16.0 CONTAMINANTS

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Some information has been published describing the chemical contamination of biota from Hudson Bay, especially animals taken by Native people for subsistence consumption (seals, whales, polar bears, birds, fish). The Northern Contaminants Program has issued annual reports describing research on northern contaminants and it has also produced two comprehensive summaries. The second summary was released in March 2003, and some of the illustrations from it provide a timely overview of the state of chemical contamination of the biota of Hudson Bay.

One important question is whether the contamination of physical strata or biological organisms is fully or partially the result of human activity. Residues of synthetic organochlorine compounds have been found consistently in animals from Hudson Bay. These compounds result exclusively from human activities. They are products of 20th-century technology and have no natural sources and no natural background concentrations.

16.1 EVIDENCE OF INPUTS OF SYNTHETIC ORGANIC COMPOUNDS

Some of the most convincing evidence of the impacts of human activities on Hudson Bay and indeed the Arctic in general, derives from studies of synthetic organic compounds. These compounds reach the Arctic by several means but one important pathway is via moving air masses (Figure 16-1). For example, studies of the composition of the air from the Canadian North have consistently identified a wide range of synthetic organic compounds that originate thousands of km away. Figure 16-2 shows the levels of one of the isomers of hexachlorocyclohexane in air from Kinngait (Cape Dorset) in winter of 1994/95 and again in 2000/01. While it is discouraging to find compounds like these in the air of northern communities, it is encouraging to see that the levels measured in 2000/01 were lower than those found in 1994/95. Its two major users, China and the former USSR, discontinued the use of HCH in 1983 and 1999, respectively (CACAR II 2003). These synthetic compounds with little or no history of use in the North have nonetheless appeared throughout the Arctic in air, water and aquatic life.

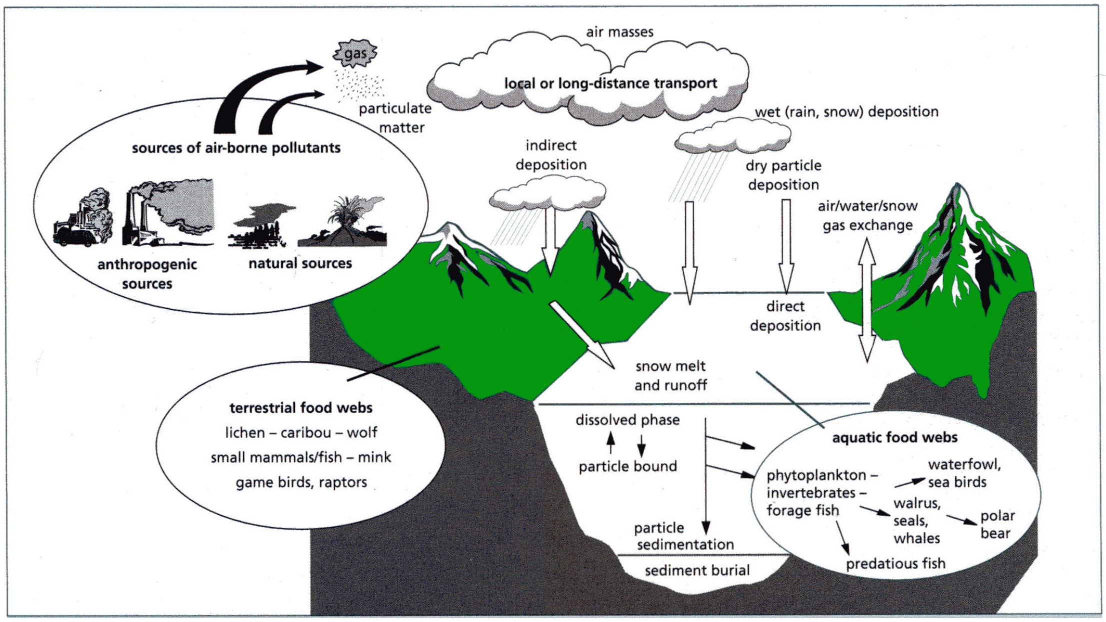


Figure 16-1. Schematic of pathways of transport and accumulation of persistent organic comtaminants and some metals to arctic and marine ecosystems. (From CACAR I 1997, Figure 3.2.1, page 193).

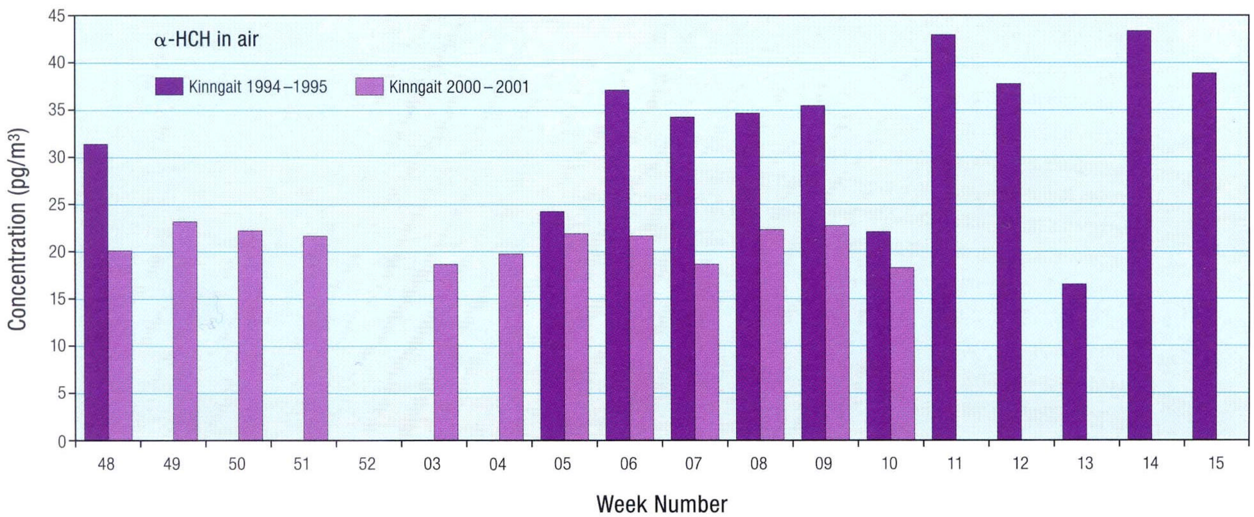


Figure 16-2. HCH in air from Kinngait (Cape Dorset) during the winters of 1994/95 and 2000/01. (From CACAR II 2003 : Physical Environment, Figure B.1.3, page 81).

16.2 EVIDENCE OF IMPACT OF ANTHROPOGENIC ACTIVITIES

Another good example of the impact of human activities is the deposition of cesium-137 (Cs-137) derived from the atmospheric testing of fission bombs. This is illustrated in Figure 16-3. This isotope does not occur naturally. It is formed by the fission of uranium atoms into smaller fragments. The figure shows a sediment profile of Cs-137 (red dots) from southeastern Hudson Bay in 1992 (Lockhart 1998). The Cs-137 must have reached Hudson Bay and other sites throughout the hemisphere with moving air masses. Each point in this graph is the result from the analysis of a slice of sediment from the core starting at the top with the sediment-water interface. The dates shown were calculated from the profile of the natural isotope lead-210 (blue triangles). The peak period for the deposition of bomb isotopes should occur in 1963 and this core shows a broad peak in slices in the 1950s and 1960s. There has been some mixing or vertical movement of Cs-137 within the core otherwise we

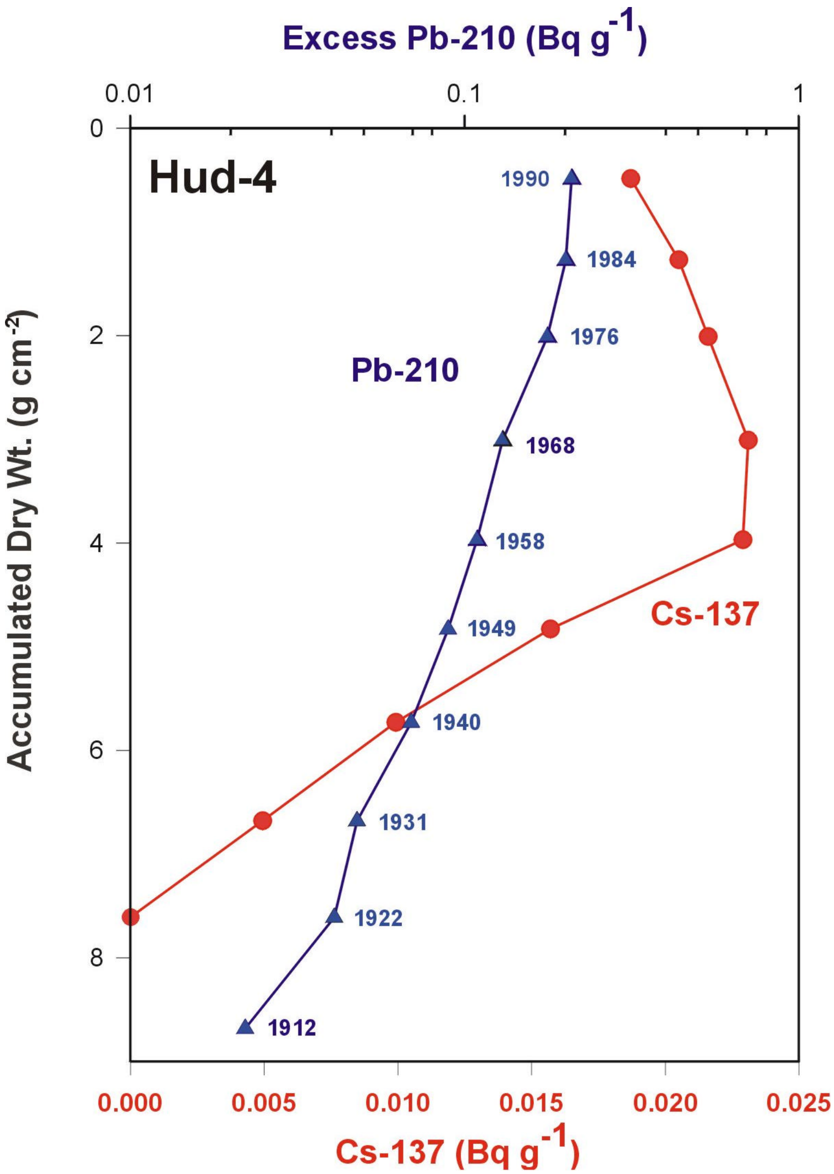


Figure 16-3. Down core profile of lead-210, a natural radioactive isotope of lead, and cesium-137, a byproduct of atmospheric testing of nuclear bombs, in a sediment core from southeastern Hudson Bay in 1992 (from Lockhart et al. 1998).

should expect a more rapid decline in slices deposited after peak deposition and we should not expect any in the slice dated at 1931. Nonetheless, the presence of Cs-137 is an unambiguous sign of the presence of products of human activity well removed from the watershed.

In 1992, the Department of Fisheries and Oceans sponsored a workshop to explore the nature of impacts on Hudson Bay with particular attention to hydroelectricity projects (Bunch and Reeves 1992). Several impacts were considered, namely the Great Whale hydroelectricity project in Quebec, the Conawapa hydroelectricity project in Manitoba, mercury contamination, nutrients and suspended matter, impacts on nearshore habitats, freshwater fish, marine mammals, endangered fish stocks, and productivity of marine species. The principal discussion was given to the cumulative impacts of combinations of individual impacts. For example, hydroelectric reservoirs alter the timing of freshwater inflows and that in turn can affect sedimentation, water stratification, temperature, salinity, and ice distribution and quality. These in turn affect the biological communities of benthic and pelagic organisms. More recently Sly (1995) listed a number of human activities within the drainage with potential to exert toxic chemical stress on Hudson Bay (Table 16-1). Most of these activities have some potential to contribute loadings of chemical substances to the drainage basins.

The study of elements in sediments is often more complex than the study of synthetic materials like cesium-137 or organic pesticides. The presence of a natural element is not, in itself, evidence of human activity because many elements are present naturally in soils and sediments. However, human activities often move elements about from place to place in the environment in ways and at rates not found naturally. The questions that arise in cases of suspected contamination by elements are whether the amounts found exceed natural background amounts and what the sources might be. Among the more toxic elements are cadmium, copper, chromium, lead, mercury, and zinc and the metalloids arsenic and selenium. In view of the potential for these elements to produce biological harm, Canada has established Interim Sediment Quality Guidelines (ISQG) and Probable Effect Levels (PEL) (Table 16-2). These guidelines describe levels that should not be exceeded in freshwater and marine sediments in order to avoid biological impacts.

The watersheds draining to Hudson Bay are being changed by human activities, notably population growth and associated business activity, agriculture, hydroelectricity, and climate change. In addition to alterations in watersheds, there is direct loading to the water surface of Hudson Bay by materials dispersed via atmospheric circulation. With the relatively limited attention contaminants have been given in Hudson Bay, existing analyses and those done in the coming few years have ‘benchmark’ quality that will help to assess the magnitude and significance of future changes.

Chemical contaminants often become incorporated into aquatic sediments. As the sediments accumulate over time, they provide an archive of past and present inputs of the contaminants (e.g., Haworth and Lund 1984). Only a few studies of this nature have been done in Hudson Bay. For example, Hermanson has studied sediment cores from small lakes in the Belcher Islands and reported histories of contamination by several elements (Hermanson 1993).

Table 16-1. Human activities and their potential to exert toxic chemical stress on the Hudson Bay ecosystem (from Sly 1995).

Activity	Potential for impact
forestry	low
agriculture	low
local mining	medium-high
distant mining	low
local oil and gas activity	low - high
distant oil and gas activity	low
pulp and paper	low - medium
local hydroelectricity	medium - high
distant hydroelectricity	low to medium
Transmission (hydroelectricity)	low to medium
Air transport	low
Shipping	low to high
Roadway	low to medium
Rail transportation	low to medium
Construction sites	medium to high
Tourism/Recreation	
Population growth	low to high
LRTAP*	low to high
Global warming	

*Long Range Transport of Atmospheric Pollutants

16.3 PRIMARY OBJECTIVE

The primary objective of this section is to compare the data on several elements reported by Henderson (1989) with Interim Sediment Quality Guidelines that were developed after the thesis was written.

16.4 SECONDARY OBJECTIVE

In addition, the Northern Contaminants Program recently released its summary of research on contaminants in the Canadian North for the past five years. Excerpts from that document and some unpublished data are included to provide a current overview of our knowledge of contaminants in biota.

Table 16-2. Interim sediment quality guidelines (ISQG) for marine sediments in Canada and concentrations described as "Probable effect levels" (PEL).

Element	Interim Sediment Quality Guideline for marine sediments ($\mu g \cdot g^{-1}$ dry weight)	Probable Effect Level for in marine sediment ($\mu \cdot g^{-1}$ dry weight)
Arsenic	7.24	41.6
Cadmium	0.7	4.2
Chromium	52.3	160
Copper	18.7	108
Lead	30.2	112
Mercury	0.13	0.7
Zinc	124	271

Canadian Council of Ministers of the Environment, 2001, Canadian sediment quality guidelines for the protection of aquatic life: Summary tables.

16.5 DATA FROM HENDERSON (1989) PH.D. THESIS

Data presented in the thesis by Henderson (1989) were derived from the analysis of dredge samples of surface sediment collected during several oceanographic surveys of Hudson Bay over the period from 1961 to 1986, mainly with geological objectives in mind. Dredge samples were fractionated into four particle-size classes (Table 16-3). Information was given on geographic position, water depth, and particle-size classes for most samples. The geographic positions where samples were obtained were presented as UTM (Universal Transverse Mercator) coordinates. Since Hudson Bay spans several UTM zones, it was convenient to express the position data in latitude and longitude. UTM coordinates were converted to longitude and latitude coordinates by M. Ouellette, Freshwater Institute, using MapInfo software. The datum used for the conversion was NAD83. Undoubtedly some of the older samples would have been described with reference to the older NAD27 datum and so the positional data listed in latitude and longitude may contain small errors in addition to any present in the original UTM coordinates. In proportion to the size of Hudson Bay, such errors are insignificant.

Table 16-3. Sediment fractions described by Henderson (1989).

Fraction	Description
>2 mm	Weight per cent of total sample larger than 2 mm in size
Sand	Weight per cent sand (0.063 – 2 mm) of matrix fraction < 2 mm
Silt	Weight per cent silt (0.004 – 0.063 mm) of matrix fraction < 2 mm
Clay	Weight per cent clay (<0.004 mm) of matrix fraction < 2 mm

Henderson, 1989, Appendix C

Several elements were analyzed in the clay-size fraction of some of the samples. This sub-set was drawn from samples collected from 1965 to 1986. Since different samples were sometimes analyzed for different variables, there are some variations in the numbers of samples available for pair-wise tabulations and mapping. Some samples were analyzed in duplicate or triplicate and reported more than once in the original tables; in those instances, the first result listed was used and any further replicates were ignored. In a few instances, apparent inconsistencies, probably of typographical origin, were found and were interpreted with the author's best judgment. This resulted in a set of 114 samples widely distributed throughout Hudson Bay for which metals data were available. Data were copied from the original thesis tables into Microsoft Excel, checked manually, and then transferred electronically to a table in Microsoft Access format. Data were sorted and drawn from the Access table as needed. Appendix 5 lists the positions, depths and particle sizes of the sediments at sites for which metals were determined. Appendix 6 lists the concentrations of metals found in the sediments.

Three types of presentation follow:

- tabular comparison with sediment quality guidelines for elements for which these guidelines have been established,
- pair-wise correlation analysis for associations between pairs of elements, and
- spatial distribution maps.

16.6 HENDERSON THESIS DATA AND INTERIM SEDIMENT QUALITY GUIDELINES (ISQG)

The Canadian Interim Sediment Quality Guidelines (ISQG) and Probable Effect Levels (PEL) were listed in Table 16-2. The guideline figures are given in terms of total sediment whereas the figures from Henderson (1989) were for the clay-size particles only. Consequently, a straightforward comparison is not possible. Lacking analyses of the metal content of other sizes of particles, it is not possible to express Henderson's results in the same units as the ISQG. However, it is likely that the error is conservative because metals are usually more abundant in the clay-size particles than in larger particles. Assuming that to be the case, the values from Henderson (1989) are higher than they would be in unfractionated, bulk sediment.

16.7 COMPARISON OF HUDSON BAY VALUES WITH ISQG

Interim Sediment Quality Guidelines have been established for five of the elements reported by Henderson (1989). Table 16-4 shows the number of sediment samples for which the clay-size particles fell in each of the three ranges defined by the ISQG: below the ISQG, between the ISQG and the PEL, and above the PEL.

Sediment quality guidelines have not been established for the remaining metals reported by Henderson (1989). In addition, some comparisons of Henderson's (1989) data are made with the same elements in sediments from lakes and streams, based on the report of many thousands of samples by Painter et al. (1994).

Table 16-4. Interim sediment quality guidelines (ISQG) and probable effect levels (PEL) for five elements in Canada and numbers of samples in Henderson (1989) in ranges defined by ISQG and PEL.

Element	ISQG ($\mu g \cdot g^{-1}$ dry weight, whole sediment)	PEL ($\mu g \cdot g^{-1}$ dry weight, whole sediment)	Number Below ISQG (in clay- size fraction)	Number above ISQG & below PEL (in clay-size fraction)	Number above PEL (in clay- size fraction)
Arsenic	7.24	41.6	103	10	1
Chromium	52.3	160	0	78	36
Copper	18.7	108	0	112	2
Lead	30.2	112	5	108	1
Zinc	124	271	4	110	0

16.7.1 Arsenic

Arsenic (As) in 114 samples averaged 3.7 $\mu g \cdot g^{-1}$ with a range of 2 to 77 $\mu g \cdot g^{-1}$. Arsenic values are plotted the maps in Figure 16-4. Most of the samples (92) were reported as 2 $\mu g \cdot g^{-1}$, probably indicating that they were too close to limits of measurement to be reliable. The analytical detection limit for arsenic was given as 5 $\mu g \cdot g^{-1}$ and so relatively little confidence might be given to values tabulated as 2 $\mu g \cdot g^{-1}$. Ten samples fell between the ISQG of 7.64 $\mu g \cdot g^{-1}$ and the PEL of 41.6 $\mu g \cdot g^{-1}$ (Table 16-4). One sample only exceeded the PEL and that one may have been an analytical or typographical error since it was far outside the range of the other samples. That sample (65TH 0466, Appendix 6) was reported to be 77 $\mu g \cdot g^{-1}$, well beyond the range of all the others. (The sample with the second highest level was only 17 $\mu g \cdot g^{-1}$). Although the values given by Henderson as 2 $\mu g \cdot g^{-1}$ may be open to some question quantitatively, there is still information in them. Results below the detection limit of 5 $\mu g \cdot g^{-1}$ are also below the ISQG of 7.74 $\mu g \cdot g^{-1}$, hence most samples contained arsenic in the clay-size fraction well below the ISQG. Twenty-two samples only had arsenic at 5 $\mu g \cdot g^{-1}$ or greater. Comparing the results with the ISQG, 11 samples exceeded 7.61 $\mu g \cdot g^{-1}$, and one of them exceeded the PEL.

Arsenic in the clay size particles had no statistical correlation to water depth and little relationship to other metals in the same particles (Appendix 7). (As was weakly correlated with potassium.) Considering the geographic distribution of points, the sites where arsenic exceeded the ISQG (Figure 16-4 (left panel)) were mostly in the central area of Hudson Bay; there was no concentration of points that might suggest a potential source of sediment enriched in arsenic. The bars on the map in Figure 16-4 (right panel) show more clearly that the points with the highest concentrations of arsenic are located mostly in the central part of Hudson Bay.

Painter et al. (1994) reported arsenic levels in 17,088 lake and stream sediment samples from widely scattered areas of Canada. The median value was 1.9 $\mu g \cdot g^{-1}$ and the 95th percentile was 22 $\mu g \cdot g^{-1}$. Their data were presented as coloured areas on maps (Figure 16-5) and also as a graph. From their graph, it is evident that about 90 per cent of samples had arsenic values below the ISQG. Figure 16-5 shows the distribution of arsenic values includes some regions in Hudson Bay drainage. Most of the values were below 6 $\mu g \cdot g^{-1}$ with the exception a striking anomaly on central Baffin Island called the Foxe Fold Belt where values were often above 21 $\mu g \cdot g^{-1}$. Virtually all values from the large region west of Hudson Bay, much of it in Churchill drainage, were below 6 $\mu g \cdot g^{-1}$.

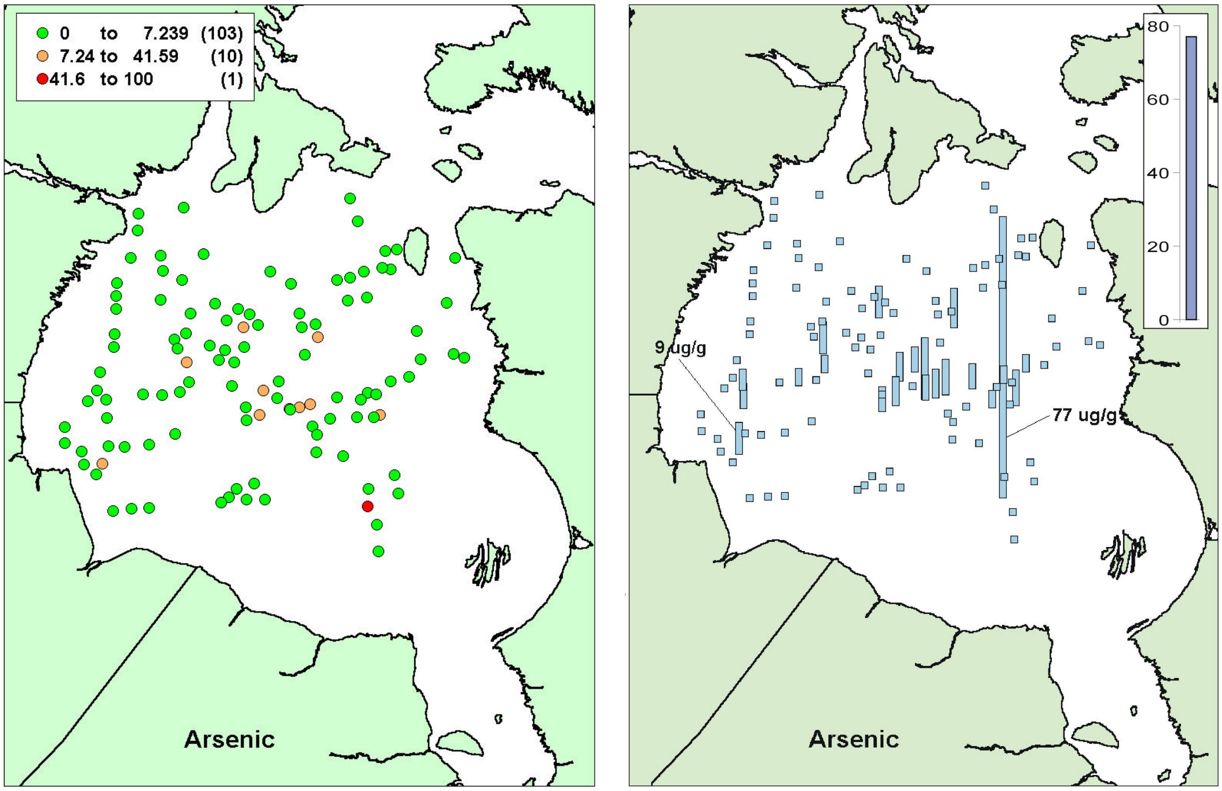


Figure 16-4. Arsenic (µg·g^{-1} dry weight) in clay-size particles of surficial sediment from Hudson Bay. Data from Henderson (1989). Ranges are those from the Canadian ISQG (=7.24 µg·g^{-1} dry weight whole, unfractionated sediment) and PEL (=41.6 µg·g^{-1}). Bar heights are scaled to concentrations of arsenic.

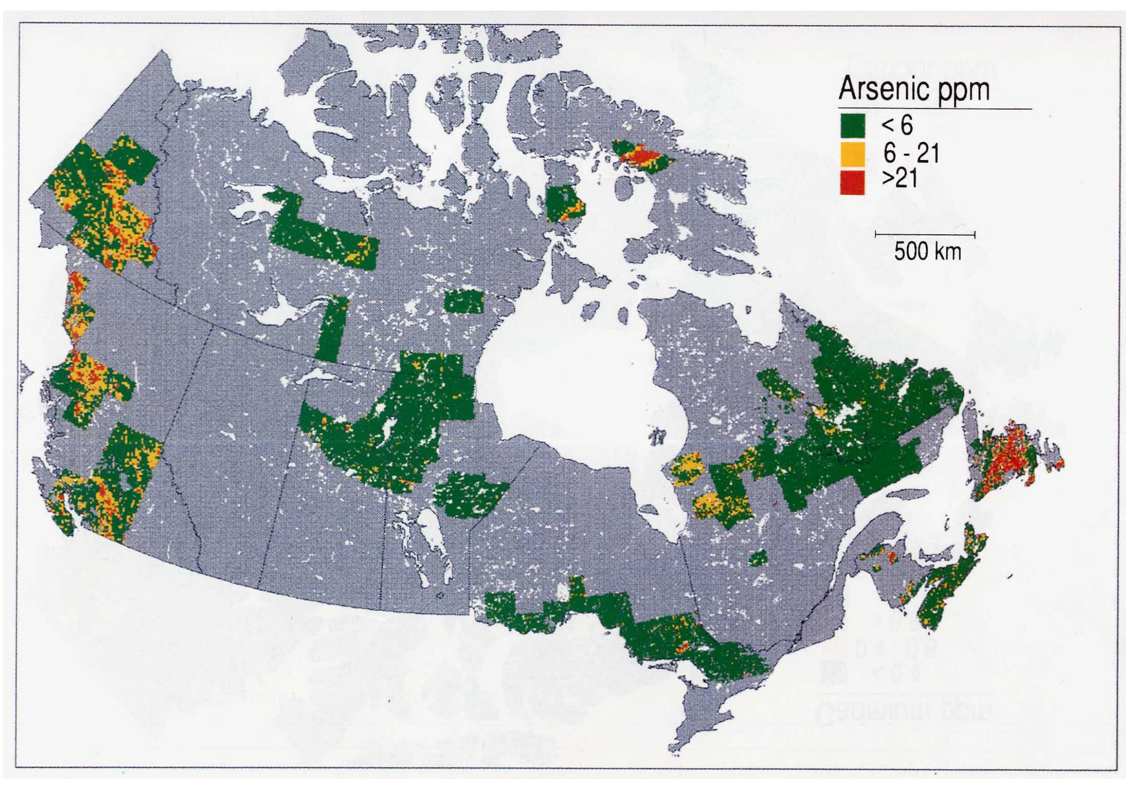


Figure 16-5. Arsenic in lake and stream sediment samples. (From Painter et al. 1994, p. 223).

It seems unlikely that the amounts of arsenic found currently in Hudson Bay clay-size sediments represent a significant risk to the biological community there. If whole, unfractionated sediment were analyzed, it would probably have lower concentrations of most elements than the clay-size particles reported by Henderson (1989). It seems likely that the levels reported indicate a natural background for arsenic in fine sediment particles distributed by natural processes.

16.7.2 Chromium

The 114 samples analyzed for chromium (Cr) ranged from 99 to 274 µg·g^{-1} with a mean value of 152 µg·g^{-1}. No samples contained chromium below the ISQG concentration of 52.3 µg·g^{-1}. Most of the samples (78) fell between the ISQG value and the PEL value of 160 µg·g^{-1} and 36 exceeded the PEL (Table 16-4). The detection level for chromium was 1 µg·g^{-1} and so all the values were well above the detection level and should be reliable. Chromium levels tended to be higher in shallow water than in deeper water (Appendix 7, r = –0.27). Chromium levels correlated with a suite of other elements, (Mg, Fe, Ni, Cu, Zn, Pb). It was negatively correlated with depth, Ca and Mn. The geographical distribution of samples in each concentration range is shown on the map in Figure 16-6. Most of the sites with chromium over the PEL were offshore from the southwestern part of Hudson Bay suggesting the possibility of chromium-enriched sediments originating from drainages entering Hudson Bay from the west.

With the clay-size sediments consistently exceeding the ISQG and even the PEL, it is desirable to obtain new analyses of bulk, unfractionated sediment in order to compare results directly with the ISQG and PEL. It is not known whether the existing levels in the sediment represent a risk to the biota of southwestern Hudson Bay. A more rigorous interpretation of the results for chromium (and other metals) will await further analysis of unfractionated sediment, and characterization of the oxidation state(s) of chromium in the sediments.

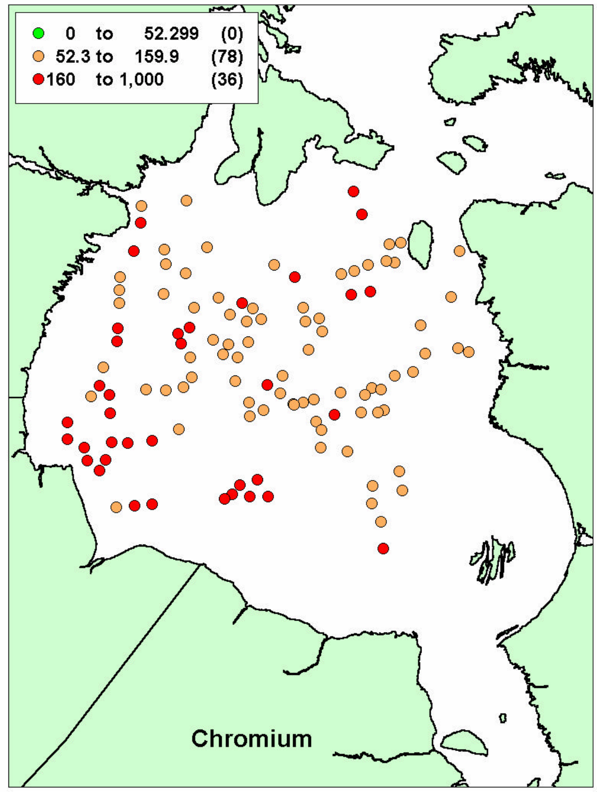


Figure 16-6. Chromium (µg·g^{-1} dry weight) in clay-size particles of surficial sediment from Hudson Bay. Data from Henderson (1989). Ranges are those from the ISQG (=52.3 µg·g^{-1}) based on unfractionated, whole sediment, and the PEL (=160 µg·g^{-1}).

Painter et al. (1994) provided some graphical information on chromium in 65,948 samples of lake and stream sediment. The median value was 32 µg·g^{-1} and the 95th percentile was 120 µg·g^{-1}. About 70% of sediment samples fell below the ISQG. Relative to these whole sediment values, the clay-size fraction in sediment from Hudson Bay is enriched in chromium.

16.7.3 Copper

Copper (Cu) concentrations in 114 samples of clay-size sediment averaged 41 µg·g^{-1} with a minimum of 22 µg·g^{-1} and a maximum of 170 µg·g^{-1}. There were no values under the ISQG of 18.7 µg·g^{-1}; almost all values (112 of 114) fell between the ISQG and the PEL of 108 µg·g^{-1} (Table 16-4). Two sites had copper concentrations over the PEL. Cu values correlated negatively with depth but not with other metals. Geographically the points are plotted in Figure 16-7 (left panel) and all but two of the points on the map appear the same because the ranges were selected to describe the ISQG and PEL. Figure 16-7 (right panel) presents the same data as bar graphs so that high and low values within a range are more apparent. In general, the highest values were found off the west coast of Hudson Bay near the Churchill River. Henderson (1989) noted the presence of copper enrichment in several places between Chesterfield Inlet and Rankin Inlet, and also north of Arviat. She suggested that the distribution of copper can be explained by sediment transport offshore from the District of Keewatin.

Painter et al. (1994) reported copper in 253,682 samples of lake and stream sediments. The median concentration was 20 $\mu g \cdot g^{-1}$, slightly over the ISQG of 18.7, and the 95th percentile value was 76 $\mu g \cdot g^{-1}$. The distribution map by Painter et al. (1994) for copper in lake and stream sediments is shown in Figure 16-8. Areas of enriched copper in sediments of Foxe Basin drainages were identified, one on central Baffin Island and the other on the Melville Peninsula. The units for the Painter et al. (1994) study are directly comparable with the ISQG and so all of the red and yellow areas in Figure 16-8 and some of the green points exceed the ISQG.

As with chromium, it is desirable to determine copper in Hudson Bay sediments in the same units of measurement as the ISQG. The clay-size particles exceed the ISQG and some also exceed the PEL. However, the sediments may not exceed these criteria on a bulk sediment basis. *McCrea* et al. (1984) determined total copper in 18 dredge samples of bulk surficial sediments from five Ontario rivers at points just before they enter Hudson Bay or James Bay. Copper concentrations in those samples averaged 6.2 $\mu g \cdot g^{-1}$ and ranged from 3 to 19 $\mu g \cdot g^{-1}$, all lower than those found in the clay-size sediments of Hudson Bay.

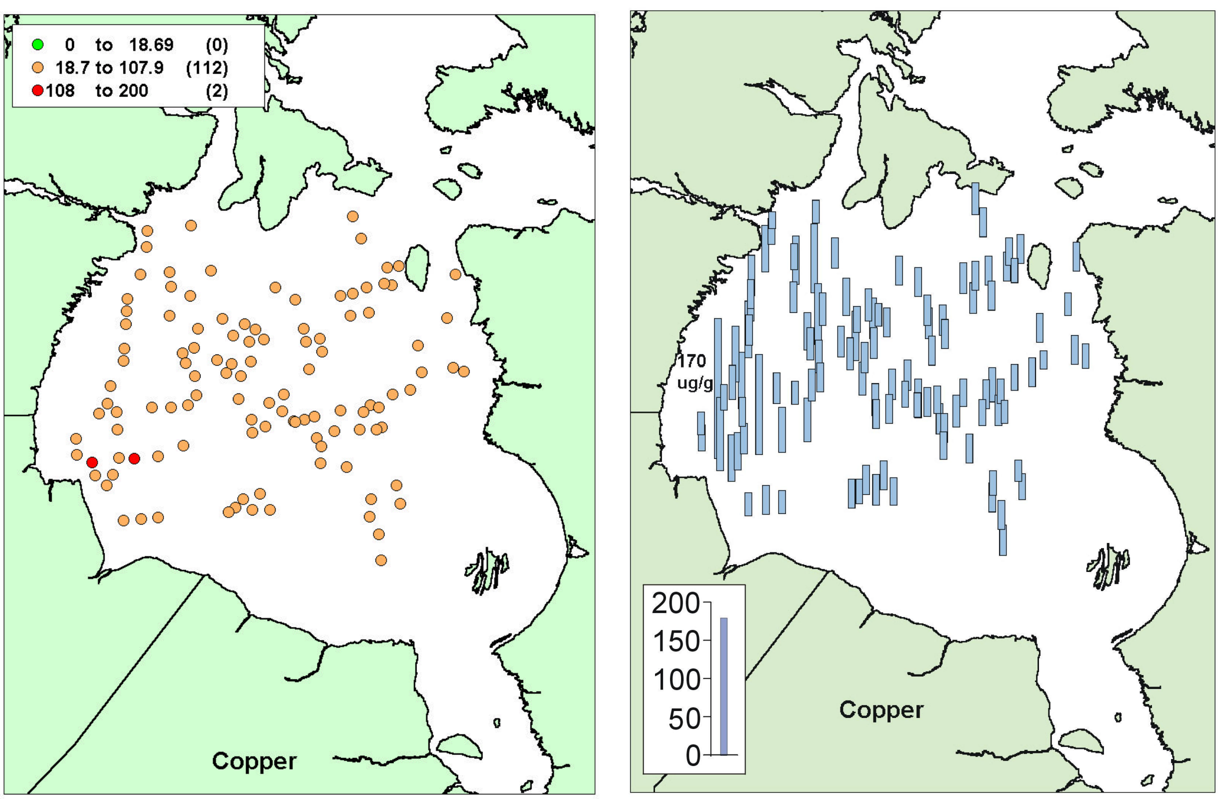


Figure 16-7. Copper ($\mu g \cdot g^{-1}$ dry weight) in clay-size particles of surficial sediment from Hudson Bay. Data from Henderson (1989). Ranges are those from the ISQG (=18.7 $\mu g \cdot g^{-1}$) and the PEL (=108 $\mu g \cdot g^{-1}$). Bar heights are scaled to concentrations of copper.

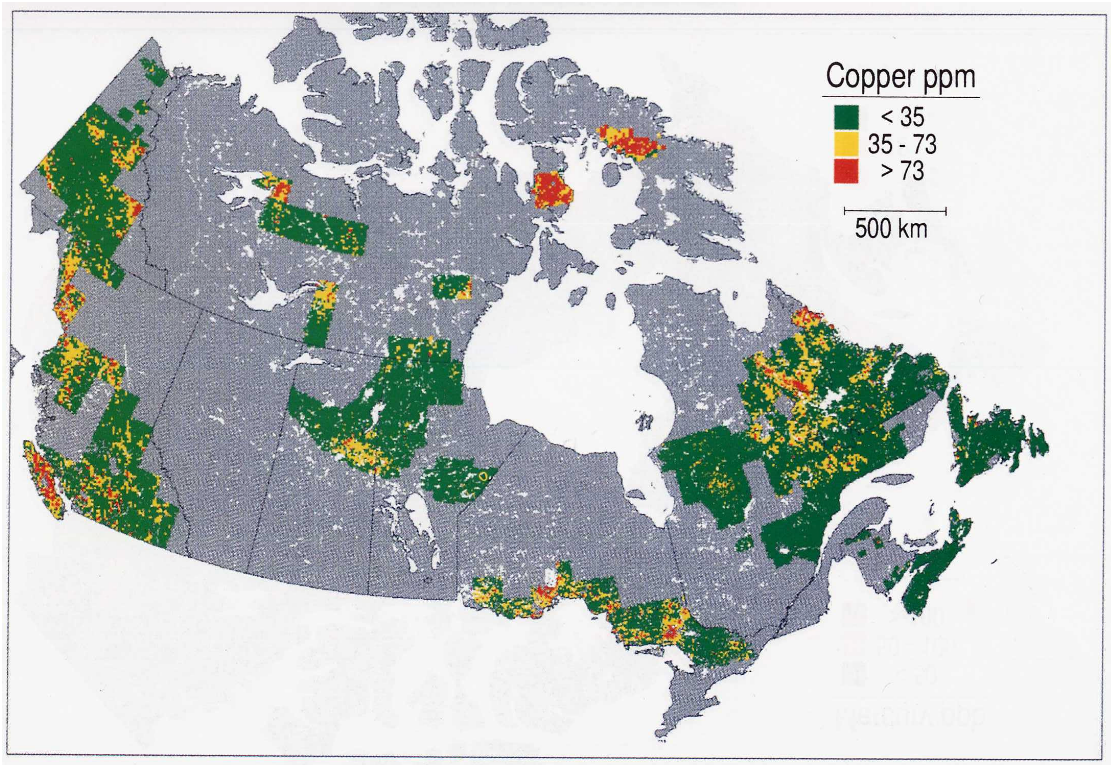


Figure 16-8. Copper in lake and stream sediment samples (From Painter et al. 1994, page 223).

16.7.4 Lead

Lead (Pb) in clay-size sediment particles ranged from 27 to 137 $\mu g \cdot g^{-1}$ with a mean value of 45 $\mu g \cdot g^{-1}$. Five sites had lead at concentrations lower than the ISQG of 30.2 $\mu g \cdot g^{-1}$ and 108 sites fell between the ISQG and the PEL of 112 $\mu g \cdot g^{-1}$ (Table 16-4). One site only exceeded the PEL. There was no relationship between lead in the sediment and the depth of water at the sites. Lead levels correlated with a few other metals (Co, Cr, Ni, Zn, Appendix 7). The geographical distribution of lead levels is shown in Figure 16-9 (left panel). This figure does not display the pattern of lead levels well because almost all the samples fell within the range between the ISQG and the PEL. Figure 16-9 (right panel) shows the concentrations of lead as bar graphs with no obvious geographic clustering of high or low values.

We have some additional data on profiles of lead in sediment cores collected in southeastern Hudson Bay in 1992 and 1993 (Figure 16-10). The values for lead in these cores ranged up to about 12 $\mu g \cdot g^{-1}$. The lead-210 profile for dating the layers of core Hud-4 was shown in Figure 16-1 and it had the expected exponential decline in Pb-210 with depth. Core Hud-4 also showed an increase in stable lead near the top of the core. Given the dating profile in Figure 16-1, the most likely interpretation of the stable lead profile in Figure 16-10 (right panel) is that inputs of lead to that site increased during the 20th century relative to pre-industrial times. The other two lead profiles shown in Figure 16-10 (Hud-10 and Fogo-4) give no indication of increasing amounts of lead in the upper slices. These two cores (Hud-10 and Fogo-4) had more extensively mixed upper layers (possibly by sediment movements and biological mixing) and so gradients in stable lead would have been unable to form or, if formed, would have been obscured or eliminated by mixing. Core Hud-4 is similar to a core from Imativik Lake in the Belcher Islands where Hermanson (1993) found that inputs of lead have increased about 3.5-fold over historical inputs. Hermanson (1993) also noted that the anticipated decline in inputs of lead due to the phasing out of lead additives to motor fuels in the early 1970s has apparently not been reflected in the core from Imativik Lake. We have results similar to those from Imativik Lake with sediment cores from Hawk Lake and Far Lake at the Saqvaqjuac research site on the west coast of Hudson Bay just north of Chesterfield Inlet (Lockhart, unpublished data). Furthermore, stable lead isotope analysis of the Far Lake core indicates that some of the inputs of lead are of anthropogenic origin (P. Outridge, unpublished data).

Painter et al. (1994) described lead in 253,846 samples of lake and stream sediments. The median value was 6 $\mu g \cdot g^{-1}$ and the 95th percentile was 25 $\mu g \cdot g^{-1}$. Figure 16-11 shows the distribution of lead in lake and stream sediments reported by Painter et al. (1994) and their results for regions near Hudson Bay and Foxe Basin are quite similar to those for copper. Given the ISQG of 30.2 $\mu g \cdot g^{-1}$, clearly very few samples of sediment from lakes and streams exceed the ISQG. McCrea et al. (1984) analyzed for lead in the sediments of the Moose, Albany, Attawapiskat, Moose, and Severn rivers at points just before entering the sea and found an average concentration of only 11.4 $\mu g \cdot g^{-1}$ (range 7-17 $\mu g \cdot g^{-1}$), below the values for clay-size sediment from Hudson Bay.

The most likely conclusion is that Hudson Bay has received inputs of anthropogenic lead over the last century probably from atmospheric fallout. The clay-size sediments frequently contain concentrations of lead above the ISQG. Probably this same argument applies to rivers flowing into Hudson Bay but the concentrations of lead in bulk sediments from the Ontario rivers were relatively low.

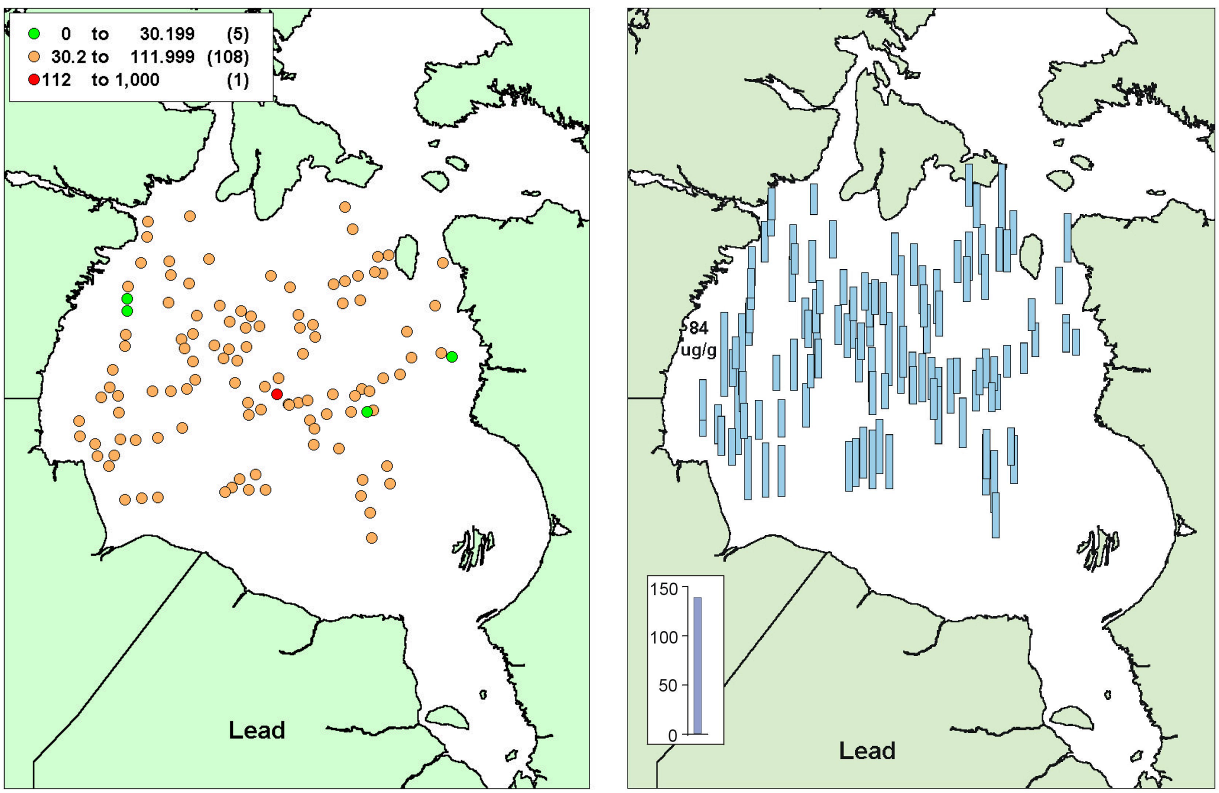


Figure 16-9. **Lead ($\mu g \cdot g^{-1}$ dry weight) in clay-size particles of surficial sediment from Hudson Bay. Data from Henderson (1989). Ranges are those from the ISQG (=30.2 $\mu g \cdot g^{-1}$) and the PEL (=112 $\mu g \cdot g^{-1}$). Bar heights are scaled to concentrations of lead.**

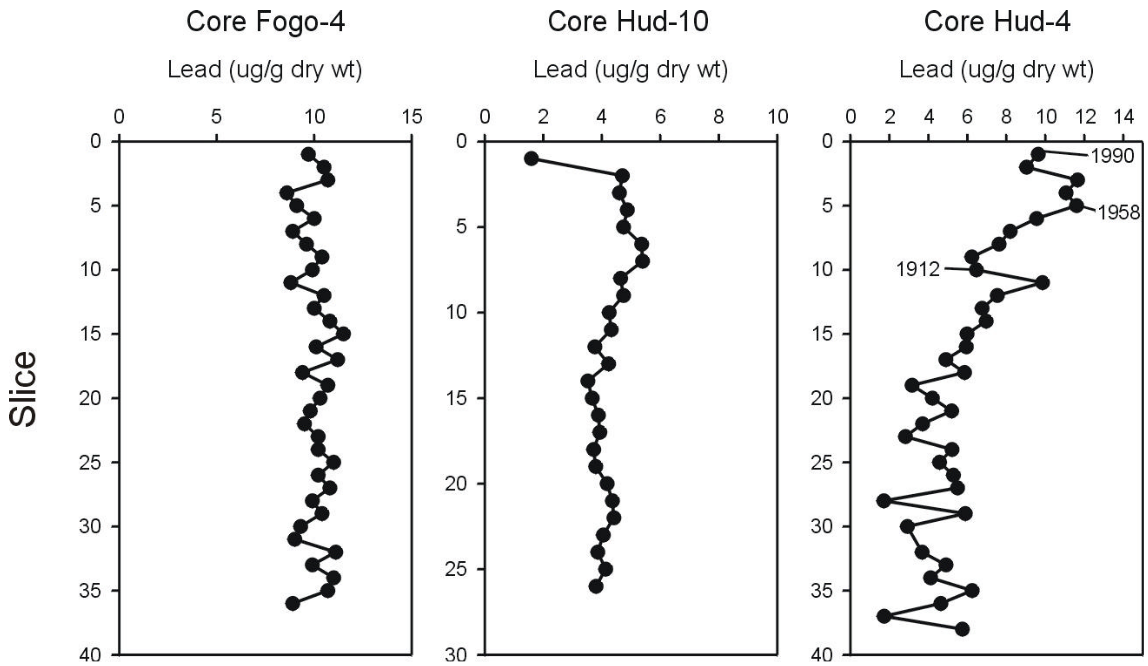


Figure 16-10. Down-core profiles of lead in three sediment cores from southeastern Hudson Bay. Cores Hud-4 and Hud-10 were collected in 1992 and core Fogo-4 was collected in 1993 (L. Lockhart, unpublished data). Sampling locations are shown in Figure 16-24.

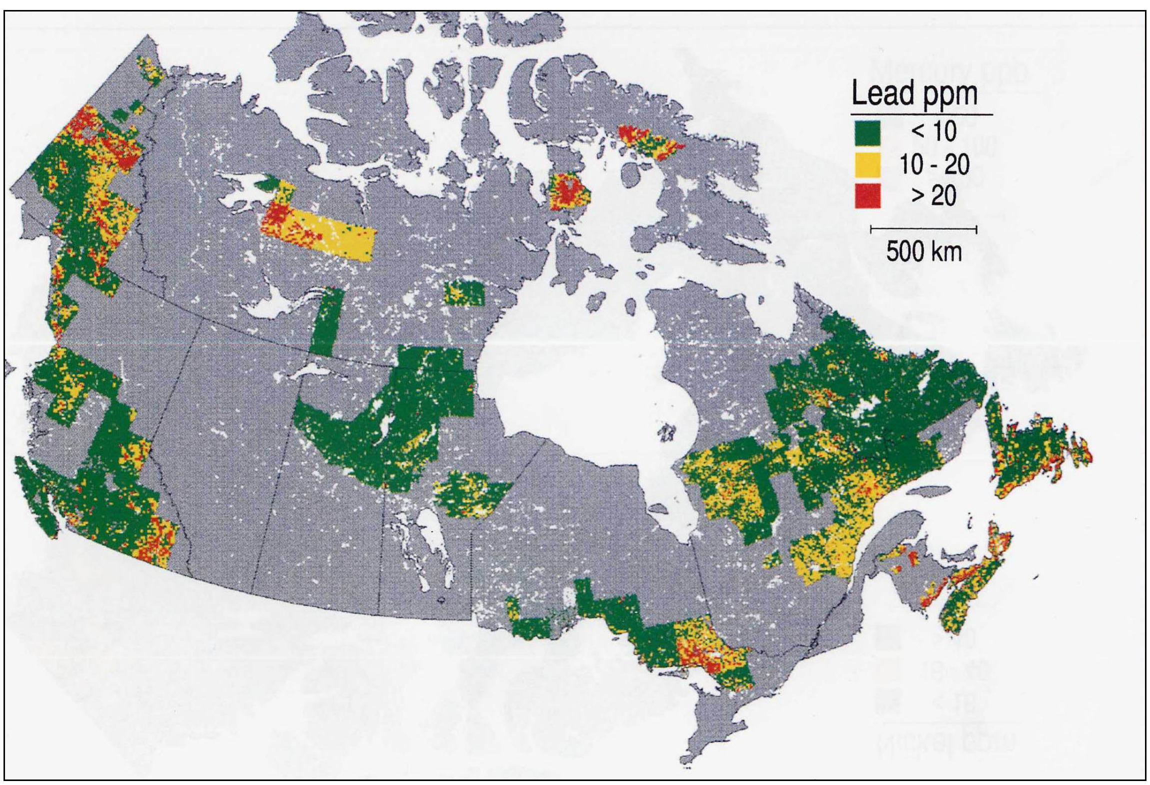


Figure 16-11. Lead in lake and stream sediment samples (From Painter et al. 1994, page 223)

16.7.5 Zinc

The range in concentrations of zinc (Zn) was from 109 µg·g^{-1} to 227 µg·g^{-1} with a mean value of 167 µg·g^{-1}. Four samples only had zinc concentrations below the ISQG of 124 µg·g^{-1}. All remaining samples exceeded the ISQG but were below the PEL of 271 µg·g^{-1}. No samples exceeded the PEL (Table 16-4). Zinc levels correlated strongly with a number of other metals, notably Mg, Co, K, Fe, Cr, and Ni (Appendix 7). Zinc correlated negatively with Ca but had no correlation with depth. The geographic distribution of zinc levels is shown in Figure 16-12 (left panel). As with some other metals, most of the samples fell in a single range of values and so a second map has been included showing the values as bar graphs (Figure 16-12 (right panel)). There is no obvious clustering of high or low zinc values.

Painter et al. (1994) summarized 256,216 sediment results for zinc and found the median concentration to be 69 µg·g^{-1} with the 95th percentile value of 191 µg·g^{-1}. From the graphical presentation by Painter et al. (1994)(Figure 16-13), about 70 per cent of the values would fall below the ISQG level of µg·g^{-1}. The geographic distribution shown in Appendix 7 shows a large region southeast of Hudson Bay with values under 60 µg·g^{-1} (green). However, other areas in the Hudson Bay/Foxe Basin drainage are mostly yellow (60-120 µg·g^{-1}) or red (>120 µg·g^{-1}). Judging from the graphical presentation by Painter et al. (1994))(Figure 16-13), about 70 per cent of lake and stream sediment samples contained zinc at concentrations below the ISQG. McCrea et al. (1984) also measured zinc in the sediments from the five Ontario rivers. Zinc concentrations ranged from 23 to 83 µg·g^{-1} with a mean of 34.1 µg·g^{-1}. As with copper and lead, zinc concentrations from the rivers were also well below those from the clay-size sediments from Hudson Bay.

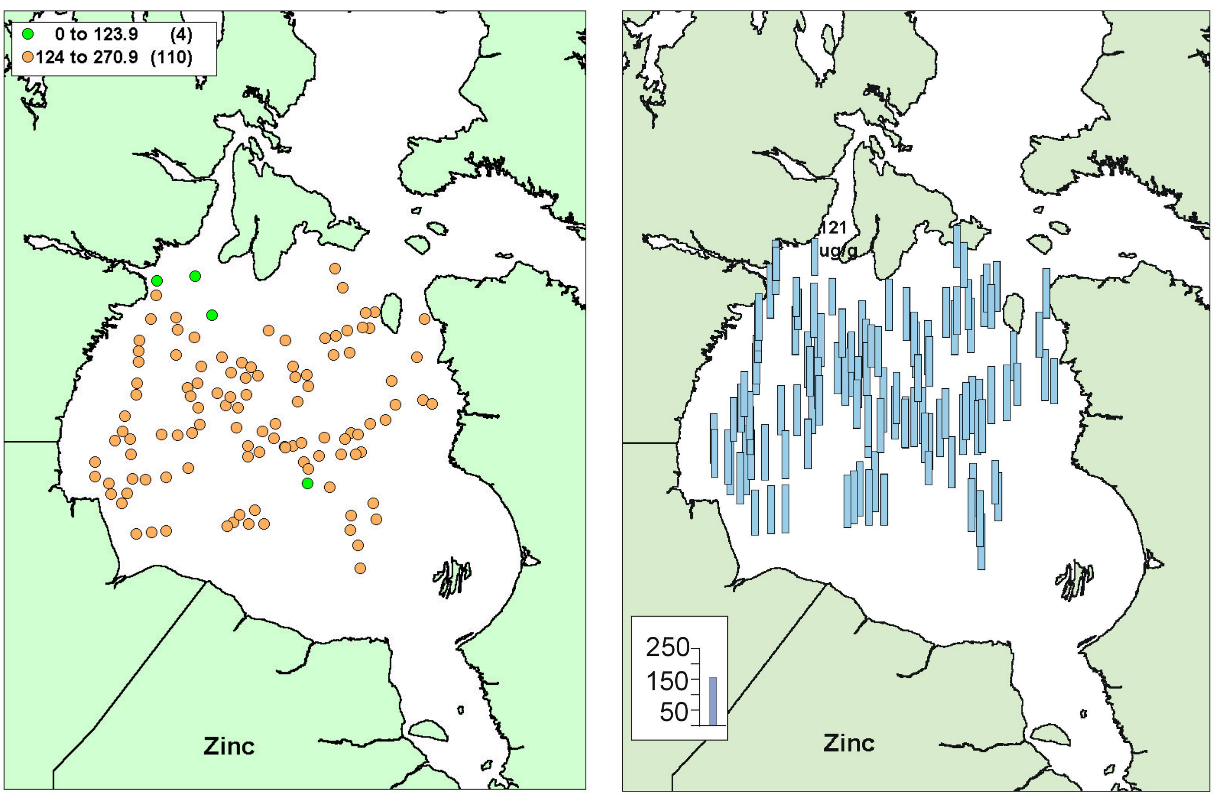


Figure 16-12. Zinc (µg·g^{-1} dry weight) in clay-size particles of surficial sediment from Hudson Bay. Data from Henderson (1989). Ranges are those from the ISAG (=124 µg·g^{-1}) and the PEL (=271 µg·g^{-1}). Bar heights are scaled to concentrations of zinc.

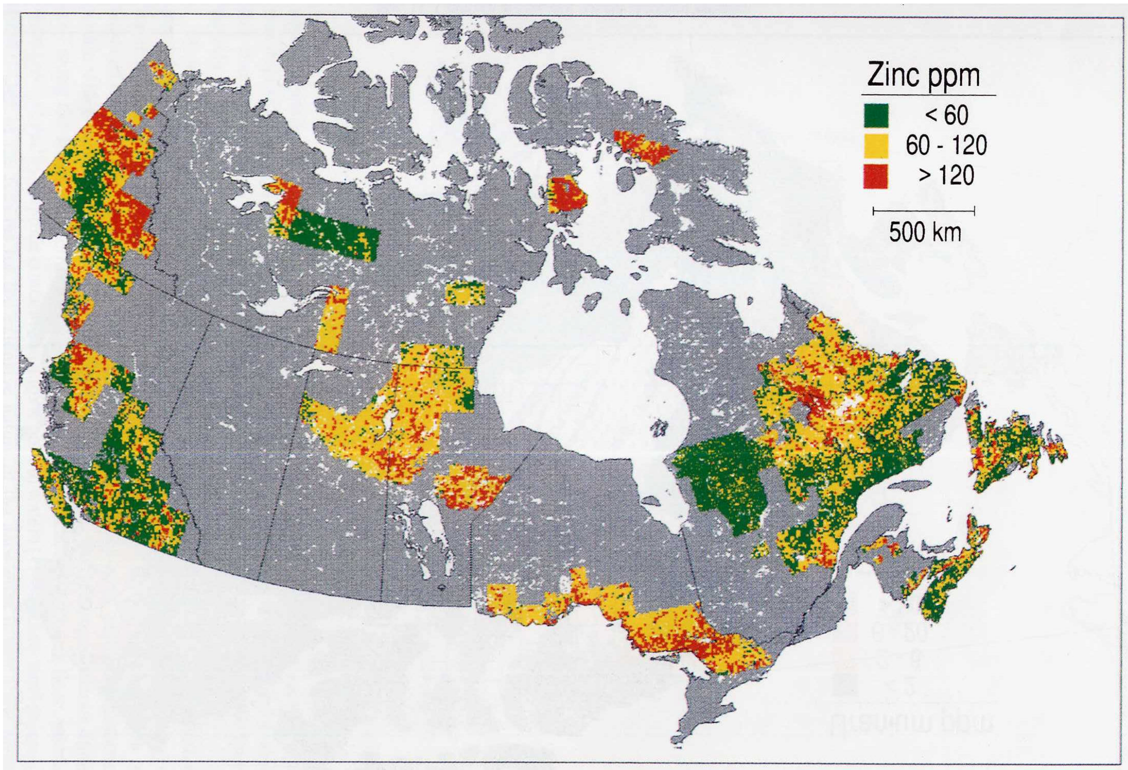


Figure 16-13. Zinc in lake and stream sediment samples (From Painter et al. 1994, page 223).

16.8 METALS FOR WHICH ISQG VALUES DO NOT APPLY

No Interim Sediment Quality Guidelines have been established for the remaining metals reported by Henderson: aluminum, calcium, cobalt, iron, magnesium, manganese, molybdenum, nickel, and potassium. Several of these are major components of the earth's crust and the concept of ISQG does not apply. Others are trace metals and ISQG values have not been established. The concentration ranges for these metals are shown in Table 16-5. The geographic distributions of these metals in the clay-size sediments of Hudson Bay are shown in Figure 16-14 through Figure 16-23. Extensive discussion of the distribution of these metals for which no guidelines have been established is beyond the scope of this report. The reader is referred to Henderson (1989) who points out that "the distribution of base metals can be related both to provenance and diagenetic processes." The major elements (measured in units of per cent by weight) all vary by about 3-fold from the minimum to the

Table 16-5. Ranges and means of metals in sediment samples from Hudson Bay. (Data from Henderson 1989).

Element	Units	n	Minimum	Maximum	Arithmetic Mean	Crustal abundance*
Aluminum	Per cent	114	3.15	9.78	6.53	8.1
Calcium	Per cent	114	0.8	5.54	1.78	3.6
Iron	Per cent	114	2.68	7.5	5.06	5.0
Magnesium	Per cent	114	1.37	3.93	2.64	2.1
Potassium	Per cent	114	1.69	4.96	3.34	2.6
Cobalt	$\mu g \cdot g^{-1}$	114	13	53	26.9	
Manganese	$\mu g \cdot g^{-1}$	114	0.04	1.0	0.12	
Molybdenum	$\mu g \cdot g^{-1}$	114	0.5	12	3.37	
Nickel	$\mu g \cdot g^{-1}$	114	51	107	74.0	

*Average crustal abundance figures from Strahler and Strahler 1978, page 202 maximum value except for calcium which varies by about 7-fold. Somewhat different spatial patterns are evident for the different major elements. These elements, along with oxygen, silicon, and sodium, are the major components of the earth's crust. Their presence and levels in the sediments of Hudson Bay do not infer anthropogenic impacts. Henderson (1989) conducted 13 mineralogical analyses of clay-size particles and most of those samples contained dolomite, feldspars, quartz, illite, kaolinite, and chlorite; some also contained calcite, amphibole, and smectite. The elements found in these minerals are shown in Table 16-6.

Table 16-6. Major elements found in the minerals identified in the clay-size fraction of surficial sediments from Hudson Bay. Sources for this information were web sites as listed.

Mineral	Major elements present in mineral
Dolomite	Calcium, magnesium, carbon, oxygen (http://wrgis.wr.usgs.gov/docs/parks/misc/glossaryAtoC.html#D)
Feldspars	Potassium, sodium, calcium, aluminum, silicon, oxygen (http://wrgis.wr.usgs.gov/docs/parks/misc/glossaryDtoI.html#F)
Quartz	Silicon, oxygen (http://wrgis.wr.usgs.gov/docs/parks/misc/glossaryDtoI.html#Q)
Illite	Potassium, magnesium, aluminum, iron, silicon, hydrogen, oxygen (http://www.webmineral.com/data/illite.shtml)
Kaolinite	Aluminum, silicon, oxygen, hydrogen (http://webmineral.com/data/Kaolinite.shtml)
Chlorite	Magnesium, iron, aluminum, silicon, oxygen, hydrogen (http://wrgis.wr.usgs.gov/docs/parks/misc/glossaryDtoI.html#C)
Calcite	Calcium, carbon, oxygen (http://wrgis.wr.usgs.gov/docs/parks/misc/glossaryAtoC.html#C)
Amphibole	Iron, magnesium, calcium, aluminum, silicon, oxygen, hydrogen (http://wrgis.wr.usgs.gov/docs/parks/misc/glossaryAtoC.html#C)
Smectite	Sodium, calcium, aluminum, magnesium, silicon, oxygen, hydrogen (http://www.reade.com/Products/Minerals_and_Ores/smectite.html)

16.9 MAJOR ELEMENTS (AL, Ca, Mg, Fe, K)

16.9.1 Aluminum

Aluminum (Al) was found in the clay-size sediment at about 3 to 10 per cent by weight with a mean of 6.5 per cent. Aluminum comprises about 8.1 per cent by weight of the earth's crust and so its average concentration in Hudson Bay fine sediments is slightly below its crustal average. Five of the minerals identified in the clay-size fraction contain aluminum (Table 16-6) and so its concentrations are expected to be relatively high. Concentrations of aluminum did not correlate with water depth although they correlated statistically with other major elements, magnesium, iron and potassium and with trace elements cobalt and nickel (Appendix 7). Aluminum is relatively uniformly distributed with high and low values scattered throughout the area sampled (Figure 16-14). Aluminum in the river sediments reported by McCrea et al. (1984) ranged from 0.9 to 4.5 per cent.

16.9.2 Calcium

Calcium (Ca), another major element, averaged almost 1.8 per cent by weight of the clay-size sediment with a range from 0.8% to 5.54% (Table 16-5). It was enriched in samples from the northern areas of Hudson Bay relative to those from central and southern areas (). Henderson (1989, page 142) reported that northern Hudson Bay sediments are characterized by high proportions of Paleozoic limestone clasts in the fine gravel fraction and so enrichment in calcium in that area is not surprising. As with aluminum, five of the mineral components of the clay-size fraction contain calcium and so its high overall abundance was expected. The crustal average for calcium is about 3.6 per cent by weight and so its average concentration in these fine sediments was lower than its crustal average. There was no correlation between calcium and depth. However, there were a number of negative pair correlations between calcium and other elements (Cr, Zn, Co, Fe, K, Ni; Appendix 7).

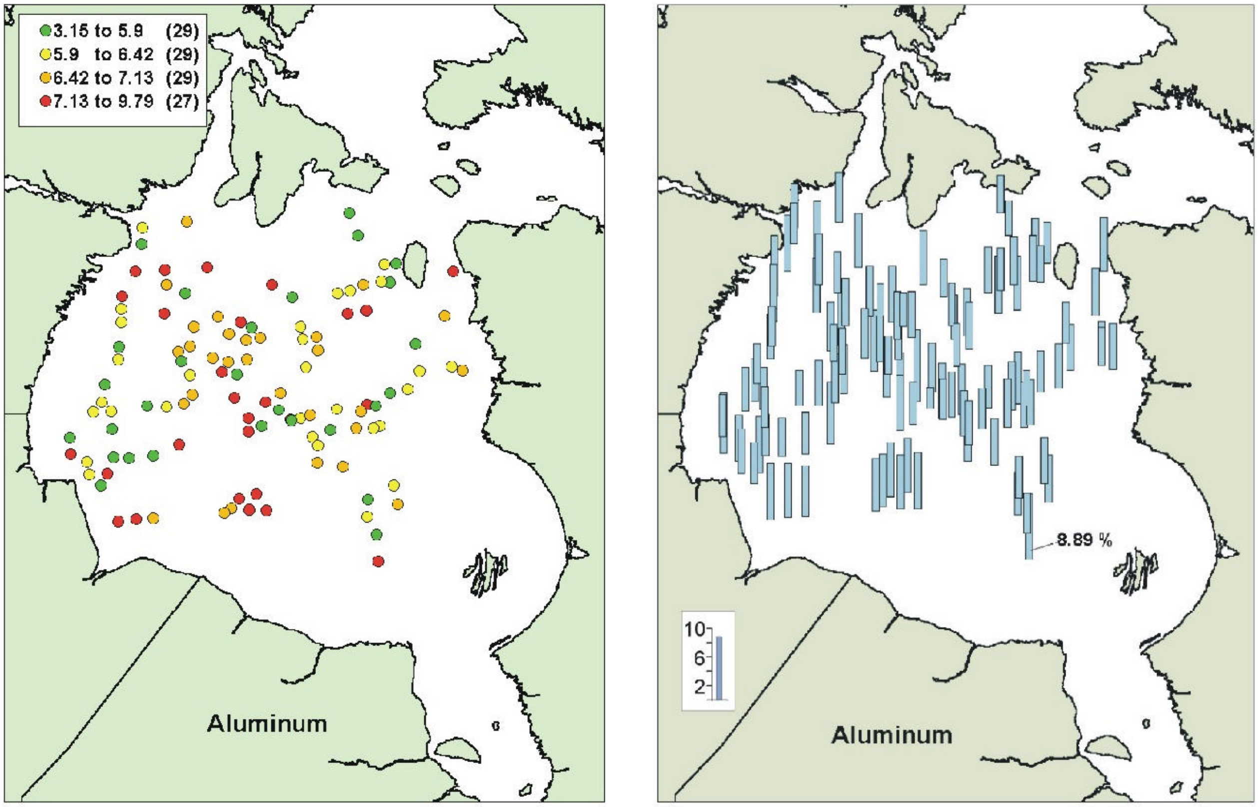


Figure 16-14. Aluminum (per cent dry weight) in clay-size particles of sediment from Hudson Bay. Data from Henderson (1989). Ranges selected to place approximately equal numbers of points in each range (left); same data shown as bars with bar height scaled to concentration of aluminum (right).

16.9.3 Magnesium

The average crustal content of magnesium (Mg) is 2.1 per cent by weight and concentrations in the fine sediments ranged from 1.37 to 3.93 per cent (Table 16-5) with a mean of 2.6 per cent. Its geographic distribution in the clay-size fraction (Figure 16-16) is somewhat similar to that for calcium but the higher values are shifted southward. Magnesium is abundant in five of the minerals listed in Table 16-6. Magnesium was not related to depth, but correlated positively with other elements (Appendix 7).

16.9.4 Iron

Iron ranged from 2.68 to 7.5 per cent with a mean value of 5.06 per cent, almost the same as the average crustal abundance of this element (Table 16-5). Iron occurs in illite and chlorite, minerals identified in the clay-size fraction of sediment (Table 16-6). Iron content was not related to depth but it was related statistically to most of the other elements. Its distribution (Figure 16-17) shows most of the high values in the central area of Hudson Bay, in common with several other elements. Like the other elements, iron was also less abundant in the river sediments than in the clay-size fraction from Hudson Bay. McCrea et al. (1984) found an average of 1.32 per cent iron in the river samples with a range from 0.56 to 3.2 $\mu g \cdot g^{-1}$.

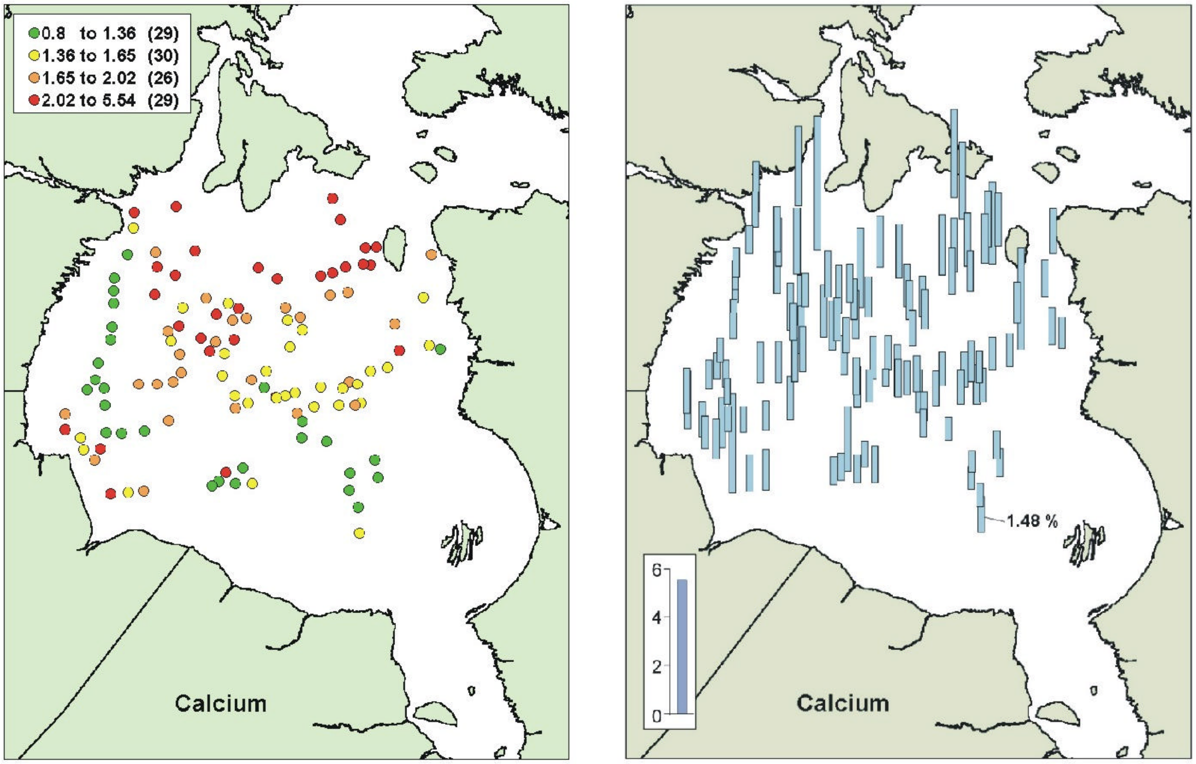


Figure 16-15. Calcium (per cent dry weight) in clay-size particles of sediment from Hudson Bay. Data from Henderson (1989). Ranges selected to place approximately equal numbers of points in each range (left); same data shown as bars with bar height scaled to concentration of calcium (right).

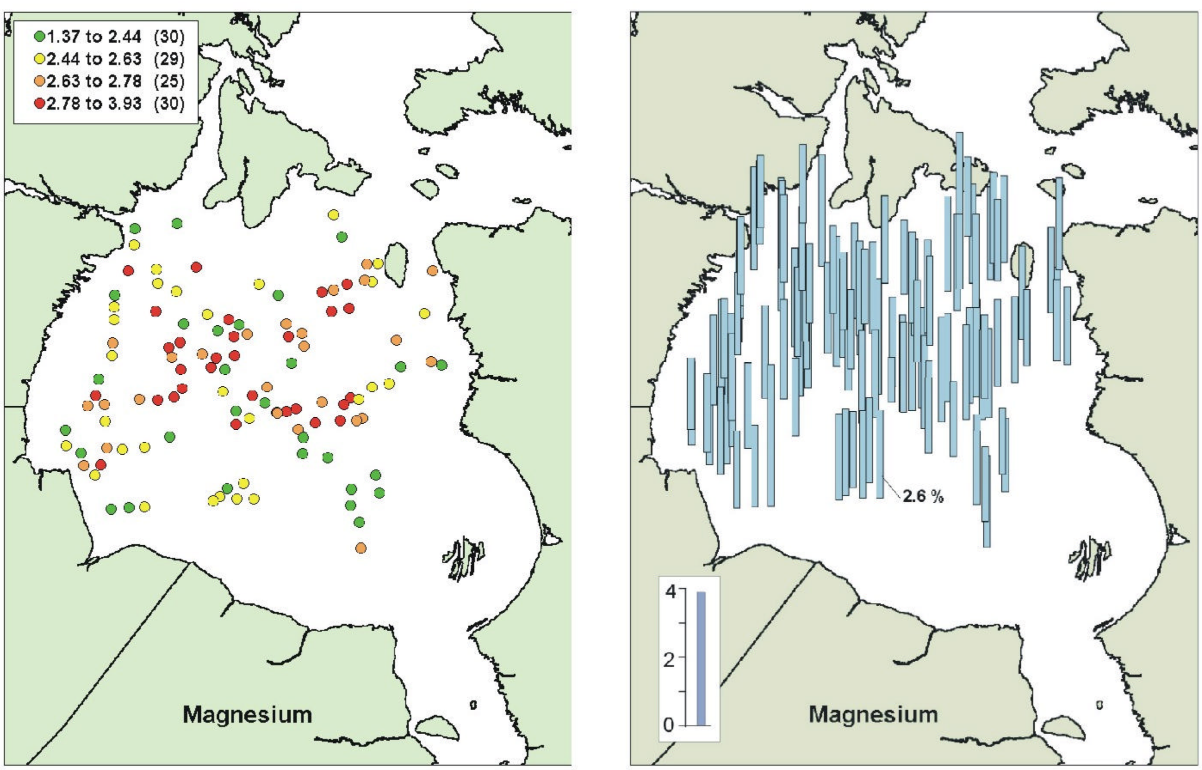


Figure 16-16. Magnesium (per cent dry weight) in clay-size particles of surficial sediment from Hudson Bay. Data from Henderson (1989). Ranges selected to place approximately equal numbers of points in each range (left); same data shown as bars with bar height scaled to concentration of magnesium (right).

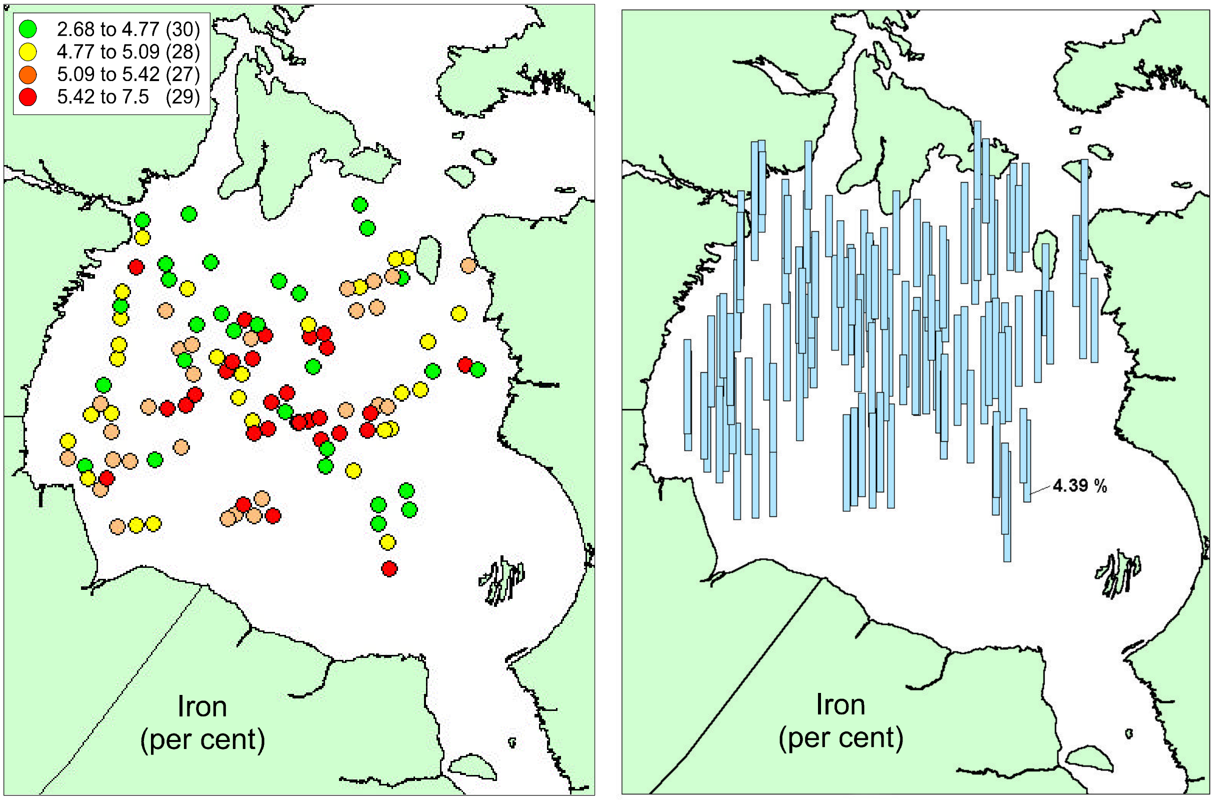


Figure 16-17. Iron (per cent dry weight) in clay-size particles of surficial sediment from Hudson Bay. Data from Henderson (1989). Ranges selected to place approximately equal numbers of points in each range (left); same data shown as bars with bar height scaled to concentration of iron (right).

16.9.5 <u>Potassium</u>

Potassium, the last of the major elements analyzed, was found at concentrations from 1.69 to 4.96 per cent with a mean of 3.34 per cent, somewhat above the crustal average of 2.6 per cent (Table 16-5). Potassium is present in feldspars and illite, two of the minerals most commonly identified in the clay-size fraction (Table 16-6). Statistically, potassium had no relation to water depth, but it correlated strongly with several other elements, especially iron (Appendix 7). Geographically, it had a cluster of high values west of the Belcher Islands unlike the other major elements (Figure 16-18) but somewhat like the trace element, nickel.

16.10 REMAINING ELEMENTS (Co, Mn, Mo, Ni)

16.10.1 <u>Cobalt</u>

The crustal abundance of elements is unhelpful as a guide for trace elements. Cobalt (Co), a trace element, was present in the 13-53 $\mu g \cdot g^{-1}$ range with a mean value of 27 $\mu g \cdot g^{-1}$ (Table 16-5). Most of the high values for cobalt were in the central part of Hudson Bay (Figure 16-19), in common with a number of other trace elements. Concentrations of cobalt correlated positively with water depth and with several other elements (Appendix 7).

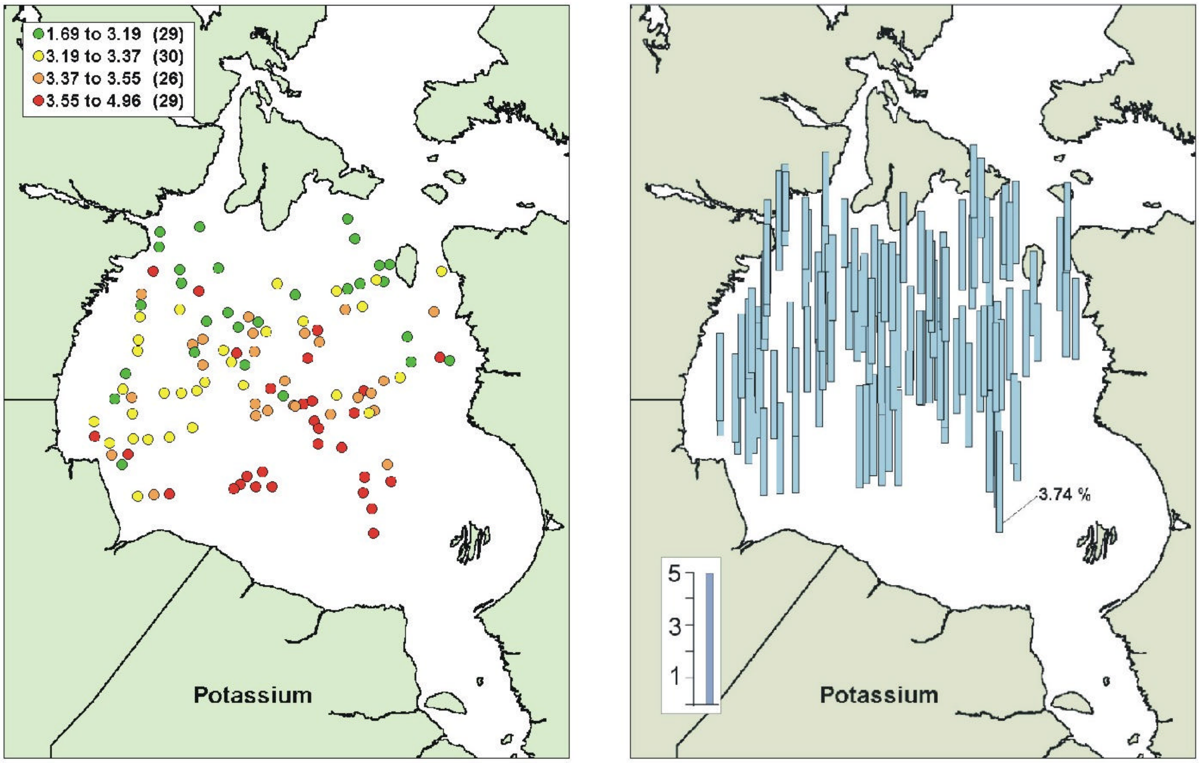


Figure 16-18. Potassium (per cent dry wt.) in clay-size particles of surficial sediment from Hudson Bay. Data from Henderson (1989). Ranges selected to place approximately equal numbers of points in each range (left); same data shown as bars with bar height scaled to the concentration of potassium (right).

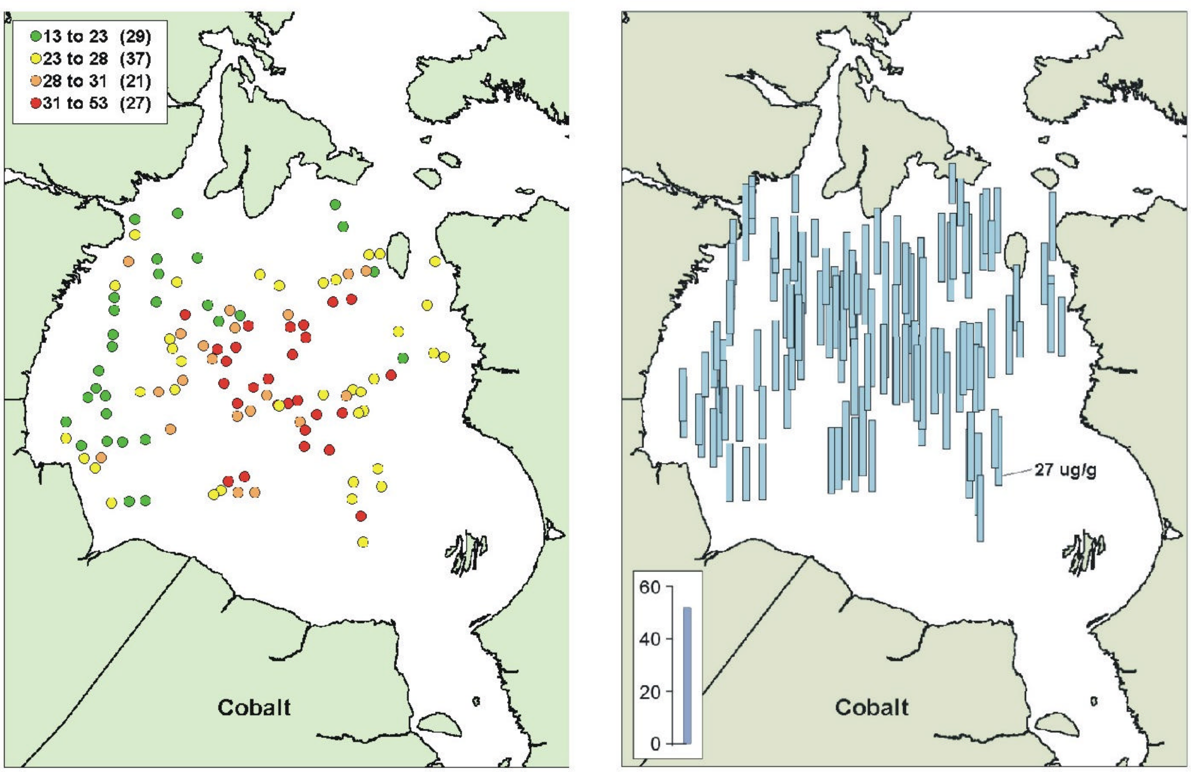


Figure 16-19. Cobalt ($\mu g \cdot g^{-1}$ dry weight) in clay-size particles of surficial sediment from Hudson Bay. Data from Henderson (1989). Ranges selected to place approximately equal numbers of points in each range (left); same data shown as bars with bar height scaled to concentration of cobalt (right).

16.10.2 Manganese

Manganese (Mn) in clay-size particles averaged 0.12 per cent with a very broad range from 0.04 to 1.0 per cent. The highest value was about 25 times the lowest value (Table 16-5). The map in Figure 16-20 (left panel) shows the high values in the central part of Hudson Bay but the bar graphs in Figure 16-20 (right panel) show this cluster more strikingly. Manganese, like cobalt, correlates positively with water depth. Manganese correlates strongly with cobalt, molybdenum and nickel and weakly with potassium and iron (Appendix 7). The cluster of high values in the central region probably implies that conditions there favour diagenesis of manganese. Manganese probably becomes concentrated in surficial sediment there by migration upward from deeper layers in response to chemical and oxidation/reduction gradients. Manganese averaged 0.035 per cent (range 0.022 – 0.062 per cent) in the sediments from the five Ontario rivers reported by McCrea et al. (1984) and so it seems unlikely that manganese originating from these rivers would make a measurable difference in the higher marine values.

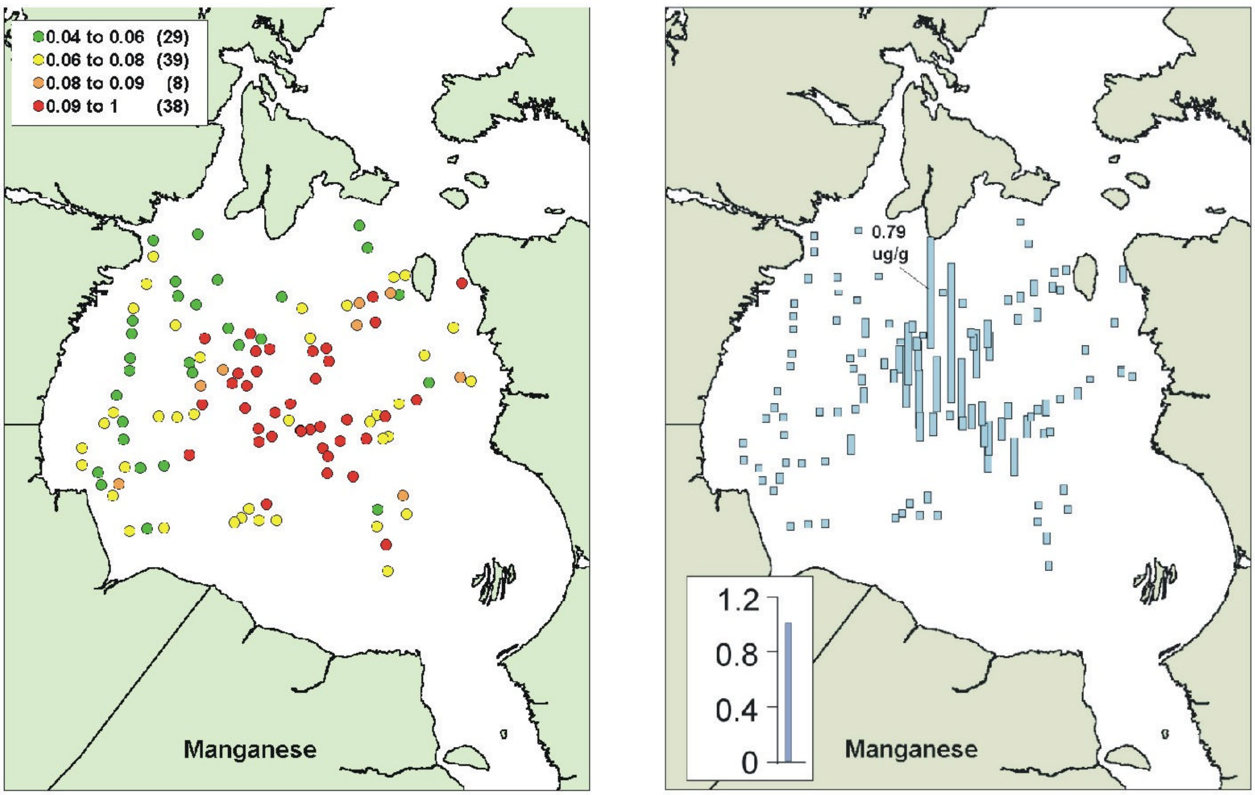


Figure 16-20. Manganese (percent dry weight) in clay-size particles of surficial sediment from Hudson Bay. Data from Henderson (1989). Ranges selected to place approximately equal numbers of points in each range (left); same data shown as bars with bar height scaled to concentration of manganese (right).

16.10.3 Molybdenum

Molybdenum (Mo) in clay-size particles had a relatively narrow range from 0.5 to 12 $\mu g{\cdot}g^{-1}$ with a mean of 3.37 $\mu g{\cdot}g^{-1}$ (Table 16-5). The high concentrations were mostly in the central area of Hudson Bay although they differed only slightly from the low values (Figure 16-21, left panel). The bar graphs in Figure 16-21 (right panel) show the relative spatial distribution of molybdenum levels. Molybdenum levels correlated with cobalt, manganese, and nickel but not with water depth (Appendix 7).

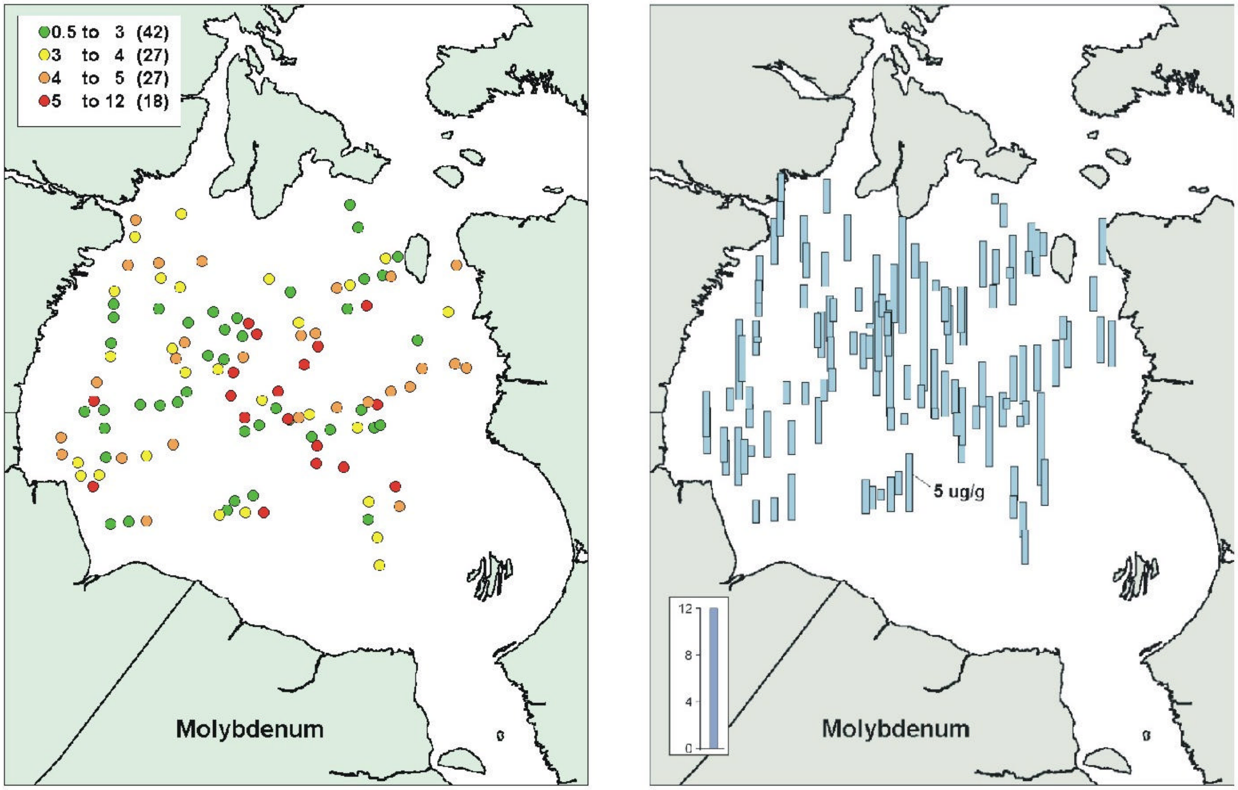


Figure 16-21. Molybdenum (µg·g^{-1} dry weight) in clay-size particles of surficial sediment from Hudson Bay. Data from Henderson (1989). Ranges selected to place approximately equal numbers of points in each range (left); same data shown as bars with bar height scaled to concentration of molybdenum.

16.10.4 Nickel

Nickel (Ni) levels in clay-size particles had a similarly narrow range also, about two-fold from 51 to 107 µg·g^{-1} with a mean of 74 µg·g^{-1}. In view of the small range in values for nickel, the coloured dots in Figure 16-22 (left panel) give a somewhat artificial impression of higher values in the central region. The bar graphs in Figure 16-22 (right panel) give a more realistic impression of relatively uniform levels throughout Hudson Bay. Nickel levels correlated with a number of other metals, notably with chromium (Appendix 7). Nickel levels were lower in the Ontario river sediments reported by McCrea et al. (1984). River sediments ranged from 'not detected' to 24 µg·g^{-1}, with a mean for the 16 positive results of 16 µg·g^{-1}. Painter et al. (1994) provided a map showing the distribution of nickel in lake and stream sediments (Figure 16-23) with areas of high values in the greenstone area around Foxe Basin on Baffin Island and the Melville Peninsula.

16.11 MERCURY

Henderson (1989) did not analyze the Geological Survey samples for mercury. The concentrations of mercury in northern animals with mercury are problematic. Mercury is present naturally in the environment of Hudson Bay but mercury is also added to Hudson Bay by a number of human activities. We have a few core profiles for mercury from southeastern Hudson Bay and also a few dredge samples from the vicinity of Rankin Inlet. Figure 16-24 shows the locations where cores were obtained, and the down-core profiles of mercury in three sediment cores from southeastern Hudson Bay. The ISQG for mercury in marine sediment is 0.13 µg·g^{-1} and the PEL for mercury is 0.7 µg·g^{-1}. Mercury levels in the cores range from 0.013 to 0.037 µg·g^{-1} (Figure 16-24). These samples were not fractionated before analyses and so the concentrations found are directly comparable to ISQG and PEL figures. This comparison suggests that levels of mercury in surficial sediments of southeastern Hudson Bay are well below ISQG and PEL values. Core Hud-4 was taken near the Great Whale River and any inputs

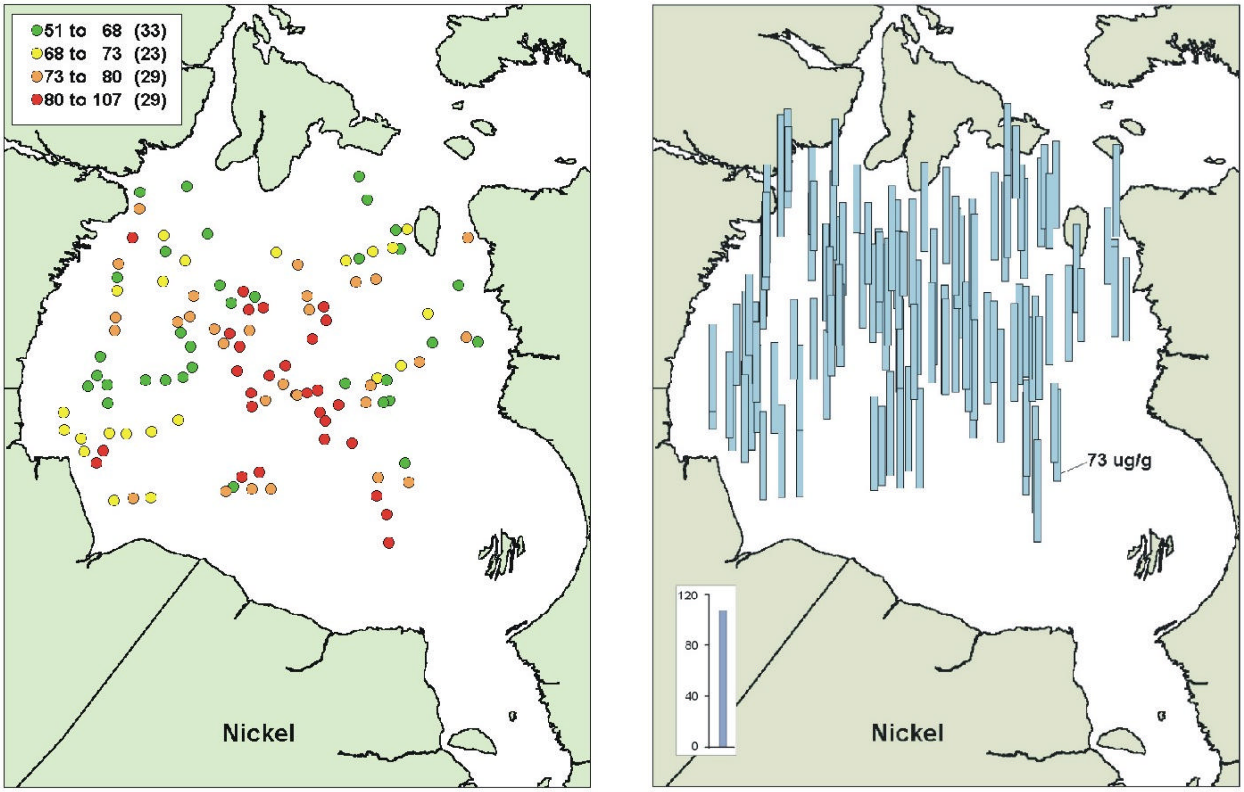


Figure 16-22. **Nickel (µg·g^{-1} dry weight) in clay-size particles of surficial sediment from Hudson Bay. Data from Henderson (1989). Ranges selected to place approximately equal numbers of points in each range (left); same data shown as bars with bar height scaled to the concentration of nickel (right).**

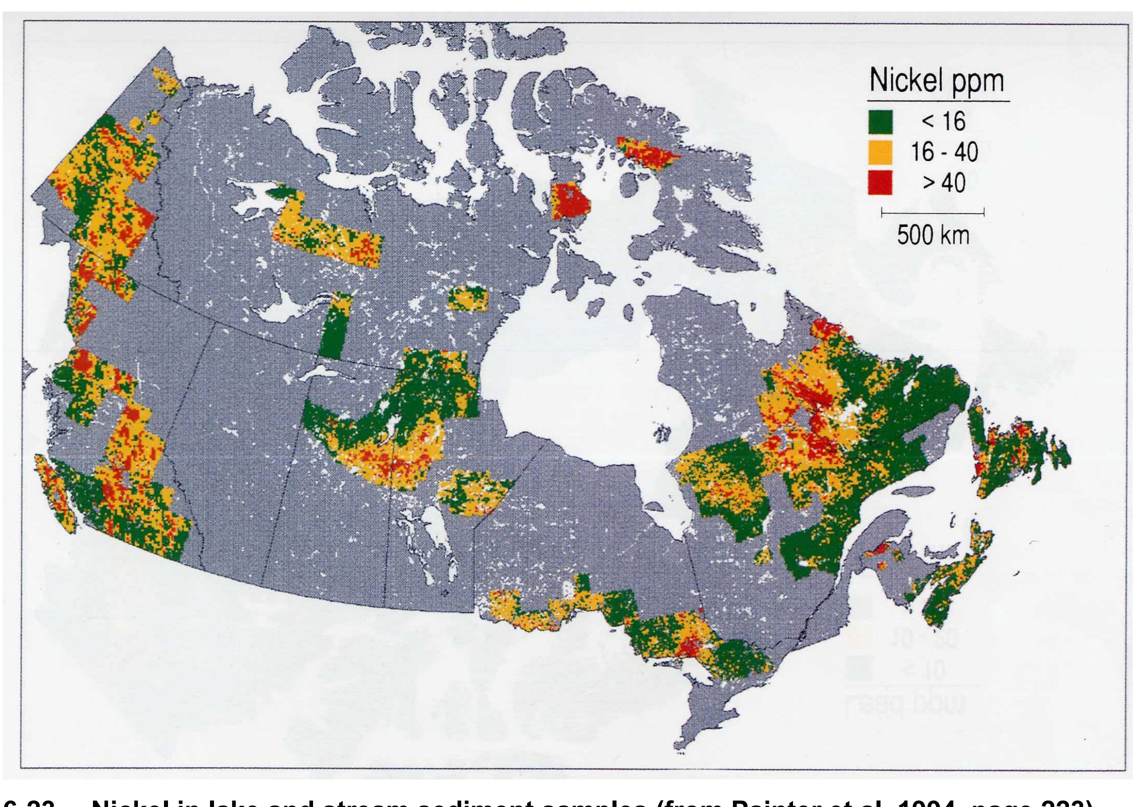


Figure 16-23. **Nickel in lake and stream sediment samples (from Painter et al. 1994, page 223).**

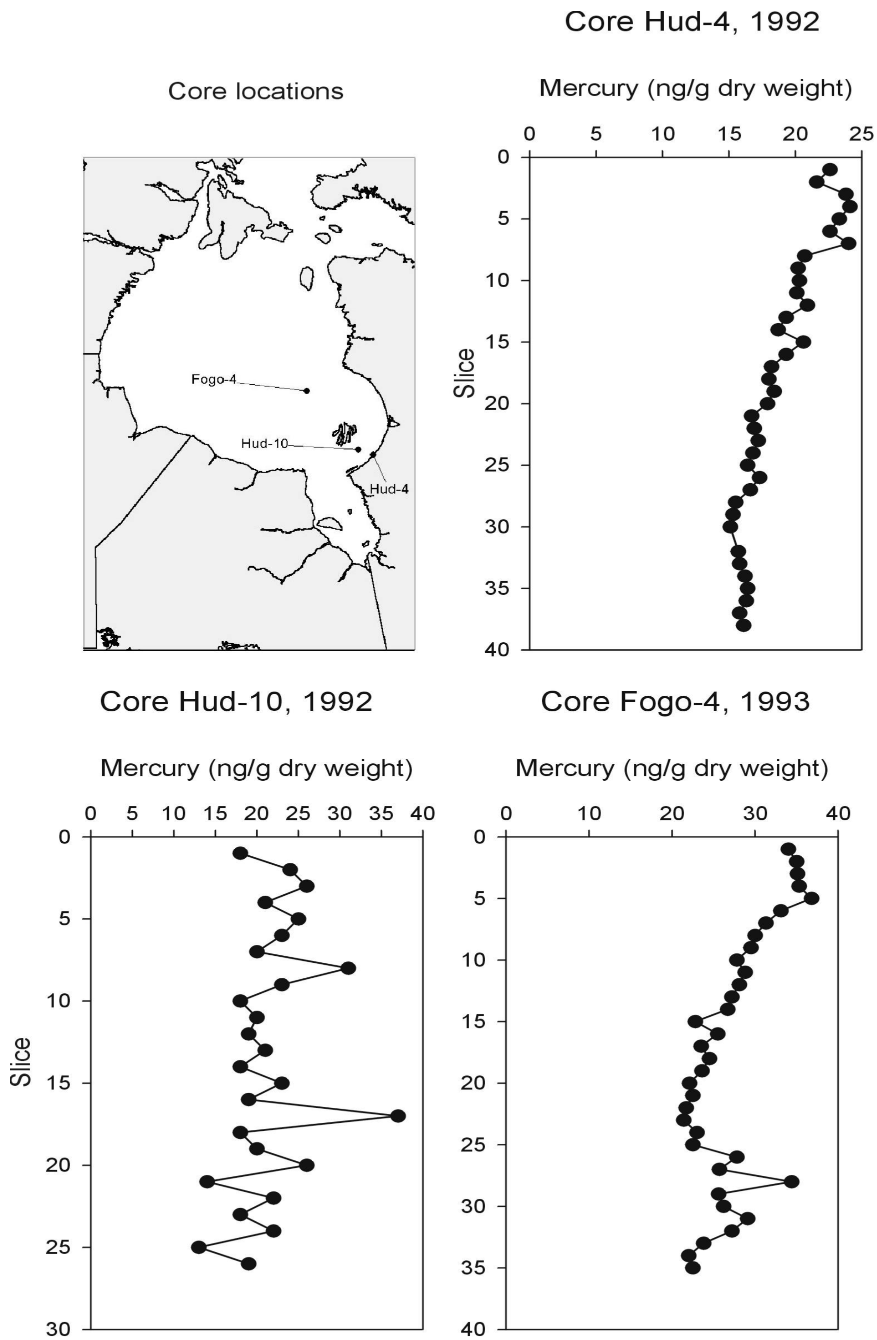


Figure 16-24. Locations of three sediment cores from Hudson Bay in 1992 and 1993 and down-core profiles of mercury ($ng \cdot g^{-1}$ dry weight). (L. Lockhart, unpublished data).

from the river have not been sufficient to bring the concentration up to the ISQG. In addition to the cores from southeastern Hudson Bay, several VanVeen dredge samples taken from near Rankin Inlet in 1995 contained mercury at levels up to 0.014 $\mu g \cdot g^{-1}$, also well below ISQG and PEL values.

Painter et al. (1994) presented data on mercury in 161,228 samples of lake and stream sediment. The median level was 0.06 $\mu g \cdot g^{-1}$ with the 95th percentile at 0.19 $\mu g \cdot g^{-1}$. Most samples (about 70 per cent) were below the ISQG for mercury. Most of the samples were not from Hudson Bay/Foxe Basin drainages, but those that were often fell below 50 $ng \cdot g^{-1}$, with some in the 50-100 $ng \cdot g^{-1}$ range, and with only a few exceeding 100 $ng \cdot g^{-1}$ (Figure 16-25). Taking these lake and stream sediment data together with the few data we have from Hudson Bay, it appears that Hudson Bay sediments are depleted in mercury relative to many others. The concentrations of mercury in samples we have of Hudson Bay sediment do not portray a problem with mercury. The river sediment samples reported by McCrea et al. (1984) were all given as either 0.01 or 0.02 $\mu g \cdot g^{-1}$ or as 'not detected.' In spite of the low levels of mercury in the sediments, there is a persistent problem with accumulations of mercury in marine animals high in the food chains. Two guideline figures are used in efforts to limit human intake of mercury. Concentrations should not exceed 0.5 $\mu g \cdot g^{-1}$ (wet weight) in fish sold commercially in Canada, and levels should not exceed 0.2 $\mu g \cdot g^{-1}$ (wet weight) in fish used for subsistence consumption (Health and Welfare Canada 1979). Levels of mercury in some organs of seals and whales, for example, frequently exceed those levels.

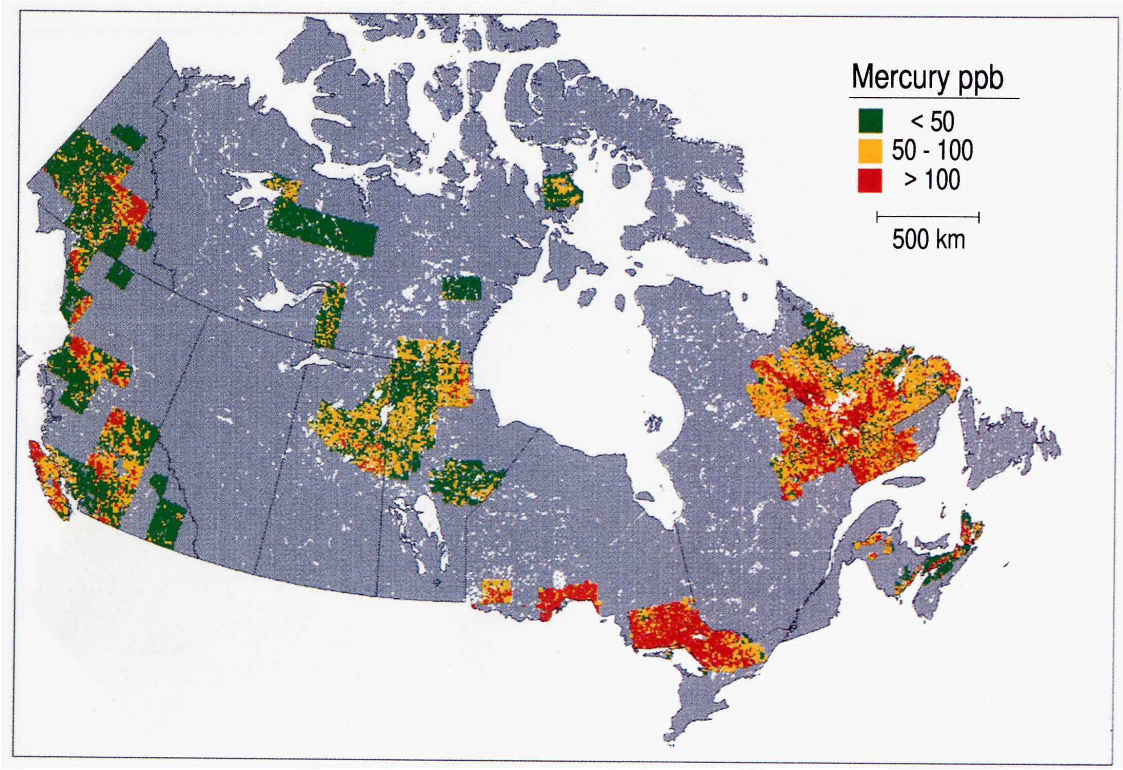


Figure 16-25. Mercury in lake and stream sediment samples (from Painter et al. 1994, page 226).

16.11.1 Mercury in the biota of Hudson Bay

When considering contaminants in animals, especially animals that range over large geographic areas, data on exact positions where animals were taken are of limited value. Rather than organize data by geographic coordinates, locations where samples were collected are generally identified as a nearby community or geographic feature. For example, samples may be identified as Sanikiluaq or Belcher Islands, with no information available on exactly where the samples were obtained.

We have a few scattered data on levels of mercury in benthic marine animals. For example, Table 16-7 and Table 16-8 list unpublished levels of mercury found in some benthic animals collected from C.S.S. Hudson in 1992 and 1996. The samples were collected in southeastern Hudson Bay in 1992 and in the vicinity of Rankin Inlet in 1996. The collection in 1992 was made by freezing specimens on the ship so that the taxonomist (P.L. Wong) worked from frozen material (Table 16-7). This was not satisfactory for some groups, especially the polychaetes, so that the animals collected in 1996 were examined freshly on board the ship by the taxonomist (M. Curtis, Table 16-8). The concentrations are expressed on a dry weight basis and so they are not readily compared with the human consumption guidelines that are expressed on a wet weight basis. As a crude guideline, the dry weight may be about 20-25 per cent of the total wet weight, and so the mercury level on a wet weight basis would be about one quarter of the concentrations shown in Table 16-7 and Table 16-8. Atwell et al. (1998) reported levels of mercury in several invertebrate species from Lancaster Sound and their values were generally lower than those we obtained for the benthic animals from Hudson Bay. For example, Atwell et al. (1998) obtained 0.09 $\mu g \cdot g^{-1}$ (dry weight) of mercury in a specimen of *Macoma calcarea* and we obtained an average of 0.22 $\mu g \cdot g^{-1}$ for six specimens of the same species (Table 16-8).

Surprisingly perhaps, the recent CACAR II (2003: Biological Environment) compendium lists no data on mercury in marine fish from Hudson Bay. Three fourhorn sculpins (*Myoxocephalus quadricornis*) were taken in August, 1992, and found to have mercury levels in muscle of 0.10, 0.27 and 0.21 $\mu g \cdot g^{-1}$wet weight (L. Lockhart, unpublished data). Schetagne and Verdon (1999) reported mercury in fourhorn sculpins from several stations

Table 16-7. Mercury (µg·g^{-1} dry weight) in benthic animals taken in box cores and VanVeen sediment dredges from southeastern Hudson Bay, August, 1992. (L. Lockhart, unpublished data).

Species	n	Minimum (µg·g^{-1} dry weight)	Maximum (µg·g^{-1} dry weight)	Mean (µg·g^{-1} dry weight)
Bivalve, *Nucula belloti*	10	0.132	0.846	0.345
Bivalve, *Thyasira gouldi*	10	0.164	1.787	0.719
Bivalve, *Yoldiella lenticula*	5	0.423	1.450	0.784
Bivalve, *Portlandia arctica*	4	0.149	0.421	0.249
Bivalve, *Periploma absyssorum*	3	0.110	0.849	0.477
Bivalve, *Nuculana penula*	pooled			0.388
Brittle stars (unclassified)	pooled			0.045
Polychaetes (unclassified)	15	0.014	0.285	0.100
Starfish (unclassified)	8	0.015	0.100	0.057
Sea urchins (unclassified)	8	0.004	0.030	0.017
Star lillies (unclassified)	3	0.040	0.048	0.043

Animal taxonomic identifications by P.L. Wong, DFO, Winnipeg.

Table 16-8. Mercury (µg·g^{-1} dry weight) in benthic animals taken in VanVeen sediment dredges from Hudson Bay near Rankin Inlet, August, 1996. (L. Lockhart, unpublished data).

Species	n	Minimum (µg·g^{-1} dry weight)	Maximum (µg·g^{-1} dry weight)	Mean (µg·g^{-1} dry weight)
Amphipod, *Ampelisca eschrichti*	15	0.025	0.048	0.035
Brittle star, *Stegophiura nodosa*	1			0.048
Bivalve, *Astarte crenata*	23	0.075	0.186	0.135
Bivalve, *Serripes groenlandicus*	7	0.067	0.179	0.122
Bivalve 1, *Macoma* spp	11	0.092	0.198	0.148
Bivalve 2, *Macoma* spp	4	0.109	0.601	0.244
Bivalve 3, *Mya pseudoarenaria*	9	0.073	0.14	0.105
Bivalve 4, *Macoma calcarea*	6	0.181	0.331	0.219
Bivalve 5, *Nucula belloti*	12	0.101	0.552	0.305
Bivalve 6, *Yoldia hyperborea*	7	0.144	0.722	0.368
Bivalve 7, *Clinocardium ciliatum*	18	0.130	0.392	0.247
Bivalve 8, *Nuculana pernula*	5	0.172	0.240	0.202
Polychaete, *Ophelia limacina*	1			0.044
Polychaete, *Praxillela gracilis*	1			0.075
Polychaete, *Ammotrypane cylindricaudatus*	6	0.14	0.209	0.168
Polychaete, *Maldane sarsi* (large)	2	0.178	0.197	0.188
Polychaete, *Nephtys*	7	0.094	0.262	0.136
Polychaete, *Lumbrineris*	12	0.068	0.228	0.126
Slender eelblenny, *Lumpenus fabricii*	1			0.080

Animal taxonomic identifications by M. Curtis, DFO, Winnipeg.

along the James Bay coast; station means ranged from 0.10 to 0.55 µg·g^{-1} wet weight for sculpins of standardized 250-mm length. Similarly, greenland cod (*Gadus ogac*) from the same stations on eastern James Bay had station means values from 0.14 to 0.42 µg·g^{-1} wet weight for cod 400 mm in length. The stations on the James Bay coast were part of a large study of mercury in fish from habitats affected by hydroelectric developments in Quebec. The authors reported effects of the La Grande complex of dams and reservoirs extended downstream, but that effects were limited to a relatively small area influenced by the summer freshwater plume from the river. Hydro Quebec (Hayeur 2001) mentioned that the area of influence extends 10-15 km on each side of the mouth of the river. Some station means for fourhorn sculpin and greenland cod from eastern James Bay included values that exceed the guideline value of 0.2 µg·g^{-1} (wet weight) for subsistence consumption but it is not clear whether hydroelectric facilities were responsible.

The CACAR II (2003: Biological Environment, Annex Table 5) lists concentrations of mercury in muscle of anadromous Arctic charr from a number of locations in northern Quebec and Labrador including one on the east side of Hudson Bay. Mean values ranged from 0.027 to 0.072 µg·g^{-1} (wet weight), all well below the guidelines for human consumption and also below the figures for sculpins and greenland cod noted above. These values for Arctic charr are shown in Figure 16-26. Included in the figure is one collection of landlocked charr from Kangiqsujuag and those charr were about three-fold higher in mercury content than the anadromous charr from the same area. Similar data on the bivalve Mytilus edulis (whole body) had a range from 0.01 to 0.03 µg·g^{-1} wet weight. The CACAR II table also listed cadmium, arsenic, selenium and lead for most of the collections of charr and mussels. The earlier CACAR I (1997: page 206, Table 3.3.5) report gave levels of mercury in muscle of several species of fish from the Grande Baleine River: lake trout, 0.71 µg·g^{-1}; lake whitefish, 0.14 µg·g^{-1}; and northern pike, 0.63 µg·g^{-1}. Lake trout and northern pike exceeded human consumption guidelines. However, lake trout from the Rankin Inlet-Arviat area of western Hudson Bay had considerably lower levels, 0.064 µg·g^{-1}. Other lakes (Hawk Lake, Peter Lake) also in the area of western Hudson Bay had mean levels of 0.24 and 0.67 µg·g^{-1}.

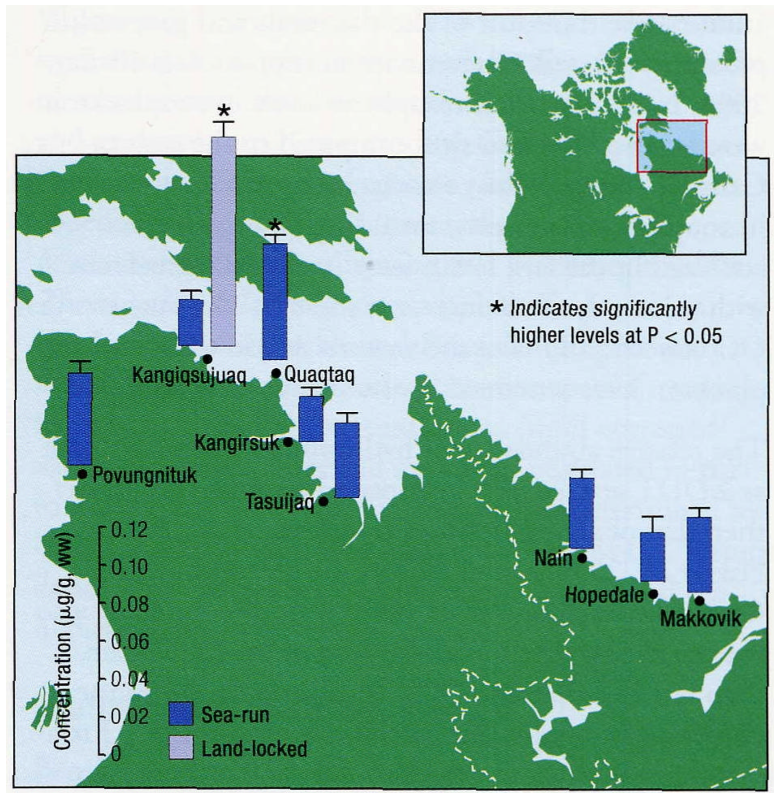


Figure 16-26. Mercury concentrations (µg·g^{-1} wet weight) in muscle of anadromous and landlocked Arctic charr from Nunavik and Labrador (1998-1999). Mercury concentrations have been adjusted for age and length of fish. (From CACAR II 2003: Biological Environment, Figure 3.3.2, page 29).

Mercury levels in some of the landlocked predatory fish like lake trout are relatively high; however, those in anadromous charr are considerably lower, implying that feeding at sea supplies less mercury than feeding in lakes. So little is known about levels in fully marine fish of Hudson Bay, that the few scattered reports available do little more than establish the need for a systematic survey of levels of mercury in marine fish and their supporting food chains. Such a survey would also help isolate the real and potential contributions of freshwater sources such as hydroelectricity projects and possibly regional climate warming in western Hudson Bay watersheds.

There have been several studies of mercury in northern birds and these have often included collections from Hudson Bay. For example, Figure 16-27, taken from CACAR II (2003) shows the levels of mercury in liver of eider ducks from two locations in Hudson Bay and from Holman Island in the western Arctic. The common eiders contained mercury in liver at concentrations in the 0.4 to 0.51 µg·g^{-1} wet weight range with little difference among geographic areas. Mercury in king eiders was slightly higher. Seals and whales from the western Arctic generally contain higher levels of mercury than those from Hudson Bay, but this is not the case with eiders. One potential explanation for the low levels in the eiders from Holman may be that the levels reflect wintering areas in the northern Bering Sea, Bering Strait, and southern Chuckchi Sea. The apparent difference between common eiders and king eiders may be related to their diets. Both species eat mussels, but common eiders are more specialized feeders while king eiders eat a varied diet including echinoderms and other invertebrates. Figure 16-28 shows similar data for long-tailed ducks (oldsquaw). The levels of mercury in liver are considerably lower than those in eiders and they vary more widely among sites. Long-tailed ducks (oldsquaw) from Sanikiluaq had the lowest levels among the collections from marine areas.

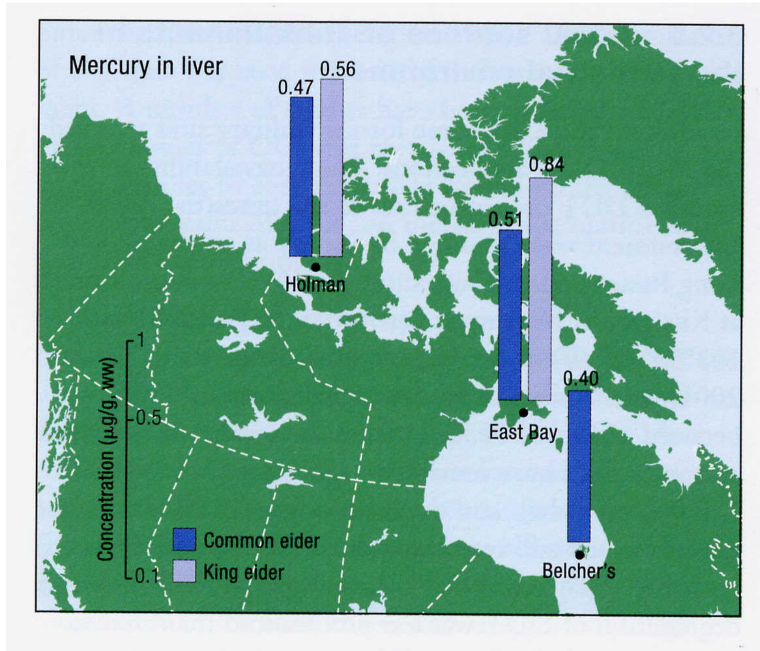


Figure 16-27. Mercury (geometric means in µg·g^{-1} wet weight) in liver of common eiders and king eiders from two locations in Hudson Bay and from Holman in the western Arctic. (From CACAR II 2003: Biological Environment, Figure 3.2.2, page 25).

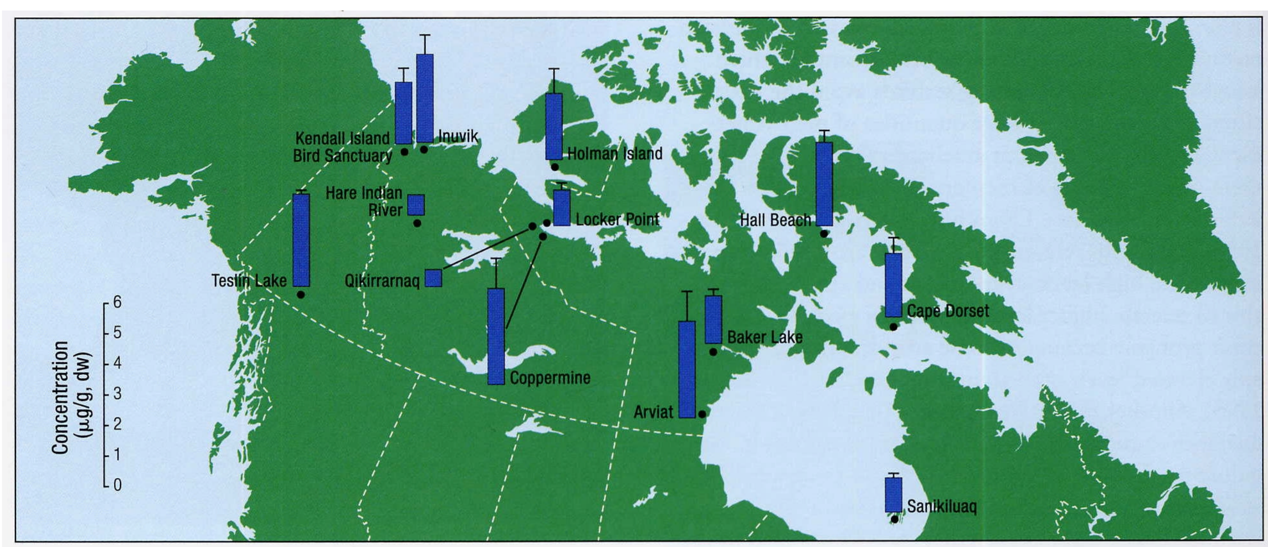


Figure 16-28. Total mercury (µg·g^{-1} dry weight) in liver of long-tailed ducks collected between 1991 and 1994. (From CACAR II 2003: Biological Environment, Figure 3.3.6, page 34).

Mercury levels have increased since the 1970s some arctic birds. Figure 16-29 shows graphs of temporal trends in eggs of three species from Prince Leopold Island, northern fulmar, black-legged kittiwake, and thick-

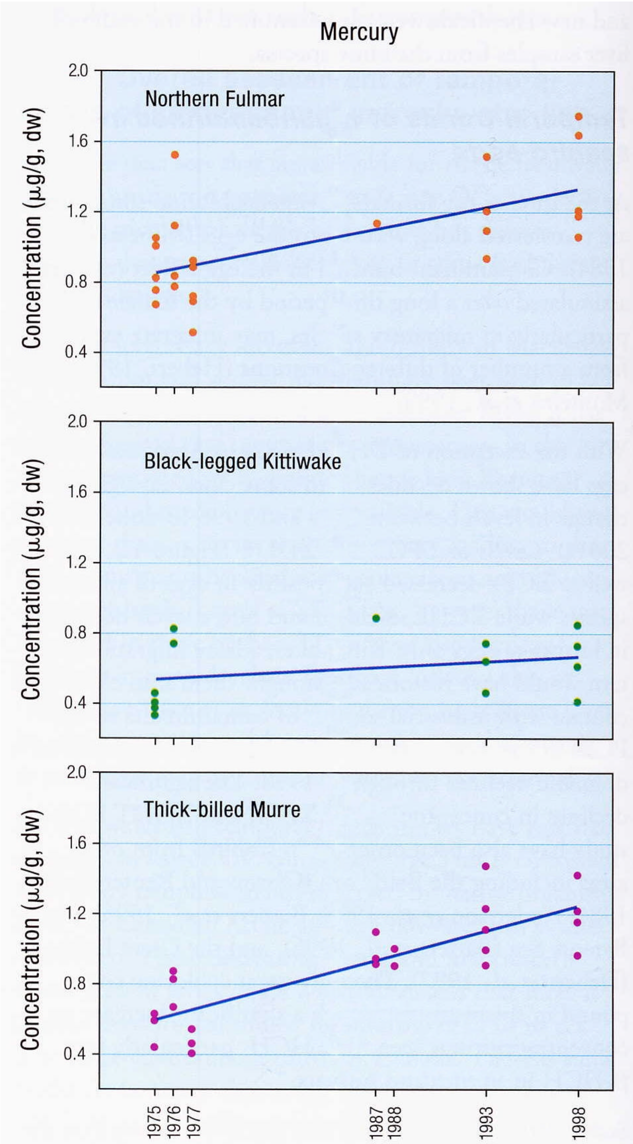


Figure 16-29. Mercury in eggs of three species of sea birds from Prince Leopold Island between 1975 and 1998. (From CACAR II 2003: Biological Environment, Figure 4.2.3, page 70).

Of all northern animals, marine mammals generally have the highest levels of mercury. Most chemical residue analyses have been done with ringed seals, beluga whales and polar bears. Figure 16-30 shows concentrations of mercury, arsenic, cadmium, and lead in liver of ringed seals from a number of locations including one in Hudson Bay. The age-adjusted levels of mercury appear to be higher in the seals from Arviat than from other locations but the high statistical variance in the data means that the apparent difference was not meaningful. The levels in liver are very high in billed murre. Eggs of northern fulmar and thick-billed murre show a trend to higher levels of mercury but those of the black-legged kittiwake do not. The difference may arise from the overwintering habitats. Kittiwakes winter at lower latitudes where levels of mercury have been decreasing but the murres and fulmars remain in northern waters (North Atlantic) throughout the year.

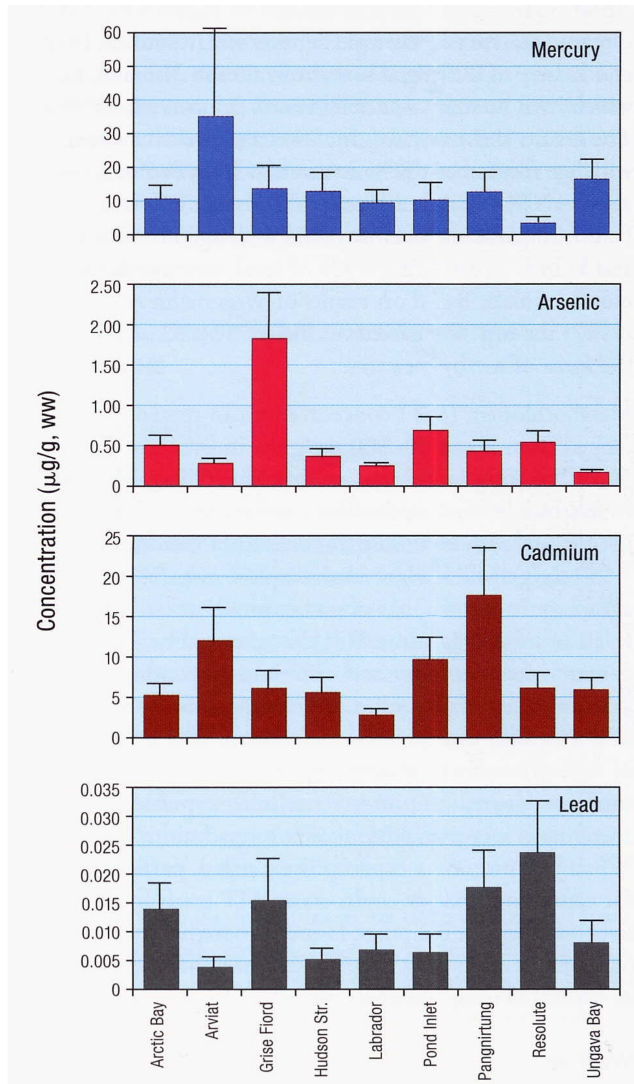


Figure 16-30. Mercury, arsenic, cadmium and lead concentrations (µg·g^{-1} wet weight) in liver of ringed seal from nine locations in the eastern/central Canadian Arctic. Results are based on seals from age 2-15 years. Mercury concentrations are adjusted for age. (From CACAR II 2003: Biological Environment, Figure 3.3.1.1, page 39).

comparison with the human consumption guidelines for fish (0. 5 µg·g^{-1} wet weight for commercial sale) but the biochemical form assumed by mercury in seal liver is different from that in fish muscle. In fish, most of the mercury is present as the neurotoxic form, methylmercury, but in seals and beluga whales the form of mercury varies from organ to organ. Methylmercury comprises only a small proportion of the total mercury in the liver of ringed seals (Wagemann et al. 2000). CACAR II (2003, page 99) reports that the threshold concentration for liver damage in marine mammals is 60 µg·g^{-1}. Figure 16-30 indicates that the average concentration of mercury in liver of seals from Arviat was about 34 µg·g^{-1} with an error bar extending over 60 µg·g^{-1} and so some seals in Hudson Bay may have liver concentrations of mercury high enough to result in biological harm.

Beluga whale liver also contains high levels of mercury but, again, only a small proportion of it is present as methylmercury. There appears to be some variation in mercury levels among beluga from different regions. Figure 16-31 shows mean levels of total mercury in liver of beluga (adjusted for age) from locations where multiple collections have been made over the period from 1981 to 2002. The whales from the Beaufort Sea coast generally had higher levels than those from any of the locations in Hudson Bay. Mercury levels seem to have increased since the mid- 1980s in several of the collections. Statistically, the apparent increase is evident at Arviat where the two collections were separated by 15 years. The collection in 1999 had concentrations of mercury more than twice as high as those in the collection in 1984. There was no apparent trend in whales from Coral Harbour but the first collection was in 1993. There appears to have been an increase also at Sanikiluaq but the collections there were only four years apart and no trend can be established rigorously. The same can be said of the whales from Iqaluit where the collections were only one year apart, although the figures give the appearance of an increase in levels. The samples from Pangnirtung do show an apparent increase since the first collection in 1984. The whales taken at Arviat summer downstream from the Churchill-Nelson hydroelectricity development but the other whales analyzed to date would not be expected to summer near any hydroelectricity development. CACAR II (2003) reported that the threshold for biological effects in marine mammals was 60 µg·g^{-1} and some 32 beluga liver samples exceeded that value of the 528 for which we have data. If this threshold is applied to beluga, then some beluga contain sufficient mercury in liver that it should pose a toxicity risk to them.

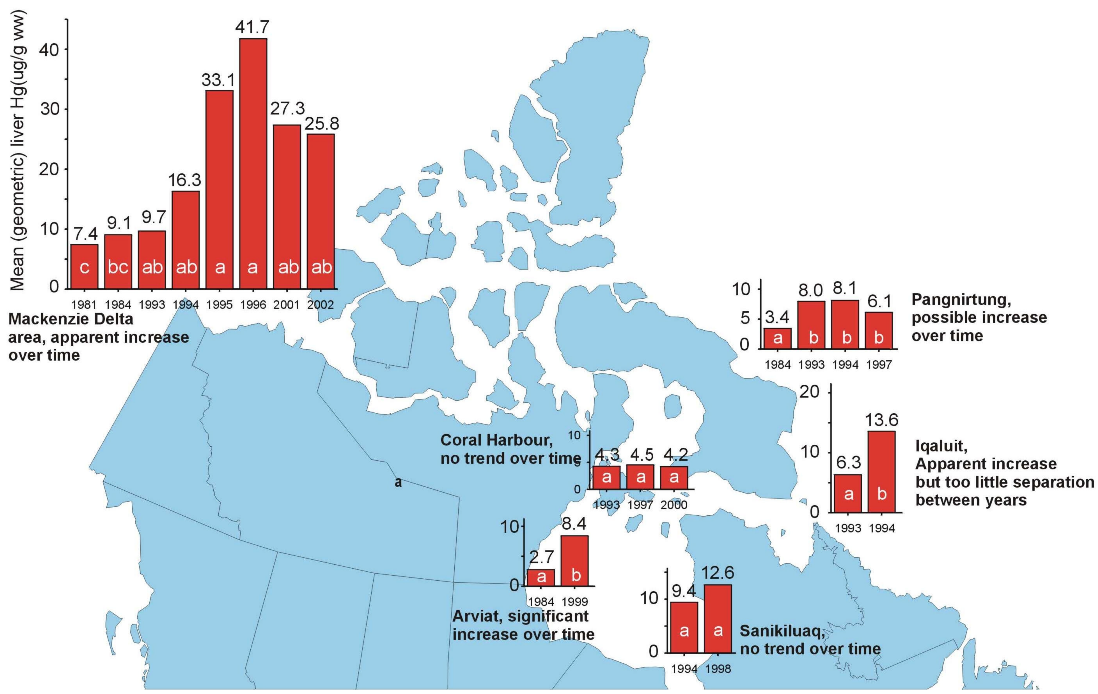


Figure 16-31. Mean concentrations of total mercury (µg·g^{-1} wet weight, adjusted for whale age) in liver of beluga whales from several sites in northern Canada from 1981 to 2002 (Figure modified from Lockhart et al., presented at Northern Contaminants Program symposium, Ottawa). Figures at the bases of the bars are years when samples were obtained; figures at the tops of the bars are least square geometric means; letters on the bars indicate statistical differences by Duncan's test.

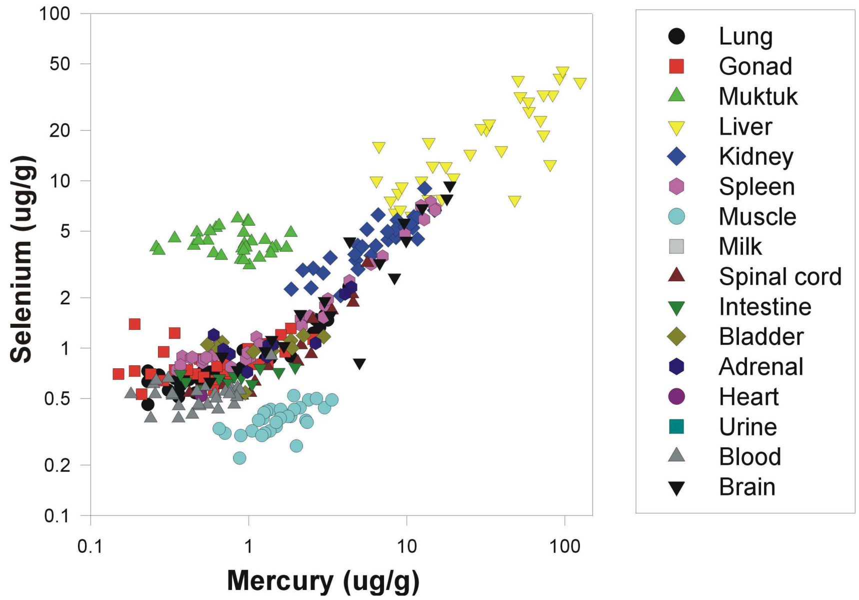


Figure 16-32. Total mercury and selenium (µg·g^{-1} wet weight) in organs of beluga whales from the Mackenzie Delta area. (From Hyatt et al. 1999).

The levels of mercury in the whales vary greatly among different organs. Figure 16-32 shows the levels of mercury and selenium in different organs of beluga from the Mackenzie Delta. Although these whales were not from Hudson Bay, it seems unlikely that the beluga from Hudson Bay would process and store mercury very differently from other beluga. Mercury concentrations were considerably higher in liver than in other organs with kidney and brain having the next highest levels. Mercury was lowest in urine, milk, heart, and lung. Two organs separated clearly from the others, namely muscle with relatively low selenium for its level of mercury, and muktuk with high selenium for its level of mercury (Figure 16-32). The relationship between mercury and selenium may be important because several studies have shown that the presence of selenium can ameliorate the toxicity of mercury. For example, Eaton et al. (1981) reported that cats given selenium with mercury did not develop the symptoms of mercury poisoning found in cats given the same amount of mercury without the accompanying selenium.

The high level of mercury in liver has been fractionated into several different forms of mercury. Figure 16-33 shows a pie chart indicating the proportions found in each different form. Only about 6 per cent of the total mercury in these samples was present as the neurotoxic methylmercury. The speciation varies strikingly from one organ to another. For example, in one study the proportion of methylmercury in beluga muscle from the eastern Arctic was 93 % while that in liver from the same animals it was only 11.7 % (Wagemann et al. 1998). The major form is thought to be mercury selenide, an inert form that is found naturally as the mineral tiemannite, HgSe. Since the whales major intake of mercury is from dietary methylmercury, the presence of a high proportion of HgSe in the liver may represent a metabolic detoxification mechanism to bind the mercury as an inert form and render it non-toxic or at least less toxic. The speciation of mercury appears also to differ geographically. The proportion of total liver mercury represented by methylmercury was 11.7 % in the eastern Arctic, but only 5.9 % in the western Arctic (G.A. Stern et al. 2003 Northern Contaminants Program Synopsis Report for 2001 and 2002, in press).

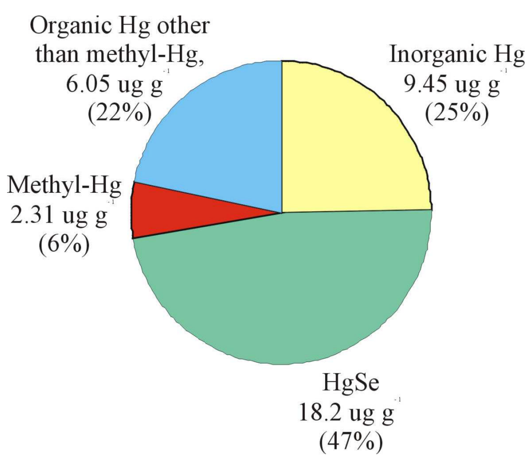


Figure 16-33. Pie chart showing arithmetic mean concentrations (µg·g^{-1} wet weight; with percentages in parentheses) of biochemical forms of mercury in liver of beluga whales. (From Lockhart et al. 1999).

Considering seals and whales together, our most comprehensive set of analyses is on liver. The most recent tabulation of DFO data on levels of mercury in liver from animals in or around Hudson Bay is shown in Table 16-9 (without statistical corrections for ages of the animals). With the exception of the two bowhead whales, all samples exceeded the human consumption guideline (0.05 µg·g^{-1} wet weight in commercial fish) for mercury.

Table 16-9. Ages and levels of mercury, selenium and cadmium (µg·g^{-1} wet weight) in liver of bowhead whales, ringed seals, beluga, narwhal and walrus taken from locations in or around Hudson Bay. (n=number of samples for which a measurement was made).

Species	Location	Year	Age (y)		Total Hg (µg·g^{-1} ww)			Se (µg·g^{-1} ww)			Cd (µg·g^{-1} ww)		
			n	mean	n	mean	S.D.	n	mean	S.D.	n	mean	S.D.
Beluga	Arviat	1984	22	11.2	23	6.62	6.94	23	4.16	2.40	23	6.73	6.69
Beluga	Arviat	1999	37	11.2	37	12.68	9.93	37	8.47	6.77	37	11.9	6.72
Beluga	Coral Harbour	1993	11	16.1	11	6.54	2.97	11	3.99	2.27	11	8.98	5.22
Beluga	Coral Harbour	1997	19	13.1	20	13.38	28.6	20	9.08	15.3	20	8.35	3.78
Beluga	Coral Harbour	2000	24	8.9	25	3.95	2.48	25	4.23	2.51	25	9.22	5.79
Beluga	Igloolik	1995	35	11.5	35	10.59	8.49	35	8.19	3.11	35	6.63	4.05
Beluga	Lake Harbour	1994	19	10.8	20	9.30	6.39	20	7.18	4.04	20	5.48	4.71
Beluga	Lake Harbour	1997	10	16.1	9	11.72	4.17	9	5.92	1.71	9	6.93	3.97
Beluga	Lake Harbour	2001	13	14.1	13	16.42	9.83				13	5.91	2.19
Beluga	Nastapoca	1984	14	13.2	15	10.70	13.7	10	4.60	2.31	15	5.12	2.70
Beluga	Repulse Bay	1993	2	8.0	2	3.43	3.09	2	4.83	2.06	2	9.75	5.20
Beluga	Sanikiluaq	1994	30	13.7	30	12.90	9.53	30	9.76	4.82	30	7.62	3.96
Beluga	Sanikiluaq	1998	22	13.0	22	21.05	25.2	22	16.04	14.9	22	1.32	0.60
Bowhead	Igloolik	1994			1	0.02		1	0.37		1	0.04	
Bowhead	Repulse Bay	1996			1	0.01		1	0.35		1	0.74	
Narwhal	Repulse Bay	1993			4	7.91	6.77	4	5.45	4.35	4	30.7	27.92
Narwhal	Repulse Bay	1999			18	11.39	7.21						
Ringed seal	Arviat	1992	73	13.7	69	22.97	30.0	69	10.91	9.28			
Ringed seal	Arviat	1998	24	18.7	24	21.80	17.2						
Ringed seal	George River		2	6.0	2	11.03	2.37	2	6.81	0.74	2	25.11	25.64
Ringed seal	George River	1989	3	10.0	3	12.90	13.5	3	7.27	5.87	3	6.70	9.13
Ringed seal	George River	1990	2	6.0	2	14.48	10.5	2	5.66	0.02	2	3.44	0.07
Ringed seal	George River	1991	2	8.0	2	4.62	2.47	2	3.12	1.48	2	5.41	4.64
Ringed seal	Inukjuak	1989	2	5.0	2	2.36	2.38	2	3.88	1.95	2	7.80	7.83
Ringed seal	Inukjuak	1990	8	5.3	8	5.56	2.71	8	5.82	1.73	8	17.51	10.34
Ringed seal	Saluit	1989	2	10.0	2	19.17	10.2	2	7.95	1.66	2	10.74	4.91
Ringed seal	Sanikiluaq	1991	27	8.4	26	8.28	6.55	26	6.60	3.36	26	14.26	7.06
Ringed seal	Umiujaq	1994			52	16.43	25.4	52	9.96	7.67			
Ringed seal	Wakeham Bay	1989	23	3.1	23	3.85	5.81	23	3.12	3.06	23	7.74	8.76
Ringed seal	Wakeham Bay	1990	10	6.7	10	6.39	5.88	10	3.91	1.78	10	5.11	7.88
Walrus	Akulivik	1990	4	5.7	4	4.86	0.85	4	3.99	0.57	4	10.95	2.65
Walrus	Hall Beach	1988	16	9.7	16	1.31	1.25	16	3.01	1.16	16	11.25	5.02
Walrus	Hall Beach	1996	16	14.7	16	1.64	1.26	16	2.66	1.61	15	10.84	6.69
Walrus	Igloolik	1982	13	11.7	16	1.30	1.38	15	2.85	0.92	15	12.35	5.73
Walrus	Igloolik	1983	27	12.6	25	1.34	0.99	24	2.57	1.05	25	9.84	3.81
Walrus	Igloolik	1987	16	8.8	16	1.09	1.02	16	2.73	1.37	15	13.82	8.27
Walrus	Igloolik	1988	15	8.5	13	1.45	1.05	13	2.97	1.52	11	13.25	10.33
Walrus	Igloolik	1996	14	16.6	14	2.41	1.98	14	2.91	1.92	14	12.01	4.71
Walrus	Inukjuaq	1990	8	12.4	9	1.12	0.93	9	2.28	1.07	9	5.28	5.81

The levels of mercury in beluga, narwhal, and ringed seal liver consistently exceeded those in walrus. Beluga samples from a number of locations are included and there is almost as much variability within a site as between different sites. Table 16-9 also includes levels of selenium and cadmium in the same samples but there is no concentration guideline for human consumption with which to compare these values. Levels of cadmium were considerably higher in kidney than in liver.

The ultimate non-human predator in the arctic is the polar bear. Figure 16-34 shows concentrations of mercury in liver of polar bears from 12 areas in the Canadian Arctic. The levels of mercury in bears from the two collections in Hudson Bay were the lowest of all the values obtained. The values given are expressed on a dry weight basis and so they are not directly comparable with human consumption guidelines. Polar bears eat ringed seals but levels of mercury in livers of ringed seals from the Arviat area (Figure 16-30, top panel, wet weight basis) appear to be higher than levels in bear livers (Figure 16-34, dry weight basis). If the seal values were expressed on a dry weight basis, they would be higher yet and the species difference would be even more striking. One might expect the predator to contain higher amounts of mercury than the prey but this is not the case. This discrepancy has been noted by several scientists and explained by the fact that the bears eat only the blubber of the seals, not the protein-rich organs like muscle and liver where more mercury is found (Atwell et al. 1998).

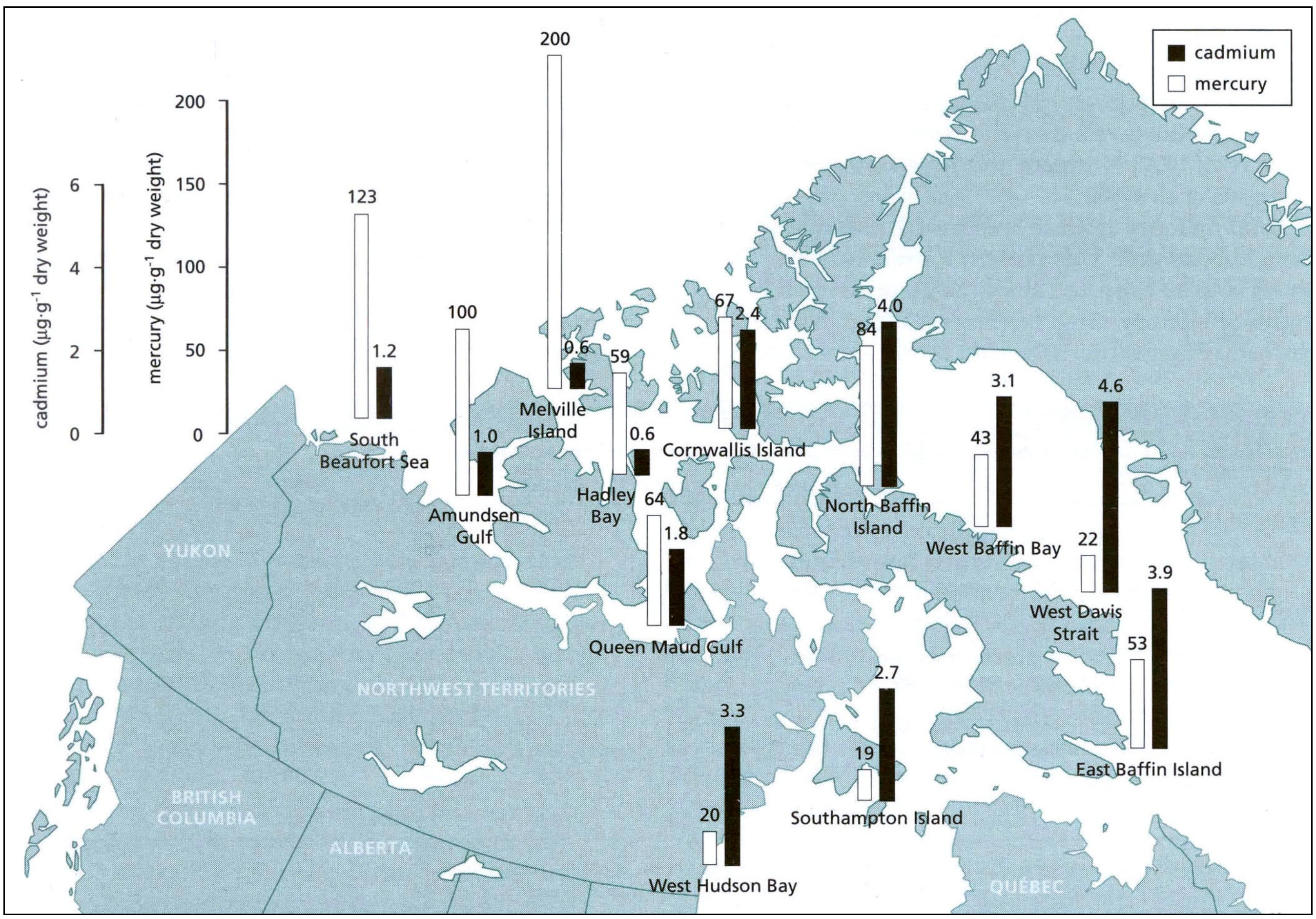


Figure 16-34. Concentrations (age and sex adjusted) of mercury and cadmium ($\mu g \cdot g^{-1}$ dry weight) in liver of polar bears from 12 areas of the Canadian Arctic. (From CACAR I 1997: Figure 3.3.24, page 247).

16.12 CADMIUM

Henderson (1989) did not report any analyses of sediments for cadmium. We obtained core profiles for cadmium on the same three cores from southeastern Hudson Bay in 1992 and 1993 mentioned above with regard for lead (Figure 16-10) and mercury (Figure 16-24). The depth profiles for cadmium are shown in Figure 16-35 with most values under 0.3 $\mu g \cdot g^{-1}$ (300 $ng \cdot g^{-1}$). Two aberrant values exceeded 500 $ng \cdot g^{-1}$ and may represent contamination errors but even so, no samples reached the ISQG of 700 $ng \cdot g^{-1}$ (0.7 $\mu g \cdot g^{-1}$). Since these analyses refer to unfractionated sediment, they can be compared directly with the ISQG value. Twenty-five additional dredge samples were taken in 1995 from the area around Rankin Inlet. The range for these samples was from 0.014 to 0.210 $\mu g \cdot g^{-1}$, all well below the ISQG values. McCrea et al. (1984) did not detect cadmium in the sediments of five rivers in Ontario just above the tidal influence of Hudson Bay or James Bay. The sediments available to date do not suggest a contamination problem with cadmium.

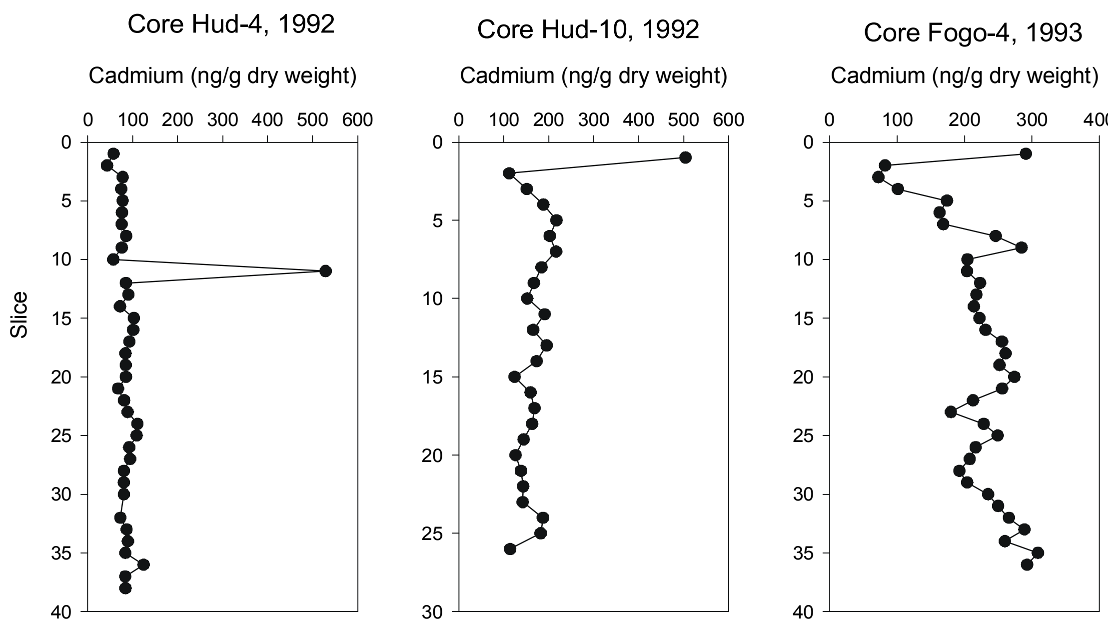


Figure 16-35. Down-core profiles of cadmium ($ng \cdot g^{-1}$ dry weight) in three sediment cores from southeastern Hudson Bay. Cores Hud-4 and Hud-10 were collected in 1992 and core Fogo-4 was collected in 1993. Locations of core sites are shown in Figure 16-24. (L. Lockhart, unpublished data).

Cadmium does reach the biota of Hudson Bay. Figure 16-36 shows the concentrations of cadmium in kidneys of eider ducks from two locations in Hudson Bay and from Holman Island in the western Arctic. The highest mean value was 40.8 $\mu g \cdot g^{-1}$ wet weight in king eider from Southampton Island. The threshold for cadmium poisoning in birds is reported to be greater than 100 $\mu g \cdot g^{-1}$ wet weight in liver (CACAR II 2003: Biological Environment, Table 5.3.2, page 99) and so the present levels in eiders do not appear to represent a risk to the birds. No indication of the variance was given in Figure 16-36 and so it is possible that some small proportion of the birds reach the threshold concentration. Figure 16-30 includes a panel showing levels of cadmium in liver of ringed seals from several locations in northern Canada. The value given for cadmium in liver of seals from Arviat was about 12 $\mu g \cdot g^{-1}$. Again, this appears to be below the threshold for biological effects which is 20-200 $\mu g \cdot g^{-1}$ (CACAR II 2003: Biological Environment, Table 5.3.2, page 99). The error bar on the graph suggests that a small proportion of the seals would reach or exceed the lowest part of the effects range. The seals from Pangnirtung, however, had mean liver cadmium of about 18 $\mu g \cdot g^{-1}$ and the error bar extended well above 20 $\mu g \cdot g^{-1}$ and so cadmium probably poses a greater risk to that stock of seals.

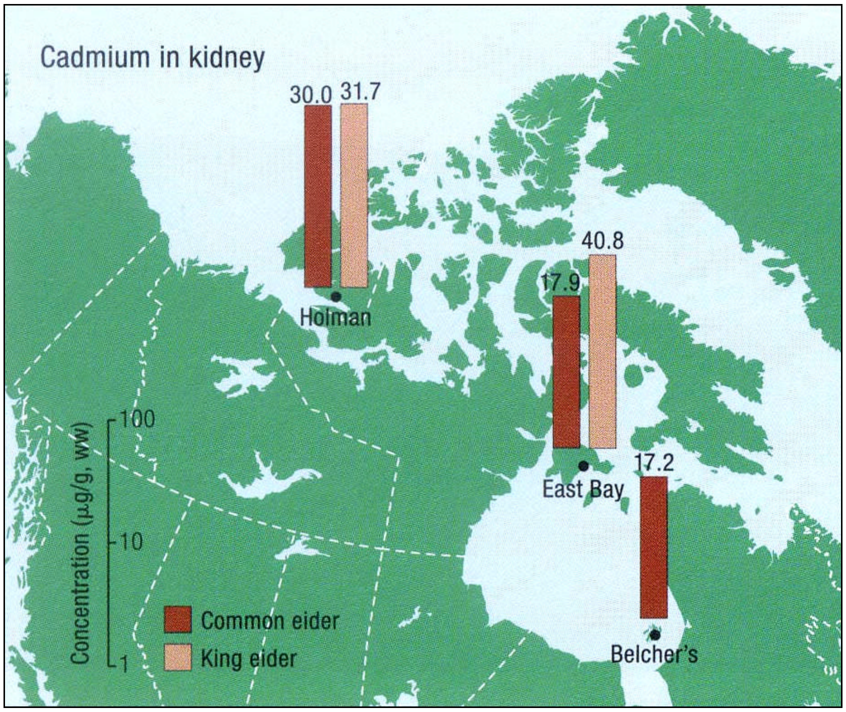


Figure 16-36. Mean levels of cadmium (µg·g^{-1} wet weight) in kidneys of eider ducks. (From CACAR II 2003: Biological Environment, Figure 3.2.2, page 25).

16.13 RADIONUCLIDES

CACAR I (1997) reviewed the status of radiunuclides in the Canadian Arctic. The greatest human exposure derives from the natural radionuclides lead-210 and its decay product, polonium-210. Both are derived from radon-222 gas, which in turn derives from radium-226 and ultimately from uranium in the soils. Presumably the exposure of northern ecosystems to these natural radionuclides has been occurring for thousands of years. The fission of uranium in bombs and nuclear reactors results in production of several artificial radionuclides and three of these are of concern in the environment, namely iodine-131, cesium-137 and strontium-90. Most atmospheric testing of nuclear bombs was stopped in 1963 by international agreement, however, France and China were not parties to the agreement and they continued atmospheric testing until 1980. Cesium isotopes, Cs-134 and Cs-137 are considered to be the artificial isotopes of greatest concern. The Chernobyl reactor accident in Ukraine in 1986 also contributed Cs-137 and Cs-134 to the Canadian North. Very small amounts of these isotopes have reached Hudson Bay. Figure 16-37 includes a profile of Cs-137 in sediment from Core Hud-4 from southeastern Hudson Bay in 1992. Judging by that profile, peak inputs occurred in the 1960s with a gradual decline since then.

The natural radionuclides polonium-210, radium-226, thorium-230, thorium-232, uranium-234, and uranium-238 were measured in a series of samples collected from the surface sediments of Hudson Bay in 1995 from the area near Rankin Inlet. These levels are plotted in the maps shown in Figure 16-37. The ranges for the different colours of dots were selected by MapInfo software to show approximately equal numbers of points in each range. The red dots represent samples with the highest levels of these radionuclides and they were usually closest to the community of Rankin Inlet. The levels of Ra-226 and its distant decay daughter Po-210 had some values in the high range further offshore near the island to the east of the community. These samples were examined analyzed for particle size distribution in the Geology Department, University of Manitoba, and the radionuclide concentrations were examined for statistical relationships to water depth and proportions of sand, silt and clay (Appendix 8). The two isotopes of uranium were associated statistically with fine particles (silt and clay), but there were few other statistical relationships between individual radionuclides and particle sizes. The cesium-137 was negatively correlated with the sand content but the correlation coefficient was significant at $p<0.06$, just short of the usual criterion for significance of $p<0.05$. However, there was no relationship between Cs-137 and either silt or clay. The correlation calculations do not take geographic position into account and so they offer an incomplete description of any relationships among radionuclides and sediment size classes. No sediment quality guidelines have been established for the natural radionuclides.

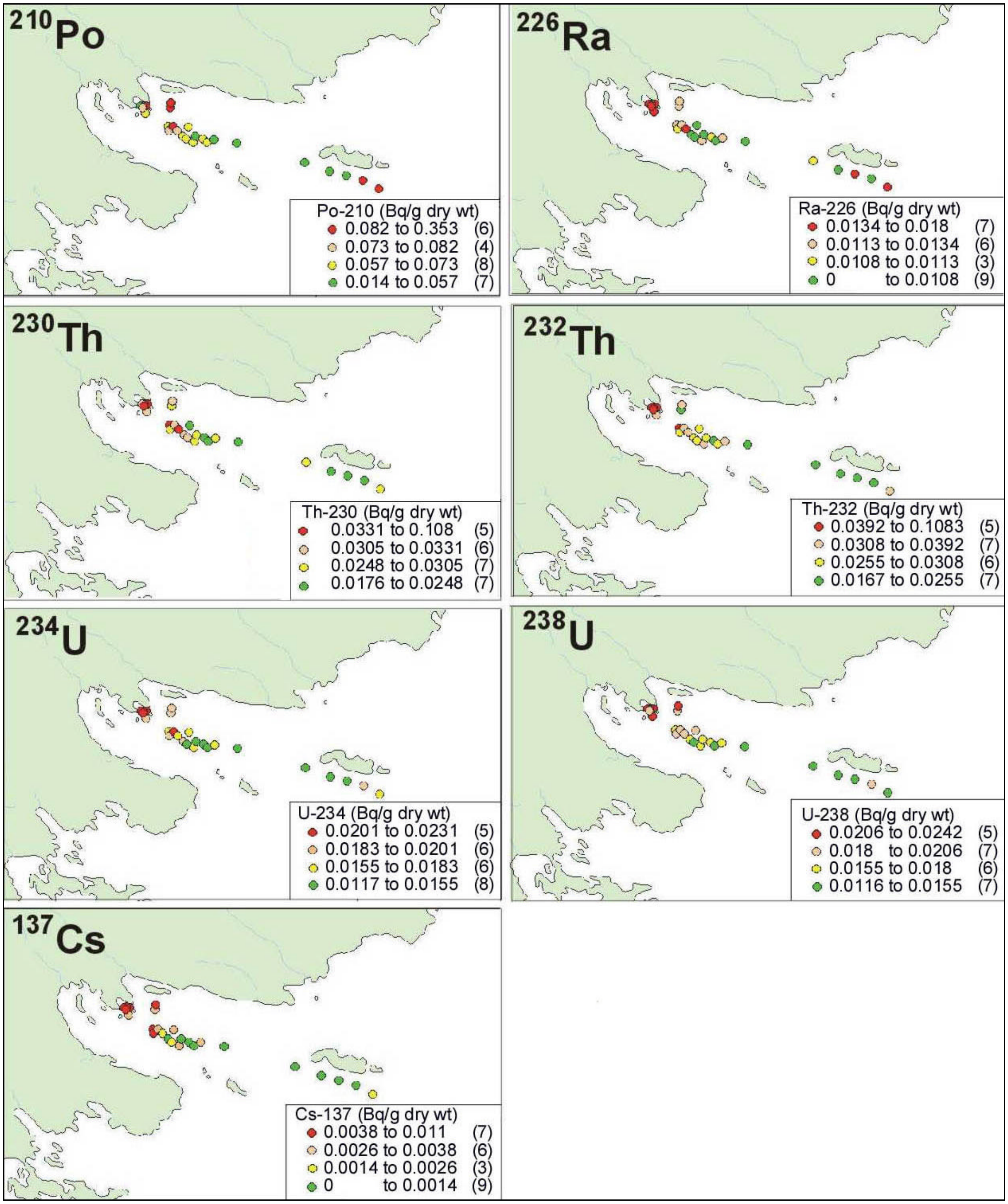


Figure 16-37. Radionuclides ($Bq \cdot g^{-1}$ dry weight) in surface sediment from Hudson Bay in the vicinity of Rankin Inlet, 1995. The colour ranges were selected by MapInfo software to contain approximately the same number of points in each range (L. Lockhart, unpublished data).

The distribution of the man-made radionuclide, cesium-137 is also shown in Figure 16-37 and its distribution is similar to that for the uranium and thorium isotopes. The down-core profile of cesium-137 in a core from southeastern Hudson Bay was shown also in Figure 16-3 and cited as evidence that anthropogenic activities have effects on Hudson Bay. The source of cesium-137 in this area was atmospheric testing of nuclear bombs and most of that stopped in 1963 by international agreement; subsequent inputs have fallen dramatically. There have been point source discharges to the ocean (e.g., Sellafield, U.K.) but it is unlikely that these are responsible for the Cs-137 found in Hudson Bay. The levels shown in Figure 16-37 may be compared with earlier measurements from the area around Baker Lake reported by Svoboda et al. (1985) on samples collected in 1981 and 1982. The earlier figures are generally higher than those for marine sediments collected in 1995 near Rankin Inlet (Figure 16-38). The radiological half-life of cesium-137 is 30 years and more than half of the amounts present in 1981/82 would still be present in 1995. Radioactive decay would be expected to bring the freshwater samples by Svoboda et al. (1985) into close agreement with the marine sediments in 1995. The same comparison is also possible for radium-226 (Figure 16-39) and the values for the freshwater drainages are highly variable as would be expected in a region with locally enriched deposits of uranium. Radium-226 has a long radiological half-life of 1622 years and so the difference in the times when the samples were collected is insignificant for this isotope. The main value in these measurements may be as basal information for evaluation when commercial developments of uranium ores in regions around Baker Lake or Foxe Basin take place.

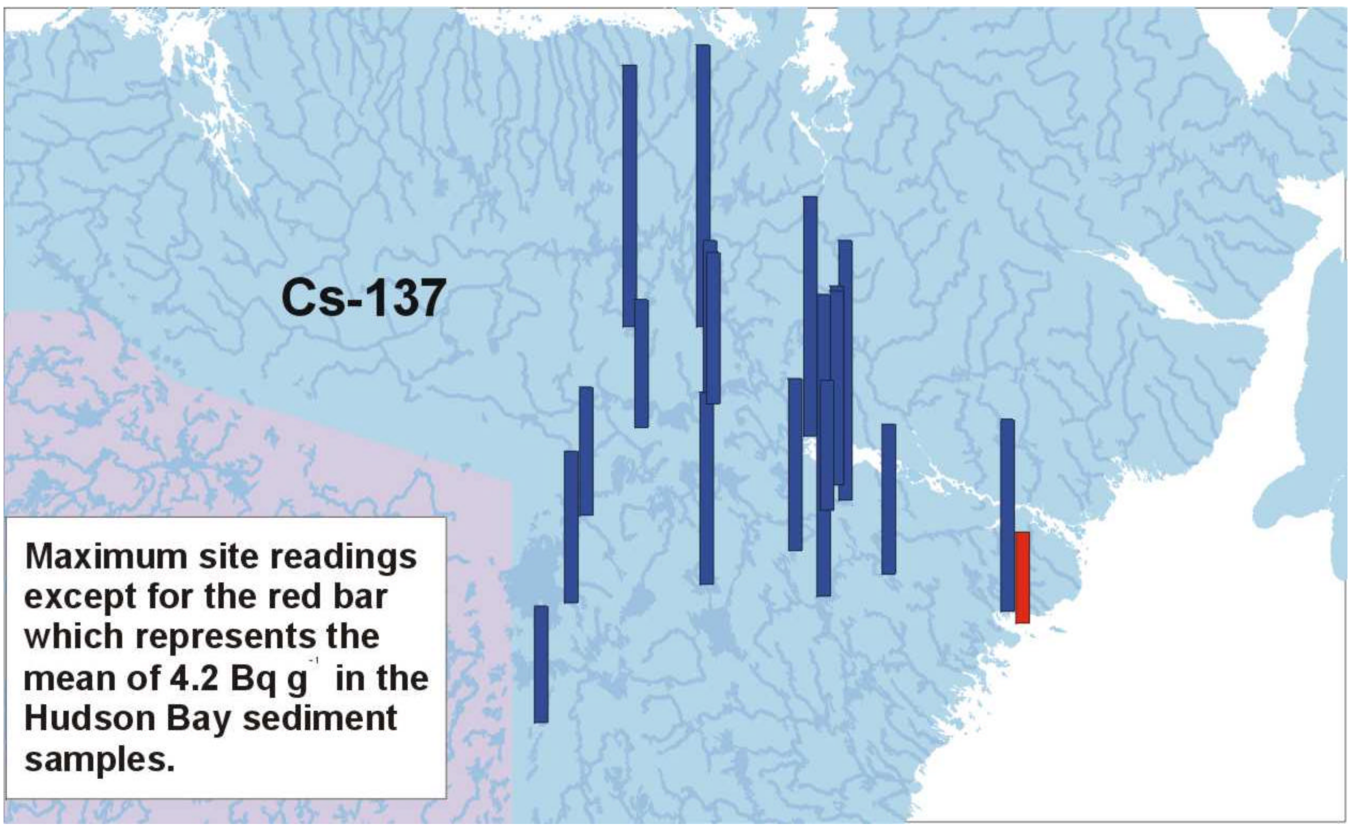


Figure 16-38. Cesium-137 (Bq·g^{-1}) in vegetation and substrate as reported by Svoboda et al. 1985 (blue bars), from collections made in 1981 and 1982. The red bar is the mean for marine sediment samples collected near Rankin Inlet in 1995 (L. Lockhart, unpublished data).

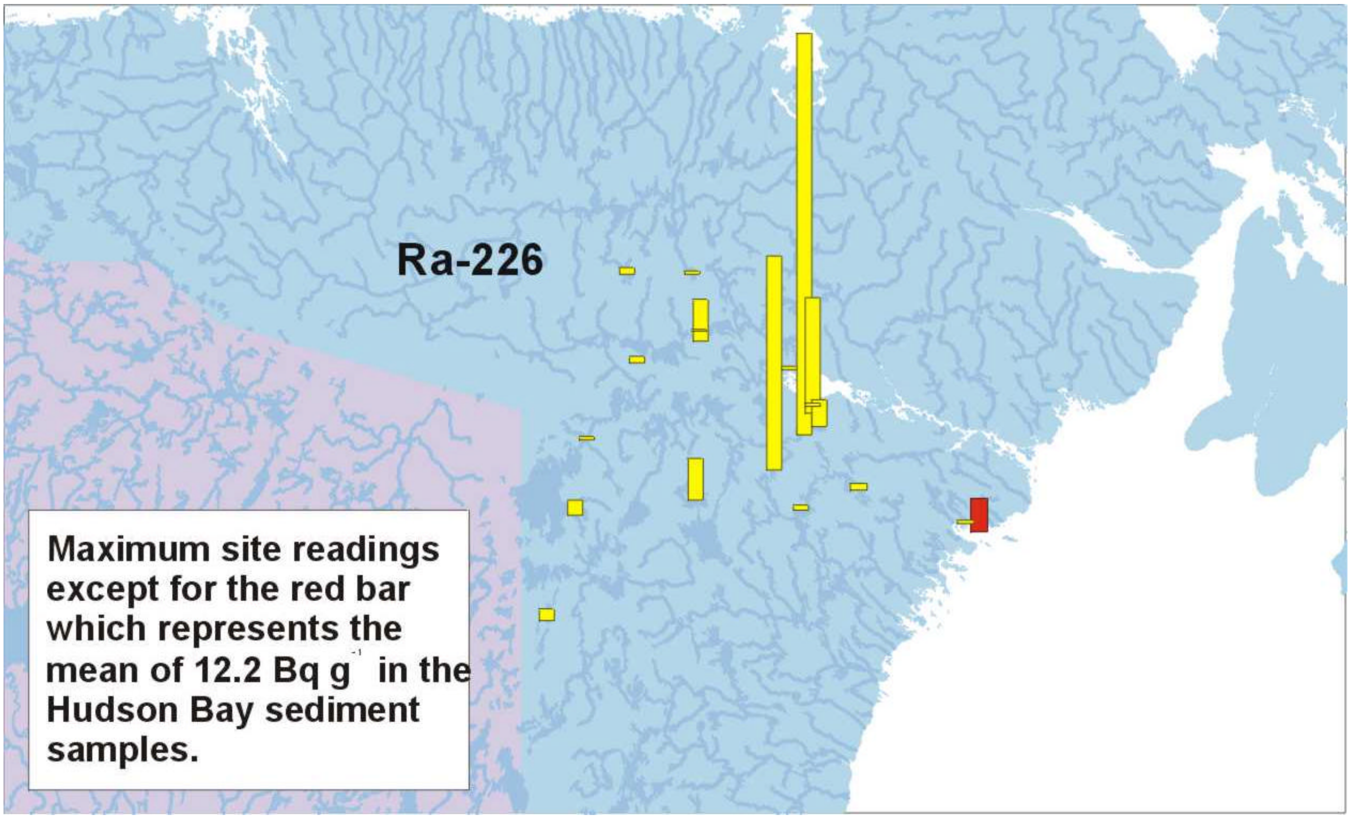
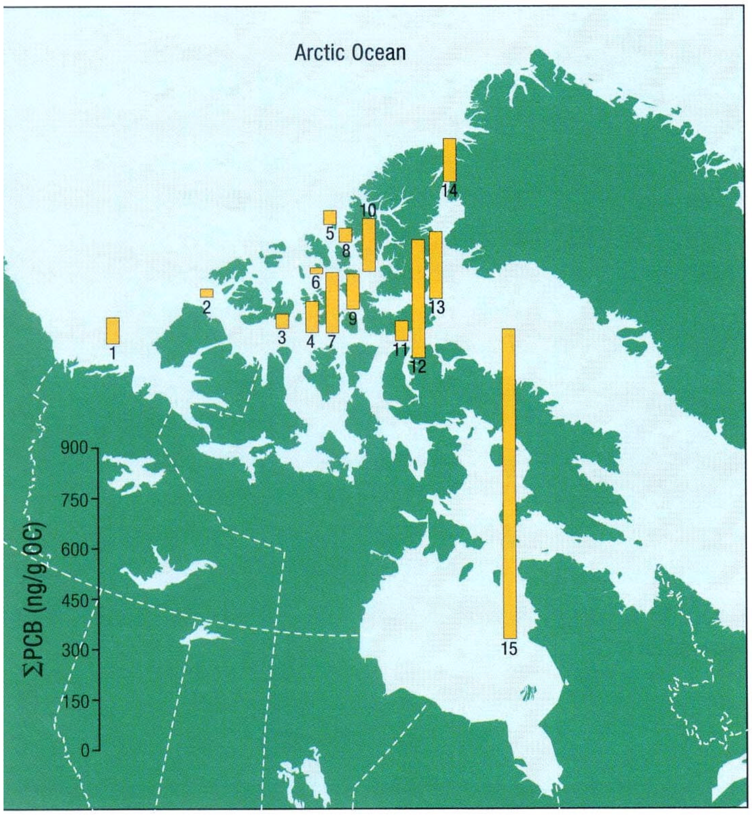


Figure 16-39. Radium-226 (Bq·g^{-1}) in vegetation and substrate as reported by Svoboda et al. 1985 (yellow bars), from collections made in 1981 and 1982. The red bar is the mean for marine sediment samples collected near Rankin Inlet in 1995 (L. Lockhart, unpublished data).

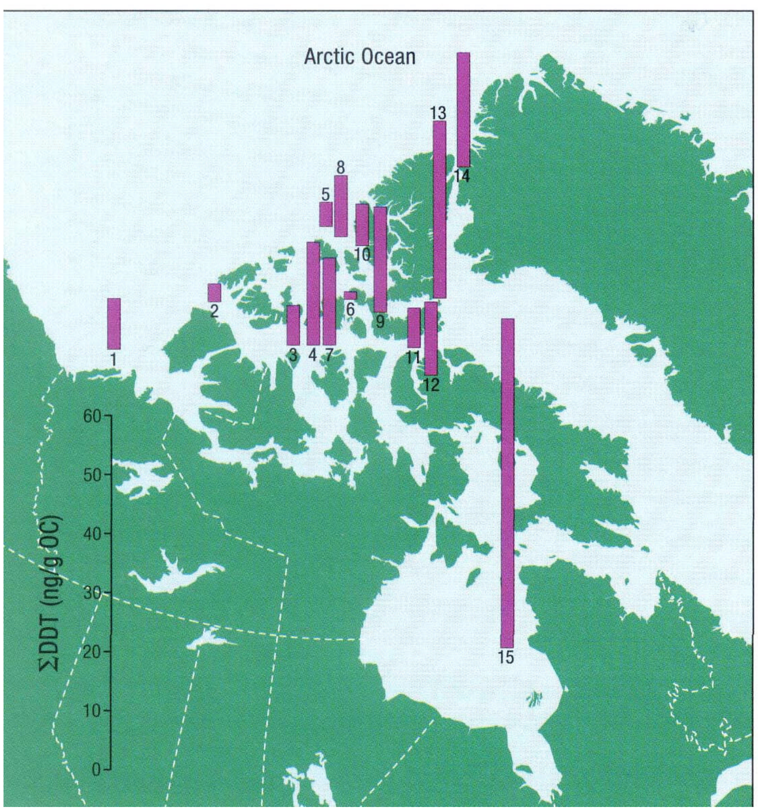
Little information exists on radionuclides in the aquatic biota of Hudson Bay. Appendix 8 lists some unpublished data from the early 1980s for several radionuclides in animals taken in the vicinity of Saqvaqjuaq. With the exception of cesium-137, the time elapsed since sampling would not be expected to have changed the exposure since all the other radionuclides are of natural origin and their presence is sustained by the presence of long-lived parent radionuclides. For example, the levels of the relatively short-lived isotope, polonium-210 are sustained by the presence of its parent isotope, lead-210, which is in turn supported by radon gas and then by radium-226. Levels of cesium-137 in animals today will have fallen below the values recorded in the early 1980s because the half-life of cesium-137 is only 30 years and inputs of new Cs-137 have fallen dramatically.

16.14 SYNTHETIC ORGANIC COMPOUNDS IN THE BIOTA OF HUDSON BAY

DDT and its metabolite, DDE, were identified in samples of soil, vegetation, and a variety of animals, mostly birds, from the Churchill area in 1967. Brown and Brown (1970) compared residues in areas sprayed with DDT (for control of mosquitoes) and in nearby unsprayed areas and found DDT and DDE present consistently in both areas. Having reached the Arctic environment by various pathways (e.g., Figure 16-1), synthetic compounds have become incorporated into both the physical and biological components of the Arctic. Several contaminants move through food chains to become concentrated in animals at high trophic levels. Following are a few examples that show levels of organochlorine compounds in arctic samples. Few data exist describing organic contaminants in Hudson Bay sediments. Figure 16-40 shows bar graphs of PCBs and DDT in Arctic marine sediments and only one sample is reported from Hudson Bay (from core Fogo-4, location shown in Figure 16-24). That sample, however, had the highest levels of both DDT and PCBs (per gram of organic carbon) among the locations reported.



	ΣPCB	Site
1	**80** ng/g OC	Beaufort
2	**15** ng/g OC	M'Clure St.
3	**40** ng/g OC	V. Melville Sd
4	**88** ng/g OC	Barrow St.
5	**33** ng/g OC	Peary Ch.
6	**9.6** ng/g OC	Penny St.
7	**170** ng/g OC	Barrow St.
8	**37** ng/g OC	Massey Sd.
9	**98** ng/g OC	Wellington Ch.
10	**150** ng/g OC	Norwegian Bay
11	**54** ng/g OC	Lancaster Sd.
12	**350** ng/g OC	Strathcona Sd.
13	**190** ng/g OC	Jones Sd.
14	**120** ng/g OC	N. Baffin Bay
15	**920** ng/g OC	E. Hudson Bay



	ΣDDT	Site
1	**8.6** ng/g OC	Beaufort
2	**2.5** ng/g OC	M'Clure St.
3	**6.0** ng/g OC	V. Melville Sd
4	**17** ng/g OC	Barrow St.
5	**3.7** ng/g OC	Peary Ch.
6	**1.2** ng/g OC	Penny St.
7	**14** ng/g OC	Barrow St.
8	**10** ng/g OC	Massey Sd.
9	**18** ng/g OC	Wellington Ch.
10	**6.8** ng/g OC	Norwegian Bay
11	**6.2** ng/g OC	Lancaster Sd.
12	**12** ng/g OC	Strathcona Sd.
13	**30** ng/g OC	Jones Sd.
14	**19** ng/g OC	N. Baffin Bay
15	**56** ng/g OC	E. Hudson Bay

Figure 16-40. PCB (above) and DDT (below) ($ng \cdot g^{-1}$ organic carbon) in Arctic marine surface sediments. (from CACAR II 2003: Physical Environment, Figure B.3.2, page 101).

16.14.1 Plankton and fish

Organochlorine contaminants have been measured in plankton samples (Figure 16-41). For example, PCBs have been measured in zooplankton from the Rankin Inlet area at about 50 ng·g^{-1} (dry weight), slightly higher than other northern Canadian locations, although the number reports on plankton to date is very small. The levels of PCBs in anadromous Arctic charr were somewhat higher than in plankton (Figure 16-42), the value for charr from Sanikiluaq in the early 1990s being about 25 µg·g^{-1} wet weight. Considering that the plankton was about 50 ng·g^{-1} dry weight, and taking the moisture content to be about 80 per cent, the value for plankton on a wet weight basis might have been of the order of 10 ng·g^{-1}. Figure 16-43 shows PCBs and several other organochlorines in sea birds from the Northwater polynya with values for PCBs about 200 to 9000 ng·g^{-1} of fat in liver. The CACAR II (2003: Biological Environment) report tabulated the same or similar data in units of ng·g^{-1} wet weight and the values ranged from about 15-450 ng·g^{-1}. The most highly contaminated tissue was body fat from ivory gull, glaucous gull, and northern fulmar, the maximum value being 11719 ng·g^{-1} in glaucous gull. We do not have comparable data for birds from Hudson Bay, but similar values would be anticipated.

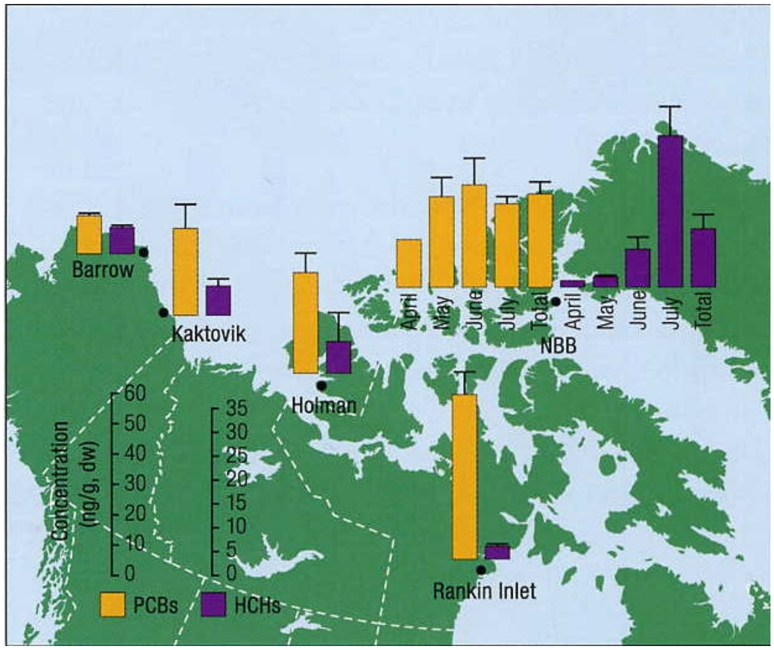


Figure 16-41. PCBs and HCHs (ng·g^{-1} dry weight) in zooplankton from several northern locations including Rankin Inlet. (From CACAR II 2003: Biological Environment, Figure 3.3.3, page 30).

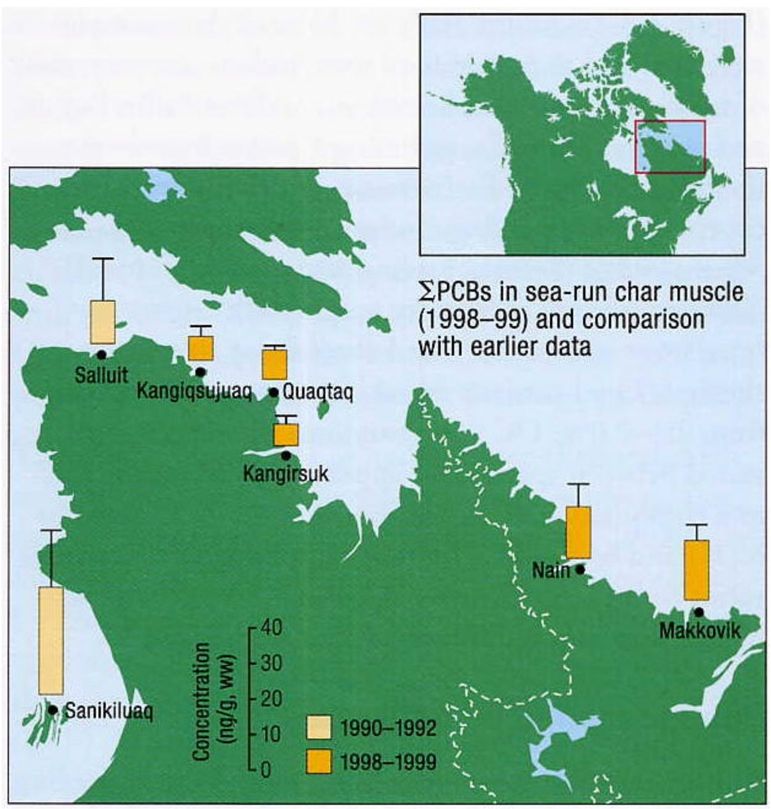


Figure 16-42. PCBs (ng·g^{-1} wet weight) in muscle of sea-run Arctic charr from several communities in 1998-1999 and from Salluit and Sanikiluaq in 1990-1992. (From CACAR II 2003: Biological Environment, Figure 3.3.5, page 32).

16.14.2 Ringed seal

The first Canadian Arctic Contaminants Assessment Report (1997) tabulated levels of chlorobenzenes, hexachlorocyclohexanes, chlordanes, DDTs, PCBs, toxaphenes, and dieldrin in blubber of ringed seals, beluga, and walrus from Hudson Bay. Figure 16-44 shows the levels of PCBs and DDT in seal blubber collected from 1989-1994. The data were presented as two bars for each location showing male and female seals separately. The levels of both PCBs and DDT were usually higher in male seals than in females. This gender difference has been observed by several investigators and is thought to be the result of females secreting

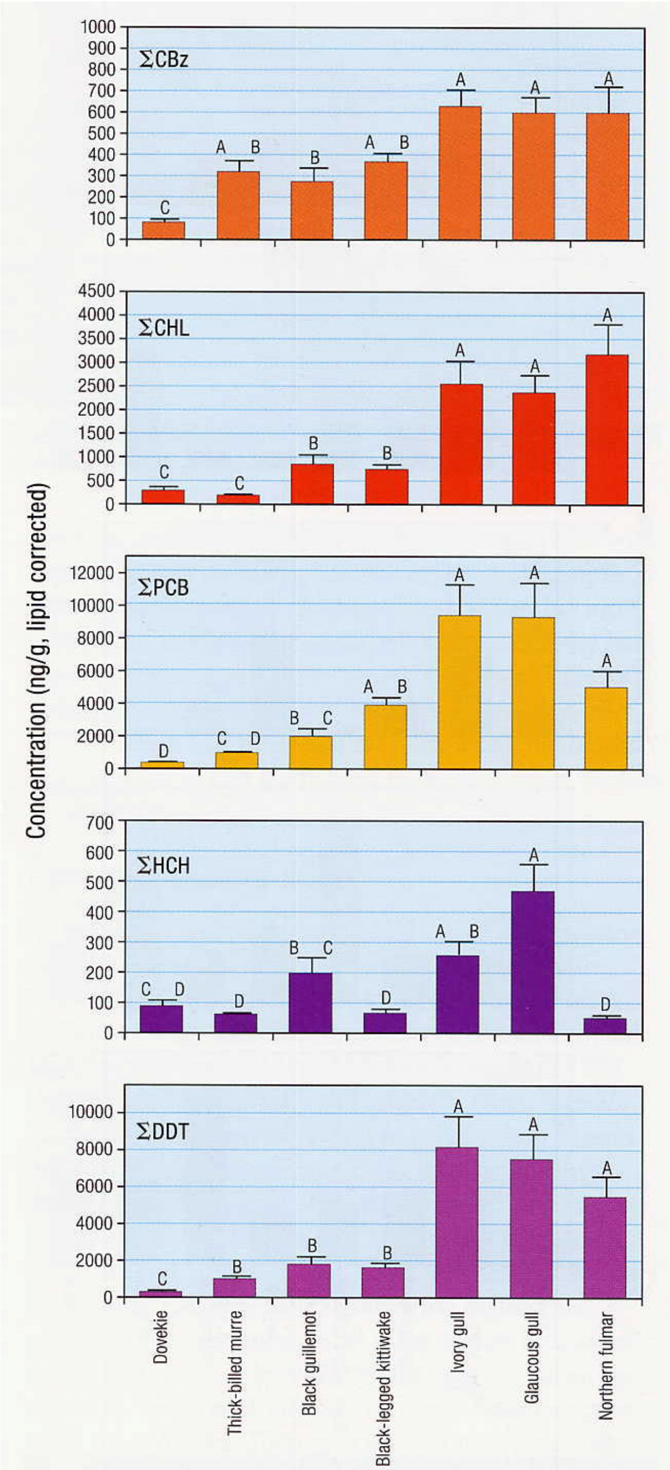


Figure 16-43. Organochlorine residues (ng·g^{-1}, lipid corrected) in liver of sea birds from the Northwater Polynya in northern Baffin Island. Bars with the same letter do not differ statistically. (From CACAR II 2003: Biological Environment, Figure 3.3.10, page 36).

contaminants with milk during lactation. The values for ΣPCB in blubber from male and female seals from Arviat were about 2.1 and 1.1 µg·g^{-1} wet weight respectively (2100 and 1100 ng·g^{-1}). The levels of ΣPCB found in seals from Inukjuak in 1989-94 were similar to those from Arviat, but the gender difference was not evident (Figure 16-44). Seals from Sanikiluaq had similar levels of ΣPCB with males again having levels about twice as high as females. With ΣDDT the results were similar except that strikingly lower levels were found in seals from Sanikiluaq than in seals from the other locations in Hudson Bay. Furthermore, the small gender difference was in the unexpected direction. More recent information on PCBs and HCHs in ringed seals collected from 1998-2000 was summarized in the CACAR II (2003) (Figure 16-45). Levels of both PCBs and HCHs in female ringed seals were quite similar across the range of locations sampled. The value for ΣPCB in seals from western Hudson Bay of about 700 ng·g^{-1} in 1998-2000 may be compared with a value of 2100 ng·g^{-1} for female seals from the same area in 1989-94. This implies a decrease in PCB contamination over the interval. The value for HCHs in blubber of seals from western Hudson Bay in 1998-2000 (about 100 ng·g^{-1} wet weight) was one of the lowest found.

16.14.3 Beluga

PCB levels in blubber of beluga from Hudson Bay were higher than those in ringed seals (Figure 16-46). The mean concentration in male beluga from western Hudson Bay in 1992-95 was about three times higher than it was in the seals. Beluga males from eastern Hudson Bay had blubber PCBs at 6.4 µg·g^{-1} but females had only 2.2 µg·g^{-1}. Only a few more recent data on beluga from the Hudson Bay region are available. The CACAR II (2003) report lists mean concentrations of PCBs in beluga blubber from the Nastapoca area of eastern Hudson Bay in 1999 as 2220 ng·g^{-1} (2.25 µg·g^{-1}) in females and 3550 ng·g^{-1} (3.55 µg·g^{-1}) in males (CACAR II 2003: Biological Environment, Annex Table 10). This suggests the unlikely situation of a decline in levels in males with no change in females. A more complete series of beluga samples from Pangnirtung (1982, 1986, 1992, 1996/97) was reported by Stern and Addison (1999). In that study, PCB congeners were measured individually in blubber of male beluga and adjusted for effects of age and fat content. The levels of all the different PCBs declined over the 15-year period from 1982 to 1997. For example, the concentration of PCB-

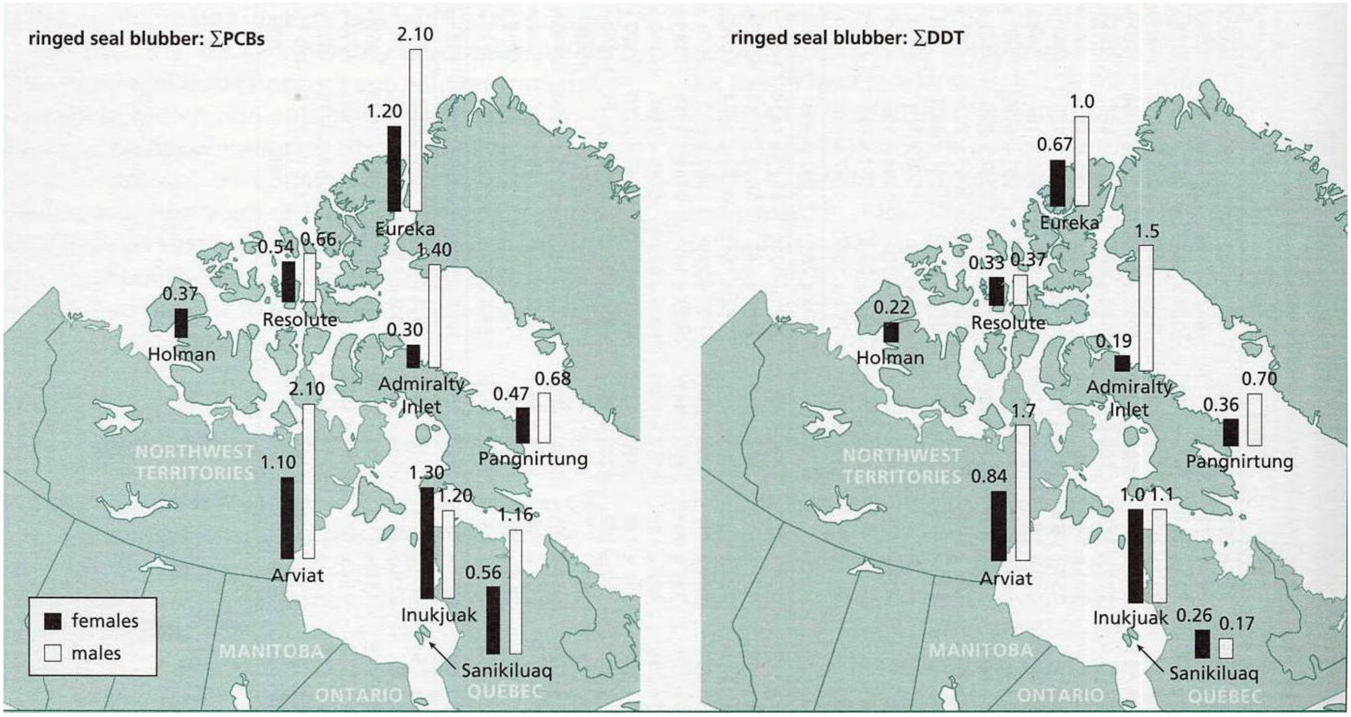


Figure 16-44. PCBs and DDT ($\mu g \cdot g^{-1}$ wet weight) in blubber of ringed seals collected from 1989 to 1994 from northern communities. (From CACAR I 1997: Figure 3.3.20 A, page 238).

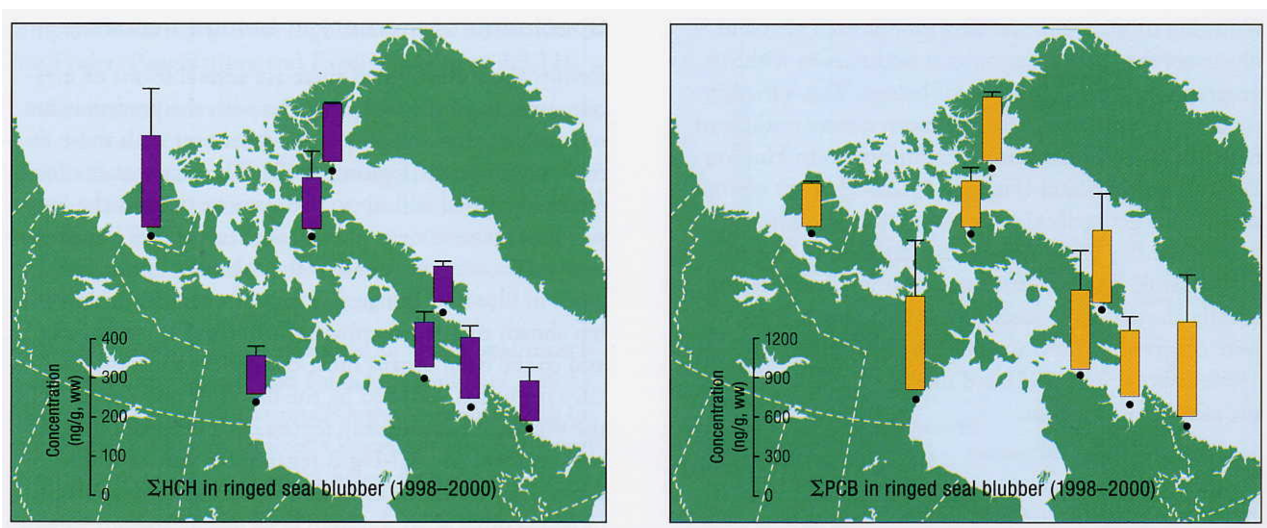


Figure 16-45. PCBs and HCHs ($ng \cdot g^{-1}$ wet weight; mean and 95% confidence interval) in blubber of female ringed seals collected between 1998 and 2000. (From CACAR II 2003: Biological Environment, Figure 3.3.14, page 42).

169 was about 220 $pg \cdot g^{-1}$ lipid in 1982 and it had fallen to about 115 $pg \cdot g^{-1}$ lipid in 1996/97 (Stern and Addison, 1999, page 208). For DDT, however, no clear temporal change was observed in the levels of total DDT, although the ratio between p,p'-DDT and ΣDDE increased suggesting that 'old' DDT was sustaining the levels rather then inputs of 'new' DDT. Levels of chlordane components have increased over the period while levels of dieldrin have fallen. Overall, there is no single pattern applicable to organochlorine compounds in general. Some have increased in the Pangnirtung samples, some have decreased, and some have remained unchanged. One can speculate that an equally complex picture will emerge from studies of Hudson Bay beluga.

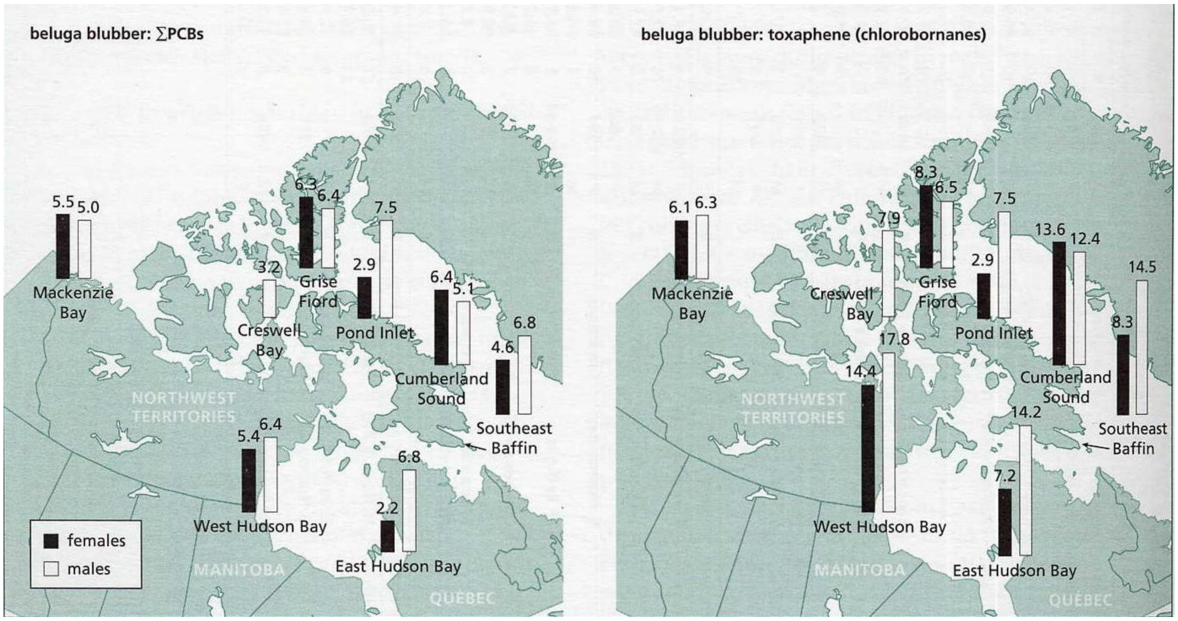


Figure 16-46. PCBs and toxaphene (µg·g^{-1} wet weight) in blubber of beluga whales collected from 1992 to 1995 from northern communities (From CACAR I 1997, Figure 3.3.20B, page 238.).

16.14.4 Polar Bear

Concentrations of several organochlorine compounds in fat of mature polar bears from the Churchill area have been measured several times over the period 1969-1998 (Figure 16-47). Recent levels of ΣPCBs were under 3000 ng·g^{-1} and all the other organochlorines reported were lower yet. The most consistent change is the decline in ΣDDT over the period from about 850 ng·g^{-1} in 1968 to about 250 ng·g^{-1} in 1999. CACAR II (2003) authors speculated that this decline is not typical of other arctic data and might have been related to cessation of spraying DDT for the control of forest insects in the Hudson Bay watershed. ΣCBz appears to have increased between 1969 and 1984 and to have declined continuously since then. Similarly, levels of σHCH have declined since 1984 but levels of ßHCH have not, with the result that the blend of the components ΣHCH has changed over the period. Similarly, PCBs have declined during the 1990s but with a change in the composition of the mixture of PCB congeners in the bears. The proportion of less chlorinated congeners has increased while the proportion of highly chlorinated congeners has fallen with the result that no long-term trend in ΣPCBs since 1968 is evident.

16.15 POTENTIAL FOR BIOLOGICAL EFFECTS OF ORGANOCHLORINE CONTAMINANTS

Some of the measurements of ΣPCB and ΣDDT in northern fish are shown in Figure 16-48 along with current estimates of the amounts required to cause biological harm. Several of the levels of both ΣPCB and ΣDDT exceed the guidelines set for the protection of birds and mammals that consume aquatic life. Levels in two groups of fish (burbot from Kusawa Lake, YK, and Greenland shark from Cumberland Sound, NU) exceed levels required for induction of biochemical responses (EROD, liver enzyme Ethoxyresorufin-O-deethylase). Figure 16-49 shows similar information for birds. Fortunately, in this instance all of the chemical residues reported are well below thresholds for biological effects. The situation is less clear in marine mammals (Figure 16-50) where ringed seal, beluga, narwhal, arctic fox and polar bear all exceed at least one no-effect level. Overall these comparisons suggest that animals at the top of arctic food chains may be at some risk of subtle biological effects from their intakes of organochlorine compounds.

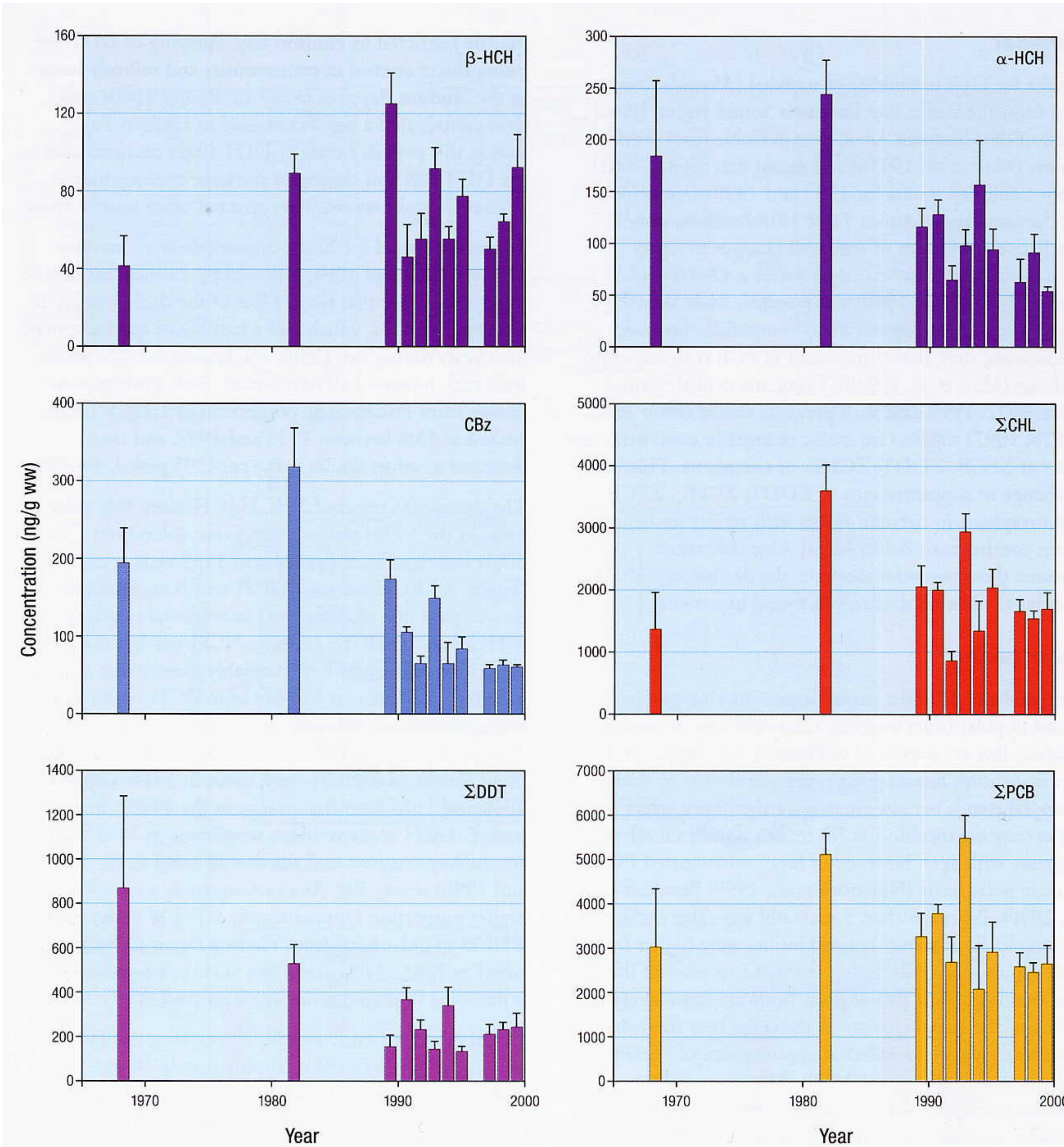


Figure 16-47. Organochlorine compounds (ng·g^{-1} wet weight) in fat of polar bears from the Churchill area from 1968 to 1999. Samples from 1991-1999 are fat biopsies; earlier samples are adipose tissue. (From CACAR II 2003: Biological Environment, Figure 4.3.6, page 76).

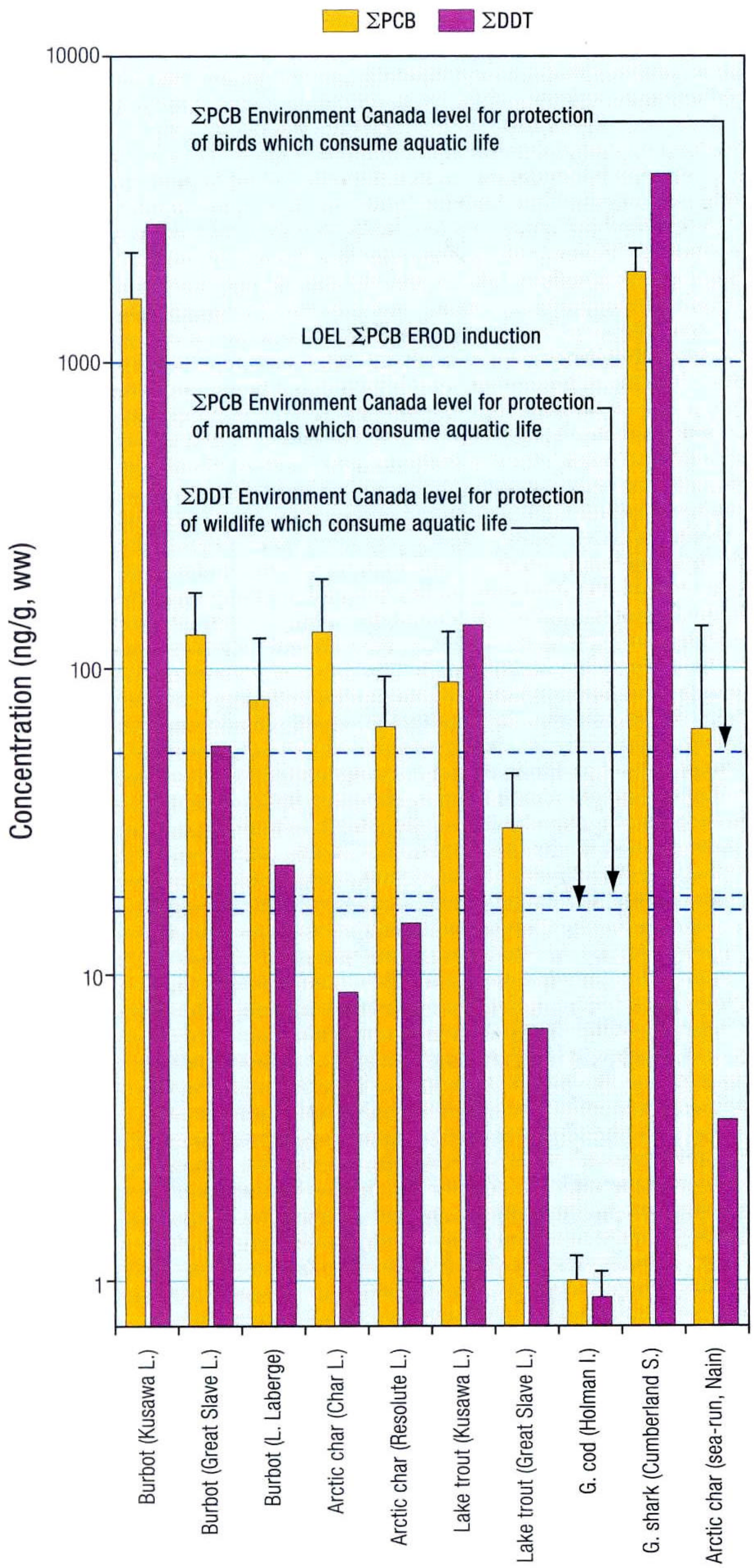


Figure 16-48. Average ΣPCB and ΣDDT levels in freshwater and marine fish compared with threshold effects levels and Environment Canada guidelines for the protection of aquatic life. References to Environment Canada guidelines and LOEL information are given in CACAR II (2003). This comparison should be used with caution because of extrapolations across tissues and species. (From CACAR II 2003: Biological Environment, Figure 5.3.1, page 96).

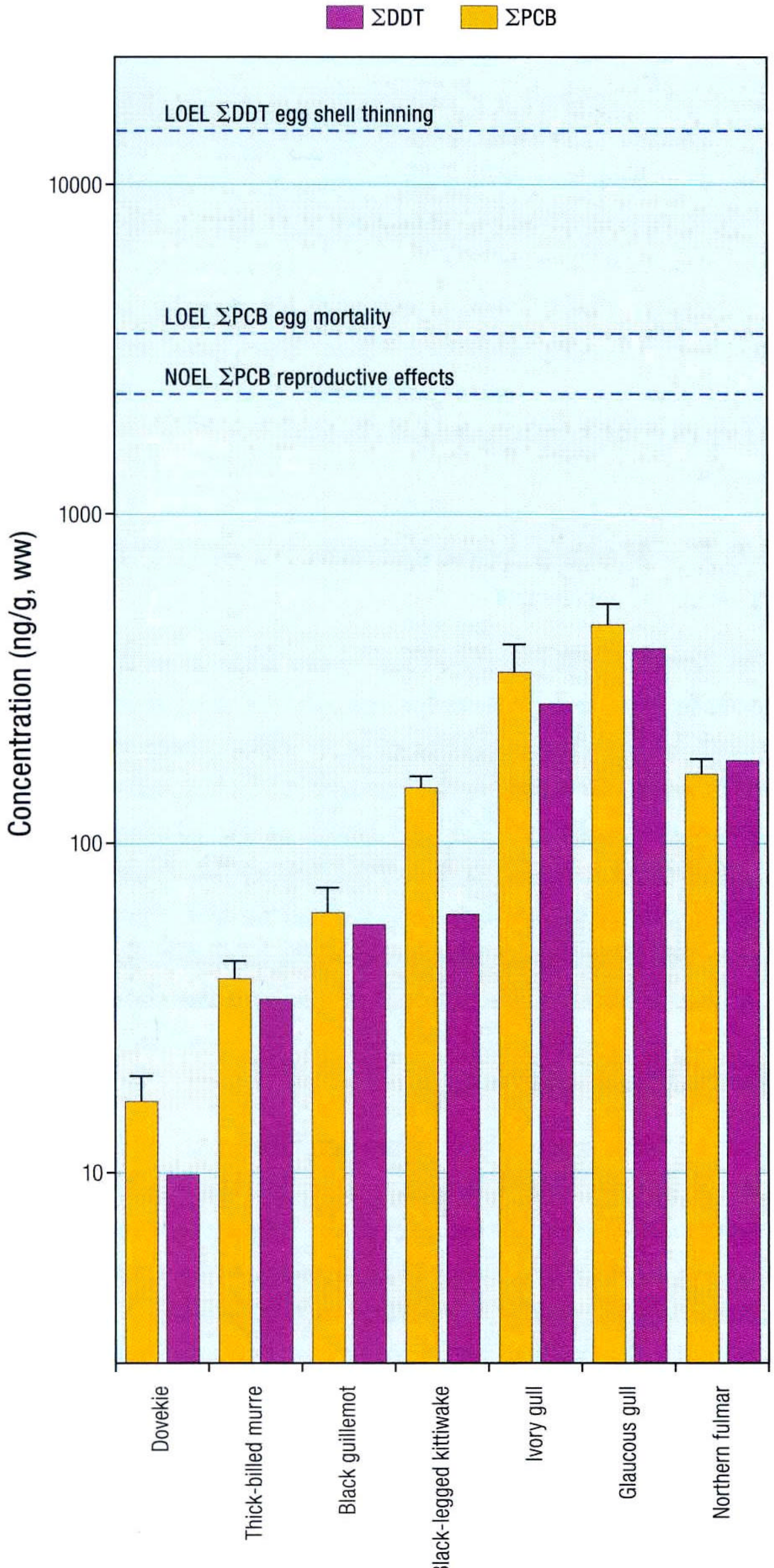


Figure 16-49. Average ΣPCB and ΣDDT levels in livers of seabirds compared with threshold effect levels. This comparison should be used with caution because of extrapolations across tissues and species. (From CACAR II 2003: Biological Environment, Figure 5.3.2, page 97).

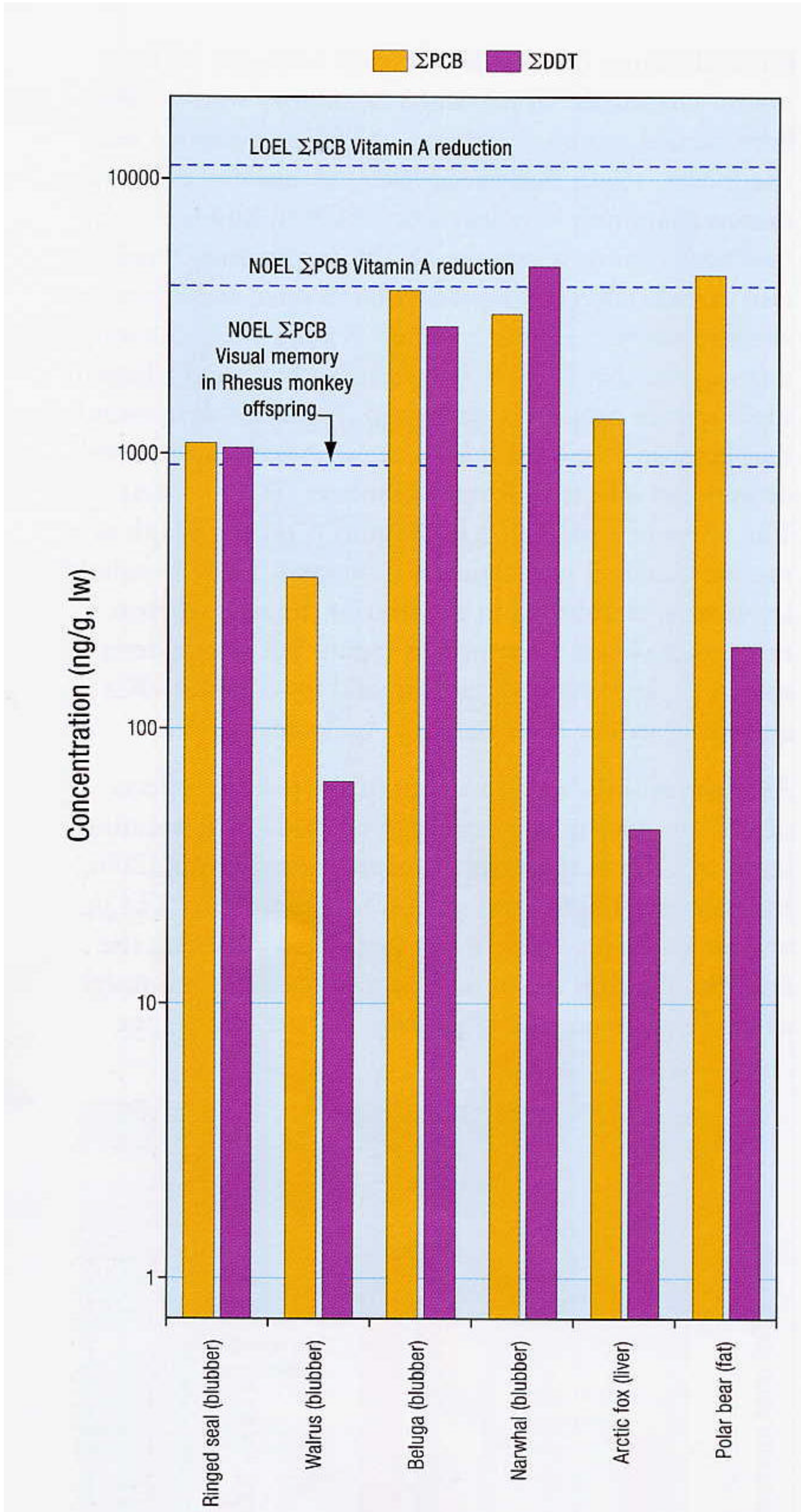


Figure 16-50. Average ΣPCB and ΣDDT levels in marine mammals compared with threshold levels for biological effects. This comparison should be used with caution because of extrapolations across tissues and species. (From CACAR II 2003: Biological Environment, Figure 5.3.4, page 98).

16.16 STABLE ISOTOPES AND ACCUMULATION OF CONTAMINANTS IN ANIMALS

A recent technique has become an important part of studies aimed at understanding the levels of contaminants in arctic animals. Through the use of stable isotopes of nitrogen, it has become possible to make a quantitative estimate of the trophic level at which an animal has been feeding. The CACAR II (2003: Biological Environment) report shows the graphical relationship between trophic level and the concentration of PCB-180 in a range of aquatic organisms and sea birds (Figure 16-51). It is apparent that the trophic level was an important predictor of the level of this PCB congener to be expected in the animals and it seems likely that this will be an important component of future studies.

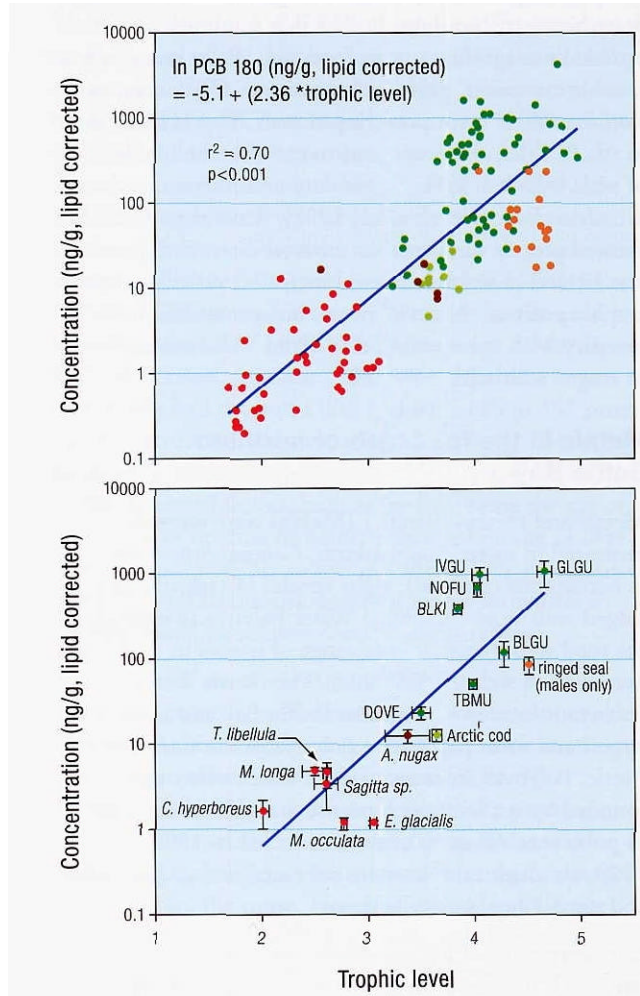


Figure 16-51. PCB 180 concentrations ($ng{\cdot}g^{-1}$, lipid corrected) and trophic status in species from the North Water Polynya. The top graph contains all the data points and the lower graph contains the mean (± S.E.) for each species. Trophic levels are assigned based on the nitrogen isotope ^{15}N. Red dots, pelagic zooplankton; Brown dots, benthic amphipods; Light green dots, arctic cod; Orange dots, ringed seals; Dark green dots, seabirds (GLGU, glaucous gull; IVGU, ivory gull, NOFU, northern fulmar, BLGU, black-legged gull; BLKI, black-legged kittiwake; TBMU, thick-billed murre; DOVE, dovekie) (From CACAR II 2003: Biological Environment, Figure 3.3.19, page 54).

16.17 RECOMMENDATIONS

The comparison of results from Henderson (1989) with ISQG and PEL values is compromised because they refer to different size fractions. Henderson (1989) analyzed the clay-size fraction but the ISQG and PEL are expressed in terms of whole, unfractionated sediment. In order to make legitimate comparisons, the measurements need to be expressed in the same units as the ISQG and PEL. This cannot be done on the basis of information available. Given the problem with mercury in arctic animals generally, it would be desirable to select an appropriate sub-set of sediment samples held by the Geological Survey and analyze them for mercury.

Hudson Bay is strongly influenced by inputs of freshwater. Data on contaminants in the water of Hudson Bay are very rare but Yates (1993) estimated that Hudson Bay supplies about as much of several metals (Mn, Fe, Co, Ni, Cu, Zn) to the Atlantic as does the Gulf of St. Lawrence. Additional future effort will be required to describe mass budgets of several key elements (notably Hg, Cd and synthetic organic compounds).

The large body of data produced by the Geological Survey of Canada and described briefly by Painter et al. (1994) is remarkable for its scope. From the maps included in this report, it is clear that a significant proportion of those data apply to areas draining into Hudson Bay or Foxe Basin. If the raw data are available, then further analysis of them would likely lead to additional insights about Hudson Bay sediments.

Data on mercury in marine fish and invertebrates from Hudson Bay are fragmentary. Levels are sometimes high enough to raise questions regarding subsistence consumption. There is evidence that hydroelectricity projects have exported mercury downstream to marine habitat and more of these projects can be anticipated in view of the projected shortages of certain fossil fuels in the coming decades. A systematic survey of mercury in marine invertebrates and fish will be required to determine the impacts of existing and future developments in the watershed.

There is the potential for impacts of regional climate warming in drainages to the west of Hudson Bay. This trend can result not only in habitat change but also in changing fluxes of materials to Hudson Bay. In the instance of mercury, for example, warming of whole permafrost-dominated basins to the west of Hudson Bay would be expected to result in increased erosion of particles and increased methylation and hence biological accumulation of mercury. This might be detected most readily by sedimentation studies in selected estuaries.

Organochlorine compounds differ somewhat from site to site even within the Hudson Bay/Foxe Basin/Hudson Strait area. With the exception of DDT, which seems to be declining at least in some species, temporal trends are still difficult to determine. With differences from one location to another, future work will have to be done on a site-by-site basis. Sentinel organisms and sites need to be selected for repeated monitoring over the coming years. Otherwise, even relatively large studies risk becoming too diffuse to detect temporal changes with high statistical confidence.

Levels of some organochlorines and mercury in some of the animals are high enough in some instances to pose a risk of biological injury. There is an almost complete lack of experimental work to find out whether existing levels are meaningful biologically. While this is understandable with some of the large species, toxicology experiments can and should be done with some of the smaller animals like fish and invertebrates, and with common laboratory surrogates for the large mammals.

Studies in areas outside Hudson Bay have revealed growing inputs of new stable chemicals like polybrominated diphenylethers (Figure 16-52). Future work on Hudson Bay biota should include assessment of these and other new chemicals.

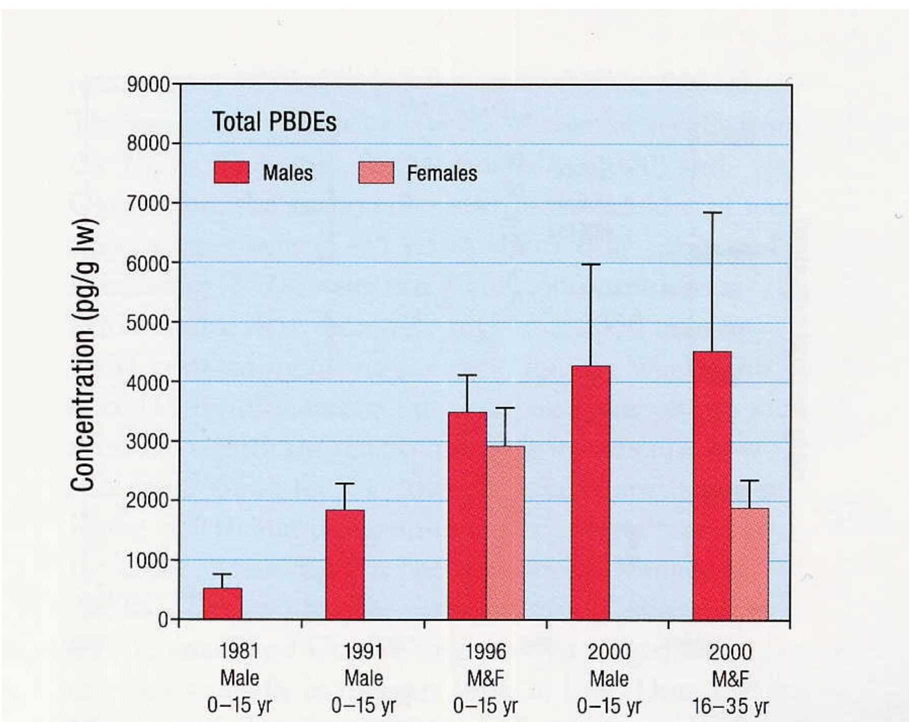


Figure 16-52. Increasing concentrations of polybrominated diphenylethers (dibrominated to hepta-brominated congeners) in ringed seal blubber from Holman in the western Canadian Arctic. (From CACAR II 2003: Biological Environment, Figure 4.3.7, page 77).

The watersheds draining to Hudson Bay are being changed by human activities, notably population growth and associated business activity, agriculture, hydroelectric development, and climate change. In addition to alterations in watersheds, there is direct loading to the water surface of Hudson Bay by materials dispersed via atmospheric circulation. With the relatively limited attention contaminants have been given in Hudson Bay, existing analyses and those done in the coming few years have ‘benchmark’ quality that will help to assess the magnitude and significance of future changes.

16.18 SUMMARY

Synthetic organochlorines and radionuclides produced by nuclear fission are two groups of contaminants found in the Hudson Bay marine ecosystem that result exclusively from human activities. They are products of 20^{th}-century technology and have no natural sources or natural background concentrations. These compounds reach the Arctic by several means but one important pathway is via moving air masses. The have little or no history of use in the North but occur throughout the Arctic in air, water, sediment, and aquatic life.

Studies of the composition of the air from the Canadian North have consistently identified a wide range of synthetic organic compounds that originate thousands of km away. Organochlorine compounds differ somewhat from site to site even within the Hudson Bay/Foxe Basin/Hudson Strait area. Several contaminants move through food chains to become concentrated in animals at high trophic levels. With the exception of DDT, which seems to be declining in some species, temporal trends are still difficult to determine. With differences from one location to another, future work will have to be done on a site-by-site basis. Sentinel organisms and sites need to be selected for repeated monitoring over the coming years. Otherwise, even relatively large studies risk becoming too diffuse to detect temporal changes with high statistical confidence.

Few data exist describing organic contaminants in Hudson Bay sediments. But, high levels of both DDT (56 $ng \cdot g^{-1}$ OC) and PCBs (920 $ng \cdot g^{-1}$ OC) have been reported from eastern Hudson Bay relative to other Arctic locations.

Levels of both PCBs and HCHs in female ringed seals were quite similar across the range of Arctic locations sampled. The value for ΣPCB in seals from western Hudson Bay of about 700 $ng \cdot g^{-1}$ wet weight (ww) in 1998-2000 may be compared with a value of 2100 $ng \cdot g^{-1}$ ww for female seals from the same area in 1989-94. This implies a decrease in PCB contamination over the interval. The value for HCHs in blubber of seals from western Hudson Bay in 1998-2000 (about 100 $ng \cdot g^{-1}$ ww) was one of the lowest found. PCB levels in blubber of belugas from Hudson Bay were higher than those in ringed seals. The mean concentration in male belugas from western Hudson Bay in 1992-95 was about three times higher than it was in the seals.

Concentrations of several organochlorine compounds in fat of mature polar bears from the Churchill area have been measured several times over the period 1969-1998. Recent levels of ΣPCBs were under 3000 $ng \cdot g^{-1}$ ww and all the other organochlorines reported were lower yet. The most consistent change is the decline in ΣDDT over the period from about 850 $ng \cdot g^{-1}$ ww in 1968 to about 250 $ng \cdot g^{-1}$ ww in 1999. CACAR II (2003) authors speculated that this decline is not typical of other Arctic data and might have been related to cessation of spraying DDT for the control of forest insects in the Hudson Bay watershed. ΣCBz appears to have increased between 1969 and 1984 and to have declined continuously since then. Similarly, levels of σHCH have declined since 1984 but levels of ßHCH have not, with the result that the blend of the components ΣHCH has changed over the period. Similarly, PCBs have declined during the 1990s but with a change in the composition of the mixture of PCB congeners in the bears. The proportion of less chlorinated congeners has increased while the proportion of highly chlorinated congeners has fallen with the result that no long-term trend in ΣPCBs since 1968 is evident.

Studies in areas outside Hudson Bay have revealed growing inputs of new stable chemicals like polybrominated diphenylethers. Future work on Hudson Bay biota should include assessment of these and other new chemicals.

The source of anthropogenic cesium-137 in this area was atmospheric testing of nuclear bombs, most of which ended in 1963 by international agreement. Little information exists on radionuclides in the aquatic biota of Hudson Bay. Levels of cesium-137 in animals today will have fallen below the values recorded in the early 1980s because the half-life of cesium-137 is only 30 years and inputs of new Cs-137 have fallen dramatically. The natural radionuclides polonium-210, radium-226, thorium-230, thorium-232, uranium-234, and uranium-238 were measured in a series of surface sediment samples collected in 1995 near Rankin Inlet. The highest levels of these radionuclides usually were found closest to the community of Rankin Inlet.

Human activities may also redistribute naturally occurring elements, including toxic heavy metals and metal-like compounds (e.g., arsenic), to the Hudson Bay marine ecosystem. Their presence is not, in itself, evidence of human activity because many elements are present naturally in soils and sediments. However, human activities often move elements about in the environment in ways and at rates not found naturally. The questions that arise in cases of suspected contamination by elements are whether the amounts found exceed natural background amounts and what the sources might be. Among the more toxic elements are cadmium, copper, chromium, lead, mercury, and zinc and the metalloids arsenic and selenium. In view of the potential for these elements to produce biological harm, Canada has established Interim Sediment Quality Guidelines (ISQG) and Probable Effect Levels (PEL). These guidelines describe levels that should not be exceeded in freshwater and marine sediments in order to avoid biological impacts.

Chemical contaminants often become incorporated into aquatic sediments. Their horizontal and vertical distribution in the sediments provides a valuable record of where and when contamination has occurred. The Geological Survey of Canada (GSC) has collected seafloor sediments throughout Hudson Bay, using grab samplers. Henderson (1989) reported the data from these samples, which provide an overview of the surficial distribution of many important elements. These results cannot be compared directly with ISQG and PEL values because they refer only to the clay fraction and not in the unfractionated sediment, but the error is likely conservative since metals are usually more abundant in clay-sized particles than in larger particles. Vertical core samples of the bottom sediment have been used to examine the deposition history at a few locations.

The distribution of elements for which sediment quality guidelines have been established found little evidence of contamination. The highest concentrations of arsenic are mostly located in central Hudson Bay. The levels reported likely indicate a natural background for arsenic in fine sediment particles distributed by natural processes. Most of the sites with chromium over the PEL were offshore in southwest Hudson Bay, suggesting the possibility of chromium-enriched sediments originating from drainages entering Hudson Bay from the west. It is not known whether the existing levels in the sediment represent a risk to the biota of southwestern Hudson Bay. In general, the highest copper values were found off the west coast of Hudson Bay near the Churchill River. Copper enrichment has also been found in the sediment north of Arviat and between Chesterfield Inlet and Rankin Inlet. The distribution of copper may be explained by sediment transport offshore from Kivalliq. Only one site exceeded the PEL for lead. The core profile data suggest that Hudson Bay has received inputs of anthropogenic lead over the last century, probably from atmospheric fallout. There is no obvious clustering of high or low zinc values. Copper, lead, and zinc concentrations from bulk surficial sediment samples taken at five Ontario rivers, immediately before they enter Hudson Bay or James Bay, were well below those from the clay-size sediments from Hudson Bay.

No Interim Sediment Quality Guidelines have been established for aluminum, calcium, cobalt, iron, magnesium, manganese, molybdenum, nickel, or potassium. Several of these are major components of the earth's crust and the concept of ISQG does not apply. Others are trace metals for which ISQG values have not been established. These elements, along with oxygen, silicon and sodium, are the major components of the earth's crust. Conditions in the deeper areas of central Hudson Bay likely favour the diagenesis of manganese. The presence and levels of these elements in the sediments of Hudson Bay do not infer anthropogenic impacts.

Mercury is present naturally in the environment of Hudson Bay but is also added by a number of human activities. The GSC samples were not analyzed for mercury, so its distribution the surface sediment of the Hudson Bay seafloor is unknown. Given the problem with mercury in Arctic animals generally, an appropriate sub-set of sediment samples held by GSC should be analyzed for mercury. Mercury levels in core samples taken from southeast Hudson Bay and from VanVeen dredge samples taken near Rankin Inlet were well below the ISQG and PEL values. They do not portray a problem with mercury. Surveys of mercury in aquatic sediments have identified some high values in the drainages to the southwest of Hudson Bay. However, regional climate warming might result not only in habitat change but also in changing fluxes of materials to Hudson Bay. In the case of mercury, for example, warming of permafrost-dominated basins to the west of Hudson likely increases the erosion of particles and methylation of inorganic mercury, and hence biological accumulation of mercury. This might be detected most readily by sedimentation studies in selected estuaries.

Despite the low levels of mercury in the sediments, there is a persistent problem with accumulations of mercury in marine animals high in the food chains. Two guideline figures are used in efforts to limit human intake of mercury. Concentrations should not exceed 0.5 $\mu g \cdot g^{-1}$ (wet weight) in fish sold commercially in Canada, and levels should not exceed 0.2 $\mu g \cdot g^{-1}$ in fish used for subsistence.

So little is known about mercury levels in marine fish of Hudson Bay that the few scattered reports available do little more than establish the need for a systematic survey of levels of mercury in marine fish and their supporting food chains. Levels are sometimes high enough to raise questions regarding subsistence consumption. However, mercury levels in anadromous charr are considerably lower than those in predatory landlocked fishes, such as lake trout, implying that feeding at sea supplies less mercury than feeding in lakes. There is evidence that hydroelectricity projects on the La Grande River have exported mercury downstream to marine habitat. A systematic survey of mercury in marine invertebrates and fishes will be required to determine the impacts of existing and future developments in the watershed, and the effects of climatic warming in western Hudson Bay watersheds.

Of all northern animals, marine mammals generally have the highest levels of mercury. Most chemical residue analyses have been done with ringed seals, beluga whales, and polar bears. Ringed seal and beluga whale livers contain high levels of mercury but only a small proportion of it is present as neurotoxic methylmercury. Mercury levels seem to have increased since the mid- 1980s at Arviat and possibly at Sanikiluaq, although the latter collections were only four years apart. The whales harvested near Arviat summer downstream from the Churchill-Nelson hydroelectric development. The levels of mercury in the whales vary greatly among organs.

Since the whales' major intake of mercury is from dietary methylmercury, the presence of a high proportion of apparent HgSe in the liver may represent a metabolic detoxification mechanism to bind the mercury as an inert form and render it non-toxic or at least less toxic. The levels of mercury in beluga, narwhal and ringed seal livers consistently exceeded those in walrus. One might expect polar bears to contain higher amounts of mercury than their prey but this is not the case. This apparent discrepancy is explained by the fact that the bears eat only the blubber of the seals, not the protein-rich organs like muscle and liver where more mercury is found.

The data from sediments available to date do not suggest a contamination problem with cadmium. The value given for cadmium in liver of seals from Arviat was about 12 $\mu g \cdot g^{-1}$ ww. This appears to be below the threshold for biological effects, which is 20-200 $\mu g \cdot g^{-1}$ ww. A small proportion of the seals would reach or exceed the lowest part of the effects range.

Levels of some organochlorines and mercury in some of the animals are high enough in some instances to pose a risk of biological injury. There is an almost complete lack of experimental work to find out whether existing levels are meaningful biologically. While this is understandable with some of the large species, toxicology experiments can and should be done with some of the smaller animals like fish and invertebrates, and with common laboratory surrogates for the large mammals.

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