Chapter 4
Spatial Patterns

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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 Amdéma (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)
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/m² 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 \((\text{Hg/Mn})_{\text{snow}} / (\text{Hg/Mn})_{\text{crust}}\) 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) 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/m²/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/m²/yr, but varied between 2 to 10 µg/m²/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/m²/yr) at Pallas, Finland.

Deposition of total particulate Hg averaged 1.4 ng/m³ at Pallas. Deposition was higher and more variable at five more southerly Scandinavian stations (7.6 to 34.6 ng/m³; 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 Rövrikt 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/m³), and wet deposition three times higher (4.25 to 8.97 versus 1.91 to 2.52 µg/m²/yr), in the
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 biomonitorators 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-
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 Zapoljarnii, spatially extensive studies on *Hylocomium splendens* performed as part of the Kola Ecogeochronology 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
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 (Frosli 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 demonstrated 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.
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 (Qeqartarsuaq 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. Froslie 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 km² 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 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 0.23 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 m³/s and a drainage basin of 50000 km², 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 (Dauwalter, 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; Dauwalter, 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 Ecogeochemy 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 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 4-6. 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).
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; Skorvold 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 Jonsson, 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 (Field 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.

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 (Äbeskójâvre; 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),
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, 15N, 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 15N to 14N (δ15N) 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 δ15N 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).
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.

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 studies at the Kanuti National Wildlife Refuge (Alaska) on northern pike and Arctic grayling (Thymal-lus 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 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).

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<th></th>
<th>V</th>
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<th>Ni</th>
<th>Zn</th>
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Effects range, median, low

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resulting 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; Tisbulski 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.

<table>
<thead>
<tr>
<th>Location</th>
<th>Species (tissue)</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>Southern Alaska, USA</td>
<td>Flathead sole (liver)</td>
<td>Meador, 1994</td>
</tr>
<tr>
<td>Northern Alaska</td>
<td>Fourhorn sculpin (liver and stomach)</td>
<td>Meador, 1994</td>
</tr>
<tr>
<td>Alaska</td>
<td>Arctic flounder (stomach contents)</td>
<td>Meador, 1994</td>
</tr>
<tr>
<td>Labrador and northern Quebec, Canada</td>
<td>Sea-run Arctic char (muscle)</td>
<td>Muir et al., 1999b; 2000</td>
</tr>
<tr>
<td>Greenland</td>
<td>Greenland cod (muscle)</td>
<td>F. Riget, unpbl. data, 2002</td>
</tr>
<tr>
<td>Greenland</td>
<td>Atlantic salmon (muscle and liver)</td>
<td>F. Riget, unpbl. data, 2002</td>
</tr>
<tr>
<td>Greenland</td>
<td>Greenland halibut (muscle and liver)</td>
<td>F. Riget, unpbl. data, 2002</td>
</tr>
<tr>
<td>Greenland</td>
<td>Capelin (muscle and whole fish)</td>
<td>Olsen et al., 2003</td>
</tr>
<tr>
<td>Faroe Islands</td>
<td>Shorthorn sculpin (liver)</td>
<td>Olsen et al., 2003</td>
</tr>
<tr>
<td>Faroe Islands</td>
<td>Atlantic cod (liver)</td>
<td>Larsen and Dam, 1999</td>
</tr>
<tr>
<td>Iceland</td>
<td>Common dab (liver)</td>
<td>Dam, 2000</td>
</tr>
<tr>
<td>Norway</td>
<td>Atlantic cod (liver)</td>
<td>Audunsson et al., 1997; Yngvadóttir and Halldórsdóttir, 1998, 1999, Yngvadóttir et al., 2002</td>
</tr>
<tr>
<td>Norway</td>
<td>Common dab (whole fish)</td>
<td>Green et al., 2001</td>
</tr>
<tr>
<td>Pechora Sea</td>
<td>Arctic cod and navaga (muscle and liver)</td>
<td>Tisbulski et al., 2001</td>
</tr>
</tbody>
</table>
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 815N 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, 815N, and 813C (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. gorbuscha*). 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>
<thead>
<tr>
<th>Location</th>
<th>Species</th>
<th>Reference</th>
</tr>
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<tbody>
<tr>
<td>Barents Sea</td>
<td>13 species</td>
<td>Savinov et al., 2003</td>
</tr>
<tr>
<td>Chaun, northeast Siberia</td>
<td>11 species</td>
<td>Kim et al., 1996</td>
</tr>
<tr>
<td>Alaska, Arctic Russia</td>
<td>4 eider</td>
<td>Stout et al., 2002</td>
</tr>
<tr>
<td>Arctic Canada</td>
<td>2 eider</td>
<td>Wayland et al., 1999a, 2001</td>
</tr>
<tr>
<td>Western Alaska</td>
<td>Spectacled eider</td>
<td>Trust et al., 2000</td>
</tr>
<tr>
<td>Arctic Canada</td>
<td>4 species (eggs)</td>
<td>Braune et al., 2002</td>
</tr>
<tr>
<td>Great Slave Lake, Canada</td>
<td>Herring gull (eggs)</td>
<td>Wayland et al., 1999b</td>
</tr>
<tr>
<td>Faroe Islands</td>
<td>Black guillemot (liver/eggs)</td>
<td>Olsen et al., 2003</td>
</tr>
<tr>
<td>Greenland</td>
<td>Black guillemot (liver)</td>
<td>F. Riget, pers. comm., 2002</td>
</tr>
<tr>
<td>Faroe Islands</td>
<td>Fulmar</td>
<td>Larsen and Dam, 2003</td>
</tr>
<tr>
<td>Barents Sea (northern Norway),</td>
<td>10 species (eggs)</td>
<td>Barrett et al., 1996</td>
</tr>
<tr>
<td>Svalbard, northwest Russia</td>
<td>4 species</td>
<td>T. Nygård, pers. comm., 2002</td>
</tr>
<tr>
<td>Northern Norway</td>
<td>Bald eagle (eggs)</td>
<td>Wenzel and Gabrielsen, 1995</td>
</tr>
<tr>
<td>Aleutian Islands, Alaska, USA</td>
<td></td>
<td>Anthony et al., 1999</td>
</tr>
</tbody>
</table>
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, oxychlorodane, 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 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).

Siberian seabirds 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-
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., 2000; Muir et al., 2000; Lockhart, 1999; Dam, 2001; Dam and Bloch, 2000; Larsen and Dam, 1999; Olsen et al., 2003; F. Riget, unpubl. data, 2002; Woshner et al., 2001a, 2003).

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,
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

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).
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 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. melas)) 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-
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 m³/hr by flameless AA. Cadmium concentrations averaged 0.02 ng/m³, with a maximum of 0.038 ng/m³ (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/km², 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

![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).]
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).

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

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 relative to most of the north. However, latitudinal gradients 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 Zapolyarnii, 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 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 updwind of Norilsk in remote (unroaded) areas.
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 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; Froslie et al., 1984; Selinus et al., 1996 cited in Nyholm and Rübling, 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-
concentrations 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 Kangerlussuaq 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 western Scandinavia, 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 Pallass (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 : 0.13 : 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 combined 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).

<table>
<thead>
<tr>
<th>Location</th>
<th>Coordinates</th>
<th>pH</th>
<th>Alk, meq/L</th>
<th>Ca, meq/L</th>
<th>Mg, meq/L</th>
<th>Cd, µg/L</th>
<th>Cu, µg/L</th>
<th>As, µg/L</th>
</tr>
</thead>
<tbody>
<tr>
<td>Ivavik National Park</td>
<td>69°26’N, 139°36’W</td>
<td>7.4</td>
<td>0.4</td>
<td>0.988</td>
<td>0.27</td>
<td>0.086</td>
<td>1.2</td>
<td>0.22</td>
</tr>
<tr>
<td>Cape Bathurst</td>
<td>70°46’N, 127°45’W</td>
<td>8.1</td>
<td>3.7</td>
<td>1.98</td>
<td>3.2</td>
<td>0.009</td>
<td>0.6</td>
<td>2.34</td>
</tr>
<tr>
<td>Banks Island N</td>
<td>73°40’N, 116°12’W</td>
<td>7.8</td>
<td>0.88</td>
<td>0.616</td>
<td>0.392</td>
<td>0.002</td>
<td>0.3</td>
<td>0.09</td>
</tr>
<tr>
<td>Melville Island S</td>
<td>75°03’N, 107°51’W</td>
<td>8.1</td>
<td>1.8</td>
<td>0.93</td>
<td>5.36</td>
<td>0.002</td>
<td>0.6</td>
<td>1.28</td>
</tr>
<tr>
<td>Ellesmere Island</td>
<td>78°49’N, 103°40’W</td>
<td>7.0</td>
<td>0.12</td>
<td>0.206</td>
<td>0.177</td>
<td>0.01</td>
<td>0.9</td>
<td>0.08</td>
</tr>
<tr>
<td>Devon Island S</td>
<td>76°28’N, 86°50’W</td>
<td>8.0</td>
<td>1.3</td>
<td>1.11</td>
<td>0.322</td>
<td>0.004</td>
<td>0.1</td>
<td>0.07</td>
</tr>
<tr>
<td>Baffin Island S</td>
<td>74°36’N, 82°24’W</td>
<td>8.1</td>
<td>3.1</td>
<td>1.89</td>
<td>1.97</td>
<td>0.002</td>
<td>0.2</td>
<td>0.23</td>
</tr>
<tr>
<td>Baffin Island S</td>
<td>68°26’N, 66°50’W</td>
<td>6.5</td>
<td>0.039</td>
<td>0.034</td>
<td>0.159</td>
<td>0.004</td>
<td>0.3</td>
<td>0.03</td>
</tr>
<tr>
<td>Baffin Island S</td>
<td>68°26’N, 66°50’W</td>
<td>6.4</td>
<td>0.021</td>
<td>0.034</td>
<td>0.178</td>
<td>0.006</td>
<td>0.6</td>
<td>0.02</td>
</tr>
</tbody>
</table>
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 Ivavik 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 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 sipunculoid and nemertine from the Pechora Sea (Savinov et al., 1998) revealed highest Cd levels in epifaunal suspension feeders such as Iceland scallop (Oklamys islandicus; 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.
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 4.24. 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).

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 8.53 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., 2000c).

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).

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.
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 Juelshamm et al. (1987) but higher than more recent findings for Newfoundland, Canada (Muir et al., 1998). 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, Juelshamm 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-
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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 204Pb-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/m³ for data reported during the late 1990s. Higher values (generally 1 to 10 ng/m³) 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 Zemlya 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, Åyräs 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., 1997). 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/km²/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 Zapolarjaniuq cause only weak Pb gradients in moss (Åyräs et al., 1997; Reimann et al., 1997b) and O-horizon soils (Reimann et al., 1997b).

On the Taymir Peninsula, concentrations of Pb in Hylcocitum 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
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
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 (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; Dauwalter and Rognerud, 2002), but not for Pb. Lead enrichment increased westward, possibly indicating source contributions by vehicle emissions from northern Finland and northeastern Norway (Dauwalter, 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.

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 (Abiskojáváre) 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 (Sirois 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 sulfur, 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 Ecogeochmistry 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 Hylcomium 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 Ecogeochmistry 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 (Åyräs 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 Ecogeochmistry Project, large regional processes do not influence moss concentrations of Zn, and local variability is as high as regional variability (Åyräs 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 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 (Åyräs 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.

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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 Sør-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 Pechenga-nickel 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 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 alkalinitities (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 Ringnes 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 (Rognrud 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).

<table>
<thead>
<tr>
<th>Sample site</th>
<th>Copper</th>
<th>Nickel</th>
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<tbody>
<tr>
<td></td>
<td>Bilberry (green leaves)</td>
<td>Crowberry (green leaves)</td>
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<tr>
<td>Polluted sites</td>
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<td>5</td>
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<tr>
<td>Background sites</td>
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<td>22</td>
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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 (Dauwalter, 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 Tasuiaq (Nunavik, Canada) than Labrador and other locations (Muir et al., 2000). A study of land-locked 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 homogeneous at about 1.5 ng/m³. 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

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