/94 to 1999 showed no significant trends at most sites for Cd, Cu, Hg, Pb, or Zn, with the exception that Hg declined significantly at two sites and showed a marginal decline ($p = 0.092$) at another (Table 5·1). Consistent although non-significant declines in Zn also occurred at several sites (ICES, 2002).

Blue mussels have been monitored for Hg, Cd, and Cr at eight coastal sites around Iceland since 1990. Concentrations of Hg and Cd were relatively constant at most locations except at two sites where, respectively, an increase and a decrease in Cd were recorded. Because the trends are in opposite directions, it is likely that they are due to random error (Table 5·1, Appendix Table 5·1). Cadmium concentrations in soft tissue of blue mussels collected annually from Nuuk (1987 to 1990) and Uummannaq (1988 to 1995) on the west coast of Greenland showed no significant trends (Riget and Dietz, 2000). Recent data from Qeqertarsuaq (Appendix Table

5·2) showed no clear trend in Cd in blue mussels from 1994 to 1999, as concentrations increased in the 5 to 7-cm and 8 to 9-cm length classes, and decreased in the 4 to 5-cm and 7 to 8-cm length classes. Mercury decreased in all but the largest specimens, significantly in the 7 to 8-cm length class, and Se decreased in all samples, significantly in the 4 to 7-cm length class.

5.4.3.2. **Marine fish**

There were few temporal trend data for marine fish in the first AMAP assessment. Although new data sets are available here, there are still no convincing, long-term data sets for Arctic marine fish.

Cadmium concentrations in liver of shorthorn sculpin (*Myoxocephalus scorpius*) collected annually between 1980 and 1993 from Uummannaq on the west coast of Greenland appeared to decrease although the trend was not significant (Riget and Dietz, 2000). However, significantly higher Hg concentrations occurred in liver of shorthorn sculpin from Avanersuaq and Nanortalik in the mid-1990s than the mid-1980s (Riget and Dietz, 2000). The opposite was the case for liver of Arctic cod (*Boreogadus saida*) with lower Hg concentrations in the mid-1990s than the mid-1980s (Riget and Dietz, 2000). More recent data for sculpin collected in 1994, 1999, and 2000 from Qeqertarsuaq, Greenland, show no clear trends for Cd, Hg, or Se (Appendix Table 5·2). Mercury concentrations in herring (*Clupea harengus*) from Landsort, and cod from southeast Gotland, both in the Baltic Sea south of the Arctic area, increased over the period 1979/80 to 1999 (Bignert pers com., 2001).

There has been no significant change in hepatic Cd or Hg in the muscle of Atlantic cod (*Gadus morhua*) at sites off northern Norway since 1994 (Table 5·1). A significant decrease in Hg and Cu concentrations was seen at one site in data unadjusted for fish length, but not in length-adjusted data (ICES, 2002).

Data are also available for cod and dab (*Limanda limanda*) sampled at a number of different sites around Iceland during the 1990s. For the purposes of trend assessment, the data were grouped into three geographical areas (Appendix Table 5·1). Both species showed significant declines in muscle Hg off northwest Iceland, with concentrations generally constant elsewhere. Dab also showed a significant increase in Zn. Cadmium levels remained unchanged.

Power estimates for the Norwegian and Icelandic invertebrate and fish data sets indicated that 20 to 30 years of data would be required in most cases to provide an adequate level of statistical power for detecting trends corresponding to an annual 5% change in the concentrations of Hg, Cd, or Pb (ICES, 2002).

5.4.3.3. **Seabirds**

Retrospective analyses of museum and archived samples have resulted in several new time-series data sets for Arctic seabirds. The first AMAP assessment discussed a study by Appelquist *et al.* (1985) which showed that Hg concentrations in feathers of black guillemot (*Cepphus grylle*) and Brünnich's guillemot (thick-billed murre; *Uria lomvia*) from Greenland were almost constant over the past 100 years, although a slight increase was evident in black guillemot. The outcome of a similar study on Hg in feathers of black guillemot from Alaska is now available.

The black guillemot is a High Arctic seabird that, unlike most birds that breed in the Arctic, has a limited annual migration, making it an excellent indicator of contamination in the region. Concentrations of methylmercury (MeHg) and other trace metals sequestered in feathers during the biannual molt (feather replacement) reflect the amount of trace metals ingested since the last molt. In black guillemots, breeding plumage (obtained in March to April) reflects the trace metal burden of prey consumed in winter while winter plumage (obtained in September to October) reflects prey consumed during the breeding season. Historical (1897 to 1985) and contemporary (2000) black guillemot feathers from Arctic Alaska were recently analyzed for MeHg and other trace metals as well as for $\delta^{13}C$ and $\delta^{15}N$ to assist the determination of trophic level and location(s) of contaminant uptake via feeding. Methylmercury in breeding plumage increased from an average of 581 ± 229 µg/kg (range 99 to 1410 µg/kg, n = 5) in 1897 to 930 ± 146 µg/kg (range 576 to 1950 µg/kg, n = 9) in 2000 (G. Divoky, unpubl. data, 2001). The increase over the last 100 years is not statistically significant, but is similar to the approximately 1% increase per year in black guillemot from Greenland (Appelquist *et al.*, 1985). Methylmercury concentrations in winter plumage for birds collected in 1897 were similar to those in breeding plumage for 1897 (mean 565 ± 232 µg/kg, range 184 to 985 µg/kg, n = 3) indicating that prey in overwintering and summer areas had similar Hg levels.

Similar increases in Hg were seen for white-tailed sea eagle (*Haliaeetus albicilla*) along the northern Norwegian coast (Nygård, 1997). The innermost primary feathers of specimens taken during three periods between 1850 and 1995 were analyzed. Although sample sizes were small for the two earlier periods, a steep increase was seen in the 1950s and 1960s, after which concentrations decreased, appearing to stabilize at about twice background. This pattern was probably influenced by changes in the use of MeHg-based seed dressings as well as changes in Hg discharges from pulp mills and chlor-alkali plants to rivers and fjords, both of which were phased out in Norway during the late-1960s and early-1970s.

In a retrospective study, Braune *et al.* (2001) analyzed eggs of thick-billed murres, northern fulmars (*Fulmarus glacialis*) and black-legged kittiwakes (*Rissa tridactyla*) collected from Prince Leopold Island in Lancaster Sound, Nunavut, Canada, between 1975 and 1998. Total Hg concentrations almost doubled between 1975 and 1998 in thick-billed murre eggs, and a 50% increase was observed in eggs of northern fulmar (Figure 5·12). Stable isotope analyses ($\delta^{15}N$) indicated that the temporal trends were not the result of shifts in trophic level. Concentrations of total Hg in black-legged kittiwake eggs did not change significantly over time, possibly because, unlike thick-billed murres and northern fulmars which overwinter in northern waters, black-legged kittiwakes overwinter at lower latitudes where Hg concentrations are lower owing to recent reductions

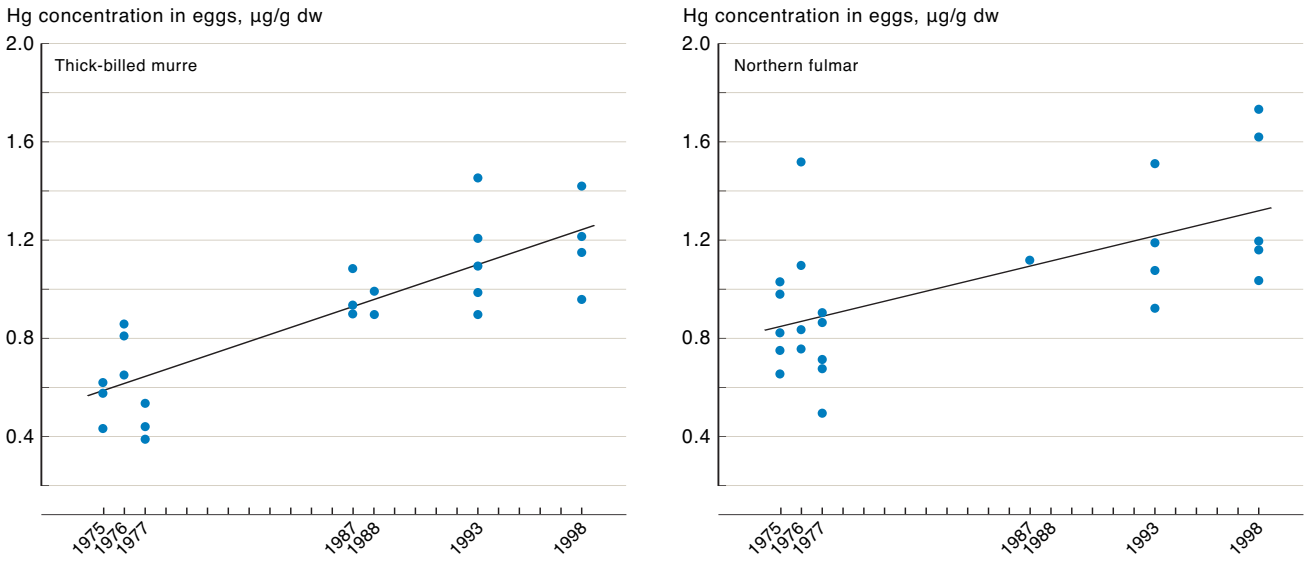


Figure 5·12. Mercury concentrations in eggs of thick-billed murre and northern fulmar collected between 1975 and 1998 from Prince Leopold Island (Braune *et al.*, 2001). Each point represents the average concentration for a pool of three eggs.

in Hg emissions and discharges from point sources (see Section 2.2).

No significant temporal trends were found for Hg or Cd concentrations between the mid-1980s and the mid-1990s in liver of glaucous gull (*Larus hyperboreus*) from Greenland (Riget and Dietz, 2000).

5.4.3.4. Marine mammals

The first AMAP assessment, considered several time-series data sets for metals in marine mammals. Continuation of the work on ringed seal (*Phoca hispida*) and beluga (*Delphinapterus leucas*) in the Canadian Arctic has extended the time series reported by Wagemann *et al.* (1996). Time series data are also available for ringed seal and polar bear (*Ursus maritimus*) from Greenland. Historical trends in Hg concentration are examined using modern and pre-industrial teeth of beluga and hair of polar bear.

The teeth of beluga from the Beaufort Sea, harvested in the Mackenzie Delta in 1993 as part of the traditional Inuit hunt, contained significantly higher Hg concentrations than archeological samples dated 1450 to 1650 AD (Figure 5·13). The increases were age-dependant, with older beluga having greater increases than younger animals, a pattern predicted by previous modeling studies. The increases in geometric mean tooth Hg ranged from 4-fold in 10-year old animals, to 17-fold in 30-year old animals. Because tooth Hg in modern beluga was significantly and linearly correlated with soft tissue Hg concentrations (including concentrations in muscle and muktuk eaten by Inuit), it is likely that soft tissue Hg increased to a similar degree. The Hg increase could not be explained by changes in diet (as inferred from stable C and N isotope analyses), sex differences in the harvested sample, or any other natural phenomenon. The modern beluga also showed significant decreases in stable Pb isotopes compared to pre-industrial animals, with the trend attributed to the introduction of Pb of Eurasian origin (Outridge *et al.*, 1997). In contrast to the Beaufort Sea beluga, walrus (*Odobenus rosmarus rosmarus*) at Igloolik in Foxe Basin had similar concentrations of Hg, Cd, and Pb in pre-industrial and modern teeth (Outridge *et al.*, 1997, 2002b). Stable Pb isotope ratios were also identical between modern and pre-industrial samples, suggesting an absence of industrial Pb uptake in this species at this location. This finding supports the bivalve shell data reported by Outridge *et al.* (2000) for other High Arctic sites (see Section 5.4.3.1).

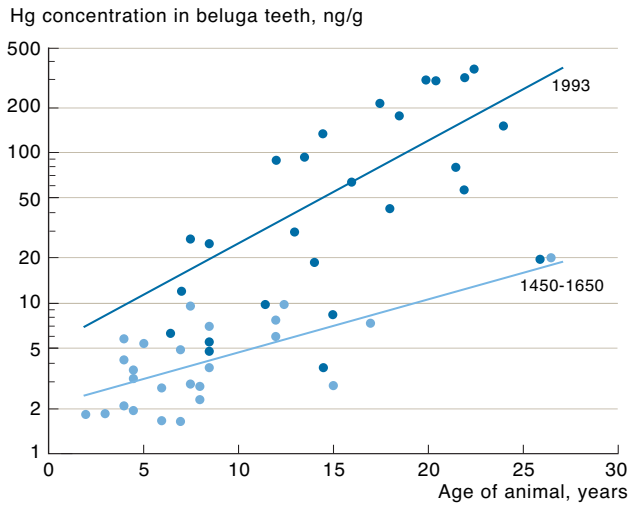


Figure 5·13. Mercury concentrations in the teeth of Beaufort Sea beluga, harvested in the Mackenzie Delta in 1993 as part of the traditional Inuit hunt and in archeological samples dated 1450-1650 AD (Outridge *et al.*, 2002b).

Wagemann *et al.* (1996) showed that Hg concentrations in liver of beluga increased significantly in the western and eastern Canadian Arctic from 1981-1984 to 1993-1994. Similarly, liver of narwhal (*Monodon monoceros*) from Pond Inlet in the eastern Canadian Arctic in 1992/94 had significantly higher mean concentrations than narwhal from the same site in 1978/79 (Wagemann *et al.*, 1996). There was no indication of a trend in Cd concentration in liver or kidney of beluga in the eastern or western Canadian Arctic, nor in narwhal from Pond Inlet (Wagemann *et al.*, 1996). Mercury data for the late-1990s are now available for beluga from several sites in the Canadian Arctic, extending the time series, particu-

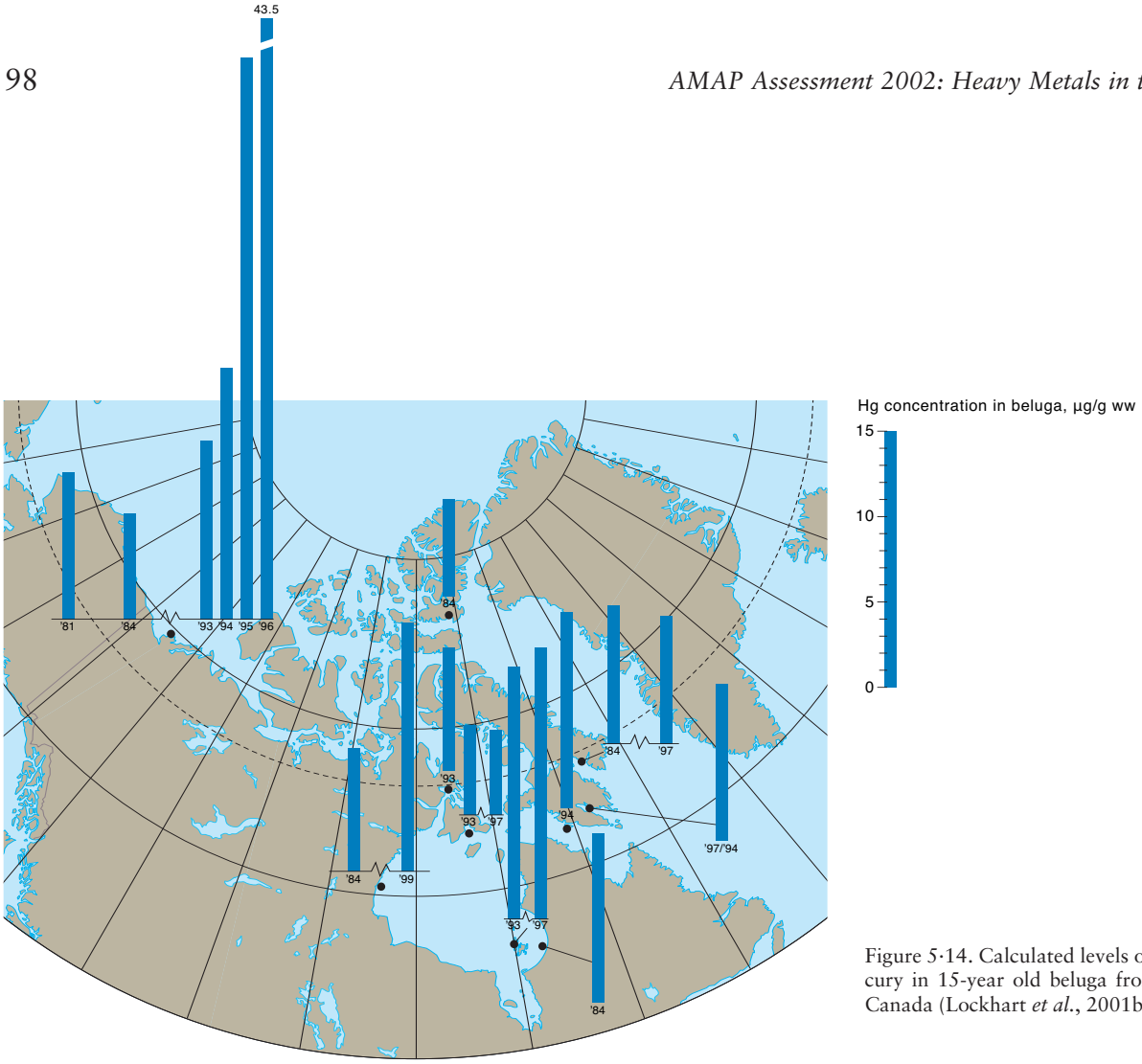


Figure 5·14. Calculated levels of total mercury in 15-year old beluga from northern Canada (Lockhart *et al.*, 2001b).

larly for the Beaufort coast (Figure 5·14). Most of the samples showed a significant correlation between age and Hg concentration in liver, with the older specimens having higher Hg concentrations (Lockhart *et al.*, 2001b). There was little difference between levels in males and females, and so the specimens were treated as a single data set. Robust regression equations were used to standardize the data to the level of Hg in a 15-year old from each collection (Figure 5·14). There was a dramatic increase in Hg levels in beluga from the Beaufort coast in the western Canadian Arctic in the 1990s. Prior to this, levels in the eastern and western regions were similar. Beluga in the eastern Canadian Arctic did not show a consistent pattern; those from Arviat (western Hudson Bay) had higher levels in 1999 than 1984, but that was not the case at Pangnirtung. Samples from Coral Harbor (northern Hudson Bay) were taken in 1993 and 1997, and from Sanikiluaq (southeastern Hudson Bay) in 1994 and 1998, in both cases the interval was probably insufficient to detect change. On the Beaufort coast and at Arviat, where samples were taken 13 years or more apart, an increased rate of Hg uptake was evident in the more recently collected animals (Figures 5·15 and 5·16). Beluga showing the greatest changes in uptake of Hg oc-

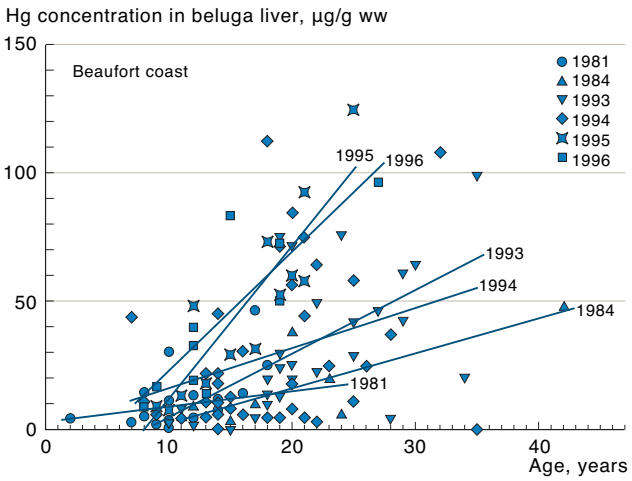


Figure 5·15. Scatter plot of mercury concentrations in liver of beluga in different years on the Beaufort coast of northern Canada (Lockhart *et al.*, 2001b). Least squares regression lines are shown.

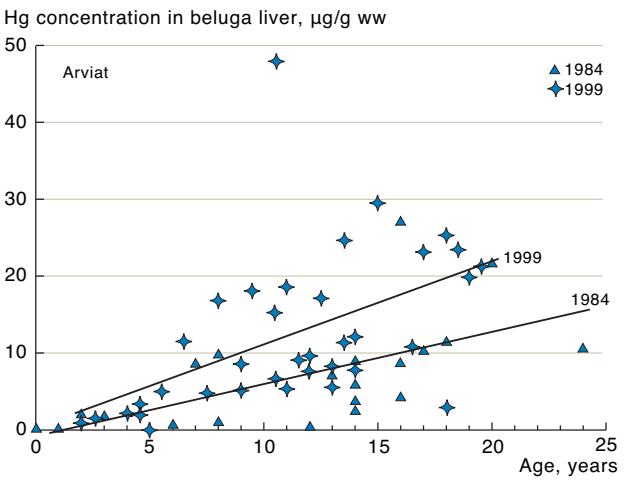


Figure 5·16. Scatter plot of mercury concentrations in liver of beluga in different years at Arviat on the western coast of Hudson Bay (Lockhart *et al.*, 2001b). Least squares regression lines are shown.

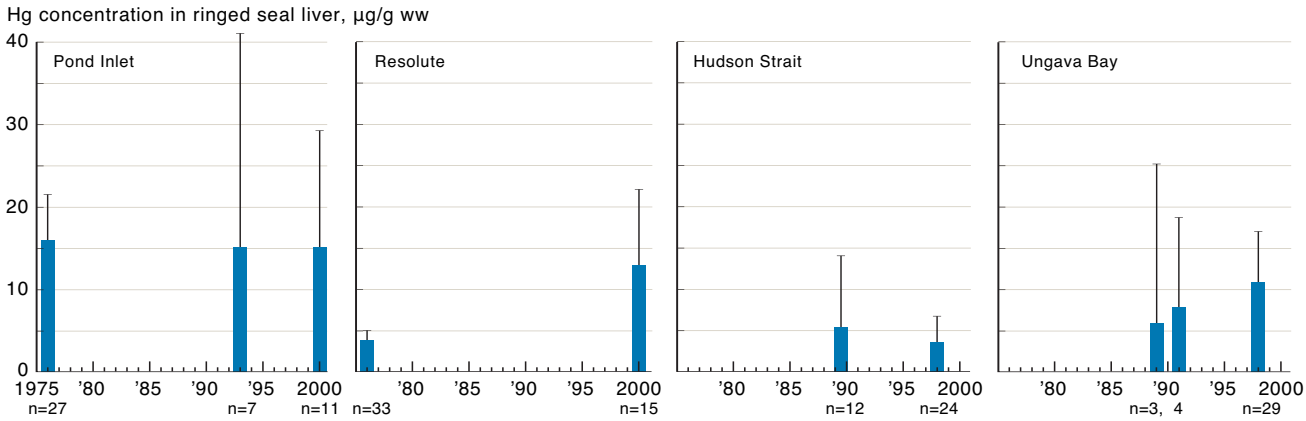


Figure 5·17. Preliminary results of temporal trend analysis for mercury in ringed seal liver from the eastern Canadian Arctic (D. Muir, unpubl. data, 2001). Bars represent geometric mean concentrations with upper 95% confidence intervals (vertical lines) for seals aged 2 to 15 years, except for Pond Inlet, where both bars represent arithmetic means, and the Resolute data for 1976. Results from 1989-91 and 1993 are reported in Dietz *et al.* (1998).

curred in areas influenced by large freshwater drainage (Mackenzie River, Nelson River, James Bay rivers). This suggests that changes in Hg uptake may be related more to Hg in freshwater inputs than to direct atmospheric inputs (Lockhart *et al.*, 2001b). As the effects of dietary change on Hg levels in beluga are currently unknown, these should be investigated using stable C and N isotope analyses.

The ringed seal is the most abundant Arctic pinniped and a top predator in near-shore pelagic food webs. Recent (1998 to 2001) collections of ringed seal from the same sites as were sampled in the 1970s, 1980s, and early-1990s have enabled trends in Hg to be established. Analysis of covariance showed no significant difference between males and females. Results for four sites are shown in Figure 5·17. Geometric mean concentrations from Wagemann *et al.* (1996) that were reported in the AMAP assessment were plotted for the 1980s and early-1990s. Geometric means were not available for the 1970s data. So, for Pond Inlet (2000) the arithmetic mean Hg concentrations in adult animals (5- to 15-year olds) were compared with data reported by Smith and Armstrong (1978) for the same average-aged animals. Concentrations at Pond Inlet were about three-fold higher in 2000 than 1976, which supports the conclusions of Wagemann *et al.* (1996) of increasing Hg concentrations over time. However, no significant trends in Hg were found at the other sites (D. Muir, pers. comm., 2001). These results should be considered preliminary, however, particularly in the absence of information about corresponding dietary changes.

Mercury concentrations in liver of young ringed seals from Greenland also increased from the mid-1980s to the mid-1990s (Riget and Dietz, 2000) and late-1990s (Appendix Table 5·2). In contrast, Cd concentrations in ringed seal liver increased from the late-1970s to the mid-1980s, and then decreased from the mid-1980s to the mid-1990s (Riget and Dietz, 2000). No change in Cd levels was observed between the mid-1990s and the late-1990s. The opposing trends in Hg and Cd concentration suggest changes in dietary habit rather than change in anthropogenically-related exposure (Riget and Dietz, 2000).

Polar bears occupy a high trophic position in the Arctic marine food web because they prey almost exclusively on other marine mammals, particularly ringed seals. In a long-term study using historical samples, Hg levels in the hair of modern polar bears in the Canadian Arctic were several times higher than in pre-industrial samples recovered from archeological sites (Wheatley and Wheatley, 1988). Possible differences in age (which is a significant factor in the Hg content of many mammals) between the modern and pre-industrial populations could not be assessed and may have influenced the results.

Various samples of muscle, liver, kidney, and hair of polar bears collected between 1984 and 2000 from Ittoqqortoormiit in northeastern Greenland were analyzed for Hg, Cd, and Se. A marginally significant increase in muscle Hg was found in adults between 1987 and 2000 ($p = 0.053$). There was no consistent change in Hg or Cd concentrations in other tissues (Table 5·1). However, power analysis suggests that 12 to 38 years of monitoring are required to detect trends owing to the relatively high variance in the data (ICES, 2002). In this study, muscle and hair were better indicator tissues for Hg than liver or kidney, while there was no clear advantage of any particular tissue for Cd.

5.4.3.5. Assessment of temporal trends in marine biota

Long-term data sets for mussels, seabirds, beluga, and polar bears in various regions of the Canadian Arctic and Greenland all show significant increases in Hg concentration between pre-industrial or historical times and the 1970s to 1990s. These increases continue in recent (short-term) data sets for seabird eggs, seals, beluga, and polar bear that are longer than ten years. Evidence of an increasing trend in bioavailable Hg in the marine environment is stronger than for the terrestrial or freshwater environments, and is particularly consistent for several higher-order consumers (birds and mammals) in areas of the Canadian Arctic and Greenland. In contrast, time series for Iceland and the European Arctic generally exhibit a pattern of recently stable or declining Hg concentrations in lower-order marine biota over the last 10 to 30 years. Data for higher-order marine biota in Europe and Iceland are currently lacking.

There is little or no evidence in either long- or short-term data sets of significant Cd increases anywhere in the Arctic marine environment. The increase in Pb levels between a thousand or more years ago and the 1970s in blue mussels from Arctic Canada, and several examples

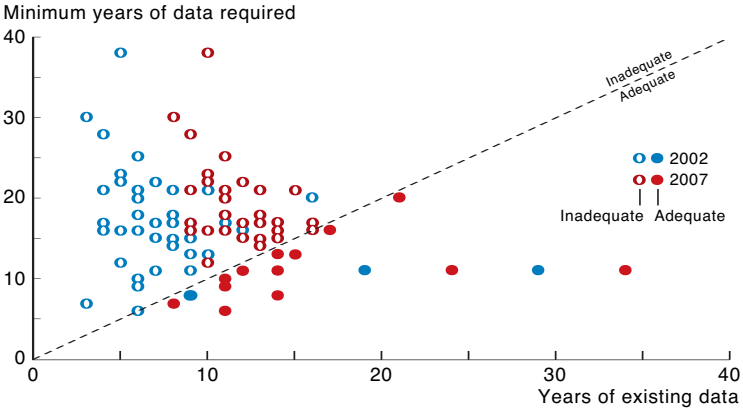


Figure 5·18. The 'adequacy' of current short-term data sets for mercury concentrations in Arctic biota and the 'adequacy' of short-term data sets for mercury concentrations in Arctic biota if current programs continue for another five years (Bignert *et al.*, 2004). Each point represents a species monitored at the same location over time. The minimum number of years of data required is the number of years needed to statistically detect a trend of 5% annual change in mercury concentration with 80% power (i.e., certainty). Differences between data sets are mainly related to varying degrees of within- and between-year variation in mercury concentration.

of changing Pb isotope ratios in Arctic biota, are consistent with significant modern inputs of anthropogenic Pb. The post-1970s decline in Pb concentrations in other environmental compartments has not been observed in Arctic marine biota, for reasons that are presently not understood.

5.5. The adequacy of temporal data sets

Conclusions reached at the AMAP Workshop on Statistical Analysis of Temporal Trend Data were that the longevity of existing short-term temporal trend data sets is generally inadequate to reliably detect 5% annual changes in trace metal concentrations with a power level of 80% (ICES, 2002; Bignert *et al.*, 2004). With certain exceptions, in most countries and for most types of biota, the variability in concentrations of metals in individual animals and the relatively small numbers of samples collected make it difficult to establish trends – or the absence of trends – with a reasonable degree of certainty. At least 10 years and up to 30 years of annual data are generally required to provide a sufficiently powerful database to detect temporal change. The evaluation of data requirements, including the decision to sample annually or at regular intervals of several years, must be made on a case-by-case basis and be based on at least several consecutive years of monitoring. Efforts should be made to extend existing temporal monitoring studies of biota, and to initiate the collection and specimen-banking of biota in areas not currently covered (e.g., North Atlantic marine mammals and birds; Russian biota in general). Calculations show that while only about 10% of short-term time-series data sets are adequate at present, if data collection continues for another five years, 34% of data sets will achieve minimum standards of reliability and statistical power (Figure 5·18).

5.6. Summary and conclusions

5.6.1. Long-term change

Evidence from peat cores and lake sediments suggests that atmospheric Hg deposition has increased significantly in remote regions of the Arctic since the Industrial Revolution (1800 AD). In the Canadian Arctic and Greenland, increases in Hg concentration above pre-industrial levels are of the order of 2- to 4-fold, amounting to a presumed anthropogenic flux of 2 to 10 μg/m^2/yr. Much higher Hg increases (7- to 17-fold) occur in Arctic peat cores, but until flux estimates are made these data cannot be compared with lake sediments. It should be noted that these conclusions assume that Arctic lake sediments and peat bogs provide reliable archives of atmospheric Hg deposition, and that such data are not confounded by sediment diagenesis or the possible influence of climate change, both of which require further study.

Anthropogenic Pb deposition in remote environments such as the Greenland Summit and Hudson Bay (Canada) has also increased dramatically over the past 200 years; however, levels appear to have decreased over recent decades owing to successful emission reduction strategies in Europe and North America. Lead isotope studies have confirmed these conclusions.

Industrial Cd sources do not appear to have resulted in increasing levels of Cd in most of the remote areas of the Arctic, although regions close to industrial areas in the Eurasian Arctic have been affected by Cd and other metal emissions from these sources.

Platinum group elements (Pt, Pd, Rh) from catalytic converters on cars have shown relatively large increases (40- to 120-fold) in Greenland Summit snow and ice and may be a significant future issue. Such increases in deposition are comparable to those observed for Pb at its maximum. The ecotoxicology of the PGEs is largely unknown.

Long-term data sets with which to quantify the relative importance of natural and anthropogenically-derived metals in biota over time, including species important for the human diet, are few. Some progress has been made since the first AMAP assessment in obtaining data for the Arctic regions of Canada and Norway, mainly using calcified tissues (shells and teeth) to identify change since the pre-industrial period. Mussels in Hudson Bay in the 1970s showed several-fold higher Pb and Hg concentrations than during the Holocene period (more than 1000 years ago). Increases in Hg concentration since the period 1300 to 1500 AD have been observed in species at the top of food chains; increases of around an order of magnitude have been observed in Beaufort Sea beluga (4- to 17-fold by the 1990s), and in the Norwegian human population (13-fold by the 1970s). The data suggest that anthropogenic Hg accounts for >80% of the total Hg in these species, although Hg levels in Norwegian human teeth have since declined. Cadmium concentrations in higher order biota have not increased significantly since the pre-1800 era.

5.6.2. Short-term trends

For biota, the majority of recent temporal trend data sets focus on Hg and Cd (Figures 5·19 and 5·20). The

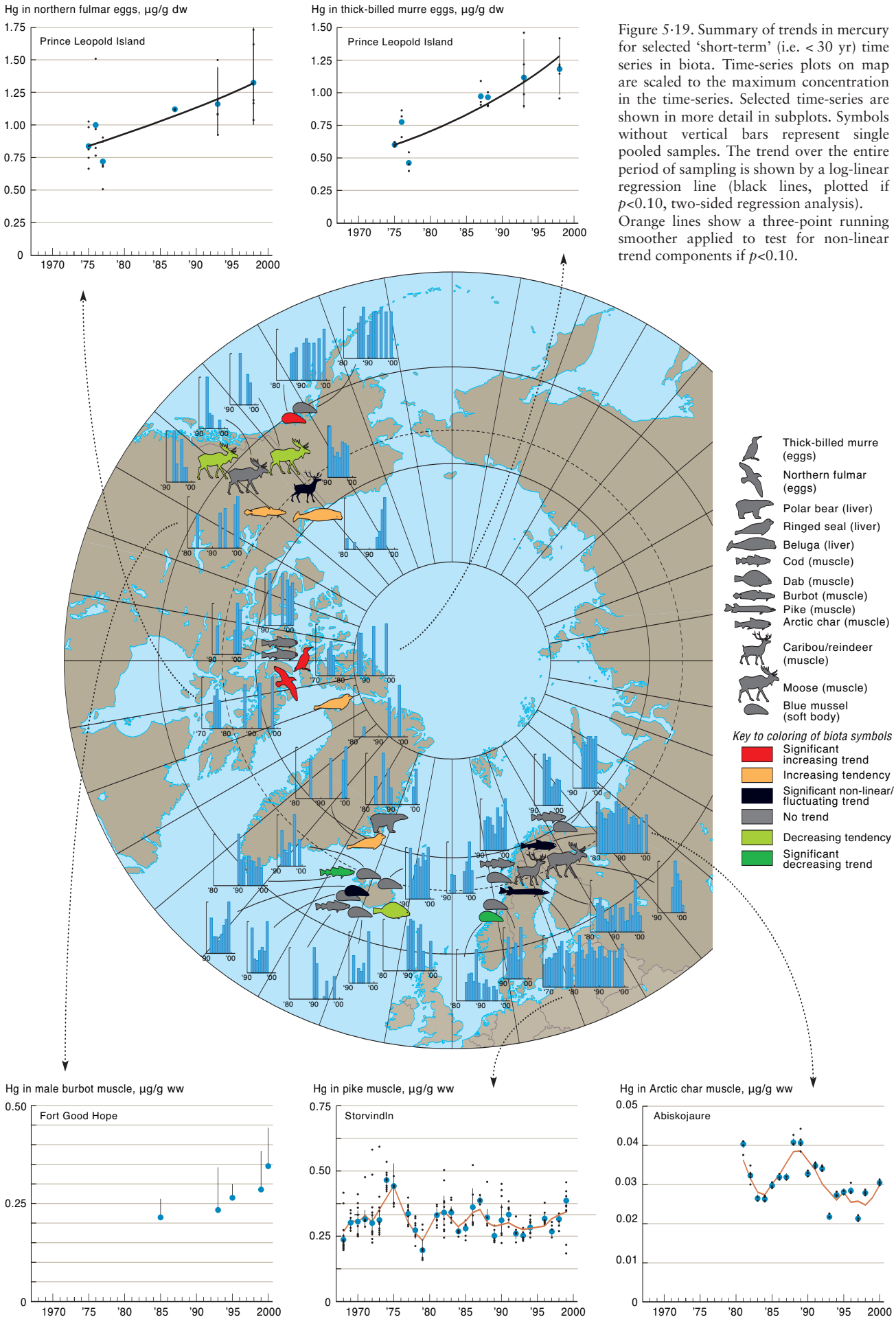


Figure 5·19. Summary of trends in mercury for selected 'short-term' (i.e. < 30 yr) time series in biota. Time-series plots on map are scaled to the maximum concentration in the time-series. Selected time-series are shown in more detail in subplots. Symbols without vertical bars represent single pooled samples. The trend over the entire period of sampling is shown by a log-linear regression line (black lines, plotted if $p<0.10$, two-sided regression analysis). Orange lines show a three-point running smoother applied to test for non-linear trend components if $p<0.10$.

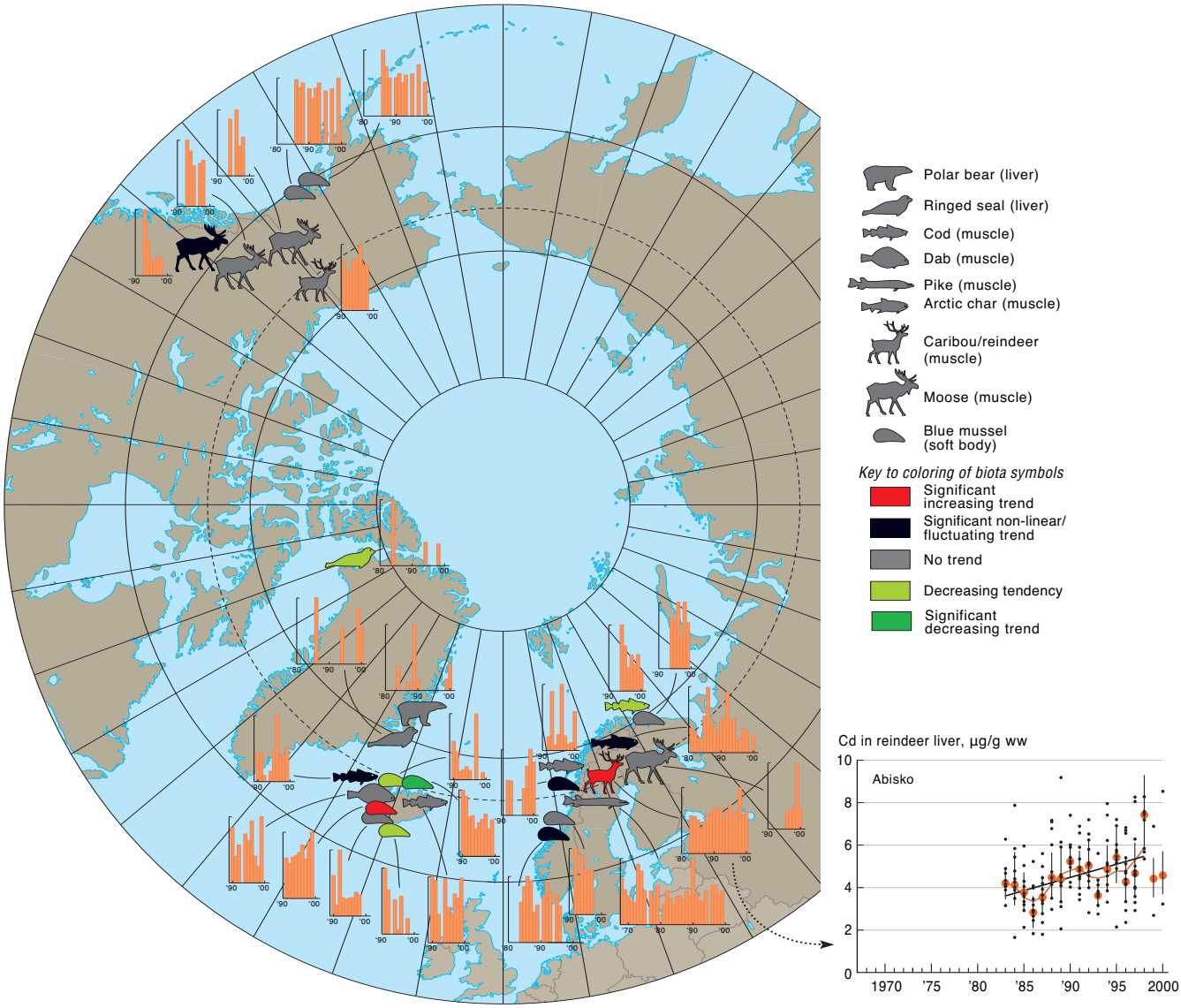


Figure 5·20. Summary of trends in cadmium for selected 'short-term' (i.e. < 30 yr) time series in biota. Time-series plots on map are scaled to the maximum concentration in the time-series. A subplot shows more detail for the time-series of Cd in reindeer liver at Abisko. The trend over the entire period of sampling is shown by a log-linear regression line (black line, plotted if $p<0.10$, two-sided regression analysis). The orange line shows a three-point running smoother applied to test for non-linear trend components if $p<0.10$.

evidence for increasing trends in Hg is compelling in the Canadian and Greenland Arctic, reflected in a number of species of marine birds and mammals. In contrast, Hg in the European Arctic in lower-order marine biota and terrestrial mammals is either stable or declining. The reason for this decoupling between the Canadian/Greenland and European Arctic is unknown, particularly as emission inventories suggest anthropogenic Hg output is declining across the Northern Hemisphere. In freshwater biota the only increase in Hg concentration was in burbot in the Mackenzie River, Canada.

Cadmium levels in biota and in the abiotic environment are either stable or declining, with the exception of a recent Cd increase in reindeer liver in northern Sweden.

Lead levels appear to be declining in atmospheric particulates and precipitation throughout the Arctic. Although this pattern is also evident in mosses, the decreases have not been observed in Arctic fauna, in which Pb concentrations are generally stable.

The only temporal trend data sets for biota from Russia or Finland are for mosses, and in Russia these only cover the Kola Peninsula. Temporal monitoring of biota throughout northern Russia would be useful for making geographical comparisons of temporal trends.

Appendix Table 5·1. Detailed results for the statistical analyses performed during the AMAP workshop (ICES, 2002) as summarized in Table 5.1.

		Hg		Cd		Pb		
		Trend	Adeq.	Trend	Adeq.	Trend	Adeq.	Source
Canada								
Caribou								
kidney	Porcupine Herd, Yukon	fluc.	0.56	none	0.73	none	0.56	MacDonald *et al.*, 2002
Moose								
kidney	Dawson, Yukon	none	0.43	none	0.25	none	0.125	P. Roach, unpubl. data, 2001
kidney	Ross River, Yukon	none	0.19	none	0.5	none	0.235	P. Roach, unpubl. data, 2001
kidney	Teslin, Yukon	none	0.10	none	0.18	none	0.143	P. Roach, unpubl. data, 2001
kidney	Watson Lake, Yukon	none	0.23	fluc.	0.32	none	0.171	P. Roach, unpubl. data, 2001
kidney	Whitehorse, Yukon	none	0.14	none	0.56	none	0.263	P. Roach, unpubl. data, 2001
Thick-billed murre								
eggs	Prince Leopold Isl, Nunavut	incr.	0.64	n.a.		n.a.		Braune *et al.*, 2001
Northern Fulmar								
eggs	Prince Leopold Isl, Nunavut	incr.	0.67	n.a.		n.a.		Braune *et al.*, 2001
Black-legged Kittiwake								
eggs	Prince Leopold Isl, Nunavut	none	0.31	n.a.		n.a.		Braune *et al.*, 2001
Greenland								
Polar bear								
muscle	Ittoqqortoormiit	none	0.42	none	0.13	n.a.		F. Riget, unpubl. data, 2002
liver	Ittoqqortoormiit	none	0.30	none	0.21	n.a.		F. Riget, unpubl. data, 2002
kidney	Ittoqqortoormiit	none	0.24	none	0.18	n.a.		F. Riget, unpubl. data, 2002
hair	Ittoqqortoormiit	none	0.60	n.a.		n.a.		F. Riget, unpubl. data, 2002
Iceland								
Blue mussel								
soft body	Grímsey	none	0.53	decr.	0.77	n.a.		ICES TDC
soft body	Hvalstöd, Hvalfjord	none	0.38	incr.	1.00	n.a.		ICES TDC
soft body	Hvítanes, Hvalfjord	fluc.	0.44	none	0.50	n.a.		ICES TDC
soft body	Hvassahraun	none	0.32	none	0.41	n.a.		ICES TDC
soft body	Straumsvík	fluc.	0.32	none	0.80	n.a.		ICES TDC
soft body	Hvaleyri, Hvalfjord	none	0.47	fluc.	0.75	n.a.		ICES TDC
soft body	Úlfsá by Skutulsfjord	none	0.29	none	0.28	n.a.		ICES TDC
soft body	Vestmannaeyjar	none	0.22	none	0.20	n.a.		ICES TDC
Cod								
muscle	NW Iceland	decr.	0.69	n.a.		n.a.		ICES TDC
liver	NW Iceland	n.a.		fluc.	0.36	n.a.		ICES TDC
muscle	SW Iceland	none	0.13	n.a.		n.a.		ICES TDC
liver	SW Iceland	n.a.		none	0.33	n.a.		ICES TDC
muscle*	E and NE Iceland	fluc.	0.82	n.a.		n.a.		ICES TDC
muscle**	E and NE Iceland	none	0.69	n.a.		n.a.		ICES TDC
liver	E and NE Iceland	n.a.		none	0.53	n.a.		ICES TDC
Dab								
muscle	NW Iceland	decr.	0.57	n.a.		n.a.		ICES TDC
liver	NW Iceland	n.a.		none	0.53	n.a.		ICES TDC
muscle	SW Iceland	none	0.60	n.a.		n.a.		ICES TDC
liver	SW Iceland	n.a.		none	0.45	n.a.		ICES TDC
muscle*	SE Iceland	none	0.41	n.a.		n.a.		ICES TDC
muscle**	SE Iceland	none	0.47	n.a.		n.a.		ICES TDC
liver	SE Iceland	n.a.		none	0.29	n.a.		ICES TDC
Norway								
Blue mussel								
soft body	Skagodden	none	1.0	none	0.38	none	0.24	ICES TDC
soft body	Sildkroneset, Bokfjord	none	0.25	none	0.36	none	0.21	ICES TDC
soft body	Brashavn	none	0.24	none	0.21	none	0.17	ICES TDC
soft body	Flakk	none	0.65	none	0.58	fluc.	0.42	ICES TDC
soft body	Trossavika	decr.	0.69	fluc.	0.79	none	0.52	ICES TDC
soft body	Ingdalsbukt	decr.	0.48	none	0.73	none	0.41	ICES TDC
soft body	Stokken	none	0.33	none	0.46	none	0.30	ICES TDC
soft body	Husvagen	none	0.38	fluc.	0.40	none	0.55	ICES TDC
Cod								
muscle*	Varangerfjorden	decr.	0.50	n.a.		n.a.		ICES TDC
muscle**	Varangerfjorden	none	0.47	n.a.		n.a.		ICES TDC
liver	Varangerfjorden	n.a.		none	0.47	n.a.		ICES TDC
muscle*	Lille Molla	none	0.60	n.a.		n.a.		ICES TDC
muscle**	Lille Molla	none	0.53	n.a.		n.a.		ICES TDC
liver	Lille Molla	n.a.		none	0.28	n.a.		ICES TDC
Sweden								
Char								
muscle*	Abiskojaure	fluc.	1.9	n.a.		n.a.		Bignert, 2001
muscle***	Abiskojaure	fluc.	1.7	n.a.		n.a.		Bignert, 2001
liver	Abiskojaure	n.a.		fluc.	0.90	none	1.0	Bignert, 2001

Appendix Table 5·1 continued.

		Hg		Cd		Pb		
		Trend	Adeq.	Trend	Adeq.	Trend	Adeq.	Source
Pike								Bignert, 2001
muscle*	Storvindeln	fluc.	2.4	n.a.		n.a.		Bignert, 2001
muscle***	Storvindeln	fluc.	2.6	n.a.		n.a.		Bignert, 2001
liver	Storvindeln	n.a.		none	1.9	fluc.	1.7	Bignert, 2001
Reindeer								
liver	Abisko	none	0.80	incr.	1.5	decr.	0.81	Odsjö, 2003
muscle	Abisko	none	0.75	none	0.46	none	0.42	Odsjö, 2003
Moose								
liver	Norrbotten	n.a.		none	0.22	n.a.		Odsjö *et al.*, 2001
kidney	Norrbotten	n.a.		none	0.19	n.a.		Odsjö *et al.*, 2001
United States								
Blue mussel								
soft body	Unakwit Inlet (Siwash Bay), Alaska	none	0.77	none	0.83	none	0.56	NOAA
soft body	Port Valdez (Mineral Creek Flats), Alaska	incr.	1.1	none	1.25	none	0.77	NOAA

* unadjusted data; ** length-normalized data; *** weight-normalized data.
Adeq. = ratio of number of years of data : number of years of data required to detect an average trend of 5% change per year;
incr. = significant increasing (log-linear) trend; decr. = significant decreasing (log-linear) trend; none = no significant linear or non-linear trend;
fluc. = a significant fluctuating (non-linear) trend was detected, with no general increasing or decreasing tendency; n.a.: data not available.

Appendix Table 5·2. Trend observations for biota datasets for which time series were too short or inadequate for their inclusion in the detailed statistical analyses performed during the AMAP workshop (see Appendix Table 5.1). Alternative statistical analyses were applied to some of these trend series.

		Collection years	Period, yr	Concentration changes	Source
TERRESTRIAL					
Long-term					
Human					
teeth	Norway	1100s, 1970s	~800	13-fold increase in Hg, possible recent decline over past 20 years	Eide *et al.*, 1993; Tvinnereim *et al.*, 1997
hair	Greenland	1400s, 1900s	~500	2.5-fold increase in Hg	Hansen *et al.*, 1991
hair	Canada	400-460, 1150, 1970s	~1500	Hg several times higher in post- than in pre-industrial samples	Wheatley and Wheatley, 1988
Short-term					
Moss	Faroe Islands	1996, 2000	4	Lower Pb levels in samples collected in 2000	M. Dam, unpubl. data, 2001
Lichen	Greenland	1994, 1999	6	No significant change in Hg or Cd; significantly lower Se in 1999	F. Riget, unpubl. data, 2002
Peregrine falcon					
egg	Alaska	1988-90, 1991-95	8	Possible Hg increase in American peregrine falcon, but Hg not significantly different between year groups	Ambrose *et al.*, 2000
Predatory birds (golden eagle, goshawk, merlin, peregrine falcon) feather	Norway	pre-1950s, 1950s-1960s, 1970s, present day	~50	Increased Hg in 1950s to 1960s, then a decline after 1970. Modern levels stabilized at higher levels than in the pre-1950s	Nygård, 1997
Mountain hare					
liver	Faroe Islands	1997, 1999	3	No significant differences observed	Olsen *et al.*, 2003
Reindeer/caribou					
kidney and liver	Lake Harbour Herd (South Baffin Island), Canada	1992,1999	8	Cd unchanged	Elkin, 2000; Macdonald *et al.*, 2002
liver	Isortoq, Greenland	1994, 1999	6	No significant differences in Cd. Hg and Se significantly lower in 1999.	F. Riget, unpubl. data, 2002
liver	Kitaa, Greenland	1995/97, 1999	5	Lower Hg in 1999.	F. Riget, unpubl. data, 2002

Appendix Table 5·2 continued.

		Collection years	Period, yr	Concentration changes	Source
FRESHWATER					
Short-term					
Lake Trout					
muscle	Lake Laberge, Canada	1993, 1996, 1998	6	30% decrease in (length-adjusted) Hg between 1993 and 1996 and no change between 1996 and 1998.	Stern *et al.*, 2001b
muscle	Quiet Lake, Canada	1992, 1999	8	No significant difference in Hg.	Stern *et al.*, 2001b
muscle	Great Slave L. (East Arm), Canada	1995, 1999	5		Evans and Muir, 2001
Burbot					
liver	Lake Laberge, Canada	1996, 1999	4	No change in Hg.	Stern *et al.*, 2001b
muscle	Fort Good Hope, Canada	1985, 1993, 1995, 1999, 2000	16	Mean Hg in male burbot increased by 36% between 1985 and 2000	Stern *et al.*, 2001a
muscle	Great Slave L. (West Basin), Canada	1995, 1996, 1999	5		Evans and Muir, 2001
Char					
muscle	Resolute Lake, Canada	1992, 1997, 1999, 2000, 2001	10 10	No significant change in Hg. But, mean concentrations in char from 2000/2001 were about 20% lower than in 1992 and 1997.	Muir *et al.*, 2001a; Muir and Lockhart, 1994
muscle	Char Lake, Canada	1992, 1999, 2000, 2001	10	Hg higher in (larger) fish in 2000 than fish in 1999.	Muir *et al.*, 2001a; Muir and Lockhart, 1994
muscle	Isortoq, Greenland	1994, 1995, 1999	6	No significant change in (length-adjusted) Hg.	Riget, *et al.*, 2000a
Pike					
muscle	Great Slave L. (West Basin), Canada	1996, 1999	4		Evans and Muir, 2001
MARINE					
Long-term					
Mussel					
shell	Axel Heiberg and Cornwallis Islands, Canada	9000-1000 BP, 1980s-1990s	~9000	Similar Hg, Pb and Cd in Holocene and modern samples. Cd, Zn and other trace metals similar or lower in modern samples.	Outridge *et al.*, 2000
shell	SE Hudson Bay, Canada	9000-1000 BP, 1970s	~9000	Significantly higher Hg and Pb in modern shells. Cd, Zn and other trace metals similar or lower in modern samples.	Outridge *et al.*, 2000
Black guillemot					
feather	Alaska	1897-2000	104	Increased MeHg in 2000 compared to 1897-1985 feathers.	G. Divoky, unpubl. data, 2001
White-tailed sea eagle					
feather	Norway	3 periods between 1850 and 1995	~150	Pronounced increase in Hg in 1950s and 1960s, then a decrease, stabilizing at about twice background	Nygård, 1997
Beluga					
teeth	Mackenzie Delta, Canada	1450-1650, 1993	~500	Significantly higher (4- to17-fold) Hg in modern samples	Outridge *et al.*, 2002b
Walrus					
teeth	Igloolik, Canada	1200-1500, 1987-1988	~700	Similar Hg, Cd and Pb in pre-industrial and modern teeth	Outridge *et al.*, 1997, 2002b
Polar bear					
hair	Canada	300BC-1500AD, 1930-1977	~2000	Significantly higher methly-Hg in modern samples	Wheatley and Wheatly, 1998

Appendix Table 5·2 continued.

		Collection years	Period, yr	Concentration changes	Source
Short-term					
Mussel	Qeqertarsuaq, Greenland	1994, 1999	6	Cd higher in some length classes, lower in others. Hg lower in all but the largest mussels. Se lower in all length classes	Riget, *et al.*, 1997b; F. Riget, F. Riget, unpubl. data, 2001
	Nuuk, Greenland	1987-1990	4	No significant trends	Riget and Dietz 2000
	Uummannaq, Greenland	1988-1995	8	No significant trends	Riget and Dietz 2000
Shorthorn sculpin					
liver	Qeqertarsuaq, Greenland	1994, 1999, 2000	7	No clear differences in Cd, Hg or Se.	F. Riget, unpubl. data, 2002
liver	Uummannaq, Greenland	1980 to 1993, annually	13	Cd decreased, but no significant trend	Riget and Dietz, 2000
liver	Avanersuaq, Greenland	mid-1980s, mid-1990s	10	Hg significantly higher in mid-1990s	Riget and Dietz, 2000
liver	Nanortalik, Greenland	mid-1980s, mid-1990s	10	Hg significantly higher in mid-1990s	Riget and Dietz, 2000
Dab					
liver	Norway	1992/94-1999, annually	8		Green *et al.*, 2001
Glaucous gull					
liver	Greenland	1984/86, 1995	12	No significant differences in Hg or Cd	Riget and Dietz, 2000
Black guillemot					
egg	Faroe Islands	1999-2001, annually	3	Hg decrease from 1999 to 2000 and no change 2000 to 2001	Olsen *et al.*, 2003; M. Dam, M. Dam, unpubl. data, 2001
Beluga					
liver	W and E Canadian Arctic	1981-1984, 1993-1994	10 10	Significantly higher Hg in 1993-1994. No differences in Cd.	Wagemann *et al.*, 1996
Narwhal					
liver	Pond Inlet, Canada	1978-1979, 1992-1994	17	Significantly higher Hg in 1992-1994. No differences in Cd.	Wagemann *et al.*, 1996
Beluga					
liver	Mackenzie Delta, Canada	1981-1984, 1993, 1994, 1995, 1996, 2001	21	Incease in Hg during 1990s (possibly diet-related)	G. Stern, unpubl. data, 2001
liver	Arviat, Canada	1984, 1999	16	Much higher Hg in 1999	G. Stern, unpubl. data, 2001
liver	Pangnirtung, Canada	1984, 1993, 1994, 1997	14	Similar Hg in 1984 and later	G. Stern, unpubl. data, 2001
liver	Coral Harbour, Canada	1993, 1997, 2000	8		G. Stern, unpubl. data, 2001
liver	Sanikiluaq, Canada	1994, 1998 1994, 1998	5		G. Stern, unpubl. data, 2001
liver	Iqaluit, Canada	1993, 1994	2		G. Stern, unpubl. data, 2001
Ringed seal					
liver	Pond Inlet, Canada	1976, 2000	25	Hg ca. 3-fold higher in 2000	Smith and Armstrong, 1978; Muir *et al.*, 2001b
liver	Resolute, Canada	1976, 1993, 2000	25	No significant change in Hg	D. Muir, unpubl. data, 2001
liver	Hudson Strait, Canada	1989/90, 1998	10	No significant change in Hg.	D. Muir, unpubl. data, 2001
liver	Ungava Bay, Canada	1989, 1991, 1998	10	No significant change in Hg.	D. Muir, unpubl. data, 2001
liver	Avanersuaq, Greenland	1984/85, 1994, 1998	15		F. Riget, unpubl. data, 2001
liver	Ittoqqortoormiit, Greenl.	1986, 1994, 1999, 2000	15		F. Riget, unpubl. data, 2001
Polar bear (adult)					
hair, muscle, liver	Ittoqqortoormiit, Greenl.	1984, 1986/87 1989, 1990, 1999, 2000		Marginally significant increase in muscle Hg between 1987 and 2000, but not reflected in other tissues. No significant differences in Cd or Se.	F. Riget, unpubl. data, 2001

Chapter 6

Biological Effects

John Derome, Anne Fairbrother, Suzanne Marcy, Jeffrey Wirtz and Katie Harding

Arctic biota are exposed to heavy metals of anthropogenic origin, yet exposure does not always constitute an effect. The potential for biological effects is directly influenced by the amount and type of exposure and this is determined by many factors including transport mechanisms, abiotic processes, forms of heavy metals, food abundance, and receptor behaviors such as feeding habits and movement across the landscape (especially long-distance migration). Although detecting effects remains a challenge, progress has been made.

This assessment targets effects from exposure to mercury (Hg), cadmium (Cd), and lead (Pb) in terrestrial, freshwater, and marine ecosystems, and reviews new data and information on the biological effects of heavy metals on Arctic species and ecosystems that have become available since the first AMAP assessment.

The effects considered in this chapter include obvious effects from high levels of exposure, and a few cases where effects were detected in biota subject to chronic exposure. In those cases where exposure occurred but effects were not detected, or could not be linked directly to exposure, an estimate of the likelihood of an effect occurring is made by comparing metal concentrations in tissues to known effects levels in non-Arctic species through the use of established toxicity thresholds. For metals in environmental media (e.g., water, sediment, soil) or the diet, concentrations are compared to toxicity-based criteria or thresholds. Appendix 6.1 provides information on toxicity-based criteria and thresholds from published sources that include information on the derivation of the thresholds and related uncertainties.

Conclusions concerning effects are based primarily on concentrations relative to toxicity thresholds and make no links to or assumptions about sources, pathways, spatial patterns, or temporal trends. The chapter provides a screening level summary of recently observed effects and potential exceedences of regulatory criteria or toxicity benchmarks. It is important to note that such benchmarks have significant limitations owing to the laboratory testing of non-Arctic species under temperate conditions and are often based on acute effects (e.g., death) that are unlikely to provide screening for subtle reproductive, behavioral, or neurological effects.

The first AMAP assessment (AMAP, 1998) raised concern about the concentrations of some heavy metals in the tissues of some species. Cadmium concentrations in reindeer/caribou (*Rangifer tarandus*), moose (*Alces alces*), and ptarmigan from the Yukon Territory, and in seabirds and marine mammals from northwest Greenland were considered sufficient to result in possible kidney damage. Marine mammals in the western Canadian Arctic and Faroe Islands showed elevated Hg levels, although selenium (Se) was considered to provide some protection against Hg poisoning. However, although raising concern, evidence of effects was generally lacking during the first AMAP assessment. AMAP was thus charged with filling data gaps, with priority to be given to Hg, Cd, and Pb in species at risk of biological effects, and with collecting data for key regions for which no data were available (e.g., marine mammals in Russian waters).

6.1. Detecting the environmental effects of heavy metals

Effects arising from exposure to heavy metals may take many forms. Available methods for detecting effects include body condition, lesions, histopathology, and reproductive status (AMAP, 1999). The challenges associated with identifying these observable effects in Arctic biota and linking them to heavy metal exposure are considerable. In addition, many subtle effects are not readily observed.

6.1.1. Limitations on detecting effects

Metal concentrations in tissues can be determined routinely and with high precision. However, tissue concentrations alone do not provide accurate predictions of biological effects. The sensitivity of tissues to metal accumulation, for example, varies with tissue type and for different species. Some species accumulate high metal levels without apparent ill effect. Several metals are essential in low concentrations (e.g., Se, zinc (Zn), copper (Cu)), while others are toxic even at trace levels (e.g., Hg, Cd, Pb). Metals may also be sequestered or depurated in tissues or wastes (e.g., feathers, feces) such that measurable levels in tissues have no apparent relationship to effects or can be stored for long periods before causing effects. Some invertebrate species accumulate metals in lysosomes and granules, thereby rendering these metals unavailable for toxic interactions with other cellular components. However, binding metals to proteins such as metallothionein (MT) provides only temporary 'detoxification', rather than permanent storage of the metal, and at times may actually increase metal toxicity (e.g., Cd bound to MT in the blood is more nephrotoxic than unbound Cd). Storage and excretion of non-essential metals follow different paths in different species, making generalizations difficult.

All metals can become toxic at high concentrations, whether exposure is direct or through the food web. In most contexts, an assessment of heavy metal effects requires an understanding of biological responses to known levels of exposure. Where possible, exposure levels should be linked to tissue metal concentrations.

6.1.2. Measures to detect effects

Three types of effects measures are used in this assessment.

- Observed effects. These may occur at the individual, population, or community level (e.g., reproductive failure, change in biotic community structure).

- Indicators of effects (e.g., physiological responses to exposure such as hemoglobin synthesis, or bioassays).
- Indicators of exposure (e.g., tissue levels, biomarkers).

Observed effects and indicators of effects are direct measures of biotic response. However, data most readily available for Arctic species are measures of ambient environmental concentrations, and tissue concentrations. Such indicators of exposure provide information on where to anticipate biotic response. The most powerful data are those where observed effects occur under conditions of known exposure, and cause–effect relationships can be established (see Section 6.1.3). Such data are rare.

Since there are few effects data for Arctic species, it is useful to compare ambient concentrations and tissue levels of metals in the Arctic with published criteria and tissue levels that are known to cause effects in similar species (e.g., as for persistent organic pollutants in the first AMAP assessment; see Section 6.6).

6.1.3. Linking effects to heavy metal exposure

A number of factors influence the bioavailability, uptake, and resulting effects of heavy metals in Arctic flora and fauna. Interactions between stressors make it difficult to distinguish the specific cause of an observed effect in terrestrial ecosystems (e.g., effects of smelter-related sulfur dioxide (SO_2) emissions and heavy metal deposition). Other environmental influences can increase stress (e.g., high levels of solar radiation) or create local variations in deposition (e.g., wind direction, precipitation frequency). In freshwater ecosystems, interactions between several factors (e.g., lake biogeochemistry, rate of methylation by sulfur-reducing bacteria, trophic position) appear to determine the extent of methylmercury (MeHg) bioaccumulation, but such interactions are not clearly understood. Complex interactions also dictate the extent of exposure experienced by marine wildlife in comparable situations. Organisms exposed to heavy metals are subject to direct and indirect toxicity effects (e.g., dramatic changes in soil processes and vegetation linked to high levels of heavy metals in the local environment). Direct and indirect effects are both important when determining effects on animal species.

Thus, even when heavy metal exposure is confirmed and effects are observed, establishing the cause is still difficult. To establish cause and effect relationships, a link must be made between known exposure and observed changes in the organism or population. A set of established and verified criteria for identifying cause and effect relationships (Hill, 1965) are instructive for interpreting information.

- Strength. A high magnitude of effect is associated with exposure to the stressor.
- Consistency. The association is repeatedly observed under different circumstances.
- Specificity. The effect is diagnostic of a stressor.
- Temporality. The stressor precedes the effect in time.
- Presence of a biological gradient. A positive correlation between the stressor and response.
- A plausible mechanism of action.
- Coherence. The hypothesis does not conflict with knowledge of natural history and biology.
- Experimental evidence.
- Analogy. Similar stressors cause similar responses.

Not all criteria must necessarily be satisfied, but each reinforces evidence for causality. Negative evidence does not rule out causal associations, it may only indicate an incomplete knowledge of the relationship (Rothman, 1986).

The importance of measuring tissue concentrations and biotic condition during sampling cannot be overemphasized when trying to establish cause and effect (AMAP, 1999). Such data will help with the future detection of effects that can be linked to specific heavy metal exposure.

6.2. Effects of point sources in terrestrial ecosystems

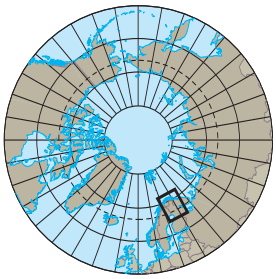
Although metals are naturally found in soils, elevated levels derived from anthropogenic sources can have adverse effects on terrestrial ecosystems. Soil processes (e.g., nitrogen mineralization and decomposition) may be affected when soil microorganisms or invertebrates are adversely impacted by metals. Vegetation may be altered through reduced growth, early senescence of leaves, or death. In addition, metal-induced change in plant communities can indirectly affect wildlife owing to a reduced food supply or refuge. Wildlife may also be directly affected by metal-induced illness, decreased reproductive ability, or death.

Biological effects directly attributable to heavy metal pollution in Arctic terrestrial ecosystems mainly occur around smelters and mine sites. Studies show that high levels of metals around point sources can have significant impacts on soil microorganisms, soil invertebrates, and vegetation, although it can be extremely difficult to differentiate these effects from those caused by the high levels of SO_2 that generally accompany smelter emissions. In the absence of toxic SO_2 concentrations, damage to above-ground plant compartments is primarily due to the direct toxic effects of heavy metals. Heavy metals can also affect macronutrient availability (Derome and Lindroos, 1998) and nutrient uptake (Helmisaari *et al.*, 1999).

Other studies report on effects observed in regions of the Arctic where large point source emissions do not occur. These include the identification of effects on wildlife from diffuse sources (see sections 6.3 to 6.5).

6.2.1. Effect zones around smelters

Three zones of effect from smelter emissions have been defined for terrestrial ecosystems: the 'forest-death' zone, the 'visible-damage' zone, and the 'non-visible damage' zone. Terrestrial ecosystems in the forest-death zone surrounding smelters are completely destroyed (Figures 6·1 and 6·2). In this zone (extending 15 to 20 km at Nikel and Monchegorsk, and 80 to 100 km at Norilsk), trees and ground vegetation are dead, vertebrates and invertebrates are almost completely absent, soil microbial activity is extremely low, and the organic layer in many places is absent as a result of fire and erosion (Kozlov *et al.*, 1999). This is surrounded by the visible-damage zone (extending to 50 km at Nikel and Monchegorsk, and 200 km at Norilsk). In this zone, the trees suffer marked defoliation and a decline in growth (Kharuk, 2000; Kharuk *et al.*, 1996; Nöjd, 1996), needle-



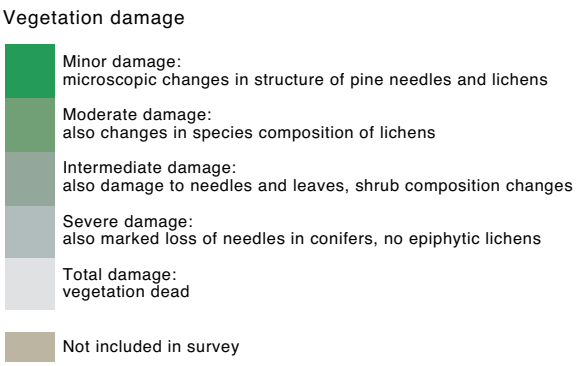
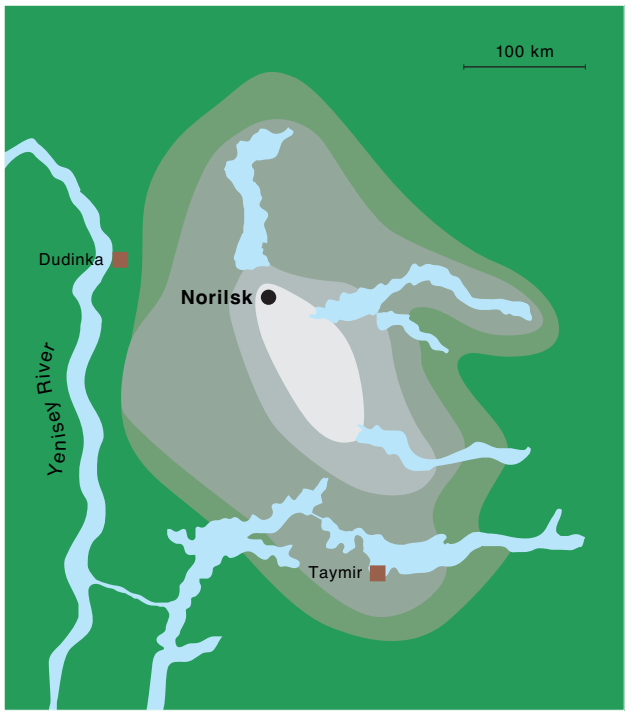
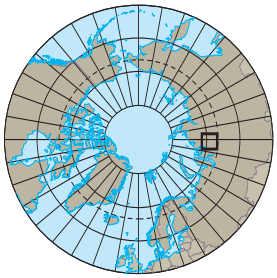


Figure 6·1. Extent of vegetation damage on the Kola Peninsula due to the combined effects of metals and acidifying substances.





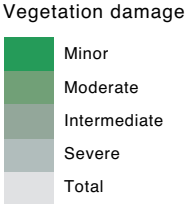


Figure 6·2. Extent of vegetation damage around Norilsk due to the combined effects of metals and acidifying substances.

tip necrosis (Salemaa *et al.*, 1992), reduced frost hardiness (Sutinen *et al.*, 1996), and reduced root biomass (Yarmishko and Yarmishko, 1993). Epiphytic lichens are absent from tree trunks and branches, and changes have occurred in the species composition and coverage of the ground vegetation, as well as in the chemical and microbiological properties of the soil. In the non-visible damage zone (extending about 150 km at Nikel and Monchegorsk), the effects are mainly restricted to changes in the physiological functioning and microscopic structure of plant tissues (Kharuk, 2000; Rigina and Kozlov, 2000; Tikkanen and Niemelä, 1995).

Biological effects directly attributable to heavy metal pollution are mainly restricted to the areas around Cu-nickel (Ni) smelters on the Kola Peninsula (northwest Russia; Figure 6·1), and the extremely large Cu-Ni smelter complex at Norilsk (northwest Siberia; Figure 6·2). In the boreal coniferous zone, similar damage has been reported around a Cu-Ni smelter at Sudbury in Canada (Winterhalder, 1995) and, on a relatively restricted

scale, at a Cu smelter at Sulitjelma in northern Norway (Løbersli and Steinnes, 1988), at a brass foundry at Gusum in southern Sweden (Bengtsson *et al.*, 1984), and at a Cu-Ni smelter at Harjavalta in southwest Finland (Helmisaari *et al.*, 1995).

6.2.2. Effects on soil processes, microorganisms, and vegetation

The main cause of forest decline and death in the vicinity of smelters is the direct toxic effect of SO_2 on trees and plants, combined with the inhibition of soil microbial activity, and the death of mycorrhizae and fine roots resulting from the accumulation of toxic levels of heavy metals in the soil. In addition, changes in the species diversity of saprophytic bacteria, algae, and fungi in the soil are thought to influence nutrient availability.

6.2.2.1. Soil processes and microorganisms

High concentrations of Cu and Ni in forest soils around the smelters on the Kola Peninsula may have caused the decrease in species diversity observed in non-spore-forming saprophytic bacteria, algae, and fungi (Evdokimova, 2000; Evdokimova and Mozgova, 1993). The proportion of fungi has increased, and complete functional groups of bacteria and cyanobacteria have disappeared (Kabirov, 1993; Lebedeva, 1993). The change in species composition and biodiversity of soil microorganisms resulting from elevated levels of metals alters the functions carried out by these organisms. Reduced microbial respiration has been observed in heavily polluted soils near smelters (Fritze *et al.*, 1996; Nordgren *et al.*, 1986). Respiration is a measure of the activity of the soil microbial community. Heavy metal accumulation in soil is also known to inhibit litter decomposition (Freedman and Hutchinson, 1980; Fritze *et al.*, 1989; Rühling and Tyler, 1974), resulting in the development of a thick layer of undecomposed material containing large amounts of plant nutrients such as nitrogen, calcium (Ca), and magnesium (Lukina *et al.*, 1993; Salemaa *et al.*, 2001). Inhibition of the mineralization of litter reduces nutrient availability for plant growth (Derome and Lindroos, 1998; Kabirov, 1993; Lebedeva, 1993), which further reduces the ability of vegetation to withstand the toxic effects of heavy metal accumulation.

6.2.2.2. Mycorrhizae and fine roots

Heavy metals can have direct toxic effects on roots and mycorrhizae (Colpaert and van Assche, 1992; Gobl and Mutsch, 1985; Helmisaari *et al.*, 1999), seriously reducing nutrient uptake. According to Yarmishko and Yarmishko (1993), the accumulation of heavy metals in forest soil around the smelters on the Kola Peninsula initially resulted in an increase in the formation of mycorrhizae, followed by a decrease in the longitudinal and diameter growth of the fine roots and, finally, by the damage and death of the mycorrhizae and fine roots. The growth of living roots in heavily polluted areas changes from the normal horizontal growth pattern to a horizontal form extending to a depth of 1 m. This is a strategy adopted by the root system to avoid the high heavy metal concentrations in the surface soil layers (Derome *et al.*, 1998; Kharuk, 2000; Lukina and Nikonov, 1995). Displacement of the active part of the root system to greater soil depth means that the trees are no longer able to access the pools of plant-available macro- and micronutrients in the upper soil layers, and may thus suffer nutrient deficiencies (Derome and Lindroos, 1998; Derome and Saarsalmi, 1999). Also, changes in the biomass of mycorrhizal fungi or shifts in community structure are likely to have important consequences for carbon cycling (Cairney and Meharg, 1999). However, the ability to predict the ecological consequences of observed pollution-driven change in mycorrhizal fungal communities is currently limited by a lack of understanding of the functional significance of mycorrhizal fungal diversity, particularly as there may be a degree of functional redundancy within diverse communities of mycorrhizal fungi (Allen *et al.*, 1995; Cairney and Meharg, 1999).

6.2.2.3. Ground vegetation

Sulfur dioxide and heavy metal emissions have caused dramatic changes in the ground vegetation over extensive areas around the point sources on the Kola Peninsula and at Norilsk (Lukina *et al.*, 1993; Rigina and Kozlov, 2000). Around one million hectares of forest and tundra have visible signs of damage (including dead forest) around the smelter complex at Norilsk, and around seven million hectares have damage that is not readily observed (Oleksyn and Innes, 2000). At Monchegorsk on the Kola Peninsula, the forest-death zone covers about 70 000 ha (Tikkanen and Niemelä, 1995). The species most sensitive to heavy metal deposition are those dependent on rainwater and snowmelt for their nutrient supply (i.e., lichens and mosses) (Gorshkov, 1993; Neshatayev and Neshatayeva, 1993), but grasses, forbs, and shrubs have also been severely affected (Deyeva and Maznaja, 1993; Lukina *et al.*, 1993; Rigina *et al.*, 1999). Some shrubs (e.g., crowberry; *Empetrum* spp.) are very resistant to heavy metal pollution and can grow close to smelters (Monni *et al.*, 2000), but overall the ground vegetation community diversity is severely reduced. Some of the resistant shrubs are hyperaccumulators, and retain high concentrations of metals (e.g., Cu and Ni) in their leaves (Reimann *et al.*, 2001a). However, in some areas even these species are negatively affected and show a distinct decrease in vitality (Monni *et al.*, 2000, 2001).

Lichens are a dominant component of the vegetation in the tundra ecosystem. The sensitivity of Arctic lichens to air pollution (Nash and Gries, 1995), together with their importance to the Arctic food web, has caused special concern about potential large-scale effects in Arctic regions. Reimann *et al.* (1998) compared damage to vegetation from the Kola smelter emissions and that due to centuries of reindeer/caribou herding, and suggested that herding was ecologically more problematic than smelter emissions. In the Nikel-Pechenga area, however, remote sensing coupled with ground truth measurements has shown that the decrease in lichen-dominated vegetation between 1973 and 1988 is related to a doubling of SO_2 emissions from local smelters (200 000 to 400 000 t respectively; Tømmervik *et al.*, 1995).

In areas remote from smelters, however, heavy grazing may be a significant factor in the degradation of reindeer/caribou ranges. A reduction in the lichen cover is most acute in areas where fences constructed since the Second World War restrict the movement of animals between traditional summer (coastal lowland) and winter (inland fjell) pastures (B. Forbes pers. comm., 2002). Along the Finnish/Norwegian border, the absence of lichen is more related to trampling than consumption, because the lichens of dwarf-shrub heaths are brittle and highly susceptible to breakage at the time reindeer move across the area to access riparian and wetland areas during the dry season. Lichen, when broken, is highly susceptible to wind erosion.

6.2.2.4. Trees

Forests around heavy metal point sources in the Arctic region exhibit a range of effects (see Section 6.2.1). As distance from the point source increases, there are changes in the physiological functioning and microscopic structure of plant tissues (Sutinen and Koivisto, 1992). Although tree damage and mortality are mainly caused by the direct toxic effects of elevated SO_2 concentrations, deposition and accumulation of heavy metals also contribute indirectly to forest damage and, in the long-term, prevent the recovery of forest ecosystems when emissions cease or are reduced. Damage to the trees and ground vegetation around a brass foundry in southern Sweden was attributed to the long-term accumulation of Zn and Cu in the forest floor (Tyler, 1984), and in southwest Finland to the accumulation of Ni and Cu (Helmisaari *et al.*, 1995).

Coniferous species (e.g., Scots pine (*Pinus sylvestris*), and spruces (*Picea abies* and *P. obovata*)) are the most sensitive to SO_2 and heavy metal emissions. Damage to the lowest branches of spruce is usually less severe owing to the protection provided by snow cover during the long winter (Kharuk, 2000). Species that shed their leaves (e.g., European white birch (*Betula pendula*) and willow (*Salix* spp.)) or needles (larches, e.g., *Larix russica* and the Dahurian larch *L. gmelinii*) during winter are more resistant to emissions. Larch, which grows extensively in the Norilsk region, can replace damaged needles and compensates for the loss of needles by producing needles along the stem and primary branches (Kharuk, 2000). Birch and willow are usually the last forest trees and shrubs to disappear from the most polluted areas (Kharuk, 2000; Kozlov, 1993).

Close to pollution sources, even the most resistant species are dead or severely damaged. Damage outside these areas depends on the prevailing wind direction, topography, and soil fertility. Stands on sheltered slopes and in river valleys, as well as those on more fertile sites, are considerably less affected (Kharuk, 2000). Additional stress factors, such as severe climatic conditions and exposure to high levels of solar radiation (e.g., UV-B) typical of the Arctic, increase the susceptibility of trees to damage.

Emissions from point sources also reduce sexual reproductive capacity and regeneration success, sometimes by as much as 50% (Daletzkaja *et al.*, 1993; Fedorkov, 1993). The number of pine seedlings in young (35- to 45-year old) and old-growth (250-year old) Scots pine stands on dry sites was lower at severely polluted sites (8 to 12 km from Monchegorsk) than unpolluted sites (75 km away). Also, the age distribution in the young stand was relatively even, whereas the youngest seedling age classes were absent from the old-growth stand (Stavrova, 1993).

6.2.3. Effects on invertebrates, birds, and mammals

High leaf Cu and Ni concentrations (>20 to 30 mg/kg) in birch reduced the density and survival of larval *Epirrita autummata* (Lepidoptera: Geometrida) in areas close to the Cu-Ni smelters at Nikel and Monchegorsk (Ruohomäki *et al.*, 1996). In another study at Monchegorsk during 1996 to 2000 (Zenkova, 2001), soil invertebrate biodiversity was reduced and species dominance patterns were altered. Groups of soil invertebrates with narrow ecological niches were entirely absent. Among the species missing were earthworms and molluscs, although all trophic groups had reduced numbers of species near the smelter.

Decreases were observed in the population density of microtine rodents with decreasing distance from the Monchegorsk smelter. These may have resulted from decreased lichen cover (see Section 6.2.2.3) and/or direct toxic effects of heavy metals in the soil (Kataev *et al.*, 1994).

Birds may avoid exposure to toxic levels of heavy metal pollutants on the Kola Peninsula and in the Norilsk region because the most severely affected areas do not have sufficient prey or forage. For example, areas close to the Monchegorsk smelter have significantly reduced populations of microtine rodents due to the decreased lichen cover and possible toxic effects of heavy metals (Kataev *et al.*, 1994). Thus, it is likely that raptors and other bird species that depend on rodents as a source of food do not remain in an area where rodents are not abundant. Research on avoidance behavior has not been completed in this region.

6.3. Effects of diffuse sources in terrestrial ecosystems

Heavy metal concentrations are relatively low in most areas of the Arctic environment (AMAP, 1998). Where higher concentrations are found in tissues of Arctic species they normally occur in species feeding at higher trophic levels such as birds and mammals. In a few cases, higher levels have been associated with observed effects in peregrine falcon (*Falco peregrinus*) and Arctic ground squirrel (*Spermophilus parryii*).

6.3.1. Peregrine falcon

Ambrose *et al.* (2000) measured heavy metal concentrations in the eggs of American and Arctic peregrine falcon (*Falco peregrinus anatum* and *F. p. tundrius*, respectively) in several areas of Alaska between 1988 and 1995. Copper, Fe, and Hg concentrations were significantly higher in the eggs of American peregrine falcon from unsuccessful nests (no young at the expected age of one to three weeks) compared to eggs from successful

Table 6·1. Comparison of heavy metal concentrations in the eggs of American and Arctic peregrine falcon from successful and unsuccessful nests. Geometric mean and range in concentration (mg/kg dw) between 1988 and 1995 (Ambrose *et al.*, 2000).

	American peregrine falcon		Arctic peregrine falcon	
	1988 to 1990 (n = 22)	1991 to 1995 (n = 31)	1988 to 1990 (n = 23)	1991 to 1995 (n = 19)
Copper *				
successful	2.9 (2.1 - 4.6)	2.3 (1.8 - 3.6)	2.6 (1.5 - 3.6)	2.5 (2.1 - 3.1)
unsuccessful	4.3 (2.7 - 6.8)	2.7 (1.9 - 3.7)	2.7 (1.7 - 4.3)	2.6 (1.7 - 3.9)
Iron *				
successful	103 (67 - 140)	67 (36 - 107)	92 (58 - 135)	62 (42 - 94)
unsuccessful	125 (81 - 207)	91 (41 - 174)	82 (42 - 140)	79 (28 - 163)
Magnesium				
successful	461 (358 - 582)	433 (198 - 885)	440 (335 - 601)	469 (378 - 602)
unsuccessful	448 (273 - 689)	413 (217 - 734)	450 (372 - 548)	387 (166 - 572)
Mercury *				
successful	1.50 (0.82 - 4.04)	1.75 (0.48 - 5.68)	2.06 (0.91 - 7.69)	1.77 (1.20 - 2.60)
unsuccessful	1.89 (1.06 - 3.19)	3.20 (1.10 - 9.58)	1.74 (1.48 - 2.25)	1.99 (1.32 - 3.12)
Zinc				
successful	44 (31 - 59)	34 (26 - 46)	40 (32 - 49)	31 (22 - 44)
unsuccessful	52 (35 - 90)	37 (24 - 55)	40 (25 - 79)	35 (30 - 44)

* significant ($p < 0.05$) difference between the concentrations in eggs from successful and unsuccessful nests for the American subspecies. No significant differences occurred between the concentrations in eggs from successful and unsuccessful nests for the Arctic subspecies.

nests (>1 young), while Mg and Zn were not (Table 6·1). Arctic peregrine falcon showed no significant differences in metal concentrations between eggs from unsuccessful and successful nests (Table 6·1). The number and percentage of peregrine falcon eggs exceeding critical Hg thresholds for egg survival (0.5 mg/kg) were 3 of 22 (13%) and 2 of 23 (9%) in 1988 to 1990 for American peregrine falcon and 10 of 33 (30%) and 6 of 20 (30%) in 1991 to 1995 for Arctic peregrine falcon (Peakall *et al.*, 1990).

Although these subspecies overwinter in similar locations in Central and South America, band returns indicate extreme site fidelity to specific breeding grounds in Alaska. The Arctic subspecies nests on the north slopes of the Brooks Range, the American subspecies in the Yukon River valley to the south of the Brooks Range. Elevated metal levels in the American subspecies probably reflect the natural enrichment of the Yukon River valley, and the effects of over 100 years of gold (Au) mining (A. Matz, pers. comm., 2001). Mercury was used as an amalgam in gold mining in the Yukon River valley until the mid-1900s.

No other metal exceeded threshold values for population effects. Concentrations of Cu, Fe, and Zn in eggs decreased between 1988 and 1995, while Hg levels increased (Ambrose *et al.*, 2000). Copper, Fe, and persistent organochlorine compounds may contribute to the reduced productivity observed in American peregrine falcon. However, since Hg levels in eggs exceed published thresholds for reproductive impairment, and levels have increased, Hg is of concern for peregrine falcon breeding in Alaska.

6.3.2. Arctic ground squirrel

Heavy metal concentrations were measured in Arctic ground squirrel at three sites in the Brooks Range between 1991 and 1993 (Allen-Gil *et al.*, 1997b). Heavy metals (e.g., As, Cd, Hg, Ni, Pb) were present in most of the livers collected (averaging <1 mg/kg ww). This indicates that small mammals are accumulating contaminants at considerable distances from point sources, although at levels likely to be below available toxicity thresholds.

6.4. Effects in freshwater ecosystems

The ecological impact of heavy metals in freshwater ecosystems is complex. Heavy metals are subject to many variables in freshwater systems that influence where metals accumulate, how they move in the substrate, and how they are taken up by biota. Freshwater communities are particularly sensitive to heavy metals and other pollutants (Ford, 1989). Dominated by short-lived species with high reproductive rates, exposure to heavy metals at toxic levels often results in changes in community structure and species richness. When exposure occurs, interactions between metals can exacerbate or alleviate biotic effects. For example, divalent cationic metals compete for binding sites on gills, resulting in additive effects (Playle, 1998). In addition, diet-borne exposure has varying interactive effects. Data from Maarmorilik (Elberling *et al.*, 2003) suggest that high Pb levels are less toxic if accompanied by high Zn levels, which is consistent with other studies suggesting that some metals have detoxification effects on heavy metal uptake in aquatic organisms (Mason and Jenkins, 1995). Zinc is one of the essential metals that have documented detoxification effects on Pb toxicity. Campbell (1995) discussed these detoxification effects and other interactions known to occur within aquatic organisms exposed to multiple trace metals.

Studies on the Kola Peninsula showed that the vulnerability of species in high latitude surface waters to the toxic impact of heavy metals from Cu-Ni smelters increased during spring (Moiseenko, 1999). This is due to

a pulse of ionic metal forms that enters surface waters during spring snowmelt. Moiseenko (1999) also showed the long polar winter to be an additional stressor, since a wide spectrum of metals is involved in the redox cycle in lower water layers under the eutrophic and oxygen deficient conditions that exist during the long, dark period. Such cycling may increase rates of bioaccumulation in Arctic fauna (Moiseenko, 1999).

This assessment contains brief descriptions of studies on effects in Arctic species. These include unicellular plants (algae, plankton, and diatoms), invertebrates, fish, and mammals. Effects data are not available on metals in macrophytes (submerged and emergent) or on obligate freshwater species of birds.

6.4.1. Algae and invertebrates

In a study conducted in southern Norway, changes were seen in algal communities at Zn concentrations as low as 20 to 30 µg/L (Hylland *et al.*, 1998). Benthic diatoms appeared to be the most sensitive group. A plankton photosynthesis bioassay performed with sediment eluate from different Mackenzie River Delta locations showed no effects of low Cd levels on photosynthesis in aquatic algae (Nalewajko, 1995). Cadmium concentrations in the sediments were below the level reported to have direct toxic effects on algae.

Hylland *et al.* (1998) found changes in the species composition of communities of benthic (sediment) larvae when Zn concentrations exceeded 200 to 300 µg/L in the water of a mountain brook in southern Norway.

6.4.2. Fish

Freshwater fish are key biomonitors for contaminants due to their importance in natural food webs and their role in the human diet. Owing to their sensitivity to pollutants in surface waters they are also frequently used as test organisms for establishing toxicity thresholds. However, most studies on fish toxicity are experimental laboratory studies. Few Arctic species have been evaluated for the effects of heavy metal exposure, particularly in terms of the more subtle effects. Those that have been studied include European grayling (*Thymallus thymallus*), Arctic char (*Salvelinus alpinus*), walleye (*Stizostedion vitreum*), lake trout (*Salvelinus namaycush*), pike (*Esox lucius*), perch (*Perca fluviatilis*), and salmon (*Oncorhynchus* spp.). No studies have wide spatial coverage.

Fish pathologies related to heavy metal exposure are well established for some Arctic species. Moiseenko (1999) found that increased risk of disease and decreased fitness resulted from atmospheric deposition and the indirect leaching of heavy metals to surface waters from localized mining and metallurgical activities (e.g., locations in the Kola Peninsula, Russia).

6.4.2.1. Bioavailability, uptake and bioaccumulation

The bioaccumulation of metals in Arctic freshwater biota is highly variable and influenced by species characteristics, feeding habit, trophic position, age, and sex. Bioaccumulation is also influenced by seasonal changes, lake or river biogeochemistry (e.g., alkalinity, pH), the presence of other contaminants, humic substances and other complexing agents, and for Hg, the rate of methylation.

Allen-Gil *et al.* (1997a) and Allen-Gil and Martynov (1995) analyzed tissues from fish collected from Arctic lakes and rivers to try to relate metal concentrations in water and sediment to concentrations in fish. With the exception of Hg, the correlation between metal concentrations in sediment and freshwater fish tissue for four Alaskan lakes was not consistent (Allen-Gil *et al.*, 1997a). A number of physical, chemical, and physiological parameters appeared to mediate metal bioavailability and uptake in these systems. In particular, diet and trophic level, as evidenced by the higher concentrations of some metals in lake trout than Arctic grayling (*Thymallus arcticus*), and differences between lakes for lake trout. Cadmium, Cu, and Zn tissue burdens were higher in trout from lakes where snails were consumed, than in lakes without snails. Other researchers have found that in oligotrophic alpine lakes, water temperature appears to drive metal accumulation in fish, and that there is a positive correlation between fish age and Cd and Pb concentrations in liver and kidney (Köck *et al.*, 1996; see also Section 4.5.4.4).

Moiseenko and Kudryavtseva (2001) studied the impact of acidified conditions on the bioaccumulation of a wide range of metals by fish in lakes of the northern Kola region of Russia. Aluminum (Al), Ni Cu, Pb, and Cd appeared to be more actively accumulated under acidified conditions. This increased toxic effects. The levels of bioaccumulation found enabled fish to be used as indicators of atmospheric loads in the Kola region.

6.4.2.2. Effects in Arctic freshwater fish

The effects of heavy metals in Arctic fish species are varied and differ for species and age groups. Based on laboratory tests, the relative toxicities of metals to Arctic salmonid species (European grayling, coho salmon (*Oncorhynchus kisutch*), and rainbow trout (*O. mykiss*)) varied by up to four orders of magnitude. Ranging from most toxic to least toxic, the rank order is: Cd, Ag, Hg, Ni, Au, arsenite, selenite, selenate, and hexavalent chromium. In general, juvenile life stages are more sensitive than the alevin life stage, and among juveniles, no single species was consistently more sensitive to the metals than another. However, among alevins, Arctic grayling was more sensitive than coho salmon and rainbow trout (Buhl and Hamilton, 1991).

Using juvenile walleye, a species commonly found in North American lakes, Friedmann *et al.* (1996) showed that low levels of dietary MeHg reduce survival by impairing growth and immune function. Furthermore, their results suggest that MeHg might also affect reproductive potential of teleosts by impairing testicular development in young fish. Arctic grayling embryos exposed to water with elevated MeHg concentrations during egg development showed a slight increase in morphological abnormalities (Vollestad *et al.*, 1998). Cutthroat trout (*O. clarki*) exposed to various heavy metals through their diet show reduced feeding activity, changes in immune function, and kidney pathology (Farag *et al.*, 1999). Farag *et al.* (1999) considered that these effects would probably reduce growth and survival of fish in the wild, particularly in early life stage fish whose diet

consists wholly of benthic macroinvertebrates. Collectively, these studies suggest that Arctic freshwater fish species may be adversely affected if heavy metal concentrations in the water column are elevated.

Kislalioglu *et al.* (1996) and Scherer *et al.* (1997) studied Cd-induced change in foraging behavior of lake trout. Cadmium-exposed fish made fewer attacks and consumed fewer prey than unexposed fish. Furthermore, Hansen *et al.* (1999) demonstrated that exposure to low levels of Cd in the water caused histopathological changes in the olfactory system of chinook salmon (*O. tshawytscha*), effectively eliminating their ability to detect and avoid metal-contaminated waters. This indicates that chinook salmon have a low response threshold for Cd-induced behavioral change. However, closely related species such as rainbow trout have been shown to accumulate high concentrations of Cd in the wild (Kislalioglu *et al.*, 1996), suggesting that these species still forage effectively.

In addition to a potential behavioral avoidance of metal-contaminated waters, some fish populations have adapted to high environmental concentrations of metals by increasing the amount of MT available for binding and subsequent storage. Gerpe *et al.* (1998, 2000) studied MT gene expression in Arctic char and found Cd-induced MT mRNA in the liver, kidney, and brain. Dallinger *et al.* (1997) found that MT accounted for the sequestration of virtually all Cd, and considerable proportions of Cu and Zn, found in the liver and kidneys of Arctic char from high alpine lakes. Dallinger *et al.* (1997) also found significant, positive correlations between age and hepatic concentrations of MT in Arctic char.

6.4.3. Mammals

No new work on Arctic freshwater mammals is included in this assessment. However, river otters (*Lontra canadensis* (= *Lutra canadensis*)) in the Prince William Sound watershed showed significantly elevated Hg concentrations in liver. Location (naturally contaminated site versus reference site), diet, and age were found to influence the extent of contamination (Ben-David *et al.*, 2001). The elevated levels are probably due to a diet of intertidal fish with naturally elevated Hg concentrations. No changes in survival rates of otters inhabiting the contaminated area were detected.

6.5. Effects in marine ecosystems

The Arctic marine environment is a significant repository of heavy metals. Most data concern effects in fish, birds, and mammals. Although bacteria and unicellular organisms are essential components of nutrient and carbon budgets in pelagic systems, there is limited information on the extent to which they are affected by metals in the Arctic marine environment. There are no data on the effects of metals on marine vegetation. Information on the relative sensitivity of Arctic systems compared to subarctic or temperate systems is also not available. Recent research highlights the importance of trophic level on the potential for bioaccumulation. Observed effects in marine species, however, are not always consistent with measures of exposure.

6.5.1. Primary producers

The main primary producers in marine ecosystems are algae. Some algal species are very sensitive to metals, especially Cu. However, there are few data on effects induced in primary producers in Arctic marine systems. Since ice algae rather than phytoplankton form the base of the food chain in far northern seas, contaminant pathways probably differ significantly from those in other regions (Alexander, 1995). Contaminants deposited on sea ice can be transported directly to the ice/seawater interface, where the ice algae occur, via brine channels or spring melt water. Metal concentrations in open ocean waters are thought unlikely to affect primary production, although sufficiently high levels may occur in estuaries or near point sources.

6.5.2. Invertebrates and the benthos

Larsen *et al.* (2001) investigated Pb and Zn levels in sediments and biota from fjords surrounding the Pb/Zn mine at Maarmorilik, West Greenland, to evaluate the impact of waste rock and marine-deposited tailings on marine biota. Metal concentrations in sediment were up to 8900 ± 620 mg/kg dw for Pb and 19 300 ± 476 mg/kg dw for Zn. Lead and Zn were also elevated in a range of organisms, including *Fucus distichus* (a seaweed), *Littorina saxatilis* var. *laevigata* (a herbivore), *Mya truncata* and *Musculus discors* (suspension feeders), *Gammarus* spp. (a scavenger), *Macoma calcarea* (a deposit feeder), and *Pholis fasciatus* (a carnivorous fish). Trophic level appeared to explain the sources of metals to the organisms. Metal concentrations in the upper centimeters of sediment decreased following mine closure, although marine biota in the area continued to have elevated tissue concentrations. Despite these high metal levels, effects were not detected in any biota.

A variety of marine invertebrates were examined for their potential as biomonitors of trace metal pollution. Regoli *et al.* (1998) sampled the clam *Macoma balthica* at seven stations in the White and Pechora Seas to define baseline concentrations of As, Cd, Cr, Cu, Fe, Mn, Pb, and Zn and to evaluate antioxidant responses as biomarkers of anthropogenic stress. Their results indicate the suitability of clam antioxidant responses as indicators of pollutant exposure in the Arctic. Regoli *et al.* (2000) extended this work to three species of scallop and concluded that they were also suitable sentinel species.

Foraminifera occur in large numbers (>100 individuals per 100 g of sediment) in most marine environments. Several species are known to be very sensitive to some environmental factors and this is reviewed by Murray (1991). Calcium carbonate shell material survives burial under most natural conditions and can be extracted from cores to evaluate temporal trends in naturally-occurring metal accumulation (Murray, 1991) or anthropogenic pollution (e.g., Alve, 1995a, 1995b, 1999; Nagy and Alve, 1987; Yanko *et al.*, 1999). Studies of pollution-induced effects include morphological abnormalities as well as change in species composition (Alve, 1995a; Ellisson *et al.*, 1986; Nagy and Alve, 1987; Samir and El-Din, 2001). In areas of heavy metal pollution, deformed specimens have been shown to contain

higher levels of heavy metals than non-deformed specimens (Samir and El-Din, 2001).

Species of zooplankton from the Greenland Sea were evaluated for use as biomonitors (Ritterhoff and Zauke, 1997). The amphipods *Themisto libellula* and *T. abyssorum* and the copepod *Calanus hyperboreus* had substantially different concentrations of metals between juvenile and adult life stages, suggesting that adults should be used in routine biomonitoring studies.

Ritterhoff and Zauke (1998) investigated the potential role of metal-binding proteins in amphipods (*T. libellula* and *T. abyssorum*) from the Greenland Sea. Results were inconclusive as only 66% of the total Cd could be found in insoluble fractions. Based on tissue samples of *Macoma balthica* and blue mussel (*Mytilus edulis*) from several estuaries in the White and Pechora Seas, these authors concluded that biological factors (e.g., size of sampled organisms) influenced MT concentration more than Cd exposure. Thus, biological factors should be considered when assessing the implications of MT measurements (Amiard-Triquet *et al.*, 1998). As storage and excretion of non-essential metals follow different paths in different species, generalizations are difficult.

Hutchins *et al.* (1996) examined the effects of temperature on the uptake and retention of heavy metals by the brittle star *Ophiothrix fragilis*, which is typical of species that dominate Arctic benthic communities. Lower temperatures significantly reduced the rate of uptake for all elements considered, but had little effect on loss rates. This raises questions about the validity of extrapolating to polar ecosystems, studies on biological dynamics of dissolved contaminants largely carried out at temperatures experienced in temperate zones. Hutchins *et al.* (1996) suggested that the effects of low Arctic temperatures may need to be investigated in order to understand food web accumulation of toxic metals in high-latitude seas.

6.5.3. **Fish**

Although studies are few, metal concentrations in Arctic marine fish are generally lower than in marine fish from temperate areas. Hepatic Cd concentrations in Atlantic cod (*Gadus morhua*) from Varanger and Svolvir, Norway, were monitored from 1995 to 2000 (Hylland and Green, 2001); concentrations at Varanger were generally < 0.2 mg/kg ww and fairly consistent across years, while those at Svolvir were generally < 0.1 mg/kg ww (except for 1996 and 2000). Preliminary results indicate a trend towards lower δ-aminolevulinic acid dehydratase (ALA-D) activity in fish with higher hepatic (and presumably blood) Cd concentrations (Hylland and Green, 2001). ALA-D is also sensitive to Pb exposure. The activity of ALA-D in blood cells of cod from northern Norway was close to or at the levels viewed as normal for cod, so there was no obvious impact of Pb on Atlantic cod from this area. However, results were inconclusive.

6.5.4. **Seabirds**

Effects of heavy metal exposure have been identified in seabirds. Diet is probably the main factor influencing exposure and bioaccumulation (Monteiro *et al.*, 1998; Stewart *et al.*, 1997). Although Stewart and Furness (1998) found insufficient evidence to show that either Cd or Hg concentrations increase with age in seabird liver or kidney, seabirds can accumulate high levels of both (Muirhead and Furness, 1988; Stewart and Furness, 1998; Walsh, 1990), which has been correlated with cellular kidney damage in some pelagic seabird species (Nicholson and Osborn, 1983).

Elevated blood and tissue Pb levels, morbidity, and mortality from Pb poisoning have been reported for spectacled eider (*Somateria fischeri*) and common eider (*S. mollissima*) on the Yukon-Kuskokwim Delta (Flint *et al.*, 1997; Franson *et al.*, 1995a,b). The proportion of spectacled eider hens exposed to Pb increased over the breeding season. Low overwinter survival together with decreased survival of Pb-exposed young may influence population dynamics (Flint *et al.*, 1997).

Steller's eider (*Polysticta stelleri* (= *S. stelleri*)), sea ducks that breed on Arctic and subarctic coastal plains and overwinter in subarctic nearshore waters, are considered threatened in Alaska. Causes of recent breeding population declines may include exposure to environmental contaminants, particularly Pb from lead shot (Flint and Herzog, 1999). One specimen found dead near Barrow in 1993 had liver and kidney Pb concentrations that suggested Pb poisoning had been the cause of mortality (Trust *et al.*, 1997). Whole blood samples from seven incubating hens trapped in 1999 near Barrow all had Pb concentrations (0.689 to 6.03 mg/L) above waterfowl toxicity thresholds for exposure (Locke and Thomas, 1996). Because hens were incubating, they had been on the breeding area in approximately the same location for three to four weeks thus suggesting 'local' exposure. All were above waterfowl toxicity thresholds for Pb exposure (Locke and Thomas, 1996). Lead levels in liver for three birds found shot at Barrow in 1999 included two collected in June with concentrations below detection and 0.32 mg/kg ww respectively; while the third collected in July had a concentration of 0.72 mg/kg ww. Thus, 10 of 11 birds sampled in the Barrow area had Pb levels well above 'background' (0.2 mg/kg ww). More data are needed to determine the significance of Pb to breeding Steller's eider in Barrow. Studies of spectacled eider along the Kashunuk River, Yukon Delta, demonstrated that the likelihood of exposure to lead shot increased with time spent on or near tundra pond breeding areas where significant hunting with lead shot occurs (Flint *et al.*, 1997). Franson *et al.* (1998) noted that in spectacled eider, an overall increase in mean blood Pb concentration over time could be attributed to an increase in the number of females exposed to lead shot, and an increase in the proportion of females with blood Pb levels above detection but still at background levels. More recently, Grand *et al.* (2002) sampled female spectacled and common eider on the Yukon-Kuskokwim Delta, and found that the exposure rates of females sampled at early incubation were already elevated suggesting that most exposure occurred in spring prior to egg laying when females feed extensively. However, the overall exposure rate of spectacled eider females peaked at 37% during brood rearing (Flint *et al.*, 1997), which appears to be well below the exposure rate of the small sample of Steller's eider near Barrow.

Henny and Rudis (1995) reviewed nesting sea duck population declines in Alaska over recent decades, exploring the possibility that contaminants were involved. Aerial surveys of surf scoter (*Melanitta perspicillata*), white-winged scoter (*M. fusca*), black scoter (*M. nigra*), long-tailed duck, spectacled eider, and Steller's eider showed long-term breeding population declines, especially the latter three species. Of these, both the Steller's eider and the spectacled eider are now considered threatened in the United States. Large die-offs of the three scoter species during molt, a period of high-energy demand, were documented in August 1990, 1991, and 1992 at coastal reefs in southeast Alaska. There was no evidence of infectious disease. Many scoters had elevated renal concentrations of Cd (e.g., a high of 375 mg/kg dw). Selenium concentrations in liver of nesting white-winged scoter were high. However, eggs contained less Se than expected. Histological analysis showed a high prevalence of hepatocellular vacuolation (49%), a degenerative change frequently associated with sublethal toxic insult. However, Henny and Rudis (1995) were unable to establish a relationship between metals (or other contaminants) and population declines in sea ducks.

Wayland *et al.* (2001) found that levels of Hg, and to some extent Cd, were negatively correlated with body weight and abdominal fat in sea ducks. A positive correlation was found between nematode infection and Hg concentration in common eider, possibly reflecting individual feeding preferences. However, relationships between tissue metal levels (Cd, Hg) and immune responses could not be found.

6.5.5. Marine mammals

Marine mammals bioaccumulate heavy metals to levels that would probably cause effects in many species. However, detecting effects in marine mammals is particularly challenging.

6.5.5.1. Bioaccumulation in marine mammals

Heavy metal bioaccumulation in Arctic marine food webs has been studied using stable carbon ($\delta^{13}C$) and nitrogen ($\delta^{15}N$) isotopes for species feeding at different trophic levels (L. Dehn, unpubl. data, 2002). Liver, kidney, and muscle tissue were collected from marine mammal species including top level consumers (e.g., beluga; *Delphinapterus leucas*), intermediate consumers (e.g., ringed seal; *Phoca hispida*), and low level consumers (e.g., bowhead whale; *Balaena mysticetus*). Prey items were also sampled. Isotopic evidence confirmed trophic status. Mercury and Se accumulated in kidney and liver with age and trophic level. The assumed Hg vector was fish because seals and beluga feed on fish. Bowhead whale showed the lowest concentration of Hg, consistent with a diet of krill. Hepatic Se followed the same trend as MeHg, although renal Se showed no biomagnification with trophic level. Cadmium in kidney and liver accumulated with age, and appeared to indicate a diet of crustaceans or shellfish. Cadmium concentrations were higher in seals and bowhead whale than in top level consumers, which is consistent with diets containing invertebrates such as krill (L. Dehn, unpubl. data, 2002).

6.5.5.2. Effects in marine mammals

Steller sea lions (*Eumetopias jubatus*) are upper trophic level predators that are classified as endangered in the western Alaska part of their U.S. range (Loughlin *et al.*, 1992) and threatened but increasing in southeast Alaska (Calkins *et al.*, 1998). Many factors are likely to have been responsible for the declines observed over the last 30 years (Calkins *et al.*, 1998; Merrick, 1995; Pitcher *et al.*, 1998), including environmental contaminants (Merrick, 1995; Merrick *et al.*, 1987). Porphyrins have been used as biomarkers for potential Hg (and polychlorinated biphenyl) exposure in Steller sea lions (Beckmen *et al.*, 2001a,b,c). Preliminary data for fecal composites from rookeries of both the western and eastern populations (Beckmen *et al.*, 2001b) show detectable levels of all six porphyrins in samples from the eastern population. In contrast, three of the six porphyrins were not detected in some samples from the western population. Mercury concentrations in sea lion hair were inversely correlated with fecal porphyrins. Total Hg analysis showed significantly higher exposure in Prince William Sound sea lion pups than in pups from southeast Alaska. Mercury levels in the fur of northern fur seal (*Callorhinus ursinus*) were significantly higher than in fur from Steller sea lion from the Pribilof Islands (Beckmen *et al.*, 2002). Total Hg concentrations in fur of Steller sea lion pups from Prince William Sound and the Pribilof Islands ranged from 0.90 to 3.14 mg/kg ww and in juveniles from 0.56 to 6.75 mg/kg ww in 1998 and 2000. In contrast, northern fur seal pups had concentrations of 3.15 to 8.14 mg/kg ww in 2000.

Cadmium concentrations in kidney of ringed seal from northwest Greenland are among the highest recorded Cd tissue concentrations in Arctic marine mammals, reaching levels that may induce renal histopathological changes. In a study of 100 ringed seals from the Qaanaq area, renal histopathological changes were found in ten. However, none of the changes were consistent with Cd-induced renal damage. The three lowest lumbar vertebrae were scanned to measure the Ca content to evaluate possible demineralization. No correlation was found between the degree of skeletal mineralization and Cd concentration, renal damage, age, or sex (Sonne-Hansen *et al.*, 2000).

Contaminant and histopathological analyses were performed on several marine mammal species obtained in conjunction with the subsistence harvest in Barrow, Alaska. Tissues from five bowhead whales, 24 beluga whales, and 17 ringed seals were examined, with particular attention given to lesions consistent with chronic heavy metal toxicosis (Woshner, 2000; Woshner *et al.*, 2001a, 2001b). Cadmium, Hg, Ag, and Zn occurred in some tissues at concentrations that would be considered toxic in domestic animals. However, all the subsistence-harvested whales and seals had good body condition and an absence of lesions consistent with chronic heavy metal toxicosis, despite concentrations of Hg, Cd, and Se that were elevated relative to normal concentration ranges for domestic terrestrial species. In conjunction with the histopathological assessment, a semi-quantitative autometallographic (AMG) histochemical technique was used to localize inorganic Hg in light microscopic sections of bowhead and beluga liver (Woshner *et al.*,

2002). No AMG staining occurred in bowhead liver, consistent with the extremely low concentration of total Hg determined by chemical analyses (0.011 to 0.038 mg/kg ww). In beluga, total Hg ranged from 0.30 to 17.11 mg/kg ww and 0.33 to 82.47 mg/kg ww in liver and kidney, respectively. In beluga liver, AMG staining was restricted to specific cellular locations, which implies that Hg was confined to lysosomes and that binding to vital organelles is minimal and that Hg-induced toxicosis is not likely.

A histopathological study has also been undertaken on kidney tissue obtained from five Atlantic white-sided dolphin (*Lagenorhynchus acutus*) (two immature and three mature) from the Faroe Islands (Gallien *et al.*, 2001). Cadmium was not detected in the immature animals and ranged from 22.7 to 31.1 mg/kg ww in the three mature dolphin. Mercury concentrations ranged from 0.1 to 2.5 mg/kg ww, with lower concentrations in immature dolphin. There were kidney abnormalities in two of the three mature animals, as well as calcium phosphate concretions containing Cd in the kidney tissue of these animals. Metal-containing granules have also been observed in liver of marine mammals and birds and in the respiratory tract of cetaceans. In invertebrates, these granules are for the storage and immobilization of Cd and/or for its detoxification. Further study is required on the long-term effects of Cd exposure in marine top trophic level consumers.

6.6. Toxicity thresholds for effects

Most organisms have mechanisms to detoxify, sequester, or eliminate toxic substances. These are protective at low levels of exposure since organisms are able to regulate their internal dose across a range of exposure levels. As exposure increases, regulatory mechanisms can be overwhelmed and toxicity occurs, leading to effects. Metals have different dose-response curves depending on the type of metal, the organism, and mechanisms of action. Metals that are essential micronutrients tend to have a biphasic dose-response curve; with deficiencies occurring when levels are too low and toxicity occurring when levels are too high. Other metals, such as Hg, Pb, and Cd that have no known biological benefit show only toxicity thresholds.

This section summarizes information on heavy metal concentrations in the Arctic environment and biota, and compares these levels to pre-existing toxicity thresholds established for heavy metal exposure in different media and organisms. Threshold data are used to identify levels of exposure that can be expected to result in biotic effects. It is recognized that existing toxicity thresholds and acute and chronic criteria may be insufficient to identify all potential effects of heavy metal exposure. Despite uncertainties associated with toxicity thresholds, they do provide an accepted approach for identifying concentrations of concern and biota at potential risk. Background information and sources for the toxicity thresholds are provided in Appendix 6.1. Detailed information concerning the data summarized in this chapter may be found in the relevant sections of Chapter 4 and in Annex Tables A1 to A16.

Extrapolating from data on non-Arctic species to Arctic species living under very different environmental conditions introduces significant uncertainty. Comparisons between current environmental concentrations in the Arctic to effects thresholds must be undertaken with caution, and are presented as benchmarks for identifying potential research priorities only, not as evidence of effects. Appendix 6.2 includes more discussion on the uncertainty associated with establishing thresholds, provides a description of data conversions, and shows how Figures 6·3, 6·4, and 6·5 (see next spread) were derived.

Thresholds and data on environmental and tissue concentrations of Hg, Pb, and Cd are presented in Figures 6·3, 6·4, and 6·5 respectively. These illustrations are supplemented by the summaries of available data on organisms living in the terrestrial, freshwater, and marine environments presented in Section 6.6.1, 6.6.2, and 6.6.3. While the figures are organized by metal, the text is organized first by ecosystem type and then by metal in the order: Hg, Pb, and Cd, to provide an alternative way of viewing the information.

6.6.1. Terrestrial ecosystems

Thresholds for toxicity in terrestrial Arctic ecosystems are available for soil microorganisms, invertebrates, birds, and mammals.

6.6.1.1. Soil microorganisms and invertebrates

Recent studies show that accumulations of Hg and Pb, especially in the forest topsoils of Sweden, are already above levels at which soil respiration and decomposition of mor layers are adversely affected (Bringmark and Bringmark, 2001a,b; Johansson *et al.*, 2001). Because such studies report that Hg concentrations are still increasing in the forest soils of northern Scandinavia, as well as in other parts of the Arctic region, there is an increasing risk of effects on soil processes and biotic communities in the Arctic region.

Of the 45 soil Pb concentrations reported in the first AMAP assessment, none were above the benchmark for protection of plants (50 mg/kg dw) or invertebrates (500 mg/kg dw). The highest value was 26.3 mg/kg dw, located in Russia. The highest soil Hg concentration, also in Russia, was 0.25 mg/kg dw, well below the benchmark of 5 mg/kg dw for the protection of soil organisms. This is shown in Figure 6·4.

6.6.1.2. Birds

Mercury concentrations in the kidney, liver, and eggs of birds other than waterfowl are well below toxicity thresholds (Figure 6·3).

The first AMAP assessment reported Pb concentrations in various tissues for a number of terrestrial bird species. None were above toxic threshold concentrations. For Falconiformes, only feathers were assessed for Pb concentration, for which there are no toxicity-related threshold values (feathers, however, are useful for indicating that exposure occurred sometime during the last molt). In the Galliformes, three values for ptarmigan liver were indicative of sublethal Pb effects, and one indicated severe Pb poisoning.

In the first AMAP assessment, 2 of 148 plant tissue Cd concentrations were above the avian (chicken) dietary

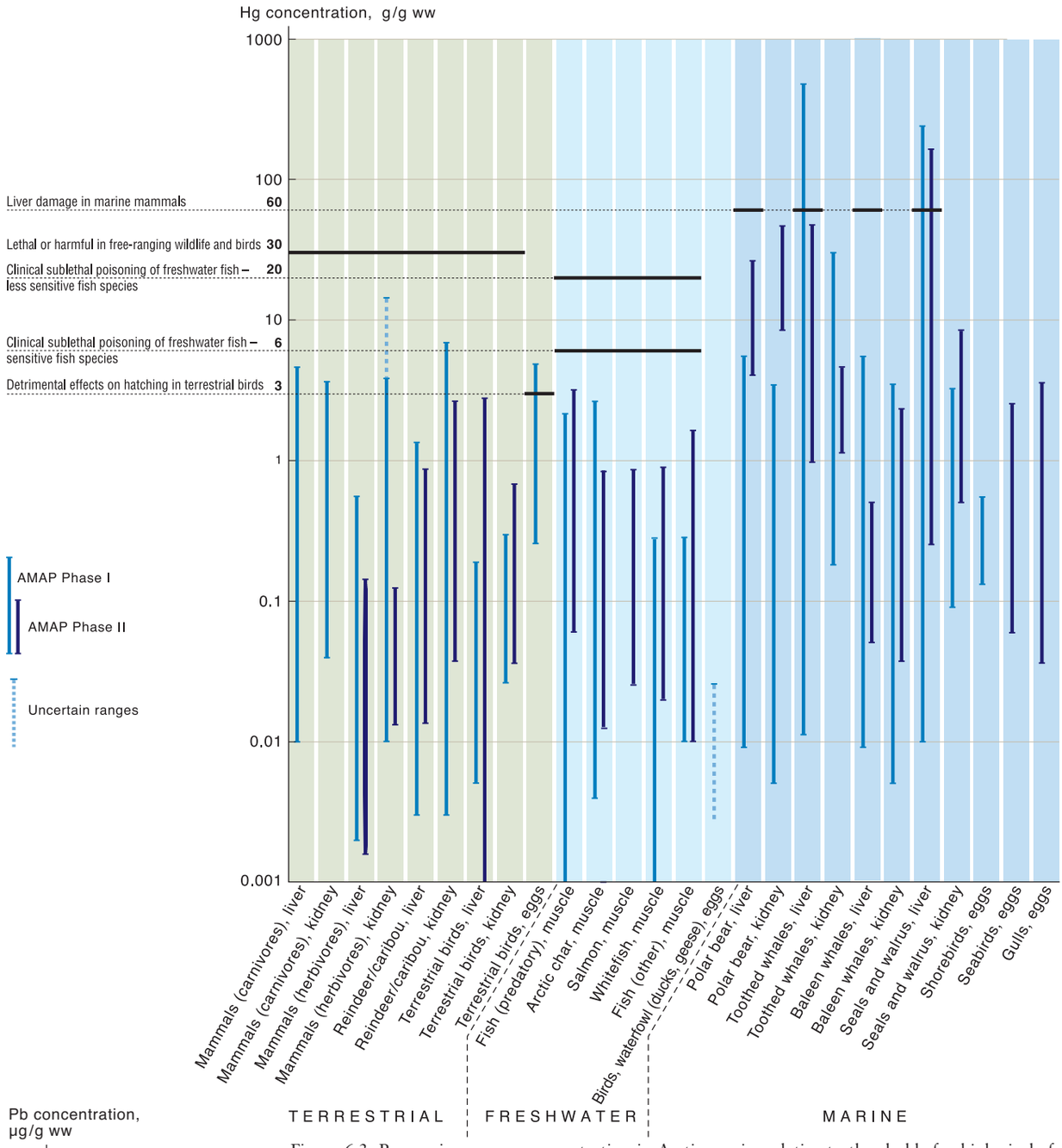


Figure 6·3. Ranges in mercury concentration in Arctic species relative to thresholds for biological effects. The comparison should be used with caution owing to problems with extrapolating data across species. For references to thresholds see Table 6·2.

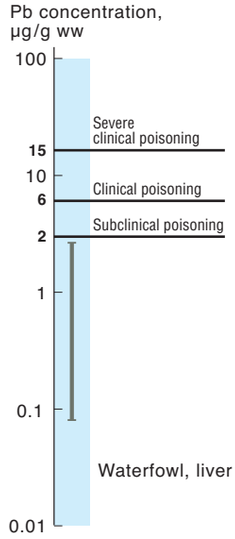


Figure 6·4. Ranges in lead concentration in Arctic species relative to thresholds for biological effects. The comparison should be used with caution owing to problems with extrapolating data across species. For references to thresholds see Table 6·2.

threshold of 12 mg/kg dw. However, other birds, including quail and ducks, show reduced egg production at dietary levels of about 65 to 70 mg/kg ww. One plant sample from Russia had a Cd level of 140 mg/kg dw, the only sample out of eight above the 12 mg/kg dw threshold. The next highest concentration (22 mg/kg ww) was found in plants from Norway. Figure 6·5 shows Cd kidney and liver concentrations in birds other than waterfowl, and liver concentrations for waterfowl, in relation to tissue thresholds.

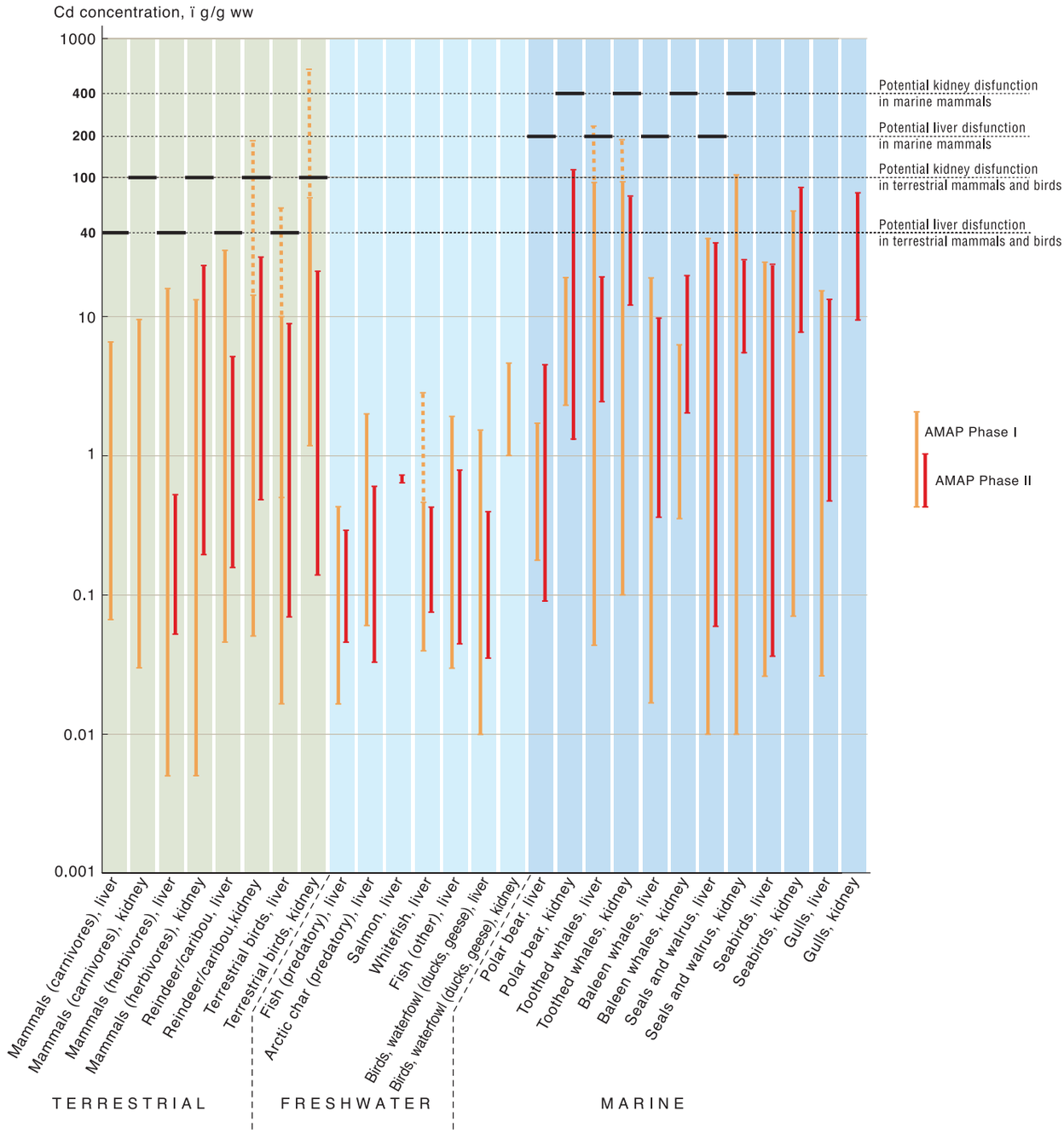


Figure 6·5. Ranges in cadmium concentration in Arctic species relative to thresholds for biological effects. The comparison should be used with caution owing to problems with extrapolating data across species. For references to thresholds see Table 6·2.

6.6.1.3. **Mammals**

Of 169 mammalian liver Hg concentrations, none were above the toxicity threshold. All were collected from Swedish reindeer. Likewise, none of the 100 concentrations of Hg in plants were above the dietary threshold of 2 to 6 mg/kg dw (based on MeHg) for most mammals recommended by the National Research Council (NRC, 1980). Feather moss from Greenland had the highest Hg concentration, at 0.22 mg/kg dw.

Lead levels consistent with subclinical effects (i.e., effects only detectable through histopathological evaluation of organs) were found in several species, including reindeer/caribou, moose, muskrat, squirrel, and shrew. Only one sample from a squirrel suggested the possibility of clinical Pb poisoning (37.2 mg/kg dw in a ground squirrel from Canada; AMAP, 1998). All data reported for the present assessment on Pb in reindeer/caribou liver (e.g., Figure 4·32) fell within the normal range.

The highest liver Cd concentration occurred in shrew (3.7 mg/kg dw), well below the threshold value of 40 mg/kg ww. None of the 11 values reported in the first AMAP assessment or in the present assessment are above this threshold. As consumers of soil invertebrates, shrews are likely to have the highest tissue concentrations of metals, especially Cd. Of 183 values reported in the first AMAP assessment only three were above terrestrial mammal thresholds for Cd in liver (40 mg/kg ww) and kidney (100 mg/kg ww).

6.6.2. Freshwater ecosystems

Toxicity thresholds for metals in freshwater ecosystems are available for water, sediments, and fish. Some new data on concentrations in freshwater are above some water quality criteria. This is probably due to the use of total rather than dissolved concentrations.

6.6.2.1. Freshwater

In the first AMAP assessment one of 26 water samples had Hg concentrations above the freshwater criterion for chronic exposure (0.77 µg/L dissolved Hg; see Table 6·3). The highest values were found in Russia, although the two highest values (0.1 and 0.066 µg/L) were below the threshold. New Hg data for this assessment includes a high value of 0.069 µg/L also from a Russian river (Melnikov *et al.*, 2002).

Concentrations of Pb in Alaskan ponds and lakes were <40 to 50 µg/L in 1988 and <15 to 470 µg/L in 1989, some of which were substantially higher than the chronic freshwater criteria for dissolved Pb. Some samples also exceeded the higher acute water quality criteria for dissolved Pb, suggesting the potential for substantial impact on the aquatic ecosystem. Of the 16 Pb concentrations for Canada, none were above chronic water quality criteria; even assuming the concentrations reported were for the dissolved fraction. Likewise, in Finland, neither of the two concentrations was above available criteria. In Russian rivers, one out of nine reported values was above chronic criteria, and only a few samples from 18 wetlands had water concentrations above the chronic value (range: >2.5 to 6.1 µg/L; AMAP, 1998). No samples from three rivers in Russia had Pb concentrations above the chronic criteria value.

Cadmium levels in 29 freshwater samples were reported for northern Alaska, Canada, Finland, and Russia (AMAP, 1998). Some were above the dissolved concentration criteria for chronic toxicity to aquatic organisms. However, it is likely that the values reported are for total recoverable Cd. The dissolved fraction would be significantly less.

New data reported for the present assessment on heavy metal concentrations in Russian freshwater systems show some values above water quality criteria. However, these probably represent total rather than dissolved concentrations.

6.6.2.2. Freshwater sediments

Sediments samples from 21 wetlands in Russia included five that exceeded the apparent effects threshold for metals (AMAP, 1998). New data from Finland indicate that none of 52 freshwater sediment samples exceed the Hg threshold.

6.6.2.3. Freshwater fish

Mercury uptake in freshwater fish is influenced by feeding habit and growth rate, among other factors (see Section 4.3.4.3). To reflect this, data are presented according to two categories: predatory fish (e.g., Arctic char, salmon) and non-predatory fish (e.g., whitefish (*Coregonus* spp.)). In the present assessment, data are available for 487 muscle Hg concentrations, of which Arctic char, other predatory fish, and some non-predatory fish exceed the threshold for toxicity. Data for 437 samples presented in the first AMAP assessment also include concentrations for char and other predators that exceed toxicity thresholds. Mercury concentrations in freshwater fish muscle are compared to fish toxicity thresholds in Figure 6·3.

6.6.3. Marine ecosystems

Thresholds for toxicity are available for marine sediments, marine fish, seabirds, and marine mammals. Thresholds and available data for biota are compared in Figures 6·3, 6·4, and 6·5.

6.6.3.1. Marine sediments

The maximum mean value of 1.6 ± 1.2 mg/kg dw for Hg in Arctic marine sediments was found in Canada. None of the 306 marine sediment samples analyzed exceeded saltwater criteria for Cd, Hg, or Pb (AMAP, 1998). Some new sediment data in this assessment exceed threshold values.

6.6.3.2. Marine fish

The first AMAP assessment included 136 tissue concentrations for metals in marine fish. None of the marine fish data exceeded the toxicity threshold concentration for Hg in muscle. Hepatic Cd concentrations in Atlantic cod from Varanger and Svolvir, Norway were <0.2 mg/kg ww.

6.6.3.3. Seabirds

Of 376 seabird samples presented in the first AMAP assessment, the maximum liver Hg concentration was 10.6 mg/kg ww and the maximum kidney concentration 28.1 mg/kg ww in great cormorant (*Phalacrocorax carbo*) from Greenland. These data were below the toxicity thresholds of 20 mg/kg ww and 30 mg/kg ww for Hg in the liver and kidney of birds. Of the 152 new seabird samples reported in this assessment, none exceed these toxicity thresholds. The highest mean value for liver was 27.1 mg/kg dw (appoximately 13.5 mg/kg ww) in long-tailed duck (*Clangula hyemalis*), and for kidney 6.19 mg/kg dw (approximately 3.1 mg/kg ww) in Arctic tern (*Sterna paradisaea*), both from Chuan, Russia.

Mercury concentrations in freshwater fish tissues were below 3 mg/kg ww, the dietary level known to cause reproductive disorders in sensitive bird species. It has also been shown that seabirds are at least an order of magnitude less sensitive than freshwater birds, such as herons and bald eagle (*Haliaeetus leucocephalus*) (Thompson, 1996). However, in seabird eggs, 16 of 43 samples had Hg concentrations above a critical threshold of 0.5 mg/kg ww.

As noted previously Steller's eider, spectacled eider (Flint *et al.*, 1997; Franson *et al.*, 1995b), and common eider (Franson *et al.*, 1995b) in the Yukon-Kuskokwim Delta are exposed to Pb through lead shot, and have elevated levels in liver and blood. Whole blood from seven incubating Steller's eider hens in Barrow had Pb levels

above the 0.6 mg/L toxicity threshold (Locke and Thomas, 1996) (range 0.689 to 6.03 mg/L ww; Trust *et al.*, 1997).

Cadmium concentrations for 320 seabird tissue samples were reported in the first AMAP assessment. An additional 152 samples are presented in this report. All data are shown in Figure 6·5. In the first AMAP assessment, nine samples exceeded toxicity thresholds for Cd in liver and 22 exceeded those for kidney. In this assessment, five kidney samples exceeded thresholds (in gulls, eiders, and jaegers (*Stercorarius* spp.) from Norway and Greenland) and two kittiwakes from Norway exceeded liver toxicity threshold values.

6.6.3.4. **Marine mammals**

Most data on metal levels in marine mammals in the first AMAP assessment were for Canada and Greenland. There were few data for marine areas of Alaska and Norway and almost no data for Russia. In this assessment, new data are available for metal levels in ringed seal, bowhead whale, beluga, and polar bear (*Ursus maritimus*) from Alaska, and ringed seal, minke whale (*Balaenoptera acutorostrata*), narwhal (*Monodon monoceros*), and polar bear for Greenland. No additional data are available for the Russian Arctic. For the Faroe Islands, metal data are available for muscle and kidney from long-finned pilot whale (*Globicephala melas*) and white-sided dolphin (*Lagenorhynchus acutus*), and liver and muscle from grey seal (*Halichoerus grypus*). In long-finned pilot whale, kidney Cd concentrations as high as 239 mg/kg ww were found in one of the adults from a group of adult males with an average of 155 mg/kg ww. Females also had high kidney Cd concentrations, e.g., 146 mg/kg ww (see Annex Table A16). These recent values are consistent with data for a school of pilot whales sampled in 1999 (M. Dam, pers. comm., 2003).

Mercury concentrations in liver are compared to toxicity thresholds in Figure 6·3. Exceedance occurred in polar bear, toothed whales, seals, and walrus (*Odobenus rosmarus*), based on concentrations in liver and kidney. However, fish tissue concentrations reported in the first AMAP assessment, and in the present assessment, are all below estimated dietary thresholds of 2 to 6 mg/kg ww (i.e. based on studies with mink) for sensitive marine mammals.

The effects threshold for Cd in marine mammal tissues is 200 mg/kg ww in kidneys, and 40 mg/kg ww in liver. Of around 1000 samples reported in the first AMAP assessment and the 264 new samples reported here, there were 154 and 16 exceedances, respectively. All types of marine mammal were represented in the exceedance group (whales, seals, bear, walrus) and from various locations (Alaska, Canada, Russia, Faroe Islands).

Appendix 6.1

Information on toxicity-based criteria and thresholds from published sources

Substantial research has been conducted to identify toxicity thresholds for a variety of species exposed under controlled conditions to varying levels of heavy metals. This appendix provides general information on how thresholds are established, and includes information on toxicity threshold concentrations for Hg, Pb, and Cd for plants, wildlife, and aquatic organisms. The appendix is not a comprehensive review, but relies on summary sources and previous calculations. In particular, water quality and sediment criteria developed to protect

Table 6·2. Thresholds for effects used in comparisons presented in Figures 6·3, 6·4, and 6·5, and associated literature references.

	Value, µg/g ww	Tissue	Effect	Reference
Mercury thresholds	60	Marine mammal – liver	Liver damage	Law 1996
	30	Terrestrial mammal – liver and kidney	Lethal or harmful in free-ranging wildlife and birds	Thompson 1996
	20	Freshwater fish – muscle	Clinical, sublethal poisoning – less sensitive species	Wiener and Spry 1996
	6	Freshwater fish – muscle	Clinical, sublethal poisoning – sensitive species	Wiener and Spry 1996
	3	Terrestrial birds – eggs	Detrimental effects on hatching	Thompson 1996
Cadmium thresholds	400	Marine mammal – kidney	Potential renal disfunction	Law 1996
	200	Marine mammal – liver	Potential hepatic disfunction	Law 1996
	100	Terrestrial mammal – kidney	Potential renal disfunction	Cooke and Johnson 1996
	100	Terrestrial birds – kidney	Potential renal disfunction	Furness 1996
	40	Terrestrial mammal – liver	Potential hepatic disfunction	Cooke and Johnson 1996
	40	Terrestrial birds – liver	Potential hepatic disfunction	Furness 1996
Lead thresholds	15	Waterfowl – liver	Severe clinical poisoning	Pain 1996
	6	Waterfowl – liver	Clinical poisoning	Pain 1996
	2	Waterfowl – liver	Subclinical poisoning	Pain 1996

aquatic life, and soil benchmark values designed to protect terrestrial organisms, are relied upon for estimating environmental thresholds. Critical body burdens are taken from summary literature.

Although this information was not developed for Arctic species, and frequently contains no specific reference to Arctic organisms, it is regarded as a reasonable first approximation for effects levels in all species, and will suffice until the development of more specific data. Comparison of threshold values to body residues and environmental concentrations in the Arctic, combined with field studies on observed effects, provides a weight-of-evidence analysis of the potential risk to the Arctic environment from current levels of heavy metals.

The purpose of this appendix is to support the assessment. Additional information on thresholds may be found in the literature (e.g., Beyer *et al.*, 1996; Fairbrother *et al.*, 1996; NRC, 1980). A summary of threshold levels and literature sources is provided in Table 6·2.

Thresholds

Thresholds may be quantified by measuring critical residue levels in organisms or by identifying concentrations in environmental media (sediment, soil, or water) that result in uptake of toxic amounts. While concentrations of metals in plants and many vertebrates have been correlated with toxic responses, body burdens in aquatic invertebrates frequently show no relationship to effects. These organisms can, in many instances, store large amounts of various metals in their bodies, sequestering these in granules or in large molecule proteins; thus protecting themselves from harmful effects. Therefore, determination of tissue residue thresholds for toxic responses in aquatic invertebrates currently is not possible, and effects thresholds must be based on concentrations in environmental media (water or sediment). Water column concentrations are less controversial in this regard than sediment values. However, while both are likely to be protective of such species, neither is highly predictive in site-specific situations. Likewise, soil threshold concentrations are generally applicable but local geochemistry may significantly affect bioavailability of metals and subsequent toxicity to plants and animals.

Mercury

The toxicity of Hg to humans is well known, and has been studied extensively (e.g., Goyer and Clarkson, 2001; Merrill *et al.*, 2001). Consequently, tolerance levels for tissue residues in fish or wildlife have been established based on human consumption advisories. Surprisingly, much less is known about the effects of these residues on the animals themselves. The following information on soils, mammals, and birds is a summary of reviews by Eisler (1987) and, more recently, Thompson (1996).

Elemental Hg entering the environment can be converted to the highly toxic form, MeHg, through a variety of natural processes. Exposure to MeHg causes central nervous system effects, including loss of coordination, inability to feed, reduced responsiveness to stimuli, and starvation. Methylmercury is also an immunosuppressant. Maternal deposition of Hg into eggs or embryos can cause reproductive failure or reduction in growth and development. Methylmercury tends to accumulate in brain tissue, and in organs, muscles, feathers, and hair. There are significant differences in species sensitivity to MeHg, particularly in fish. The rate of exposure also influences toxicity. Animals that are exposed slowly can tolerate considerably higher levels than those that are exposed suddenly.

Soil organisms

Earthworms are unaffected by MeHg in soil at levels of 2.5 mg/kg dw (ORNL, 1997a). A concentration of 5 mg/kg dw caused sublethal effects in earthworms and can be considered a reasonable effects threshold (Eisler, 1987). Inorganic Hg may be more toxic to earthworms, as 50% died when placed in soil containing 0.8 mg/kg dw, and all died at the 5 mg/kg dw level (Eisler, 1987). The benchmark value for inorganic Hg in invertebrates has been set at 0.1 mg/kg dw (ORNL, 1997b). The corresponding value for plants is 0.3 mg/kg dw, although some species can tolerate significantly higher concentrations (ORNL, 1997b).

Terrestrial birds

Toxic effect thresholds associated with tissue residues of Hg vary widely among species. Non-marine birds are likely to suffer severe toxic effects when Hg exceeds 20 mg/kg ww in liver or 30 mg/kg ww in kidney. Symptoms include altered and erratic behavior, weight loss due to appetite suppression, ataxia, and death. This level of effect appears to be caused by dietary Hg concentrations of 10 mg/kg ww. Food concentrations of up to 3 mg/kg dw cause no apparent effects in adult birds, but result in reduced egg production, egg viability, and hatchability, and reduced embryo and chick survival. These effects were associated with Hg concentrations in the eggs of 0.5 to 2 mg/kg ww. Toxicity dose thresholds for most birds, based on depressed growth and reproduction, are between 0.05 and 0.1 mg/kg/day (Bouton *et al.*, 1999). Inorganic Hg is significantly less toxic, with threshold dietary concentrations of about 250 mg/kg dw (NRC, 1980).

Terrestrial mammals

Non-marine mammals with Hg concentrations in liver or kidney of more than 20 mg/kg ww are likely to show signs of Hg toxicity. The toxicity threshold for non-marine mammals exposed to inorganic Hg is 1 mg/kg/day, based on reproductive effects in mink (Aulerich *et al.*, 1974). The organic Hg threshold is 0.1 mg/kg/day, also based on mink (Wobeser *et al.*, 1976). This is equivalent to a dietary concentration of 1.8 mg/kg dw. Mink appear to be one of the most sensitive species to effects of Hg exposure (NRC, 1980). A dietary MeHg concentration of 2 to 6 mg/kg ww induces Hg poisoning in most mammals.

Fish

Concentrations of Hg in most surface waters are probably too low to cause direct toxic effects in fish gills. Methylmercury concentrations in uncontaminated surface waters range from about 0.01 to 0.8 ng/L, with total Hg (in unfiltered samples) in the range 0.6 to 4

Table 6·3. Water quality criteria for mercury (µg/L) for the protection of aquatic life (USEPA, 1999).

	Dissolved Hg
Acute freshwater	1.4
Chronic freshwater	0.77
Acute marine	1.8
Chronic marine	0.94

ng/L. Water quality criteria for the protection of aquatic life are shown in Table 6·3 for Hg (USEPA, 1999).

Sediment threshold concentrations are shown in Table 6·4. Actual effects thresholds vary by site, depending upon sediment conditions, food chain length, and the potential for methylation of Hg. These include the Threshold Effects Level where some response has been observed in some areas, and the Apparent Effects Threshold where effects are always observed (Buchman, 1999).

Fish accumulate a significant amount of Hg from their diet. As with other animals, the methylated form of Hg is considerably more toxic than the inorganic form. Also, MeHg has substantially greater biomagnification potential. Mercury in fish tissue ranges from <10 to >80% MeHg depending upon the level of the food chain at which the fish feed. The remainder of the Hg is inorganic (Law, 1996).

In adult fish, brain concentrations of ≥7 mg/kg ww are potentially lethal for all species. Some species, such as pike have a greater sensitivity to Hg, and brain concentrations >3 mg/kg ww are associated with severe effects. Muscle tissue residues of 5 to 8 mg/kg ww in walleye and 10 to 20 mg/kg ww in salmonids are associated with toxic effects. Toxic thresholds based on whole body concentrations are 5 mg/kg ww for brook trout (*Salvelinus kontinalis*) and 10 mg/kg ww for rainbow trout. In trout, egg concentrations of 0.07 to 0.1 mg/kg ww may be associated with embryo mortality (maternal fish tissue levels are generally 100 times greater than egg concentrations). For all salmonids, whole body concentrations >3 mg/kg ww and brain or muscle concentrations >5 mg/kg ww are considered thresholds for toxicity (see Thompson, 1996).

Seabirds

Toxic effects thresholds associated with tissue residues of Hg vary widely among species. Seabirds can tolerate at least an order of magnitude greater Hg exposure than non-marine birds, with highest accumulations in pelagic birds. Tissue threshold concentrations are also an order of magnitude greater than for non-marine birds (Thompson, 1996).

Marine mammals

Law (1996) reviewed available information on metal toxicity and threshold concentrations in marine mammals. Controlled studies with pinnipeds suggest a toxic threshold in liver at total Hg concentrations of >130 mg/kg ww, with subclinical toxicity at >65 mg/kg dw. The liver contains approximately 95% of the body burden of Hg; with 3 to 62% in the methylated form. This corresponds to a dietary intake of 0.01 mg/kg body weight, or approximately 7 mg/kg dw in edible fish tissue. Normal concentrations in fetal and neonatal animals are low, with <1 mg/kg dw in liver. However, these thresholds are very conservative. Marine mammals are less affected by Hg than terrestrial species, as they can demethylate Hg and immobilize it in liver tissue as mercuric selenide. In this form, marine mammals can accumulate Hg to very high levels without detrimental effect. For example, liver concentrations in live adult striped dolphin (*Stenella coeruleoalba*) from the Tyrrhenian Sea have been measured at 4400 mg/kg dw, while bottle-nosed dolphin (*Tursiops truncatus*) from this area had 13000 mg/kg dw. This is approximately equal to 1300 and 3800 mg/kg ww. Various pinnipeds have been shown to have no indication of Hg poisoning although liver concentrations were as high as 420 mg/kg dw.

Lead

Lead is a nonessential element that causes a wide range of toxic effects. Neurotoxicity resulting in altered behavioral responses is the most widely recognized effect of Pb poisoning. Lead interferes with many enzymes, most notably those associated with the production of hemoglobin. Other effects include renal damage and dysfunction, anemia, intestinal dysfunction, and reproductive problems including abnormal growth and development. At exposure, Pb is taken up from the gut into the blood stream and distributed preferentially to the liver and kidney, with bones, brain, and muscle receiving considerably less. Lead may also be found at low concentrations in hair and feathers, and over time tends to be stored in bone. Lead absorption and deposition within body tissues, are influenced by the type of Pb ingested (e.g., Pb shot versus Pb salts in soils or Pb incorporated into plant material). For example, laying hens deposit more Pb in bones than non-laying hens, due to interactions between Pb and Ca metabolism. Animals on nutrient-poor diets absorb more Pb than those with adequate nutrition. The following information on Pb in biota is summarized principally from Law (1996).

Aquatic organisms

Water quality criteria for Pb for the protection of freshwater aquatic organisms are based on water hardness. Using a default hardness of 100 mg/L for freshwater, the US EPA set the following criteria for the protection of aquatic life, including fish, invertebrates, and aquatic

Table 6·4. Sediment threshold concentrations (µg/kg dw) for mercury (Buchman, 1999).

	Threshold Effects Level	Effects Range Low	Probable Effects Level	Effects Range Median	Apparent Effects Threshold
Freshwater	174	–	488	–	560
Marine	130	150	696	710	410

Table 6·5. Water quality criteria for lead (µg/L) for the protection of aquatic life (adjusted for a water hardness of 100 mg/L for freshwater) (USEPA, 1999).

	Dissolved Pb
Acute freshwater	65
Chronic freshwater	2.5
Acute marine	210
Chronic marine	8.1

Table 6·6. Water quality criteria for lead (µg/L) for the protection of aquatic life (adjusted for a water hardness of 100 mg/L for freshwater) (USEPA, 1999).

	Threshold Effects Level	Effects Range Low	Probable Effects Level	Effects Range Median	Apparent Effects Threshold
Freshwater	35 000	–	91 300	–	127 000
Marine	30 240	46 700	112 180	218 000	400 000

plants, except possibly where a locally important species is unusually sensitive (Table 6·5). Values are given for the dissolved fraction as this is the fraction to which the organisms respond.

Sediment threshold concentrations for Pb are given in Table 6·6. The Threshold Effects Level is the level at which some response has been observed in some areas. Above the Apparent Effects Threshold, effects are always observed.

Soil organisms

The soil Pb benchmark level for plants is 50 mg/kg dw and for soil organisms 500 mg/kg dw (ORNL, 1997a).

Birds

Lead poisoning in waterfowl has been recognized as a significant mortality factor since the early part of the twentieth century, owing to ingestion of spent lead shot. As a result, toxic thresholds are well known. Table 6·7 details the threshold concentrations in various tissues for animals that are poisoned but without obvious effects (subclinical poisoning), animals that are obviously ill (clinical poisoning), and animals with severe poisoning (generally not expected to recover, or found dead).

Lead poisoning has been reported in over 30 species of bird other than waterfowl, although effects in these species have not been studied in as much detail. Nevertheless, toxic thresholds for Pb residues in tissues have been established for Falconiformes (raptors, such as hawks and eagles), Columbiformes (pigeons), and Galliformes (grouse, partridge, chukar) as shown in Table 6·8.

Toxicity thresholds for birds have been estimated at a dose of 0.16 mg/kg body weight per day for birds such as quail, chickens, or small songbirds. Larger birds, including raptors, are more tolerant, with dietary thresholds of about 2 mg/kg body weight per day (Pattee, 1984). Dietary threshold concentrations are approximately 500 mg/kg dw (NRC, 1980).

Terrestrial mammals

Absorption rates for Pb in mammals vary from 2 to 20% when ingested, and from 10 to 50% when inhaled. Blood Pb levels are the most effective measure of Pb poisoning in mammals. Clinical Pb poisoning occurs only at levels >35 µg/dL, and is generally not found until blood Pb reaches 60 µg/dL. A level of 20 µg/dL is indicative of subclinical exposure (i.e., elevated intake but levels below those that will cause effects to the whole organism). Liver concentrations of 30 mg/kg dw or kidney Pb of 90 mg/kg dw are associated with clinical signs of Pb

Table 6·7. Threshold concentrations for lead in biota (Pain, 1996).

	Normal	Subclinical poisoning	Clinical poisoning	Severe clinical poisoning
Blood (µg/dl)	<20	20 - <50	50 - 100	>100
Liver (mg/kg ww)	<2	2 - <6	6 - 15	>15
Bone (mg/kg dw)	<10	10 - 20	>20	

Table 6·8. Toxic thresholds for lead in Falconiformes (raptors), Columbiformes (pigeons) and Galliformes (grouse, partridge, chukar) (Franson, 1996).

Tissue (mg/kg, ww)	Subclinical poisoning	Clinical poisoning	Severe clinical poisoning
Falconiformes			
blood	0.2 - 1.5	>1	>5
liver	2 - 4	>3	>5
kidney	2 - 5	>3	>5
Columbiformes			
blood	0.2 - 2.5	>2	>10
liver	2 - 6	>6	>20
kidney	2 - 20	>15	>40
Galliformes			
blood	0.2 - 3	>5	>10
liver	2 - 6	>6	>15
kidney	2 - 20	>15	>50

poisoning. Subclinical thresholds are 10 mg/kg dw in liver and 25 mg/kg in kidney (Franson, 1996) (approximately equivalent to 50 and 125 mg/kg ww, respectively).

Cattle and sheep have consumed 10 mg/kg soluble Pb (on a dw basis) in their feed for extended periods without adverse effects. Some ruminants can tolerate up to 1000 mg/kg Pb in feed for several months. However, other species (e.g., horses) are more sensitive, with clinical signs observable at 300 mg/kg dw dietary Pb (NRC, 1980).

Marine mammals

Lead absorption rates in mammals vary from 2 to 20% when ingested and from 10 to 50% when inhaled. Lead levels in blood, liver, and kidney are used to identify Pb concentrations associated with clinical signs of Pb poisoning and subclinical thresholds (Franson, 1996). In marine mammals, Pb liver concentrations are generally low (<1 mg/kg dw), although concentrations as high as 5 mg/kg have been measured in industrialized areas in apparently normal animals (Law, 1996). Toxic thresholds specific to marine mammals have not been determined.

Cadmium

Cadmium is a nonessential element widely distributed in the Earth's crust, most frequently in association with zinc. The following information on Cd is taken from Cooke and Johnson (1996) and Furness (1996). Cadmium body burdens are very low at birth and Cd accumulates slowly throughout an animal's life. Absorption from the diet is low (<5% of ingested cadmium), but low Ca and Fe, as well as deficient levels of protein, Zn, and Cu, can significantly increase absorption. Excretion of absorbed Cd is very low, resulting in a long retention time relative to the life span (e.g., 26 years in humans, one year in rodents). Cadmium binds readily to MT and is rapidly transported to various organs. In most animals, the greatest amount of Cd occurs in kidney, then liver, and with a significant amount also bound to intestinal mucosa. However, because Cd may be excreted through the kidney (albeit in small amounts), liver concentrations tend to be more stable while those in kidney fluctuate. Fish, crabs, oysters, shrimp, and other aquatic invertebrates may accumulate significant amounts of Cd.

Signs of Cd toxicity are most frequently related to kidney dysfunction, with the hyperexcretion of Ca and phosphorus resulting in porosity and reduced calcification of bones. Other signs include reduced food intake, decreased growth, and hypertension. Very high levels cause hepatotoxicity. Cadmium induces the production of MT, a sulfide-rich transport protein, which sequesters Cd in liver and kidney. Levels of MT may be correlated with potential for toxic effects (Peakall, 1992), but no threshold has been established (Furness, 1996).

Soil organisms

The benchmark for soil Cd in plants is 4 mg/kg dw, and for invertebrates 20 mg/kg dw (ORNL, 1997a). Soil benchmark values for plants and soil invertebrates are established at the fifth percentile within the distribution of available data on plant and invertebrate toxicity responses to metals in soils. Because different soil types, soil geochemistry, and measurement endpoints significantly affect measured response levels, these benchmark values may not be locally appropriate. However, they provide a general guide for screening areas to determine whether the possibility of risk can be ruled out.

Birds

The highest cadmium concentrations in birds occur in kidney, are lower in liver (~50% of total body burden), and very low in muscle (Furness, 1996). Cadmium causes little to no pathological effects in liver, but at sufficiently high levels will cause tubular dysfunction in kidney, with subsequent reduction in Cd concentrations. Cadmium also induces bone damage through inhibition of calcification resulting in osteoporosis. Cadmium is not appreciably deposited in eggs, which typically have extremely low levels. It is not known whether Cd deposits preferentially in feathers, but it is unlikely.

Uptake of Cd from the diet is dose-dependent, but is generally very low (<2%). Decreased dietary concentrations of Ca, Zn, or Fe potentiates Cd uptake from the gut lumen in birds. Copper and Se also may interfere with Cd uptake.

Diets containing Cd as low as 2 mg/kg increase MT synthesis and result in the accumulation of Cd but do not result in toxicity. Egg production may be inhibited and body weight reduced at 60 to 75 mg/kg in Japanese quail (*Coturnix japonica*; Richardson *et al.*, 1974; Spivey Fox *et al.*, 1971) or mallard duck (*Anas platyrhynchos*; White and Finley, 1978) (equivalent to 15 to 20 mg/kg/day). Chicken appear more sensitive, with a threshold dose resulting in reduced eggshell thickness of about 3.5 mg/kg/day (Scheuhammer, 1991), or about 12 mg/kg dw of diet (NRC, 1980). Toxic effects include kidney damage, suppression of egg production, duodenal epithelium damage, anemia, bone marrow hyperplasia, and cardiac and adrenal hypertrophy.

About 40 mg/kg ww in liver or 100 mg/kg ww in kidney have been suggested as threshold levels, above which toxic effects would be observed. Concentrations of Cd in liver of healthy birds are generally well below these threshold levels, ranging from <1 mg/kg ww in peregrines and raptors; 1 to 5 mg/kg ww in wading birds and waterfowl, and 5 to 35 mg/kg ww in many shorebirds. Some seabird populations appear able to tolerate even higher levels (above the 40 mg/kg ww threshold) without notable reduction in fitness or population size, suggesting the possibility of higher thresholds for pelagic seabirds owing to the naturally elevated levels of Cd in seawater.

Terrestrial mammals

Based on studies of voles and mice associated with areas of biosolid application or smelter sites, Ma (1996) concluded that kidney Cd levels generally range from 2 to 90 mg/kg dw, and are two to eight times the corresponding liver value. Co-located shrews have considerably higher tissue concentrations, presumably due to feeding almost exclusively on soil invertebrates. Values range from 100 to 250 mg/kg dw in kidney and 200 to 600 mg/kg dw in liver (Cooke and Johnson, 1996). Critical threshold concentrations in kidney are estimated at 350 mg/kg ww, equivalent to 100 mg/kg dw (Cooke and Johnson, 1996).

Onset of clinical signs (e.g., mild renal dysfunction and loss of bone calcification) is associated with dietary levels of 10 mg/kg dw, with severe pathology at dietary Cd concentrations of 50 mg/kg dw. In rats, this is equivalent to a dose of 1 and 7 mg/kg body weight per day. A study with bank vole (*Clethrionomys glareolus*) had a threshold for effects at 15 mg/kg dw in the diet, equivalent to a dose of 2.1 mg/kg body weight per day (Swiergosz *et al.*, 1998).

For larger mammals, clinical effects have been seen at or above 30 mg/kg dw in the diet with subclinical changes at dietary concentrations of 5 mg/kg dw (NRC, 1980).

Aquatic organisms

Water quality criteria for Cd for freshwater aquatic organisms are linked to water hardness. Using the default hardness value of 100 mg/L for freshwater, the US EPA established the following criteria for the protection of all aquatic life, including fish, invertebrates, and aquatic plants, except possibly where a locally important species is unusually sensitive (Table 6·9). Values are given for the dissolved fraction as this is the fraction to which organisms respond.

Sediment threshold concentrations have been suggested as shown in Table 6·10. The Threshold Effects Level is the level at which some response has been observed in some areas. Above the Apparent Effects Threshold, effects always are observed.

Marine mammals

Marine mammal Cd tissue concentrations and threshold effects values were reviewed by Law (1996). High concentrations of Cd in marine mammals in areas remote from pollution sources are not uncommon. Greater than 20 mg/kg dw in liver and 34 mg/kg dw in kidney have been found in Dall's porpoise (*Phocoenoides dalli*). In long-finned pilot whale the average kidney Cd concentration in four groups of adults ranged from 110 to 160 mg/kg ww in two schools sampled in 1999 and 2000. The highest individual kidney Cd concentration was 239 mg/kg. There are significant differences in Cd tissue concentration among species, probably reflecting differences in feeding preferences. Liver of Pacific walrus (*Odobenus rosmarus divergens*) from the Bering Sea had up to 50 mg/kg ww with no obvious adverse effects (Taylor *et al.*, 1989). Law (1996) suggests that the toxic thresholds of 200 to 400 mg/kg ww in kidney that has been established for humans be applied to marine mammals as well. This corresponds to a liver concentration of 40 to 200 mg/kg dw, based on studies that measured relative concentrations in both tissues. Transplacental transfer of Cd is negligible, so embryos and neonates should contain little to no Cd. Significant amounts may be transferred in milk, suggesting that juveniles may begin accumulating Cd in liver and kidney.

Table 6·9. Water quality criteria for cadmium (µg/L) for the protection of aquatic life (adjusted for a water hardness of 100 mg/L for freshwater) (USEPA, 2001).

	Dissolved Cd
Acute freshwater	2.0
Chronic freshwater	0.25
Acute marine	40
Chronic marine	8.8

Appendix 6.2

Threshold figures: sources of uncertainty

Toxicity thresholds are generally established within a laboratory setting using standard test species. Relating these data to data collected from wild populations using different research protocols introduces significant uncertainty that directly influences confidence in any conclusions about potential effects based on environmental and tissue concentrations. This appendix discusses uncertainties in the thresholds established in the literature, conversions generated for this assessment, and the construction of the figures.

Establishing thresholds

Toxicity thresholds cited in this chapter are based on published threshold values. Generally these are established through the exposure of laboratory animals to controlled levels of heavy metals. Most data are based on standard test species that are then compared to similar species (e.g., chicken to other birds). Wherever possible, toxicity thresholds based on Arctic species, or closely related species are used.

Thresholds derived in the laboratory are likely to differ in natural populations because of the many factors that influence toxicity in the wild. Most animals have strong preferences about where and what they eat which influences exposure. Thus, concentrations of contaminants in dietary items (e.g., plants, invertebrates) provide only a guideline for where to assess the potential for adverse effects in higher trophic level animals. In addition, the likelihood of observed effects at threshold concentrations are a function of body size and food consumption rates. Environmental factors in the abiotic environment (e.g., freshwater pH) also influence uptake. These variables add to the uncertainty of comparing concentrations in Arctic species with laboratory derived toxicity thresholds.

Thus thresholds, when compared to tissue concentrations in Arctic biota, provide general guidance for re-

Table 6·10. Sediment toxicity thresholds (µg/kg dw) for cadmium (Buchman, 1999).

	Threshold Effects level	Effects Range Low	Probable Effects Level	Effects Range Median	Apparent Effects Threshold
Freshwater	596	–	3530	–	3000
Marine	676	1200	4210	9600	3000

gions and species where there is potential for concern. Threshold exceedences do not demonstrate, *a priori*, an undue risk to Arctic biota. In addition, failure to identify an exceedance does not, *a priori*, constitute an area of no concern. Comparisons of spatial trend data with threshold concentrations should be used to highlight those areas where there is reason for further research.

Data conversion

As part of this assessment, concentrations of heavy metals in the tissues of Arctic species were compared to tissue concentration thresholds. In order to increase the robustness of comparing data to toxicity thresholds, as many data as possible were incorporated Figures 6·3, 6·4, and 6·5. Some conversions were required, principally between wet weight (ww) and dry weight (dw) for each tissue type. For comparison purposes, when thresholds were developed in ww, data for Arctic species were retained as, or converted to ww. When thresholds were developed in dw, data for Arctic species were retained as, or converted to dw for comparison. Conversions factors used in this assessment to convert from dw to ww were: 2 for birds (Dunning, 1993), 5 for marine mammals (Law, 1996), and 5 for terrestrial mammals (ECOFRAM, 1999). These conversion factors were selected on the basis of what appeared to be the median (i.e., most common) value used for classes of organisms. The same conversion factor was then applied in all cases. As there are differences in the moisture content of the various tissues for the many types of bird and mammal evaluated, this is a source of uncertainty in the use of thresholds. Values in the text, however, were retained as originally reported without conversion.

Formulation of the illustrations

Three illustrations are used to show threshold values relative to environmental concentrations in the Arctic biotic environment, one each for Hg (Figure 6·3), Pb (Figure 6·4), and Cd (Figure 6·5).

Data sets used to generate the bars were of four types: 1) ranges of actual values, 2) a mean and standard deviation, 3) a mean with a 'relative' standard deviation (see AMAP, 1998), 4) only the mean. When a range was given, it was used to show the maximum-minimum of the measured samples. When a standard deviation or a relative standard deviation was given and the sample size was eight or more, a range was generated using the mean, 2 times the standard deviation, to establish a 95% confidence interval. Sample sizes of 8 or more were used to eliminate small sample size biases.

The illustrations are based on data from the first AMAP assessment as well as that of the present assessment. Although presented on the same bar, the data are distinguished in order to highlight new data. No attempt was made to explain differences between the two datasets for the 20 cases in which data from both assessments were available.

The toxicity threshold values shown in the illustrations are based on standards or criteria established in countries or other jurisdictions. Dietary thresholds for birds and mammals have not achieved this status and are generally estimated from dose information (i.e., converted from mg/kg body weight per day to mg/kg diet) and estimated food consumption rates. Therefore, wildlife dietary thresholds are not shown on the illustrations, although they are mentioned in the text when such information is available.

Chapter 7

Conclusions and Recommendations

Suzanne Marcy, Rune Dietz, Anders Bignert, Hans Borg, Birgit Braune, Steve Brooks, Jesper Christensen, Maria Dam, John Derome, Anne Fairbrother, Jesse Ford, Viacheslav Gordeev, Alexey Gusev, Steve Lindberg, Robie Macdonald, Peter Outridge, Jozef Pacyna, Keith Puckett, Frank Riget, Oleg Travnikov, and Simon Wilson

The distribution of heavy metals among the various environmental compartments of the Arctic is dynamic and reflects natural sources, environmental processes, and a range of environmental factors that affect biological uptake. Superimposed on these natural patterns are anthropogenic inputs of metals. A continuing challenge in assessing the levels, trends, and effects of heavy metal contamination in the Arctic is showing the extent to which observed patterns reflect natural processes or anthropogenic inputs.

The second AMAP assessment has extended the understanding of anthropogenic sources, transport processes, trends, and effects of metals in the Arctic. This chapter provides a summary of the principal findings that emerged from the evaluation of the AMAP Phase II data and, where appropriate, in combination with the AMAP Phase I data. Many observations were consistent between the two assessments; as summarized in section 7.1. The principal results of the second AMAP assessment are provided in section 7.2. Tables 7.1 and 7.2 present an overview of the combined AMAP Phase II and AMAP Phase I results for the terrestrial, freshwater and marine environments. Conclusions and recommendations are presented in section 7.3 based on scientific results.

7.1. Phase I and Phase II – common observations

Data collected for the second AMAP assessment complement rather than duplicate earlier work. Many of the findings from AMAP Phase I were confirmed during the second assessment. These include the following.

1. The metals in the Arctic biosphere of greatest toxicological concern continue to be mercury (Hg) and cadmium (Cd). These metals occur in some Arctic biota at concentrations that may have health implications for individual animals or for human consumers. Reductions in atmospheric lead (Pb) levels are promising but due to problems identified with the use of Pb shot in waterfowl hunting, Pb remains a concern in some areas.
2. Levels of metals in Arctic soil, lakes, rivers, and oceans generally fall within global ranges for the same elements. The geographical distribution of Pb, Cd, Hg, and copper (Cu) in marine sediments appears related to the geological provinces of the Arctic.
3. While still difficult to establish spatial patterns and temporal trends for metals in the Arctic due to data limitations, increasing trends in Hg in some areas and more widespread decreasing trends in Pb have been documented.
4. The current assessment presents important new data from Alaska and Russia, filling geographical gaps in spatial coverage that were identified in the first AMAP assessment. Additional data were gathered for the evaluation of temporal trends. To improve detection of trends, statistical analyses suggest that many ongoing studies involving annual sampling need to be extended for reliable time trend evaluation.
5. Challenges associated with finding observable effects in Arctic biota and linking these to heavy metals remain considerable. Current tools are limited.

7.2. Phase II – summary of results

New data collected for the second AMAP assessment contributed significantly to the available information on heavy metals in the Arctic. Key new results on Hg, Pb, and Cd, and on other heavy metals, are described in the following sections. In addition, Tables 7.1 and 7.2 provide a compilation of data available from both AMAP Phase I and AMAP Phase II. This compilation summarizes the current state of knowledge and highlights data gaps.

7.2.1. Mercury

Mercury emissions appear to be increasing globally, despite temporary reductions in the 1990s from the implementation of better emission controls in some regions. When Hg reaches the Arctic it undergoes chemical transformations that lead to enhanced deposition during polar sunrise in the spring. A well-known neurotoxicant in its methylated form, there is now evidence that in certain regions of the Arctic, Hg environmental levels are increasing.

7.2.1.1. Sources

1. Coal burning, waste incineration, and industrial processes worldwide affect environmental inputs of Hg, even to remote regions, including the Arctic.
2. Based on 1995 emission inventories, a total of 1475 t/yr (66%) of Hg were released to the atmosphere from stationary fossil fuel combustion, with other sources including non-ferrous metal production (164 t, 7%) cement production (133 t, 6%), waste disposal (109 t, 5%), iron and steel production (29 t, 1%) as well as other sources (325 t, 15%) for a total of 2235 t emitted in 1995.

3. Emissions data from 1995 show that the majority was released from Asia (1121t, 50%). Emissions from other source regions include Africa (389t, 17%), Europe (313t, 14%), North America (215t, 10%), Australia and Oceania (113t, 5%), and South America (84 t, 4%).

4. Major emissions from fossil fuel combustion declined in Europe and North America, primarily during the 1980s and 1990s from emission controls. However, these declines are now being offset by increased emissions in Asia.

7.2.1.2. Transport pathways and transformation processes

5. Following emission to the atmosphere in gaseous form, Hg interacts, transforms, and is transported in the atmosphere over very short (e.g., one to two days in the spring in the Arctic) to very large time and space scales (e.g., one to two years for elemental Hg). Since elemental Hg has a long residence time in the atmosphere, it undergoes global scale atmospheric transport. This represents a major pathway to the Arctic environment.

6. Based on recent models, there are differences between the relative contributions of source regions to different receptor regions. Surface air is more affected by local sources. Contributions from global sources are transported at higher atmospheric levels.

7. Recently discovered transformation of elemental Hg during polar sunrise suggests that the Arctic is a global sink for Hg. Ground level ozone and elemental Hg depletions at Alert in the Canadian High Arctic provided the first evidence that conditions exist in the upper Arctic following polar sunrise that promote depletion of atmospheric elemental Hg and could explain springtime episodic Hg depletion events. Mercury depletion events have now been documented in multiple coastal locations during Arctic spring including Alert, Canada; Barrow, Alaska; Ny-Ålesund, Svalbard; and Station Nord, Greenland.

8. At Barrow, the Hg deposition mechanism has been linked with the formation of reactive gaseous mercury in the near surface air, the formation of fine particulate Hg in air, and the deposition to, and accumulation of Hg in the snowpack.

9. Mercury depletion events observed during polar sunrise are estimated to lead to the deposition of 100 to 300 tonnes of total Hg per year in the Arctic.

10. An estimated 50% ± 25% of Hg deposited on the snowpack is retained until snowmelt when it is released to the environment. In Barrow, large pulses of Hg with measured concentrations in the range 10-80 ng/L are delivered to tundra in percolation and runoff waters. Mercury that is bioavailable during periods of high biotic activity can be taken up by bacteria and potentially methylated to forms that can bioaccumulate and cause effects in developing biota. This is potentially a key step in the pathway for Hg accumulation in Arctic biota.

7.2.1.3. Spatial patterns and temporal trends

11. Long-term time series (more than 100 years) derived from sediment, ice, and peat bog cores provide evidence of increased Hg fluxes in many but not all areas of the Arctic since the pre-industrial period.

12. Of the few long-term temporal trend data sets available on Hg levels in biota, tissue levels have increased since pre-industrial times. Mercury concentrations in contemporary beluga whale (*Delphinapterus leucas*) teeth collected in the Mackenzie Delta region were 4 to 17 times higher than levels found in teeth from the pre-industrial period (1450-1650 AD). Present-day Hg concentrations in seal and human hair from Greenland are two to four times higher than in samples from the fifteenth century. Mercury levels in modern human hair from the Canadian Arctic are also several times higher than in pre-industrial (fifth and twelfth century) hair samples. Mercury in human teeth (without Hg amalgam fillings) collected in the 1970s in Norway was 13 times higher than in human teeth from the twelfth century.

13. In studies where dated lake sediments or peat bog cores were used, increased Hg levels in the surface layers were found and interpreted as evidence of increased Hg fluxes. While confounding factors must be considered when interpreting trends using lake sediment and peat bog data, increased Hg concentrations in sediments of the Pechora River and lake sediments in the region between Russia and Norway present a potential risk to freshwater organisms.

14. Shorter (up to ca. 30 year) time series were used to identify more recent trends. Significant increasing temporal trends in Hg over the last 20 to 30 year period were found in marine mammals and seabirds from the Canadian Archipelago (Beaufort Sea to Northwest Greenland) and are cause for concern. In the northeast Atlantic region, decreasing trends in Hg were observed in some fish and mussels around Iceland and western Norway. Decreasing trends were also found in some terrestrial mammals in inland western Canada (Yukon). The European Arctic shows no increasing trends in Hg.

15. The cause for apparent differences in temporal trend patterns between regions is unknown. However, the pattern coincides with the distribution of atmospheric bromine, ice cover, and other factors that could be influenced by the recently discovered atmospheric Hg depletion events that appear to increase atmospheric-surface Hg flux rates.

16. The utility of retrospective temporal trend studies based on archived samples has been clearly demonstrated. Time series of at least 10 years and up to 30 years are required to provide sufficiently powerful trend detection. With a few notable exceptions, available time series of metal concentrations in biota in the Arctic are too short to reliably detect changes of interest. However, the number of series now approaching ten or more years in length is encouraging.

Table 7.1. Summary of regional and local impacts on marine ecosystems.

	Seawater	Sediment	Algae
1. Concentrations of metals exceeding average global background			
1.1 Regional	None	None	None
1.2 Local	At point sources such as mining areas in Canada and Greenland as well as some Russian estuaries (scale <30 km)	At point sources such as mining areas in Canada and Greenland as well as some Russian estuaries (scale <30 km)	At point sources such as mining areas in Canada and Greenland as well as some Russian estuaries (scale <30 km)
2. Spatial patterns within the Arctic			
2.1 Regional	None	Sediment seems to folow geological provinces over the Arctic	None
2.2 Local	Enrichment near point sources. Increasing natural Cd from inner fjords toward the sea	Enrichment near point sources. Increasing natural Cd from inner fjords toward the sea	Enrichment near point sources. Increasing natural Cd in individuals from inner fjords toward the sea
3. Temporal trends within the Arctic			
3.1.Regional	Higher Pb levels than in prehistoric time	Hg concentrations in sediment from a number of Arctic areas show levels elevated or increasing in recent sediments	Unknown
4. Observed biological effects attributable to metals			
4.1 Regional	Not applicable	Not applicable	Not likely
4.2 Local	Not applicable	Not applicable	Possible combined effect in some Russian estuaries.

Legend:
Blue AMAP Phase I findings
Black AMAP Phase II findings

Unknown: Insufficient data to reach a conclusion.
None: No trend or effect documented from a fair amount of data.
Combined: Contamination by metals may and probably does contribute to some effects caused primarily by other factors.

Invertebrates	Fish	Seabirds	Marine mammals
Cd high in Arctic mussels and prawns	None	None	Cd is elevated in some Arctic marine mammals
At point sources such as mining areas in Canada and Greenland as well as some Russian estuaries (scale <30 km)	At point sources such as mining areas in Canada and Greenland as well as some Russian estuaries (scale <30 km)	None	None
Cd concentrations appear higher in Alaska than in the other Arctic areas, except in central West Greenland, which is known to have locally high Cd levels in biota. Hg in mussels shows no clear spatial pattern	Cd concentrations are highest in fish from Greenland and Alaska, intermediate in Canada, and lowest in the Faroe Islands. No Hg trends documented.	Cd and Hg concentrations in seabirds from the Barents Sea are lower than in Greenland, Canada and northwest Siberia. Within the Canadian Arctic, Hg levels in seabird eggs are generally higher at high latitude sites.	The highest Cd concentrations are found in ringed seals from West Greenland and eastern Canada, concentrations in Alaska are intermediate, and are lowest at sites in Labrador and Svalbard. Hg concentrations are highest in central Arctic Canada, intermediate in Alaska and Greenland, and lowest in the Norwegian Arctic.
Enrichment near point sources. Increasing natural Cd in individuals from inner fjords toward the sea	Enrichment near point sources. Increasing natural Cd in stationary fish from inner fjords toward the sea	Higher Cd and Hg concentrations were found in seabirds from Ny Ålesund compared to the rest of the Barents Sea	Not likely as few marine mammals are stationary
Unknown. Decreasing Hg trend observed in some mussels around Iceland and western Norway.	Unknown. Decreasing Hg trend observed in some fish around Iceland and western Norway.	Moderate to no increase in Hg. Significant increase of Hg in seabirds from the Canadian Archipelago.	Hg levels in seal tissue from northern Canada and Greenland as well as in toothed whales, are increasing. Concentrations of Hg in beluga teeth have increased 4- to 17-fold since the pre-industrial period. Cd has not increased.
Not likely	Not likely	None of the seabird Hg values reported under Phase II, were above toxicity thresholds. Cd high enough in some areas to pose a threat for kidney damage	Cd high enough in some areas to pose a threat for kidney damage. Pathological investigations on ringed seal, beluga, and bowhead whale have not identified Cd-related effects.
Possible combined effect in some Russian estuaries.	Possible combined effect in some Russian estuaries.	Hg levels in up to 30% of eggs from American peregrine falcon suffering reduced productivity exceeded the critical threshold for reproductive effects. Cd high enough in some areas to pose a threat for kidney damage	Cd high enough in some areas to pose a threat for kidney damage.

Table 7.2. Summary of regional and local impacts on terrestrial and freshwater ecosystems.

	Air/atmospheric deposition	Freshwater	Sediment	Soil
1. Concentrations of metals exceeding average global background				
1.1 Regional	Recently discovered transformation of elemental Hg during polar sunrise suggests that the Arctic is a global sink for Hg; 100-300 tonnes of total Hg are deposited per year in the Arctic.	None	None	None
1.2 Local	Kola Peninsula	At 'hot spots' of human/ industrial activity such as smelter complexes of the Kola Peninsula (scale 10-100 km)	At 'hot spots' of human/ industrial activity such as smelter complexes of the Kola Peninsula (scale 10-100 km)	At 'hot spots' of human/ industrial activity such as smelter complexes of the Kola Peninsula (scale 10-100 km)
2. Spatial patterns				
2.1 Regional	South to north decrease Seasonal patterns indicate different sources in different seasons. Hg high in Chukchi Seas snow	None	High Hg concentrations in Arctic Lakes. Regional differences in Hg time signal	Kola/northern Scandinavia enrichment in Cu and Ni
2.2 Local	Decrease with distance from the source region. Concentrations and effects due to Kola Peninsula industry are highest around Monchegorsk	Enrichment near point sources (Russian rivers, lakes of Kola Peninsula)	Gradients near point sources (Cu and Ni)	Gradients near point sources (Cu and Ni)
3. Temporal trend				
3.1 Regional	A decrease over the last 2 decades. Strong seasonal variation; the highest values seen in winter.	Unknown in freshwater, but large increases found relative to pre-industrial levels in ice cores	Recent Arctic-wide increase in surficial Hg concentrations in Arctic Lakes	Unknown
3.2 Local	A decrease over the last 5 years in the Kola Peninsula. Winter concentrations higher than in summer.	Unknown; however, a 20% increase in Hg has been detected in Ice cores between 1967 and 1989	Unknown	Unknown
4. Observed biological effects attributable to metals				
4.1 Regional	Not Applicable	Not Applicable	Not Applicable	Not Applicable
4.2 Local	Not Applicable	Not Applicable	Not Applicable	Not Applicable

Legend:

BlueAMAP Phase I findings
Black AMAP Phase II findings

Unknown: Insufficient data to reach a conclusion.
None: No trend or effect documented from a reasonable amount of available data.
Combined: Contamination by metals may and probably does contribute to some effects caused primarily by other factors.

Vegetation	Birds	Fish	Mammals
None	High Cd in kidney of ptarmigan	None	High Cd in reindeer/caribou kidney
At 'hot spots' of human/ industrial activity such as smelter complexes of the Kola Peninsula (scale 10-100 km)	Cd in kidney of ptarmigan high in Yukon/NWT	High Hg in fish of NWT	Unknown
Enrichment of Cu and Ni in Kola/northern Scandinavia	None	None	None
Gradients near point sources (Cu and Ni)	Cd in ptarmigan high in YukonPb in herbivores in Russia greater in east than west	Highest values for Hg occur in Canada	Cd in caribou kidney higher in Yukon than in NWT. In Norway, Cd in reindeer/moose kidney increases along N-S gradient. In Russia, Pb in reindeer liver/muscle are higher in east than in west
Unknown Post-industrial increase of Hg in peat bogs from 7-14 times	Unknown	Inconsistent pattern for Hg in fish from different regions	Unknown in land mammals due to short data series. Indications of Hg increase in humans since pre industrial times. Possible decrease in Norway since 1970
In Scandinavia, decreasing Pb and Cd levels measured using the moss monitoring technique clearly reflect changes in atmospheric deposition associated with emission reductions	Unknown	No trends in Hg, Cd, Pb, and Cu in fish based on up to 28 years of monitoring in northern Sweden	Unknown in most areas due to short time series. Decreasing Hg trend in some terrestrial mammals from Yukon, Canada. Indications of increase of Cd in liver, but stable in muscle. Pb concentrations decreased significantly in liver but showed no change in muscle. Hg concentrations in liver and muscle showed no change in reindeer from northwestern Sweden from 1983 to 2000
Combined effects	Combined effects Reduced reproduction in eiders in Alaska was reported as a result of ingestion of lead shot. Other reports included incidents of Pb poisoning in other waterfowl.	Combined effects	Combined effects
Combined effects close to point sources can cause tree death, defoliation and decline in growth	Cd in some ptarmigans from Yukon is high enough to cause kidney damage. Local displacement of birds close to point sources og metals due to lack of food and area degradation	Combined effects observed. Additional information from experimental studies available	Cd in some moose and caribou from Yukon are high enough to cause kidney damage. Possible still births due to Hg

17. Spatial patterns of Hg levels in fish and shellfish are not yet clear. However, recent work has revealed that Canadian subsistence consumption advisory limits for Hg in muscle tissue of piscivorous freshwater fish are exceeded over extensive areas of Canada. Increased circumpolar data coverage is needed to identify potential spatial patterns for these species.

18. Mercury levels in seabirds were generally lower in the Barents Sea than in Greenland, Canada, and northwest Siberia, except for four species of eider, which had similar Hg levels in all these areas. In northeast Siberia, long-tailed ducks (oldsquaw, *Clangula hyemalis*) and herring gulls (*Larus hyperboreus*) show elevated Hg levels compared to Greenland and eastern Canada. Within the Canadian Arctic, Hg levels in seabird eggs are generally higher at sites at higher latitudes.

19. Mercury levels in ringed seal (*Phoca hispida*), beluga whales, and polar bear (*Ursus maritimus*) in western Arctic Canada were found to decrease to the east. New data from Alaska and from western Canada for beluga whales, and new data from East Greenland (Tuna) for polar bear, are generally consistent with this spatial pattern. However, polar bear from Alaska had lower Hg levels than polar bear from western Arctic Canada. Mercury levels in ringed seal from Labrador were variable and in some cases similar to levels reported for the western Canadian Arctic in the first AMAP assessment, and higher than those found in Greenland, Svalbard, and Alaska.

7.2.1.4. **Effects**

20. Limited work on the effects of Hg on Arctic biota has been completed. Of the 152 new seabird liver and kidney samples reported, none were above toxicity thresholds previously established for Hg in non-Arctic species tested in the laboratory. Further assessment will require more sensitive measures of effects in Arctic species.

21. Mercury levels in up to 30% of eggs from American peregrine falcon (*Falco peregrinus anatum*) exceeded the critical threshold for reproductive effects. The falcon is suffering from reduced productivity, but no causal association has yet been established.

22 It is uncertain whether Hg poses a health threat to the most highly exposed groups of marine mammals found in the western Canadian Arctic, and pilot whales (*Globicephala melas*) from the Faroe Islands.

7.2.2. **Lead**

The reduction of Pb emissions through the decreased use of leaded gasoline around the globe is among the more successful regulatory actions in recent times. As a result, Pb deposition has decreased, although Pb levels in Arctic biota do not yet reflect this change. Lead shot from hunting remains a problem.

7.2.2.1. **Sources and pathways**

1. Vehicular traffic represents the primary source of atmospheric emissions of Pb (88739 t, 74%). Non-ferrous metal production is second (14815 t, 12%), then stationary fossil fuel combustion (11690 t, 10%), iron and steel production (2926 t, 2%), waste disposal (821 t, <1%), and cement production (268 t, <1%) for a total of 119259 t.

2. The largest contributions of Pb are emitted from Asia (51212 t, 43%). Other emissions come from Europe (28091 t, 24%), North America (17015 t, 14%), Africa (11349 t, 10%), South America (9118 t, 8%), and Australia and Oceania (2474 t, 2%).

3. Of global emissions, the regions emitting the most airborne Pb to the Arctic include Europe and the Asian part of Russia. Between 2% and 5% of the total anthropogenic Pb emissions from these areas is deposited in the Arctic.

4. Riverine transport of Pb to the Arctic Basin is comparable to the amount transported by the atmosphere.

7.2.2.2. **Spatial patterns and temporal trends**

5. Based on the Greenland Summit deep-drilling program ice core-based reconstruction of Arctic maximum metal deposition fluxes, there was a 12-fold increase in Pb deposition during the 1960s and 1970s. However, due to significant earlier contributions of Pb that had occurred by 1800, maximum deposition fluxes in the 1970s represent a 200-fold increase over pre-industrial levels.

6. Elimination of Pb additives in gasoline by environmental regulation during the 1970s and 1980s, particularly in Europe and North America, is believed directly responsible for recent decreasing trends in Pb emissions. Industrial emissions are also being reduced. These reductions are, in part, a side effect of measures introduced to reduce emissions of particles and acidifying gases.

7. Based on trends in stable Pb isotope ratios at the Greenland Summit site, the region is influenced by both North American and Eurasian air masses. The North American Pb contribution showed a rapid decline following the introduction of unleaded gasoline.

8. In Scandinavia, decreasing Pb levels measured using the moss monitoring technique clearly reflect changes in atmospheric deposition associated with emission reductions. While Pb levels in moss samples provide an effective monitor for changes in atmospheric deposition, decreasing Pb emissions are not yet reflected in the tissues of Arctic biota, where Pb levels appear to be stable.

9. Waterfowl and human subsistence exposures appear to continue in some areas (e.g., Alaska, Greenland) due to the presence in the environment of Pb shot used for hunting.

7.2.2.3. **Effects**

10. Reduced reproduction in eiders (*Somateria molissima*) in Alaska was reported as a result of ingestion of Pb shot. Other reports included incidents of Pb poisoning in other waterfowl.

7.2.3. Cadmium

Cadmium levels in the Arctic environment have increased since pre-industrial times but it remains difficult to separate anthropogenic and natural sources where higher concentrations are found in wildlife. Although levels in some wildlife and marine organisms are high enough for concern, effects have not yet been detected in wild populations.

7.2.3.1. **Sources and pathways**

1. Based on 1995 emission inventories, non-ferrous production of zinc (Zn) and Pb is the major source of global anthropogenic Cd emissions to the atmosphere (2171 t, 73%). Other sources include stationary fossil fuel combustion (691 t, 23%) with relatively small amounts released from iron and steel production (64 t, 2%), waste disposal (40 t, 1%), and cement production (17 t, <1%) for a total of 2983 t.
2. The major contributor of Cd emissions is Asia (1463 t, 49%), then North America (482 t, 16%), South America (452 t, 15%), Europe (362 t, 12%), Africa (172 t, 6%), and Australia and Oceania (52 t, 2%). Emissions are declining in Europe and North America.
3. Less than 2% of airborne Cd emitted into the global atmosphere is deposited in the Arctic.
4. Riverine transport of Cd to the Arctic Basin is comparable to the amount transported by the atmosphere.

7.2.3.2. **Spatial patterns and temporal trends**

5. Data from the Greenland Summit ice core indicate that since the onset of the Industrial Revolution (i.e., 1800 AD), an eight-fold increase in deposition fluxes for Cd occurred with maximum levels reached during the 1960s and 1970s.
6. Since the 1970s (i.e., subsequent to the ice core maximum), monitoring records of moss in northern Sweden indicate that atmospheric deposition of Cd is decreasing. Recent declines may be related to a two- to three-factor decline in Cd global emissions from non-ferrous metal processing during the 1980s and 1990s.
7. In one of the longest records available on tissue concentrations of metals in Arctic biota, a statistically significant increase in Cd was found in liver of reindeer (*Rangifer tarandus*) collected in northern Sweden between 1983 and 1999. Concentrations in muscle tissue, however, did not increase.
8. Cadmium concentrations in tissues of some terrestrial birds and mammals are high compared with global background levels. Cadmium concentrations in marine organisms from large parts of the Arctic exceed levels in other regions of the globe. However, Cd levels in Arctic biota and the biotic environment, in general, do not appear to be increasing.
9. Cadmium in shellfish may be associated with local sources, as found in central west Greenland. Due to poor circumpolar coverage for fish, only limited spatial comparisons are possible. Some differences in Cd concentration are found among sub-regions, but these probably reflect natural rather than anthropogenic causes.
10. Spatial patterns in concentrations of Cd found in seabirds may be partly explained by differences in overwintering areas used by different populations. In general, seabirds appear to have higher Cd levels in northeast Siberia and lower levels in the Barents Sea.
11. The highest concentrations of Cd in ringed seals, beluga whales and polar bears reported from the eastern Canadian Arctic and northwest Greenland during the first AMAP assessment are partly confirmed by new data. Cadmium concentrations in ringed seals from the eastern Canadian Arctic and Greenland were higher than in Alaska and Svalbard. Cadmium levels in polar bears from Alaska were similar to those observed previously in western Canada.

7.2.3.3. **Effects**

12. Cadmium levels in some reindeer/caribou, moose (*Alces alces*), and ptarmigan (*Lagopus mutus*) from the Yukon Territory (Canada) as well as those in seabirds and marine mammals from northwest Greenland and the Faroe Islands may be high enough to cause kidney damage.
13. Pathological investigations on ringed seal, beluga and bowhead whale (*Balaena mysticetus*), including examinations of animals with Cd levels above expected effects thresholds, have not identified Cd-related effects.

7.2.4. All metals

7.2.4.1. **Sources and pathways**

1. The three main anthropogenic sources of heavy metals to the atmosphere are fossil fuel combustion, non-ferrous metal production, and waste incineration. Of these, emissions from waste incineration are the least understood and probably underestimated, introducing uncertainty in source inventory emission estimates for all metals and particularly for Hg.
2. Asian sources of emissions to the global environment are clearly the largest for all metals. Of these emissions, an estimated five to ten percent are deposited in the Arctic. Given expected population increases and predominant use of small coal heating fires in China, this region is likely to become an in

creasingly important anthropogenic source of heavy metals to the Arctic, even as industrial emissions are reduced in other regions owing to technological advances.

3. The highest concentrations of atmospheric heavy metals in Arctic air occur in the vicinity of smelter complexes on the Kola Peninsula and at Norilsk in Russia, and result from emissions from these smelters.

4. All heavy metal concentrations show strong seasonal variation above the Arctic Circle (High Arctic). During winter, about two-thirds of the heavy metals (except Hg) in air in the High Arctic are transported from Eurasia, particularly from the Kola Peninsula, the Norilsk region, the Urals, and the Pechora Basin. Five to ten percent of these emissions are deposited in the High Arctic. The remaining third in High Arctic air in winter is transported from industrial regions in Europe and North America. In summer, local sources dominate the contamination of the High Arctic.

5. Although atmospheric transport represents a major transport mechanism to the Arctic, particularly for Hg, rivers provide another major transport mechanism for some metals reaching the Arctic Ocean. Riverine transport pathways are equally important for Pb and Cd and rivers provide the main transport mechanism for Zn.

7.2.4.2. **Spatial patterns and temporal trends**

6. Concentrations of heavy metals measured in air on the Kola Peninsula are comparable to concentrations found in the most polluted regions of Europe and North America. With the exception of regions around point sources, however, concentrations in High Arctic air are one order of magnitude lower than concentrations in other remote non-Arctic locations and about two orders of magnitude lower than concentrations around major point sources on the Kola Peninsula. Exceptions in the Arctic include both reactive gaseous Hg and particulate Hg that can exceed non-Arctic values by several orders of magnitude.

7. Concentrations of most heavy metals measured in subarctic air have decreased over the last two decades. Exceptions are vanadium (V) and nickel (Ni), where increased emissions have been attributed to increased use of oil combustion to replace coal combustion in the power generating industry.

8. Soils respond slowly to changes in atmospheric deposition and provide a reservoir of contaminant elements that can be taken up by organisms as they interact with soils. However, moss reflects contemporary atmospheric deposition of heavy metals.

7.2.4.3. **Effects**

9. Evidence of biological effects is clearest around smelters and foundries in the Arctic where obvious changes occur in the surrounding ecosystem from a combination of sulfur fumigation and deposition of both acids and heavy metals (e.g., Cu, Ni).

10. Other than point source problems, there are few studies on biological effects on Arctic plants and animals, and fewer still that link effects in Arctic biota to heavy metal pollution.

11. Critical tissue effect thresholds are unknown for most Arctic species making the potential risks associated with observed tissue levels difficult to interpret or predict. In several cases, observed tissue levels in Arctic biota exceed exposure thresholds for effects in non-Arctic species.

7.2.4.4. **Emerging issues**

12. Greenland ice cores and recent snow samples reveal a recent and significant increase in the platinum group elements platinum (Pt), palladium (Pd), and rhodium (Rh). Concentrations of Rh in the late 1990s were 40 to 120 times higher than in ancient ice indicating that deposition of these elements in the Greenland Arctic is virtually all from industrial sources. Deposition is probably related to the increasing use of catalytic converters in automobiles. There is no indication of deposition rates stabilizing or decreasing. The toxicity and bioaccumulation potential for these elements is largely unknown, thus there is little information from which to assess likely implications of environmental increases for soils, plants, wildlife, and humans.

13. Hydrology, currents, ice, winds, and temperature are the major factors in determining the main pathways of contaminant transport, and consequently past and present fluxes of heavy metals to the Arctic. These factors are already being influenced by observed climatic changes, and more changes are predicted to occur in the future. As a result, significant alterations in metal transport pathways can be expected. This may be particularly important for Hg flux since the Hg cycle may be impacted in several ways (e.g., the influence of annual ice extent and open water on reactive halogen chemistry, changes in organic carbon cycling in the upper ocean).

7.3. Conclusions and recommendations

Based on scientific findings, the following is concluded, and recommendations for action are suggested.

7.3.1. **Mercury**

1. In the Arctic, Hg is removed from the atmosphere and deposits on snow in a form that is bioavailable (i.e., can be taken up by some microorganisms). This recently discovered process is linked to polar sunrise, and appears to be unique to high latitude areas. The resulting enhanced deposition may mean that the Arctic plays a previously unrecognized role as an important sink in the global Hg cycle. Some of the Hg deposited on snow is released to the environment at snowmelt in bioavailable forms at the onset of animal and plant reproduction and rapid growth.

Although poorly understood, this process may be the chief mechanism for transferring atmospheric Hg to Arctic food webs.

Recommendation:

- Expand and accelerate research on critical aspects of the Hg cycle and budget in the Arctic. Such research should include long-range transport and transformation processes, Hg deposition mechanisms, Hg transformation at snowmelt to bioavailable forms, the dynamics of uptake by biota leading to exposure, and effects on reproduction and growth.

2. Despite substantial Hg emission reductions in North America and Western Europe during the 1980s, global Hg emissions may, in fact, be increasing. Mercury emissions from waste incineration are probably underestimated. The burning of coal in small-scale power plants and residential heaters, principally in Asia, are major potential sources of current Hg emissions. These emissions are likely to increase significantly due to economic and population growth in this region.

Recommendation:

- Promote efforts at global, regional and national levels to better quantify all sources of Hg and report results in a consistent and regular manner to improve emission inventories. Particular effort should focus on estimating contributions made by burning coal for residential heating and small-scale power plants as well as by waste incineration.
  Continue to implement technologies that reduce Hg emissions and look for alternative strategies to reduce emissions not readily amenable to current approaches.

3. There is a trend of increasing Hg levels in marine birds and mammals in some regions of the Canadian Arctic, and some indications of increases in West Greenland (Kitaa). The effects of these levels are not well understood. However, there are also examples of stable or decreasing levels in other regions, perhaps indicating the importance of local or regional processes.

Recommendation:

- Continue temporal trend monitoring to obtain longer, more statistically reliable trend analysis, and assess the effects of Hg in key indicator media and biota. These data will enable assessment of whether the measures taken in accordance with the LRTAP (Long-range Transboundary Air Pollution) Protocol are effective in reducing Hg levels in the Arctic.

4. Current Hg exposures may pose a health risk to some people and animals in the Arctic. These risks may include subtle neurobehavioral effects as well as possible reproductive effects. Reducing exposure to Hg can only be addressed by regional and global action to reduce worldwide emissions.

Recommendations:

- Acknowledge the assessment for global action undertaken by the United Nations Environmental Program and its resulting proposals and take appropriate steps to ensure that Arctic concerns are adequately addressed. Promote the development of regional and global actions.
- Study effects in Arctic species that exhibit Hg at tissue levels of concern. Focus on effects monitoring, critical tissue effect thresholds, relationships between indicators of exposure (e.g., biomarkers) and observed effects in Arctic biota. Fill existing data gaps.

7.3.2. Lead

1. Significant reduction in the deposition of atmospheric Pb occurred in Arctic regions where the use of leaded gasoline was banned. Arctic-wide elimination of leaded gasoline use will reduce Pb exposure in other regions of the Arctic. Although levels in wildlife and fish have not measurably declined, probably reflecting continued uptake from the large reservoir of Pb deposited in soils and sediments, Pb levels in the environment are expected to diminish over time if current trends continue.

Recommendation:

- Support continued efforts to eliminate the use of leaded gasoline in all Arctic regions.

2. Certain regions of the Arctic (e.g., Alaska, Greenland) contain elevated Pb levels in the environment because of past or current use of Pb shot by hunters.

Recommendation:

- Encourage a complete ban on the use of Pb shot in the Arctic and improve enforcement.

7.3.3. Cadmium

1. Based on existing toxicity thresholds for non-Arctic species, Cd levels in some Arctic seabirds are high enough to cause kidney damage. However, monitoring data on Cd in the abiotic and biotic environment to date provide no conclusive evidence of trends or effects. None-the-less, Cd accumulates in birds and mammals and not enough is known about possible effects.

Recommendation:

- Continue monitoring Cd in the Arctic to support human and wildlife exposure estimates.

7.3.4. All heavy metals

1. Levels of Pt, Pd, and Rh have increased rapidly in Greenland snow and ice since the 1970s. These elements are used in automobile catalytic converters to reduce hydrocarbon pollution. The toxicity and bioaccumulation of these elements is largely unknown, which prevents assessment of their potential impact in the Arctic.

Recommendation:

- Monitor trends of Pt, Pd, and Rh in the Arctic.

2. Climate change is already influencing Arctic hydrology, currents, ice, winds, and temperature which are the major factors determining pathways of contaminant transport, and consequently past and present fluxes of heavy metals to the Arctic. More changes are predicted to occur in the future. As a result, significant alterations in metal transport pathways can be expected.

Recommendations:

- Continue mass balance studies and assessment of transport pathways for heavy metals, in air, rivers, and oceans across the Arctic.
- Expand research to better understand the processes of climate change that may influence spatial patterns and temporal trends.

3. Conclusions from this assessment are constrained by the limited availability of data for establishing spatial patterns and temporal trends, and understanding transport pathways and transformation processes that lead to environmental exposure. To improve detection of trends, better temporal and spatial coverage, harmonization of sampling, analytical, and reporting protocols, and improved understanding of key processes are required.

Recommendations:

- Generate contemporary inventories for Hg, Pb, Cd and the emissions of other priority contaminants, spatially distributed for use in hemispheric-scale models.
- Continue process studies to evaluate observed geographical differences and spatial patterns in the Arctic to obtain the 10 to 20 years of data using consistent protocols that are required to recognize emerging patterns. Increase efforts on assessing spatial patterns for more spatially fixed environmental media such as soils and vegetation.
- Support long-term time series data collection for biotic tissues. Develop studies that evaluate the relationship between soft and hard tissue concentrations in biota that will improve links between long-term time series and contemporary concentrations.
- Extend the length of existing temporal monitoring studies of biota, and initiate the collection and banking of biota specimens in areas not currently covered (e.g., North Atlantic marine mammals and birds; Russian biota in general).
- Continue work to develop standard protocols for sampling, analysis, and reporting among the AMAP countries to ensure inter-comparability of data sets.

4. The current absence of observable effects in Arctic biota from exposure to heavy metals does not confirm that effects are not occurring. Data sets that link exposure pathways, tissue concentrations, and a range of subtle responses to heavy metal exposure are needed to better identify effects.

Recommendation:

- Conduct effects studies in priority species experiencing heavy metal exposures of concern that link exposure profiles with tissue concentrations within individual animals and include an assessment of histopathological, behavioral, and reproductive parameters.

Table A1. Metals in air.

Country	Location	Sampling place	Coordinate	Year	Unit	Lead		Cadmium		Mercury		Remark	Reference
						mean	SD	mean	SD	mean	SD		
Canada	Alert, Nunavut		62°30'N, 28°28'W	1995	ng/m^3					1.63	0.14	Winter (Jan-Mar)	1
”	” ”		” ”	1996	”					1.48	0.4	”	1
”	” ”		” ”	1997	”					1.48	0.19	”	1
”	” ”		” ”	1998	”					1.62	0.23	”	1
”	” ”		” ”	1999	”					1.59	0.28	”	1
”	” ”		” ”	2000	”					1.66	0.28	”	1
”	” ”		” ”	1995-2000	”					1.57		”	1
”	” ”		” ”	1995	”					1.41	0.57	Spring (Apr-Jun)	1
”	” ”		” ”	1996	”					1.4	0.63	”	1
”	” ”		” ”	1997	”					1.21	0.62	”	1
”	” ”		” ”	1998	”					1.38	0.49	”	1
”	” ”		” ”	1999	”					1.26	0.65	”	1
”	” ”		” ”	2000	”					1.13	0.6	”	1
”	” ”		” ”	1995-2000	”					1.3		”	1
”	” ”		” ”	1995	”					1.88	0.31	Summer (Jul-Sep)	1
”	” ”		” ”	1996	”					1.79	0.21	”	1
”	” ”		” ”	1997	”					1.88	0.28	”	1
”	” ”		” ”	1998	”					1.67	0.24	”	1
”	” ”		” ”	1999	”					1.7	0.24	”	1
”	” ”		” ”	2000	”					1.7	0.28	”	1
”	” ”		” ”	1995-2000	”					1.77		”	1
”	” ”		” ”	1995	”					1.54	0.04	Autumn (Oct-Dec)	1
”	” ”		” ”	1996	”					1.55	0.06	”	1
”	” ”		” ”	1997	”					1.53	0.08	”	1
”	” ”		” ”	1998	”					1.64	0.09	”	1
”	” ”		” ”	1999	”					1.49	0.14	”	1
”	” ”		” ”	2000	”					1.47	0.09	”	1
”	” ”		” ”	1995-2000	”					1.54		”	1
”	” ”		” ”	1995	”					1.62	0.41	Annual	1
”	” ”		” ”	1996	”					1.56	0.42	”	1
”	” ”		” ”	1997	”					1.52	0.44	”	1
”	” ”		” ”	1998	”					1.58	0.31	”	1
”	” ”		” ”	1999	”					1.5	0.43	”	1
”	” ”		” ”	2000	”					1.47	0.44	”	1
”	” ”		” ”	1995-2000	”					1.54		”	1
Russia	Russian Arctic	Wrangel Island	71°N, 180°E	1986	”			0.96					2
”	” ”	Northern Land	71°N, 100°E	1988	”			0.09					2
”	” ”	East Arctic		1985-1989	”			0.59					2
”	” ”	Ob-Yenisey coastal waters		1994	”	4.2		3.0					3

References

1. B. Schroeder unpubl. data, 2002; 2. Shevchenko *et al*., 1995, 1997, 1999, 2000; Vinogradova, 1996; Review, 1996.

Table A2. Metals in snow/ice.

Country	Location	Coordinate	Date	Elevation, m asl	Depth, cm from	Depth, cm to	Snow density, g/cc	Unit	Zinc	Cadmium	Lead	Remark	Reference
Russia	Dikson (Coastal Siberia)	73°30'N, 80°36'E	26 Apr. 1993	50	0	47	0.45	pg/g	106.0	2.30	111.40	A	1
Russia	Academii Nauk (O. Komsomolets)	80°36'N, 94°58'E	13 May 1993	850	0	45	0.47	pg/g	225.0	5.10	107.20	A	1
	Arctic Ocean 1	84°09'N, 95°47'E	11 May 1993	1	0	34	0.39	pg/g	401.0	9.70	314.70	B	1
	Arctic Ocean 2	86°29'N, 96°02'E	11 May 1993	1	0	47	0.26	pg/g	247.0	2.30	123.40	B	1
	Arctic Ocean 3	89°54'N, 10°44'W	12 May 1993	1	0	39	0.30	pg/g	162.0	2.33	134.20	B	1
	Arctic Ocean 4	89°49'N, 82°49'W	13 May 1993	1	0	38	0.29	pg/g	177.0	2.80	112.20	B	1
Canada	Devon Ice Cap (Devon Island)	75°24'N, 82°42'W	5 Apr. 1993	1700	0	34	0.40	pg/g	111.0	2.10	47.30	C	1
Canada	Melville Ice Cap (Melville Island)	75°24'N, 114°54'W	29 Mar. 1993	700	0	30	0.43	pg/g	310.2	6.40	161.90	C	1
Canada	Agassiz Ice Cap (Ellesmere Island)	80°42'N, 72°53'W	26 Mar. 1993	1600	0	31	0.37	pg/g	118.0	1.10	79.10	C	1
Russia	Leningradskii Ice Cap (O. Bolshevik)	78°42'N, 103°49'E	22 May 1994	940	0	5	0.48	pg/g	505.0	13.75	443.30	D	1
Russia	Leningradskii Ice Cap (O. Bolshevik)	78°42'N, 103°49'E	22 May 1994	940	5	18	0.48	pg/g	303.0	8.75	219.50	D	1
Russia	Leningradskii Ice Cap (O. Bolshevik)	78°42'N, 103°49'E	22 May 1994	940	18	28	0.48	pg/g	346.0	9.17	241.05	D	1
Russia	Leningradskii Ice Cap (O. Bolshevik)	78°42'N, 103°49'E	22 May 1994	940	28	37	0.48	pg/g	130.0	0.42	78.87	D	1
Russia	Leningradskii Ice Cap (O. Bolshevik)	78°42'N, 103°49'E	22 May 1994	940	37	48	0.48	pg/g	122.0	2.50	51.39	D	1
Russia	Leningradskii Ice Cap (O. Bolshevik)	78°42'N, 103°49'E	22 May 1994	940	48	60	0.48	pg/g	209.0	1.67	22.79	D	1
Russia	Total accumulation = 60 cm	78°42'N, 103°49'E	22 May 1994	940*				pg/g	249.1	5.43	150.49	D	1
Russia	Academii Nauk (O. Komsomolets)	80°35'N, 94°57'E	13 May 1994	850	0	6	0.43	pg/g	786.0	21.38	648.32	D	1
Russia	Academii Nauk (O. Komsomolets)	80°35'N, 94°57'E	13 May 1994	850	6	12	0.43	pg/g	178.0	2.39	142.59	D	1
Russia	Academii Nauk (O. Komsomolets)	80°35'N, 94°57'E	13 May 1994	850	12	22	0.43	pg/g	88.0	2.19	70.22	D	1
Russia	Academii Nauk (O. Komsomolets)	80°35'N, 94°57'E	13 May 1994	850	22	31	0.43	pg/g	62.0	1.00	52.49	D	1
Russia	Academii Nauk (O. Komsomolets)	80°35'N, 94°57'E	13 May 1994	850	31	46	0.43	pg/g	397.0	3.36	89.07	D	1
Russia	Academii Nauk (O. Komsomolets)	80°35'N, 94°57'E	13 May 1994	850	46	70	0.43	pg/g	181.0	0.49	52.10	D	1
Russia	Total accumulation = 70 cm	80°35'N, 94°57'E	13 May 1994	850*				pg/g	292.0	3.93	141.78	D	1
Russia	Z. Aleksandry	80°36'N, 46°30'E	16 May 1994	345	0	5	0.45	pg/g	343.0	13.30	261.42	D	1
Russia	Z. Aleksandry	80°36'N, 46°30'E	16 May 1994	345	5	22	0.45	pg/g	1356.0	164.60	5482.29	D	1
Russia	Z. Aleksandry	80°36'N, 46°30'E	16 May 1994	345	22	40	0.45	pg/g	589.0	23.30	690.51	D	1
Russia	Z. Aleksandry	80°36'N, 46°30'E	16 May 1994	345	40	75	0.45	pg/g	332.0	7.50	307.72	D	1
Russia	Z. Aleksandry	80°36'N, 46°30'E	16 May 1994	345	75	100	0.45	pg/g	53.0	0.83	34.91	D	1
Russia	Z. Aleksandry	80°36'N, 46°30'E	16 May 1994	345	100	129	0.45	pg/g	43.0	0.42	11.55	D	1
Russia	Total accumulation = 129 cm	80°36'N, 46°30'E	16 May 1994	345*				pg/g	826.0	59.66	1981.89	D	1
Russia	O. Greem Bell	80°47'N, 63°35'E	15 May 1994	532	0	12	0.38	pg/g	408.0	7.98	418.79	D	1
Russia	O. Greem Bell	80°47'N, 63°35'E	15 May 1994	532	12	19	0.38	pg/g	415.0	6.86	446.13	D	1
Russia	O. Greem Bell	80°47'N, 63°35'E	15 May 1994	532	19	31	0.38	pg/g	593.0	14.89	319.44	D	1
Russia	O. Greem Bell	80°47'N, 63°35'E	15 May 1994	532	31	47	0.38	pg/g	59.0	0.42	23.53	D	1
Russia	O. Greem Bell	80°47'N, 63°35'E	15 May 1994	532	47	99	0.38	pg/g	137.0	5.54	104.66	D	1
Russia	Total accumulation = 99 cm	80°47'N, 63°35'E	15 May 1994	532*				pg/g	383.1	10.29	296.67	D	1
Russia	O. Ushakova	80°54'N, 80°00'E	15 May 1994	290	0	8	0.44	pg/g	567.0	15.99	757.19	D	1
Russia	O. Ushakova	80°54'N, 80°00'E	15 May 1994	290	8	22	0.44	pg/g	733.0	24.34	612.24	D	1
Russia	O. Ushakova	80°54'N, 80°00'E	15 May 1994	290	22	36	0.44	pg/g	46.0	0.42	27.36	D	1
Russia	O. Ushakova	80°54'N, 80°00'E	15 May 1994	290	36	54	0.44	pg/g	278.0	12.59	205.42	D	1
Russia	Total accumulation = 54 cm	80°54'N, 80°00'E	15 May 1994	290*				pg/g	340.8	11.69	311.82	D	1
Canada	Penny Ice Cap (Baffin Island)	67°15'N,65°48W	10 Apr. 1995	1850	0	28	0.41	pg/g	191.5	2.02	138.16	E	1
Canada	Penny Ice Cap (Baffin Island)	67°15'N, 65°48'W	10 Apr. 1995	1850	28	63	0.31	pg/g	111.7	0.68	63.77	E	1
Canada	Penny Ice Cap (Baffin Island)	67°15'N, 65°48'W	10 Apr. 1995	1850*				pg/g	147.2	1.37	102.01	E	1
Canada	Penny Ice Cap (Baffin Island)	67°15'N, 65°48'W	10 Apr. 1995	1850	0	28	0.41	pg/g	285.9	4.39	169.44	E	1
Canada	Penny Ice Cap (Baffin Island)	67°15'N, 65°48'W	10 Apr. 1995	1850	28	63	0.31	pg/g	67.8	1.39	45.75	E	1
Canada	Penny Ice Cap (Baffin Island)	67°15'N, 65°48'W	10 Apr. 1995	1850*				pg/g	164.7	2.93	109.34	E	1
Canada	Penny Ice Cap (Baffin Island)	67°15'N, 65°48'W	10 Apr. 1995	1850	0	28	0.41	pg/g	174.2	3.29	117.79	E	1
Canada	Penny Ice Cap (Baffin Island)	67°15'N,65°48' W	10 Apr. 1995	1850	28	63	0.31	pg/g	114.4	1.65	50.57	E	1
Canada	Penny Ice Cap (Baffin Island)	67°15'N, 65°48'W	10 Apr. 1995	1850*				pg/g	140.9	2.50	85.13	E	1
Canada	Total accumulation = 63 cm	67°15'N, 65°48'W	10 Apr. 1995	1850**				pg/g	150.9	1.7	74.12	E	1
Canada	Agassiz Ice Cap (Ellesmere Island)	80°48'N, 72°54'W	20 Mar. 1995	1600	0	47	0.33	pg/g	131.6	1.06	55.74	E	1
Canada	Agassiz Ice Cap (Ellesmere Island)	80°48'N, 72°54'W	20 Mar. 1993	1600	0	47	0.33	pg/g	103.7	0.65	48.25	E	1
Canada	Total accumulation = 47 cm	80°48'N, 72°54'W	20 Mar. 1995	1600***				pg/g	117.7	0.86	52.00	E	1
Canada	Devon Ice Cap (Devon Island)	75°24'N, 82°42'W	23 Mar. 1995	1700	0	35	0.31	pg/g	161.2	0.64	72.45	E	1
Canada	Devon Ice Cap (Devon Island)	75°24'N, 82°42'W	23 Mar. 1995	1700	0	36	0.31	pg/g	141.0	1.36	50.83	E	1
Canada	Devon Ice Cap (Devon Island)	75°24'N, 82°42'W	23 Mar. 1995	1700	0	36	0.31	pg/g	110.4	1.10	43.00	E	1
Canada	Total accumulation = 36 cm	75°24'N, 82°42'W	23 Mar. 1995	1700***				pg/g	137.5	1.03	55.43	E	1
Canada	Meighen Ice Cap (Meighen Island)	80°00'N, 99°09'W	10 Mar. 1995	265	0	27	0.44	pg/g	385.2	6.38	285.85	E	1
Canada	Meighen Ice Cap (Meighen Island)	80°00'N, 99°09'W	10 Mar. 1995	265	27	50	0.44	pg/g	265.0	3.49	137.50	E	1
Canada	Meighen Ice Cap (Meighen Island)	80°00'N, 99°09'W	10 Mar. 1995	265	50	79	0.44	pg/g	138.0	2.13	68.81	E	1
Canada	Total accumulation = 79 cm	80°00'N, 99°09'W	10 Mar. 1995	265*				pg/g	259.5	3.98	162.99	E	1
Canada	Melville Ice Cap (Melville Island)	75°24'N, 114°54'W	12 Mar. 1995	700	0	50	0.28	pg/g	153.0	0.51	38.66	E	1
Canada	Melville Ice Cap (Melville Island)	75°24'N, 114°54'W	12 Mar. 1995	700	0	50	0.28	pg/g	191.3	5.19	101.27	E	1
Canada	Melville Ice Cap (Melville Island)	75°24'N, 114°54'W	12 Mar. 1995	700	0	50	0.28	pg/g	260.9	8.49	159.73	E	1
Canada	Melville Ice Cap (Melville Island)	75°24'N, 114°54'W	12 Mar. 1995	700	0	50	0.28	pg/g	159.8	4.27	100.68	E	1
Canada	Total accumulation = 50 cm	75°24'N, 114°54'W	12 Mar. 1995	700***				pg/g	191.3	4.49	100.09	E	1
Canada	Penny Ice Cap (Baffin Island)	67°15'N,65°48'W	30 Mar. 1996	1850	0	10	0.35	pg/g	58.5	0.89		F	1
Canada	Penny Ice Cap (Baffin Island)	67°15'N,65°48'W	30 Mar. 1996	1850	10	20	0.35	pg/g	142.4	2.78		F	1
Canada	Penny Ice Cap (Baffin Island)	67°15'N,65°48'W	30 Mar. 1996	1850	20	30	0.35	pg/g	179.3	8.24		F	1
Canada	Penny Ice Cap (Baffin Island)	67°15'N,65°48'W	30 Mar. 1996	1850	30	40	0.35	pg/g	100.3	0.68		F	1

Table A2.. Metals in snow/ice, continued.

Country	Location	Coordinate	Date	Elevation, m asl	Depth, cm from	Depth, cm to	Snow density, g/cc	Unit	Zinc	Cadmium	Lead	Remark	Reference
Canada	Penny Ice Cap (Baffin Island)	67°15'N,65°48'W	30 Mar. 1996	1850	40	50	0.35	pg/g	347.0	10.78		F	1
Canada	Penny Ice Cap (Baffin Island)	67°15'N,65°48'W	30 Mar. 1996	1850	50	60	0.35	pg/g	328.2	4.58		F	1
Canada	Penny Ice Cap (Baffin Island)	67°15'N,65°48'W	30 Mar. 1996	1850	60	70	0.35	pg/g	330.7	12.53		F	1
Canada	Penny Ice Cap (Baffin Island)	67°15'N,65°48'W	30 Mar. 1996	1850	70	80	0.35	pg/g	508.3	10.45		F	1
Canada	Penny Ice Cap (Baffin Island)	67°15'N,65°48'W	30 Mar. 1996	1850	80	90	0.35	pg/g	276.2	5.73		F	1
Canada	Penny Ice Cap (Baffin Island)	67°15'N,65°48'W	30 Mar. 1996	1850	90	100	0.35	pg/g	377.2	12.23		F	1
Canada	Total accumulation = 100 cm	67°15'N, 65°48'W	30 Mar. 1996	1850*				pg/g	264.8	6.90		F	1
Canada	Agassiz Ice Cap (Ellesmere Island)	80°48'N, 72°54'W	25 Mar. 1996	1600	0	10	0.32	pg/g	139.00	3.54		F	1
Canada	Agassiz Ice Cap (Ellesmere Island)	80°48'N, 72°54'W	25 Mar. 1996	1600	10	20	0.32	pg/g	281.90	1.47		F	1
Canada	Agassiz Ice Cap (Ellesmere Island)	80°48'N, 72°54'W	25 Mar. 1996	1600	20	32	0.32	pg/g	260.20	4.80		F	1
Canada	Agassiz Ice Cap (Ellesmere Island)	80°48'N, 72°54'W	25 Mar. 1996	1600	32	38	0.32	pg/g	181.60	2.98		F	1
Canada	Agassiz Ice Cap (Ellesmere Island)	80°48'N, 72°54'W	25 Mar. 1996	1600****				pg/g	221.60	3.30		F	1
Canada	Agassiz Ice Cap (Ellesmere Island)	80°48'N, 72°54'W	25 Mar. 1996	1600	0	11	0.32	pg/g	191.80	2.84		F	1
Canada	Agassiz Ice Cap (Ellesmere Island)	80°48'N, 72°54'W	25 Mar. 1996	1600	11	23	0.32	pg/g	360.60	3.36		F	1
Canada	Agassiz Ice Cap (Ellesmere Island)	80°48'N, 72°54'W	25 Mar. 1996	1600	23	30	0.32	pg/g	165.10	3.32		F	1
Canada	Agassiz Ice Cap (Ellesmere Island)	80°48'N, 72°54'W	25 Mar. 1996	1600	30	40	0.32	pg/g	526.30	9.05		F	1
Canada	Agassiz Ice Cap (Ellesmere Island)	80°48'N, 72°54'W	25 Mar. 1996	1600*****				pg/g	383.3	4.90		F	1
Canada	Total accumulation = 39 cm (avg)	80°48'N, 72°54'W	25 Mar. 1996	1600******				pg/g	302.45	4.10		F	1
Canada	Devon Ice Cap (Devon Island)	75°24'N, 82°42'W	10 Mar. 1996	1700	0	12	0.34	pg/g	60.3	2.70		F	1
Canada	Devon Ice Cap (Devon Island)	75°24'N, 82°42'W	10 Mar. 1996	1700	12	24	0.34	pg/g	246.6	10.70		F	1
Canada	Devon Ice Cap (Devon Island)	75°24'N, 82°42'W	10 Mar. 1996	1700	24	36	0.34	pg/g	262.4	11.00		F	1
Canada	Devon Ice Cap (Devon Island)	75°24'N, 82°42'W	10 Mar. 1996	1700	36	48	0.34	pg/g	353.0	14.00		F	1
Canada	Total accumulation = 48 cm	75°24'N, 82°42'W	10 Mar. 1996	1700*				pg/g	230.6	9.60		F	1
Canada	Meighen Ice Cap (Meighen Island)	80°00'N, 99°09'W	23 Mar. 1996	265	0	5	0.37	pg/g	235.8	3.47		F	1
Canada	Meighen Ice Cap (Meighen Island)	80°00'N, 99°09'W	23 Mar. 1996	265	5	17	0.37	pg/g	142.1	0.28		F	1
Canada	Meighen Ice Cap (Meighen Island)	80°00'N, 99°09'W	23 Mar. 1996	265	17	28	0.37	pg/g	119.9	1.92		F	1
Canada	Meighen Ice Cap (Meighen Island)	80°00'N, 99°09'W	23 Mar. 1996	265	28	38	0.37	pg/g	259.9	13.60		F	1
Canada	Meighen Ice Cap (Meighen Island)	80°00'N, 99°09'W	23 Mar. 1996	265****				pg/g	179.0	4.70		F	1
Canada	Meighen Ice Cap (Meighen Island)	80°00'N, 99°09'W	23 Mar. 1996	265	0	10	0.37	pg/g	255.3	3.27		F	1
Canada	Meighen Ice Cap (Meighen Island)	80°00'N, 99°09'W	23 Mar. 1996	265	10	22	0.37	pg/g	111.6	1.61		F	1
Canada	Meighen Ice Cap (Meighen Island)	80°00'N, 99°09'W	23 Mar. 1996	265	22	33	0.37	pg/g	112.4	4.35		F	1
Canada	Meighen Ice Cap (Meighen Island)	80°00'N, 99°09'W	23 Mar. 1996	265	33	37	0.37	pg/g	90.0	8.01		F	1
Canada	Meighen Ice Cap (Meighen Island)	80°00'N, 99°09'W	23 Mar. 1996	265*****				pg/g	148.3	3.60		F	1
Canada	Total accumulation = 37.5 cm (avg)	80°00'N, 99°09'W	23 Mar. 1996	265*				pg/g	141.3	5.97		F	1
Canada	Melville Ice Cap (Melville Island)	75°24'N, 114°54'W	19 Mar. 1996	700	0	8	0.32	pg/g	365.2	8.86		F	1
Canada	Melville Ice Cap (Melville Island)	75°24'N, 114°54'W	19 Mar. 1996	700	0	8	0.32	pg/g	212.3	5.96		F	1
Canada	Melville Ice Cap (Melville Island)	75°24'N, 114°54'W	19 Mar. 1996	700	0	10	0.32	pg/g	377.0	9.79		F	1
Canada	Melville Ice Cap (Melville Island)	75°24'N, 114°54'W	19 Mar. 1996	700	8	18	0.32	pg/g	203.6	1.15		F	1
Canada	Melville Ice Cap (Melville Island)	75°24'N, 114°54'W	19 Mar. 1996	700	9	21	0.32	pg/g	255.0	2.89		F	1
Canada	Melville Ice Cap (Melville Island)	75°24'N, 114°54'W	19 Mar. 1996	700	8	18	0.32	pg/g	235.0	1.32		F	1
Canada	Melville Ice Cap (Melville Island)	75°24'N, 114°54'W	19 Mar. 1996	700	10	22	0.32	pg/g	245.4	2.19		F	1
Canada	Melville Ice Cap (Melville Island)	75°24'N, 114°54'W	19 Mar. 1996	700	18	30	0.32	pg/g	121.6	1.06		F	1
Canada	Melville Ice Cap (Melville Island)	75°24'N, 114°54'W	19 Mar. 1996	700	20	32	0.32	pg/g	98.4	0.57		F	1
Canada	Melville Ice Cap (Melville Island)	75°24'N, 114°54'W	19 Mar. 1996	700	28	40	0.32	pg/g	125.3	2.60		F	1
Canada	Melville Ice Cap (Melville Island)	75°24'N, 114°54'W	19 Mar. 1996	700	29	40	0.32	pg/g	99.5	0.36		F	1
Canada	Melville Ice Cap (Melville Island)	75°24'N, 114°54'W	19 Mar. 1996	700	28	40	0.32	pg/g	110.5	0.42		F	1
Canada	Total accumulation = 40 cm	75°24'N, 114°54'W	19 Mar. 1996	700*				pg/g	186.2	2.77		F	1
Russia	Pechora Sea	69°N,55°E	1998					µg/g		0.21	1.02	G	2
Russia	Pechora Sea	69°N,55°E	1998					µg/g		0.21	0.21	H	3

* weighted mean; ** mean of three pits. *** mean. **** weighted mean for pit #1. ***** weighted mean for pit #2. ****** mean of two pits.

Remarks

A. RUSSIAN ARCTIC 1993. These data are from composite samples representing one year's snow accumulation. Snow depth is the total depth of the snowpack down to the previous year's summer surfaceDensity is the bulk density of the snowpack for the previous year's accumulation.

B. ARCTIC OCEAN 1993. Samples were collected by scooping the top 10 cm of the snowpack. Snow depth is the total depth of the snowpack down to the underlying sea ice surface. Density is the bulk density of the snowpack for the previous year's accumulation.

C. CANADIAN ARCTIC 1993. These data are from composite samples representing one year's snow accumulation. Snow depth is the total depth of the snowpack down to the previous year's summer surface. Density is the bulk density of the snowpack for the previous year's accumulation.

D. RUSSIAN ARCTIC 1994. These data are from discrete incremental depth samples collected in shallow snow pits. The total depth (thickness) of the previous years' accumulation is given for each site. Density is the bulk density of the snowpack for the previous year's accumulation.

E. CANADIAN ARCTIC 1995. These data are from composite samples representing one year's snow accumulation (except for Penny Ice Cap). The total depth (thickness) of the previous years' accumulation is given for each site. For Penny ice cap, the density of each sample is given. For all other sites, density is the bulk density of the snowpack for the previous year's accumulation.

F. CANADIAN ARCTIC 1996. Data are from discrete incremental depth samples collected in shallow snow pits. The total depth (thickness) of the previous years' accumulation is given for each site. Density is the bulk density of the snowpack for the previous year's accumulation. Lead was not analyzed in these samples.

G. Snow cover.

H. Marine ice.

References

1. Koerner *et al.*, 2002;

2. Review, 1996; 3. Review, 2000.

Table A3. Metals in soil.

Type	Country	Location	Coordinate	Year	n	units	Lead		Cadmium		Mercury		Reference
							min-max	mean	min-max	mean	min-max	mean	
Fine sandy loam	Russia	Western shore of Yamal Peninsula	72°N,66°E	1998		µg/g dw	13.7-30		0.32-3.42		0.15-0.46		1
Loam	Russia	Western shore of Yamal Peninsula	72°N,66°E	1998		µg/g dw	12-19.3		0.1-0.37		0.08-0.10		1
Loamy soils	Russia	Eastern shore of Yugor Peninsula	71°N,65°E	1998		µg/g dw	8.3-17.5		0.43-0.9		0.10-0.15		1
Sand	Russia	Western shore of Yamal Peninsula	72°N,66°E	1998		µg/g dw	3.5-7.2		0.09-0.12		<0.05		1
Soil	Russia	Chukotka, Kanchalan	65.048-65.518N, 176.315-177.380E	2001	30	µg/g dw		0.751		<0.005		0.096	2
Soil	Russia	Chukotka, Lavrentiya	65.115-65.715N, 170.942-171.786W	2001	30	µg/g dw		7.274		0.242		0.046	2
Soil	Russia	Eastern shore of Yugor Peninsula		1998		µg/g dw	10.1-25.8		0.36-0.92		0.13-0.2		1
Soil	Russia	Pechora Basin	67.955-68.226N, 52.906-53.292E	2001	30	µg/g dw		2.436		0.154		0.066	2
Soil	Russia	Pechora Basin	67.955-68.226N, 52.906-53.292E	2001	30	µg/g dw		2.370		0.176		0.074	2
Soil	Russia	Pechora Basin	67.955-68.226N, 52.906-53.292	2001	30	µg/g dw		2.645		0.182		0.060	2
Soil	Russia	Taymir Peninsula, Dudinka	69.366-70.696N, 83.633-70.696W	2001	30	µg/g dw		3.433		<0.005		0.067	2
Soil	Russia	Taymir Peninsula, Dudinka	69.366-70.696N, 83.633-70.696W	2001	30	µg/g dw		3.152		<0.005		0.050	2
Soil-litter	Russia	Kola Peninsula		2001	7	µg/g dw		5.870		0.195		0.094	2
Soil-litter	Russia	Taymir Peninsula, Khatanga	71.957-74.519N, 100.527-105.788E	2001	27	µg/g dw		5.334		0.090		0.062	2
Soil-loamy sand	Russia	Kola Peninsula	67.798-68.765N 34.303-36.102E	2001	4	µg/g dw		2.130		<0.005		0.085	2
Soil-loamy sand	Russia	Taymir Peninsula, Khatanga	71.957-74.519N, 100.527-105.788E	2001	3	µg/g dw		8.297		0.010		0.081	2
Soil-peat	Russia	Kola Peninsula	67.798-68.765N 34.303-36.102E	2001	9	µg/g dw		6.925		0.141		0.123	2
Soil-peat litter	Russia	Kola Peninsula	67.798-68.765N 34.303-36.102E	2001	6	µg/g dw		7.192		0.163		0.105	2
Soil-sand	Russia	Kola Peninsula	67.798-68.765N 34.303-36.102E	2001	4	µg/g dw		2.855		<0.005		0.042	2

References

1. Review, 2000; 2. Melnikov *et al*., 2002.

Table A4. Metals in vegetation.

Species/type/location	Coordinate	Year	n	Unit	Lead min-max	Lead arith. mean	Lead SD	Cadmium min-max	Cadmium arith. mean	Cadmium SD	Mercury min-max	Mercury arith. mean	Mercury SD	Selenium min-max	Selenium arith. mean	Selenium SD	Reference
Cetraria cucullata (lichen)																	
Russia, Chukotka, Kanchalan		2001	10	µg/g dw		1.260			0.047			0.036					1
Russia, Chukotka, Lavrentiya		2001	20	µg/g dw		2.370			0.056			0.062					1
Russia, Kola Peninsula		1991	3	mg/kg dw	4.69-6.54			0.099-0.160									2
Russia, Taymir Peninsula		1992	13-16	mg/kg dw	0.78-5.78			0.066-0.382			0.005-0.057						2
Russia, Urals		1991	3	mg/kg dw	2.44-3.66			0.138-0.207									2
USA, Alaska		1990-93	39-43	mg/kg dw	0.18-4.33			0.015-0.770			0.015-0.085						2
Cetraria islandica (lichen)																	
Russia, Taymir Peninsula, Dudinka		2001	10	µg/g dw		1.550			0.086			0.019					1
Russia, Taymir Peninsula, Khatanga		2001	10	µg/g dw		3.960			0.086			0.062					1
Cetraria nivalis (lichen)																	
Greenland, Nuuk	64.16N, 51.75W	1994	5	mg/kg dw					0.510	0.025		0.0398	0.0097		0.068	0.023	3
Greenland, Nuuk	64.16N, 51.75W	1999	4	mg/kg dw					0.170	0.238		0.0356	0.0012		0.04	0.006	3
Cladina alpica (lichen)																	
Russia, Kola Peninsula		2001	3	µg/g dw		2.530			0.113			0.032					1
Cladina islandica (lichen)																	
Russia, Kola Peninsula		2001	6	µg/g dw		2.780			0.123			0.042					1
Cladina mitis (lichen)																	
Russia, Pechora Basin		2001	20	µg/g dw		1.780			0.051			0.039					1
Cladina rangiferina (lichen)																	
Russia, Chukotka, Kanchalan		2001	10	µg/g dw		0.580			0.038			0.021					1
Russia, Kola Peninsula		2001	3	µg/g dw		2.770			0.093			0.028					1
Cladina stellaris (lichen)																	
Russia, Kola Peninsula		2001	4	µg/g dw		3.130			0.138			0.056					1
Cladonia mitis (lichen)																	
Faroe Islands		1997		mg/kg dw		2.350			0.080			0.136			<0.5		4
Cladonia rangiferina (lichen)																	
Russia, Taymir Peninsula, Dudinka		2001	10	µg/g dw		1.840			0.106			0.023					1
Russia, Taymir Peninsula, Khatanga		2001	10	µg/g dw		4.210			0.074			0.059					1
Cladonia spp. (reindeer lichen)																	
Faroe Islands, Norðuri á Fossum		1997		mg/kg dw		2.4			0.08			0.14					5
Russia, Background regions (Kola Peninsula)		1996		mg/kg dw		1.4			0.05								6
Russia, East part of Lapland nature reserve, Kola Peninsula		1996		mg/kg dw		6.6			0.1								6
Russia, Near Imandra slope of Khibini, Kola Peninsula		1996		mg/kg dw		11			0.2								6
Russia, SE of Monchegorsk, Kola Peninsula		1996		mg/kg dw		8.8			0.7								6
Dicranum spp. (moss)																	
Russia, Chukotka, Kanchalan		2001	10	µg/g dw		3.650			0.096			0.073					1
Russia, Chukotka, Lavrentiya		2001	20	µg/g dw		2.850			0.688			0.231					1
Empetrum hermaphroditum (crowberry)																	
Greenland, Southwest		1999	5	mg/kg ww					0.004	0.006		<0.002			<0.05		3
Empetrum nigrum (crowberry)																	
Russia, Chukotka, Lavrentiya		2001	8	µg/g ww		0.050			0.008			<0.001					1
Hylocomium splendens (moss)																	
Faroe Islands		1996	8	mg/kg dw	5.37-10.31	7.580	1.98	0.090-0.150	0.120	0.020							7
Faroe Islands		2000	8	mg/kg dw	2.18-5.94	3.780	1.06	0.040-0.090	0.060	0.020	0.02-0.07	0.050	0.020				4
Norway, Amotsdalen		1990-93		µg/g dw		3.6											8
Norway, Dividalen		1990-93		µg/g dw		2.4											8
Norway, Lund		1990-92		µg/g dw		46											8
Norway, Solhomfjell		1990-92		µg/g dw		41											8
Russia, Pechora Basin		2001	20	µg/g dw		4.480			0.103			0.053					1
Russia, Taymir Peninsula, Dudinka		2001	14	µg/g dw		2.650			0.123			0.095					1
Russia, Taymir Peninsula, Khatanga		2001	10	µg/g dw		4.190			0.138			0.083					1
Leccinum aurantiacum (mushroom)																	
Russia, Kola Peninsula		2001	8	µg/g ww		0.041			0.082			0.018					1
Russia, Pechora Basin		2001	10	µg/g ww		0.022			0.064			0.009					1

Table A4. Metals in vegetation, continued.

Species/type/location	Coordinate	Year	n	Unit	Lead min-max	Lead arith. mean	Lead SD	Cadmium min-max	Cadmium arith. mean	Cadmium SD	Mercury min-max	Mercury arith. mean	Mercury SD	Selenium min-max	Selenium arith. mean	Selenium SD	Reference
Leccinum scabrum (mushroom)																	
Russia, Pechora Basin		2001	10	µg/g ww		0.058			0.101			0.010					1
Russia, Taymir Peninsula, Dudinka		2001	4	µg/g ww		0.072			0.072			0.041					1
Russia, Taymir Peninsula, Khatanga		2001	12	µg/g ww		0.097			0.060			0.023					1
Pleurozium schreberi (moss)																	
Russia, Kola Peninsula		2001	7	µg/g dw		3.990			0.215			0.053					1
Polytrichum commune (moss)																	
Russia, Kola Peninsula		2001	6	µg/g dw		3.730			0.232			0.046					1
Racomitrium lanuginosum (moss)																	
Faroe Islands		1997		mg/kg dw		13.200			0.070			0.164			1.01		4
Rubus chamaemorus (cloudberry)																	
Russia, Chukotka, Lavrentiya		2001	12	µg/g ww		0.008			0.008			<0.001					1
Sphagnum balticum (moss)																	
Russia, Chukotka, Kanchalan		2001	10	µg/g dw		3.740			0.088			0.069					1
Russia, Taymir Peninsula, Dudinka		2001	6	µg/g dw		2.580			0.096			0.083					1
Russia, Taymir Peninsula, Khatanga		2001	10	µg/g dw		3.960			0.147			0.079					1
Sphagnum magellanicum (moss)																	
Russia, Kola Peninsula		2001	3	µg/g dw		3.890			0.253			0.022					1
Vaccinium uliginosum (blueberry)																	
Russia, Chukotka, Kanchalan		2001	10	µg/g ww		0.037			<0.005			<0.001					1
Russia, Taymir Peninsula, Dudinka		2001	20	µg/g ww		0.012			<0.005			<0.001					1
Russia, Taymir Peninsula, Khatanga		2001	8	µg/g ww		0.025			0.009			<0.001					1
Vaccinium myrtillus (berry)																	
Russia, Kola Peninsula		2001	4	µg/g ww		0.017			<0.005			<0.001					1
Russia, Pechora Basin		2001	10	µg/g ww		0.014			<0.005			<0.001					1
Vaccinium uliginosum (Arctic blueberry)																	
Greenland, Southwest		1999	4	mg/kg ww					0.028	0.007		<0.001			<0.03		3
Vaccinium vitis-idaea (lingonberry/lowbush cranberry)																	
Russia, Chukotka, Kanchalan		2001	10	µg/g ww		0.021			<0.005			<0.001					1
Russia, Kola Peninsula		2001	16	µg/g ww		0.019			<0.005			<0.001					1
Russia, Pechora Basin		2001	10	µg/g ww		0.017			<0.005			<0.001					1
Russia, Taymir Peninsula, Khatanga		2001	10	µg/g ww		0.019			0.006			<0.001					1
Xerocomus sp. (mushroom)																	
Russia, Chukotka, Lavrentiya		2001	12	µg/g ww		0.072			0.032			0.0065					1
Grass																	
Russia, Pechora Sea shore		1999		µg/g dw	1.2-4.6			0.26-0.35				<0.02					6
Russia, Yamal Peninsula, Western shore		1999		µg/g dw					2.660			0.120					6
Musci																	
Russia, Yugor Peninsula, Eastern shore		1999		ng/g dw		2.480			0.600			0.060					6
Musci, lichen																	
Russia, Pechora Sea shore		1999		µg/g dw	3.1-6.5			0.24-0.41				<0.02					6
Shrubs																	
Russia, Yamal Peninsula, Western shore		1999		µg/g dw		7.090											6
Various vegetation																	
Russia, Yamal Peninsula, Western shore		1998		µg/g dw		4.2			2.15			0.1					6

References

1. Melnikov *et al*., 2002;
2. Ford *et al*., 1997;
3. F. Riget, unpubl. data, 2002;
4. M. Dam, unpubl. data, 2002;
5. Larsen and Dam, 1999;
6. Tsibulski *et al*., 2001;
7. Rühling *et al*., 1996; M. Dam, unpubl. data, 2002;
8. Kålås *et al*., 2000.

Table A5. Metals in terrestrial/aquatic birds.

Species/location	Coordinate	Year	Tissue	n	Sex	Age, yr	Unit	Lead			Cadmium			Mercury					Selenium			Remark	Reference
								min-max	mean	SD	min-max	mean	SD	min-max	mean	SD	geo. mean	GSD	min-max	mean	SD		
Accipiter gentilis (goshawk)																							
Norway	+62 N	1850-1939	Feathers	4		Juvenile	µg/g								0.20								1
Norway	+62 N	1850-1939	Feathers	3		Adult	µg/g								2.06								1
Norway	+62 N	1940-1967	Feathers	6		Juvenile	µg/g								26.25								1
Norway	+62 N	1940-1967	Feathers	2		Adult	µg/g								0.57								1
Norway	+62 N	1968-1995	Feathers	11		Juvenile	µg/g								2.20								1
Norway	+62 N	1968-1995	Feathers	23		Adult	µg/g								5.20								1
Anas acuta (northern pintail)																							
Canada, Northern Quebec		1981-1990	Liver	13			µg/g ww								0.48								2
Canada, Northern Quebec		1981-1990	Muscle	14			µg/g ww		0.12			<0.07			0.23					0.33			2
Russia, Chukotka, Kanchalan		2001	Liver	1	F	2-3	µg/g ww		0.163			0.392			0.0390								3
Russia, Chukotka, Kanchalan		2001	Muscle	1	F	2-3	µg/g ww		0.467*			0.008			0.0430							A	3
Russia, Pechora Basin		2001	Liver	4	F	1-3	µg/g ww		0.189			0.104			0.1760								3
Russia, Pechora Basin		2001	Liver	6	M	1-3	µg/g ww		0.165			0.130			0.2430								3
Russia, Pechora Basin		2001	Muscle	4	F	1-3	µg/g ww		0.167			0.008			0.0640								3
Russia, Pechora Basin		2001	Muscle	6	M	1-3	µg/g ww		0.129			0.005			0.0560								3
Russia, Taymir Peninsula, Dudinka		2001	Liver	4	F	1-3	µg/g ww		0.145			0.073			0.1280								3
Russia, Taymir Peninsula, Dudinka		2001	Liver	1	M	2-3	µg/g ww		0.136			0.059			0.0860								3
Russia, Taymir Peninsula, Dudinka		2001	Muscle	4	F	1-3	µg/g ww		0.035			0.034			0.0190								3
Russia, Taymir Peninsula, Dudinka		2001	Muscle	1	M	2-3	µg/g ww		0.027			0.031			0.0260								3
Russia, Taymir Peninsula, Khatanga		2001	Liver	1	M	1-2	µg/g ww		0.775			0.160			0.1540								3
Russia, Taymir Peninsula, Khatanga		2001	Muscle	1	M	1-2	µg/g ww		0.095			0.002			0.0710								3
Anas crecca (green-winged teal)																							
Canada, Northern Quebec		1981-1990	Liver	29			µg/g ww								0.37								2
Canada, Northern Quebec		1981-1990	Muscle	24			µg/g ww		0.03			<0.07			0.29					0.18			2
Russia, Taymir Peninsula, Dudinka		2001	Liver	2	M	1-2	µg/g ww		0.135			0.211			0.2110								3
Russia, Taymir Peninsula, Dudinka		2001	Muscle	2	M	1-2	µg/g ww		0.032			0.102			0.1020								3
Russia, Taymir Peninsula, Khatanga		2001	Liver	2	M	1-2	µg/g ww		0.105			0.066			0.1210								3
Russia, Taymir Peninsula, Khatanga		2001	Muscle	2	M	1-2	µg/g ww		0.047			0.005			0.0590								3
Anas penelope (widgeon)																							
Russia, Chukotka, Kanchalan		2001	Liver	1	F	1-2	µg/g ww		0.280			0.211			0.0480								3
Russia, Chukotka, Kanchalan		2001	Liver	1	M	1-2	µg/g ww		0.302			0.056			0.0430								3
Russia, Chukotka, Kanchalan		2001	Muscle	1	F	1-2	µg/g ww		1.204			0.015			0.0140								3
Russia, Chukotka, Kanchalan		2001	Muscle	1	M	1-2	µg/g ww		0.152			0.019			0.0120								3
Russia, Pechora Basin		2001	Liver	3	F	1-3	µg/g ww		0.117			0.454			0.0650								3
Russia, Pechora Basin		2001	Liver	2	M	1-3	µg/g ww		0.145			0.211			0.0820								3
Russia, Pechora Basin		2001	Muscle	3	F	1-3	µg/g ww		6.513*			0.015			0.1180							A	3
Russia, Pechora Basin		2001	Muscle	2	M	1-3	µg/g ww		0.112			0.036			0.0350								3
Anas platyrhynchos (mallard)																							
Canada, Northern Quebec		1981-1990	Liver	2			µg/g ww								0.53								2
Canada, Northern Quebec		1981-1990	Muscle	2			µg/g ww								0.18								2
Anas rubripes (black duck)																							
Canada, Northern Quebec		1981-1990	Liver	19			µg/g ww								0.71								2
Canada, Northern Quebec		1981-1990	Muscle	18			µg/g ww								0.25								2
Anser albifrons (white fronted goose)																							
Russia, Taymir Peninsula, Dudinka		2001	Liver	9	F	1-3	µg/g ww		0.156			0.044			0.0440								3
Russia, Taymir Peninsula, Dudinka		2001	Liver	4	M	1-3	µg/g ww		0.193			0.035			0.0350								3
Russia, Taymir Peninsula, Dudinka		2001	Muscle	9	F	1-3	µg/g ww		0.060			0.011			0.0110								3
Russia, Taymir Peninsula, Dudinka		2001	Muscle	4	M	1-3	µg/g ww		0.054			0.013			0.0130								3
Russia, Taymir Peninsula, Khatanga		2001	Liver	5	F	1-3	µg/g ww		0.580			0.234			0.0300								3
Russia, Taymir Peninsula, Khatanga		2001	Liver	4	M	1-4	µg/g ww		0.776			0.332			0.0380								3
Russia, Taymir Peninsula, Khatanga		2001	Muscle	5	F	1-3	µg/g ww		0.112			0.021			0.0060								3
Russia, Taymir Peninsula, Khatanga		2001	Muscle	4	M	1-4	µg/g ww		0.071			0.005			0.0060								3
Anser fabalis (bean goose)																							
Russia, Chukotka, Kanchalan		2001	Liver	2	F	1-2	µg/g ww		0.261			0.144			0.0970								3
Russia, Chukotka, Kanchalan		2001	Muscle	2	F	1-2	µg/g ww		0.526*			0.028			0.0100							A	3
Aquila chrysaetos (golden eagle)																							
Norway	+62 N	1850-1939	Feathers	3		Juvenile	µg/g								0.22								1
Norway	+62 N	1850-1939	Feathers	4		Adult	µg/g								0.55								1
Norway	+62 N	1940-1967	Feathers	7		Juvenile	µg/g								0.72								1
Norway	+62 N	1940-1967	Feathers	2		Adult	µg/g								12.67								1
Norway	+62 N	1968-1995	Feathers	25		Juvenile	µg/g								1.09								1

Table A5. Metals in terrestrial/aquatic birds, continued.

Species/location	Coordinate	Year	Tissue	n	Sex	Age, yr	Unit	Lead			Cadmium			Mercury					Selenium			Remark	Reference
								min-max	mean	SD	min-max	mean	SD	min-max	mean	SD	geo. mean	GSD	min-max	mean	SD		
Norway	+62 N	1968-1995	Feathers	46		Adult	µg/g								1.32								1
Aythya collaris (ring-necked duck)																							
Canada, Northern Quebec		1981-1990	Liver	10			µg/g ww								0.43								2
Canada, Northern Quebec		1981-1990	Muscle	10			µg/g ww		8.24			<0.06			0.28					0.31			2
Aythya marila (greater scaup)																							
Canada, Northern Quebec		1981-1990	Liver	26			µg/g ww								0.48								2
Canada, Northern Quebec		1981-1990	Muscle	26			µg/g ww								0.30								2
Russia, Kola Peninsula		2001	Liver	2	F	1-3	µg/g ww		0.326			0.133			0.1820								3
Russia, Kola Peninsula		2001	Liver	2	M	2-3	µg/g ww		0.265			0.163			0.1500								3
Russia, Kola Peninsula		2001	Muscle	2	F	1-3	µg/g ww		0.098			0.021			0.0460								3
Russia, Kola Peninsula		2001	Muscle	2	M	2-3	µg/g ww		0.122			0.019			0.0900								3
Russia, Taymir Peninsula, Khatanga		2001	Liver	1	F	1-2	µg/g ww		1.662			0.404			0.0680								3
Russia, Taymir Peninsula, Khatanga		2001	Muscle	1	F	1-2	µg/g ww		0.208			0.003			0.0340								3
Branta canadensis (Canada goose)																							
Canada, Northern Quebec		1981-1990	Liver	27			µg/g ww								0.09								2
Canada, Northern Quebec		1981-1990	Muscle	25			µg/g ww								0.06								2
Canada, Northern Quebec		1991-1997	Muscle	9			µg/g ww								<0.05								2
Bucephala clangula (common goldeneye)																							
Canada, Northern Quebec		1981-1990	Liver	3			µg/g ww								2.97								2
Canada, Northern Quebec		1981-1990	Muscle	3			µg/g ww								0.59								2
Russia, Kola Peninsula		2001	Liver	5	F	1-3	µg/g ww		0.670			0.130			0.2810								3
Russia, Kola Peninsula		2001	Liver	7	M	1-3	µg/g ww		0.349			0.164			0.3160								3
Russia, Kola Peninsula		2001	Muscle	5	F	1-3	µg/g ww		0.515			0.085			0.1130								3
Russia, Kola Peninsula		2001	Muscle	7	M	1-3	µg/g ww		1.380*			0.011			0.0970							A	3
Chen caerulescens (snow goose)																							
Canada, Northern Quebec		1991-1997	Muscle	8			µg/g ww		0.270			<0.07			<0.06					0.45			2
Clangula hyemalis (long tailed duck; oldsquaw)																							
Canada, Northern Quebec		1991-1997	Muscle	1			µg/g ww		<0.05			<0.10			0.24					0.82			2
Canada, Northern Quebec		1991-1997	Muscle	3			µg/g ww		0.05			<0.10			0.10					1.05			2
Falco columbarius (merlin)																							
Norway	+62 N	1850-1939	Feathers	2		Juvenile	µg/g								0.47								1
Norway	+62 N	1850-1939	Feathers	3		Adult	µg/g								1.75								1
Norway	+62 N	1940-1967	Feathers	2		Juvenile	µg/g								0.06								1
Norway	+62 N	1940-1967	Feathers			Adult	µg/g																1
Norway	+62 N	1968-1995	Feathers	6		Juvenile	µg/g								0.60								1
Norway	+62 N	1968-1995	Feathers	35		Adult	µg/g								2.30								1
Falco peregrinus (peregrine falcon)																							
Canada, Northern Quebec		1981-1990	Liver	1			µg/g ww								0.46								2
Canada, Northern Quebec		81-90	Liver	1			µg/g ww								0.46								2
Canada, Northern Quebec		81-90	Muscle	1			µg/g ww								0.20								2
Canada, Northern Quebec		81-90	Muscle	1			µg/g ww								0.2								2
Norway	+62 N	1850-1939	Feathers	3		Juvenile	µg/g								2.02								1
Norway	+62 N	1850-1939	Feathers	1		Adult	µg/g																1
Norway	+62 N	1940-1967	Feathers			Juvenile	µg/g																1
Norway	+62 N	1940-1967	Feathers	1		Adult	µg/g																1
Norway	+62 N	1968-1995	Feathers	40		Juvenile	µg/g								1.83								1
Norway	+62 N	1968-1995	Feathers	23		Adult	µg/g								7.33								1
Falco peregrinus anatum (American peregrine falcon)																							
USA, Alaska		1988-1990	Eggs	22			mg/kg dw										0.82-4.04		1.61				4
USA, Alaska		1991-1995	Eggs	33			mg/kg dw										0.48-9.58		1.96				4
Falco peregrinus tundrius (Arctic peregrine falcon)																							
USA, Alaska		1988-1990	Eggs	23			mg/kg dw										0.91-7.69		1.95				4
USA, Alaska		1991-1995	Eggs	20			mg/kg dw										1.20-3.12		1.88				4
Gavia immer (common loon)																							
Canada, Northern Quebec		1981-1990	Muscle	1			µg/g ww								0.93								2
Canada, Northern Quebec		1991-1997	Kidney	3			µg/g ww								1.549					2.77			2
Canada, Northern Quebec		1991-1997	Kidney	5			µg/g ww								1.954					2.39			2
Canada, Northern Quebec		1991-1997	Liver	3			µg/g ww								4.27								2
Canada, Northern Quebec		1991-1997	Liver	3			µg/g ww								1.90					3.42			2

Table A5. Metals in terrestrial/aquatic birds, continued

Species/location	Coordinate	Year	Tissue	n	Sex	Age, yr	Unit	Lead			Cadmium			Mercury					Selenium			Remark	Reference
								min-max	mean	SD	min-max	mean	SD	min-max	mean	SD	geo. mean	GSD	min-max	mean	SD		
Canada, Northern Quebec		1991-1997	Liver	5			µg/g ww								1.935					2.11			2
Canada, Northern Quebec		1991-1997	Muscle	3			µg/g ww								0.64					0.67			2
Canada, Northern Quebec		1991-1997	Muscle	10			µg/g ww					<0.05			0.782					0.67			2
Gavia stellata (red-throated loon)																							
Canada, Northern Quebec		1991-1997	Muscle	1			µg/g ww		<0.06			<0.1			0.297					0.85			2
Lagopus lagopus (willow ptarmigan)																							
Canada, Northern Quebec		1997	Kidney	31			µg/g ww					2.60											2
Canada, Northern Quebec		1997	Liver	31			µg/g ww					8.94											2
Canada, Northern Quebec		1997	Muscle	31			µg/g ww		0.06														2
Canada, Northern Quebec		1981-1990	Liver	2			µg/g ww		0.24			5.90			0.18					0.25			2
Canada, Northern Quebec		1981-1990	Muscle	7			µg/g ww		<0.10			0.21			<0.05					0.16			2
Canada, Northern Quebec		1991-1997	Muscle	1			µg/g ww		10.75			<0.10			<0.05					0.09			2
Canada, Northern Quebec		1991-1997	Muscle	1			µg/g ww		<0.02			<0.04			0.20					0.20			2
Norway		1990-1992	Liver	207	F/M	>14 months	µg/g dw		1.42	1.51													5
Russia, Chukotka, Kanchalan		2001	Liver	9	F	1-3	µg/g ww		0.450			0.325			0.0130								3
Russia, Chukotka, Kanchalan		2001	Liver	11	M	1-3	µg/g ww		0.570			0.340			0.0060								3
Russia, Chukotka, Kanchalan		2001	Muscle	9	F	1-3	µg/g ww		0.072			0.036			<0.001								3
Russia, Chukotka, Kanchalan		2001	Muscle	11	M	1-3	µg/g ww		0.120			0.061			<0.001								3
Russia, Chukotka, Lavrentiya		2001	Liver	14	F	1-3	µg/g ww		0.425			0.385			0.0120								3
Russia, Chukotka, Lavrentiya		2001	Liver	6	M	1-3	µg/g ww		0.395			0.430			0.0140								3
Russia, Chukotka, Lavrentiya		2001	Muscle	14	F	1-3	µg/g ww		0.082			0.038			<0.001								3
Russia, Chukotka, Lavrentiya		2001	Muscle	6	M	1-3	µg/g ww		0.180			0.066			<0.001								3
Russia, Kola Peninsula		2001	Liver	12	F	1-3	µg/g ww		0.570			0.470			0.0075								3
Russia, Kola Peninsula		2001	Liver	9	M	1-3	µg/g ww		0.645			0.520			0.0164								3
Russia, Kola Peninsula		2001	Muscle	12	F	1-3	µg/g ww		0.095			0.042			<0.001								3
Russia, Kola Peninsula		2001	Muscle	9	M	1-3	µg/g ww		0.215			0.074			<0.001								3
Russia, Taymir Peninsula, Khatanga		2001	Liver	2	F	1-2	µg/g ww		0.385			0.305			0.0120								3
Russia, Taymir Peninsula, Khatanga		2001	Liver	18	M	1-2	µg/g ww		0.310			0.440			<0.001								3
Russia, Taymir Peninsula, Khatanga		2001	Muscle	2	F	1-2	µg/g ww		0.068			0.045			<0.001								3
Russia, Taymir Peninsula, Khatanga		2001	Muscle	18	M	1-2	µg/g ww		0.176			0.053			<0.001								3
Lagopus mutus (rock ptarmigan)																							
Canada, Northern Quebec		1991-1997	Muscle	1			µg/g ww		6.13			0.06			<0.05					0.23			2
Canada, Northern Quebec		1991-1997	Muscle	2			µg/g ww		4.09			0.25			<0.05					0.16			2
Greenland, Nuuk		1999	Kidney	12			mg/kg ww					21.6	24.3		0.0373	0.0081				0.525	0.156		6
Greenland, Nuuk		1999	Liver	9			mg/kg ww					2.87	3.83		0.0219	0.0068				0.181	0.089		6
Greenland, Qeqertarsuaq		1999	Kidney	20			mg/kg ww					15.7	12.9		0.0456	0.0188				0.524	0.201		6
Greenland, Qeqertarsuaq		1999	Liver	19			mg/kg ww					1.91	1.26		0.0356	0.0182				0.174	0.052		6
Greenland, Nuuk	64.1N, 51.45 W	1999	Muscle	5	F		mg/kg ww					0.116	0.150		<0.002					0.169	0.068		6
Greenland, Nuuk	64.1N, 51.45 W	1999	Liver	5	F		mg/kg ww					5.20	4.41		0.0260	0.008				0.246	0.080		6
Russia, Pechora Basin		2001	Liver	8	F	0-3	µg/g ww		0.860			0.595			0.0140								3
Russia, Pechora Basin		2001	Liver	12	M	1-3	µg/g ww		1.130			0.540			0.0930								3
Russia, Pechora Basin		2001	Muscle	8	F	0-3	µg/g ww		0.142			0.072			<0.001								3
Russia, Pechora Basin		2001	Muscle	12	M	1-3	µg/g ww		0.286			0.063			<0.001								3
Russia, Taymir Peninsula, Dudinka		2001	Liver	8	F	1-2	µg/g ww		0.72			0.765			0.0250								3
Russia, Taymir Peninsula, Dudinka		2001	Liver	12	M	1-3	µg/g ww		0.950			0.870			0.0180								3
Russia, Taymir Peninsula, Dudinka		2001	Muscle	8	F	1-2	µg/g ww		0.162			0.610			<0.001								3
Russia, Taymir Peninsula, Dudinka		2001	Muscle	12	M	1-3	µg/g ww		0.365			0.135			<0.001								3
Lophodytes cucullatus (hooded merganser)																							
Canada, Northern Quebec		1981-1990	Liver	11			µg/g ww								3.697								2
Canada, Northern Quebec		1981-1990	Muscle	11			µg/g ww								0.689								2
Melanitta fusca (white-winged scoter)																							
Canada, Northern Quebec		1981-1990	Liver	3			µg/g ww								0.87								2
Canada, Northern Quebec		1981-1990	Muscle	3			µg/g ww								0.3								2
Melanitta nigra (black scoter)																							
Canada, Northern Quebec		1981-1990	Liver	23			µg/g ww								0.69								2
Canada, Northern Quebec		1981-1990	Muscle	24			µg/g ww								0.20								2
Canada, Northern Quebec		1991-1997	Liver	7			µg/g ww													4.35			2
Canada, Northern Quebec		1991-1997	Muscle	1			µg/g ww		<0.02			<0.05			0.19					0.66			2

Table A5. Metals in terrestrial/aquatic birds, continued.

Species/location	Coordinate	Year	Tissue	n	Sex	Age, yr	Unit	Lead			Cadmium			Mercury					Selenium			Remark	Reference
								min-max	mean	SD	min-max	mean	SD	min-max	mean	SD	mean	GSD	min-max	mean	SD		
Melanitta perspicillata (surf scoter)																							
Canada, Northern Quebec		1981-1990	Liver	29			µg/g ww								0.51								2
Canada, Northern Quebec		1981-1990	Muscle	28			µg/g ww								0.29								2
Canada, Northern Quebec		1991-1997	Liver	1			µg/g ww								12.90								2
Canada, Northern Quebec		1991-1997	Muscle	1			µg/g ww		12.56			0.07			0.11					1.52			2
Mergus merganser (common merganser)																							
Canada, Northern Quebec		81-90	Liver	10			µg/g ww								24.36								2
Canada, Northern Quebec		81-90	Muscle	18			µg/g ww								1.398					0.49			2
Mergus serrator (red-breasted merganser)																							
Canada, Northern Quebec		81-90	Liver	32			µg/g ww								2.693								2
Canada, Northern Quebec		81-90	Muscle	22			µg/g ww								0.518								2
Canada, Northern Quebec		91-97	Muscle	2			µg/g ww		<0.04			<0.02			0.653					0.65			2
Canada, Northern Quebec		91-97	Muscle	3			µg/g ww		<0.04			<0.02			0.678					0.64			2
Pandion haliaetus (osprey)																							
Canada, Northern Quebec		81-90	Kidney	3			µg/g ww					0.13			0.69					2.30			2
Canada, Northern Quebec		81-90	Liver	3			µg/g ww					0.07			0.52					3.39			2
Canada, Northern Quebec		81-90	Muscle	3			µg/g ww		<0.10			0.07			0.27					0.57			2
Somateria mollissima (common eider)																							
Canada, Northern Quebec		91-97	Muscle	1			µg/g ww		0.21			0.18			0.24					1.22			2
Canada, Northern Quebec		91-97	Muscle	2			µg/g ww		0.07			0.12			0.15					0.71			2
Somateria spectabilis (king eider)																							
Canada, Northern Quebec		91-97	Muscle	3			µg/g ww		0.16			0.10			0.22					0.71			2
Tetrao tetrix (black grouse)																							
Norway		1990-92	Liver	37	F/M	>14 months	µg/g dw		2.27	2.04													5
Norway		1990-92	Liver	32	F/M	3-6 months	µg/g dw		1.23	1.33													5

Remarks

A. This sample is contaminated by lead rolls.

References

1. Nygård, 1997;
2. Champoux *et al.*, 1999;
3. Melnikov *et al.*, 2002;
4. Ambrose *et al.*, 2000;
5. Kålås *et al.*, 2000;
6. F. Riget, unpubl. data, 2002.

Table A6. Metals in terrestrial mammals.

Species/location	Coordinate	Year	Tissue	n	Sex, U = unknown	Age (yrs)	Unit	Lead min-max	Lead arithm. mean	Lead SD	Lead geom. mean	Lead GSD	Cadmium min-max	Cadmium arithm. mean	Cadmium SD	Cadmium geom. mean	Cadmium GSD	Mercury min-max	Mercury arithm. mean	Mercury SD	Mercury geom. mean	Mercury GSD	Selenium min-max	Selenium arithm. mean	Selenium SD	Selenium geom. mean	Selenium GSD	Remark	Reference
Alces alces (moose)																													
Sweden, Norrbotten county (BD)	69.15N, 53.33W	1996-1999	Kidney	40			µg/g ww				0.010-0.039	0.02				0.426-5.49	1.53												1
Sweden, Norrbotten county (BD)		1996-1999	Liver	42			µg/g ww				0.009-0.056	0.022				0.105-0.824	0.294												1
Sweden, Norrbotten county (BD)		1996-1999	Liver	42			ng/g ww											3.91-5.26	4.10				101-458	215					1
Sweden, Norrbotten county (BD)		1996-1999	Muscle	37			ng/g ww											0.594-9.15	0.737										1
Sweden, Norrbotten county (BD)		1996-1999	Muscle	42			ng/g ww																34.7-69.8	49.2					1
Sweden, Norrbotten county (BD)		1996	Kidney	8-10		1.5-11.5	µg/g ww		0.030	0.018				1.850	1.690														13
Sweden, Norrbotten county (BD)		1996	Liver	8-10		1.5-11.5	µg/g ww		0.035	0.017				0.410	0.245									0.137	0.069				13
Sweden, Norrbotten county (BD)		1997	Kidney	8-10		1.5-11.5	µg/g ww		0.033	0.018				1.316	0.722														13
Sweden, Norrbotten county (BD)		1997	Liver	8-10		1.5-11.5	µg/g ww		0.022	0.005				0.260	0.138									0.251	0.195				13
Sweden, Norrbotten county (BD)		1998	Kidney	8-10		1.5-11.5	µg/g ww		0.015	0.006				1.971	0.995														13
Sweden, Norrbotten county (BD)		1998	Liver	8-10		1.5-11.5	µg/g ww		0.014	0.005				0.303	0.170									0.377	0.228				13
Sweden, Norrbotten county (BD)		2000	Kidney	8-10		1.5-11.5	µg/g ww		0.024	0.022				1.857	0.900														13
Sweden, Norrbotten county (BD)		2000	Liver	8-10		1.5-11.5	µg/g ww		0.023	0.028				0.302	0.193									0.113	0.038				13
Alopex lagopus (Arctic fox)																													
Canada, northern Quebec		1981-90	Kidney				µg/g ww							0.43															2
Canada, northern Quebec		1981-90	Liver	2(2)			µg/g ww		<0.10					0.25					0.25					0.70					2
Canis lupus (grey wolf)																													
Canada, northern Quebec		1981-90	Kidney	4(4)			µg/g ww							0.58															2
Canada, northern Quebec		1981-90	Liver	2(2)			µg/g ww							0.42					4.83										2
Lepus americanus (snowshoe hare)																													
Canada, northern Quebec		1981-90	Kidney	2(2)			µg/g ww							23.83															2
Canada, northern Quebec		1981-90	Liver	2(2)			µg/g ww		0.22					0.53					0.4					0.43					2
Canada, northern Quebec		1981-90	Muscle	5(5)			µg/g ww		<0.10					<0.20					0.12					0.15					2
Canada, northern Quebec		1991-97	kidney	2(2)			µg/g ww							6.19															2
Canada, northern Quebec		1991-97	Liver	3(3)			µg/g ww		2.07					0.37					0.05					0.33					2
Canada, northern Quebec		1991-97	Muscle	2(2)			µg/g ww		<0.10					<0.20					<0.05					0.30					2
Lepus arcticus (Arctic hare)																													
Greenland, Qeqertarsuaq	69.15N, 53.33W	1999	Kidney	5	M/F	Adult	µg/g ww							3.81	1.71				0.052	0.006				0.841	0.174				3
Greenland, Qeqertarsuaq	69.15N, 53.33W	1999	Liver	5	M/F	Adult	µg/g ww							0.186	0.021				0.029	0.010				0.102	0.046				3
Greenland, Qeqertarsuaq	69.15N, 53.33W	1999	Muscle	5	M/F	Adult	µg/g ww							0.005	0.003				<0.002					<0.05					3
Lepus timidus (mountain hare)																													
Faroe Islands		1997	Liver	1	M	Adult	µg/g dw							1.818					0.259										4
Faroe Islands		1997	Liver	1	M	Juvenile	µg/g dw							0.579					0.386										4
Faroe Islands		1997	Liver	1	F	Adult	µg/g dw							0.856					0.272										4
Faroe Islands		1999	Liver	6	M	Adult	µg/g dw							0.362					0.226					2.230					4
Faroe Islands		1999	Liver	5	F	Adult	µg/g dw							0.582					0.150					2.470					4
Faroe Islands		1999	Liver	15		Juvenile	µg/g dw							0.786					0.240					2.730					4
Norway		1990-92	Liver	49	M/F	>14 mths	µg/g dw		1.15	1.5																			5
Norway		1990-92	Liver	71	M/F	2-8 mths	µg/g dw		0.55	0.74																			5
Russia, Chukotka, Kanchalan		2001	Kidney	4	M	1-2	µg/g ww		0.031					0.980					0.048										6
Russia, Chukotka, Kanchalan		2001	Liver	4	M	1-2	µg/g ww		0.078					0.160					0.024										6
Russia, Chukotka, Kanchalan		2001	Muscle	4	M	1-2	µg/g ww		<0.005					0.007					<0.001										6
Russia, Chukotka, Kanchalan		2001	Kidney	6	F	1-3	µg/g ww		0.058					1.100					0.053										6
Russia, Chukotka, Kanchalan		2001	Liver	6	F	1-3	µg/g ww		0.096					0.250					0.015										6
Russia, Chukotka, Kanchalan		2001	Muscle	6	F	1-3	µg/g ww		0.059					0.009					<0.5										6
Russia, Kola Peninsula		2001	Kidney	4	M	0-2	µg/g ww		0.039					1.350					0.0600										6
Russia, Kola Peninsula		2001	Liver	4	M	0-2	µg/g ww		0.102					0.210					0.0130										6
Russia, Kola Peninsula		2001	Muscle	4	M	0-2	µg/g ww		<0.005					0.0084					<0.001										6
Russia, Kola Peninsula		2001	Kidney	6	F	1-2	µg/g ww		0.057					2.550					0.0720										6
Russia, Kola Peninsula		2001	Liver	6	F	1-2	µg/g ww		0.135					0.360					0.0017										6
Russia, Kola Peninsula		2001	Muscle	6	F	1-2	µg/g ww		0.0079					0.0110					<0.001										6
Russia, Pechora Basin		2001	Kidney	7	M	1-3	µg/g ww		0.056					1.750					0.0210										6
Russia, Pechora Basin		2001	Liver	7	M	1-3	µg/g ww		0.176					0.198					0.0920										6
Russia, Pechora Basin		2001	Muscle	7	M	1-3	µg/g ww		0.011					0.013					<0.001										6
Russia, Pechora Basin		2001	Kidney	8	F	0-3	µg/g ww		0.088					3.340					0.0180										6
Russia, Pechora Basin		2001	Liver	8	F	0-3	µg/g ww		0.212					0.520					0.1260										6
Russia, Pechora Basin		2001	Muscle	8	F	0-3	µg/g ww		0.012					0.018					<0.001										6
Russia, Taymir Peninsula, Dudinka		2001	Kidney	5	M	0-3	µg/g ww		0.062					1.630					0.112										6
Russia, Taymir Peninsula, Dudinka		2001	Liver	5	M	0-3	µg/g ww		0.204					0.440					0.022										6
Russia, Taymir Peninsula, Dudinka		2001	Muscle	5	M	0-3	µg/g ww		0.009					0.012					<0.001										6
Russia, Taymir Peninsula, Dudinka		2001	Kidney	10	F	0-3	µg/g ww		0.093					2.840					0.083										6
Russia, Taymir Peninsula, Dudinka		2001	Liver	10	F	0-3	µg/g ww		0.146					0.350					0.019										6
Russia, Taymir Peninsula, Dudinka		2001	Muscle	10	F	0-3	µg/g ww		0.014					0.016					<0.001										6

Table A6. Metals in terrestrial mammals, continued

Species/location	Coordinate	Year	Tissue	n	Sex, U = unknown	Age (yrs)	Unit	Lead min-max	Lead arithm. mean	Lead SD	Lead geom. mean	Lead GSD	Cadmium min-max	Cadmium arithm. mean	Cadmium SD	Cadmium geom. mean	Cadmium GSD	Mercury min-max	Mercury arithm. mean	Mercury SD	Mercury geom. mean	Mercury GSD	Selenium min-max	Selenium arithm. mean	Selenium SD	Selenium geom. mean	Selenium GSD	Remark	Reference
Russia, Taymir Peninsula, Khatanga		2001	Kidney	8	M	1-3	µg/g ww		0.043					0.830					0.047										6
Russia, Taymir Peninsula, Khatanga		2001	Liver	8	M	1-3	µg/g ww		0.074					0.150					0.007										6
Russia, Taymir Peninsula, Khatanga		2001	Muscle	8	M	1-3	µg/g ww		0.042					0.006					<0.001										6
Russia, Taymir Peninsula, Khatanga		2001	Kidney	6	F	1-3	µg/g ww		0.032					1.230					0.059										6
Russia, Taymir Peninsula, Khatanga		2001	Liver	6	F	1-3	µg/g ww		0.127					0.230					0.012										6
Russia, Taymir Peninsula, Khatanga		2001	Muscle	6	F	1-3	µg/g ww		<0.005					0.008					<0.001										6
Martes americana (American marten)																													
Canada, northern Quebec		1981-90	Liver	2(4)			µg/g ww		<0.10					0.47					0.41					0.85					2
Canada, northern Quebec		1981-90	Liver	5(5)			µg/g ww												0.45										2
Canada, northern Quebec		1981-90	Muscle	5(6)			µg/g ww		0.08					0.26					0.25					0.242					2
Canada, northern Quebec		1991-97	Kidney	5(5)			µg/g ww							1.41															2
Canada, northern Quebec		1991-97	Kidney	5(5)			µg/g ww							2.45															2
Canada, northern Quebec		1991-97	Liver	6(6)			µg/g ww		<0.10					0.90					0.52					1.00					2
Canada, northern Quebec		b. 1981	Kidney	1			µg/g ww												5.17										2
Canada, northern Quebec		b. 1981	Liver	1			µg/g ww												1.48										2
Canada, northern Quebec		b. 1981	Muscle	1			µg/g ww												1.51										2
Mustela erminea (shorttail weasel)																													
Canada, northern Quebec		1981-90	Liver	1(2)			µg/g ww												0.19					0.75					2
Canada, northern Quebec		1981-90	Muscle	1(2)			µg/g ww		<0.10					<0.30					0.11					0.30					2
Mustela vison (mink)																													
Canada, northern Quebec		1981-90	Liver	1			µg/g ww		<0.10					0.80					3.47					1.60					2
Canada, northern Quebec		1981-90	Muscle	1			µg/g ww		<0.10					<0.20					1.94					0.35					2
Canada, northern Quebec		b. 1981	Kidney	3(3)			µg/g ww												2.54										2
Canada, northern Quebec		b. 1981	Liver	1			µg/g ww												7.60										2
Canada, northern Quebec		b. 1981	Muscle	2(2)			µg/g ww												2.13										2
Ovibos moschatus (muskox)																													
Greenland, Kangerlussuaq, W Greenland	67.01N, 50.40W	1999	Fat	4			µg/g ww							<0.005					<0.005					<0.2					3
Greenland, Kangerlussuaq, W Greenland	67.01N, 50.40W	1999	Kidney	4			µg/g ww							0.284	0.202				0.072	0.025				0.757	0.246				3
Greenland, Kangerlussuaq, W Greenland	67.01N, 50.40W	1999	Liver	4			µg/g ww							0.051	0.007				0.023	0.004				0.062	0.044				3
Greenland, Kangerlussuaq, W Greenland	67.01N, 50.40W	1999	Muscle	9			µg/g ww							0.002	0.004				0.002	0.001				<0.05					3
Ovis spp. (domestic sheep)																													
Greenland, South		1999	Fat	5		lamb	µg/g ww							<0.005					<0.005					<0.2					3
Greenland, South		1999	Kidney	5		lamb	µg/g ww							0.200	0.081				0.012	0.01				0.707	0.165				3
Greenland, South		1999	Liver	5		lamb	µg/g ww							0.108	0.059				0.005	0.001				0.098	0.054				3
Greenland, South		1999	Muscle	10		lamb	µg/g ww							<0.002					<0.002					<0.05					3
Rangifer tarandus (reindeer/caribou)																													
Canada, northern Quebec		1981-90	Muscle	37(37)			µg/g ww												0.02										2
Canada, northern Quebec		1994-1996	Liver	260	M/F		µg/g ww	0.792					1.020					0.594										A	7
Canada, NWT and Nunavut, Bathurst Herd		1992	Kidney	19-20	10M/10F	6.8(2.6)	µg/g ww	0.015	(0.01-0.05)				1.609	(0.96-2.68)				0.513	(0.33-0.8)									B	8
Canada, NWT and Nunavut, Bathurst Herd		1992	Liver	20	10M/10F	6.8(2.6)	µg/g ww	0.084	(0.04-0.19)				0.484	(0.21-1.14)				0.131	(0.07-0.25)										8
Canada, NWT and Nunavut, Beverly Herd		1994	Kidney	10-11	6M/5F	6.5(2.4)	µg/g ww	0.023	(0.01-0.04)				6.153	(4.05-9.35)				1.978	(1.37-2.86)									C	8
Canada, NWT and Nunavut, Beverly Herd		1994	Liver	10	6M/5F	6.5(2.4)	µg/g ww	0.033	(0.03-0.05)				0.996	(0.77-1.29)				0.351	(0.26-0.49)										8
Canada, NWT and Nunavut, Beverly Herd		2000	Kidney	20	20M	-	µg/g ww	0.116	(0.07-0.19)				9.576	(5.23-17.6)				1.292	(1.18-1.71)										8
Canada, NWT and Nunavut, Beverly Herd		2000	Liver	20	20M	-	µg/g ww	0.313	(0.17-0.57)				1.508	(0.97-2.34)				0.232	(0.13-0.4)										8
Canada, NWT and Nunavut, Bluenose Herd		1994	Kidney	9	2M/8F	-	µg/g ww	0.013	(0.01-0.02)				6.644	(2.4-18.4)				1.819	(1.52-2.35)										8
Canada, NWT and Nunavut, Bluenose Herd		1994	Liver	10	2M/8F	-	µg/g ww	0.027	(0.01-0.06)				1.431	(0.67-3.06)				0.444	(0.29-0.69)										8
Canada, NWT and Nunavut, Bluenose Herd		1998	Kidney	11	10 M/2F	-	µg/g ww	0.053	(0.03-0.09)				3.213	(1.77-5.84)				0.403	(0.22-0.75)										8
Canada, NWT and Nunavut, Bluenose Herd		1998	Liver	12	10M/2F	-	µg/g ww	0.117	(0.08-0.17)				0.936	(0.6-1.46)				0.138	(0.09-0.21)										8
Canada, NWT and Nunavut, Cambridge Bay Herd		1993	Kidney	8-10	6M/4F	5.5(3.7)	µg/g ww	0.078	(0.04-0.16)				1.695	(0.94-3.07)				0.798	(0.56-1.13)									D	8
Canada, NWT and Nunavut, Cambridge Bay Herd		1993	Liver	8-10	6M/4F	5.5(3.7)	µg/g ww	0.154	(0.09-0.26)				0.331	(0.19-0.58)				0.212	(0.17-0.27)									D	8
Canada, NWT and Nunavut, Cape Dorset Herd		1992	Kidney	10	5M/5F	4.7(3.0)	µg/g ww	0.067	(0.03-0.13)				2.280	(1.2-4.35)				1.252	(1.01-1.55)										8
Canada, NWT and Nunavut, Cape Dorset Herd		1992	Liver	10	5M/5F	4.7(3.0)	µg/g ww	0.625	(0.44-0.89)				0.455	(0.23-0.9)				0.280	(0.14-0.58)										8
Canada, NWT and Nunavut, Lake Harbour Herd		1992	Kidney	10	5 M/5F	4.6(1.8)	µg/g ww	0.080	(0.04-0.15)				5.220	(2.48-11)				2.560	(1.74-3.74)										8
Canada, NWT and Nunavut, Lake Harbour Herd		1992	Liver	10	5M/5F	4.6(1.8)	µg/g ww	0.761	(0.42-1.4)				1.034	(0.58-1.84)				0.551	(0.3-1.01)										8
Canada, NWT and Nunavut, Lake Harbour Herd		1999	Kidney	17-19	15M/2F/2U	5.4(2.8)	µg/g ww	0.405	(0.32-0.51)				4.950	(2.57-9.55)				0.689	(0.5-0.95)									E	8
Canada, NWT and Nunavut, Lake Harbour Herd		1999	Liver	17-19	15M/2F/2U	5.4(2.8)	µg/g ww	2.406	(1.65-3.5)				1.076	(0.69-1.67)				0.233	(0.17-0.33)									E	8
Canada, NWT and Nunavut, Pond Inlet Herd		1993	Kidney	10	6M/4F	7.1(1.8)	µg/g ww	0.044	(0.01-0.25)				2.625	(1.6-4.28)				1.793	(1.0-3.21)										8
Canada, NWT and Nunavut, Pond Inlet Herd		1993	Liver	10	6M/4F	7.1(1.8)	µg/g ww	0.133	(0.07-0.25)				0.255	(0.15-0.44)				0.371	(0.26-0.53)										8
Canada, NWT and Nunavut, Qaminuriaq Herd		1992	Kidney	10	10M	5.6(2.8)	µg/g ww	0.010	(0.002-0.05)				5.700	(2.96-11)				2.820	(2.26-3.52)										8
Canada, NWT and Nunavut, Qaminuriaq Herd		1992	Liver	10	10M		µg/g ww	0.035	(0.01-0.17)				0.925	(0.51-1.68)				0.861	(0.59-1.26)										8
Canada, NWT and Nunavut, Southampton Island		1991	Kidney	10	8M/2F	2.8(1.5)	µg/g ww	0.040	(0.01-0.18)				3.400	(2.2-5.26)				0.436	(0.37-0.52)										8
Canada, NWT and Nunavut, Taloyoak Herd		1993	Kidney	10	5M/2F/3U		µg/g ww	0.065	(0.05-0.09)				1.497	(1.13-1.98)				0.200	(0.13-0.32)										8

Table A6. Metals in terrestrial mammals, continued

Species/location	Coordinate	Year	Tissue	n	Sex, U = unknown	Age (yrs)	Lead Unit	Lead min-max	Lead arithm. mean	Lead SD	Lead geom. mean	Lead GSD	Cadmium min-max	Cadmium arithm. mean	Cadmium SD	Cadmium geom. mean	Cadmium GSD	Mercury min-max	Mercury arithm. mean	Mercury SD	Mercury geom. mean	Mercury GSD	Selenium min-max	Selenium arithm. mean	Selenium SD	Selenium geom. mean	Selenium GSD	Remark	Reference
Canada, NWT and Nunavut, Taloyoak Herd		1993	Liver	10	5M/2F/3U		µg/g ww				0.133	(0.07-0.25)				0.299	(0.24-0.38)				0.218	(0.15-0.32)							8
Canada, NWT, Beverly Herd (Nonacho L.)	61°59'N, 109°28'W	2000	Liver	19-20	M		µg/g ww	0.41-4.45			1.100	(0.60-1.95)	1.19-9.7			5.200	(3.4-8.1)	<0.5-5.0			0.700	(0.28-1.78)	0.80-2.40			1.00	(0.76-1.45)	F	9
Canada, South Baffin Isl., Lake Harbour Herd		1999	Liver	19	14M/5F	1-10 yrs	µg/g ww				2.27	(1.56-3.32)				1.020	(0.65-1.58)				0.220	(0.16-0.3)				0.530	(0.45-0.62)		10
Canada, South Baffin Isl., Lake Harbour Herd		1999	Kidney	19	14M/5F	1-10 yrs	µg/g ww				0.38	(0.31-0.48)				4.710	(2.45-9.07)				0.020	(0.01-0.03)				0.910	(0.82-1.02)		10
Canada, Yukon, Bonnet Plume Herd		1993	Kidney	19-21	21M	6.1(2.9)	µg/g ww				0.072	(0.031-0.0.17)				7.30	(3.89-13.7)				0.4032	(0.31-0.53)						G	8
Canada, Yukon, Bonnet Plume Herd		1993	Liver	21	21M		µg/g ww				0.177	0.13				0.97	(0.6-1.56)												8
Canada, Yukon, Finlayson Herd		1992	Kidney	32	30M/2F	5.3(2.8)	µg/g ww				0.100	(0.062-0.17)				26.40	(12.2-57.2)				0.74	(0.54-1.01)							8
Canada, Yukon, Finlayson Herd		1992	Liver	30-31	30M/1F	5.2(2.8)	µg/g ww				0.186	(0.12-0.28)				3.19	(1.89-5.39)				0.1624	(0.11-0.24)						H	8
Canada, Yukon, Finlayson Herd		1993	Kidney	22-23	20M/2F/1U	4.3(1.8)	µg/g ww				0.147	(0.083-0.26)				23.23	(14.7-36.8)				0.7153	(0.54-0.95)						I	8
Canada, Yukon, Finlayson Herd		1993	Liver	22-23	20M/2F/1U	4.3(1.8)	µg/g ww				0.105	(0.054-0.21)				2.13	(1.09-4.18)												I	8
Canada, Yukon, Finlayson Herd		1993	Muscle	1-7	20M/2F	2.0	µg/g ww				0.023	(0.01 -0.06)				0.0286	(0.02-0.04)												J	8
Canada, Yukon, Finlayson Herd		1993	Muscle	4		-	µg/g ww				< 0.01					< 0.04													8
Canada, Yukon, Porcupine Herd		1991	Kidney	10-20	20M	5.9(2.0)	µg/g ww				< 1.20					7.39	(3.51-15.6)				0.6468	(0.48-0.88)						K	8
Canada, Yukon, Porcupine Herd		1991	Liver	10-20	20M		µg/g ww				< 1.20					1.31	(0.7-2.43)												K	8
Canada, Yukon, Porcupine Herd		1992	Kidney	14-26	26M	3.6(2.9)	µg/g ww				< 1.20					5.81	(2.97-11.4)				0.3872	(0.26-0.59)						L	8
Canada, Yukon, Porcupine Herd		1992	Liver	14-26	26M		µg/g ww				< 0.40					1.03	(0.6-1.76)												L	8
Canada, Yukon, Porcupine Herd		1993	Kidney	4	2M/2F		µg/g ww				< 1.00					6.13	(4.64-8.09)				0.2604	(0.2-0.34)							8
Canada, Yukon, Porcupine Herd		1993	Liver	4	2M/2F		µg/g ww				< 0.80					1.25	(0.85-1.83)													8
Canada, Yukon, Porcupine Herd		1993	Muscle	2	2M/2F		µg/g ww				< 0.96					<0.06													8
Canada, Yukon, Porcupine Herd		1994	Kidney	13-28	4M/5F/19U	3.8(1.9)	µg/g ww				0.0462	(0.022-0.1)				7.92	(4.09-15.4)				0.3982	(0.14-1.15)							8
Canada, Yukon, Porcupine Herd		1994	Liver	12-13	4M/5F/19U	4.1(1.7)	µg/g ww				0.0551	(0.038-0.06)				1.40	(0.76-2.59)				0.0841	(0.05-0.15)							8
Canada, Yukon, Porcupine Herd		1994	Muscle	6	4M/5F	4.2(1.0)	µg/g ww				0.0150	(0.003-0.07)				<0.01					0.0075	(0.01-0.02)							8
Canada, Yukon, Porcupine Herd		1995	Kidney	40	16M/24U		µg/g ww				0.0374	(0.024-0.06)				8.07	(4.27-15.3)				0.176	(0.04-0.89)							8
Canada, Yukon, Porcupine Herd		1995	Liver	18	16M/24 U		µg/g ww				0.0525	(0.039-0.07)				1.19	(0.88-1.62)				0.0315	(0.01-0.1)							8
Canada, Yukon, Porcupine Herd		1996	Kidney	42	23M/2F/17U		µg/g ww				0.0396	(0.029-0.05)				10.34	(5.41-19.8)				0.4664	(0.35-0.62)							8
Canada, Yukon, Porcupine Herd		1997	Kidney	11-53	41M/12F	4.0(1.9)	µg/g ww				0.0396	(0.022-0.07)				7.11	(3.37-15)				0.4466	(0.33-0.61)						M	8
Canada, Yukon, Porcupine Herd		1998	Kidney	17-56	39M/13F/4U	3.2(2.9)	µg/g ww				0.0462	(0.025-0.08)				3.68	(0.33-41.4)				0.2919	(0.09-0.96)						N	8
Canada, Yukon, Tay Herd		1993	Kidney	20	20M	4.8(2.6)	µg/g ww				0.3335	(0.23-0.47)				23.92	(14.3-40)				0.7337	(0.56-0.96)							8
Canada, Yukon, Tay Herd		1993	Liver	20	20M	4.8(2.6)	µg/g ww				0.5066	(0.35-0.74)				2.85	(1.59-5.13)													8
Canada, Yukon, Tay Herd		1998	Kidney	4	1M/3F	3.2(2.2)	µg/g ww				0.0378	(0.021-0.07)				9.94					0.1827	(0.08-0.44)							8
Greenland, Akia		1996	Liver	24	M/F		µg/g ww							0.631	0.285				0.128	0.056				0.414	0.101				3
Greenland, Akia		1996	Muscle	22	M/F		µg/g ww							0.005	0.004				0.011	0.003				0.128	0.018				3
Greenland, Akia		1997	Liver	25	M/F		µg/g ww							0.829	0.536				0.282	0.103				0.305	0.099				3
Greenland, Akia		1997	Muscle	25	M/F		µg/g ww							0.003	0.002				0.03	0.007				0.175	0.03				3
Greenland, Akia		1997	Liver	29		5	µg/g ww		1.008	0.575				0.782	0.519				0.280	0.097									3
Greenland, Isortoq	60°59'.06N, 47°30'.63W	1995	Liver	7			µg/g ww							0.443	0.136				0.651	0.206				1.005	0.226				11
Greenland, Isortoq	60°59.06'N, 47°30'.63W	1999	Kidney	20	M/F		µg/g ww							1.73	1.08				0.115	0.03				0.82	0.094				3
Greenland, Isortoq	60°59.06'N, 47°30'.63W	1999	Liver	20	M/F		µg/g ww		0.588	0.196				0.496	0.398				0.142	0.112				0.261	0.081				3
Greenland, Isortoq	60°59.06'N, 47°30'.63W	2000	Fat	5	M		µg/g ww							0.003	0.002				<0.003					<0.2					3
Greenland, Isortoq	60°59.06'N, 47°30'.63W	2000	Kidney	5	M		µg/g ww							0.276	0.117				0.038	0.008				0.563	0.065				3
Greenland, Itinnera	64.38 N, 50.38 W	1996	Liver	7	M/F		µg/g ww							0.350	0.251				0.071	0.023				0.171	0.048				3
Greenland, Itinnera	64.38 N, 50.38 W	1996	Muscle	7	M/F		µg/g ww							0.001	0.001				0.003	0.001				0.078	0.012				3
Greenland, Itinnera	64.38 N, 50.38 W	1997	Liver	10	M/F		µg/g ww							0.280	0.195				0.063	0.033				0.171	0.042				3
Greenland, Itinnera	64.38 N, 50.38 W	1997	Muscle	10	M/F		µg/g ww							0.002	0.002				0.006	0.002				0.100	0.017				3
Greenland, Itinnera	64.38 N, 50.38 W	1999	Kidney	18	M/F		µg/g ww							1.424	0.822				0.093	0.046				0.639	0.16				3
Greenland, Itinnera	64.38 N, 50.38 W	1999	Liver	18	M/F		µg/g ww							0.265	0.123				0.032	0.022				0.163	0.108				3
Greenland, Kangerlussuaq		1996	Liver	23	M/F		µg/g ww							0.154	0.101				0.041	0.012				0.086	0.017				3
Greenland, Kangerlussuaq		1996	Muscle	22	M/F		µg/g ww							0.002	0.005				0.005	0.001				0.031	0.008				3
Greenland, Kangerlussuaq		1997	Liver	24	M/F		µg/g ww							0.211	0.106				0.064	0.019				0.095	0.019				3
Greenland, Kangerlussuaq		1997	Muscle	24	M/F		µg/g ww							0.004	0.004				0.010	0.002				0.046	0.009				3
Greenland, Kangerlussuaq	97.01N, 50.40W	1997	Liver	27		7	µg/g ww		0.068	0.059				0.210	0.098				0.064	0.018									3
Greenland, South Greenland		2000	Fat	5	M		µg/g ww							0.003	0.002				<0.003					<0.2					3
Greenland, South Greenland		2000	Kidney	5	M		µg/g ww							0.276	0.117				0.0380	0.008				0.563	0.065				3
Norway, Hardangervidda		1987-1994	Liver	4			µg/g ww		0.550	0.085				1.120	0.610				0.130	0.040									12
Norway, Rondane		1986-1993	Liver	60			µg/g ww		1.150	0.820				1.100	0.790				0.160	0.110									12
Russia, Chukotka		1991	Liver	6			µg/g ww		0.480	0.170				1.730	0.520				0.170	0.035									12
Russia, Chukotka		2000	Liver			2-5	µg/g ww		0.166	0.022				0.343	0.053				0.035	0.010									6
Russia, Chukotka, Kanchalan		2001	Kidney	4	M	3-4	µg/g ww		0.122					0.860					0.145										6
Russia, Chukotka, Kanchalan		2001	Liver	4	M	3-4	µg/g ww		0.160					0.280					0.032										6
Russia, Chukotka, Kanchalan		2001	Muscle	4	M	3-4	µg/g ww		0.015					0.031					0.001										6
Russia, Chukotka, Kanchalan		2001	Kidney	6	F	3-5	µg/g ww		0.097					0.720					0.120										6
Russia, Chukotka, Kanchalan		2001	Liver	6	F	3-5	µg/g ww		0.138					0.335					0.026										6
Russia, Chukotka, Kanchalan		2001	Muscle	6	F	3-5	µg/g ww		0.012					0.021					0.001										6
Russia, Chukotka, Lavrentiya		2001	Kidney	4	M	2-4	µg/g ww		0.124					0.890					0.2400										6

Table A6. Metals in terrestrial mammals, continued

Species/location	Coordinate	Year	Tissue	n	Sex, U = unknown	Age (yrs)	Lead						Cadmium						Mercury						Selenium						Remark	Reference
							Unit	min-max	arithm. mean	SD	geom. mean	GSD	min-max	arithm. mean	SD	geom. mean	GSD	min-max	arithm. mean	SD	geom. mean	GSD	min-max	arithm. mean	SD	geom. mean	GSD		
Russia, Chukotka, Lavrentiya		2001	Liver	4	M	2-4	µg/g ww		0.190					0.350					0.0490											6
Russia, Chukotka, Lavrentiya		2001	Muscle	4	M	2-4	µg/g ww		0.012					0.028					0.0010											6
Russia, Chukotka, Lavrentiya		2001	Kidney	6	F	2-5	µg/g ww		0.135					0.760					0.1750											6
Russia, Chukotka, Lavrentiya		2001	Liver	6	F	2-5	µg/g ww		0.177					0.410					0.0320											6
Russia, Chukotka, Lavrentiya		2001	Muscle	6	F	2-5	µg/g ww		0.011					0.027					0.0010											6
Russia, Karelia		1992	Liver	3			µg/g ww		0.900	0.640				0.220	0.140				0.190	0.040										12
Russia, Kola Peninsula		1992	Liver	7			µg/g ww		0.900	0.420				0.720	0.410				0.060	0.020										12
Russia, Kola Peninsula		2000	Liver	6		2-5	µg/g ww		0.219	0.070				0.410	0.121				0.050	0.016										6
Russia, Kola Peninsula		2001	Kidney	3	M	2-3	µg/g ww		0.120					0.475					0.140											6
Russia, Kola Peninsula		2001	Kidney	5	M	3-4	µg/g ww		0.148					1.050					0.230											6
Russia, Kola Peninsula		2001	Kidney	2	M	4-5	µg/g ww		0.230					2.400					0.260											6
Russia, Kola Peninsula		2001	Liver	3	M	2-3	µg/g ww		0.170					0.360					0.027											6
Russia, Kola Peninsula		2001	Liver	5	M	3-4	µg/g ww		0.265					0.520					0.054											6
Russia, Kola Peninsula		2001	Liver	2	M	4-5	µg/g ww		0.235					0.495					0.061											6
Russia, Kola Peninsula		2001	Muscle	3	M	2-3	µg/g ww		0.017					0.014					0.0015											6
Russia, Kola Peninsula		2001	Muscle	5	M	3-4	µg/g ww		0.029					0.028					0.0074											6
Russia, Kola Peninsula		2001	Muscle	2	M	4-5	µg/g ww		0.063					0.044					0.0100											6
Russia, Kola Peninsula		2001	Kidney	2	F	1-2	µg/g ww		0.122					0.670					0.115											6
Russia, Kola Peninsula		2001	Kidney	2	F	2-3	µg/g ww		0.175					1.230					0.170											6
Russia, Kola Peninsula		2001	Kidney	1	F	4-5	µg/g ww		0.250					2.670					0.225											6
Russia, Kola Peninsula		2001	Liver	2	F	1-2	µg/g ww		0.113					0.235					0.038											6
Russia, Kola Peninsula		2001	Liver	2	F	2-3	µg/g ww		0.224					0.315					0.047											6
Russia, Kola Peninsula		2001	Liver	1	F	4-5	µg/g ww		0.310					0.520					0.072											6
Russia, Kola Peninsula		2001	Muscle	2	F	2-3	µg/g ww		0.048					0.033					0.0065											6
Russia, Kola Peninsula		2001	Muscle	1	F	4-5	µg/g ww		0.064					0.056					0.0120											6
Russia, Kola Peninsula		2001	Muscle	2	F	1-2	µg/g ww		0.039					0.028					<0.001											6
Russia, Pechora Basin		2000	Liver			2-5	µg/g ww		0.279	0.154				0.487	0.189				0.079	0.043										6
Russia, Pechora Basin		2001	Kidney	1	M	1-2	µg/g ww		0.115					0.585					0.2100											6
Russia, Pechora Basin		2001	Kidney	2	M	2-3	µg/g ww		0.161					1.620					0.2700											6
Russia, Pechora Basin		2001	Kidney	5	M	3-4	µg/g ww		0.410					3.350					0.4250											6
Russia, Pechora Basin		2001	Liver	1	M	1-2	µg/g ww		0.192					0.220					0.0380											6
Russia, Pechora Basin		2001	Liver	2	M	2-3	µg/g ww		0.247					0.396					0.0750											6
Russia, Pechora Basin		2001	Liver	5	M	3-4	µg/g ww		0.380					0.762					0.1260											6
Russia, Pechora Basin		2001	Muscle	5	M	3-4	µg/g ww		0.112					0.083					0.0180											6
Russia, Pechora Basin		2001	Muscle	1	M	1-2	µg/g ww		0.026					0.022					<0.001											6
Russia, Pechora Basin		2001	Muscle	2	M	2-3	µg/g ww		0.047					0.038					<0.001											6
Russia, Pechora Basin		2001	Kidney	3	F	2-3	µg/g ww		0.173					0.860					0.1700											6
Russia, Pechora Basin		2001	Kidney	3	F	3-4	µg/g ww		0.168					1.520					0.2650											6
Russia, Pechora Basin		2001	Kidney	1	F	4-5	µg/g ww		0.446					2.540					0.3760											6
Russia, Pechora Basin		2001	Liver	3	F	2-3	µg/g ww		0.134					0.420					0.0450											6
Russia, Pechora Basin		2001	Liver	3	F	3-4	µg/g ww		0.178					0.495					0.0540											6
Russia, Pechora Basin		2001	Liver	1	F	4-5	µg/g ww		0.543					0.630					0.1380											6
Russia, Pechora Basin		2001	Muscle	3	F	2-3	µg/g ww		0.052					0.068					0.0080											6
Russia, Pechora Basin		2001	Muscle	3	F	3-4	µg/g ww		0.074					0.065					0.0090											6
Russia, Pechora Basin		2001	Muscle	1	F	4-5	µg/g ww		0.091					0.073					0.0170											6
Russia, Taymir Peninsula		1993	Liver	20			µg/g ww		0.120	0.090				0.270	0.410				0.033	0.020										12
Russia, Taymir Peninsula, Dudinka		2000	Liver			2-5	µg/g ww		0.270	0.118				0.459	0.191				0.061	0.024										6
Russia, Taymir Peninsula, Dudinka		2001	Kidney	2	M	4-5	µg/g ww		0.122					0.670					0.1670											6
Russia, Taymir Peninsula, Dudinka		2001	Kidney	1	M	6-7	µg/g ww		0.175					1.230					0.2600											6
Russia, Taymir Peninsula, Dudinka		2001	Liver	2	M	4-5	µg/g ww		0.133					0.235					0.0550											6
Russia, Taymir Peninsula, Dudinka		2001	Liver	1	M	6-7	µg/g ww		0.244					0.315					0.0840											6
Russia, Taymir Peninsula, Dudinka		2001	Muscle	2	M	4-5	µg/g ww		0.067					0.038					0.0110											6
Russia, Taymir Peninsula, Dudinka		2001	Muscle	1	M	6-7	µg/g ww		0.086					0.047					0.0150											6
Russia, Taymir Peninsula, Dudinka		2001	Kidney	2	F	1-3	µg/g ww		0.190					0.740					0.2270											6
Russia, Taymir Peninsula, Dudinka		2001	Kidney	2	F	4-6	µg/g ww		0.285					1.480					0.3450											6
Russia, Taymir Peninsula, Dudinka		2001	Kidney	3	F	6-8	µg/g ww		0.470					3.670					0.3800											6
Russia, Taymir Peninsula, Dudinka		2001	Liver	2	F	1-3	µg/g ww		0.260					0.570					0.0320											6
Russia, Taymir Peninsula, Dudinka		2001	Liver	2	F	4-6	µg/g ww		0.324					0.465					0.0470											6
Russia, Taymir Peninsula, Dudinka		2001	Liver	3	F	6-8	µg/g ww		0.430					0.710					0.0890											6
Russia, Taymir Peninsula, Dudinka		2001	Muscle	2	F	4-6	µg/g ww		0.083					0.058					0.0120											6
Russia, Taymir Peninsula, Dudinka		2001	Muscle	3	F	6-8	µg/g ww		0.110					0.079					0.0160											6
Russia, Taymir Peninsula, Dudinka		2001	Muscle	2	F	1-3	µg/g ww		0.051					0.041					<0.001											6
Russia, Taymir Peninsula, Khatanga		2000	Liver			2-5	µg/g ww		0.113	0.033				0.268	0.077				0.032	0.016										6
Russia, Taymir Peninsula, Khatanga		2001	Kidney	1	M	1-2	µg/g ww		0.089					0.730					0.105											6
Russia, Taymir Peninsula, Khatanga		2001	Kidney	3	M	4-6	µg/g ww		0.110					1.250					0.130											6
Russia, Taymir Peninsula, Khatanga		2001	Liver	1	M	1-2	µg/g ww		0.092					0.270					0.013											6
Russia, Taymir Peninsula, Khatanga		2001	Liver	3	M	4-6	µg/g ww		0.154					0.365					0.027											6
Russia, Taymir Peninsula, Khatanga		2001	Muscle	1	M	1-2	µg/g ww		0.009					0.015					<0.001											6
Russia, Taymir Peninsula, Khatanga		2001	Muscle	3	M	4-6	µg/g ww		0.012					0.017					<0.001											6

Table A6. Metals in terrestrial mammals, continued

Species/location	Coordinate	Year	Tissue	n	Sex, U = unknown	Age (yrs)	Unit	Lead min-max	Lead arithm. mean	Lead SD	Lead geom. mean	Lead GSD	Cadmium min-max	Cadmium arithm. mean	Cadmium SD	Cadmium geom. mean	Cadmium GSD	Mercury min-max	Mercury arithm. mean	Mercury SD	Mercury geom. mean	Mercury GSD	Selenium min-max	Selenium arithm. mean	Selenium SD	Selenium geom. mean	Selenium GSD	Remark	Reference
Russia, Taymir Peninsula, Khatanga		2001	Kidney	3	F	1-3	µg/g ww		0.106					0.450					0.085									6
Russia, Taymir Peninsula, Khatanga		2001	Kidney	3	F	6-8	µg/g ww		0.138					0.890					0.110									6
Russia, Taymir Peninsula, Khatanga		2001	Liver	3	F	1-3	µg/g ww		0.081					0.175					0.036									6
Russia, Taymir Peninsula, Khatanga		2001	Liver	3	F	6-8	µg/g ww		0.125					0.260					0.051									6
Russia, Taymir Peninsula, Khatanga		2001	Muscle	3	F	1-3	µg/g ww		0.021					0.010					0.001									6
Russia, Taymir Peninsula, Khatanga		2001	Muscle	3	F	6-8	µg/g ww		0.035					0.014					0.001									6
Russia, Wrangell Island		1991	Liver	6			µg/g ww		1.150	0.420				0.210	0.060				0.240	0.080								12
Sweden, Norrbotten county (BD)	68°4.5'N, 19°50.2'E	1983	Liver	10	F	3	µg/g ww									0.43	0.099											13
Sweden, Norrbotten county (BD)	68°4.5'N, 19°50.2'E	1984	Liver	10	F	3	µg/g ww									0.45	0.168											13
Sweden, Norrbotten county (BD)	68°4.5'N, 19°50.2'E	1985	Liver	10	F	3	µg/g ww									0.39	0.115											13
Sweden, Norrbotten county (BD)	68°4.5'N, 19°50.2'E	1986	Liver	10	F	3	µg/g ww									0.30	0.099											13
Sweden, Norrbotten county (BD)	68°4.5'N, 19°50.2'E	1987	Liver	10	F	3	µg/g ww									0.37	0.118											13
Sweden, Norrbotten county (BD)	68°4.5'N, 19°50.2'E	1988	Liver	10	F	3	µg/g ww									0.47	0.148											13
Sweden, Norrbotten county (BD)	68°4.5'N, 19°50.2'E	1989	Liver	10	F	3	µg/g ww									0.47	0.197											13
Sweden, Norrbotten county (BD)	68°4.5'N, 19°50.2'E	1990	Liver	10	F	3	µg/g ww									0.53	0.104											13
Sweden, Norrbotten county (BD)	68°4.5'N, 19°50.2'E	1991	Liver	10	F	3	µg/g ww									0.50	0.119											13
Sweden, Norrbotten county (BD)	68°4.5'N, 19°50.2'E	1992	Liver	10	F	3	µg/g ww									0.52	0.14											13
Sweden, Norrbotten county (BD)	68°4.5'N, 19°50.2'E	1993	Liver	10	F	3	µg/g ww									0.37	0.09											13
Sweden, Norrbotten county (BD)	68°4.5'N, 19°50.2'E	1994	Liver	10	F	3	µg/g ww									0.50	0.163											13
Sweden, Norrbotten county (BD)	68°4.5'N, 19°50.2'E	1995	Liver	10	F	3	µg/g ww									0.56	0.146											13
Sweden, Norrbotten county (BD)	68°4.5'N, 19°50.2'E	1996	Liver	10	F	3	µg/g ww									0.45	0.141											13
Sweden, Norrbotten county (BD)	68°4.5'N, 19°50.2'E	1997	Liver	10	F	3	µg/g ww									0.50	0.19											13
Sweden, Norrbotten county (BD)	68°4.5'N, 19°50.2'E	1998	Liver	10	F	3	µg/g ww									0.71	0.384											13
Sweden, Central Lapland	65°58.2'N, 16°14.2'E	1996-2001	Liver	10	F	3	µg/g ww				0.134	(0.079-0.224)		0.488		0.479	(0.394-0.582)		0.035	(0.020-0.060)								14
Sweden, Northern Lapland	68°4.5'N, 19°50.3'E	1983-2001	Liver	10	F	3	µg/g ww				0.124	(0.098-0.157)		0.443		0.443	(0.403-0.487)		0.039	(0.031-0.049)								14
USA, Alaska, Barrow	71°17'N, 156°47'W	1995	Liver	6			µg/g ww	0.70-1.60	1.080	0.350			0.87-2.10	1.270	0.480				0.395	0.504								15
USA, Alaska, Point Hope	68°15'N, 166°30'W	1995	Liver	6			µg/g ww	0.41-1.00	0.730	0.210			1.4-2.60	1.870	0.510				0.635	0.412								15
USA, Alaska, Red Dog Mine	68°33'N, 164°W	1996	Liver	15		2-10	µg/g ww	0.82-2.88	1.650	0.600			0.11-0.70	0.430	0.180													16
USA, Alaska, Teshekpuk Lake	70°43'N, 153°50'W	1995	Liver	18		7-12	µg/g ww	0.11-1.11	0.770	0.290			0.56-2.52	1.200	0.540				0.040	0.340								15
Sorex araneus (Arctic shrew)																												
Finland	68°30'N, 24°09'E	1999	Kidney	26			µg/g ww	0.20-1.25					0.3-1.5						1.2-1.8								O	17
Finland	68°30'N, 24°09'E	2000	Kidney	27			µg/g ww	0.5-1.0					0.5-1.5						1.0-2.0								O	17
Vulpes vulpes (red fox)																												
Canada, northern Quebec		1981-90	Kidney	2(2)			µg/g ww							0.33														2
Canada, northern Quebec		1981-90	Liver	3(3)			µg/g ww		0.29					0.22					0.38					0.61				2
Canada, northern Quebec		1981-90	Muscle	3(3)			µg/g ww		0.13					<0.20					0.11					0.17				2
Canada, northern Quebec		1991-97	Liver	2(2)			µg/g ww		0.10					0.46					0.41					0.75				2
Canada, northern Quebec		1991-97	Muscle	2(2)			µg/g ww		<0.10					<0.20					0.22					0.20				2

Remarks

A. Calculated geo mean by G. Beauchamp, pers. comm. 2002 based on published data in Robillard *et al*. 2002;
B. Total 20 samples, 19 analyzed for Cd;
C. Total 11 samples, 10 analyzed for age;
D. Total 10 samples, 8 analyzed for age;
E. Total 19 samples, 17 analyzed for age;
F. Value in figure 4.5 is divided by 4 (dw to ww conversions);
G. Total 21 samples, 19 analyzed for Hg;
H. Total 31 samples, 30 analyzed for Hg;
I. Total 23 samples, 22 analyzed for age;
J. Total 7 samples, 1 analyzed for age;
K. Total 20 samples, 10 analyzed for age;
L. Total 26 samples, 14 analyzed for age;
M. Total 53 samples, 11 analyzed for age;
N. Total 56 samples, 17 analyzed for age;
O. Range of means.

References

1. Odsjö *et al*., 2001;
2. Champoux *et al*., 1999;
3. F. Riget, unpubl. data, 2002;
4. Olsen *et al*., 2003;
5. Kålås *et al*., 2000;
6. Melnikov *et al*., 2002;
7. Beauchamp, pers. comm., 2002;
8. Macdonald *et al*., 2002;
9. Elkin, 2001;
10. Elkin, 2000;
11. Aastrup *et al*., 2000;
12. Espelien *et al*., 1999;
13. T. Odsjö, unpubl. data, 2002;
14. Odsjö, 2003;
15. O'Hara *et al*., 2003; T. O'Hara, unpubl. data, 2002;
16. O'Hara *et al*., 2003;
17. J. Mannio, unpubl. data, 2001.

Table A7. Metals in freshwater.

Country/location	Coordinate	Year	Depth, m	n	Unit	Copper min-max	Copper mean	Zinc min-max	Zinc mean	Lead min-max	Lead mean	Cadmium min-max	Cadmium mean	Mercury min-max	Mercury mean	Remark	Reference
Lake water																	
Canada, Baffin Island S	68°26'N, 66°50'W				µg/L		0.3						0.004				1
Canada, Baffin Island S	68°26'N, 66°50'W				µg/L		0.6						0.006				1
Canada, Banks Island N	73°40'N, 116°12'W				µg/L		0.3						0.002				1
Canada, Cape Bathurst	70°46'N, 127°45'W				µg/L		0.6						0.009				1
Canada, Devon Island S	74°36'N, 82°24'W				µg/L		0.2						0.002				1
Canada, Ellef Ringnes Island	78°49'N, 103°40'W				µg/L		0.9						0.01				1
Canada, Ellesmere Island	76°28'N, 86°50'W				µg/L		0.1						0.004				1
Canada, Ivvavk National Park	69°26'N, 139°36'W				µg/L		1.2						0.086				1
Canada, Melville Island S	75°03 N, 107°51'W				µg/L		0.6						0.002				1
Russia, Imandra lake		1995			µg/L	2.5-4.6	3.3	0.2-2.5	0.7	0.004-0.09	0.03	0.01-0.08	0.04				2
Russia, Kola North		1995			µg/L	0.2-6.1	0.9	0.2-9.3	1.8							A	2
Russia, Kola North, Pechenganikel		1995			µg/L	1-20	5.5	0.7-3.2	1.6							B	2
Russia, Kola North, Pechenganikel		1995			µg/L	0.2-4.3	1.3	0.2-6.7	1.7							C	2
Russia, Kola North, Severonikel		1995			µg/L	1.8-17.9	6.8	1.1-5.8	2.2							B	2
Russia, Kola North, Severonikel		1995			µg/L	0.1-20	1.4	0.2-8.5	1.9							C	2
Russia, Kola Peninsula		2001		9	µg/L						0.277		0.096		0.018		3
Russia, Kola Peninsula		2001		3	µg/L						0.252		0.050		0.015		3
Russia, Kola Peninsula		2001		3	µg/L						0.289		0.072		0.035		3
Russia, Kola Peninsula		2001		3	µg/L						0.205		0.045		0.010		3
River water																	
Russia, Chukotka, Kanchalan		2001		8	µg/L						0.495		0.055		0.042		3
Russia, Lena		1991, 1993		8	µg/L	0.47-1.0	0.81	0.08-1.37	0.42	0.015-0.083	0.035	0.0022-0.012	0.006				4
Russia, Lena		1991		2	ng/L							2.2-12.0	6.0	0.9-1.08	1.0		5
Russia, Ob		1993		2	µg/L	1.85-2.43	2.12	<0.1-0.43	0.3	0.011-0.017	0.014	0.00056-0.0009	0.0007				6
Russia, Ob		1993		2	ng/L							0.56-0.90	0.7	0.48-0.64	0.56		5
Russia, Ob-Irtysh		1994, 1995		18	µg/L	1.83-4.81	2.82					0.0001-0.0151	0.0037				7
Russia, Pechora Basin		2001		4	µg/g						0.312		0.032		0.019		3
Russia, Pechora Basin		2001		4	µg/g						0.227		0.027		0.055		3
Russia, Pechora Basin		2001		4	µg/g						0.382		0.057		0.021		3
Russia, Taymir Peninsula, Dudinka		2001		4	µg/g						0.244		0.030		0.009		3
Russia, Taymir Peninsula, Dudinka		2001		4	µg/g						0.192		0.027		0.021		3
Russia, Taymir Peninsula, Khatanga		2001		4	µg/g						0.207		0.045		0.069		3
Russia, Taymir Peninsula, Khatanga		2001		4	µg/g						0.272		0.028		0.022		3
Russia, Yenisey		1993		3	µg/L	1.41-1.85	1.62	0.51-2.0	1.32	0.005-0.006	0.0055	0.0012-0.0018	0.0015				6
Russia, Yenisey		1993		3	ng/L							1.2-1.8	1.5	0.16-0.42	0.3		5

Remarks

A. At the distance of >100 km from integrated works.
B. In the radius <30 km from integrated works.
C. At the distance of 30-100 km from integrated works.

References

1. Borg *et al.*, 2001;
2. Moiseenko *et al.*, 1998;
3. Melnikov *et al.*, 2002;
4. Martin *et al.*, 1993; Guieu *et al.*, 1996;
5. Coquery *et al.*, 1995;
6. Dai and Martin 1995; Kravtsov *et al.*, 1994;
7. Moran and Woods, 1997.

Table A8. Metals in freshwater particulates.

Country/location	Coordinate	Year	Depth, m	n	Unit	Copper	Zinc	Lead	Mercury	Cadmium	Reference
Russia, Khatanga		1996		12	mg/kg dw	82	104	12		0.22	1
Russia, Lena		1991		2	mg/kg dw	28	143	23			2
Russia, Lena		1991		2	mg/kg dw				0.12		3
Russia, Lena		1991		6	mg/kg dw	28	160	36		0.25	4
Russia, Lena		1993		5	mg/kg dw	42	185	42		0.96	5
Russia, Lena		1994, 1995		31	mg/kg dw	35	141	24		0.65	1
Russia, Ob		1993		6	mg/kg dw	50	104	16		0.53	6
Russia, Ob		1993		2	mg/kg dw				0.05		3
Russia, Ob-Irtysh		1994, 1995		18	mg/kg dw	25.8		14.9		0.58	7
Russia, Yana		1995		7	mg/kg dw	30	130	23		0.32	1
Russia, Yenisey		1993		6	mg/kg dw	144	220	30		2.2	6
Russia, Yenisey		1993		3	mg/kg dw				0.05		3

References

1. Rachold, 1999;
2. Martin *et al.*, 1993;
3. Coquery *et al.*, 1995;
4. Gordeev and Shevchenko, 1995;
5. Nolting *et al.*, 1996;
6. Gordeev *et al.*, 1995;
7. Moran and Woods, 1997.

Table A9. Metals in freshwater sediments.

Country/location	Coordinate	Year	Depth, m	n	Unit	Copper min-max	Copper mean	Copper SD	Zinc min-max	Zinc mean	Zinc SD	Lead min-max	Lead mean	Lead SD	Cadmium min-max	Cadmium mean	Cadmium SD	Mercury min-max	Mercury mean	Mercury SD	Remark	Reference
Russia																						
Chukotka, Kanchalan	65.048-65.518N, 176.315-177.380E	2001		10	µg/g dw								10.335			0.149			0.044			1
Chukotka, Lavrentiya	65.115-65.715N, 170.942-171.786W	2001		10	µg/g dw								2.901			0.141			0.055			1
Imandra Lake		1995			µg/g dw	60-471	211		42-303	124		3.4-30	16		0.22-1.3	0.61						2
Imandra Lake		1997-2001			µg/g dw		50.6	29		101.2	29		18.1	6.2		1.72	0.73		0.03			3
Kola North		1995			µg/g	3-72	23		18-195	74		3-49	16		0.11-4.24	1.24		0.009-0.20	0.11		A	2
Kola North, Pechenganikel		1995			µg/g	9-6495	627		45-439	136		9-49	25		0.20-3.84	1.21		0.005-0.51	0.25		B	2
Kola North, Pechenganikel		1995			µg/g	5-250	88		17-1327	206		1-176	39		0.10-5.38	0.57		0.005-0.27	0.12		C	2
Kola North, Severonikel		1995			µg/g	28-3779	595		97-374	195		8-62	32		0.40-2.50	1.44		0.03-0.39	0.16		B	2
Kola North, Severonikel		1995			µg/g	35-89	63		112-200	155		8-24	16		0.40-0.77	0.59					C	2
Kola Peninsula		2001		10	µg/g dw								2.901			0.092			0.265			1
Pasvic system		1997-2001			µg/g dw		59	17.3		123.4	24		18.7	11.8		1.45	1.14		0.023	0.005		3
Pechora Basin		2001		10	µg/g dw								6.028			0.075			0.297			1
River Ob Estuary					ppm								36									3
River Yenisey Estuary					ppm								30									3
Small lakes		1997-2001			µg/g dw		44.2	47.4		101.9	83.8		7.4	10.3		0.96	1.18		0.71	0.048		3
Taymir Peninsula, Dudinka		2001		10	µg/g dw								5.624			0.058			0.236			1
Taymir Peninsula, Khatanga		2001		10	µg/g dw								4.824			0.047			0.189			1

Remarks

A. At the distance of >100 km from integrated works.
B. In the radius <30 km from integrated works.
C. At the distance of 30-100 km from integrated works.

References

1. Melnikov *et al.*, 2002;
2. Moiseenko *et al.*, 1998;
3. Tsibulski *et al.*, 2001.

Table A10. Metals in freshwater fish.

Species/location	Coordinate	Year	Tissue	n	Sex	Age, yrs (SD)	Length, cm (SD)	Weight, g (SD)	Unit	Lead min-max	Lead arithm. mean	Lead SD	Lead geom. mean	Lead GSD	Cadmium min-max	Cadmium arithm. mean	Cadmium SD	Cadmium geom. mean	Cadmium GSD	Mercury min-max	Mercury arithm. mean	Mercury SD	Mercury geom. mean	Mercury GSD	Selenium min-max	Selenium arithm. mean	Selenium SD	Selenium geom. mean	Re-mark	Refer-ence
Abramis brama (bream)																														
Finland, Kemijoki River		1994	Muscle	3			25.5	201	µg/g ww												0.06	0.012								1
Coregonus albula (vendace)																														
Finland, Lake Inari	69°00'N, 27°45'E	2000	Muscle	10	F/M			83	µg/g ww												0.16									2
Coregonus autumnalis (Arctic cisco)																														
Russia, Taymir Peninsula, Dudinka			Liver	4	F	8-10			µg/g ww		0.113					0.062					0.114									3
Russia, Taymir Peninsula, Dudinka			Liver	6	F	10-12			µg/g ww		0.204					0.102					0.136									3
Russia, Taymir Peninsula, Dudinka			Muscle	6	F	10-12			µg/g ww		0.024					0.014					0.036									3
Russia, Taymir Peninsula, Dudinka			Muscle	4	F	8-10			µg/g ww		0.018					0.011					0.048									3
Russia, Taymir Peninsula, Dudinka			Liver	8	M	7-10			µg/g ww		0.094					0.086					0.094									3
Russia, Taymir Peninsula, Dudinka			Liver	2	M	10-12			µg/g ww		0.081					0.079					0.107									3
Russia, Taymir Peninsula, Dudinka			Muscle	8	M	7-10			µg/g ww		0.019					0.016					0.025									3
Russia, Taymir Peninsula, Dudinka			Muscle	2	M	10-12			µg/g ww		0.024					0.015					0.052									3
Coregonus clupeaformis (lake whitefish)																														
Canada, Jacques Lake	66°10'N, 127°24'W	1994, 95	Muscle	20			42.0		µg/g ww												0.285									4
Canada, Aubry Lake	67°23'52"N, 126°26'44"W	1999	Muscle	14			50.7		µg/g ww												0.048									4
Canada, Bovie Lake	60°10'N, 122°56' W	1988	Muscle	5			51.3		µg/g ww												0.152									4
Canada, Burnt Lake	67°26'N, 128°10' W	1989	Muscle	2			44.7		µg/g ww												0.03									4
Canada, Carcajou Lake	64°41'N, 127°54' W	1978	Muscle	5			48.0		µg/g ww												0.152									4
Canada, Cli Lake	61°59'N., 123°18' W	1996, 2000	Muscle	43			49.5		µg/g ww												0.078									4
Canada, Colville Lake	67°09'32"N, 125°58'34"W	1993, 1999	Muscle	54			44.7		µg/g ww												0.03									4
Canada, Deep Lake	61°12.5'N, 120°54.5' W	2000	Muscle	28			44.4		µg/g ww												0.249									4
Canada, Dubawnt Lake, NWT	63°03'45"N, 102°05'45"W	1971	Muscle	3			55.4		µg/g ww												0.107									4
Canada, Ekali Lake	61°17'N, 120°35'W	1996	Muscle	20			47.0		µg/g ww												0.082									4
Canada, Ellice River	68°02'N, 103°58'W	1977	Muscle	24			46.8		µg/g ww												0.054									4
Canada, Ferguson Lake, Nunavut	62°55'N, 96°53'W	1971	Muscle	1			40.6		µg/g ww												0.13									4
Canada, Giauque Lake	63°11'N, 113°51'W	1977, 1992	Muscle	30			50.5		µg/g ww												0.881									4
Canada, Gordon Lake	63°05'N, 113°11'W	1981	Muscle	5			33.4		µg/g ww												0.034									4
Canada, Gray Lake	61°52'N, 108°15'W	1986	Muscle	15			46.5		µg/g ww												0.177									4
Canada, Great Slave Lake	62°50'N, 113°50'W	1975, 76, 77, 78, 88, 89, 95	Muscle	239			40.8		µg/g ww												0.062									4
Canada, Hay River	60°51'50"N, 115°44'00"W	1984, 89, 90, 92	Muscle	85			38.8		µg/g ww												0.056									4
Canada, Hidden Lake, NWT	62°33'N, 113°33'W	1978	Muscle	2			51.5		µg/g ww												0.3									4
Canada, Hjalmar Lake	61°33'N, 109°25'W	1986	Muscle	15			49.1		µg/g ww												0.106									4
Canada, Hottah Lake	65°04'N,118°30' W	1972	Muscle	5			50.1		µg/g ww												0.038									4
Canada, Island	66°54'N, 126°34'48"W	1989	Muscle	1			58.0		µg/g ww												0.34									4
Canada, Kaminuriak Lake	92°57'N, 95°46'12"W	1971, 75	Muscle	2			48.4		µg/g ww												0.13									4
Canada, Kelly Lake	65°23'51"N, 126°09'39"W	1998	Muscle	79			49.5		µg/g ww												0.165									4
Canada, Leland Lake	60°00'N, 110°59'W	1989, 90, 92	Muscle	30			43.7		µg/g ww												0.111									4
Canada, Liard River NWT	61°50'55"N, 121°18'30"W	1977	Muscle	8			36.4		µg/g ww												0.053									4
Canada, Little Doctor Lake	61°53'N, 123°16'W	1996	Muscle	18			40.7		µg/g ww												0.13									4
Canada, Loche Lake	67°36'N, 132°20'W	1978	Muscle	5			44.4		µg/g ww												0.17									4
Canada, Mackenzie Delta	68°50'N, 136°25'W	1977, 81, 92	Muscle	42			47.1		µg/g ww												0.061									4
Canada, Mackenzie River	69°21'N, 133°54'W	1971, 85, 89, 95	Muscle	16			36.0		µg/g ww												0.091									4
Canada, Maguire Lake	63°13'N, 113°54'W	1977	Muscle	2			55.0		µg/g ww												0.28									4
Canada, Mahoney Lake	65°30'51"N, 125°21'30"W	1996	Muscle	20			51.0		µg/g ww												0.131									4
Canada, Manuel Lake	66°58'N, 128°54'W	1978, 89, 93, 95, 97, 98	Muscle	50			50.5		µg/g ww												0.111									4
Canada, McCrea, Lake NWT	63°33'N, 112°35'W	1977	Muscle	1			56.0		µg/g ww												0.3									4
Canada, McEwan Lake	60°49'N, 119°57'W	2000	Muscle	30			45.0		µg/g ww												0.088									4
Canada, McGill Lake	61°18'N, 121°00'45"W	2000	Muscle	29			39.2		µg/g ww												0.15									4
Canada, Mirror Lake, NWT	64°51'N, 126°55'W	2000	Muscle	107			45.7		µg/g ww						0.212-0.616	0.351	0.071													4.5
Canada, Nonacho Lake	61°59'N, 109°28'W	1975, 86	Muscle	24			50.0		µg/g ww												0.162									4
Canada, Quartzite Lake	67°10'N, 81°40'W	1971	Muscle	2			32.6		µg/g ww												0.22									4
Canada, Reade Lake	60°54'N, 119 55'W	2000	Muscle	30			42.7		µg/g ww												0.146									4
Canada, Rorey Lake	66°55'N, 128°24'W	1978	Muscle	1			37.0		µg/g ww												0.04									4
Canada, Sibbeston Lake	61°45'N, 122°45'W	1997	Muscle	143			42.2		µg/g ww												0.071									4
Canada, Sibbeston Lake, NWT	61°45'N, 122°45W	1998/99	Muscle	143			42.2		µg/g ww												0.071	0.024								5
Canada, Slave River Lake	61°18'N, 113°40'W	1990, 92	Muscle	64			37.9		µg/g ww												0.057									4
Canada, St. Thérèse Lake	64°37'41"N, 121°35'52"W	1992	Muscle	23			46.3		µg/g ww												0.131									4
Canada, Tagatui Lake	64°57'20, "N125°13'50"W	1996	Muscle	20			33.7		µg/g ww												0.035									4
Canada, Thistlethwaite Lake	63°10'N, 113°34'W	1977	Muscle	10			44.7		µg/g ww												0.063									4

Table A10. Metals in freshwater fish, continued.

Species/location	Coordinate	Year	Tissue	n	Sex	Age, yrs (SD)	Length, cm (SD)	Weight, g (SD)	Unit	Lead min-max	Lead arithm. mean	Lead SD	Lead geom. mean	Lead GSD	Cadmium min-max	Cadmium arithm. mean	Cadmium SD	Cadmium geom. mean	Cadmium GSD	Mercury min-max	Mercury arithm. mean	Mercury SD	Mercury geom. mean	Mercury GSD	Selenium min-max	Selenium arithm. mean	Selenium SD	Selenium geom. mean	Re-mark	Refer-ence
Canada, Thompson Lake	62°37'N, 113°30'W	1978	Muscle	33			37.7		µg/g ww												0.201									4
Canada, Tree River	67°15'15"N, 132°34'10"W	1977	Muscle	4			41.6		µg/g ww												0.04									4
Canada, Tsetso Lake	61°51'N, 123°01'W	1997	Muscle	102			42.1		µg/g ww												0.075									4
Canada, Turton Lake	65°48'25"N, 126°57'W	1996	Muscle	12			41.9		µg/g ww												0.113									4
Canada, Wagenitz Lake	63°03'N, 11°52'W	1977	Muscle	15			44.5		µg/g ww												0.057									4
Canada, Willow Lake	62°10'40"N, 119°06'32"W	1999	Muscle	167			40.5		µg/g ww												0.085									4
Canada, Yathkyed Lake	62°40'N, 97°58'W	1971	Muscle	1			50.8		µg/g ww												0.12									4
Canada, Yeltea Lake	66°55'N, 129°22'W	1978	Muscle	3			46.7		µg/g ww												0.137									4
Coregonus lavaretus (European whitefish)																														
Russia, Kola Peninsula			Liver	4	F	8-11			µg/g ww		0.117					0.277					0.439									3
Russia, Kola Peninsula			Liver	4	F	11-13			µg/g ww		0.148					0.305					0.697									3
Russia, Kola Peninsula			Muscle	4	F	8-11			µg/g ww		0.012					0.029					0.142									3
Russia, Kola Peninsula			Muscle	4	F	11-13			µg/g ww		0.012					0.047					0.161									3
Russia, Kola Peninsula			Liver	6	M	9-11			µg/g ww		0.152					0.220					0.375									3
Russia, Kola Peninsula			Liver	6	M	11-14			µg/g ww		0.173					0.256					0.418									3
Russia, Kola Peninsula			Muscle	6	M	9-11			µg/g ww		0.021					0.034					0.122									3
Russia, Kola Peninsula			Muscle	6	M	11-14			µg/g ww		0.019					0.032					0.177									3
Russia, Pechora Basin			Liver	4	F	9-11			µg/g ww		0.140					0.143					0.124									3
Russia, Pechora Basin			Liver	3	F	11-14			µg/g ww		0.154					0.136					0.152									3
Russia, Pechora Basin			Muscle	4	F	9-11			µg/g ww		0.012					0.016					0.051									3
Russia, Pechora Basin			Muscle	3	F	11-14			µg/g ww		0.015					0.018					0.059									3
Russia, Pechora Basin			Liver	5	M	7-9			µg/g ww		0.179					0.110					0.124									3
Russia, Pechora Basin			Liver	7	M	9-11			µg/g ww		0.192					0.206					0.132									3
Russia, Pechora Basin			Liver	1	M	12-13			µg/g ww		0.186					0.242					0.134									3
Russia, Pechora Basin			Muscle	5	M	7-9			µg/g ww		0.015					0.016					0.052									3
Russia, Pechora Basin			Muscle	1	M	12-13			µg/g ww		0.018					0.037					0.056									3
Russia, Pechora Basin			Muscle	7	M	9-11			µg/g ww		0.015					0.021					0.057									3
Russia, Taymir Peninsula, Dudinka			Liver	6	F	8-11			µg/g ww		0.293					0.073					0.158									3
Russia, Taymir Peninsula, Dudinka			Liver	4	F	11-14			µg/g ww		0.282					0.097					0.218									3
Russia, Taymir Peninsula, Dudinka			Muscle	4	F	11-14			µg/g ww		0.052					0.021					0.078									3
Russia, Taymir Peninsula, Dudinka			Muscle	6	F	8-11			µg/g ww		0.029					0.013					0.080									3
Russia, Taymir Peninsula, Dudinka			Liver	7	M	11-14			µg/g ww		0.256					0.114					0.194									3
Russia, Taymir Peninsula, Dudinka			Liver	3	M	8-11			µg/g ww		0.186					0.084					0.202									3
Russia, Taymir Peninsula, Dudinka			Muscle	7	M	11-14			µg/g ww		0.037					0.036					0.061									3
Russia, Taymir Peninsula, Dudinka			Muscle	3	M	8-11			µg/g ww		0.034					0.023					0.090									3
Russia, Taymir Peninsula, Khatanga			Liver	1	F	8-9			µg/g ww		0.256					0.125					0.280									3
Russia, Taymir Peninsula, Khatanga			Liver	3	F	10-12			µg/g ww		0.198					0.160					0.356									3
Russia, Taymir Peninsula, Khatanga			Liver	3	F	12-14			µg/g ww		0.280					0.410					0.460									3
Russia, Taymir Peninsula, Khatanga			Muscle	3	F	12-14		214	µg/g ww		0.042					0.058					0.086									3
Russia, Taymir Peninsula, Khatanga			Muscle	1	F	8-9			µg/g ww		0.026					0.015					0.110									3
Russia, Taymir Peninsula, Khatanga			Muscle	3	F	10-12			µg/g ww		0.036					0.021					0.126									3
Russia, Taymir Peninsula, Khatanga			Liver	3	M	6-9			µg/g ww		0.124					0.140					0.154									3
Russia, Taymir Peninsula, Khatanga			Liver	6	M	9-12			µg/g ww		0.250					0.106					0.160									3
Russia, Taymir Peninsula, Khatanga			Liver	4	M	12-14			µg/g ww		0.264					0.257					0.258									3
Russia, Taymir Peninsula, Khatanga			Muscle	3	M	6-9			µg/g ww		0.005					0.012					0.073									3
Russia, Taymir Peninsula, Khatanga			Muscle	4	M	12-14			µg/g ww		0.010					0.019					0.112									3
Russia, Taymir Peninsula, Khatanga			Muscle	6	M	9-12			µg/g ww		0.012					0.010					0.095									3
Coregonus nasus (broad whitefish)																														
Canada, Mackenzie Delta	68°50'N, 136°25'W	1981	Muscle	6			48.8		µg/g ww												0.023									4
Canada, Mackenzie River	69°21'N, 133°54'W	1971, 95	Muscle	144			48.8		µg/g ww												0.056									4
Canada, Peel River, NWT	67°41'50"N, 134°31'50"W	1999	Muscle	10	F/M	11	52.0	2200	µg/g ww							n.d.					0.082	0.051							6
Canada, Peel River, Yukon	67°00'N, 135°00'W	1999	Muscle	1					µg/g ww												0.08									4
Russia, Chukotka, Kanchalan			Liver	13	F				µg/g ww		0.105					0.123					0.154									3
Russia, Chukotka, Kanchalan			Muscle	13	F				µg/g ww		0.010					0.010					0.083									3
Russia, Chukotka, Kanchalan			Liver	7	M				µg/g ww		0.115					0.246					0.290									3
Russia, Chukotka, Kanchalan			Muscle	7	M				µg/g ww		0.018					0.032					0.136									3
Russia, Taymir Peninsula, Khatanga			Liver	2	F	8-9			µg/g ww		0.155					0.179					0.124									3
Russia, Taymir Peninsula, Khatanga			Liver	5	F	9-12			µg/g ww		0.170					0.123					0.142									3
Russia, Taymir Peninsula, Khatanga			Liver	2	F	12-14			µg/g ww		0.120					0.205					0.242									3
Russia, Taymir Peninsula, Khatanga			Muscle	2	F	8-9			µg/g ww		0.007					0.022					0.055									3
Russia, Taymir Peninsula, Khatanga			Muscle	5	F	9-12			µg/g ww		0.005					0.020					0.090									3
Russia, Taymir Peninsula, Khatanga			Muscle	2	F	12-14			µg/g ww		0.015					0.042					0.122									3
Russia, Taymir Peninsula, Khatanga			Liver	6	M	9-11			µg/g ww		0.202					0.299					0.189									3
Russia, Taymir Peninsula, Khatanga			Liver	1	M	8-9			µg/g ww		0.149					0.170					0.224									3
Russia, Taymir Peninsula, Khatanga			Liver	4	M	13-14			µg/g ww		0.160					0.285					0.269									3
Russia, Taymir Peninsula, Khatanga			Muscle	1	M	8-9			µg/g ww		0.005					0.006					0.056									3
Russia, Taymir Peninsula, Khatanga			Muscle	6	M	9-11			µg/g ww		0.012					0.015					0.082									3

Table A10. Metals in freshwater fish, continued.

Species/location	Coordinate	Year	Tissue	n	Sex	Age, yrs (SD)	Length, cm (SD)	Weight, g (SD)	Unit	Lead min-max	Lead arithm. mean	Lead arithm. SD	Lead geom. mean	Lead geom. GSD	Cadmium min-max	Cadmium arithm. mean	Cadmium arithm. SD	Cadmium geom. mean	Cadmium geom. GSD	Mercury min-max	Mercury arithm. mean	Mercury arithm. SD	Mercury geom. mean	Mercury geom. GSD	Selenium min-max	Selenium arithm. mean	Selenium arithm. SD	Selenium geom. mean	Remark	Reference
Russia, Taymir Peninsula, Khatanga			Muscle	4	M	13-14			µg/g ww		0.014					0.013					0.087									3
USA, Alaska, Chipp and Meade River			Liver						µg/g ww												0.080								A	7
USA, Alaska, Chipp and Meade River			Muscle						µg/g ww							<0.01					0.030								A	7
USA, Alaska, Lake 1, North Slope	70°19'N, 156°19'W	2000, 2001	Muscle	19					mg/kg ww				0.006					0.001					0.024					0.310		8
USA, Alaska, Lake 12, North Slope	70°08'N, 151°05'W	2000, 2001	Muscle	14					mg/kg ww				0.009					0.001					0.047					0.492		8
USA, Alaska, Lake 13, North Slope	70°15'N, 156°07'W	2000, 2001	Muscle	14					mg/kg ww				0.001					0.001					0.020					0.322		8
USA, Alaska, Lake 3, North Slope	70°19'N, 155°25'W	2000, 2001	Muscle	20					mg/kg ww				0.002					0.001					0.021					0.259		8
USA, Alaska, Lake 4, North Slope	70°13'N, 155°28'W	2000, 2001	Muscle	14					mg/kg ww				0.002					0.001					0.030					0.260		8
Coregonus nelsoni (Alaska whitefish)																														
USA, Alaska, Kuskokwim River		2000	Muscle	6					µg/g ww												0.032	0.013								9
USA, Alaska, Kuskokwim River			Liver	6					µg/g ww												0.057	0.020								9
Coregonus peled (peled)																														
Finland, Kemijoki river, reservoirs		1994	Muscle	5			30.9	318	µg/g ww												0.16	0.019								1
Coregonus sardinella (least cisco)																														
USA, Alaska, Lake 0, North Slope	71°14'N, 156°38'W	2000, 2001	Muscle	12					mg/kg ww				0.005					0.001					0.049					0.322		8
USA, Alaska, Lake 12, North Slope	70°08'N, 151°05'W	2000, 2001	Muscle	15					mg/kg ww				0.010					0.003					0.102					0.478		8
USA, Alaska, Lake 3, North Slope	70°19'N, 155°25'W	2000, 2001	Muscle	21					mg/kg ww				0.001					0.003					0.041					0.266		8
USA, Alaska, Lake 9, North Slope	70°24'N, 151°04'W	2000, 2001	Muscle	14					mg/kg ww				0.002					0.001					0.015					0.340		8
Coregonus spp. (whitefish) (cisco)																														
Canada, Baker Lake, YT	61°00'N, 134°04'W	1998	Muscle	6					µg/g ww		<0.019					<0.002					0.072	0.007								10
Canada, Bandy Lake	65°16'05"N, 126°29'45"W	2000	Muscle	19			32.5		µg/g ww												0.229									4
Canada, Bandy Lake, western NWT	65°16,083'N, 126°29,75'W	2000	Muscle	19					µg/g ww											0.040-0.507	0.229	0.149								5
Canada, Cli Lake	61°59'N, 123°18'W	2000	Muscle			7.5 (2.6)	49.85 (4.4)		µg/g ww												0.080	0.030								11
Canada, Deep Lake	61°12.5'N, 120°54.5'W	2000	Muscle	1			15.5		µg/g ww												0.26									4
Canada, Ekali Lake	61°17'N, 120°35'W	1996	Muscle	20			33.8		µg/g ww												0.118									4
Canada, Great Bear Lake	65°50'N, 120°45'W	1978	Muscle	25			30.4		µg/g ww												0.02									4
Canada, Great Bear Lake	65°50'N, 120°45'W	2001	Muscle	10			30.66 (1.90)	252.21 (45.7)	µg/g ww												0.0245	0.0050								12
Canada, Hidden Lake, NWT	66°00'N, 117°51'W	1978	Muscle	2			28.5		µg/g ww												0.09									4
Canada, Kam Lake	62°25'15"N, 114°24'10"W	1976	Muscle	13			36.8		µg/g ww												0.106									4
Canada, Little Doctor Lake	61°53'N, 123°16'W	2000	Muscle			10.0 (1.8)	40.69 (2.57)		µg/g ww												0.130	0.060								11
Canada, McEwan Lake	60°49'N, 119°57'W	2000	Muscle	35			19.7		µg/g ww												0.093									4
Canada, Sanguez Lake	61°15'N, 120°28'W	1996	Muscle	12			26.7		µg/g ww												0.158									4
Canada, Sibbeston Lake	61°45'N, 122°45'W	2000	Muscle			11.9 (5.0)	42.1 (0.33)		µg/g ww												0.07	0.02								11
Canada, St. Thérèse Lake	64°37'N, 121°52.9'W	2001	Muscle	9			50.83 (3.41)	1442.7 (315.7)	µg/g ww												0.2592	0.1238								12
Canada, Tsetso Lake	61°51'N, 123°01'W	2000	Muscle			14.6 (5.2)	42.11 (2.25)		µg/g ww												0.070	0.030								11
Canada, Tunago Lake	66°19'N, 125°50'W	1978	Muscle	1			42.0		µg/g ww												0.14									4
Canada, Willow Lake	62°10'40"N, 119°06'32"W	2000	Muscle			8.8 (2.5)	40.80 (4.64)		µg/g ww												0.090	0.040								11
Finland, Kemijoki River		1994	Muscle	53			30.5	275	µg/g ww											0.07-0.27	0.16	0.051								1
Finland, Kemijoki River, reservoirs		1994	Muscle	47			30.6	279	µg/g ww												0.16	0.053								1
Finland, Kemijoki River, upstream		1994	Muscle	6			29.6	242	µg/g ww												0.16	0.037								1
USA, Alaska, Kuskokwim river			Muscle						µg/g ww												0.026	0.011							B	13
Esox lucius (northern pike)																														
Canada, a Jacques Lake	66°10'N, 127°24'W	1994, 1995	Muscle	30			65.8		µg/g ww												0.505									4
Canada, Bandy Lake, western NWT	65°16.083'N, 126°29,75'W	Apr. 2000	Muscle	19					µg/g ww												0.323	0.192								5
Canada, Basler Lake	63°57'N, 115°58'W	1982, 1983	Muscle	10			73.4		µg/g ww												0.434									4
Canada, Carcajou Lake	64°41'N, 127°54'W	1978	Muscle	1			66.0		µg/g ww												0.44									4
Canada, Cli Lake	61°59'N, 123°18'W	2000	Muscle	5			66.0		µg/g ww												0.353									4
Canada, Colville Lake	67°09'32"N, 125°58'34"W	1999	Muscle	7			63.0		µg/g ww												0.244									4
Canada, Deep Lake	61°12.5'N, 120°54.5'W	2000	Muscle	6			58.4		µg/g ww												0.67									4
Canada, Ekali Lake	61°17'N, 120°35'W	1996	Muscle	7			57.8		µg/g ww												0.3									4
Canada, Fork Lake	60°53'N, 111°05'W	1992	Muscle	5			31.4		µg/g ww												0.262									4
Canada, Gargon Lake	61°15'N, 120°22'30"W	1996	Muscle	2			66.8		µg/g ww												0.587									4
Canada, Giauque Lake	63°11'N, 113°51'W	1977, 92	Muscle	39			58.5		µg/g ww												1.705									4
Canada, Great Slave Lake	62°50'N, 113°50'W	1976, 77, 78, 88, 89, 96	Muscle	111			59.1		µg/g ww												0.242									4
Canada, Great Slave Lake, East Arm	62°24'N, 110°44'W	1999	Muscle						µg/g ww												0.20	0.14				0.35	0.1			14
Canada, Great Slave Lake, West Basin	61°10'N, 113°40'W	1996	Muscle						µg/g ww												0.25	0.02								14
Canada, Great Slave Lake, West Basin	61°10'N, 113°40'W	1999	Muscle						µg/g ww												0.24	0.10				0.25	0.02			14

Table A10. Metals in freshwater fish, continued.

Species/location	Coordinate	Year	Tissue	n	Sex	Age, yrs (SD)	Length, cm (SD)	Weight, g (SD)	Unit	Lead min-max	Lead arithm. mean	Lead SD	Lead geom. mean	Lead GSD	Cadmium min-max	Cadmium arithm. mean	Cadmium SD	Cadmium geom. mean	Cadmium GSD	Mercury min-max	Mercury arithm. mean	Mercury SD	Mercury geom. mean	Mercury GSD	Selenium min-max	Selenium arithm. mean	Selenium SD	Selenium geom. mean	Re-mark	Refer-ence
Canada, Hanson Lake	64°00'40"N, 125°20'50"W	1992	Muscle	4			52.1		µg/g ww												0.134									4
Canada, Hay River	60°51'50"N, 115°44'00"W	1984, 89, 92	Muscle	66			61.4		µg/g ww												0.295									4
Canada, Hjalmar Lake	61°33'N, 109°25'W	1986	Muscle	5			68.5		µg/g ww												0.72									4
Canada, Hottah Lake	65°04'N, 118°30'W	1972	Muscle	2			73.8		µg/g ww												0.265									4
Canada, Kakisa Lake	60°56'N, 117°43'W	1977, 78, 81, 97, 98	Muscle	43			53.8		µg/g ww												0.282									4
Canada, Kelly Lake	65°23'51"N, 126°09'39"W	1998	Muscle	13			67.3		µg/g ww												0.546									4
Canada, Koksoak River, Quebec	58°32'N, 68°10'W		Muscle	3					µg/g ww												0.530							0.277	C	4
Canada, Lake Bandy, NWT	65°16.083'N, 126°29.75'W	2000	Muscle	19			59.1		µg/g ww											0.063-0.751	0.323	0.192							4,5
Canada, Leland Lake	61°00'N, 111°59'W	1989, 90, 92	Muscle	42			55.7		µg/g ww												0.347									4
Canada, Little Doctor Lake	61°53'N, 123°16'W	1996	Muscle	10			69.5		µg/g ww												0.772									4
Canada, Loche Lake	67°36'N, 132°20'W	1978	Muscle	5			68.8		µg/g ww												0.508									4
Canada, Loon Lake	66°36'30"N, 128°43'15"W	1997	Muscle	14			64.8		µg/g ww												0.505									4
Canada, Mackenzie Delta	68°50'N, 136°25'W	1977, 92	Muscle	36			98.0		µg/g ww												0.49									4
Canada, Mackenzie River	69°21'N, 133°54'W	1971, 77, 85, 97	Muscle	34			49.8		µg/g ww												0.148									4
Canada, Maguire Lake	63°13'N, 113°54'W	1977	Muscle	4			59.3		µg/g ww												0.488									4
Canada, Mahoney Lake	65°30'51"N, 125°21'30"W	1996	Muscle	20			66.0		µg/g ww												0.257									4
Canada, Manuel Lake	66°58'N, 128°54'W	1978, 95, 97, 98	Muscle	42			63.7		µg/g ww												0.441									4
Canada, Marian River	63°04'N, 116°21'W	1979	Muscle	23			64.4		µg/g ww												0.437									4
Canada, Mc Crea Lake, NWT	63°33'N, 112°35'W	1977	Muscle	8			60.0		µg/g ww												0.494									4
Canada, McEwan Lake	60°49'N, 119°57'W	2000	Muscle	15			56.9		µg/g ww												0.331									4
Canada, McGill Lake	61°18'N, 121°00'45"W	2000	Muscle	28			57.7		µg/g ww												0.713									4
Canada, Nares Lake	60°09'30"N, 134°39'35"W	1996	Muscle	3			87.0		µg/g ww												0.258									4
Canada, Nonacho Lake	61°59'N, 109°28'W	1986	Muscle	20			58.8		µg/g ww												0.391									4
Canada, Reade Lake	60°54'N, 119°55'W	2000	Muscle	45			58.4		µg/g ww												0.43									4
Canada, Sanguez Lake	61°15'N, 120°28'W	1996	Muscle	20			68.3		µg/g ww												0.703									4
Canada, Sibbeston Lake	61°45'N, 122°45'W	1997	Muscle	2			67.1		µg/g ww												0.165									4
Canada, Slave River Lake	61°18'N, 113°40'W	1988, 89, 90, 92	Muscle	92			57.3		µg/g ww												0.352									4
Canada, Sparrow Lake	62°37'N, 113°38'W	1983	Muscle	3			72.5		µg/g ww												0.253									4
Canada, St. Thérèse Lake	64°37'41"N, 121°35'52"W	1980, 92	Muscle	20			79.0		µg/g ww												1.105									4
Canada, Steep Bank Bay	63°35'N, 91°38'W	1992	Muscle	5			39.4		µg/g ww												0.214									4
Canada, Tagatui Lake	64°57'20"N, 125°13'50"W	1996	Muscle	16			63.4		µg/g ww												0.17									4
Canada, Taltson River	64°24'N, 112°45'W	1982	Muscle	5			66.1		µg/g ww												0.464									4
Canada, Tathlina Lake	60°33'N, 117°32'W	1981, 86	Muscle	10			51.6		µg/g ww												0.287									4
Canada, Thistlethwaite	63°10'N, 113°34'W	1977	Muscle	8			58.6		µg/g ww												0.233									4
Canada, Thompson Lake	62°37'N, 113°30'W	1978	Muscle	16			53.5		µg/g ww												1.692									4
Canada, Trout Lake	60°34'20"N, 121°15'33"W	1977	Muscle	3			81.3		µg/g ww												0.27									4
Canada, Tsetso Lake	61°51'N, 123°01'W	1997	Muscle	3			76.3		µg/g ww												0.393									4
Canada, Tunago Lake	66°19'N, 125°50'W	1978	Muscle	2			56.8		µg/g ww												0.375									4
Canada, Wagenitz Lake	63°03'N, 11°52'W	1977	Muscle	10			66.9		µg/g ww												0.277									4
Canada, Willow Lake	62°10'40"N, 119°06'32"W	1999	Muscle	30			63.4		µg/g ww												0.283									4
Finland, Kemijoki River		1994	Muscle	130			52.1	1113	µg/g ww											0.11-1.68	0.47	0.222								1
Finland, Kemijoki River, upstream		1994	Muscle	23			44.4	1157	µg/g ww												0.42	0.416								1
Finland, Kemijoki River, reservoirs		1994	Muscle	107			53.4	1104	µg/g ww												0.49	0.154								1
Finland, Kemijoki River		1994	Muscle	1		11		2450	µg/g ww												1.22									1
Finland, Kemijoki River		1994	Muscle	1		16		7840	µg/g ww												1.60									1
Finland, Kemijoki River		1994	Muscle	1		10		3910	µg/g ww												1.68									1
Finland, Lake Inari	69°00'N, 27°45'E	1997	Liver	1	F				µg/g ww		<0.03					0.24														2
Finland, Lake Inari	69°00'N, 27°45'E	1997	Muscle	1	F		58.0	1321	µg/g ww		0.04					<0.03														2
Finland, Lake Inari	69°00'N, 27°45'E	2000	Muscle	10	1F/9M		50.3	865	µg/g ww												0.18	0.06								2
Finland, Lake Inari	69°00'N, 27°45'E	2000	Muscle	10	1F/9M			1000	µg/g ww												0.19								D	2
Finland, Lake Kemijärvi	66°35'N, 27°25'E	2001	Muscle	5	3F/2M		54.9	949	µg/g ww		0.0009	0.00039				0.0002	0.0001				0.45	0.17				0.17	0.04			2
Finland, Lake Kuusijärvi	65°51'N, 28°10'E	2001	Muscle	10	M		49.2	809	µg/g ww												0.19	0.02								2
Finland, Lake Kuusijärvi	65°51'N, 28°10'E	2001	Muscle	10	M				µg/g ww												0.20								D	2
Finland, Lake Kynsijärvi	65°52'N, 28°20'E	2001	Muscle	10	M				µg/g ww												0.37								D	2
Finland, Lake Kynsijärvi	65°52'N, 28°20'E	2001	Muscle	10	M		54.5	1057	µg/g ww												0.38	0.04								2
Finland, Lake Muddusjärvi	69°00'N, 26°50'E	2000	Muscle	10	8F/2M				µg/g ww												0.28								D	2
Finland, Lake Muddusjärvi	69°00'N, 26°50'E	2000	Muscle	10	8F/2M		61.3	1691	µg/g ww												0.42	0.26								2
Finland, Lake Unari	67°10'N, 25°45'E	2000	Muscle	9	1F/8M		53.5	1024	µg/g ww												0.29	0.08								2
Finland, Lake Unari	67°10'N, 25°45'E	2000	Muscle	9	1F/8M				µg/g ww												0.29								D	2
Finland, Reservoir Lokka	67°55'N, 27°40'E	1997	Muscle	1	F		46.0	681	µg/g ww		<0,03					<0.03														2
Finland, Reservoir Lokka	67°55'N, 27°40'E	2001	Muscle	5	2F/3M		51.0	836	µg/g ww		0.001	0.00022									0.23	0.08				0.09	0.005			2
Finland, River Tornionjoki	65°55'N, 24°10'E	2000	Muscle	10	3F/7M		56.2	1108	µg/g ww												0.38	0.06								2

Table A10. Metals in freshwater fish, continued.

Species/location	Coordinate	Year	Tissue	n	Sex	Age, yrs (SD)	Length, cm (SD)	Weight, g (SD)	Unit	Lead min-max	Lead arithm. mean	Lead arithm. SD	Lead geom. mean	Lead geom. GSD	Cadmium min-max	Cadmium arithm. mean	Cadmium arithm. SD	Cadmium geom. mean	Cadmium geom. GSD	Mercury min-max	Mercury arithm. mean	Mercury arithm. SD	Mercury geom. mean	Mercury geom. GSD	Selenium min-max	Selenium arithm. mean	Selenium arithm. SD	Selenium geom. mean	Re-mark	Refer-ence
Finland, River Tornionjoki	65°55'N, 24°10'E	2000	Muscle	10	3F/7M				µg/g ww												0.38								D	2
Finland, River Tornionjoki	65°55'N, 24°10'E	2001	Muscle	10	1F/9M				µg/g ww												0.39								D	2
Finland, River Tornionjoki	65°55'N, 24°10'E	2001	Muscle	10	1F/9M		56.3	1099	µg/g ww												0.41	0.12							2
Russia, Chukotka, Kanchalan			Liver	4	F	7-9			µg/g ww		0.310					0.060					0.054								3
Russia, Chukotka, Kanchalan			Liver	6	F	9-11			µg/g ww		0.533					0.068					0.190								3
Russia, Chukotka, Kanchalan			Liver	2	F	11-13			µg/g ww		0.536					0.125					0.215								3
Russia, Chukotka, Kanchalan			Muscle	4	F	7-9			µg/g ww		0.125					0.015					0.024								3
Russia, Chukotka, Kanchalan			Muscle	6	F	9-11			µg/g ww		0.196					0.029					0.073								3
Russia, Chukotka, Kanchalan			Muscle	2	F	11-13			µg/g ww		0.250					0.058					0.120								3
Russia, Chukotka, Kanchalan			Liver	5	M	6-9			µg/g ww		0.437					0.068					0.062								3
Russia, Chukotka, Kanchalan			Liver	3	M	9-11			µg/g ww		0.327					0.084					0.082								3
Russia, Chukotka, Kanchalan			Liver	1	M	12-13			µg/g ww		0.505					0.110					0.199								3
Russia, Chukotka, Kanchalan			Muscle	5	M	6-9			µg/g ww		0.124					0.021					0.024								3
Russia, Chukotka, Kanchalan			Muscle	3	M	9-11			µg/g ww		0.219					0.023					0.069								3
Russia, Chukotka, Kanchalan			Muscle	1	M	12-13			µg/g ww		0.196					0.037					0.103								3
Russia, Kola Peninsula			Liver	5	F	7-9			µg/g ww		0.417					0.218					0.264								3
Russia, Kola Peninsula			Liver	7	F	10-13			µg/g ww		0.448					0.171					0.357								3
Russia, Kola Peninsula			Muscle	7	F	10-13			µg/g ww		0.112					0.039					0.135								3
Russia, Kola Peninsula			Muscle	5	F	7-9			µg/g ww		0.093					0.035					0.159								3
Russia, Kola Peninsula			Liver	5	M	7-10			µg/g ww		0.374					0.203					0.267								3
Russia, Kola Peninsula			Liver	3	M	10-13			µg/g ww		0.493					0.192					0.474								3
Russia, Kola Peninsula			Muscle	3	M	10-13			µg/g ww		0.136					0.037					0.173								3
Russia, Kola Peninsula			Muscle	5	M	7-10			µg/g ww		0.122					0.028					0.182								3
Sweden, Storvindeln	65°42'N, 17°8'E	1968-98	Muscle	307					µg/g ww														0.31	0.03					15
Sweden, Storvindeln	65°42'N, 17°8'E	1969-2000	Liver	196					µg/g dw				0.031	0.005															15
Sweden, Storvindeln	65°42'N, 17°8'E	1969-98	Liver	276					µg/g dw									0.049	0.006										15
USA, Alaska, Kuskokwim river		2000	Liver	15					µg/g ww												0.471	0.594							9
USA, Alaska, Kuskokwim river		2000	Muscle	15					µg/g ww												0.628	0.359							9
USA, Alaska, Kuskokwim river			Muscle						µg/g ww											0.129-1.591	0.579	0.371						B	13
USA, Alaska, Yukon river		2000	Liver	6					µg/g ww												1.731	0.861							9
USA, Alaska, Yukon river		2000	Muscle	6					µg/g ww												1.506	0.298							9
USA, Alaska, Yukon river			Muscle						µg/g ww											1.23-2.14	1.56	0.34						B	13
USA, Alaska, Lake 12, North Slope	70°08'N, 151°05'W	2000, 2001	Muscle	13					µg/g ww			0.002					0.001						0.452				0.350			8
USA, Alaska, Lake 13, North Slope	70°15'N, 156°07'W	2000, 2001	Muscle	17					µg/g ww			0.003					0.001						0.234				0.212			8
Leuciscus idus (ide)																														
Russia, Pechora Basin			Liver	2	F	7-8			µg/g ww	0.157					0.168					0.116								3
Russia, Pechora Basin			Liver	5	F	9-11			µg/g ww	0.151					0.216					0.123								3
Russia, Pechora Basin			Muscle	5	F	9-11			µg/g ww	0.032					0.024					0.054								3
Russia, Pechora Basin			Muscle	2	F	7-8			µg/g ww	0.027					0.031					0.061								3
Russia, Pechora Basin			Liver	5	M	9-11			µg/g ww	0.122					0.295					0.119								3
Russia, Pechora Basin			Liver	5	M	6-9			µg/g ww	0.098					0.189					0.135								3
Russia, Pechora Basin			Liver	3	M	11-13			µg/g ww	0.127					0.306					0.142								3
Russia, Pechora Basin			Muscle	5	M	6-9			µg/g ww	0.029					0.018					0.068								3
Russia, Pechora Basin			Muscle	3	M	11-13			µg/g ww	0.034					0.033					0.075								3
Russia, Pechora Basin			Muscle	5	M	9-11			µg/g ww	0.032					0.023					0.082								3
Lota lota (burbot)																														
Canada, Fort Good Hope	66°15'25"N, 128°38'W	1985	Liver	6	F		71.4 (14.0)		µg/g ww												0.097	0.098				1.272	0.715		16
Canada, Fort Good Hope	66°15'25"N, 128°38'W	1985	Muscle	6	F		71.4 (14.0)		µg/g ww												0.337	0.136				0.48	0.126		16
Canada, Fort Good Hope	66°15'25"N, 128°38'W	1985	Liver	9	M		64.3 (8.2)		µg/g ww												0.044	0.019				1.759	0.558		16
Canada, Fort Good Hope	66°15'25"N, 128°38'W	1985	Muscle	10	M		63.3 (8.4)		µg/g ww												0.222	0.035				0.358	0.087		16
Canada, Fort Good Hope	66°15'25"N, 128°38'W	1988	Liver	2	F		62.3 (8.6)		µg/g ww												0.072	0.035				1.460	1.529		16
Canada, Fort Good Hope	66°15'25"N, 128°38'W	1988	Liver	8	M		70.6 (8.4)		µg/g ww												0.054	0.026				1.23	0.555		16
Canada, Fort Good Hope	66°15'25"N, 128°38'W	1993	Liver	3	F		81.2 (12.9)		µg/g ww													---					---		16
Canada, Fort Good Hope	66°15'25"N, 128°38'W	1993	Muscle	3	F		81.2 (13.3)		µg/g ww												0.297	0.035				0.321	0.009		16
Canada, Fort Good Hope	66°15'25"N, 128°38'W	1993	Liver	7	M		67.7 (10.9)		µg/g ww													---					---		16
Canada, Fort Good Hope	66°15'25"N, 128°38'W	1993	Muscle	7	M		67.7 (10.9)		µg/g ww												0.231	0.113				0.534	0.163		16
Canada, Fort Good Hope	66°15'25"N, 128°38'W	1995	Muscle	2	F		---		µg/g ww												0.180	0.085					---		16
Canada, Fort Good Hope	66°15'25"N, 128°38'W	1995	Muscle	2	M		---		µg/g ww												0.265	0.035					---		16
Canada, Fort Good Hope	66°15'25"N, 128°38'W	1999	Liver	20	F		74.9 (7.7)		µg/g ww												0.064	0.069				0.687	0.552		16
Canada, Fort Good Hope	66°15'25"N, 128°38'W	1999	Muscle	21	F		73.5 (10.1)		µg/g ww												0.259	0.108				0.219	0.104		16
Canada, Fort Good Hope	66°15'25"N, 128°38'W	1999	Liver	21	M		67.6 (10.7)		µg/g ww												0.046	0.024				1.071	0.628		16
Canada, Fort Good Hope	66°15'25"N, 128°38'W	1999	Muscle	21	M		67.6 (10.7)		µg/g ww												0.286	0.095				0.395	0.107		16
Canada, Fort Good Hope	66°15'25"N, 128°38'W	2000	Muscle	21	M		69.9 (10.4)		µg/g ww												0.345	0.097				0.478	0.136		16
Canada, Fort Good Hope	66°15'25"N, 128°38'W	Apr-85	Muscle	14			64.4		µg/g ww												0.233	0.054							17
Canada, Fort Good Hope	66°15'25"N, 128°38'W	Dec-88	Liver	7			72.6		µg/g ww												0.051	0.021							18
Canada, Fort Good Hope	66°15'25"N, 128°38'W	Dec-93	Muscle	10			71.8		µg/g ww												0.251	0.099							18
Canada, Fort Good Hope	66°15'25"N, 128°38'W	Sep-95	Muscle	4					µg/g ww												0.223	0.072							18

Table A10. Metals in freshwater fish, continued.

Species/location	Coordinate	Year	Tissue	n	Sex	Age, yrs (SD)	Length, cm (SD)	Weight, g (SD)	Unit	Lead min-max	Lead arithm. mean	Lead SD	Lead geom. mean	Lead GSD	Cadmium min-max	Cadmium arithm. mean	Cadmium SD	Cadmium geom. mean	Cadmium GSD	Mercury min-max	Mercury arithm. mean	Mercury SD	Mercury geom. mean	Mercury GSD	Selenium min-max	Selenium arithm. mean	Selenium SD	Selenium geom. mean	Re-mark	Refer-ence
Canada, Fort Good Hope	66°15'25"N, 128°38'W	Dec-99	Muscle	39			70.8		µg/g ww												0.254	0.075								18
Canada, Fort Good Hope	66°15'25"N, 128°38'W	2000	Liver	15	F		73.2 (12.7)		µg/g ww												0.094	0.056				1.203	0.469			16
Canada, Fort Good Hope	66°15'25"N, 128°38'W	2000	Muscle	15	F		73.2 (12.2)		µg/g ww												0.364	0.140				0.460	0.175			16
Canada, Fort Good Hope	66°15'25"N, 128°38'W	2000	Liver	21	M		69.9 (10.4)		µg/g ww												0.064	0.026				1.646	0.733			16
Canada, Fort Good Hope	66°15'25"N, 128°38'W	1999	Liver	38			71.0		µg/g ww												0.044	0.019								18
Canada, Fort Good Hope	66°15'25"N, 128°38'W	1999	Liver	39			70.9		µg/g ww																	0.879	0.615			18
Canada, Fort Good Hope	66°15'25"N, 128°38'W	1999	Muscle	42			70.5		µg/g ww																	0.312	0.137			18
Canada, Fort Good Hope	66°15'25"N, 128°38'W	1985	Liver	13			65.4		µg/g ww												0.042	0.018								17
Canada, Fort Good Hope	66°15'25"N, 128°38'W	1985	Liver	15			67.2		µg/g ww																	1.564	0.649			17
Canada, Fort Good Hope	66°15'25"N, 128°38'W	1988	Liver	10			68.9		µg/g ww																	1.276	0.713			18
Canada, Fort Good Hope	66°15'25"N, 128°38'W	1985	Muscle	16			66.3		µg/g ww																	0.404	0.116			17
Canada, Great Slave Lake, East Arm	62°24'N, 110°44'W	1999	Muscle						µg/g ww												0.08	0.04				0.31	0.06			14
Canada, Great Slave Lake, West Basin	61°10'N, 113°40'W	1995	Muscle						µg/g ww												0.09	0.04								14
Canada, Great Slave Lake, West Basin	61°10'N, 113°40'W	1996	Muscle						µg/g ww												0.08	0.05								14
Canada, Great Slave Lake, West Basin	61°10'N, 113°40'W	1999	Muscle						µg/g ww												0.11	0.03				0.23	0.03			14
Canada, Lake Laberge	64°11'N, 138°12'W	1996	Liver	15			42.9 (6.4)		µg/g ww												0.10								E	19
Canada, Lake Laberge	65°11'N, 139°12'W	1999	Liver	11			64.8 (7.3)		µg/g ww												0.12								E	19
Canada, Peel River, NWT	67°41'50"N, 134°31'50"W	1999	Liver	10	F/M	12	66.0	2200	µg/g ww		n.d.					0.044	0.036				0.04	0.02								6
Finland, Kemijoki		1994	Muscle	76			43.7	727	µg/g ww											0.07-1.09	0.43	0.21								1
Finland, Kemijoki River, upstream		1994	Muscle	13			30.8	211	µg/g ww											0.07-1.05	0.41	0.276								1
Finland, Kemijoki River, reservoirs		1994	Muscle	63			46.3	833	µg/g ww											0.11-1.09	0.43	0.196								1
Finland, Kemijoki River		1994	Muscle	1		11		240	µg/g ww												1.05									1
Finland, Kemijoki River		1994	Muscle	1		12		2030	µg/g ww												1.09									1
Russia, Kola Peninsula			Liver	7	F	6-9			µg/g ww		0.113					0.213					0.313									3
Russia, Kola Peninsula			Liver	6	F	9-13			µg/g ww		0.248					0.184					0.458									3
Russia, Kola Peninsula			Muscle	6	F	9-13			µg/g ww		0.063					0.032					0.175									3
Russia, Kola Peninsula			Muscle	7	F	6-9			µg/g ww		0.045					0.035					0.178									3
Russia, Kola Peninsula			Liver	2	M	6-8			µg/g ww		0.184					0.174					0.369									3
Russia, Kola Peninsula			Liver	5	M	9-12			µg/g ww		0.152					0.291					0.384									3
Russia, Kola Peninsula			Muscle	5	M	9-12			µg/g ww		0.037					0.045					0.121									3
Russia, Kola Peninsula			Muscle	2	M	6-8			µg/g ww		0.039					0.022					0.158									3
Russia, Taymir Peninsula, Dudinka			Liver	5	F	9-11			µg/g ww		0.244					0.184					0.165									3
Russia, Taymir Peninsula, Dudinka			Liver	2	F	7-9			µg/g ww		0.159					0.249					0.191									3
Russia, Taymir Peninsula, Dudinka			Liver	6	F	11-13			µg/g ww		0.235					0.267					0.235									3
Russia, Taymir Peninsula, Dudinka			Muscle	5	F	9-11			µg/g ww		0.105					0.061					0.096									3
Russia, Taymir Peninsula, Dudinka			Muscle	2	F	7-9			µg/g ww		0.102					0.016					0.123									3
Russia, Taymir Peninsula, Dudinka			Muscle	6	F	11-13			µg/g ww		0.124					0.054					0.126									3
Russia, Taymir Peninsula, Dudinka			Liver	5	M	6-9			µg/g ww		0.218					0.189					0.157									3
Russia, Taymir Peninsula, Dudinka			Liver	1	M	10-11			µg/g ww		0.371					0.131					0.176									3
Russia, Taymir Peninsula, Dudinka			Liver	1	M	13-14			µg/g ww		0.362					0.178					0.203									3
Russia, Taymir Peninsula, Dudinka			Muscle	5	M	6-9			µg/g ww		0.268					0.021					0.093									3
Russia, Taymir Peninsula, Dudinka			Muscle	1	M	10-11			µg/g ww		0.154					0.034					0.102									3
Russia, Taymir Peninsula, Dudinka			Muscle	1	M	13-14			µg/g ww		0.227					0.027					0.103									3
Russia, Taymir Peninsula, Khatanga			Liver	2	F	7-9			µg/g ww		0.033					0.081					0.143									3
Russia, Taymir Peninsula, Khatanga			Liver	5	F	9-12			µg/g ww		0.149					0.136					0.153									3
Russia, Taymir Peninsula, Khatanga			Muscle	5	F	9-12			µg/g ww		0.04					0.031					0.071									3
Russia, Taymir Peninsula, Khatanga			Muscle	2	F	7-9			µg/g ww		0.081					0.115					0.112									3
Russia, Taymir Peninsula, Khatanga			Liver	3	M	9-12			µg/g ww		0.174					0.103					0.183									3
Russia, Taymir Peninsula, Khatanga			Liver	10	M	6-8			µg/g ww		0.067					0.095					0.244									3
Russia, Taymir Peninsula, Khatanga			Muscle	10	M	6-8			µg/g ww		0.01					0.030					0.088									3
Russia, Taymir Peninsula, Khatanga			Muscle	3	M	9-12			µg/g ww		0.035					0.043					0.089									3
Perca fluviatilis (perch)																														
Finland, Kemijoki River		1994	Muscle	104			22.8	172	µg/g ww											0.03-1.56	0.32	0.254								1
Finland, Lake Inari	69°00'N, 27°45'E	1997	Muscle	1	M		27.0	267	µg/g ww	<0.03					<0.03															2
Finland, Lake Inari	69°00'N, 27°45'E	2000	Muscle	2	M			89	µg/g ww												0.1									2
Finland, Lake Inari	69°00'N, 27°45'E			3	M			104	µg/g ww												0.13									2
Finland, Lake Inari	69°00'N, 27°45'E			5	F/M			158	µg/g ww												0.19									2
Finland, Lake Kuusijärvi	65°51'N, 28°10'E	2001		2	F			137	µg/g ww												0.21									2
Finland, Lake Kuusijärvi	65°51'N, 28°10'E	2001	Muscle	8	M			74	µg/g ww												0.16									2
Finland, Lake Kuusijärvi	65°52'N, 28°20'E	2001	Muscle	7	F/M			76	µg/g ww												0.2									2
Finland, Lake Kuusijärvi	65°52'N, 28°20'E	2001		3	F/M			117	µg/g ww												0.14									2
Finland, Lake Muddusjärvi	69°00'N, 26°50'E	2000	Muscle	3	M			89	µg/g ww												0.15									2
Finland, Lake Muddusjärvi	69°00'N, 26°50'E			6	M			110	µg/g ww												0.14									2
Finland, Lake Muddusjärvi	69°00'N, 26°50'E			1	M			142	µg/g ww												0.29									2
Finland, Lake Unari	67°10'N, 25°45'E	2000	Muscle	8	F/M			139	µg/g ww												0.37									2

Table A10. Metals in freshwater fish, continued.

Species/location	Coordinate	Year	Tissue	n	Sex	Age, yrs (SD)	Length, cm (SD)	Weight, g (SD)	Unit	Lead min-max	Lead arithm. mean	Lead arithm. SD	Lead geom. mean	Lead geom. GSD	Cadmium min-max	Cadmium arithm. mean	Cadmium arithm. SD	Cadmium geom. mean	Cadmium geom. GSD	Mercury min-max	Mercury arithm. mean	Mercury arithm. SD	Mercury geom. mean	Mercury geom. GSD	Selenium min-max	Selenium arithm. mean	Selenium arithm. SD	Selenium geom. mean	Remark	Reference
Finland, Lake Unari	67°10'N, 25°45'E			2	F/M			220	µg/g ww												0.35									2
Finland, Reservoir Lokka	67°55'N, 27°40'E		Muscle	1	F		26.0	179	µg/g ww		<0.03					<0.03														2
Norway, Lake Gavdujav´ri (Finnmark)	68°742'N, 24°632'E	1992/95	Muscle	20		10.4	22.0	189	µg/g ww												0.189	0.076								20
Russia, Pechora Basin			Liver	9	F	5-7			µg/g ww		0.438					0.128					0.172									3
Russia, Pechora Basin			Liver	6	F	9-12			µg/g ww		0.576					0.165					0.204									3
Russia, Pechora Basin			Liver	5	F	7-9			µg/g ww		0.374					0.143					0.215									3
Russia, Pechora Basin			Muscle	9	F	5-7			µg/g ww		0.149					0.032					0.072									3
Russia, Pechora Basin			Muscle	5	F	7-9			µg/g ww		0.168					0.030					0.094									3
Russia, Pechora Basin			Muscle	6	F	9-12			µg/g ww		0.170					0.041					0.129									3
Prosopium cylindraceum (round whitefish)																														
Canada, Giauque Lake	63°11'N, 113°51'W	1977	Muscle	35			46.7		µg/g ww												1.217									4
Canada, Great Slave Lake	62°50'N, 113°50'W	1977	Muscle	14			32.6		µg/g ww												0.034									4
Canada, Mc Crea Lake, NWT	63°33'N, 112°35'W	1977	Muscle	1			55.0		µg/g ww												0.53									4
Canada, Thistlethwaite Lake	63°10'N, 113°34'W	1977	Muscle	9			42.9		µg/g ww												0.179									4
Canada, Wagenitz Lake	63°03'N, 11°52'W	1977	Muscle	1			47.0		µg/g ww												0.08									4
Finland, Lake Inari	69°00'N, 27°45'E	2001	Muscle	5	2F/3M		35.9	402	µg/g ww		0.01	0.01				0.001	0.0003				0.08	0.03				0.28	0.05			2
Finland, Lake Kilpisjärvi	69°02'N, 20°50'E	2001	Muscle	5	3F/2M		27.0	158	µg/g ww		0.001	0.0004				0.002	0.001				0.08	0.01				0.38	0.03			2
Finland, Lake Muddusjärvi	69°00'N, 26°50'E	2000	Muscle	6	F/M			164	µg/g ww												0.07									2
Finland, Lake Muddusjärvi	69°00'N, 26°50'E	2000	Muscle	4	F/M			222	µg/g ww												0.08									2
Finland, Lake Nitsijärvi	69°15'N, 28°00'E	2001	Muscle	5	1F/4M		34.7	424	µg/g ww		0.003	0.001				0.002	0.001				0.10	0.03				0.51	0.09			2
Finland, Reservoir Lokka	67°55'N, 27°40'E	2001	Muscle	10	5F/5M		31.0	272	µg/g ww		0.002	0.002				0.0001	0.0001				0.08	0.01				0.10	0.02			2
Finland, River Tornionjoki	65°55'N, 24°10'E	2001	Muscle	5	1F/4M		33.3	307	µg/g ww		0.004	0.001				0.001	0.0002				0.06	0.03				0.31	0.09			2
Norway, Lake S. Galdinjav´ri (Finnmark)	68°850'N, 23°434'E	1992/95	Muscle	20		8.0	22.2	146	µg/g ww												0.079	0.021								20
Rutilus rutilus (roach)																														
Finland, Kemijoki River, upstream		1994	Muscle	1			21.8	141	µg/g ww												0.19									1
Finland, Kemijoki River, reservoirs		1994	Muscle	19			21.1	93	µg/g ww												0.33	0.165								1
Salmo trutta (brown trout)																														
Faroe Islands, Fjallavatn	62°07'N, 07°17'W	aug/sep 97	Muscle	8		4+	23.2	108	µg/g ww												0.19									21
Faroe Islands, Fjallavatn	62°07'N, 07°17'W	aug/sep 97	Muscle	11		4+	23.4	112	µg/g ww												0.26									21
Faroe Islands, Fjallavatn	62°07'N, 07°17'W	aug/sep 97	Muscle	4		5+	28.9	210	µg/g ww												0.34									21
Faroe Islands, Fjallavatn	62°07'N, 07°17'W	aug/sep 97	Muscle	5		5+	28.5	196	µg/g ww												0.38									21
Faroe Islands, Leitisvatn	52°03'N, 07°14'W	aug/sep 97	Muscle	11		4+	23.4	125	µg/g ww												0.29									21
Faroe Islands, Leitisvatn	52°03'N, 07°14'W	aug/sep 97	Muscle	5		4+	23.3	121	µg/g ww												0.30									21
Faroe Islands, Leitisvatn	52°03'N, 07°14'W	aug/sep 97	Muscle	3		5+	27.4	204	µg/g ww												0.31									21
Faroe Islands, Leitisvatn	52°03'N, 07°14'W	aug/sep 97	Muscle	4		5+	27.3	199	µg/g ww												0.38									21
Salvelinus alpinus (Arctic char)																														
Canada, Amituk Lake	75°28'48"N, 93°49'12"W	1989, 92	Muscle	54			5.4		µg/g ww												0.698									4
Canada, Baker foreland	62°55'N, 90°48'W	1989	Muscle	5			61.8		µg/g ww												0.05									4
Canada, Baker Lake, YT	60°12'10"N, 128°27'30"W	1995	Muscle	6			60.1		µg/g ww												0.047									4
Canada, Boomerang L., Nunavut	n.a.	1999	Muscle	11			49.6	950.0	µg/g ww	0.004-0.013	0.008	0.003			0.001-0.002	0.001	0.000			0.104-0.521	0.314	0.189			0.620-0.940	0.765	0.104			22,23
Canada, Buchanan Lake	79°28'N, 87°40'W	1991	Muscle	19			51.8		µg/g ww												0.037									4
Canada, Byron Bay	68°55'N, 108°30'W	1984, 88, 90 89, 90	Muscle	19			67.7		µg/g ww												0.035									4
Canada, Byron Bay, Lauchlin River	68°55'N, 108°30'W	1993	Muscle	5			71.9		µg/g ww												0.024									4
Canada, Cambridge Bay	69°03'N, 105°07'W	1977, 78, 92, 93	Muscle	26			51.4		µg/g ww												0.048									4
Canada, Char Lake, Nunavut	74°42'N, 94°53'W	1992	Muscle	7					µg/g ww																					4,24
Canada, Char Lake, Nunavut	74°42'N, 94°53'W	1999	Muscle	3			28.3	200.6	µg/g ww	0.006-0.009	0.007	0.002			0.001-0.003	0.002	0.001			0.09-0.305	0.215	0.114			0.440-0.560	0.487	0.064			22,23
Canada, Char Lake, Nunavut	74°42'N, 94°53'W	2000	Muscle	4			37.3	338.6	µg/g ww		0.018	0.010				0.001	0.001				0.469	0.178				0.727	0.091			23
Canada, Char Lake, Nunavut	74°42'N, 94°53'W	2001	Muscle	4			36.4	595.5	µg/g ww		0.002	0.002				0.001	0.002				0.409	0.179				0.430	0.116			23
Canada, Char Lake, Nunavut	74°42'N, 94°53'W	1993 or 94	Muscle	7					µg/g ww												0.26									4
Canada, Chesterfield Inlet	63°43'N, 92°00'W	1977, 84, 89, 92, 94	Muscle	36			59.5		µg/g ww												0.041									4
Canada, Corbet Inlet	62°28'N, 92°20'W	1989	Muscle	5			64.5		µg/g ww												0.058									4
Canada, Dease Strait	68°50'N, 107°30'W	1977	Muscle	5			59.2		µg/g ww												0.068									4
Canada, Ekalluk River	69°24'N, 106°18'W	1990	Muscle	5			61.7		µg/g ww												0.026									4
Canada, Ellice River	68°02'N, 103°58'W	1977, 84, 88, 89, 90, 93, 94	Muscle	33			62.3		µg/g ww												0.078									4
Canada, Ferguson River	62°03'N, 93°20'W	1992	Muscle	10			64.1		µg/g ww												0.037									4
Canada, Ferguson River	62°03'N, 93°20'W	1994	Muscle						µg/g ww																					4
Canada, Foggy Bay	68°17'N, 104°47'W	1993	Muscle	5			64.0		µg/g ww												0.08									4
Canada, Gore Bay	66°19'N, 84°24'W	1993	Muscle	5			63.4		µg/g ww												0.024									4
Canada, Hall	68°41'N, 82°17'W	1978	Muscle	25			64.2		µg/g ww												0.038									4
Canada, Hall Beach	68°47'N, 81°13'W	1992	Muscle	5			69.8		µg/g ww												0.054									4
Canada, Hazen Lake	81°47'50"N, 71 01'00"W	1990	Muscle	45			40.4		µg/g ww												0.181									4

Table A10. Metals in freshwater fish, continued.

Species/location	Coordinate	Year	Tissue	n	Sex	Age, yrs (SD)	Length, cm (SD)	Weight, g (SD)	Unit	Lead min-max	Lead arithm. mean	Lead SD	Lead geom. mean	Lead GSD	Cadmium min-max	Cadmium arithm. mean	Cadmium SD	Cadmium geom. mean	Cadmium GSD	Mercury min-max	Mercury arithm. mean	Mercury SD	Mercury geom. mean	Mercury GSD	Selenium min-max	Selenium arithm. mean	Selenium SD	Selenium geom. mean	Re-mark	Refer-ence
Canada, Jayco River	69°34'12"N, 103°21'W	1979, 84, 89, 90, 93, 94	Muscle	31			64.9		µg/g ww												0.078									4
Canada, Kaminuriak Lake	62°30'N, 95°46'12"W	1975	Muscle	1			64.0		µg/g ww												0.02									4
Canada, Lake 1, Ungave Peninsula	62°22'N, 73°42'W	1999	Liver	5	F		16.7		µg/g dw		0.026	0.021				0.58	0.39												25
Canada, Lake 1, Ungave Peninsula	62°22'N, 73°42'W	1999	Muscle	5	F		16.7	35	µg/g ww												0.09									25
Canada, Lake 2, Melville Peninsula	67°33'N, 81°41'W	1999	Liver	5	F		16.4		µg/g dw		0.020	0.014				0.72	0.21												25
Canada, Lake 2, Melville Peninsula	67°33'N, 81°41'W	1999	Muscle	5	F		16.4	30	µg/g ww												0.07	0.01								25
Canada, Lake 20, Ivvavik NP, Yukon	69°26'N, 139°36'W	1999	Liver	5	F		28.7		µg/g dw		0.006	0.003				8.18	1.58												25
Canada, Lake 20, Ivvavik NP, Yukon	69°26'N, 139°36'W	1999	Muscle	5	F		28.7	169	µg/g ww												0.10	0.09								25
Canada, Lake 23, N Banks Island	73°40'N, 116°12'W	1999	Liver	5	F		17.8		µg/g dw		0.064	0.037				0.86	0.21												25
Canada, Lake 23, N Banks Island	73°40'N, 116°12'W	1999	Muscle	5	F		17.8	46	µg/g ww												0.06	0.04								25
Canada, Lake 26, S Ellesmere Island	76°28'N, 86°50'W	1999	Liver	6	F		16.0		µg/g dw		0.020	0.026				0.28	0.062												25
Canada, Lake 26, S Ellesmere Island	76°28'N, 86°50'W	1999	Muscle	6	F		16.0	35.0	µg/g ww												0.07	0.03								25
Canada, Lake 29, E Baffin Island	68°26'N, 66°50'W	1999	Liver	5	F		25.0		µg/g dw		0.027	0.011				1.89	0.97												25
Canada, Lake 29, E Baffin Island	68°26'N, 66°50'W	1999	Muscle	5	F		25.0	136.0	µg/g ww												0.07	0.03								25
Canada, Nettilling	66°28'48"N, 70°19'48"W	1990	Muscle	5			56.1		µg/g ww												0.084									4
Canada, unnamed	63°39'N, 90°39'W	1993 or 94	Muscle	3					µg/g ww												0.263									4
Canada, North Lake, Nunavut	74°46'N, 95°06' W	2000	Muscle	14			36(6)	382(143)	µg/g ww		0.026	0.036				0.002	0.001				0.202	0.115				0.374	0.044			26
Canada, P&N Lake	63°39'N, 90°39'W	1988	Muscle	15			36.8		µg/g ww												0.085									4
Canada, Pangnirtung, (Fjord)	66°07'25"N, 65°37'35"W	1990, 1992	Muscle	23			49.0		µg/g ww												0.033									4
Canada, Paulatuk	69°21'N, 124°04'W	1984	Muscle	6			54.7		µg/g ww												0.042									4
Canada, Pistol Bay	62°25'N, 92°40'30"W	1988	Muscle	6			70.0		µg/g ww												0.033									4
Canada, Rankin Inlet	62°44'N, 91°59'W	1984, 92, 93	Muscle	21			55.1		µg/g ww												0.058									4
Canada, Resolute Lake, Nunavut	74°41'N, 94°57'W	1992	Muscle	7					µg/g ww																	0.200				4,24
Canada, Resolute Lake, Nunavut	74°41'N, 94°57'W	1997	Muscle	10			32.8	305.3	µg/g ww	0.002-0.020	0.008	0.005			0.001-0.019	0.005	0.007			0.102-0.426	0.200	0.103			0.510-0.780	0.649	0.092			22,23
Canada, Resolute Lake, Nunavut	74°41'N, 94°57'W	1999	Muscle	10			39.6	506.6	µg/g ww	0.003-0.015	0.008	0.004			0.001-0.002	0.002	0.001			0.07-0.262	0.172	0.067			0.660-0.810	0.728	0.040			22,23
Canada, Resolute Lake, Nunavut	74°41'N, 94°57'W	2000	Muscle	18			39(2)	433(78)	µg/g ww		0.030	0.020				0.002	0.001				0.162	0.056				0.684	0.049			26
Canada, Resolute Lake, Nunavut	74°41'N, 94°57'W	2001	Muscle	17			39.3	466.8	µg/g ww		0.129	0.486				0.001	0.001				0.159	0.109				0.711	0.123			23
Canada, Resolute Lake, Nunavut	74°41'N, 94°57'W	1993 or 94	Muscle	7					µg/g ww												0.198									4
Canada, Sandy Point	60°26'N, 109°51'W	1992, 93	Muscle	10			58.8		µg/g ww												0.055									4
Canada, Sapphire Lake, Nunavut	n.a.	1999	Muscle	8			40.2	864.0	µg/g ww	0.003-0.012	0.007	0.003			0.001-0.005	0.002	0.001			0.116-0.577	0.291	0.189			0.610-0.740	0.670	0.053			22,23
Canada, Saputing Lake	70°42'N, 85°25'W	1979	Muscle	5			73.3		µg/g ww												0.052									4
Canada, Small Lake	74°45'N, 95°04'W	1993 or 94	Muscle	2					µg/g ww												0.054									4
Canada, Sophia Lake	62°55'40"N, 114°08'45"W	1989	Muscle	3			63.7		µg/g ww												0.204									4
Canada, Stony Point area	63°51'N, 92°48'W	1988	Muscle	5			63.6		µg/g ww												0.03									4
Canada, Surrey	69°40'N, 107°13'W	1984	Muscle	5			65.4		µg/g ww												0.036									4
Canada, Surrey River/ Palikyuak River	69°27'N, 107°13'12"W	1989, 90, 93	Muscle	15			68.4		µg/g ww												0.033									4
Canada, Sylvia Grinell River	63°43'48"N, 68°34'12"W	1992	Muscle	5			30.2		µg/g ww												0.082									4
Canada, Tagniuknitak Lake	69°15'N, 102°4'48"W	1995	Muscle	5			63.0		µg/g ww												0.062									4
Canada, Tessikakjuak Lake	64°18'N, 76°46'W	1977	Muscle	5			37.9		µg/g ww												0.014									4
Canada, Thirty mile Lake	61°32'N, 135°08'W	1988, 89, 90, 94	Muscle	20			65.5		µg/g ww												0.045									4
Canada, Thirty mile River/Halovik	61°32'N, 135°08'W	1993	Muscle	5			73.8		µg/g ww												0.024									4
Canada, Tree River		1977	Muscle	8			66.0		µg/g ww												0.02									4
Canada, Victory Lake	62°40'N, 113°05'W	1993 or 94	Muscle	1					µg/g ww												0.421									4
Canada, Wellington Bay	69°20'N, 106°35'W	1984, 88, 89	Muscle	14			63.9		µg/g ww												0.041									4
Canada, Wilson River	62°19'N, 93°03'W	1988, 93	Muscle	10			60.4		µg/g ww												0.051									4
Faroe Islands, Myranar		28 Jun 2000	Muscle	8	F	8.5	36.7	596.5	µg/g ww												0.17	0.04				1.45	0.12			27
Faroe Islands, Myranar		28 Jun 2000	Muscle	17	M	8.0	38.1	650.24	µg/g ww												0.17	0.06				1.45	0.16			27
Faroe Islands, Myranar		Jun/Jul 2001	Muscle	40	F/M	7.7	36.2	505.25	µg/g ww												0.22					1.836			F	27
Faroe Islands, Leynavatn		May/Jun '98	Muscle	6	F	7.5	23.0		µg/g ww												0.16									21
Faroe Islands, Leynavatn		May/Jun '98	Muscle	5	F	8.6	23.3		µg/g ww												0.20									21
Faroe Islands, Leynavatn		May/Jun '98	Muscle	4	M	6.5	21.3		µg/g ww												0.19									21
Finland, Lake Pahtajärvi	68°10'N, 24°00'E	1999	Muscle	5	F		30.4	300	µg/g ww	<0.03					<0.03						0.06	0.01			0.71	0.11			2
Finland, Lake Pahtajärvi	68°10'N, 24°00'E	1999	Muscle	5	M		34.6	389	µg/g ww	<0.03					<0.03						0.09	0.07			0.81	0.06			2
Finland, Lake Pahtajärvi	69°02'N, 20°50'E	1999	Muscle	5			21.0	82	µg/g ww	0.001	0.0004				0.001	0.0006				0.062	0.03			0.35	0.01			28
Greenland, Isortoq	60°59'N, 47°30'W	1994	Muscle	27			40.0		µg/g ww												0.816	0.521						G	29
Greenland, Isortoq	60°59'N, 47°30'W	1995	Muscle	27			40.0		µg/g ww												0.655	0.418						G	29
Greenland, Isortoq	60°59'N, 47°30'W	1999	Muscle	20			40.0		µg/g ww												0.761	0.486						G	30
Greenland, Ittoqqortootmiit		2000	Muscle	20			40.0		µg/g ww												0.074	0.058			0.856	0.621		G	30
Greenland, Zackenberg		1999	Muscle	18			40.0		µg/g ww												0.135	0.108						G	30
Greenland, Zackenberg		1999	Muscle	20			40.0		µg/g ww							0.002	0.001								1.3	0.89		G	30
Iceland, Lake Thingvallavatn, Vatnskot		Dec 1995	Liver						µg/g ww		0.050	0.006				0.059	0.005								3.33	0.10			31
Iceland, Lake Thingvallavatn, Vatnskot		Dec 1995	Liver						µg/g ww		0.020	0.008				0.076	0.018								3.56	1.37			31
Iceland, Lake Thingvallavatn, Vatnskot		Dec 1995	Liver						µg/g ww		0.160	0.025				0.115	0.005								4.00	0.18			31
Iceland, Lake Thingvallavatn, Vatnskot		Dec 1995	Muscle	22		2.4			µg/g ww												0.013	0.003							31
Iceland, Lake Thingvallavatn, Vatnskot		Dec 1995	Muscle	10		4			µg/g ww												0.026	0.001							31
Iceland, Lake Thingvallavatn, Vatnskot		Dec 1995	Muscle	10		6.8			µg/g ww												0.031	0.002							31
Iceland, Lake Thingvallavatn, Vatnskot		May 1996	Liver	14		3.7			µg/g ww		0.040	0.006				<0.065									2.71	0.24			31

Table A10. Metals in freshwater fish, continued.

Species/location	Coordinate	Year	Tissue	n	Sex	Age, yrs (SD)	Length, cm (SD)	Weight, g (SD)	Unit	Lead min-max	Lead arithm. mean	Lead SD	Lead geom. mean	Lead GSD	Cadmium min-max	Cadmium arithm. mean	Cadmium SD	Cadmium geom. mean	Cadmium GSD	Mercury min-max	Mercury arithm. mean	Mercury SD	Mercury geom. mean	Mercury GSD	Selenium min-max	Selenium arithm. mean	Selenium SD	Selenium geom. mean	Re-mark	Refer-ence
Iceland, Lake Thingvallavatn, Vatnskot		May 1996	Liver						µg/g ww		0.116	0.046				0.065										1.78	0.06			31
Iceland, Lake Thingvallavatn, Vatnskot		May 1996	Muscle	6		6.7			µg/g ww												0.019	0.003								31
Iceland, Lake Thingvallavatn, Vatnskot		May/Jul '00	Liver						µg/g ww		<0.020					0.108	0.001									3.9	0.1			31
Iceland, Lake Thingvallavatn, Vatnskot		May/Jul '00	Liver						µg/g ww		<0.020					0.049										4.9				31
Iceland, Lake Thingvallavatn, Vatnskot		May/Jul '00	Muscle	9		3.8			µg/g ww												0.013	0.001								31
Iceland, Lake Thingvallavatn, Vatnskot		May/Jul '00	Muscle	7		8.9			µg/g ww												0.018	0.001								31
Norway, Store Raudvatnet	66°16'41"N, 14°31'1"E	19995-1999	Muscle	12			24.5 (6.2)	174 (196)	µg/g ww												0.061									32
Norway, Fjellfrösvatnet	69°5'10"N, 19°20'2"E	1995-1999	Muscle	20			27.2 (3.6)	206 (88)	µg/g ww												0.035									32
Norway, Takvatnet	69°6'54", 19°5'17"E	1995-1999	Muscle	19			25.2 (4.6)	150 (104)	µg/g ww												0.037									32
Norway, Storvatnet	71°2'12"N, 27°55'37"E	1995-1999	Muscle	20			21.0 (2.6)	82 (39)	µg/g ww												0.075									32
Norway, Björnöya, Ellasjön	74°23'35"N, 19°2'24"E	2001-2003	Muscle	22			34.0 (18.6)		µg/g ww												0.218	0.159								33
Norway, Björnöya, Öyangen	74°26'49"N, 19°0'25"E	2001-2003	Muscle	21			29.4 (17.7)		µg/g ww												0.109	0.068								33
Norway, Björnöya, Ellasjön	74°23'35"N, 19°2'24"E	2001-2003	Muscle	22			34.0 (18.6)		µg/g ww												0.171	0.128							B	33
Norway, Björnöya, Öyangen	74°26'49"N, 19°0'25"E	2001-2003	Muscle	21			29.4 (17.7)		µg/g ww												0.09	0.06							B	33
Russia, Chukotka (Lavrentiya)			Liver	8	F	3-7			µg/g ww		0.388					0.550					0.470									3
Russia, Chukotka (Lavrentiya)			Muscle	8	F	3-7			µg/g ww		0.071					0.149					0.151									3
Russia, Chukotka (Lavrentiya)			Liver	12	M	3-6			µg/g ww		0.316					0.600					0.610									3
Russia, Chukotka (Lavrentiya)			Muscle	12	M	3-6			µg/g ww		0.067					0.124					0.285									3
Sweden, Lake Abiskojaure	68°18'N, 18°39'E	1981-2000	Liver	117					µg/g dw				0.028																H	15
Sweden, Lake Abiskojaure	68°18'N, 18°39'E	1981-2000	Liver	190					µg/g dw									1.0											H	15
Sweden, Lake Abiskojaure	68°18'N, 18°39'E	1981-97	Muscle	170					µg/g ww														0.031						H	15
Sweden, Lake Tjulträsk	65°58'N, 16°4'E	1995-96	Liver	20					µg/g dw				0.045																H	15
Sweden, Lake Tjulträsk	65°58'N, 16°4'E	1995-96	Liver	20					µg/g dw									0.31											H	15
Sweden, Lake Tjulträsk	65°58'N, 16°4'E	1995-96	Muscle	20					µg/g ww														0.128						H	15
Salvelinus namaycush (lake trout)																														
Canada, Atlin Lake	60°00'N, 133°49'30"W	1993, 98	Muscle	15			57.7		µg/g ww												0.218									4
Canada, Aubry Lake	67°23'52"N, 126°26'44"W	1999	Muscle	88			57.6		µg/g ww												0.254									4
Canada, Baker Lake, YT	60°12'10"N, 128°27'30"W	1995	Muscle	5			64.4		µg/g ww												0.365									4
Canada, Baker Lake, YT	61°00'N, 134°04'W	1998	Muscle	5					µg/g ww	<0.019					<0.002						0.157	0.035								10
Canada, Basler Lake	63°57'N, 115°58'W	1981, 82, 83	Muscle	31			59.7		µg/g ww												0.434									4
Canada, Belot Lake	66°53'50"N, 126°17'33"W	1993, 99	Muscle	73			61.8		µg/g ww												0.192									4
Canada, Carcajou Lake	64°41'N, 127°54'W	1978	Muscle	2			52.8		µg/g ww												0.555									4
Canada, Cli Lake	61°59'N, 123°18'W	1983, 96, 2000	Muscle	59			48.1		µg/g ww												0.79									4
Canada, Coal Lake	60°29'35"N, 135°09'35"W	1999	Muscle	16			34.4		µg/g ww												0.072									4
Canada, Colville Lake	67°09'32"N, 125°58'34"W	1993, 99	Muscle	250			51.2		µg/g ww												0.204									4
Canada, Dease Strait	68°50'N,107°30'W	1977	Muscle	7			51.1		µg/g ww												0.297									4
Canada, Discovery Lake	62°53'30"N,114°19'15"W	1983	Muscle	7			44.3		µg/g ww												2.293									4
Canada, Dubawnt, NWT	63°03'45"N,102°05'45"W	1971	Muscle	28			52.0		µg/g ww												0.415									4
Canada, Ellice River	68°02'N,103°58'W	1977	Muscle	4			52.5		µg/g ww												0.13									4
Canada, Ferguson Lake, Nunavut	69°25'N,105°15'W	1971	Muscle	10			51.3		µg/g ww												1.037									4
Canada, Fox Lake	61°13'48"N, 135°28'12"W	1998	Muscle	2			40.0		µg/g ww												0.397									4
Canada, Garry Lake	65°58'N, 100°18'W	1976	Muscle	8			53.3		µg/g ww												0.755									4
Canada, Giauque Lake	63°11'N, 113°51'W	1977, 82, 92	Muscle	63			55.2		µg/g ww												3.06									4
Canada, Gray Lake	61°52'N, 108°15'W	1986	Muscle	12			50.1		µg/g ww												0.531									4
Canada, Great Bear Lake	65°50'N, 120°45'W	1978, 79	Muscle	30			60.9		µg/g ww												0.183									4
Canada, Great Bear Lake	65°50'N, 120°45'W	2001	Muscle	10			73.69 (12.3)	4086.36 (1587.7)	µg/g ww												0.350	0.309								4
Canada, Great Slave Lake	62°50'N, 113°50'W	1979, 88, 89, 95	Muscle	74			58.4		µg/g ww												0.128									4
Canada, Great Slave Lake, East Arm	62°24'N, 110°44'W	1995	Muscle						µg/g ww												0.13	0.06								14
Canada, Great Slave Lake, East Arm	62°24'N, 110°44'W	1999	Muscle						µg/g ww												0.14	0.52				0.29	0.10			14
Canada, Great Slave Lake, West Basin	61°10'N, 113°40'W	1999	Muscle						µg/g ww												0.11	0.02				0.29	0.02			14
Canada, Hall Lake	68°41'N,82°17'W	1977, 78	Muscle	25			67.0		µg/g ww												0.707									4
Canada, Hawk Lake	66°10'N,94°25'W	1988	Muscle	9			49.5		µg/g ww												0.238									4
Canada, Hidden Lake	66°00'N,117°51'W	1978	Muscle	6			46.2		µg/g ww												0.323									4
Canada, Hjalmar Lake	61°33'N,109°25'W	1986	Muscle	11			67.5		µg/g ww												0.759									4
Canada, Hottah Lake	65°04'N,118°30'W	1972	Muscle	5			58.8		µg/g ww												0.236									4
Canada, Itsi Lake	62°50'N,130°12'W	2001	Muscle	5			0.0		µg/g ww												0.086									4
Canada, Jayco River	n.a.	1979	Muscle	3			83.6		µg/g ww												0.517									4

Table A10. Metals in freshwater fish, continued.

Species/location	Coordinate	Year	Tissue	n	Sex	Age, yrs (SD)	Length, cm (SD)	Weight, g (SD)	Unit	Lead min-max	Lead arithm. mean	Lead SD	Lead geom. mean	Lead GSD	Cadmium min-max	Cadmium arithm. mean	Cadmium SD	Cadmium geom. mean	Cadmium GSD	Mercury min-max	Mercury arithm. mean	Mercury SD	Mercury geom. mean	Mercury GSD	Selenium min-max	Selenium arithm. mean	Selenium SD	Selenium geom. mean	Re-mark	Refer-ence
Canada, Kaminak Lake	61°10'N,95°00'W	1971, 95 or 96	Muscle	25			59.6		µg/g ww												0.955									4
Canada, Kaminuriak Lake	n.a.	1971, 72, 75	Muscle	77			62.0		µg/g ww												0.651									4
Canada, Kelly Lake	65°23'51"N, 126 09'39"W	1998	Muscle	31			59.8		µg/g ww												0.482									4
Canada, Koksoak River,	58°32'N, 68°10'W		Muscle	8			0.0		µg/g ww												0.368									4
Canada, Kusawa	60°21'N, 136°22'W	1999	Muscle	14			51.5		µg/g ww												0.55								E	19
Canada, Kusawa	60°21'N, 136°22'W	1999	Muscle	14			51.5 (10.6)		µg/g ww												0.55									19
Canada, Lake Laberge	61°11'N, 135°12'W	1993, 96, 98, 99	Muscle	39			51.8		µg/g ww												0.413									4
Canada, Lake Laberge	61°11'N, 135°12'W	1993	Muscle	12			48.3 (11.1)		µg/g												0.49								E	19
Canada, Lake Laberge	62°11'N, 136°12'W	1996	Muscle	14			48.9 (9.5)		µg/g												0.35								E	19
Canada, Lake Laberge	63°11'N, 137°12'W	1998	Muscle	7			70 (12.5)		µg/g												0.38								E	19
Canada, Little Doctor Lake	61°53'N, 123°16'W	1996	Muscle	10			54.7		µg/g ww												0.393									4
Canada, Loon Lake	66°36'30"N, 128°43'15W	1997	Muscle	6			60.0		µg/g ww												0.369									4
Canada, Maguire Lake	63°13'N, 113°54'W	1977	Muscle	34			47.9		µg/g ww												0.461									4
Canada, Maguse Lake	61°37'N, 95°10'W	1995	Muscle	13			54.3		µg/g ww												0.729									4
Canada, Mahoney Lake	65°30'51"N, 125°21'30"W	1996	Muscle	20			67.2		µg/g ww												0.367									4
Canada, Manuel Lake	66°58'N, 128°54'W	1997	Muscle	19			48.5		µg/g ww												0.299									4
Canada, Mattberry Lake	64°05'W, 115°54'W	1981	Muscle	11			54.9		µg/g ww												0.399									4
Canada, Mc Crea Lake, NWT	63°33'N, 112°35'W	1977	Muscle	17			48.2		µg/g ww												0.588									4
Canada, Mirror Lake	64°15'N, 126°55'W	2000	Muscle	63			46.0		µg/g ww											0.356-1.651	0.606	0.256							4,5
Canada, Nares Lake	60°09'30"N, 134°39'35"W	1996	Muscle	1			53.0		µg/g ww												0.343									4
Canada, Nonacho Lake	61°59'N,1 09°28'W	1975, 86	Muscle	59			55.3		µg/g ww												0.605									4
Canada, P&N Lake	63°39'N, 90°39'W	1988	Muscle	9			49.4		µg/g ww												0.328									4
Canada, Pitz Lake	63°59'N, 96°32'W	1976	Muscle	3			65.3		µg/g ww												0.453									4
Canada, Quartzite Lake	62°22'N, 94°32'W	1971	Muscle	5			36.5		µg/g ww												0.43									4
Canada, Quiet Lake	61°05'N, 133°05'W	1992, 99	Muscle	14			55.3		µg/g ww												0.302									4
Canada, Quiet Lake	61°05'N, 133°05'W	1999	Muscle	9			55.3 (5.2)		µg/g ww												0.29									19
Canada, Rorey Lake	66°55'N, 128°24'W	1978, 97	Muscle	51			53.6		µg/g ww												0.457									4
Canada, Ross Lake	62°41'N, 113°15'W	1973	Muscle	18			57.9		µg/g ww												0.266									4
Canada, Saturday Night Lake	63°39'N, 90°39'W	1998	Muscle	14			39.9		µg/g ww												0.221									4
Canada, Schultz Lake	64°45'N, 97°30'W	1976	Muscle	7			53.3		µg/g ww												0.551									4
Canada, Slemon Lake	63°13'N, 116°02'W	1983	Muscle	5			56.8		µg/g ww												0.276									4
Canada, Small Lake	74°45'N, 95°04'W	1976	Muscle	4			65.9		µg/g ww												0.588									4
Canada, Sparks Lake	61°12'N, 109°40'W	1981	Muscle	15			62.0		µg/g ww												0.293									4
Canada, Sparrow Lake	62°37'N, 113°38'W	1983	Muscle	7			63.4		µg/g ww												0.256									4
Canada, St. Thérèse Lake	64°37'N, 121°35.9'W	2001	Muscle	11			70.81 (6.07)	3217.9 (888.0)	µg/g ww												0.771	0.472								12
Canada, St. Thérèse Lake	64°37'N, 121°35.9'W	1980, 92	Muscle	16			79.4		µg/g ww												1.174									4
Canada, Thekulthili Lake	61°00'N, 110°06'W	1981	Muscle	15			60.5		µg/g ww												0.313									4
Canada, Thistlethwaite Lake	63°10'N, 113°34'W	1977	Muscle	20			55.3		µg/g ww												0.341									4
Canada, Tree River Lake	67°41'N, 111°53'W	1977	Muscle	3			60.7		µg/g ww												0.09									4
Canada, Trout Lake	60°34'20"N, 121°15'33"W	1977, 82	Muscle	30			59.6		µg/g ww												0.291									4
Canada, Tulemalu Lake	62°58'N, 99°25'W	1971	Muscle	9			54.3		µg/g ww												0.504									4
Canada, Turton Lake	65°48'25"N, 126°57'W	1996	Muscle	55			56.6		µg/g ww												0.60									4
Canada, Wagenitz Lake	63°03'N, 11°52'W	1977	Muscle	15			54.7		µg/g ww												0.305									4
Canada, Watson Lake	60°06'15"N, 128°49'00"W	1997	Muscle	2			51.5		µg/g ww												0.152									4
Canada, Willow Lake	62°10'40"N, 119°06'32"W	1999	Muscle	32			61.4		µg/g ww												0.38									4
Canada, Yathkyed Lake	62°40'N, 97°58'W	1971	Muscle	14			50.1		µg/g ww												0.484									4
Canada, Yaya Lake	69°10'50"N, 134°59'55"W	1995	Muscle	28			58.1		µg/g ww												0.21									4
Stenodus leucichthys (inconnu)																														
Canada, Great Slave Lake	62°50'N,113°50'W	1977, 78, 88, 96	Muscle	44			61.3		µg/g ww												0.08									4
Canada, Kelly Lake	65°23'51"N,126°09'39"W	1998	Muscle	4			76.0		µg/g ww												0.396									4
Canada, Mackenzie Delta	68°50'N,136°25'W	1977, 81, 92	Muscle	21			66.1		µg/g ww												0.137									4
Canada, Mackenzie River	69°21'N,133°54'W	1985, 95, 97	Muscle	82			63.7		µg/g ww												0.161									4
Canada, Peel River, NWT	67°41'50"N,134°31'50"W	1999	Muscle	10	M	13	78.0	3800	µg/g ww	0.068	0.031					n.d.				0.317	0.10								6
Canada, Peel River, Yukon	67°00'N,135°00'W	1999	Muscle	10					µg/g ww												0.256									4
Canada, Slave River	61°18'N,113°40'W	1989	Muscle	11			72.4		µg/g ww												0.108									4
Canada, Yaya Lake	69°10'50"N,134°59'55"W	1995	Muscle	30			65.6		µg/g ww												0.168									4
Stenodus leucichthys nelma (inconnu)																														
Russia, Chukotka, Kanchalan			Liver	6	M	11-14			µg/g ww	0.305					0.784					0.140									3
Russia, Chukotka, Kanchalan			Liver	3	M	7-10			µg/g ww	0.280					0.536					0.152									3
Russia, Chukotka, Kanchalan			Muscle	6	M	11-14			µg/g ww	0.096					0.114					0.042									3
Russia, Chukotka, Kanchalan			Muscle	3	M	7-10			µg/g ww	0.088					0.098					0.025									3

Table 10. Metals in freshwater fish, continued.

Species/location	Coordinate	Year	Tissue	n	Sex	Age, yrs (SD)	Length, cm (SD)	Weight, g (SD)	Unit	Lead min-max	Lead arithm. mean	Lead SD	Lead geom. mean	Lead GSD	Cadmium min-max	Cadmium arithm. mean	Cadmium SD	Cadmium geom. mean	Cadmium GSD	Mercury min-max	Mercury arithm. mean	Mercury SD	Mercury geom. mean	Mercury GSD	Selenium min-max	Selenium arithm. mean	Selenium SD	Selenium geom. mean	Remark	Reference
Stizostedion vitreum vitreum (walleye)																														
Canada, Jacques Lake	66°10'N, 127°24'W	1994, 95	Muscle	31			50.1		µg/g ww												0.992									4
Canada, Bovie Lake	60°10'N, 122°56'W	1988	Muscle	5			36.2		µg/g ww												0.18									4
Canada, Deep Lake	61°12.5'N, 120°54.5'W	2000	Muscle	4			44.8		µg/g ww												1.105									4
Canada, Ekali Lake	61°17'N, 120°35'W	1996	Muscle	14			41.0		µg/g ww												0.256									4
Canada, Grainger River	61°08'N, 123°04'W	1977	Muscle	15			46.8		µg/g ww												0.463									4
Canada, Great Slave Lake	62°50'N, 113°50'W	1977, 88, 89, 90, 96	Muscle	114			44.2		µg/g ww												0.268									4
Canada, Hay River	60°51'50"N, 115°44'00"W	1984, 89, 92	Muscle	78			38.7		µg/g ww												0.211									4
Canada, Kakisa	60°56'N, 117°43'W	1977, 81, 88, 89, 90, 92, 98	Muscle	80			36.5		µg/g ww												0.291									4
Canada, Leland	62°00'N, 112°59'W	1989, 90, 92	Muscle	29			48.0		µg/g ww												0.441									4
Canada, Little Doctor	61°53'N, 123°16'W	1996	Muscle	18			47.4		µg/g ww												0.753									4
Canada, Mackenzie River	69°21'N, 133°54'W	1997	Muscle	3			35.2		µg/g ww												0.191									4
Canada, McEwan	60°49'N, 119°57'W	2000	Muscle	17			43.0		µg/g ww												0.356									4
Canada, McGill	61°18'N, 121°00'45"W	2000	Muscle	22			48.0		µg/g ww												1.125									4
Canada, Muskeg River	60°19'N, 123°21'W	1977	Muscle	25			43.5		µg/g ww												0.511									4
Canada, Sanguez Lake	61°15'N, 120°28'W	1996	Muscle	20			44.0		µg/g ww												0.539									4
Canada, Sibbeston Lake	61°45'N, 122°45'W	1997	Muscle	5			46.8		µg/g ww												0.327									4
Canada, Slave River	61°18'N, 113°40'W	1988, 89, 90, 92	Muscle	130			41.3		µg/g ww												0.371									4
Canada, St. Thérèse Lake	64°37'N, 121°35.9'W	2001	Muscle	6			54.17 (3.08)	1521 (173.2)	µg/g ww												1.427	0.182								12
Canada, St. Thérèse Lake	64°37'N, 121°35.9'W	1975, 80, 92	Muscle	58			47.9		µg/g ww												1.258									4
Canada, Tathlina Lake	60°33'N, 117°32'W	1981, 90, 93, 93, 94, 98	Muscle	34			39.3		µg/g ww												0.452									4
Canada, Trout Lake	60°34'20"N, 121°15'33"W	1977	Muscle	7			59.1		µg/g ww												0.833									4
Canada, Tsetso Lake	61°51'N, 123°01'W	1997	Muscle	99			49.3		µg/g ww												0.485									4
Thymallus arcticus (Arctic grayling)																														
USA, Alaska, Kuskokwim River		2000	Muscle	6					µg/g ww												0.078	0.015								9
USA, Alaska, Kuskokwim river			Muscle						ng/g ww											64-87										13
USA, Alaska, Yukon river		2000	Muscle	4					µg/g ww												0.264	0.030								9
USA, Alaska, Yukon river			Muscle						ng/g ww											218-295										13
Thymallus thymallus (grayling)																														
Finland, Kemijoki river		1994	Muscle	4			25.2	236	µg/g ww												0.11	0.045								1
Russia, Chukotka, Lavrentiya			Liver	1	F				µg/g ww	0.082			0.044							0.180										3
Russia, Chukotka, Lavrentiya			Muscle	1	F				µg/g ww	0.013			0.003							0.090										3
Russia, Chukotka, Lavrentiya			Liver	6	M				µg/g ww	0.112			0.056							0.119										3
Russia, Chukotka, Lavrentiya			Muscle	6	M				µg/g ww	0.021			0.004							0.066										3

Remarks

A. Hg is approx. 100% as MeHg;
B. Values reported as MeHg;
C. Year and length not given from source;
D. Standardized to a 1kg fish;
E. Length-adjusted mean;
F. Five pooled samples, age estimated from length;
G. Normalized to fish length 40 cm;
H. Landlocked.

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Table A11. Metals in seawater.

Country/location	Year	Depth, m	Unit	Lead		Cadmium		Mercury	Remark	Reference
				min-max	mean	min-max	mean	mean		
Russia, Barents Sea	1994	surface	µg/L	0.03-2.06						1
Russia, Chukotka, Lavrentiya	2001	3	µg/L		<0.05		0.030	0.018		2
Russia, Chukotka, Lavrentiya	2001	3	µg/L		<0.05		0.510	0.013		2
Russia, Chukotka, Lavrentiya	2001	3	µg/L		0.215		0.095	0.036		2
Russia, East Siberian Sea	1994	surface	µg/L	0.3-2.0		0.01-0.02				1
Russia, Gulf of Anabar	1998	surface	µg/L		0.58		0.06			3
Russia, Gulf of Ob	1998	surface	µg/L		1.1		0.19	<0.02		3
Russia, Kara Sea	1994	surface	µg/L	0.01-9.6		0.01-0.5				1
Russia, Kara Sea, Baidaratskaya Bay and shelf of Yamal	1998	surface	µg/L		1.15		0.09			3
Russia, Kara Sea, Baidaratskaya Bay and shelf of Yamal	1998	bottom	µg/L		1.87		0.09			3
Russia, Khatanga Bay	1998	surface	µg/L		0.51		0.06			3
Russia, Laptev Sea (open part)	1998	surface	µg/L		0.58		0.06	<0.02		3
Russia, Pechora Bay	1998	surface	µg/L		0.52		0.07			3
Russia, Pechora Bay	1999	surface	µg/L		0.41		0.09			3
Russia, Pechora Sea	1994	surface	µg/L	0.08-1.7		0.01-0.08				1
Russia, Pechora Sea	1998	surface	µg/L		0.25		0.17			3
Russia, Pechora Sea	1998	surface	µg/L		0.24		0.06	0.02	costal	3
Russia, Pechora Sea	1998	bottom	µg/L		0.23		0.06	0.03	costal	3
Russia, Pechora Sea	1999	surface	µg/L		0.35		0.05			3
Russia, Sea of Chukotsk	1994	surface	µg/L	1.0-3.0		0.01-0.02				1

References

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Table A12. Metals in marine sediments.

Country/location	Coordinate	Date/year	Sediment type, depth, cm	Water depth, m	n	Unit	Copper mean	Copper SD	Zinc mean	Zinc SD	Cadmium mean	Cadmium SD	Lead mean	Lead SD	Mercury mean	Mercury SD	Selenium mean	Aluminium	Remark	Reference
Canada, W. Baffin Bay				12		µg/g	12	29	8	61	14									1,17
Faroe Islands, Sandoyarbanki	61.85N, 5.733W	26 July 2000	0-1	330		ng/g									7.48	0.17				2
Faroe Islands, Sandoyarbanki	61.85N, 5.733W	26 July 2000	1-2	330		ng/g									9.23	0.75				2
Faroe Islands, Sandoyarbanki	61.85N, 5.733W	26 July 2000	2-3	330		ng/g									9.75	0.36				2
Faroe Islands, Sandoyarbanki	61.85N, 5.733W	26 July 2000	3-4	330		ng/g									8.07	0.16				2
Faroe Islands, Sandoyarbanki	61.85N, 5.733W	26 July 2000	4-5	330		ng/g									7.76	0.00				2
Faroe Islands, Sandoyarbanki	61.85N, 5.733W	26 July 2000	5-6	330		ng/g									7.15	0.28				2
Faroe Islands, Sandoyarbanki	61.85N, 5.733W	26 July 2000	6-7	330		ng/g									6.37	0.41				2
Faroe Islands, Sandoyarbanki	61.85N, 5.733W	26 July 2000	7-8	330		ng/g									5.4	0.36				2
Faroe Islands, Sandoyarbanki	61.85N, 5.733W	26 July 2000	8-9	330		ng/g									4.78	0.45				2
Faroe Islands, Sandoyarbanki	61.85N, 5.733W	26 July 2000	9-10	330		ng/g									4.05	0.78				2
Faroe Islands, Sandoyarbanki	61.85N, 5.733W	26 July 2000	10-11	330		ng/g									4.12	0.69				2
Faroe Islands, Sandoyarbanki	61.85N, 5.733W	26 July 2000	0-1	330		ng/g									4.31	0.84				2
Faroe Islands, Sandoyarbanki	61.85N, 5.733W	26 July 2000	1-2	330		ng/g									7.4	0.93				2
Faroe Islands, Sandoyarbanki	61.85N, 5.733W	26 July 2000	2-3	330		ng/g									10	1.04				2
Faroe Islands, Sandoyarbanki	61.85N, 5.733W	26 July 2000	3-4	330		ng/g									8.62	0.43				2
Faroe Islands, Sandoyarbanki	61.85N, 5.733W	26 July 2000	4-5	330		ng/g									8.54	0.29				2
Faroe Islands, Sandoyarbanki	61.85N, 5.733W	26 July 2000	5-6	330		ng/g									9.55	1.11				2
Faroe Islands, Sandoyarbanki	61.85N, 5.733W	26 July 2000	6-7	330		ng/g									7.41	0.08				2
Faroe Islands, Sandoyarbanki	61.85N, 5.733W	26 July 2000	7-8	330		ng/g									7.11	0.78				2
Faroe Islands, Sandoyarbanki	61.85N, 5.733W	26 July 2000	8-9	330		ng/g									6.47	0.53				2
Faroe Islands, Sandoyarbanki	61.85N, 5.733W	26 July 2000	0-1	330		ng/g									8.43	0.22				2
Faroe Islands, Sandoyarbanki	61.85N, 5.733W	26 July 2000	1-2	330		ng/g									11.8	0.02				2
Faroe Islands, Sandoyarbanki	61.85N, 5.733W	26 July 2000	2-3	330		ng/g									11.6	0.10				2
Faroe Islands, Sandoyarbanki	61.85N, 5.733W	26 July 2000	3-4	330		ng/g									11.2	0.21				2
Faroe Islands, Sandoyarbanki	61.85N, 5.733W	26 July 2000	4-5	330		ng/g									10.9	0.05				2
Faroe Islands, Sandoyarbanki	61.85N, 5.733W	26 July 2000	5-6	330		ng/g									6.34	0.50				2
Faroe Islands, Sandoyarbanki	61.85N, 5.733W	26 July 2000	6-7	330		ng/g									5.08	0.52				2
Faroe Islands, Sandoyarbanki	61.85N, 5.733W	26 July 2000	7-8	330		ng/g									5.27	0.71				2
Faroe Islands, Sandoyarbanki	61.85N, 5.733W	26 July 2000	8-9	330		ng/g									3.94	0.50				2
Faroe Islands, Sandoyarbanki	61.85N, 5.733W	26 July 2000	0-1	330		ng/g									13.6	0.02				2
Faroe Islands, Sandoyarbanki	61.85N, 5.733W	26 July 2000	1-2	330		ng/g									13.7	0.33				2
Faroe Islands, Sandoyarbanki	61.85N, 5.733W	26 July 2000	2-3	330		ng/g									12.4	0.10				2
Faroe Islands, Sandoyarbanki	61.85N, 5.733W	26 July 2000	3-4	330		ng/g									12.1	0.17				2
Faroe Islands, Sandoyarbanki	61.85N, 5.733W	26 July 2000	4-5	330		ng/g									11.3	0.02				2
Faroe Islands, Sandoyarbanki	61.85N, 5.733W	26 July 2000	5-6	330		ng/g									10.5	0.26				2
Faroe Islands, Sandoyarbanki	61.85N, 5.733W	26 July 2000	6-7	330		ng/g									10.9	0.52				2
Faroe Islands, Sandoyarbanki	61.85N, 5.733W	26 July 2000	7-8	330		ng/g									9.22	0.06				2
Faroe Islands, Sandoyarbanki	61.85N, 5.733W	26 July 2000	8-9	330		ng/g									7.45	0.21				2
Faroe Islands, Sandoyarbanki	61.85N, 5.733W	26 July 2000	9-10	330		ng/g									7.16	0.10				2
Greenland, W Greenland					22	µg/g	49	40	77	19	0.15	0.16	18	8						3,17
Greenland, E. Greenland					10	µg/g	46	32	89	20	0.11	0.05	19	7						3,4,17
Norway, Svalbard					15	µg/g			107	3	0.22	0.03								4,17
Russia, Barents Sea			bottom			µg/g					5.3		24.6		4.5					5
Russia, Beaufort Sea					23	µg/g	33	9	98	18										6
Russia, Beaufort Sea					13	µg/g	24	4	93	13	0.19	0.06	15	4						7
Russia, Beaufort Sea					62	µg/g	27	7	96	18	0.27	0.11	15	6	0.017	0.007				8
Russia, Beaufort Sea					7	µg/g	29	8	115	22	0.18	0.13	17	3	0.074	0.034				9
Russia, Bering Sea					28	µg/g	11	9	56	26	0.09	0.03	8.5	2.8	0.025	0.023				10
Russia, Chukchi Sea					12	µg/g	22	6	79	18										11
Russia, East Siberian Sea					24	µg/g	16	6	85	25	0.14	0.08	18	5	0.037	0.019				12
Russia, East Siberian Sea		1994	silt and sandy silts			µg/g					2.4		23.0							13
Russia, Gulf of Ob		1998	sandy silt			µg/g					0.2		22.7							14
Russia, Kara Sea		1994	silt and sandy silts			µg/g					0.2		33.1							13
Russia, Kara Sea					36	µg/g	20	6			0.11	0.07								4,17
Russia, Kara Sea					16	µg/g	27	14	80	22	0.09	0.03	14	3	0.028	0.009				12
Russia, Khatanga Bay		1998	sandy silt			µg/g					0.31		24.2							14
Russia, Laptev Sea					11	µg/g	19	7	92	15	0.11	0.06	20	5	0.037	0.016				12
Russia, Laptev Sea					10	µg/g	15	4	98	22	0.06	0.02	18	3						15
Russia, Laptev Sea (open part)		1998	silty sand			µg/g					0.14		6.6							14
Russia, Pechora Bay		1998	sands and silty sands			µg/g					0.12		10.3							14
Russia, Pechora Bay		1999	silt and sandy silts			µg/g					0.15		8.8							14
Russia, Pechora Sea		1999	silt and sandy silts			µg/g					0.12		7.8							14
Russia, Pechora Sea					40	µg/g	21	2	84	9	0.11	0.05								4,17
Russia, Pechora Sea					109	µg/g	17	10					24	18						16

Table A12. Metals in marine sediments, continued.

Country/location	Coordinate	Date/year	Sediment type	Water depth, m	n	Unit	Copper mean	Copper SD	Zinc mean	Zinc SD	Cadmium mean	Cadmium SD	Lead mean	Lead SD	Mercury mean	Mercury SD	Selenium mean	Aluminium	Remark	Reference
Russia, Pechora Sea			bottom			ppm					9		24							5
Russia, Prirazlomnoye oil field			bottom			ppm					1		18							5
Russia, Shtokmanovskoye gas condensate field			bottom			ppm					2		22							5
Russia, St. Anna Trough Area			bottom			ppm							20							5
Russia, White Sea			bottom			ppm							71							5
USA, Alaska, Port Valdez, Mineral Creek Flats	61.14N, 146.46W	1986			17	µg/g	65		150		0.1		18		0.01		0.42	89000		10
USA, Alaska, Port Valdez, Mineral Creek Flats	61.14N, 146.46W	1986			17	µg/g	66		150		0.17		18		0.01		0.42	88000		10
USA, Alaska, Port Valdez, Mineral Creek Flats	61.14N, 146.46W	1986			17	µg/g	66		150		0.1		16		0.01		0.52	89000		10
USA, Alaska, Port Valdez, Mineral Creek Flats	61.14N, 146.46W	1987			17	µg/g	58		140		0.12		17		0.05		0.24	90000		10
USA, Alaska, Port Valdez, Mineral Creek Flats	61.14N, 146.46W	1987			17	µg/g	68		140		0.12		17		0.04			89000		10
USA, Alaska, Port Valdez, Mineral Creek Flats	61.14N, 146.46W	1987			17	µg/g	70		170		0.11		18		0.06			82000		10
USA, Alaska, Unakwit Inlet, Siwash Bay	60.96N, 147.66W	1986			17	µg/g	45		110		0.13		15		0.06		0.34	74000		10
USA, Alaska, Unakwit Inlet, Siwash Bay	60.96N, 147.66W	1986			17	µg/g	43		110		0.11		13		0.06		0.43	73000		10
USA, Alaska, Unakwit Inlet, Siwash Bay	60.96N, 147.66W	1986			17	µg/g	43		100		0.11		13		0.05		0.34	72000		10
USA, Alaska, Unakwit Inlet, Siwash Bay	60.96N, 147.66W	1987			17	µg/g	49		120		0.08		18		0.06		0.24	69000		10
USA, Alaska, Unakwit Inlet, Siwash Bay	60.96N, 147.66W	1987			17	µg/g	46		120		0.08		19		0.06		0.24	57000		10
USA, Alaska, Unakwit Inlet, Siwash Bay	60.96N, 147.66W	1987			17	µg/g	35		100		0.07		12		0.05		0.32	64000		10

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Table A13. Metals in marine invertebrates.

Type of invertebrate	Species	Location	Coordinate	Year	Tissue	n	Sex	Age, yrs	Size, cm	Unit	Lead min-max	Lead arithm. mean	Lead SD	Cadmium min-max	Cadmium arithm. mean	Cadmium SD	Mercury arithm. mean	Mercury SD	Selenium arithm. mean	Selenium SD	Remark	Reference
Amphipoda	*Anonyx nugax*	Russia, Pechora Sea	69°50'N, 59.10E	1992	Pooled	1 pool				mg/kg ww					0.430							1
Amphipoda	*Byblis gaimardi*	Russia, Pechora Sea	69°50'N, 59°10'E	1992	Pooled	1 pool				mg/kg ww					0.042							1
Nemertini	*Cerebratulus barentsi*	Russia, Pechora Sea	69°50'N, 59°10'E	1992	Pooled	1 pool				mg/kg ww					1.090							1
Malacostraca	*Chionoecetes opilio* (queen crab)	Greenland, Qeqertarsuaq	69°00'N, 53°19'W	1999	Hepatopancreas	5	M		8.9-9.8	mg/kg ww					5.060	1.710	0.074	0.023	3.27	1.15		2
		Greenland, Qeqertarsuaq	69°00'N, 53°19'W	1999	Muscle	10	M		8.9-11.9	mg/kg ww					0.035	0.032	0.096	0.066	0.599	0.157		2
Bivalvia	*Chlamys islandica* (Iceland scallop)	Canada, Nain, Labrador		1999	Gonad	19		19		mg/kg ww					1.680		0.02					3
		Canada, Nain, Labrador		1999	Gut	18		19		mg/kg ww					6.150		0.05		2.83			3
		Canada, Nain, Labrador		1999	Muscle	18		19		mg/kg ww					0.950		0.03		0.19			3
		Greenland, Qeqertarsuaq	69°00'N, 53°19'W	2000	Muscle	8 pools			5.9-8.5	mg/kg ww					2.040	0.650	0.022	0.002	0.81			2
		Russia, Pechora Sea	68°30'N, 38°08'E	1992	Soft tissue	10				mg/kg ww					4.300				0.129	0.010		1
		Russia, Pechora Sea	70°11'N, 58°10'E	1992	Soft tissue	5				mg/kg ww					2.920							1
Gastropoda	*Chlamys opercularis* (queen scallop)	Faroe Islands	62°06'N, 6°29'W	1997	Soft tissue	50			6.3	mg/kg ww					0.420		<0.02					4
		Faroe Islands	62°04'N, 6°38'W	1997	Soft tissue	56			6.2	mg/kg ww					0.420		<0.02					4
Bivalvia	*Clinocardium ciliatum*	Russia, Pechora Sea	70°11N, 58°10E	1992	Soft tissue	1 pool				mg/kg ww					0.170							1
		Russia, Pechora Sea	69°59N, 57°15E	1993	Soft tissue	2				mg/kg ww					0.340	0.040						1
		Russia, Pechora Sea	69°38N, 58°42E	1993	Soft tissue	1				mg/kg ww					0.160							1
		Russia, Pechora Sea	69°10N, 57°51E	1993	Soft tissue	3				mg/kg ww					0.130	0.500						1
		Russia, Pechora Sea	69°50N, 54°00E	1993	Soft tissue	3				mg/kg ww					0.280	0.030						1
		Russia, Pechora Sea	68°50N, 50°43E	1993	Soft tissue	2				mg/kg ww					0.320	0.040						1
Gastropoda	*Gibbula* spp. (topsnails)	Faroe Islands, Kirkjubøur	61°57'N, 6°47'W	1996/97	Whole animal	297				µg/g ww		0.138			0.483						A	5
Siphunculoida	*Golfingia* spp.	Russia, Pechora Sea	69°50'N, 59°10'E	1992	Pooled	1 pool				mg/kg ww					0.020							1
Gastropoda	*Lacuna vincta* (banded chink shell)	Faroe Islands, Kirkjubøur	61°57'N, 6°47'W	1996/97	Whole animal	502				µg/g ww		0.040			0.713						B	5
Amphipoda	*Lembos arcticus*	Russia, Pechora Sea	69°50'N, 59°10'E	1992	Pooled	1 pool				mg/kg ww					0.070							1
Polychaeta	*Lumbriconereis fragellis*	Russia, Pechora Sea	69°50'N, 59°10'E	1992	Pooled	1 pool				mg/kg ww					0.030							1
Bivalvia	*Macoma baltica*	Russia, Pechora Sea	68°35'N, 55°13'E	1992	Soft tissue	9				mg/kg ww					0.060							1
Bivalvia	*Macoma calcarea*	Russia, Pechora Sea	70°11'N, 58°10'E	1992	Soft tissue	25				mg/kg ww					0.220							1
Bivalvia	*Musculus corrugatus*	Russia, Pechora Sea	70°11'N, 58°10'E	1992	Soft tissue	1				mg/kg ww					1.000							1
Bivalvia	*Musculus niger*	Russia, Pechora Sea	70°11'N, 58°10'E	1992	Soft tissue	1 pool				mg/kg ww					2.180							1
Bivalvia	*Mytilus edulis* (blue mussel)	Canada, Deception Bay, Labrador		1998-99	Soft tissue					mg/kg ww					0.390		0.01		0.68			3
		Canada, Kangiqsualujjuaq, Labrador		1998-99	Soft tissue					mg/kg ww					0.230		0.03		0.41			3
		Canada, Kangiqsujuaq, Labrador		1998-99	Soft tissue					mg/kg ww					0.350		0.01		0.48			3
		Canada, Kuujjuaq, Labrador		1998-99	Soft tissue					mg/kg ww					0.200		0.02		0.50			3
		Canada, Makkovik, Labrador		1998-99	Soft tissue					mg/kg ww					0.270		0.02		0.60			3
		Canada, Nain, Labrador		1998-99	Soft tissue					mg/kg ww					0.330		0.01		0.59			3
		Canada, Quaqtaq, Labrador		1998-99	Soft tissue					mg/kg ww					1.100		0.01		0.69			3
		Faroe Island	62°03'N, 6°52.2'W	1997	Soft tissue	1 pool			3.7	mg/kg ww					0.170		0.03					4
		Greenland, Qeqertarsuaq		1994	Soft tissue	3			4-5	mg/kg ww					0.694	0.074	0.0139	0.0011	0.942	0.097		2
		Greenland, Qeqertarsuaq		1994	Soft tissue	3			5-6	mg/kg ww					0.826	0.054	0.0143	0.0009	0.82	0.124		2
		Greenland, Qeqertarsuaq		1994	Soft tissue	3			6-7	mg/kg ww					1.120	0.150	0.0165	0.0015	0.726	0.122		2
		Greenland, Qeqertarsuaq		1994	Soft tissue	3			7-8	mg/kg ww					1.260	0.720	0.0174	0.0019	0.666	0.226		2
		Greenland, Qeqertarsuaq		1994	Soft tissue	3			8-9	mg/kg ww					1.100	0.160	0.0162	0.0014	0.533	0.101		2
		Greenland, Qeqertarsuaq	61°N, 53°W	1999	Soft tissue	2 pools			3-4	mg/kg ww					1.210	1.020	0.0122	0.0036	0.485	0.100		2
		Greenland, Qeqertarsuaq	61°N, 53°W	1999	Soft tissue	3 pools			4-5	mg/kg ww					0.552	0.152	0.0106	0.0036	0.517	0.018		2
		Greenland, Qeqertarsuaq	61°N, 53°W	1999	Soft tissue	3 pools			5-6	mg/kg ww					0.892	0.198	0.0133	0.0013	0.467	0.042		2
		Greenland, Qeqertarsuaq	61°N, 53°W	1999	Soft tissue	3 pools			6-7	mg/kg ww					1.170	0.190	0.0156	0.0046	0.417	0.032		2
		Greenland, Qeqertarsuaq	61°N, 53°W	1999	Soft tissue	3 pools			7-8	mg/kg ww					1.200	0.230	0.0199	0.0012	0.474	0.160		2
		Greenland, Qeqertarsuaq	61°N, 53°W	1999	Soft tissue	1 pools			8-9	mg/kg ww					2.270		0.0196		0.365			2
		Iceland, north	66°34'N, 18°W	1995	Soft tissue	1 pool			4.3	mg/kg dw					3.220	0.200	0.083	0.002	3.06	0.41		6
		Iceland, north	66°34'N, 18°W	1996	Soft tissue	1 pool			5.0	mg/kg dw					3.550	0.130	0.066	0.007	3.09	0.55		7
		Iceland, north	66°34'N, 18°W	1997	Soft tissue	1 pool			4.9	mg/kg dw		0.471	0.011		4.070	0.005	0.094	0.004	0.82	0.25		7
		Iceland, north	66°34'N, 18°W	1998	Soft tissue	1 pool			4.6	mg/kg dw					2.640	0.100	0.079	0.003	2.83	0.05		8

Table A13. Metals in marine invertebrates, continued.

Type of invertebrate	Species	Location	Coordinate	Year	Tissue	n	Sex	Age, yrs	Size, cm	Unit	Lead min-max	Lead arithm. mean	Lead SD	Cadmium min-max	Cadmium arithm. mean	Cadmium SD	Mercury arithm. mean	Mercury SD	Selenium arithm. mean	Selenium SD	Remark	Reference
Bivalvia	*Mytilus edulis* (blue mussel)	Iceland, north	66°34'N, 18°W	1999	Soft tissue	1 pool			4.8	mg/kg dw					3.520	0.090	0.106	0.008	2.06	0.04		8
		Iceland, southeast	64°23'N, 21°26'W	1995	Soft tissue	1 pool			4.9	mg/kg dw					4.400	0.043	0.05	0.006	1.83	0.11		6
		Iceland, southeast	64°21'N, 21°29'W	1995	Soft tissue	1 pool			4.8	mg/kg dw					1.280	0.096	0.023	0.003	2.33	0.23		6
		Iceland, southeast	64°01'N, 22°09'W	1995	Soft tissue	1 pool			4.7	mg/kg dw					1.990	0.086	0.046	0.02	2.75	0.02		6
		Iceland, southeast	64°02'N, 22°02'W	1995	Soft tissue	1 pool			4.6	mg/kg dw					2.050	0.110	0.022	0.003	3.18	0.46		6
		Iceland, southeast	64°23'N, 21°26'W	1996	Soft tissue	1 pool			4.5	mg/kg dw					1.840	0.021	0.029	0.004	2.26	0.04		9
		Iceland, southeast	64°21'N, 21°29'W	1996	Soft tissue	1 pool			4.2	mg/kg dw					2.310	0.025	0.028	0.002	2.68	0.09		9
		Iceland, southeast	64°01'N, 22°09'W	1996	Soft tissue	1 pool			4.5	mg/kg dw					2.030	0.032	0.053	0.001	2.84	0.03		9
		Iceland, southeast	64°02'N, 22°02'W	1996	Soft tissue	1 pool			4.6	mg/kg dw					2.460	0.060	0.053	0.001	2.73	0.03		9
		Iceland, southeast	64°23'N, 21°26'W	1997	Soft tissue	1 pool			5.1	mg/kg dw					1.360	0.104	0.034	0.006	1.2	0.15		7
		Iceland, southeast	64°21'N, 21°29'W	1997	Soft tissue	1 pool			4.8	mg/kg dw					2.310	0.048	0.042	0.008	1.35	0.08		7
		Iceland, southeast	64°01'N, 22°09'W	1997	Soft tissue	1 pool			4.7	mg/kg dw					2.630	0.050	0.045	0.003	4.03	0.94		7
		Iceland, southeast	64°02'N, 22°02'W	1997	Soft tissue	1 pool			5.0	mg/kg dw					2.570	0.080	0.08	0.01	1.43	0.19		7
		Iceland, southeast	64°23'N, 21°26'W	1998	Soft tissue	1 pool			4.9	mg/kg dw					1.960	0.080	0.035	0.001	3.48	0.14		8
		Iceland, southeast	64°21'N, 21°29'W	1998	Soft tissue	1 pool			4.7	mg/kg dw					1.660	0.110	0.051	0.001	3.17	0.16		8
		Iceland, southeast	64°01'N, 22°09'W	1998	Soft tissue	1 pool			5.0	mg/kg dw					2.350	0.170	0.106	0.002	3.95	0.18		8
		Iceland, southeast	64°02'N, 22°02'W	1998	Soft tissue	1 pool			5.0	mg/kg dw					3.280	0.020	0.06	0.001	3.86	0.11		8
		Iceland, southeast	64°23'N, 21°26'W	1999	Soft tissue	1 pool			5.0	mg/kg dw					2.290	0.130	0.067	0.017	2.19	0.06		8
		Iceland, southeast	64°21'N, 21°29'W	1999	Soft tissue	1 pool			4.4	mg/kg dw					1.210	0.030	0.077	0.036	1.84	0.05		8
		Norway	64°02'N, 10°01'E	1995	Soft tissue	3 pools			3 size groups	mg/kg dw					0.740		0.041				C	10
		Norway	68°11'N, 14°40'E	1995	Soft tissue	3 pools			3 size groups	mg/kg dw					0.690		0.340				C	10
		Norway	68°57'N, 16°39'E	1995	Soft tissue	3 pools			3 size groups	mg/kg dw					2.900		0.064				C	10
		Norway	70°06'N, 20°33'E	1995	Soft tissue	3 pools			3 size groups	mg/kg dw					4.300		0.095				C	10
		Norway	70°31'N, 22°15'E	1995	Soft tissue	3 pools			3 size groups	mg/kg dw					2.700		0.050				C	10
		Norway	70°58'N, 25°48'E	1995	Soft tissue	3 pools			3 size groups	mg/kg dw					2.100		0.062				C	10
		Norway	70°42'N, 28°33'E	1995	Soft tissue	3 pools			3 size groups	mg/kg dw					1.300		0.060				C	10
		Norway	70°04'N, 30°10'E	1995	Soft tissue	3 pools			3 size groups	mg/kg dw					1.700		0.049				C	10
		Norway	69°47'N, 30°11'E	1995	Soft tissue	3 pools			3 size groups	mg/kg dw					1.300		0.145				C	10
		Norway	64°02'N, 10°01'E	1996	Soft tissue	3 pools			3 size groups	mg/kg dw					0.690		0.023				C	10
		Norway	68°11'N, 14°40'E	1996	Soft tissue	3 pools			3 size groups	mg/kg dw					0.780		0.328				C	10
		Norway	68°57'N, 16°39'E	1996	Soft tissue	3 pools			3 size groups	mg/kg dw					1.600		0.064				C	10
		Norway	70°31'N, 22°15'E	1996	Soft tissue	3 pools			3 size groups	mg/kg dw					2.000		0.052				C	10
		Norway	70°58'N, 25°48'E	1996	Soft tissue	3 pools			3 size groups	mg/kg dw					2.700		0.056				C	10
		Norway	70°42'N, 28°33'E	1996	Soft tissue	3 pools			3 size groups	mg/kg dw					1.400		0.052				C	10
		Norway	70°04'N, 30°10'E	1996	Soft tissue	3 pools			3 size groups	mg/kg dw					2.300		0.059				C	10
		Norway	69°47'N, 30°11'E	1996	Soft tissue	3 pools			3 size groups	mg/kg dw					1.100		0.086				C	10
		Norway	64°02'N, 10°01'E	1997	Soft tissue	3 pools			3 size groups	mg/kg dw					0.720		0.067				C	10
		Norway	68°15'N, 14°40'E	1997	Soft tissue	3 pools			3 size groups	mg/kg dw					0.850		0.104				C	10
		Norway	68°57'N, 16°39'E	1997	Soft tissue	3 pools			3 size groups	mg/kg dw					1.900		0.085				C	10
		Norway	70°06'N, 20°33'E	1997	Soft tissue	3 pools			3 size groups	mg/kg dw					3.900		0.104				C	10
		Norway	70°31'N, 22°15'E	1997	Soft tissue	3 pools			3 size groups	mg/kg dw					1.500		0.059				C	10
		Norway	70°04'N, 30°10'E	1997	Soft tissue	3 pools			3 size groups	mg/kg dw					1.100		0.062				C	10
		Norway	69°47'N, 30°11'E	1997	Soft tissue	3 pools			3 size groups	mg/kg dw					1.600		0.146				C	10
		Norway	68°15'N, 14°40'E	1998	Soft tissue	3 pools			3 size groups	mg/kg dw					1.600		0.155				C	10
		Norway	70°04'N, 30°10'E	1998	Soft tissue	3 pools			3 size groups	mg/kg dw					2.300		0.058				C	10
		Norway	68°15'N, 14°40'E	1999	Soft tissue	3 pools			3 size groups	mg/kg dw					2.200		0.246				C	10
		Norway	70°04'N, 30°10'E	1999	Soft tissue	3 pools			3 size groups	mg/kg dw					1.600		0.063				C	10
		Norway	64°02'N, 10°01'E	1992	Soft tissue					mg/kg dw					1.1		0.055				C	10
		Norway	64°02'N, 10°01'E	1993	Soft tissue					mg/kg dw					0.54		0.034				C	10
		Norway	64°02'N, 10°01'E	1994	Soft tissue					mg/kg dw					0.94		0.052				C	10
		Norway	68°09'N, 14°39'E	1992	Soft tissue					mg/kg dw					1.1		0.087				C	10
		Norway	68°09'N, 14°39'E	1993	Soft tissue					mg/kg dw					1.1		0.086				C	10
		Norway	68°10'N, 14°40'E	1994	Soft tissue					mg/kg dw					0.71		0.035				C	10
		Norway	68°57'N, 16°38'E	1994	Soft tissue					mg/kg dw					1.9		0.069				C	10
		Norway	70°06'N, 20°33'E	1994	Soft tissue					mg/kg dw					3.5		0.084				C	10
		Norway	70°30'N, 22°15'E	1994	Soft tissue					mg/kg dw					1.7		0.055				C	10
		Norway	70°58'N, 25°48'E	1994	Soft tissue					mg/kg dw					2.7		0.039				C	10
		Norway	70°41'N, 28°33'E	1994	Soft tissue					mg/kg dw					1.4		0.073				C	10
		Norway	70°04'N, 30°10'E	1994	Soft tissue					mg/kg dw					1.7		0.053				C	10
		Norway	69°47'N, 30°11'E	1994	Soft tissue					mg/kg dw					1.3		0.182				C	10
		Russia, Pechora Sea	69°31'N, 49°10'E	1992	Soft tissue	1 pools				mg/kg ww					0.090							1
		Russia, Pechora Sea	69°31'N, 49°10'E	1992	Soft tissue	1pools				mg/kg ww					0.170							1
		Russia, Pechora Sea	69°15'N, 57°17'E	1993	Soft tissue	2				mg/kg ww					0.320	0.010						1
Bivalvia	*Mytilus* spp. (mussels)	USA, Alaska, Gulf of Alaska	60°04'N, 147°48'W	1995	Soft tissue					mg/kg dw					3.284		0.10		4.48			11
		USA, Alaska, Gulf of Alaska	60°07'N, 147°8'W	1995	Soft tissue					mg/kg dw					3.102		0.11		4.68			11

Table A13. Metals in marine invertebrates, continued.

Type of invertebrate	Species	Location	Coordinate	Year	Tissue	n	Sex	Age, yrs	Size, cm	Unit	Lead min-max	Lead arithm. mean	Lead SD	Cadmium min-max	Cadmium arithm. mean	Cadmium SD	Mercury arithm. mean	Mercury SD	Selenium arithm. mean	Selenium SD	Remark	Reference
		USA, Alaska, Gulf of Alaska	59°2'N, 151°5'W	1995	Soft tissue					mg/kg dw					2.371		0.12		2.83			11
		USA, Alaska, Gulf of Alaska	59°6'N, 151°5'W	1995	Soft tissue					mg/kg dw					1.939		0.11		2.39			11
		USA, Alaska, Gulf of Alaska	58°5'N, 152°63'W	1995	Soft tissue					mg/kg dw					3.166		0.05		2.39			11
		USA, Alaska, Gulf of Alaska		1997	Soft tissue					mg/kg dw					1.720		0.13		2.69			11
		USA, Alaska, Gulf of Alaska		1999	Soft tissue					mg/kg dw					1.37		0.13		1.47			11
		USA, Alaska, Ketchikan	55°18'N, 131°34'W	1995	Soft tissue					mg/kg dw					4.997		0.06		3.98			11
		USA, Alaska, Ketchikan	55°18'N, 131°34'W	1997	Soft tissue					mg/kg dw					3.610		0.04		2.24			11
		USA, Alaska, Ketchikan	55°18'N, 131°34'W	1999	Soft tissue					mg/kg dw					3.62		0.215		2.35			11
		USA, Alaska, Nahku Bay	59°3'N, 135°18'W	1995	Soft tissue					mg/kg dw					3.139		0.06		4.34			11
		USA, Alaska, Nahku Bay	59°3'N, 135°18'W	1997	Soft tissue					mg/kg dw					4.060		0.09		3.02			11
		USA, Alaska, Nahku Bay	59°3'N, 135°18'W	1999	Soft tissue					mg/kg dw					4.02		0.146		3.08			11
		USA, Alaska, Port Valdez, Mineral Creek Flats	61°8'N, 146°3'W	1986	Soft tissue					µg/g dw					2.830	0.380	0.06	0.03	3.73	0.47		12
		USA, Alaska, Port Valdez, Mineral Creek Flats	61°8'N, 146°3'W	1987	Soft tissue					µg/g dw					2.470	0.120	0.06	0.01	3.10	0.36		12
		USA, Alaska, Port Valdez, Mineral Creek Flats	61°8'N, 146°3'W	1988	Soft tissue					µg/g dw					2.670	0.060	0.07	0.01	2.97	0.21		12
		USA, Alaska, Port Valdez, Mineral Creek Flats	61°8'N, 146°3'W	1990	Soft tissue					µg/g dw					2.420	0.090			2.70	0.33		12
		USA, Alaska, Port Valdez, Mineral Creek Flats	61°8'N, 146°3'W	1991	Soft tissue					µg/g dw					2.000	0.000	0.08	0.00	4.53	0.06		12
		USA, Alaska, Port Valdez, Mineral Creek Flats	61°8'N, 146°3'W	1992	Soft tissue					µg/g dw		1.510			2.440		0.09		3.37			12
		USA, Alaska, Port Valdez, Mineral Creek Flats	61°8'N, 146°3'W	1993	Soft tissue					µg/g dw		1.100			2.670		0.08		5.07			12
		USA, Alaska, Port Valdez, Mineral Creek Flats	61°8'N, 146°3'W	1995	Soft tissue					µg/g dw		1.190			2.320		0.08		5.49			12
		USA, Alaska, Port Valdez, Mineral Creek Flats	61°8'N, 146°3'W	1997	Soft tissue					µg/g dw		1.300			2.410		0.09		2.27			12
		USA, Alaska, Port Valdez, Mineral Creek Flats	61°8'N, 146°3'W	1999	Soft tissue					µg/g dw		2.210			2.89		0.108		0			12
		USA, Alaska, Prince Willian Sound	60°39N, 146°W	1995	Soft tissue					mg/kg dw					2.550	0.150	0.09	0.02	3.79	0.09		11
		USA, Alaska, Unakwit Inlet, Siwash Bay	60°58'N,147°40'W	1986	Soft tissue					µg/g dw					4.630	0.210	0.09	0.05	4.60	0.20		11
		USA, Alaska, Unakwit Inlet, Siwash Bay	60°58'N,147°40'W	1987	Soft tissue					µg/g dw					3.470	0.230	0.09	0.00	3.53	0.60		11
		USA, Alaska, Unakwit Inlet, Siwash Bay	60°58'N,147°40'W	1988	Soft tissue					µg/g dw					2.670	0.120	0.11	0.01	3.80	0.40		11
		USA, Alaska, Unakwit Inlet, Siwash Bay	60°58'N,147°40'W	1990	Soft tissue					µg/g dw					2.700	0.170	0.04	0.04	3.62	0.21		11
		USA, Alaska, Unakwit Inlet, Siwash Bay	60°58'N,147°40'W	1991	Soft tissue					µg/g dw					3.000	0.350	0.08	0.00	4.07	0.12		11
		USA, Alaska, Unakwit Inlet, Siwash Bay	60°58'N,147°40'W	1992	Soft tissue					µg/g dw		0.700			2.340		0.10		3.92			11
		USA, Alaska, Unakwit Inlet, Siwash Bay	60°58'N,147°40'W	1993	Soft tissue					µg/g dw		0.560			2.870		0.10		3.49			11
		USA, Alaska, Unakwit Inlet, Siwash Bay	60°58'N,147°40'W	1995	Soft tissue					µg/g dw		2.220			2.960		0.11		5.06			11
		USA, Alaska, Unakwit Inlet, Siwash Bay	60°58'N,147°40'W	1997	Soft tissue					µg/g dw		1.600			3.610		0.10		3.14			11
		USA, Alaska, Unakwit Inlet, Siwash Bay	60°58'N,147°40'W	1999	Soft tissue					µg/g dw		1.420			2.38		0.10		3.82			11
Bivalvia	*Nicania montagui*	Russia, Pechora Sea	69°15'N, 57°17'E	1993	Soft tissue	3				mg/kg ww					0.920	0.160						1
Gastropoda	*Nucella lapillus* (dogwhelk)	Faroe Islands, Gamlarætt	61°57'N, 6°49'W	1997	Soft parts	38				µg/g ww		0.4			158		0.1					5
Polychaeta	*Onuphis conchylega*	Russia, Pechora Sea	69°50'N, 59°10'E	1992	Pooled	1 pool				mg/kg ww					0.100							1
Polychaeta	*Ophelia limacina*	Russia, Pechora Sea	69°50'N, 59°10'E	1992	Pooled	1 pool				mg/kg ww					0.050							1
Malacostraca	*Pandalus borealis* (deep sea shrimp)	Greenland, Uummannaq	69°00'N, 53°19'W	1999	Muscle	11 pools	F		2.4-3.0	mg/kg ww							0.05	0.016	0.142	0.014		2
Malacostraca	*Paralithodes camtschaticus* (red king crab)	USA, Norton Sound	64°26'N, 165°30'W	1987	Muscle	25	F/M			µg/g dw		0.230	0.010		0.250	0.090	0.029	0.001				13
		USA, Norton Sound	64°26'N, 165°30'W	1988	Muscle	24	F/M			µg/g dw		0.710	0.050		0.240	0.120	0.046	0.008				13
		USA, Norton Sound	64°26'N, 165°30'W	1988	Muscle	37	F/M			µg/g dw		0.700	0.040		0.250	0.170	0.033	0.006				13
		USA, Norton Sound	64°26'N, 165°30'W	1989	Muscle	20	F/M			µg/g dw		0.810	0.070		0.290	0.060	0.056	0.004				13
		USA, Norton Sound	64°26'N, 165°30'W	1989	Muscle	20	F/M			µg/g dw		0.820	0.060		0.200	0.040	0.06	0.005				13
		USA, Norton Sound	64°26'N, 165°30'W	1990	Muscle	10	F/M			µg/g dw		0.230	0.040		0.150	0.070	0.105	0.027				13
		USA, Norton Sound	64°26'N, 165°30'W	1990	Muscle	10	F/M			µg/g dw		0.260	0.040		0.190	0.060	0.139	0.051				13
Amphipoda	*Rhachotropis aculeata*	Russia, Pechora Sea	69°50N, 59°10E	1992	Pooled	1 pool				mg/kg ww					0.110							1
Bivalvia	*Serripes groenlandicus*	Russia, Pechora Sea	69°15N, 57°17E	1993	Soft tissue	3				mg/kg ww					0.100	0.030						1
		Russia, Pechora Sea	69°19N, 59°04E	1993	Soft tissue	4				mg/kg ww					0.620	0.100						1
Bivalvia	*Tridonta borealis*	Russia, Pechora Sea	70°11N, 58°10E	1992	Soft tissue	1 pool				mg/kg ww					1.580							1
		Russia, Pechora Sea	69°50N, 57°15E	1993	Soft tissue	3				mg/kg ww					1.800	0.040						1
		Russia, Pechora Sea	69°38N, 58°42E	1993	Soft tissue	3				mg/kg ww					1.710	0.030						1
		Russia, Pechora Sea	69°10N, 57°51E	1993	Soft tissue	3				mg/kg ww					1.670	0.150						1
		Russia, Pechora Sea	69°50N, 54°00E	1993	Soft tissue	2				mg/kg ww					1.760	0.010						1
		Russia, Pechora Sea	68°50N, 50°43E	1993	Soft tissue	2				mg/kg ww					1.810	0.020						1
Zoobenthos		Russia, Kara Sea		1994						µg/g					8.700			0.2				14
		Russia, Ob-Yenisei costal waters		1994						µg/g		15.000		3.3-8.7				0.2				14
		Russia, Pechora Sea		1998						µg/g		1.600			0.470							15
		Russia, Pechora Sea		1999						µg/g	0.69-1.47			0.14-0.53								15

Remarks
A. Dry weight 69%;
B. Dry weight 44%;
C. Median value.

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10. Green *et al.*, 2001; 11. NOAA's Mussel Watch Programme; 12. Cantillo *et al.*, 1999, NOAA NS&T Program, unpublished data;
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Table A14. Metals in marine fish.

Species/location	Coordinate	Year	Tissue	n	Sex	Age, yrs	Size, cm (SD)	Unit	Lead		Cadmium			Mercury							Selenium			Remark	Reference
									min-max	arithm. mean	min-max	arithm. mean	SD	min-max	arithm. mean	SD	MeHg min-max	MeHg arithm. mean	Hg geo. mean	Hg GSD	min-max	arithm. mean	SD		
Amblyraja radiata (= *Raja radiata*) (thorny skate)																									
Russia, Barents Sea		1994	Liver					mg/kg dw				0.640	0.19		0.18	0.32									1
Russia, Barents Sea		1994	Muscle					mg/kg dw							0.72	0.7									1
Anarhichas denticulatus (jelly wolf-fish)																									
Russia, Barents Sea		1994	Liver					mg/kg dw				3.100	2.4		0.11	0.04									1
Russia, Barents Sea		1994	Muscle					mg/kg dw							0.47	0.17									1
Anarhichas lupus (wolf-fish)																									
Russia, Barents Sea		1994	Liver					mg/kg dw				4.300	2.2		0.13	0.1									1
Russia, Barents Sea		1994	Muscle					mg/kg dw							0.84	0.52									1
Anarhichas minor (spotted catfish)																									
Russia, Barents Sea		1994	Liver					mg/kg dw				2.400	1.1		0.15	0.16									1
Russia, Barrents Sea		1994	Muscle					mg/kg dw							0.57	0.38									1
Bathyraja spinicauda (spinytail skate)																									
Russia, Barents Sea		1994	Liver					mg/kg dw				<0.10		0.7-0.19											1
Russia, Barrents Sea		1994	Muscle					mg/kg dw						0.58-1.21											1
Boreogadus saida (polar cod or Arctic cod)																									
Russia, Pechora Sea		1997	Liver					mg/kg ww				0.485													1
Russia, Pechora Sea		1997	Muscle					mg/kg ww		0.037		0.006			0.02							0.23			1
Coregonus sardinella (least cisco)																									
USA, Alaska, Elson Lagoon, Barrow			Liver					µg/g ww							0.040									A	3
USA, Alaska, Elson Lagoon, Barrow			Muscle					µg/g ww							0.013									A	3
Cyclopterus lumpus (lumpsucker)																									
Russia, Barents Sea		1994	Liver					mg/kg dw			1.6-1.7			0.03-0.06											1
Russia, Barents Sea		1994	Muscle					mg/kg dw						0.02-0.42											1
Eleginus navaga (navaga)																									
Russia, Pechora Sea		??	Muscle					µg/g		0.133		0.023													1
Gadus morhua (Atlantic cod)																									
Faroe Islands	62°23'N, 7°30'W	1997	Liver	2 pools			59	mg/kg ww				0.180													4
Faroe Islands	62°23'N, 7°30'W	1997	Muscle	2 pools			59	mg/kg ww							0.0275										4
Faroe Islands	62°23'N, 7°30'W	1997	Liver	44				µg/g ww		<0.15		0.180													4
Faroe Islands	62°23'N, 7°30'W	1997	Muscle	44			59	µg/g ww							0.028										4
Greenland, Nuuk	64°10'N, 51°45'W	1999	Liver	5	F/M		38-42	mg/kg ww				0.050	0.024		0.007	0.002						0.779	0.146		5
Greenland, Nuuk	64°10'N, 51°45'W	1999	Muscle	9	F/M		35-43	mg/kg ww				<0.001			0.014	0.001						0.293	0.018		5
Iceland, northeast	66°N, 13.5'W	1995	Liver	2 pools			38	mg/kg ww				0.097	0.063												6
Iceland, northeast	66°N, 13°5'W	1995	Muscle	2 pools			38	mg/kg ww							0.023	0.002									6
Iceland, northeast	66°N, 13°5'W	1996	Liver	43			47.1	mg/kg ww				0.076	0.043												6
Iceland, northeast	66°N, 13°5'W	1996	Muscle	6			48.2	mg/kg ww							0.027	0.012									6
Iceland, northeast	66°N, 13°5'W	1997	Liver	11			40.2	mg/kg ww				0.111	0.028												7
Iceland, northeast	66°N, 13°5'W	1997	Muscle	2 pools			39.6	mg/kg ww							0.027	0.014									7
Iceland, northeast	66°N, 13°5'W	1998	Liver	6			37.5	mg/kg ww				0.095	0.038												7
Iceland, northeast	66°N, 13°5'W	1998	Muscle	2 pools			37	mg/kg ww							0.021±	0.01									7
Iceland, northeast	66°N, 13°5'W	1999	Liver	12			40.9	mg/kg ww				0.157	0.086												8
Iceland, northeast	66°N, 13°5'W	1999	Muscle	2 pools			39.5	mg/kg ww							0.02	0.004									8
Iceland, northwest	66°N, 24°5'W	1995	Liver	2 pools			40.1	mg/kg ww				0.352													6
Iceland, northwest	66°N, 24°5'W	1995	Muscle	1 pool			40.1	mg/kg ww							0.027										6
Iceland, northwest	66°N, 24°5'W	1996	Liver	5			37.2	mg/kg ww				0.575	0.200												6
Iceland, northwest	66°N, 24°5'W	1996	Muscle	1 pool			37.4	mg/kg ww							0.026										6
Iceland, northwest	66°N, 24°5'W	1997	Liver	5			40.1	mg/kg ww				0.342	0.071												7
Iceland, northwest	66°N, 24°5'W	1997	Muscle	1 pool			39.8	mg/kg ww							0.020										7
Iceland, northwest	66°N, 24°5'W	1998	Liver	8			41.9	mg/kg ww				0.454	0.481												8
Iceland, northwest	66°N, 24°5'W	1998	Muscle	1 pool			41.1	mg/kg ww							0.030										8
Iceland, northwest	66°N, 24°5'W	1999	Liver	5			39.8	mg/kg ww				0.445	0.210												9
Iceland, northwest	66°N, 24°5'W	1999	Muscle	1 pool			38.5	mg/kg ww							0.019										9
Iceland, southwest	64°05'N, 23°W	1995	Liver	1 pool			38.9	mg/kg ww				0.164													6
Iceland, southwest	64°05'N, 23°W	1995	Muscle	1 pool			38.9	mg/kg ww							0.022										6
Iceland, southwest	64°05'N, 23°W	1998	Liver	6			38.1	mg/kg ww				0.103	0.065												8
Iceland, southwest	64°05'N, 23°W	1998	Muscle	1 pool			37.4	mg/kg ww							0.028										8
Norway	68°12'E, 14°48'E	1992	Liver	pools				mg/kg ww				0.069												B	2
Norway	68°12'E, 14°48'E	1992	Muscle	pools			Small	mg/kg ww							0.067									B	2

Table A14. Metals in marine fish, continued.

Species/location	Coordinate	Year	Tissue	n	Sex	Age, yrs	Size, cm (SD)	Unit	Lead		Cadmium			Mercury						Selenium			Remark	Reference
									min-max	arithm. mean	min-max	arithm. mean	SD	min-max	arithm. mean	SD	MeHg min-max	MeHg arithm. mean	Hg geo. mean	Hg GSD	min-max	arithm. mean	SD		
Norway	68°12'E, 14°48'E	1992	Muscle	pools			Large	mg/kg ww							0.065								B	2
Norway	64°09'N, 09°53'E	1993	Liver	pools				mg/kg ww				0.036												
Norway	68°12'E, 14°48'E	1993	Liver	pools				mg/kg ww				0.15												
Norway	64°09'N, 09°53'E	1993	Muscle	pools			Small	mg/kg ww							0.046								B	2
Norway	64°09'N, 09°53'E	1993	Muscle	pools			Large	mg/kg ww							0.058								B	2
Norway	68°12'E, 14°48'E	1993	Muscle	pools			Small	mg/kg ww							0.054								B	2
Norway	68°12'E, 14°48'E	1993	Muscle	pools			Large	mg/kg ww							0.064								B	2
Norway	64°09'N, 09°53'E	1994	Liver	pools				mg/kg ww				0.029												
Norway	68°12'E, 14°48'E	1994	Liver	pools				mg/kg ww				0.025												
Norway	70°09'N, 20°23'E	1994	Liver	pools				mg/kg ww				0.17												
Norway	69°54'N, 29°30'E	1994	Liver	pools				mg/kg ww				0.23												
Norway	64°09'N, 09°53'E	1994	Muscle	pools			Small	mg/kg ww							0.079								B	2
Norway	64°09'N, 09°53'E	1994	Muscle	pools			Large	mg/kg ww							0.091								B	2
Norway	68°12'E, 14°48'E	1994	Muscle	pools			Small	mg/kg ww							0.069								B	2
Norway	68°12'E, 14°48'E	1994	Muscle	pools			Large	mg/kg ww							0.069								B	2
Norway	70°09'N, 20°23'E	1994	Muscle	pools			Small	mg/kg ww							0.065								B	2
Norway	70°09'N, 20°23'E	1994	Muscle	pools			Large	mg/kg ww							0.05								B	2
Norway	69°54'N, 29°30'E	1994	Muscle	pools			Small	mg/kg ww							0.044								B	2
Norway	69°54'N, 29°30'E	1994	Muscle	pools			Large	mg/kg ww							0.056								B	2
Norway	64°10'N, 9°53'W	1995	Liver	pools				mg/kg ww				0.022												
Norway	68°12'N, 14°48'W	1995	Liver	pools				mg/kg ww				0.110												
Norway	70°09'N, 21°22'W	1995	Liver	pools				mg/kg ww				0.180												
Norway	69°55'N, 29°30'W	1995	Liver	pools				mg/kg ww				0.190												
Norway	64°10'N, 9°53'W	1995	Muscle	pools			Small	mg/kg ww							0.08								B	2
Norway	64°10'N, 9°53'W	1995	Muscle	pools			Large	mg/kg ww							0.074								B	2
Norway	68°12'N, 14°48'W	1995	Muscle	pools			Small	mg/kg ww							0.069								B	2
Norway	68°12'N, 14°48'W	1995	Muscle	pools			Large	mg/kg ww							0.086								B	2
Norway	70°09'N, 21°22'W	1995	Muscle	pools			Small	mg/kg ww							0.054								B	2
Norway	70°09'N, 21°22'W	1995	Muscle	pools			Large	mg/kg ww							0.059								B	2
Norway	69°55'N, 29°30'W	1995	Muscle	pools			Small	mg/kg ww							0.034								B	2
Norway	69°55'N, 29°30'W	1995	Muscle	pools			Large	mg/kg ww							0.053								B	2
Norway	64°10'N, 9°53'W	1996	Liver	pools				mg/kg ww				0.066												
Norway	68°12'N, 14°48'W	1996	Liver	pools				mg/kg ww				0.330												
Norway	70°09'N, 21°22'W	1996	Liver	pools				mg/kg ww				0.097												
Norway	69°55'N, 29°30'W	1996	Liver	pools				mg/kg ww				0.095												
Norway	64°10'N, 9°53'W	1996	Muscle	pools			Small	mg/kg ww							0.077								B	2
Norway	64°10'N, 9°53'W	1996	Muscle	pools			Large	mg/kg ww							0.117								B	2
Norway	68°12'N, 14°48'W	1996	Muscle	pools			Small	mg/kg ww							0.04								B	2
Norway	68°12'N, 14°48'W	1996	Muscle	pools			Large	mg/kg ww							0.037								B	2
Norway	70°09'N, 21°22'W	1996	Muscle	pools			Small	mg/kg ww							0.047								B	2
Norway	70°09'N, 21°22'W	1996	Muscle	pools			Large	mg/kg ww							0.057								B	2
Norway	69°55'N, 29°30'W	1996	Muscle	pools			Small	mg/kg ww							0.029								B	2
Norway	69°55'N, 29°30'W	1996	Muscle	pools			Large	mg/kg ww							0.04								B	2
Norway	68°12'N, 14°48'W	1997	Liver	pools				mg/kg ww				0.064												
Norway	69°55'N, 29°30vW	1997	Liver	pools				mg/kg ww				0.130												
Norway	68°12'N, 14°48'W	1997	Muscle	pools			Small	mg/kg ww							0.095								B	2
Norway	68°12'N, 14°48'W	1997	Muscle	pools			Large	mg/kg ww							0.128								B	2
Norway	69°55'N, 29°30'W	1997	Muscle	pools			Small	mg/kg ww							0.011								B	2
Norway	69°55'N, 29°30'W	1997	Muscle	pools			Large	mg/kg ww							0.02								B	2
Norway	68°12'N, 14°48'W	1998	Liver	pools				mg/kg ww				0.047												
Norway	69°55'N, 29°30'W	1998	Liver	pools				mg/kg ww				0.120												
Norway	68°12'N, 14°48'W	1998	Muscle	pools			Small	mg/kg ww							0.066								B	2
Norway	68°12'N, 14°48'W	1998	Muscle	pools			Large	mg/kg ww							0.09								B	2
Norway	69°55'N, 29°30'W	1998	Muscle	pools			Small	mg/kg ww							0.014								B	2
Norway	69°55'N, 29°30'W	1998	Muscle	pools			Large	mg/kg ww							0.019								B	2
Norway	68°12'N, 14°48'W	1999	Liver	pools				mg/kg ww				0.039												
Norway	69°55'N, 29°30'W	1999	Liver	pools				mg/kg ww				0.140												
Norway	68°12'N, 14°48'W	1999	Muscle	pools			Small	mg/kg ww							0.047								B	2
Norway	68°12'N, 14°48'W	1999	Muscle	pools			Large	mg/kg ww							0.043								B	2
Norway	69°55'N, 29°30'W	1999	Muscle	pools			Small	mg/kg ww							0.017								B	2
Norway	69°55'N, 29°30'W	1999	Muscle	pools			Large	mg/kg ww							0.032								B	2
Russia, Barents Sea		1994	Liver					mg/kg dw			0.350	0.21			0.03	0.02								1
Russia, Barents Sea		1994	Muscle					mg/kg dw							0.26	0.14								1
Gadus ogac (Greenland cod)																								
Greenland, Nuuk	64.10N, 51.45W	1999	Muscle	5	F/M		29-45	mg/kg ww							0.02	0.01					0.209	0.023		5

Table A14. Metals in marine fish, continued.

Species/location	Coordinate	Year	Tissue	n	Sex	Age, yrs	Size, cm (SD)	Unit	Lead		Cadmium			Mercury							Selenium			Remark	Reference
									min-max	arithm. mean	min-max	arithm. mean	SD	min-max	arithm. mean	SD	MeHg min-max	MeHg arithm. mean	Hg geo. mean	Hg GSD	min-max	arithm. mean	SD		
Hippoglossoides elassodon (flathead sole)																									
USA, Alaska, Boca de Quadra	55.17N, 130.3W	1986	Liver	1-3	F			mg/kg dw				1.500	0.430		0.14										10
USA, Alaska, Boca de Quadra	55.17N, 130.3W	1986	Stomach cont.	1	F			mg/kg dw				0.210			0.07										10
USA, Alaska, Dutch Harbor	53.5N, 166.3W	1986	Liver	3	F			mg/kg dw				0.030	0.018		0.43	0.06									10
USA, Alaska, Dutch Harbor	53.5N, 166.3W	1986	Stomach cont.	1				mg/kg dw				0.550			0.46										10
USA, Alaska, Kamishak	59.1N, 153.4W	1986	Liver	3	F			mg/kg dw				1.560	1.130		0.27	0.18									10
USA, Alaska, Kamishak	59.1N, 153.4W	1986	Stomach cont.	1				mg/kg dw				4.880			0.14										10
USA, Alaska, Lutak Inlet	59.2N, 135.3W	1984	Liver	1-3	F			mg/kg dw				2.220	2.060		0.13										10
USA, Alaska, Lutak Inlet	59.2N, 135.3W	1986	Liver	3	F			mg/kg dw				1.170	0.420												10
USA, Alaska, Lutak Inlet	59.2N, 135.3W	1986	Stomach cont.	1				mg/kg dw				0.420			0.15										10
USA, Alaska, Nahku Bay	59.3N, 135.2W	1984	Liver	3	F			mg/kg dw				1.260	0.710												10
USA, Alaska, Nahku Bay	59.3N, 135.2W	1986	Liver	1	F			mg/kg dw				0.925													10
USA, Alaska, Port Moller	56.1N, 160.4W	1986	Liver	3	F			mg/kg dw				1.270	0.600		0.06	0.01									10
USA, Alaska, Port Moller	56.1N, 160.4W	1986	Stomach cont.	2	F			mg/kg dw				0.757	0.598		0.13	0.1									10
USA, Alaska, Port Valdez	61.1N, 146.2W	1986	Liver	3	F			mg/kg dw				2.010	0.510		0.14	0.09									10
USA, Alaska, Skagway	59.3N, 135.2W	1986	Liver	2	F			mg/kg dw				0.623	0.590		0.07	0									10
USA, Alaska, Skagway	59.3N, 135.2W	1986	Stomach cont.	1	F			mg/kg dw				1.210			0.09										10
Hippoglossoides platessoides (long rough dab)																									
Faroe Islands	61°41´N, 07°45´W	1996	Liver	21		5.6		µg/g ww		<0.15		0.410													11
Faroe Islands	62°53´N, 09°05´W	1996	Liver	34		5.6		µg/g ww		<0.15		0.400													11
Faroe Islands	61°41´N, 07°45´W	1996	Muscle	21		10.2		µg/g ww							0.05										11
Faroe Islands	62°53´N, 09°05´W	1996	Muscle	34		10.2		µg/g ww							0.12										11
Russia, Barents Sea		1994	Liver					mg/kg dw				1.600	1.7		0.07	0.06									1
Russia, Barents Sea		1994	Muscle					mg/kg dw							0.37	0.24									1
Hippoglossus stenolepis (Pacific halibut)																									
USA, Alaska, ATKA		1999 April	Muscle					ng/g ww							197.57										12
USA, Alaska, ATKA		1999 April	Muscle					ng/g ww										142.64							12
Limanda limanda (common dab)																									
Faroe Islands, Kirkjubøur	61°57'N, 6°47'00 W	1996/97	Liver	153	F/M	4.0	26.6	µg/g ww		<0.15		0.930													13
Faroe Island, Kirkjubøur	61°57'N, 6°47'00 W	1996/97	Muscle	153		4.0	26.6	µg/g ww							0.02										13
Iceland, NW of Iceland	66.6N, 18.0W	1995	Liver	1 pool			25.7	mg/kg ww				0.526													14
Iceland, NW of Iceland	66.6N, 18.0W	1995	Muscle	1 pool			25.7	mg/kg ww							0.022										14
Iceland, NW of Iceland	66.00N, 24.34W	1996	Liver	1 pool			28.3	mg/kg ww				0.876													7
Iceland, NW of Iceland	66.00N, 24.34W	1996	Muscle	1 pool			28.3	mg/kg ww							0.018										7
Iceland, NW of Iceland	66.00N, 24.34W	1997	Liver	1 pool			24.3	mg/kg ww				0.859													7
Iceland, NW of Iceland	66.00N, 24.34W	1997	Muscle	1 pool			24.3	mg/kg ww							0.021										7
Iceland, NW of Iceland	65.56N, 24.45W	1998	Liver	1 pool			29.6	mg/kg ww				0.684													8
Iceland, NW of Iceland	65.38N, 20.57W	1998	Liver	1 pool			28.6	mg/kg ww				0.971													8
Iceland, NW of Iceland	65.56N, 24.45W	1998	Muscle	1 pool			29.6	mg/kg ww							0.029										8
Iceland, NW of Iceland	65.38N, 20.57W	1998	Muscle	1 pool			28.6	mg/kg ww							0.063										8
Iceland, NW of Iceland	66.18N, 23.57W	1999	Liver	1 pool			28.2	mg/kg ww				0.810													9
Iceland, NW of Iceland	65.35N, 21.03W	1999	Liver	1 pool			28.5	mg/kg ww				1.130													9
Iceland, NW of Iceland	66.18N, 23.57W	1999	Muscle	1 pool			28.2	mg/kg ww							0.019										9
Iceland, NW of Iceland	65.35N, 21.03W	1999	Muscle	1 pool			28.5	mg/kg ww							0.063										9
Iceland, south	63.47N, 21.24W	1996	Liver	1 pool			27.7	mg/kg ww				0.503													7
Iceland, south	63.47N, 21.24W	1996	Muscle	1 pool			27.7	mg/kg ww							0.053										7
Iceland, south	63.48N, 21.08W	1997	Liver	1 pool			28.7	mg/kg ww				0.240													7
Iceland, south	63.48N, 21.08W	1997	Muscle	1 pool			28.7	mg/kg ww							0.051										7
Iceland, south	63.21N, 19.10W	1998	Liver	1 pool			30.7	mg/kg ww				0.140													8
Iceland, south	63.21N, 19.10W	1998	Muscle	1 pool			30.7	mg/kg ww							0.043										8
Iceland, south	63.33N, 20.19W	1999	Liver	1 pool			24.6	mg/kg ww				0.453													9
Iceland, south	63.33N, 20.19W	1999	Muscle	1 pool			24.6	mg/kg ww							0.043										9
Iceland, southeast	64N, 15.5W	1995	Liver	1 pool			28.8	mg/kg ww				0.470													14
Iceland, southeast	64N, 15.5W	1995	Muscle	1 pool			28.8	mg/kg ww							0.054										14
Iceland, southeast	64.13N, 14.5W	1996	Liver	1 pool			24.3	mg/kg ww				0.152													7
Iceland, southeast	64.13N, 14.5W	1996	Muscle	1 pool			24.3	mg/kg ww							0.019										7
Iceland, southeast	64.13N, 14.50W	1999	Liver	1 pool			25.5	mg/kg ww				0.421													9
Iceland, southeast	64.13N, 14.50W	1999	Muscle	1 pool			25.5	mg/kg ww							0.030										9
Iceland, southwest	64.5N, 23W	1995	Liver	1 pool			26.5	mg/kg ww				0.295													14
Iceland, southwest	64.5N, 23W	1995	Muscle	1 pool			26.5	mg/kg ww							0.070										14
Norway	68°12'N, 14°48'E	1994	Liver					mg/kg ww				0.98												B	2
Norway	68°12'N, 14°48'E	1995	Liver					mg/kg ww				0.18												B	2
Norway	68°12'N, 14°48'E	1996	Liver					mg/kg ww				0.23												B	2
Russia, Barents Sea		1994	Liver					mg/kg dw			1.7-2.8			0.16-0.21											1
Russia, Barents Sea		1994	Muscle					mg/kg dw						0.15-0.31											1

Table A14. Metals in marine fish, continued.

Species/location	Coordinate	Year	Tissue	n	Sex	Age, yrs	Size, cm (SD)	Unit	Lead		Cadmium			Mercury							Selenium			Remark	Reference
									min-max	arithm. mean	min-max	arithm. mean	SD	min-max	arithm. mean	SD	MeHg min-max	MeHg arithm. mean	Hg geo. mean	Hg GSD	min-max	arithm. mean	SD		
Mallotus villosus (capelin)																									
Greenland, Nuuk	64.10N, 51.45W	1999	Muscle	20	M		15.8-19.9	mg/kg ww				0.003	0.001		0.009	0.002						0.163	0.021		5
Greenland, Nuuk	64.10N, 51.45W	1999	Whole fish	10	M		14.3-18.7	mg/kg ww							0.007	0.002						0.131	0.014		5
Melanogrammus aeglefinus (haddock)																									
Russia, Barrents Sea		1994	Liver					mg/kg dw				1.400	3.1		0.05	0.05									1
Russia, Barrents Sea		1994	Muscle					mg/kg dw							0.33	0.37									1
Myoxocephalus quadricornis (fourhorn sculpin)																									
USA, Alaska, Endicott	70.21N, 147.6W	1985	Liver	3	F			mg/kg dw				1.010	0.190		0.47	0.13						6.29	2.13		10
USA, Alaska, Endicott	70.21N, 147.6W	1986	Liver	3	F			mg/kg dw				0.510	0.350									6.53	2.95		10
USA, Alaska, Endicott	70.21N, 147.6W	1986	Stomach cont.	1	F			mg/kg dw				0.657										1.2			10
USA, Alaska, Oliktok	70.30N, 149.5W	1985	Liver	3	F			mg/kg dw				0.276	0.045		0.4	0.28						3.70	1.44		10
USA, Alaska, Oliktok	70.30N, 149.5W	1986	Liver	3	F			mg/kg dw				0.413	0.241		0.46	0.26						6.10	2.43		10
Myoxocephalus scorpius (shorthorn sculpin)																									
Faroe Islands	62.04N, 6.54W	1999	Liver	2 pools			10-15	mg/kg ww				0.190	0.020		0.04	0						1.15	0.09		15
Faroe Islands	62.04N, 6.54W	1999	Liver	2 pools			15-20	mg/kg ww				0.150	0.010		0.13	0.08						1.47	0.19		15
Faroe Islands	62.04N, 6.54W	1999	Liver	4 pools			20-25	mg/kg ww				0.200	0.210		0.29	0.18						1.28	0.2		15
Faroe Islands	62.04N, 6.54W	1999	Liver	8			25-31	mg/kg ww				0.500	0.440		0.55	0.55						1.18	0.22		15
Faroe Islands	62.03N, 6.50W	2000	Liver	5 pools			20-25	mg/kg ww				0.384	0.273		1.38	0.821						1.318	0.215		15
Faroe Islands	62.03N, 6.50W	2000	Liver	11			25-30	mg/kg ww				0.946	0.938		0.988	0.539						1.54	0.61		15
Greenland, Ittoqqortoormiit	70.13N, 22.00W	1999	Liver	20	F/M			mg/kg ww				1.030	0.700		0.049	0.026						1.05	0.23		5
Greenland, Ittoqqortoormiit	70.13N, 22.00W	2000	Liver	13	F		24.8	mg/kg ww				0.750	0.230		0.061	0.048						0.89	0.2		5
Greenland, Ittoqqortoormiit	70.13N, 22.00W	2000	Liver	5	M		21.6	mg/kg ww				1.100	0.570		0.045	0.006						1.16	0.23		5
Greenland, Qeqertarsuaq	61N, 53W	1994	Liver	25				mg/kg dw				1.700	1.400		0.0266	0.0200						0.995	0.609	C	5
Greenland, Qeqertarsuaq	61N, 53W	1999	Liver	15	F		27.3	mg/kg ww				0.662	0.360		0.014	0.006						0.88	0.15		5
Greenland, Qeqertarsuaq	61N, 53W	1999	Liver	4	M		23.3	mg/kg ww				1.260	0.590		0.008	0.001						0.88	0.05		5
Greenland, Qeqertarsuaq	61N, 53W	1999	Liver	20				mg/kg dw				0.818	0.479		0.0162	0.0123						0.89	0.545	C	5
Greenland, Qeqertarsuaq	61N, 53W	2000	Liver	14	F		29.3	mg/kg ww				1.220	1.020		0.014	0.005						1.13	0.23		5
Greenland, Qeqertarsuaq	61N, 53W	2000	Liver	6	M		22.8	mg/kg ww				1.400	0.410		0.011	0.005						1.17	0.12		5
Greenland, Qeqertarsuaq	61N, 53W	2000	Liver	20				mg/kg dw				1.280	0.880		0.0313	0.0246						1.09	0.67	C	5
Oncorhynchus gorbuscha (pink salmon)																									
USA, Alaska, Elson Lagoon, Barrow			Liver					µg/g ww							0.035									A	3
USA, Alaska, Elson Lagoon, Barrow			Muscle					µg/g ww							0.025									A	3
Oncorhynchus keta (chum salmon)																									
Russia, Chukotak, Lavrentiya			Liver	11	F			µg/g ww	0.290			0.725			0.360										16
Russia, Chukotak, Lavrentiya			Liver	9	M			µg/g ww	0.245			0.634			0.409										16
Russia, Chukotak, Lavrentiya			Muscle	11	F			µg/g ww	0.070			0.196			0.184										16
Russia, Chukotak, Lavrentiya			Muscle	9	M			µg/g ww	0.086			0.140			0.153										16
USA, Alaska Western River		1999, 2000	Eggs	6			57.5(1.3)	ng/g ww						4.3-7.8											12
USA, Alaska Western River		1999, 2000	Liver	36			59.5 (3.2)	ng/g ww						42.0-138.0											12
USA, Alaska Western River		1999, 2000	Liver	36			59.5 (3.2)	ng/g ww								18.4-76.7									12
USA, Alaska Western River		1999, 2000	Muscle	36			59.5 (3.2)	ng/g ww						38.1-106.3											12
USA, Alaska Western River		1999, 2000	Muscle	36			59.5 (3.2)	ng/g ww								23.2-87.9									12
USA, Alaska, Kuskokwim		1999	Muscle	6				µg/g ww									0.044	0.0120	0.058	0.014					17
USA, Alaska, Kuskokwim		2000	Muscle	6				µg/g ww									0.060	0.0182	0.074	0.015					17
USA, Alaska, Kuskokwim		1999	Liver	6				µg/g ww									0.043	0.0134	0.066	0.0102					17
USA, Alaska, Kuskokwim		2000	Liver	6				µg/g ww									0.047	0.0099	0.072	0.0090					17
USA, Alaska, Nushagak		1999	Muscle	6				µg/g ww									0.058	0.0182	0.072	0.018					17
USA, Alaska, Nushagak		2000	Muscle	6				µg/g ww									0.054	0.0177	0.073	0.018					17
USA, Alaska, Nushagak		1999	Liver	6				µg/g ww									0.042	0.0114	0.064	0.0116					17
USA, Alaska, Nushagak		2000	Liver	6				µg/g ww									0.046	0.0131	0.069	0.0141					17
USA, Alaska, Yukon River		1999	Muscle	6				µg/g ww									0.056	0.0237	0.068	0.023					17
USA, Alaska, Yukon River		2000	Muscle	6				µg/g ww									0.064	0.0060	0.084	0.0113					17
USA, Alaska, Yukon River		1999	Liver	6				µg/g ww									0.041	0.0173	0.071	0.0371					17
USA, Alaska, Yukon River		2000	Liver	6				µg/g ww									0.058	0.0104	0.095	0.0167					17
Oncorhynchus kisutch (coho salmon)																									
USA, Alaska Western River		1999, 2000	Eggs	5			56.0 (2.1)	ng/g ww						6.4-8.5											12
USA, Alaska Western River		1999, 2000	Liver	35			56.0 (3.2)	ng/g ww						32-172											12
USA, Alaska Western River		1999, 2000	Liver	35			56.0 (3.2)	ng/g ww								16-116									12
USA, Alaska Western River		1999, 2000	Muscle	35			56.0 (3.2)	ng/g ww						37.3-76.4											12
USA, Alaska Western River		1999, 2000	Muscle	35			56.0 (3.2)	ng/g ww								20.6-56.7									12
USA, Alaska, Kuskokwim		1999	Muscle	6				µg/g ww									0.039	0.0059	0.049	0.006					17
USA, Alaska, Kuskokwim		2000	Muscle	6				µg/g ww									0.038	0.0127	0.057	0.014					17

Table A14. Metals in marine fish, continued.

Species/location	Coordinate	Year	Tissue	n	Sex	Age, yrs	Size, cm (SD)	Unit	Lead		Cadmium			Mercury							Selenium			Remark	Reference
									min-max	arithm. mean	min-max	arithm. mean	SD	min-max	arithm. mean	SD	MeHg min-max	MeHg arithm. mean	Hg geo. mean	Hg GSD	min-max	arithm. mean	SD		
USA, Alaska, Kuskokwim		1999	Liver	6				µg/g ww										0.062	0.0169	0.094	0.0236				17
USA, Alaska, Kuskokwim		2000	Liver	6				µg/g ww										0.069	0.0342	0.104	0.0314				17
USA, Alaska, Kvichak		1999	Muscle	5				µg/g ww										0.041	0.0062	0.047	0.0037				17
USA, Alaska, Kvichak		1999	Liver	5				µg/g ww										0.036	0.0134	0.075	0.0141				17
USA, Alaska, Nushagak		2000	Muscle	6				µg/g ww										0.042	0.0074	0.059	0.011				17
USA, Alaska, Nushagak		2000	Liver	6				µg/g ww										0.072	0.0186	0.112	0.0403				17
USA, Alaska, Yukon River		1999	Muscle	6				µg/g ww										0.036	0.0059	0.044	0.0156				17
USA, Alaska, Yukon River		2000	Muscle	6				µg/g ww										0.042	0.0104	0.058	0.012				17
USA, Alaska, Yukon River		1999	Liver	6				µg/g ww										0.050	0.0142	0.087	0.0160				17
USA, Alaska, Yukon River		2000	Liver	6				µg/g ww										0.029	0.0124	0.054	0.0140				17
Oncorhynchus nerka (sockeye salmon)																									
USA, Alaska Western River		1999, 2000	Eggs	3			54.6 (4.7)	ng/g ww						3.4-5.1											12
USA, Alaska Western River		1999, 2000	Liver	30			55.5 (5.0)	ng/g ww						38-165											12
USA, Alaska Western River		1999, 2000	Liver	30			55.5 (5.0)	ng/g ww								22-99									12
USA, Alaska Western River		1999, 2000	Muscle	30			55.5 (5.0)	ng/g ww						25.2-74.2											12
USA, Alaska Western River		1999, 2000	Muscle	30			55.5 (5.0)	ng/g ww								9.3-63.2									12
USA, Alaska, Kuskokwim		1999	Muscle	6				µg/g ww										0.023	0.0032	0.034	0.006				17
USA, Alaska, Kuskokwim		2000	Muscle	6				µg/g ww										0.033	0.0067	0.061	0.006				17
USA, Alaska, Kuskokwim		1999	Liver	6				µg/g ww										0.036	0.0125	0.058	0.0158				17
USA, Alaska, Kuskokwim		2000	Liver	6				µg/g ww										0.066	0.0185	0.100	0.0270				17
USA, Alaska, Kvichak		2000	Muscle	6				µg/g ww										0.046	0.0123	0.058	0.011				17
USA, Alaska, Kvichak		2000	Liver	6				µg/g ww										0.051	0.0165	0.094	0.0153				17
USA, Alaska, Nushagak		1999	Muscle	6				µg/g ww										0.027	0.0127	0.038	0.007				17
USA, Alaska, Nushagak		2000	Muscle	6				µg/g ww										0.044	0.0064	0.061	0.006				17
USA, Alaska, Nushagak		1999	Liver	6				µg/g ww										0.049	0.0160	0.084	0.0168				17
USA, Alaska, Nushagak		2000	Liver	6				µg/g ww										0.069	0.0255	0.105	0.0403				17
Oncorhynchus tshawytscha (chinook Salmon)																									
USA, Alaska Western River		1999, 2000	Eggs	8			88.8 (3.1)	ng/g ww						6.4-15.3											12
USA, Alaska Western River		1999, 2000	Liver	33			76.8 (12.6)	ng/g ww								17.7-117.8									12
USA, Alaska Western River		1999, 2000	Muscle	33			76.7 (13.3)	ng/g ww						29.1-137.4											12
USA, Alaska Western River		1999, 2000	Muscle	33			76.7 (13.3)	ng/g ww								15.7-120.3									12
USA, Alaska, Kuskokwim		1999	Muscle	6				µg/g ww										0.078	0.0276	0.096	0.0304				17
USA, Alaska, Kuskokwim		2000	Muscle	6				µg/g ww										0.059	0.0250	0.08	0.0261				17
USA, Alaska, Kuskokwim		1999	Liver	6				µg/g ww										0.076	0.0232	0.107	0.0360				17
USA, Alaska, Kuskokwim		2000	Liver	6				µg/g ww										0.049	0.0187	0.079	0.0250				17
USA, Alaska, Nushagak		1999	Muscle	6				µg/g ww										0.078	0.0282	0.092	0.029				17
USA, Alaska, Nushagak		2000	Muscle	6				µg/g ww										0.043	0.0191	0.06	0.02				17
USA, Alaska, Nushagak		1999	Liver	6				µg/g ww										0.059	0.0292	0.099	0.0354				17
USA, Alaska, Nushagak		2000	Liver	6				µg/g ww										0.039	0.0164	0.077	0.0246				17
USA, Alaska, Yukon		1999	Muscle	6				µg/g ww										0.039	0.0271	0.05	0.035				17
USA, Alaska, Yukon		2000	Muscle	6				µg/g ww										0.059	0.0243	0.07	0.027				17
USA, Alaska, Yukon		1999	Liver	6				µg/g ww										0.037	0.0398	0.060	0.0450				17
USA, Alaska, Yukon		2000	Liver	6				µg/g ww										0.068	0.0276	0.103	0.0340				17
Pleuronectes glacialis (= *Liopsetta glacialis*) (Arctic flounder)																									
USA, Alaska, Oliktok	70.30N, 149.5W	1985	Stomach cont.	2	F			mg/kg dw			0.263	0.117		0.34	0.2										10
USA, Alaska, Oliktok	70.30N, 149.5W	1986	Stomach cont.	1	F			mg/kg dw			0.599			0.11											10
Pleuronectes platessa (plaice)																									
Russia, Barents Sea		1994	Liver					mg/kg dw			2.100	2.2		0.18	0.12										1
Russia, Barents Sea		1994	Muscle					mg/kg dw						0.24	0.18										1
Pollachius virens (saithe)																									
Russia, Barents Sea		1994	Liver					mg/kg dw			0.230	0.19		<0.02											1
Russia, Barents Sea		1994	Muscle					mg/kg dw						0.32	0.31										1
Rajella fyllae (= *Raja fyllae*) (round skate)																									
Russia, Barrents Sea		1994	Liver					mg/kg dw			8.100	3.5		0.45	0.3										1
Russia, Barrents Sea		1994	Muscle					mg/kg dw						1.35	0.19										1
Reinhardtius hippoglossoides (Greenland halibut)																									
Greenland, Nuuk	64.10N, 51.45W	1999	Liver	5	F/M		56-105	mg/kg ww						0.151	0.151							1.49	0.76		5
Greenland, Nuuk	64.10N, 51.45W	1999	Muscle	10	F/M		54-105	mg/kg ww			<0.002			0.154	0.134							0.313	0.116		5
Russia, Barents Sea		1994	Liver					mg/kg dw			1.200	0.7		0.05	0.01										1
Russia, Barents Sea		1994	Muscle					mg/kg dw						0.1	0.04										1

Table A14. Metals in marine fish, continued.

Species/location	Coordinate	Year	Tissue	n	Sex	Age, yrs	Size, cm (SD)	Unit	Lead		Cadmium			Mercury						Selenium			Remark	Reference
									min-max	arithm. mean	min-max	arithm. mean	SD	min-max	arithm. mean	SD	MeHg min-max	MeHg arithm. mean	Hg geo. mean	Hg GSD	min-max	arithm. mean	SD		
Salmo salar (Atlantic salmon)																								
Greenland, Nuuk	64.10N, 51.45W	1999	Liver	10	F/M		61-73	mg/kg ww				0.190	0.040		0.043	0.006						7.93	1.27		5
Greenland, Nuuk	64.10N, 51.45W	1999	Muscle	20	F/M		61-73	mg/kg ww				0.003	0.001		0.04	0.01						0.269	0.03		5
Salvelinus alpinus (Arctic char (sea run))																								
Canada, Hopedale, Labrador		1999	Muscle	7		6	40.6	µg/g ww	0.001-0.010	0.002	0.001-0.002	0.001		0.01-0.04	0.03						0.32-0.41	0.37			18
Canada, Labrador , Hopedale		1998	Muscle	13		7		mg/kg ww							0.033	0.005									19
Canada, Labrador, Nain		1998	Muscle	14		8		mg/kg ww							0.035	0.011									19
Canada, Makkovik, Labrador	55°05'N, 59°11'W	1999	Muscle	9		6	42.4	µg/g ww	0.001-0.012	0.005	0.001-0.006	0.002		0.02-0.05	0.03						0.29-0.43	0.35			18
Canada, Nain, Labrador	56°32'N, 61°41'W	1999	Muscle	10		7	48.8	µg/g ww		0.001	0.001-0.001	0.001		0.02-0.05	0.03						0.31-0.51	0.38			18
Canada, Quebec, Kangiqsuk		1998	Muscle	15		8		mg/kg ww							0.032	0.026						0.235	0.069		19
Canada, Quebec, Kangiqsuk		1998	Muscle	2		12		mg/kg ww							0.037							0.26			19
Canada, Quebec, Povungnituk		1998	Muscle	11		8		mg/kg ww							0.044	0.009									19
Canada, Quebec, Quaqtaq		1998	Muscle	14		9		mg/kg ww							0.072	0.035									19
Canada, Quebec, Tasuijaq		1998	Muscle	15		9		mg/kg ww							0.04	0.011						0.144	0.023		19
Greenland, Nuuk	64.10N, 51.45W	1999	Liver	4	F/M		36-40	mg/kg ww				0.092	0.035		0.018	0.004						1.48	0.29		5
USA, Alaska, Elson Lagoon, Barrow	71°18'N, 156°30'W		Liver					µg/g ww							0.017									A	3
USA, Alaska, Elson Lagoon, Barrow	71°18'N, 156°30'W		Muscle					µg/g ww							0.013									A	3

Table A13. Metals in marine fish, continued.

Species/location	Coordinate	Year	Tissue	n	Sex	Age, yrs	Size, cm (SD)	Unit	Lead		Cadmium			Mercury						Selenium			Remark	Reference
									min-max	arithm. mean	min-max	arithm. mean	SD	min-max	arithm. mean	SD	MeHg min-max	MeHg arithm. mean	Hg geo. mean	Hg GSD	min-max	arithm. mean	SD		
Sebastes marinus (ocean perch)																								
Russia, Barents Sea		1994	Liver					mg/kg dw				1.200	0.8		0.06	0.08									1
Russia, Barents Sea		1994	Muscle					mg/kg dw							0.35	0.4									1

Remarks

A. Hg is approx. 100% as MeHg;
B. Media value;
C. Appendix Table 5.2.

References

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10. Meador *et al.*, 1994;
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16. Melnikov et al., 2002;
17. Zhang *et al.*, 2001;
18. Muir *et al.*, 2000;
19. Muir *et al.*, 1999b.

Table A15. Metals in marine birds.

Species/location	Coordinate	Year	Tissue	n	Sex	Age (yr)	Unit	Lead min-max	Lead arithm. mean	Lead SD	Cadmium min-max	Cadmium arithm. mean	Cadmium SD	Mercury min-max	Mercury arithm. mean	Mercury SD	Selenium min-max	Selenium arithm. mean	Selenium SD	Remark	Ref-erence
Alca torda (razorbill)																					
Norway, Hornøy	70.3N, 31E	1992-93	Egg	10			mg/kg ww								0.18	0.04					1
Russia, Seven Islands	68N, 38E	1992	Liver	5	F/M		mg/kg dw					2.08	0.65		1.71	0.42		18.5	8.9		2
Russia, Seven Islands	68N, 38E	1992	Muscle	5	F/M		mg/kg dw					0.32	0.17		0.88	0.19		7.68	2.61		2
Anas acuta (northern pintail)																					
Russia, Chaun	68N, 170W	1993	Feather	1	M		mg/kg dw								0.33						3
Russia, Chaun	68N, 170W	1993	Kidney	1	M		mg/kg dw					4.65			2.32						3
Russia, Chaun	68N, 170W	1993	Liver	1	M		mg/kg dw					1.22			2.8						3
Russia, Chaun	68N, 170W	1993	Muscle	1	M		mg/kg dw								0.31						3
Cepphus grylle (black guillemot)																					
Canada, Nuvuk Island	62.14N, 75.38W	1993	Egg	12			mg/kg dw								1.5	0.3		2.2	0.3		4
Canada, Prince Leopold Island	74.02N, 90.05W	1993	Egg	12			mg/kg dw								2.3	0.3		2.2	0.3		4
Canada, Prince Leopold Island	74°02'N, 90°05'W	1998	Egg	3			mg/kg dw								1.58			1.56			5
Canada, Prince Leopold Island	74°02'N, 90°05'W	1998	Egg	3			mg/kg dw								1.59			1.91			5
Canada, Prince Leopold Island	74°02'N, 90°05'W	1998	Egg	3			mg/kg dw								1.58			1.43			5
Canada, Prince Leopold Island	74°02'N, 90°05'W	1998	Egg	3			mg/kg dw								2.21			1.9			5
Canada, Prince Leopold Island	74°02'N, 90°05'W	1998	Egg	3			mg/kg dw								2.05			1.65			5
Canada, Walrus Island	63°12'N, 83°40'W	1993	Egg	15			mg/kg dw								1.5	0.3		2.7	0.3		4
Faroe Islands, Hestur og Sveipur		1995	Liver	11		Juveniles	mg/kg ww					0.64	0.21		0.33	0.09					6
Faroe Islands, Hestur og Sveipur		1996	Liver	4	F		mg/kg ww					1.36	0.17		0.95	0.26					6
Faroe Islands, Hestur og Sveipur		1996	Liver	13		Juveniles	mg/kg ww					0.98	0.76		0.54	0.26					6
Faroe Islands, Hestur og Sveipur		1995/96	Liver	5	M		mg/kg ww					1.27	0.42		1.73	1.26					6
Faroe Islands, Koltur	61.59N, 06.58W	1999	Egg	10			mg/kg ww								0.51	0.19				A	6
Faroe Islands, Koltur	61.59N, 06.58W	2000	Egg	10			mg/kg ww								0.36	0.11				A	6
Faroe Islands, Koltur	61.59N, 06.58W	2001	Egg	10			mg/kg ww							0.182-0.449	0.324	0.09					7
Faroe Islands, Skúvoy	61.47N, 06.48W	1999	Egg	8			mg/kg ww								0.51	0.12				A	6
Faroe Islands, Skúvoy	61.47N, 06.48W	2000	Egg	9			mg/kg ww								0.3	0.10				A	6
Faroe Islands, Skúvoy	61.47N, 06.48W	2001	Egg	10			mg/kg ww							0.190-0.437	0.326	0.085				A	7
Greenland, Ittoqqortoormiit	70.13N, 22.00W	1999	Egg	10			mg/kg ww					0.0013	0.0005		0.34	0.06		0.45	0.05		8
Greenland, Qeqertarsuaq	61N, 53W	1999	Kidney	3		Juvenile	mg/kg ww					22.0	6.1		0.54	0.29		3.74	0.68		8
Greenland, Qeqertarsuaq	61N, 53W	1999	Kidney	15		Adult	mg/kg ww					15.0	10.4		0.51	0.16		4.13	0.68		8
Greenland, Qeqertarsuaq	61N, 53W	1999	Liver	3		Juvenile	mg/kg ww					0.037	1.33		1.02	0.14		3.7	0.7		8
Greenland, Qeqertarsuaq	61N, 53W	1999	Liver	15		Adult	mg/kg ww					1.85	1.19		0.61	0.19		2.8	0.6		8
Greenland, Qeqertarsuaq	61N, 53W	2000	Egg	7			mg/kg ww								0.26	0.06		0.50	0.02		8
Greenland, Qeqertarsuaq	61N, 53W	2000	Liver	20		Adult	mg/kg ww					3.72	2.27		0.66	0.26		3.21	0.56		8
Russia, Franz Josef Land, Hooker		1992	Liver	2			mg/kg dw					1.04	0.21		0.76	0.25		4.90	0.50		2
Russia, Franz Josef Land, Hooker		1992	Muscle	2			mg/kg dw					0.34	0.09		0.31	0.08		2.10	0.16		2
Russia, Franz Josef Land, Northbrook		1992	Liver	1	M		mg/kg dw					5.24			1.12			9.50			2
Russia, Franz Josef Land, Northbrook		1992	Muscle	1	M		mg/kg dw					0.34			0.43			4.20			2
Russia, Franz Josef Land, Stolichky		1992	Liver	2	F		mg/kg dw					5.89	0.69		0.81	0.03		8.30	0.7		2
Russia, Franz Josef Land, Stolichky		1992	Muscle	2	M		mg/kg dw					0.5	0.14		0.27			2.30	0.3		2
Clangula hyemalis (long-tailed duck)																					
Russia, Chaun	68N,170W	1993	Feather	8	M		mg/kg dw					0.62	0.59		1.96	3.31					3
Russia, Chaun	68N,170W	1993	Kidney	8	M		mg/kg dw					79.5	31.9		3.69	5.53					3
Russia, Chaun	68N,170W	1993	Liver	8	M		mg/kg dw					12.7	6.2		27.1	69.5					3
Russia, Chaun	68N,170W	1993	Muscle	8	M		mg/kg dw					0.42	0.28		1.14	1.64					3
Russia, Chukotka, Kanchalan		2001	Liver	1	F	1-2	µg/g ww	0.331				0.289			0.1150						9
Russia, Chukotka, Kanchalan		2001	Muscle	1	F	1-2	µg/g ww	0.219				0.013			0.0290						9
Russia, Chukotka, Kanchalan		2001	Liver	1	M	1-2	µg/g ww	0.299				0.254			0.1420						9
Russia, Chukotka, Kanchalan		2001	Muscle	1	M	1-2	µg/g ww	0.244				0.018			0.0260						9
Russia, Chukotka, Lavrentiya		2001	Liver	7	F	1-3	µg/g ww	0.448				0.317			0.2200						9
Russia, Chukotka, Lavrentiya		2001	Muscle	7	F	1-3	µg/g ww	10.773*				0.017			0.0500					B	9
Russia, Chukotka, Lavrentiya		2001	Liver	2	M	1-2	µg/g ww	0.438				0.281			0.1820						9
Russia, Chukotka, Lavrentiya		2001	Muscle	2	M	1-2	µg/g ww	0.128				0.015			0.0520						9
Russia, Taymir Peninsula, Khatanga		2001	Liver	4	F	1-2	µg/g ww	0.311				0.197			0.3860						9
Russia, Taymir Peninsula, Khatanga		2001	Muscle	4	F	1-2	µg/g ww	0.565				0.049			0.0850						9
Russia, Taymir Peninsula, Khatanga		2001	Liver	2	M	1-2	µg/g ww	0.290				0.125			0.2970						9
Russia, Taymir Peninsula, Khatanga		2001	Muscle	2	M	1-2	µg/g ww	0.215				0.096			0.0630						9
Fratercula arctica (Atlantic puffin)																					
Norway, Hornøy, Finmark	70.3N, 31E	1992-93	Egg	10			mg/kg ww								0.22	0.01					1
Norway, Lofoten	68N, 13E	1992-93	Egg	10			mg/kg ww								0.06	0.02					1
Norway, Nord Norland	69.3N, 18E	1992-93	Egg	10			mg/kg ww								0.10	0.02					1
Russia, Seven Island, Kharlov	68N, 38E	1992	Liver	3	F/M		mg/kg dw					3.22	1.35		1.37	0.12		27.1	4.0		2
Russia, Seven Island, Kharlov	68N, 38E	1992	Muscle	3	F/M		mg/kg dw					0.25	0.04		0.44	0.1		15.5	3.0		2
Russia, Seven Island, Kuvshim	68N, 38E	1992	Liver	4	M		mg/kg dw					2.68	1.30		1.17	0.23		14.0	2.3		2
Russia, Seven Island, Kuvshim	68N, 38E	1992	Muscle	4	M		mg/kg dw					0.32	0.14		0.09	0.45		1.38	10.4		2

Table A15. Metals in marine birds, continued

Species/location	Coordinate	Year	Tissue	n	Sex	Age (yr)	Unit	Lead min-max	Lead arithm. mean	Lead SD	Cadmium min-max	Cadmium arithm. mean	Cadmium SD	Mercury min-max	Mercury arithm. mean	Mercury SD	Selenium min-max	Selenium arithm. mean	Selenium SD	Remark	Reference
Fulmarus glacialis (northern fulmar)																					
Canada, Prince Leopold Island	74.02N, 90.05W	1993	Egg	3			µg/g dw								1.51			4.53			10
Canada, Prince Leopold Island	74.02N, 90.05W	1993	Egg	3			µg/g dw								1.09			4.40			10
Canada, Prince Leopold Island	74.02N, 90.05W	1993	Egg	3			µg/g dw								0.93			3.57			10
Canada, Prince Leopold Island	74.02N, 90.05W	1993	Egg	3			µg/g dw								1.20			3.22			10
Canada, Prince Leopold Island	74.02N, 90.05W	1993	Egg	3			µg/g dw								1.20			4.35			10
Canada, Prince Leopold Island	74.02N, 90.05W	1998	Egg	3			µg/g dw								1.74			2.99			10
Canada, Prince Leopold Island	74.02N, 90.05W	1998	Egg	3			µg/g dw								1.20			3.15			10
Canada, Prince Leopold Island	74.02N, 90.05W	1998	Egg	3			µg/g dw								1.05			3.02			10
Canada, Prince Leopold Island	74.02N, 90.05W	1998	Egg	3			µg/g dw								1.63			4.27			10
Canada, Prince Leopold Island	74.02N, 90.05W	1998	Egg	3			µg/g dw								1.18			3.28			10
Canada, Prince Leopold Island, Lancaster Sound	74°02'N, 90°05'W	1975	Egg	15(5)			mg/kg dw								0.86	0.07		4.45	0.15	C	11
Canada, Prince Leopold Island, Lancaster Sound	74°02'N, 90°05'W	1976	Egg	12(4)			mg/kg dw								1.06	0.17		4.17	0.19	C	11
Canada, Prince Leopold Island, Lancaster Sound	74°02'N, 90°05'W	1977	Egg	15(5)			mg/kg dw								0.74	0.07		4.27	0.18	C	11
Canada, Prince Leopold Island, Lancaster Sound	74°02'N, 90°05'W	1987	Egg	6(2)			mg/kg dw								1.13			4.09		C	11
Canada, Prince Leopold Island, Lancaster Sound	74°02'N, 90°05'W	1993	Egg	15(5)			mg/kg dw								1.19	0.09		4.01	0.26	C	11
Canada, Prince Leopold Island, Lancaster Sound	74°02'N, 90°05'W	1998	Egg	15(5)			mg/kg dw								1.36	0.14		3.34	0.24	C	11
Faroe Islands, Nolsoy		1997	Liver	40		Pullus	mg/kg ww					0.39			0.15						12
Faroe Islands, Nolsoy		1998	Liver	25		Immat.+adult	mg/kg ww					8.55			2.66						12
Faroe Islands, Nolsoy		1998	Muscle	25		Immat.+adult	mg/kg ww					0.32			0.33						12
Faroe Islands, Nolsoy		1992	Liver	5	F/M		mg/kg dw					16.4	17.8		2.01	1.42		16.6	5.72		2
Faroe Islands, Nolsoy		1992	Muscle	5	F/M		mg/kg dw					1.82	0.68		0.13	0.06		3.11	0.82		2
Gavia arctica (Arctic loon)																					
Russia, Chaun	68N, 170W	1993	Feather	1	M		mg/kg dw					0.42			4.13						3
Russia, Chaun	68N, 170W	1993	Kidney	1	M		mg/kg dw					34.7			2.47						3
Russia, Chaun	68N, 170W	1993	Liver	1	M		mg/kg dw					3.33			6.48						3
Russia, Chaun	68N, 170W	1993	Muscle	1	M		mg/kg dw					0.19			0.95						3
Gavia stellata (red-throated loon)																					
Russia, Chaun	68N, 170W	1993	Feather	1	M		mg/kg dw					ND			3.65						3
Russia, Chaun	68N, 170W	1993	Kidney	1	M		mg/kg dw					20			5.94						3
Russia, Chaun	68N, 170W	1993	Liver	1	M		mg/kg dw					2.13			5.47						3
Russia, Chaun	68N, 170W	1993	Muscle	1	M		mg/kg dw					0.06			1.62						3
Haliaeetus albicilla (white-tailed sea eagle)																					
Norway	+62N	1850-1939	Feather	7		Juvenile	µg/g								0.58						13
Norway	+62N	1850-1939	Feather	4		Adult	µg/g								2.83						13
Norway	+62N	1940-1967	Feather	3		Juvenile	µg/g								2.65						13
Norway	+62N	1940-1967	Feather	3		Adult	µg/g								8.37						13
Norway	+62N	1968-1995	Feather	171		Juvenile	µg/g								2.21						13
Norway	+62N	1968-1995	Feather	135		Adult	µg/g								3.23						13
Haliaeetus leucocephalus (bald eagle)																					
Aleutian Islands, Adak		1994/95	Egg	10			ppm ww								1.341SE = .149						14
Aleutian Islands, Amchitka		1994/95	Egg	4			ppm ww								1.929SE = .060						14
Aleutian Islands, Kiska		1994/95	Egg	6			ppm ww								2.647SE = .440						14
Aleutian Islands, Tanaga		1994/95	Egg	5			ppm ww								1.064SE = .135						14
Larus argentatus (herring gull)																					
Canada, Northern Quebec		1981-1990	Liver	20 (20)			µg/g ww								2.47						15
Canada, Northern Quebec		1981-1990	Muscle	20 (20)			µg/g ww								0.88						15
Canada, Northern Quebec		1991-1997	Liver	4 (16)			µg/g ww	<0.10				0.46			1.30				0.01		15
Canada, Northern Quebec		1991-1997	Muscle	9 (29)			µg/g ww	0.12				0.50			0.69				0.93		15
Norway, Ainov Island	70.3N	1992	Liver	1			mg/kg ww					0.53			0.42				7.03		2
Norway, Ainov Island	70.3N	1992	Muscle	1			mg/kg ww					0.07			0.67				1.98		2
Norway, Hornøy	70.3N, 31E	1992-93	Egg	10			mg/kg ww								0.06	0.02					1
Norway, Lofoten	68N, 13E	1992-93	Egg	10			mg/kg ww								0.10	0.05					1
Norway, Nord Norland	69.3N, 18E	1992-93	Egg	10			mg/kg ww								0.09	0.03					1
Norway, West Finmark	71.1N, 25.3E	1992-93	Egg	10			mg/kg ww								0.07	0.02					1
Russia, Chaun	68N, 170W	1993	Feather	6			mg/kg dw					0.57	0.47		6.06	4.6					3
Russia, Chaun	68N, 170W	1993	Kidney	6			mg/kg dw					159	30		3.02	0.95					3
Russia, Chaun	68N, 170W	1993	Liver	6			mg/kg dw					26.3	16.1		4.01	2.14					3
Russia, Chaun	68N, 170W	1993	Muscle	6			mg/kg dw					0.91	0.22		0.72	0.3					3

Table A15. Metals in marine birds, continued

Species/location	Coordinate	Year	Tissue	n	Sex	Age (yr)	Unit	Lead min-max	Lead arithm. mean	Lead SD	Cadmium min-max	Cadmium arithm. mean	Cadmium SD	Mercury min-max	Mercury arithm. mean	Mercury SD	Selenium min-max	Selenium arithm. mean	Selenium SD	Remark	Reference
Larus hyperboreus (glaucous gull)																					
Canada, Anderson River	69.44N, 128.58W	1993	Egg	15			mg/kg dw								0.6	0.4		2.6	0.8		4
Canada, Browne Island	74.49N, 96.21W	1993	Egg	10			mg/kg dw								2.7	1.3		2.2	0.3		4
Canada, Northern Quebec		1991-1997	Muscle	4 (5)			µg/g ww		0.05			0.09			0.34			0.41			15
Canada, Prince Leopold Island	74.02N, 90.05W	1993	Egg	10			mg/kg dw								4.9	1.9		2.7	0.3		4
Canada, Richardson River	68.15N, 113.69W	1993	Egg	14			mg/kg dw								0.7	0.4		1.1	0.4		4
Greenland, Avanersuaq	77.5N, 70W	1984	Liver	4		Adult	mg/kg ww				14.5-44.2	25.1	13.6	2.40-4.50	3.29	0.94	3.36-6.55	4.83	1.33		16
Greenland, Avanersuaq	77.5N, 70W	1995	Liver	8		Adult	mg/kg ww				0.235-25.0	8.26	8.31	0.448-5.07	1.7	1.5	2.41-16.0	6.01	5.43		16
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1986	Liver	2		Adult	mg/kg ww				2.15-2.55	2.35	0.28	3.14-3.68	3.41	0.38	11.4-15.4	13.4	2.87		16
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1995	Liver	24		Adult	mg/kg ww				0.310-7.33	2.26	1.91	1.17-5.84	2.64	1.02	1.24-18.1	5.15	5.05		16
Greenland, Nanortalik	60N, 45W	1986	Liver	5		Adult	mg/kg ww				1.49-11.2	6.72	4.57	0.060-2.74	1.59	1.03	2.09-3.53	2.93	0.72		16
Greenland, Nanortalik	60N, 45W	1995	Liver	4		Adult	mg/kg ww				5.70-17.1	11	5.03	1.22-2.22	1.64	0.45	2.00-6.32	3.36	2.01		16
Russia, Chaun	68N, 170W	1993	Feather	1	M		mg/kg dw					ND			5.96						3
Russia, Chaun	68N, 170W	1993	Kidney	1	M		mg/kg dw					18.7			2.65						3
Russia, Chaun	68N, 170W	1993	Liver	1	M		mg/kg dw					3.9			3.28						3
Russia, Chaun	68N, 170W	1993	Muscle	1	M		mg/kg dw					0.13			0.5						3
Russia, Franz Josef Land, George Land		1992	Liver	2	F		mg/kg dw					4.28	4.09		1.36	0.03		9.54	2.35		2
Russia, Franz Josef Land, George Land		1992	Muscle	2	F		mg/kg dw					0.43	0.46		0.70	0.13		2.65	0.36		2
Russia, Franz Josef Land, Northbrook		1992	Liver	1	M		mg/kg dw					1.41			2.12			28.5			2
Russia, Franz Josef Land, Northbrook		1992	Muscle	1	M		mg/kg dw					0.09			0.92			8.16			2
Russia, Franz Josef Land, Stolichky		1992	Liver	3	M		mg/kg dw					7.38	4.44		0.59	0.16		4.39	0.33		2
Russia, Franz Josef Land, Stolichky		1992	Muscle	3	M		mg/kg dw					0.30	0.23		0.13	0.04		1.22	0.35		2
Russia, Pechora Sea, Guba Chernaya		1992	Liver	3	F/M		mg/kg dw					2.54	1.59		1.38	0.54		6.20	1.55		2
Russia, Pechora Sea, Guba Chernaya		1992	Muscle	3	F/M		mg/kg dw					0.23	0.08		0.5	0.16		2.18	0.53		2
Russia, Pechora Sea, Kolguyev		1992	Liver	5	M		mg/kg dw					1.21	0.46		1.21	0.57		6.00	0.96		2
Russia, Pechora Sea, Kolguyev		1992	Muscle	5	M		mg/kg dw					0.16	0.09		0.42	0.21		2.52	0.48		2
Russia, Pechora Sea, Vaygach		1992	Liver	2	F		mg/kg dw					1.42	0.73		0.97	0.2		4.33	0.13		2
Russia, Pechora Sea, Vaygach		1992	Muscle	2	F		mg/kg dw					0.19	0.01		0.3			1.82	0.25		2
Melanitta spp. (scoters)																					
Russia, Chukotka, Kanchalan		2001	Liver	1	F	2-3	µg/g ww		0.195			0.063			0.1960						9
Russia, Chukotka, Kanchalan		2001	Muscle	1	F	2-3	µg/g ww		0.210			0.011			0.0830						9
Russia, Chukotka, Kanchalan		2001	Liver	2	M	2-3	µg/g ww		0.104			0.132			0.3290						9
Russia, Chukotka, Kanchalan		2001	Muscle	2	M	2-3	µg/g ww		0.266*			0.013			0.0760					B	9
Morus bassanus (gannet)																					
Norway, Lofoten	68N, 13E	1992-1993	Egg	10			mg/kg ww								0.34	0.09					1
Phalacrocorax aristotelis (shag)																					
Norway, Hornøy	70.3N, 31E	1992-1993	Egg	10			mg/kg ww								0.09	0.1					1
Norway, Lofoten	68N, 13E	1992-1993	Egg	10			mg/kg ww								0.19	0.09					1
Norway, Nord Norland	69.3N, 18E	1992-1993	Egg	10			mg/kg ww								0.16	0.06					1
Polysticta stelleri (Steller's eider)																					
Russia, Alaska and Indigirka River Delta		1991-1995	Kidney	5	F		mg/kg dw					17.9			0.96			9.11			17
Russia, Alaska and Indigirka River Delta		1991-1995	Liver	6	F		mg/kg dw					5.93			2.04			17.6			17
Russia, Alaska and Indigirka River Delta		1991-1995	Kidney	3	M		mg/kg dw					39.8			1.69			12.8			17
Russia, Alaska and Indigirka River Delta		1991-1995	Liver	4	M		mg/kg dw					8.25			4.27			25.6			17
Rissa tridactyla (black-legged kittiwake)																					
Canada, Coburg Island	75.57N, 79.26W	1993	Egg	5			mg/kg dw								0.9	0.2		4.4	0.7		4
Canada, Prince Leopold Island	74.02N, 90.05W	1993	Egg	15			mg/kg dw								0.6	0.2		4.4	1.5		4
Canada, Prince Leopold Island	74.02N, 90.05W	1993	Egg	3			µg/g dw								0.46			4.82			10
Canada, Prince Leopold Island	74.02N, 90.05W	1993	Egg	3			µg/g dw								0.64			5.32			10
Canada, Prince Leopold Island	74.02N, 90.05W	1993	Egg	3			µg/g dw								0.63			3.16			10
Canada, Prince Leopold Island	74.02N, 90.05W	1993	Egg	3			µg/g dw								0.63			4.19			10
Canada, Prince Leopold Island	74.02N, 90.05W	1993	Egg	3			µg/g dw								0.74			4.39			10
Canada, Prince Leopold Island	74.02N, 90.05W	1998	Egg	3			µg/g dw								0.73			2.89			5
Canada, Prince Leopold Island	74.02N, 90.05W	1998	Egg	3			µg/g dw								0.61			1.95			5
Canada, Prince Leopold Island	74.02N, 90.05W	1998	Egg	3			µg/g dw								0.41			2.30			5
Canada, Prince Leopold Island	74.02N, 90.05W	1998	Egg	3			µg/g dw								0.61			2.43			5
Canada, Prince Leopold Island	74.02N, 90.05W	1998	Egg	3			µg/g dw								0.85			2.56			5
Canada, Prince Leopold Island, Lancaster Sound	74°02'N, 90°05'W	1975	Egg	12(4)			mg/kg dw								0.39	0.03		3.44	0.16	C	11
Canada, Prince Leopold Island, Lancaster Sound	74°02'N, 90°05'W	1976	Egg	6(2)			mg/kg dw								0.78			3.50		C	11
Canada, Prince Leopold Island, Lancaster Sound	74°02'N, 90°05'W	1987	Egg	3(1)			mg/kg dw								0.88			3.65		C	11
Canada, Prince Leopold Island, Lancaster Sound	74°02'N, 90°05'W	1993	Egg	15(5)			mg/kg dw								0.62	0.05		4.38	0.36	C	11
Canada, Prince Leopold Island, Lancaster Sound	74°02'N, 90°05'W	1998	Egg	15(5)			mg/kg dw								0.64	0.07		2.43	0.15	C	11
Norway, Hornøy	70.3N, 31E	1992-1993	Feather	10		juvenile	mg/kg dw					0.02	0.008		0.55	0.1		1.26	0.15		18
Norway, Hornøy	70.3N, 31E	1992-1993	Feather	27		adult	mg/kg dw					0.13	0.09		2.03	0.4		1.75	0.25		18

Table A15. Metals in marine birds, continued

Species/location	Coordinate	Year	Tissue	n	Sex	Age (yr)	Unit	Lead min-max	Lead arithm. mean	Lead SD	Cadmium min-max	Cadmium arithm. mean	Cadmium SD	Mercury min-max	Mercury arithm. mean	Mercury SD	Selenium min-max	Selenium arithm. mean	Selenium SD	Remark	Reference
Norway, Hornøy	70.3N, 31E	1992-1993	Liver	10		juvenile	mg/kg dw					0.18	0.03		0.24			8.87	2.99		18
Norway, Hornøy	70.3N, 31E	1992-1993	Liver	22		adult	mg/kg dw					14.4	11.3		2.85	1.02		16.9	9.5		18
Norway, Hornøy	70.3N, 31E	1992-1993	Muscle	10		juvenile	mg/kg dw					ND			0.03	0.06					18
Norway, Hornøy	70.3N, 31E	1992-1993	Muscle	27		adult	mg/kg dw					1.24	0.64		0.58	0.22					18
Norway, Hornøy	68N, 13E	1992-1993	Egg	10			mg/kg ww								0.13	0.08					1
Norway, Nord Norland	69.3N, 18E	1992-1993	Egg	10			mg/kg ww								0.13	0.07					1
Norway, Svalbard	79N, 11E	1992-1993	Egg	10			mg/kg ww								0.13	0.04					1
Russia, Franz Josef Land, George Land		1992	Liver	2	F/M		mg/kg dw					45.1	32.7		0.79	0.06		13.1	3.2		2
Russia, Franz Josef Land, George Land		1992	Muscle	2	F/M		mg/kg dw					1.98	0.33		0.21	0.06		5.30	0.87		2
Russia, Franz Josef Land, Hooker		1991	Liver	2	F/M		mg/kg dw					18.5	2.5		0.69	0.04		8.59	1.15		2
Russia, Franz Josef Land, Hooker		1991	Muscle	2	F/M		mg/kg dw					0.72	0.98		0.11	0.08		1.76	1.47		2
Russia, Franz Josef Land, Kuhn Island		1992	Liver	3	F/M		mg/kg dw					25.7	11.6		0.45	0.05		9.36	1.76		2
Russia, Franz Josef Land, Kuhn Island		1992	Muscle	3	F/M		mg/kg dw					1.31	0.17		0.15	0.05		4.43	0.47		2
Russia, Franz Josef Land, Northbrook		1992	Liver	2	F/M		mg/kg dw					18.1	5.25		0.64	0.18		14.0	2.21		2
Russia, Franz Josef Land, Northbrook		1992	Muscle	3	F/M		mg/kg dw					1.23	0.56		0.18	0.06		6.49	2.02		2
Russia, Franz Josef Land, Stolichky		1992	Liver	2	F		mg/kg dw					37	34.6		0.55	0.06		7.65	4.26		2
Russia, Franz Josef Land, Stolichky		1992	Muscle	2	F		mg/kg dw					1.68	1.1		0.16	0.04		5.45	0.84		2
Russia, Pechora Sea, Chernaya Bay		1992	Liver	5	F/M		mg/kg dw					8.14	12.3		1.44	0.29		13.7	9.8		2
Russia, Pechora Sea, Chernaya Bay		1992	Muscle	5	F/M		mg/kg dw					0.80	0.5		0.5	0.07		5.76	3.61		2
Russia, Pechora Sea, Vaygach Island		1992	Liver	6	F/M		mg/kg dw					11.0	11.9		0.66	0.37		9.10	1.8		2
Russia, Pechora Sea, Vaygach Island		1992	Muscle	6	F/M		mg/kg dw					0.42	0.32		0.28	0.07		4.78	1.05		2
Russia, Seven Islands		1992	Liver	5	F/M		mg/kg dw					22.3	16.8		0.98	0.14		17.7	6.72		2
Russia,		1992	Muscle	5	F/M		mg/kg dw					2.48	1.74		0.29	0.08		8.83	1.28		2
Somateria fischeri (spectacled eider)																					
Russia, Alaska and Indigirka River Delta		1991-1995	Kidney	10	F		mg/kg dw					41.0			0.57			25.7			17
Russia, Alaska and Indigirka River Delta		1991-1995	Liver	10	F		mg/kg dw					18.2			1.31			43.5			17
Russia, Alaska and Indigirka River Delta		1991-1995	Kidney	28	M		mg/kg dw					99.8			0.65			67.6			17
Russia, Alaska and Indigirka River Delta		1991-1995	Liver	28	M		mg/kg dw					37.0			1.18			124			17
USA, Alaska, St. Lawrence Island	63.47N, 171.45W	1995	Kidney	20			mg/kg dw					95.6	34.0		0.73	0.89		68.9	18.8		19
USA, Alaska, St. Lawrence Island	63.47N, 171.45W	1995	Liver	20			mg/kg dw					33.8	10.1		0.13	0.22		124	26		19
Somateria mollissima (common eider)																					
Canada, Belchers	56N, 80W	1997	Kidney	10			mg/kg dw					91.6	20.1								20
Canada, Belchers	56N, 80W	1997	Liver	10			mg/kg dw								1.46	0.25		10.4	0.8		20
Canada, East Bay	64N, 81W	1997	Kidney	13			mg/kg dw					70.6	5.6								20
Canada, East Bay	64N, 81W	1997	Liver	13			mg/kg dw								1.72	0.19		20.6	2.00		20
Canada, East Bay	64N, 81W	2000	Kidney	21			mg/kg dw					164.58									21
Canada, East Bay	64N, 81W	2000	Liver	21			mg/kg dw								3.32			16.15			21
Canada, Holman	71N, 118W	1997	Kidney	10			mg/kg dw					121.4	10.3								20
Canada, Holman	71N, 118W	1997	Liver	10			mg/kg dw								1.50	0.11		31.1	3.8		20
Norway, Nord Norland	69.3N, 18E	1992-1993	Egg	10			mg/kg ww								0.06	0.02					1
Russia, Chukotka, Lavrentiya		2001	Liver	11	F	1-3	µg/g ww	0.358				0.328			0.3870						9
Russia, Chukotka, Lavrentiya		2001	Muscle	11	F	1-3	µg/g ww	6.25*				0.032			0.0640					B	9
USA, Alaska		1991-1995	Kidney	21	F		mg/kg dw					77.6			0.85			10.1			17
USA, Alaska		1991-1995	Liver	21	F		mg/kg dw					16.1			1.59			7.85			17
USA, Alaska		1991-1995	Kidney	30	M		mg/kg dw					88.1			0.96			7.34			17
USA, Alaska		1991-1995	Liver	30	M		mg/kg dw					19.8			1.94			9.29			17
Somateria spectabilis (king eider)																					
Canada, East Bay	64N, 81W	1997	Kidney	10			mg/kg dw					165.3	11.5								21
Canada, East Bay	64N, 81W	1997	Liver	10			mg/kg dw								2.80	0.3		20.2	2.3		21
Canada, Holman	71N, 118W	1997	Kidney	10			mg/kg dw					125.3	18.7								21
Canada, Holman	71N, 118W	1997	Liver	10			mg/kg dw								1.71	0.14		37.4	3.9		21
Russia, Chaun	68N, 170W	1993	Feather	1	M		mg/kg dw					ND			0.57						3
Russia, Chaun	68N, 170W	1993	Kidney	1	M		mg/kg dw					124			3.92						3
Russia, Chaun	68N, 170W	1993	Liver	1	M		mg/kg dw					25.5			5.75						3
Russia, Chaun	68N, 170W	1993	Muscle	1	M		mg/kg dw					0.66			0.29						3
Russia, Pechora Sea, Dolgiy		1992	Liver	5	F/M		mg/kg dw					3.61	0.97		1.29	1.25		13.1	5.24		2
Russia, Pechora Sea, Dolgiy		1992	Muscle	5	F/M		mg/kg dw					0.07	0.04		0.21	0.07		2.37	0.53		2
Russia, Pechora Sea, Kolguyev		1992	Liver	3	F		mg/kg dw					5.65	2.23		0.72	0.15		8.75	1.79		2
Russia, Pechora Sea, Kolguyev		1992	Muscle	4	F/M		mg/kg dw					0.41	0.39		0.22	0.07		1.97	0.48		2
Russia, Pechora Sea, Vaygach		1992	Liver	1	M		mg/kg dw					5.18			3.03			10.9			2
Russia, Pechora Sea, Vaygach		1992	Muscle	1	M		mg/kg dw					0.47			0.59			3.74			2
USA, Alaska		1991-1995	Kidney	22	F		mg/kg dw					53.1			0.98			23.1			17
USA, Alaska		1991-1995	Liver	21	F		mg/kg dw					14.4			1.60			27.6			17
USA, Alaska		1991-1995	Kidney	32	M		mg/kg dw					78.0			1.3			22.1			17
USA, Alaska		1991-1995	Liver	33	M		mg/kg dw					24.6			2.49			34.5			17

Table A15. Metals in marine birds, continued

Species/location	Coordinate	Year	Tissue	n	Sex	Age (yr)	Unit	Lead min-max	Lead arithm. mean	Lead SD	Cadmium min-max	Cadmium arithm. mean	Cadmium SD	Mercury min-max	Mercury arithm. mean	Mercury SD	Selenium min-max	Selenium arithm. mean	Selenium SD	Remark	Reference
Stercorarius longicaudus (long-tailed jaeger)																					
Russia, Chaun	68N, 170W	1993	Feather	5	M		mg/kg dw					ND			1.95	0.54					3
Russia, Chaun	68N, 170W	1993	Kidney	5	M		mg/kg dw					113	65		5.05	0.93					3
Russia, Chaun	68N, 170W	1993	Liver	5	M		mg/kg dw					12.4	3.8		4.26	1.59					3
Russia, Chaun	68N, 170W	1993	Muscle	5	M		mg/kg dw					0.61	0.53		0.89	0.23					3
Stercorarius parasiticus (parasitic jaeger)																					
Russia, Chaun	68N, 170W	1993	Feather	1	M		mg/kg dw					0.48			1.84						3
Russia, Chaun	68N, 170W	1993	Kidney	1	M		mg/kg dw					39.7			3.66						3
Russia, Chaun	68N, 170W	1993	Liver	1	M		mg/kg dw					7.23			5.74						3
Russia, Chaun	68N, 170W	1993	Muscle	1	M		mg/kg dw					ND			0.63						3
Sterna paradisaea (Arctic tern)																					
Canada, Northern Quebec		81-90	Liver	2(2)			µg/g ww								1.63						15
Canada, Northern Quebec		81-90	Muscle	2(2)			µg/g ww								0.55						15
Russia, Chaun	68N, 170W	1993	Feather	10	M		mg/kg dw					ND			0.89	0.15					3
Russia, Chaun	68N, 170W	1993	Kidney	10	M		mg/kg dw					74.3	34.9		6.19	3.84					3
Russia, Chaun	68N, 170W	1993	Liver	10	M		mg/kg dw					15.3	7.7		4.84	3.66					3
Russia, Chaun	68N, 170W	1993	Muscle	10	M		mg/kg dw					0.49	0.35		1.30	0.75					3
Russia, Franz Josef Land, Etheridge Island		1992	Liver	5	F/M		mg/kg dw					16.3	4.7		1.08	0.21		7.34	0.67		3
Russia, Franz Josef Land, Etheridge Island		1992	Muscle	5	F/M		mg/kg dw					0.94	0.41		0.28	0.06		2.67	0.39		3
Uria aalge (common guillemot; common murre)																					
Norway, Hornøy	70.3N, 31E	1992-1993	Egg	10			mg/kg ww								0.10	0.04					1
Norway, Hornøy	70.3N, 31E	1992-1993	Feather	10			mg/kg dw					0.026	0.01		0.88	0.19		2.59	0.33		18
Norway, Hornøy	70.3N, 31E	1992-1993	Gonads	10			mg/kg dw					1.13	0.52		1.17	0.32		21.9	5.7		18
Norway, Hornøy	70.3N, 31E	1992-1993	Kidney	10			mg/kg dw					24.1	7.5		1.46	0.18		43.7	9.6		18
Norway, Hornøy	70.3N, 31E	1992-1993	Liver	10			mg/kg dw					3.08	1.12		1.88	0.41		17.6	5.04		18
Norway, Hornøy	70.3N, 31E	1992-1993	Lung	10			mg/kg dw					0.29	0.10		1.25	0.52					18
Norway, Hornøy	70.3N, 31E	1992-1993	Muscle	10			mg/kg dw					0.18	0.05		0.42	0.05					18
Russia, Kola Peninsula	68.5N, 37.3E	1992-1993	Egg	10			mg/kg ww								0.08	0.01					1
Russia, Seven Island, Kharlov Island	68N, 38E	1992	Liver	5	F/M		mg/kg dw					1.69	0.52		1.09	0.28		21.3	3.6		2
Russia, Seven Island, Kharlov Island	68N, 38E	1992	Muscle	5	F/M		mg/kg dw					0.15	0.02		0.33	0.05		8.74	1.26		2
Russia, Seven Island, Kuvshin Island	68N, 38E	1992	Liver	5	F/M		mg/kg dw					1.98	0.56		1.08	0.09		18.7	3.9		2
Russia, Seven Island, Kuvshin Island	68N, 38E	1992	Muscle	5	F/M		mg/kg dw					0.28	0.05		0.5	0.09		11.9	2.1		2
Uria lomvia (Brünnich's guillemot or thick-billed murre)																					
Canada, Coats Island	62.30N, 83.00W	1993	Egg	15			mg/kg dw								0.8	0.2		2.4	0.4		4
Canada, Coats Island	62.30N, 83.00W	1998	Egg	15			mg/kg dw								0.7	0.4		2.3	0.4		4
Canada, Coburg Island	75.57N, 79.26W	1993	Egg	15			mg/kg dw								1.4	0.08		2.1	0.4		4
Canada, Diggs Island	62.33N, 77.50W	1993	Egg	15			mg/kg dw								0.8	0.4		2.3	0.8		4
Canada, Northern Quebec		1991-1997	Muscle	1(5)			µg/g ww	0.10				0.04			0.32			1.02			15
Canada, Northern Quebec		1991-1997	Muscle	1(5)			µg/g ww	0.08				0.27			0.29			0.91			15
Canada, Northern Quebec		1991-1997	Muscle	2(9)			µg/g ww	0.06				0.48			0.37			0.99			15
Canada, Prince Leopold Island	74.02N, 90.05W	1993	Egg	15			mg/kg dw								1.1	0.4		2.6	0.4		4
Canada, Prince Leopold Island	74.02N, 90.05W	1998	Egg	15			mg/kg dw								1.2	0.4		2.2	0.8		4
Canada, Prince Leopold Island, Lancaster Sound	74°02'N, 90°05'W	1975	Egg	9(3)			mg/kg dw								0.56	0.04		2.76	0.24	C	11
Canada, Prince Leopold Island, Lancaster Sound	74°02'N, 90°05'W	1976	Egg	9(3)			mg/kg dw								0.78	0.06		2.57	0.10	C	11
Canada, Prince Leopold Island, Lancaster Sound	74°02'N, 90°05'W	1977	Egg	9(3)			mg/kg dw								0.46	0.04		2.52	0.28	C	11
Canada, Prince Leopold Island, Lancaster Sound	74°02'N, 90°05'W	1987	Egg	9(3)			mg/kg dw								0.98	0.06		2.44	0.15	C	11
Canada, Prince Leopold Island, Lancaster Sound	74°02'N, 90°05'W	1988	Egg	9(3)			mg/kg dw								0.97	0.03		2.90	0.34	C	11
Canada, Prince Leopold Island, Lancaster Sound	74°02'N, 90°05'W	1993	Egg	15(5)			mg/kg dw								1.13	0.10		2.57	0.14	C	11
Canada, Prince Leopold Island, Lancaster Sound	74°02'N, 90°05'W	1998	Egg	15(5)			mg/kg dw								1.19	0.07		2.20	0.2	C	11
Greenland, Nuuk	64.10N, 51.45W	1999	Liver	10	F/M		mg/kg ww					4.99	3.96		0.277	0.085		1.069	0.108		8
Greenland, Nuuk	64.10N, 51.45W	1999	Muscle	20	F/M		mg/kg ww					0.089	0.119		0.076	0.021		0.538	0.23		8
Norway, Hornøy	70.3N, 31E	1992-1993	Egg	10			mg/kg ww								0.10	0.02					1
Norway, Hornøy	70.3N, 31E	1992-1993	Feather	10		adult	mg/kg dw					0.12	0.08		0.78	0.18		2.75	1.27		18
Norway, Hornøy	70.3N, 31E	1992-1993	Gonads	5		adult	mg/kg dw					3.01	1.88		0.66	0.19		12.2	5.3		18
Norway, Hornøy	70.3N, 31E	1992-1993	Kidney	7		adult	mg/kg dw					46.7	31		1.16	0.21		15.8	1.6		18
Norway, Hornøy	70.3N, 31E	1992-1993	Liver	14		adult	mg/kg dw					5.51	4.24		1.11	0.51		7.05	3.48		18
Norway, Hornøy	70.3N, 31E	1992-1993	Lung	10		adult	mg/kg dw					1.14	0.66		0.53	0.18					18
Norway, Hornøy	70.3N, 31E	1992-1993	Muscle	15		adult	mg/kg dw					0.53	0.25		0.33	0.14					18
Norway, Svalbard	79N, 11E	1992-1993	Egg	10			mg/kg ww								0.20	0.06					1
Russia, Franz Josef Land, George Land		1992	Liver	2	M		mg/kg dw					19.3	7.91		0.65	0.11		8.16	7.1		2
Russia, Franz Josef Land, George Land		1992	Muscle	2	M		mg/kg dw					0.45	0.14		0.28	0.06		2.52	0.64		2
Russia, Franz Josef Land, Mable Island		1992	Liver	1	M		mg/kg dw					12.4			1.55			6.07			2
Russia, Franz Josef Land, Mable Island		1992	Muscle	1	M		mg/kg dw					1.79			0.60			2.92			2
Russia, Franz Josef Land, Northbrook		1992	Liver	2	F/M		mg/kg dw					13.1	1.13		0.58	0.01		4.26	0.4		2
Russia, Franz Josef Land, Northbrook		1992	Muscle	2	F/M		mg/kg dw					1.29	0.38		0.23			2.48	0.12		2

Table A15. Metals in marine birds, continued

Species/location	Coordinate	Year	Tissue	n	Sex	Age (yr)	Unit	Lead min-max	Lead arithm. mean	Lead SD	Cadmium min-max	Cadmium arithm. mean	Cadmium SD	Mercury min-max	Mercury arithm. mean	Mercury SD	Selenium min-max	Selenium arithm. mean	Selenium SD	Remark	Reference
Russia, Kola Peninsula	68.5N, 37.3E	1992-1993	Egg	10			mg/kg ww								0.07	0.02					1
Russia, Pechora Sea	69N, 59E	1992	Liver	5	F/M		mg/kg dw					3.98	1.00		0.79	0.11		6.27	0.9		2
Russia, Pechora Sea	69N, 59E	1992	Muscle	5	F/M		mg/kg dw					0.87	0.43		0.43	0.18		4.63	0.83		2
Russia, Seven Islands	68N, 38E,	1992	Liver	5	F/M		mg/kg dw					2.55	1.93		0.93	0.07		13.9	4.1		2
Russia, Seven Islands	68N, 38E	1992	Muscle	5	F/M		mg/kg dw					0.60	0.36		0.37	0.07		8.02	2.08		2
Xema sabini (Sabine's gull)																					
Russia, Chaun	68N, 170W	1993	Feather	2	M		mg/kg dw					ND			1.70	0.47					3
Russia, Chaun	68N, 170W	1993	Kidney	2	M		mg/kg dw					156	8		5.54	0.29					3
Russia, Chaun	68N, 170W	1993	Liver	2	M		mg/kg dw					16.3	3.0		3.89	1.8					3
Russia, Chaun	68N, 170W	1993	Muscle	2	M		mg/kg dw					0.68	0.18		1.06	0.25					3

Remarks
A. Appendix Table 5.2;
B. This sample is contaminated by lead rolls;
C. n = number of samples (number of pools analyzed).

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Table A16. Metals in marine mammals.

Species/location	Coordinate	Year	Tissue	n	Sex	Age*, yrs	Unit	Lead arithm. mean	Cadmium min-max	Cadmium arithm. mean	Cadmium SD	Cadmium geom. mean	Cadmium GSD	Mercury min-max	Mercury arithm. mean	Mercury SD	Mercury MeHg arithm. mean	Mercury MeHg geom. mean	Mercury MeHg GSD	Selenium arithm. mean	Selenium SD	Selenium geom. mean	Selenium GSD	Remark	Reference
Balaena mysticetus (bowhead whale)																									
USA, Alaska, Barrow	71.17N, 156.47W	1983-1997	Blubber	30-38			mg/kg ww			0.020	0.020				0.002	0.007				0.06	0.03				1
USA, Alaska, Barrow	71.17N, 156.47W	1983-1997	Kidney	47-48			mg/kg ww			20.020	17.290				0.038	0.033				1.58	0.42				1
USA, Alaska, Barrow	71.17N, 156.47W	1983-97	Liver	55			mg/kg ww			9.630	10.430				0.060	0.073				1.61	0.81				1
USA, Alaska, Barrow	71.17N, 156.47W	1983-97	Muscle	35-42			mg/kg ww			0.050	0.070				0.017	0.011				0.35	0.33				1
USA, Alaska, Barrow	71.17N, 156.47W	1992-94	Liver	9	F		mg/kg dw			28.300	20.200				0.280	0.990				3.82	1.16				2
USA, Alaska, Barrow	71.17N, 156.47W	1992-94	Liver	11	M		mg/kg dw			33.400	32.800				0.251	0.113				4.4	2.2				2
Balaenoptera acutorostrata (minke whale)																									
Greenland, Central east		1998	Baleen	2			mg/kg dw			0.030	0.029				0.280	0.150									3
Greenland, Central east		1998	Kidney	2			mg/kg dw			35.500	26.200				2.700	1.670				6.06	1.21				3
Greenland, Central east		1998	Liver	2			mg/kg dw			5.260	2.900				1.920	0.390				3.77	0.2				3
Greenland, Central east		1998	Muscle	4			mg/kg dw			0.190	0.190				0.403	0.135				0.69	0.35				3
Greenland, West		1998	Baleen	32-34			mg/kg dw			<0.018					0.068	0.060									3
Greenland, West		1998	Kidney	39-41			mg/kg dw			16.900	11.300				0.856	0.840				5.59	1.55				3
Greenland, West		1998	Liver	36			mg/kg dw			3.890	3.120				1.000	1.120				4.88	2.23				3
Greenland, West		1998	Muscle	38-42			mg/kg dw			0.210	0.370				0.288	0.185				0.65	0.19				3
Greenland, West		1998	Blubber	6	F/M		mg/kg ww			0.006	0.004				0.011	0.017				0.143	0.064				4
Greenland, West		1998	Epidermis (muktuk)	4	F/M		mg/kg ww			0.007	0.003				0.031	0.021				6.28	3.73				4
North Sea		1998	Baleen	21-22			mg/kg dw			<0.018					0.163	0.070									3
North Sea		1998	Kidney	21			mg/kg dw			13.600	8.300				2.450	1.480				9.23	2.29				3
North Sea		1998	Liver	22			mg/kg dw			1.850	0.740				2.040	1.260				7.07	2.14				3
North Sea		1998	Muscle	23			mg/kg dw			0.070	0.041				0.902	0.400				1.01	0.26				3
Norway coast		1998	Kidney	11			mg/kg dw			10.500	5.800				1.180	1.060				6.87	1.04				3
Norway coast		1998	Liver	14			mg/kg dw			2.390	1.860				1.740	2.190				5.31	2.15				3
Norway coast		1998	Muscle	10			mg/kg dw			0.062	0.058				0.593	± 0.604				0.82	0.27				3
Norway, Barents Sea		1998	Baleen	31			mg/kg dw			<0.018					0.159	0.093									3
Norway, Barents Sea		1998	Kidney	32			mg/kg dw			18.200	7.600				1.170	0.590				7.22	1.57				3
Norway, Barents Sea		1998	Liver	31			mg/kg dw			3.500	1.450				1.120	0.650				6.3	1.51				3
Norway, Barents Sea		1998	Muscle	33			mg/kg dw			0.127	0.076				0.505	0.204				0.95	0.35				3
Norway, Jan Mayen		1998	Baleen	22-23			mg/kg dw			10.600					0.255	0.166									3
Norway, Jan Mayen		1998	Kidney	23			mg/kg dw			20.400	10.600				2.380	1.630				7.06	2.34				3
Norway, Jan Mayen		1998	Liver	24			mg/kg dw			3.960	1.640				2.040	0.730				4.68	0.86				3
Norway, Jan Mayen		1998	Muscle	20-21			mg/kg dw			0.101	0.073				0.791	0.326				0.98	0.31				3
Norway, Svalbard	79N, 11E	1998	Baleen	1-2			mg/kg dw			<0.018					0.230										3
Norway, Svalbard	79N, 11E	1998	Kidney	16			mg/kg dw			15.400	8.700				0.694	0.337				6.81	1.11				3
Norway, Svalbard	79N, 11E	1998	Liver	16			mg/kg dw			3.760	1.490				0.728	0.352				5.53	1.02				3
Norway, Svalbard	79N, 11E	1998	Muscle	14			mg/kg dw			0.161	0.102				0.315	0.166				0.85	0.28				3
Delphinapterus leucas (beluga)																									
Canada, Arviat	61.2N, 94.0W	1999	Liver	36		11.2	mg/kg ww								12.5										5
Canada, Coral Harbour		1993	Liver	11		16.1	mg/kg ww								6.5										5
Canada, Coral Harbour		1997	Liver	19		13.1	mg/kg ww								13.7										5
Canada, Iqaluit		1993	Liver	23		12.9	mg/kg ww								7.6										5
Canada, Iqaluit		1994	Liver	7		12.9	mg/kg ww								16.3										5
Canada, Mackenzie Delta		1993	Liver	25		22.2	mg/kg ww								34.5										5
Canada, Mackenzie Delta		1994	Liver	41		17.7	mg/kg ww								28.4										5
Canada, Mackenzie Delta		1995	Liver	18		15.8	mg/kg ww								44.0										5
Canada, Mackenzie Delta		1996	Liver	10		14.6	mg/kg ww								43.8										5
Canada, Mackenzie Delta		2001	Liver	24		15.5	mg/kg ww								38.9										5
Canada, Pangnirtung	66.2N, 66.0W	1993	Liver	11		8.2	mg/kg ww								8.5										5
Canada, Pangnirtung	66.2N, 66.0W	1994	Liver	27		8.4	mg/kg ww								10.7										5
Canada, Pangnirtung	66.2N, 66.0W	1997	Liver	24		13.0	mg/kg ww								8.7										5
Greenland, Saqqaq	70.01N, 51.57W	2000	Blubber	5	F	245-398 cm	mg/kg ww			<0.005					0.020	0.016				0.1	0.04				4
Greenland, Saqqaq	70.01N, 51.57W	2000	Epidermis (muktuk)	5	F	245-398 cm	mg/kg ww			0.009	0.009				0.291	0.159				5.67	2.27				4
USA, Alaska		1992-1997	Blubber	11-16			mg/kg ww			ND										0.58	0.36				1
USA, Alaska		1992-1997	Epidermis	15-17			mg/kg ww								0.700	0.470				9.56	7.47				1
USA, Alaska		1992-1997	Kidney	23-45			mg/kg ww			12.150	6.020				4.580	3.090				6.27	3.24				1
USA, Alaska		1992-1997	Liver	24-50			mg/kg ww			3.810	1.760				12.420	10.320				41.41	32.67				1
USA, Alaska		1992-1997	Muscle	11-24			mg/kg ww			0.020	0.010				1.160	0.580				0.28	0.06				1
Erignathus barbatus (bearded seal)																									
Russia, Chukotka, Lavrentiya		2001	Blubber	1	M	2-3	µg/g ww	0.030		0.004					0.003										6
Russia, Chukotka, Lavrentiya		2001	Blubber	1	M	2-3	µg/g ww	0.028		0.024					0.037										6
Russia, Chukotka, Lavrentiya		2001	Kidney	1	M	2-3	µg/g ww	0.075		11.490					6.600										6
Russia, Chukotka, Lavrentiya		2001	Kidney	1	M	2-3	µg/g ww	0.110		7.330					8.320										6

Table A16. Metals in marine mammals, continued

Species/location	Coordinate	Year	Tissue	n	Sex	Age*, yrs	Unit	Lead arithm. mean	Cadmium min-max	Cadmium arithm. mean	Cadmium SD	Cadmium geom. mean	Cadmium GSD	Mercury min-max	Mercury arithm. mean	Mercury SD	Mercury MeHg arithm. mean	Mercury geom. mean	Mercury GSD	Selenium arithm. mean	Selenium SD	Selenium geom. mean	Selenium GSD	Remark	Reference
Russia, Chukotka, Lavrentiya		2001	Liver	1	M	2-3	µg/g ww	0.186		2.350					37.400										6
Russia, Chukotka, Lavrentiya		2001	Liver	1	M	2-3	µg/g ww	0.115		2.128					21.300										6
Russia, Chukotka, Lavrentiya		2001	Muscle	1	M	2-3	µg/g ww	0.021		0.011					1.200										6
Russia, Chukotka, Lavrentiya		2001	Muscle	1	M	2-3	µg/g ww	0.025		0.020					0.880										6
Eschrichtius robustus (= *E. gibbosus*) (grey whale)																									
Russia, Chukotka, Lavrentiya		2001	Blubber	1	M	2-3	µg/g ww	0.036		0.051					0.002										6
Russia, Chukotka, Lavrentiya		2001	Kidney	1	M	2-3	µg/g ww	0.059		6.320					2.100										6
Russia, Chukotka, Lavrentiya		2001	Liver	1	M	2-3	µg/g ww	0.169		0.950					0.500										6
Russia, Chukotka, Lavrentiya		2001	Muscle	1	M	2-3	µg/g ww	0.099		0.006					0.004										6
Eumetopias jubatus (Steller's seal lion)																									
USA, Alaska, Atka Island	52°04'N, 174°30'W	1999 April	Muscle	1			ng/g ww								1673.370									A	7
USA, Alaska, Atka Island	52°04'N, 174°30'W	1999 April	Muscle	1			ng/g ww										1391.67							B	7
USA, Alaska, Atka Island	52°04'N, 174°30'W	1999 Jan	Fat/little muscle	1			ng/g ww								305.870									A	7
USA, Alaska, Atka Island	52°04'N, 174°30'W	1999 Jan	Fat/little muscle	1			ng/g ww										182.59							B	7
Globicephala melas (= *G. melaena*) (pilot whale)																									
Faroe Islands, Leynar		1997	Muscle	7	F	juvenile	mg/kg ww								0.790	0.450									8
Faroe Islands, Leynar		1997	Muscle	31	F	adult	mg/kg ww								1.700	0.440									8
Faroe Islands, Leynar		1997	Muscle	9	M	juvenile	mg/kg ww								0.900	0.500									8
Faroe Islands, Leynar		1997	Muscle	3	M	adult	mg/kg ww								1.750	0.140									8
Faroe Islands, Suduroy	62N, 70W	1997	Muscle	24	F	adult	mg/kg ww			0.150					2.460										9
Faroe Islands, Suduroy	62N, 70W	1997	Muscle	8	M	adult	mg/kg ww			0.110					2.660										9
Faroe Islands, Suduroy	62N, 70W	1997	Muscle	17		juvenile	mg/kg ww			0.050					1.550										9
Faroe Islands, Tórshavn	62N, 70W	1997	Muscle	5	F	juvenile	mg/kg ww								1.450	1.000									8
Faroe Islands, Tórshavn	62N, 70W	1997	Muscle	21	F	adult	mg/kg ww								3.370	1.000									8
Faroe Islands, Tórshavn	62N, 70W	1997	Muscle	17	M	juvenile	mg/kg ww								2.290	0.760									8
Faroe Islands, Tórshavn	62N, 70W	1997	Muscle	7	M	adult	mg/kg ww								3.460	1.280									8
Faroe Islands, Tórshavn	62N, 70W	1999	Muscle	25	F	adult	mg/kg ww			0.257	0.108				1.630	1.030				0.63	0.45				10
Faroe Islands, Tórshavn	62N, 70W	1999	Muscle	11	M	adult	mg/kg ww			0.180	0.078				1.850	0.700				0.55	0.16				10
Faroe Islands, Tórshavn	62N, 70W	1999	Muscle	14		juvenile	mg/kg ww			0.113	0.082				1.310	0.790				0.61	0.23				10
Faroe Islands, Tórshavn	62N, 70W	2000	Kidney	10	F	adult	mg/kg ww		83-239	155.400	47.600														11
Faroe Islands, Tórshavn	62N, 70W	2000	Kidney	6	M	adult	mg/kg ww		100-215	146.200	47.000														11
Faroe Islands, Vestmanna	62N, 70W	1999	Kidney	5	F	adult	mg/kg ww		135.1-215.1	160.000	32.700														11
Faroe Islands, Vestmanna	62N, 70W	1999	Kidney	8	M	adult	mg/kg ww		74.7-144.7	110.400	26.400														11
Faroe Islands, Vestmanna	62N, 70W	1999	Muscle	8	F	adult	mg/kg ww			0.418	0.214				2.488	0.914				0.842	0.538				10
Faroe Islands, Vestmanna	62N, 70W	1999	Muscle	7	M	adult	mg/kg ww			0.180	0.058				2.056	0.437				0.613	0.06				10
Faroe Islands, Vestmanna	62N, 70W	1999	Muscle	7		juvenile	mg/kg ww			0.107	0.063				1.463	0.420				0.525	0.039				10
Halichoerus grypus (grey seal)																									
Faroe Islands	62N, 70W	1993-95	Muscle	10	F	juvenile	mg/kg ww								0.370	0.270									8
Faroe Islands	62N, 70W	1993-95	Muscle	11	F	adult	mg/kg ww								0.840	0.400									8
Faroe Islands	62N, 70W	1993-95	Muscle	7	M	juvenile	mg/kg ww								0.370	0.260									8
Faroe Islands	62N, 70W	1993-95	Muscle	3	M	adult	mg/kg ww								2.310	2.220									8
Faroe Islands	62N, 70W	1993-95	Liver	20	F	adult	mg/kg ww			14.600					155.000										9
Faroe Islands	62N, 70W	1993-95	Liver	4	M	adult	mg/kg ww			1.850					151.000										9
Faroe Islands	62N, 70W	1993-95	Liver	21		juvenile	mg/kg ww			0.940					11.600										9
Lagenorhynchus acutus (white-sided dolphin)																									
Faroe Islands, Klaksvík		1997	Muscle	7	F	juvenile	mg/kg ww								0.410	0.290									8
Faroe Islands, Klaksvík		1997	Muscle	8	F	adult	mg/kg ww								1.250	0.170									8
Faroe Islands, Klaksvík		1997	Muscle	1	M	juvenile	mg/kg ww								1.450										8
Faroe Islands, Klaksvík		1997	Muscle	12	M	adult	mg/kg ww								1.000	0.250									8
Faroe Islands	62N, 70 W	1997 Aug	Kidney	1	F	12-14	µg/g ww			24.320					1.580										12
Faroe Islands	62N, 70 W	1997 Aug	Kidney	1	F	<1	µg/g ww			0.020					0.120										12
Faroe Islands	62N, 70 W	1997 Aug	Kidney	1	M	9	µg/g ww			22.720					1.430										12
Faroe Islands	62N, 70 W	1997 Aug	Kidney	1	M	10	µg/g ww			31.070					2.540										12
Faroe Islands	62N, 70 W	1997 Aug	Kidney	1	M	<1	µg/g ww			0.020					0.150										12
Monodon monoceros (narwhal)																									
Greenland, Avanersuaq	77.5N, 70W	1993	Kidney	1			mg/kg ww			75.700					1.200					2.89					4
Greenland, Avanersuaq	77.5N, 70W	1993	Liver	1			mg/kg ww			11.600					7.880					4.49					4
Greenland, Avanersuaq	77.5N, 70W	1993	Muscle	4			mg/kg ww			0.342	1.700				1.150	1.140				0.38	1.2				4
Greenland, Kitsissuarsuit	69N, 53W	1990	Kidney	2			mg/kg ww			46.000	1.140				1.670	1.230				2.65	1.04				4
Greenland, Kitsissuarsuit	69N, 53W	1990	Liver	2			mg/kg ww			19.500	1.900				2.840	2.020				1.41	1.48				4
Greenland, Kitsissuarsuit	69N, 53W	1990	Muscle	3			mg/kg ww			0.156	1.980				0.630	1.270				0.26	1.15				4

Table A16. Metals in marine mammals, continued

Species/location	Coordinate	Year	Tissue	n	Sex	Age*, yrs	Unit	Lead arithm. mean	Cadmium min-max	Cadmium arithm. mean	Cadmium SD	Cadmium geom. mean	Cadmium GSD	Mercury min-max	Mercury arithm. mean	Mercury SD	Mercury MeHg arithm. mean	Mercury MeHg geom. mean	Mercury MeHg GSD	Selenium arithm. mean	Selenium SD	Selenium geom. mean	Selenium GSD	Remark	Reference
Greenland, Kitsissuarsuit	69N, 53W	1994	Liver	2			mg/kg ww			2.360	4.470				0.950	3.800				1.5	2.31				4
Greenland, Kitsissuarsuit	69N, 53W	1994	Muscle	2			mg/kg ww			0.029	3.440				0.280	2.670				0.3	1.18				4
Greenland, Uummannaq	71.50N, 52.50W	1993	Kidney	52			mg/kg ww			33.300	5.390				1.070	2.150				2.4	1.45				4
Greenland, Uummannaq	71.50N, 52.50W	1993	Liver	53			mg/kg ww			11.800	4.770				4.040	3.200				3.25	2.04				4
Greenland, Uummannaq	71.50N, 52.50W	1993	Muscle	53			mg/kg ww			0.100	2.400				0.800	1.740				0.34	1.11				4
Greenland, Uummannaq	71.50N, 52.50W	2000	Blubber	5	F		mg/kg ww			0.010	0.005				0.017	0.007				0.059	0.007				4
Greenland, Uummannaq	71.50N, 52.50W	2000	Epidermis (muktuk)	5	F		mg/kg ww			0.019	0.015				0.431	0.146				5.26	0.93				4
Odobenus rosmarus (walrus)																									
Canada, Inukjuaq	58.5N, 78W	1999	Kidney	7			mg/kg ww			16.490					0.310					3.85					13
Canada, Inukjuaq	58.5N, 78W	1999	Liver	5			mg/kg ww			3.320					2.640					2.59					13
Canada, Inukjuaq	58.5N, 78W	1999	Muscle	6			mg/kg ww			0.030					0.040					2.02					13
Phoca hispida (ringed seal)																									
Canada, Arctic Bay	72.9N, 85.0W	2000	Kidney	25		8.1	mg/kg ww			19.200	2.170				1.560	1.610				2.57	1.19				5
Canada, Arctic Bay	72.9N, 85.0W	2000	Liver	25		8.1	mg/kg ww			5.130	1.770				7.830	2.600				6.73	1.72				5
Canada, Arviat	61.2N, 94.0W	1998	Liver	24		18.7	mg/kg ww			7.570	3.050				14.000	3.390				12.5	2.38				5
Canada, Grise Fjord	76.6N, 85.2W	1998	Kidney	20		18.7	mg/kg ww			18.800	1.920				2.750	1.730				2.7	1.22				5
Canada, Grise Fjord	76.6N, 85.2W	1998	Liver	20		18.7	mg/kg ww			5.760	1.840				18.400	2.920				12.1	2.25				5
Canada, Hudson Strait	60N, 68W	1998	Kidney	24		5.2	mg/kg ww			5.510	14.000				0.686	1.740				2	1.25				5
Canada, Hudson Strait	60N, 68W	1998	Liver	24		5.2	mg/kg ww			2.750	7.390				3.500	5.160				4.57	2.43				5
Canada, Hudson Strait	60N, 68W	1998	Muscle	12		5.9	mg/kg ww			0.008	5.000				0.182	1.990				0.4	1.11				5
Canada, Labrador		1998	Kidney	28		5.9	mg/kg ww			5.710	2.800				1.040	1.430				2.22	1.23				5
Canada, Labrador		1998	Liver	28		5.9	mg/kg ww			2.850	1.870				6.320	3.470				6.22	1.6				5
Canada, Labrador		1998	Muscle	28		5.9	mg/kg ww			0.013	1.450				0.328	1.860				0.43	1.1				5
Canada, Labrador, Kangirsuk		1998	Cerebrum	2		3	mg/kg ww			0.010					0.070					0.27					14
Canada, Labrador, Kangirsuk		1998	Kidney	4		3	mg/kg ww								1.580	1.140									14
Canada, Labrador, Kangirsuk		1998	Liver	4		3	mg/kg ww			8.810	6.200				20.300	33.500				10.2	13.4				14
Canada, Labrador, Kangirsuk		1998	Muscle	4		3	mg/kg ww								0.310	0.120									14
Canada, Labrador, Kangisualujjuaq		1998	Cerebrum	5		7	mg/kg ww			0.230	0.010				0.090	0.020				0.23	0.01				14
Canada, Labrador, Kangisualujjuaq		1998	Kidney	9		7	mg/kg ww								0.860	0.230									14
Canada, Labrador, Kangisualujjuaq		1998	Liver	9		7	mg/kg ww			3.480	1.940				9.170	10.300				6.96	4.6				14
Canada, Labrador, Kangisualujjuaq		1998	Muscle	9		7	mg/kg ww								0.260	0.120									14
Canada, Labrador, Kuujjuaq		1998	Cerebrum	6		2	mg/kg ww			0.020	0.000				0.040	0.020				0.23	0.02				14
Canada, Labrador, Kuujjuaq		1998	Kidney	7		2	mg/kg ww			7.190	7.920				0.780	0.440				1.63	0.36				14
Canada, Labrador, Kuujjuaq		1998	Muscle	2		2	mg/kg ww								0.230										14
Canada, Labrador, Makkovik		1998	Cerebrum	6		7	mg/kg ww			0.010	0.000				0.190	0.120				0.34	0.06				14
Canada, Labrador, Makkovik		1998	Brain	9		5	mg/kg ww			0.010	0.010				0.130	0.070				0.3	0.08				14
Canada, Labrador, Makkovik		1998	Kidney	14		6	mg/kg ww			6.750	2.590				1.190	0.500				2.06	0.37				14
Canada, Labrador, Makkovik		1998	Liver	14		6	mg/kg ww			3.290	2.300				10.400	5.240				7.52	2.59				14
Canada, Labrador, Makkovik		1998	Muscle	14		6	mg/kg ww			0.010	0.010				0.460	0.240				0.38	0.09				14
Canada, Labrador, Nain		1998	Cerebrum	15		6	mg/kg ww			0.020	0.020				0.090	0.040				0.3	0.06				14
Canada, Labrador, Nain		1998	Kidney	13		6	mg/kg ww			8.690	6.470				1.030	0.300				2.47	0.45				14
Canada, Labrador, Nain		1998	Liver	13		6	mg/kg ww			3.660	2.570				7.250	4.650				6.06	2.26				14
Canada, Labrador, Nain		1998	Muscle	13		6	mg/kg ww			0.040	0.060				0.310	0.150				0.49	0.08				14
Canada, Labrador, Quaqtaq		1998	Cerebrum	10		7	mg/kg ww			0.020	0.010				0.090	0.060				0.24	0.03				14
Canada, Labrador, Quaqtaq		1998	Kidney	12		6	mg/kg ww			8.810	13.400				0.930	0.400				1.99	0.5				14
Canada, Labrador, Quaqtaq		1998	Liver	12		7	mg/kg ww			4.080	4.580				13.300	14.500				9.06	6.47				14
Canada, Labrador, Quaqtaq		1998	Muscle	12		7	mg/kg ww			0.020	0.030				0.250	0.180				0.4	0.04				14
Canada, Labrador, Quaqtaq		1998	Whole brain	3		6	mg/kg ww			0.030	0.020				0.070	0.050				0.22	0.01				14
Canada, Labrador, Salluit		1998	Cerebrum	6		2	mg/kg ww			0.010	0.010				0.220	0.440				0.27	0.05				14
Canada, Labrador, Salluit		1998	Kidney	12		4	mg/kg ww													0.640	0.360				14
Canada, Labrador, Salluit		1998	Liver	12		4	mg/kg ww			7.530	5.130				4.950	7.280				4.07	3.47				14
Canada, Labrador, Salluit		1998	Muscle	10		5	mg/kg ww								0.190	0.090									14
Canada, Labrador, Salluit		1998	Whole brain	2		2	mg/kg ww			0.030					0.070					0.3					14
Canada, Pangnirtung	66.2N, 66.0W	1998	Liver	24		5.3	mg/kg ww			8.900	3.790				6.040	3.870				5.33	2.72				5
Canada, Pond Inlet	72.8N, 77.0W	1976	Liver	33		5.2	mg/kg ww								3.760	3.420								C	15
Canada, Pond Inlet	72.8N, 77.0W	2000	Kidney	23		3.1	mg/kg ww			25.000	2.530				1.230	1.800				2.82	1.26				5
Canada, Pond Inlet	72.8N, 77.0W	2000	Liver	25		3.3	mg/kg ww			9.340	2.550				4.000	3.560				4.41	2.12				5
Canada, Pond Inlet	72.8N, 77.0W	2000	Liver	15		4.5	mg/kg ww							1.1- 69.8	12.900	9.100								D	16
Canada, Resolute Bay	74.7N, 94.3W	2000	Kidney	23		5.8	mg/kg ww			24.300	4.110				1.390	1.750				1.96	1.3				5
Canada, Resolute Bay	74.7N, 94.3W	2000	Liver	23		5.8	mg/kg ww			5.990	3.610				1.730	4.270				3.39	2.38			D	5
Canada, Ungava Bay	58N, 68W	1998	Kidney	31		4.9	mg/kg ww			9.310	2.830				1.080	1.830				2.03	1.29				5
Canada, Ungava Bay	58N, 68W	1998	Liver	29		5.1	mg/kg ww			4.720	2.240				10.800	3.540				7.79	2.36			D	5
Canada, Ungava Bay	58N, 68W	1998	Muscle	13		5.7	mg/kg ww								0.250	1.650									5
Greenland, Avanersuaq	77.5N, 70W	1994	Liver	7		<1	mg/kg ww			1.780	3.060				0.746	0.746								D	4,17
Greenland, Avanersuaq	77.5N, 70W	1994	Liver	4		1-3	mg/kg ww			0.860	0.810				0.579	0.291								D	4,17
Greenland, Avanersuaq	77.5N, 70W	1994	Liver	7		4-6	mg/kg ww			11.200	7.700				7.620	4.250								D	4,17

Table A16. Metals in marine mammals, continued

Species/location	Coordinate	Year	Tissue	n	Sex	Age*, yrs	Unit	Lead arithm. mean	Cadmium min-max	Cadmium arithm. mean	Cadmium SD	Cadmium geom. mean	Cadmium GSD	Mercury min-max	Mercury arithm. mean	Mercury SD	Mercury MeHg arithm. mean	Mercury MeHg geom. mean	Mercury MeHg GSD	Selenium arithm. mean	Selenium SD	Selenium geom. mean	Selenium GSD	Note	Ref
Greenland, Avanersuaq	77.5N, 70W	1994	Liver	7		≥7	mg/kg ww			15.900	11.600				7.930	4.810								D	4,17
Greenland, Avanersuaq	77.5N, 70W	1998	Liver	3		<1	mg/kg ww			1.880	3.220				1.090	1.090								D	4
Greenland, Avanersuaq	77.5N, 70W	1998	Liver	7		1-3	mg/kg ww			3.830	1.840				3.940	1.970								D	4
Greenland, Avanersuaq	77.5N, 70W	1998	Liver	4		4-6	mg/kg ww			10.400	2.400				10.100	5.700								D	4
Greenland, Avanersuaq	77.5N, 70W	1998	Liver	7		≥7	mg/kg ww			14.900	10.300				8.510	6.830								D	4
Greenland, Avanersuaq	77.5N, 70W	1998	Liver	24		9.2	mg/kg ww			6.170	3.820				4.160	2.980				3.17	2.12				4
Greenland, Avanersuaq	77.5N, 70W	1984/85	Liver	1		<1	mg/kg ww			9.110														D	4,17
Greenland, Avanersuaq	77.5N, 70W	1984/85	Liver	26/22		1-3	mg/kg ww			13.000	14.800				1.220	0.990								D/E	4,17
Greenland, Avanersuaq	77.5N, 70W	1984/85	Liver	13/7		4-6	mg/kg ww			31.700	18.000				1.910	0.930								D/E	4,17
Greenland, Avanersuaq	77.5N, 70W	1984/85	Liver	16/15		≥7	mg/kg ww			33.100	30.400				3.120	2.580								D/E	4,17
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1986	Liver	2		<1	mg/kg ww			1.770	1.850				0.607	0.230								D	4,17
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1986	Liver	27		1-3	mg/kg ww			16.700	27.200				2.270	1.090								D	4,17
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1986	Liver	18		4-6	mg/kg ww			16.800	5.100				5.770	3.890								D	4,17
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1986	Liver	22/21		≥7	mg/kg ww			13.000	9.200				12.000	9.600								D/E	4,17
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1994	Liver	1		<1	mg/kg ww			0.060					0.360									D	4,17
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1994	Liver	10		1-3	mg/kg ww			5.950	4.310				4.000	2.340								D	4,17
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1994	Liver	11		4-6	mg/kg ww			9.140	4.100				5.850	3.430								D	4,17
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1994	Liver	5		≥7	mg/kg ww			16.800	9.400				8.300	4.460								D	4,17
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1999	Liver	2		<1	mg/kg ww			3.070	3.890				0.651	0.511								D	4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1999	Liver	6		1-3	mg/kg ww			4.220	3.510				1.240	0.620								D	4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1999	Liver	7		4-6	mg/kg ww			14.000	11.300				5.610	5.130								D	4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1999	Liver	5		≥7	mg/kg ww			9.770	6.880				7.850	3.620								D	4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1999	Liver	25		4.9	mg/kg ww			5.280	5.040				2.660	3.160				3.16	1.97				4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	2000	Liver	1		<1	mg/kg ww			5.000					2.800									D	4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	2000	Liver	6		1-3	mg/kg ww			5.630	3.110				7.190	6.780								D	4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	2000	Liver	11		4-6	mg/kg ww			9.210	5.480				7.510	4.250								D	4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	2000	Liver	2		≥7	mg/kg ww			12.100	14.300				7.060	3.830								D	4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	2000	Liver	20		4.3	mg/kg ww			6.600	1.970				6.000	1.790				4.34	1.59				4
Greenland, Qeqertarsuaq	61N, 53W	1999	Liver	25		2.4	mg/kg ww			12.200	2.000				0.890	2.330				1.48	1.63				4
Greenland, Qeqertarsuaq	61N, 53W	2000	Blubber	10	F/M		mg/kg ww			0.011	0.007				<0.005					<0.2					4
Greenland, Qeqertarsuaq	61N, 53W	2000	Liver	20		1.2	mg/kg ww			8.290	1.590				0.810	1.910				1.38	1.35				4
Norway, Svalbard	79N, 11E	1996	Muscle	17	F/M	8.2	mg/kg dw			0.270	0.380				0.390	0.120									18
Norway, Svalbard	79N, 11E	1996	Kidney	17		8.2	mg/kg dw			61.800	60.000				2.210	0.620									18
Norway, Svalbard	79N, 11E	1996	Liver	17		8.2	mg/kg dw			13.000	12.100				3.240	2.190				6.58	2.44				18
Russia, Chukotka, Lavrentiya		2001	Blubber	1	F	1-2	µg/g ww	0.080		0.064					0.075										6
Russia, Chukotka, Lavrentiya		2001	Blubber	1	F	1-2	µg/g ww	0.015		0.090					0.058										6
Russia, Chukotka, Lavrentiya		2001	Blubber	1	F	2-3	µg/g ww	0.079		0.077					0.016										6
Russia, Chukotka, Lavrentiya		2001	Kidney	1	F	1-2	µg/g ww	0.085		15.660					7.150										6
Russia, Chukotka, Lavrentiya		2001	Kidney	1	F	1-2	µg/g ww	0.104		11.280					5.250										6
Russia, Chukotka, Lavrentiya		2001	Kidney	1	F	2-3	µg/g ww	0.103		7.650					10.000										6
Russia, Chukotka, Lavrentiya		2001	Liver	1	F	1-2	µg/g ww	0.098		2.150					8.360										6
Russia, Chukotka, Lavrentiya		2001	Liver	1	F	1-2	µg/g ww	0.076		1.078					4.650										6
Russia, Chukotka, Lavrentiya		2001	Liver	1	F	2-3	µg/g ww	0.111		3.610					4.300										6
Russia, Chukotka, Lavrentiya		2001	Muscle	1	F	1-2	µg/g ww	0.008		0.082					0.920										6
Russia, Chukotka, Lavrentiya		2001	Muscle	1	F	1-2	µg/g ww	0.007		0.070					0.790										6
Russia, Chukotka, Lavrentiya		2001	Muscle	1	F	2-3	µg/g ww	0.007		0.063					0.800										6
Russia, Chukotka, Lavrentiya		2001	Blubber	1	M	1-2	µg/g ww	0.014		0.045					1.590										6
Russia, Chukotka, Lavrentiya		2001	Blubber	1	M	2-3	µg/g ww	0.018		0.044					0.002										6
Russia, Chukotka, Lavrentiya		2001	Blubber	1	M	2-3	µg/g ww	0.267		0.520					1.500										6
Russia, Chukotka, Lavrentiya		2001	Kidney	1	M	1-2	µg/g ww	0.124		7.380					2.540										6
Russia, Chukotka, Lavrentiya		2001	Kidney	1	M	2-3	µg/g ww	0.096		6.360					3.120										6
Russia, Chukotka, Lavrentiya		2001	Kidney	1	M	2-3	µg/g ww	0.084		9.630					5.800										6
Russia, Chukotka, Lavrentiya		2001	Liver	1	M	1-2	µg/g ww	0.101		1.958					2.870										6
Russia, Chukotka, Lavrentiya		2001	Liver	1	M	2-3	µg/g ww	0.108		2.116					3.500										6
Russia, Chukotka, Lavrentiya		2001	Liver	1	M	2-3	µg/g ww	0.122		7.023					4.200										6
Russia, Chukotka, Lavrentiya		2001	Muscle	1	M	1-2	µg/g ww	0.068		0.030					0.125										6
Russia, Chukotka, Lavrentiya		2001	Muscle	1	M	2-3	µg/g ww	0.060		0.039					0.150										6
Russia, Chukotka, Lavrentiya		2001	Muscle	1	M	2-3	µg/g ww	0.071		0.560					1.630										6
USA, Alaska, Barrow	71.17N, 156.47W	1995-1997	Blubber	11			mg/kg ww			0.020	0.010				0.002	0.001				0.21	0.11				19
USA, Alaska, Barrow	71.17N, 156.47W	1995-1997	Kidney	17			mg/kg ww			26.010	15.590				0.500	0.280				2.51	0.97				19
USA, Alaska, Barrow	71.17N, 156.47W	1995-1997	Liver	17			mg/kg ww			5.270	3.210				3.520	3.520				7.16	5.77				19
USA, Alaska, Barrow	71.17N, 156.47W	1995-1997	Muscle	11			mg/kg ww			0.090	0.060				0.220	0.330				0.28	0.09				19
Phoca largha (spotted seal)																									
Russia, Chukotka, Lavrentiya		2001	Blubber	1	F	1-2	µg/g ww	0.045		0.016					0.031										6
Russia, Chukotka, Lavrentiya		2001	Blubber	1	F	1-2	µg/g ww	0.060		0.023					0.025										6
Russia, Chukotka, Lavrentiya		2001	Blubber	1	F	1-2	µg/g ww	0.053		0.038					0.027										6
Russia, Chukotka, Lavrentiya		2001	Blubber	1	F	1-2	µg/g ww	0.066		0.039					0.033										6
Russia, Chukotka, Lavrentiya		2001	Blubber	1	F	1-2	µg/g ww	0.071		0.014					0.031										6

Table A16. Metals in marine mammals, continued

Species/location	Coordinate	Year	Tissue	n	Sex	Age*, yrs	Unit	Lead arithm. mean	Cadmium min-max	Cadmium arithm. mean	Cadmium SD	Cadmium geom. mean	Cadmium GSD	Mercury min-max	Mercury arithm. mean	Mercury SD	Mercury MeHg arithm. mean	Mercury MeHg geom. mean	Mercury MeHg GSD	Selenium arithm. mean	Selenium SD	Selenium geom. mean	Selenium GSD	Remark	Reference
Russia, Chukotka, Lavrentiya		2001	Kidney	1	F	1-2	µg/g ww	0.035		10.250					7.670										6
Russia, Chukotka, Lavrentiya		2001	Kidney	1	F	1-2	µg/g ww	0.148		13.200					9.240										6
Russia, Chukotka, Lavrentiya		2001	Kidney	1	F	1-2	µg/g ww	0.040		9.540					4.150										6
Russia, Chukotka, Lavrentiya		2001	Kidney	1	F	1-2	µg/g ww	0.023		18.900					8.300										6
Russia, Chukotka, Lavrentiya		2001	Kidney	1	F	1-2	µg/g ww	0.058		12.700					8.320										6
Russia, Chukotka, Lavrentiya		2001	Liver	1	F	1-2	µg/g ww	0.100		4.800					26.640										6
Russia, Chukotka, Lavrentiya		2001	Liver	1	F	1-2	µg/g ww	0.115		2.690					18.300										6
Russia, Chukotka, Lavrentiya		2001	Liver	1	F	1-2	µg/g ww	0.084		0.245					18.700										6
Russia, Chukotka, Lavrentiya		2001	Liver	1	F	1-2	µg/g ww	0.164		4.740					20.400										6
Russia, Chukotka, Lavrentiya		2001	Liver	1	F	1-2	µg/g ww	0.080		4.080					20.900										6
Russia, Chukotka, Lavrentiya		2001	Muscle	1	F	1-2	µg/g ww	0.028		0.070					3.150										6
Russia, Chukotka, Lavrentiya		2001	Muscle	1	F	1-2	µg/g ww	0.024		0.112					3.280										6
Russia, Chukotka, Lavrentiya		2001	Muscle	1	F	1-2	µg/g ww	0.039		0.040					1.660										6
Russia, Chukotka, Lavrentiya		2001	Muscle	1	F	1-2	µg/g ww	0.033		0.050					2.800										6
Russia, Chukotka, Lavrentiya		2001	Muscle	1	F	1-2	µg/g ww	0.050		0.050					2.250										6
Russia, Chukotka, Lavrentiya		2001	Blubber	1	M	1-2	µg/g ww	0.057		0.011					0.026										6
Russia, Chukotka, Lavrentiya		2001	Blubber	1	M	1-2	µg/g ww	0.054		0.029					0.017										6
Russia, Chukotka, Lavrentiya		2001	Blubber	1	M	1-2	µg/g ww	0.073		0.045					0.027										6
Russia, Chukotka, Lavrentiya		2001	Blubber	1	M	1-2	µg/g ww	0.070		0.012					0.032										6
Russia, Chukotka, Lavrentiya		2001	Blubber	1	M	1-2	µg/g ww	0.051		0.020					0.031										6
Russia, Chukotka, Lavrentiya		2001	Kidney	1	M	1-2	µg/g ww	0.118		3.360					5.243										6
Russia, Chukotka, Lavrentiya		2001	Kidney	1	M	1-2	µg/g ww	0.058		8.470					8.991										6
Russia, Chukotka, Lavrentiya		2001	Kidney	1	M	1-2	µg/g ww	0.029		19.070					5.683										6
Russia, Chukotka, Lavrentiya		2001	Kidney	1	M	1-2	µg/g ww	0.025		20.900					9.528										6
Russia, Chukotka, Lavrentiya		2001	Kidney	1	M	1-2	µg/g ww	0.078		15.600					7.253										6
Russia, Chukotka, Lavrentiya		2001	Liver	1	M	1-2	µg/g ww	0.071		0.326					25.350										6
Russia, Chukotka, Lavrentiya		2001	Liver	1	M	1-2	µg/g ww	0.156		4.360					14.100										6
Russia, Chukotka, Lavrentiya		2001	Liver	1	M	1-2	µg/g ww	0.102		0.183					19.790										6
Russia, Chukotka, Lavrentiya		2001	Liver	1	M	1-2	µg/g ww	0.084		0.299					27.000										6
Russia, Chukotka, Lavrentiya		2001	Liver	1	M	1-2	µg/g ww	0.142		8.350					19.350										6
Russia, Chukotka, Lavrentiya		2001	Muscle	1	M	1-2	µg/g ww	0.032		0.080					3.053										6
Russia, Chukotka, Lavrentiya		2001	Muscle	1	M	1-2	µg/g ww	0.024		0.020					2.302										6
Russia, Chukotka, Lavrentiya		2001	Muscle	1	M	1-2	µg/g ww	0.048		0.040					1.813										6
Russia, Chukotka, Lavrentiya		2001	Muscle	1	M	1-2	µg/g ww	0.033		0.040					2.093										6
Ursus maritimus (polar bear)																									
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1983	Kidney	1		Sub-adult	mg/kg ww			1.250					6.040					3.08					4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1983	Liver	1		Sub-adult	mg/kg ww			0.090					4.410					2.54					4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1984	Hair	7		Adult	mg/kg dw								4.260	0.970									4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1984	Hair	2		Sub-adult	mg/kg dw								3.560	1.400									4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1984	Kidney	10		Adult	mg/kg ww			42.690	30.830				31.510	17.160				6.73	5.72				4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1984	Kidney	5		Sub-adult	mg/kg ww			11.510	9.930				16.380	11.800				4.69	1.56				4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1984	Liver	10		Adult	mg/kg ww			1.660	0.870				13.120	5.920				5.23	2.18				4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1984	Liver	5		Sub-adult	mg/kg ww			1.080	0.680				8.390	3.770				3.59	1.1				4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1985	Kidney	2		Sub-adult	mg/kg ww			25.650	4.450				28.050	25.100				15.19	11.19				4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1985	Liver	2		Sub-adult	mg/kg ww			1.430	0.110				7.780	1.910				4.15	1.82				4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1986	Hair	1		Sub-adult	mg/kg dw								2.780										4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1986	Kidney	2		Sub-adult	mg/kg ww			25.000	12.450				10.710	1.120				7.58	2.07				4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1986	Liver	2		Sub-adult	mg/kg ww			2.150	0.210				6.420	2.060				2.93	0.52				4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1986	Muscle	2		Sub-adult	mg/kg ww			0.024	0.010				0.048	0.004				0.465	0.064				4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1987	Hair	1		Adult	mg/kg dw								6.320										4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1987	Hair	3		Sub-adult	mg/kg dw								4.310	1.020									4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1987	Kidney	1		Adult	mg/kg ww			11.600					48.600					17.9					4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1987	Kidney	10		Sub-adult	mg/kg ww			11.640	5.160				10.840	4.540				6.78	1.7				4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1987	Liver	1		Adult	mg/kg ww			0.560					23.900										4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1987	Liver	10		Sub-adult	mg/kg ww			1.020	0.670				8.510	3.000				3.32	0.83				4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1987	Muscle	2		Adult	mg/kg ww			0.044	0.008				0.067	0.031				0.32	0.141				4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1987	Muscle	10		Sub-adult	mg/kg ww			0.016	0.009				0.077	0.045				0.301	0.103				4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1989	Hair	1		Adult	mg/kg dw								4.350										4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1989	Hair	1		Sub-adult	mg/kg dw								7.460										4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1989	Kidney	1		Adult	mg/kg ww			123.170					38.770					16.23					4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1989	Kidney	1		Sub-adult	mg/kg ww			7.480					34.070					12.76					4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1989	Liver	1		Adult	mg/kg ww			4.530					16.390					6.41					4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1989	Liver	1		Sub-adult	mg/kg ww			0.300					7.670					2.75					4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1989	Muscle	1		Adult	mg/kg ww			0.114					0.088					0.25					4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1989	Muscle	1		Sub-adult	mg/kg ww			0.006					0.167					0.18					4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1990	Hair	4		Adult	mg/kg dw								4.940	1.450									4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1990	Hair	6		Sub-adult	mg/kg dw								7.560	2.600									4

Table A16. Metals in marine mammals, continued

Species/location	Coordinate	Year	Tissue	n	Sex	Age*, yrs	Unit	Lead arithm. mean	Cadmium min-max	Cadmium arithm. mean	Cadmium SD	Cadmium geom. mean	Cadmium GSD	Mercury min-max	Mercury arithm. mean	Mercury SD	Mercury MeHg arithm. mean	Mercury geom. mean	Mercury GSD	Selenium arithm. mean	Selenium SD	Selenium geom. mean	Selenium GSD	Remark	Reference
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1990	Kidney	4		Adult	mg/kg ww			8.130	11.540				10.070	8.440				4.67	3.45				4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1990	Kidney	6		Sub-adult	mg/kg ww			5.170	4.620				10.780	3.520				5.22	1.53				4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1990	Liver	4		Adult	mg/kg ww			0.750	1.150				7.730	7.330				2.81	2.82				4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1990	Liver	6		Sub-adult	mg/kg ww			0.750	1.300				7.560	2.200				3.09	0.73				4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1990	Muscle	3		Adult	mg/kg ww			0.005	0.001				0.570	0.032				0.09	0.052				4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1990	Muscle	7		Sub-adult	mg/kg ww			0.010	0.010				0.094	0.041				0.26	0.103				4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1999	Hair	6		Adult	mg/kg dw								4.550	0.690		4.500	1.170						4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1999	Kidney	6		Adult	mg/kg ww			20.150	12.110	17.600	1.170		22.580	14.280		18.500	2.050	10.47	5.67	9.15	1.79		4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1999	Liver	6		Adult	mg/kg ww			0.730	0.410	0.650	1.680		8.200	2.000		7.990	1.290	4.29	1.09	4.17	1.32		4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	1999	Muscle	6		Adult	mg/kg ww			0.022	0.240	0.016	2.090		0.119	0.065		0.110	1.570	0.509	0.103	0.5	1.23		4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	2000	Hair	6		Adult	mg/kg dw								5.170	1.220									4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	2000	Hair	4		Sub-adult	mg/kg dw								2.670	0.730		2.590	1.360						4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	2000	Hair	6		Adult	mg/kg dw											5.030	1.300						4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	2000	Kidney	6		Adult	mg/kg ww			36.740	21.300	32.000	1.810		29.830	16.620		25.400	1.440	12.35	5.51	11.4	1.56		4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	2000	Kidney	4		Sub-adult	mg/kg ww			23.160	13.600	17.700	2.740		8.900	3.700		8.280	1.570	5.29	1.18	5.18	1.26		4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	2000	Liver	6		Adult	mg/kg ww			1.820	1.030	1.540	1.970		16.400	5.300		15.600	1.440	7.51	1.8	7.3	1.31		4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	2000	Liver	4		Sub-adult	mg/kg ww			1.730	0.970	1.420	2.280		4.020	1.630		3.790	1.490	2.71	0.85	2.62	1.33		4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	2000	Muscle	6		Adult	mg/kg ww			0.027	0.014	0.024	1.700		0.137	0.046		0.130	1.470	0.5	0.12	0.49	1.27		4
Greenland, Ittoqqortoormiit	70.3N, 22.3W	2000	Muscle	3		Sub-adult	mg/kg ww			0.016	0.010	0.014	2.240		0.089	0.027		0.090	1.330	0.5	0.027	0.5	1.05		4
USA, Alaska		1993-1999	Kidney	20	M		mg/kg dw			25.100	11.700				52.500	40.100				26.3	15.7				20
USA, Alaska		1993-1999	Liver	19	M		mg/kg dw			4.200	13.000				25.200	4.700				13.5	9.2				20
USA, Alaska		1993-1999	Muscle	18	M		mg/kg dw			0.100	0.100				0.400	1.000				2.7	0.6				20
USA, Alaska, Barrow	71.17N, 156.47W	1995-1997	Blubber	11			mg/kg ww			No data										0.04	0.02				19
USA, Alaska, Barrow	71.17N, 156.47W	1995-1997	Kidney	24			mg/kg ww			8.690	5.050									12.99	10.58				19
USA, Alaska, Barrow	71.17N, 156.47W	1995-1997	Liver	24			mg/kg ww			0.470	0.220				14.220	12.920				9.33	13				19
USA, Alaska, Barrow	71.17N, 156.47W	1995-1997	Muscle	18-23			mg/kg ww			0.010	0.020				0.090	0.070				0.54	0.15				19

* Length where indicated by 'cm'

Remarks

A. Values reported asTHg;
B. Values reported as MeHg;
C. In Appendix Table 5.2 and AMAP I;
D. In Appendix Table 5.2 ;
E. n: for Cd / for Hg;

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Abbreviations and definitions of some terms

ACIA . . . . . Arctic Climate Impact Assessment
AEPS . . . . . Arctic Environmental Protection Strategy
AET . . . . . Apparent effects thresholds
ALAD, ALA-D. . δ-aminolevulinic acid dehydratase (an enzyme)
AMAP . . . . . Arctic Monitoring and Assessment Programme
AMMTAP . . . . . Alaska Marine Mammal Tissue Archival Project
AO. . . . . Arctic Oscillation
AMG . . . . . Autometallographic
BAMS . . . . . Barrow Arctic Mercury Study
DEHM. . . . . Danish Eulerian Hemispheric Model (Danish National Environmental Research Institute)
DNMI . . . . . Norwegian Meteorological Institute
dw . . . . . Dry weight – basis of determination of concentration
ECMWF . . . . . European Centre for Medium-range Weather Forecasts
EF . . . . . Enrichment Factor
EMEP . . . . . European Monitoring and Evaluation Programme (Co-operative Programme for Monitoring and Evaluation of Long-Range Transboundary Air Pollutants in Europe, under the UN ECE LRTAP)
ER-L . . . . . Effects Range-Low
ER-M. . . . . Effects Range-Median
FPA . . . . . FPA filters
FPM. . . . . Fine particulate matter
FR . . . . . Flux ratio (sediment)
GEIA . . . . . Global Emissions Inventory Activity
GEM (and Hg^0) . Gaseous elemental mercury
GIS . . . . . Geographical Information System
GOME. . . . . Global Ozone Monitoring Experiment
GRAHM . . . . . Global/Regional Atmospheric Heavy Metals model (Meteorological Service of Canada)
Hg^0 (and GEM) . Gaseous elemental mercury
Hg(II). . . . . Oxidized inorganic mercury
Hg^{2+} . . . . . Divalent mercury
Hg-p . . . . . Particulate associated mercury
ICES . . . . . International Council for the Exploration of the Sea
ICP-MS . . . . . Inductively Coupled Plasma - Mass spectrometry
INTAS . . . . . International Association for the Promotion of Co-operation with Scientists from the New Independent States (NIS) of the Former Soviet Union (European Union)
ISQG . . . . . Interim sediment quality guidelines (Canada)
ITM. . . . . Institute of Applied Environmental Research, Stockholm University
IWC. . . . . International Whaling Commission
JAMP. . . . . Joint Assessment and Monitoring Programme (OSPAR)
LRTAP. . . . . Long-Range Transboundary Air Pollution (UN ECE Convention)
MAMCS . . . . . Mediterranean Atmospheric Mercury Cycle System
MDE . . . . . Mercury depletion event(s)
MeHg . . . . . Methylmercury
MF . . . . . Marginal filter(s)
MITE-RN . . . . . Metals in the Environment Research Network (Canada)
MM5. . . . . Mesoscale Model version 5 (Pennsylvania State University National Center for Atmospheric Research)
MMT . . . . . Methylcyclopentadienyl manganese tricarbonyl
MOE . . . . . Mercury Over Europe
mRNA. . . . . messenger RNA
MSC-E. . . . . Meteorological Synthesizing Centre - East (EMEP)
MSCE-HM-Hem MSC-E Heavy Metals Hemispheric Model
MT . . . . . Metallothionein (a protein)
NAO . . . . . North Atlantic Oscillation
NBSP . . . . . National Benthic Surveillance Project (Alaska)
NILU . . . . . Norwegian Institute for Air Research
NOAA. . . . . National Oceanic and Atmospheric Administrations (US)
NRC . . . . . National Research Council
NS&T . . . . . National Status and Trends Program (NOAA)
NWT . . . . . Northwest Territories (Canada)
OSPAR . . . . . Oslo and Paris Commission
PCB . . . . . Polychlorinated biphenyl
PEL . . . . . Probable effects limit
PGEs . . . . . Platinum group elements (platinum, Pt; palladium, Pd; and rhodium, Rh)
POPs . . . . . Persistent Organic Pollutants
PTS . . . . . Persistent Toxic Substances
QA/QC . . . . . Quality assurance/quality control
REA. . . . . Relaxed Eddy Accumulation
RGM . . . . . Reactive Gaseous Mercury
SPASIBA . . . . . Scientific Program on Arctic and Siberian Aquatorium (Russia-France-Netherlands)
TAMU . . . . . Texas A&M University
TDC . . . . . Thematic Data Centre(s) (AMAP)
TDS . . . . . Total dissolved solids
TEL . . . . . Threshold effects level
TGM . . . . . Total gaseous mercury
TOC . . . . . Total organic carbon
TPM . . . . . Total particulate matter
TSM . . . . . Total suspended matter
UAF. . . . . University of Alaska, Fairbanks
UN. . . . . United Nations
UN ECE. . . . . United Nations Economic Commission for Europe
UV-B . . . . . Ultraviolet-B
varved . . . . . annually-laminated
ww. . . . . Wet weight – basis of determination of concentration