Environmental Studies No. 70

Synopsis of Research Conducted Under the 1992/93 Northern Contaminants Program

Canada

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Northern Affairs Program

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Published under the authority of the
Hon. Pauline Browes,
Minister of Indian Affairs and
Northern Development,
Ottawa, 1993.

QS-8490-000-EF-A1
Catalogue No. R71-19/70-1993E
ISBN 0-662-21037-9

This report was prepared under the Arctic Environmental Strategy, coordinated by the Natural Resources and Environment Branch, Department of Indian Affairs and Northern Development.

The views, conclusions and recommendations expressed herein are those of the authors and not necessarily those of the department.

FOREWORD

This report summarizes the results of research and monitoring studies on contaminants in northern Canada. These studies were conducted under the auspices of the Green Plan's Arctic Environmental Strategy/Action on Contaminants Program.

The studies cover all aspects of the northern contaminants issue, including sources and transport, contamination of marine, freshwater and terrestrial ecosystems, human exposure through diet and related health implications, communication and education of northern residents, and international initiatives addressing the global aspect of the problem.

These projects were evaluated by the Technical and Science Managers Committees on Contaminants in Northern Ecosystems and Native Diets to ensure they supported the overall Northern Contaminants Program objectives.

PRÉFACE

Ce rapport présente les résultats des études de contrôle de la pollution et d'une recherche sur les contaminants dans le Nord canadien. Ces études ont été menées dans le cadre de la Stratégie pour l'environnement arctique du Plan vert du Canada, Programme d'action sur les contaminants.

Elles portent sur tous les aspects du problème des contaminants, incluant les sources et le transport, la contamination des écosystèmes marins, d'eau douce et terrestre, l'exposition de l'organisme humain en raison de son régime alimentaire et ses effets sur la santé, la communication et l'éducation des résidents nordiques et les initiatives internationales qui se penchent sur l'aspect global du problème.

Les comités de gestionnaires techniques et scientifiques sur les contaminants dans les écosystèmes du Nord et les régimes alimentaires des Autochtones ont examiné ces projets afin de s'assurer qu'ils avaient bien répondu à l'ensemble des objectifs du Programme d'action sur les contaminants.

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INTRODUCTION

An Arctic contaminants research and monitoring program was initiated in 1985 in response to concerns about persistent organic, heavy metal and radionuclide contamination in the north. This program, which was expanded in 1991/92 under the auspices of the Arctic Environmental Strategy (AES) of Canada's Green Plan, aims to identify the sources and pathways by which these pollutants enter the Arctic and to provide a scientific basis for the assessment of their effects on the northern environment, its wildlife and its people in order that appropriate actions may be taken. Particular attention is paid to evaluating health risks to northern indigenous populations which rely on traditional food sources, such as fish, marine mammals and caribou, in which toxic chemicals may concentrate.

During 1992/93, a number of federal and territorial government departments, universities, consulting firms and commercial laboratories participated in over 45 projects totalling $4.8 million of the program's approximately $35 million, six-year budget. While the majority of these projects are ongoing studies, for example measuring temporal trends of chemicals in wildlife, the program is addressing new research needs as they emerge.

In recent years of the program, the importance of northern communications regarding the contaminants issue, especially research results which directly affect native residents, has been increasingly recognised. Activities of this nature in the program have included steps to establish guidelines for responsible research in the Arctic, which will address items such as the need for consulting with native communities about proposed projects, considering traditional environmental knowledge, the training and employing, where possible, of local residents, and the communication of findings to local communities.

It has been recognized for some time that many contaminants found in the Arctic are imported by the atmosphere, ocean currents and rivers, from origins in more industrialized regions further south. Therefore, protection of the arctic environment will require global cooperation aimed at reducing or banning the use of toxic substances, such as pesticides and industrial chemicals, in industrialized countries. Over the past year, the AES/Northern Contaminants Program (NCP) has significantly contributed to several international initiatives directed towards these objectives:

1) The projects carried out and planned under the NCP form the basis of Canada's contribution to the Arctic Monitoring and Assessment Programme (AMAP), which is a component of the Arctic Environmental Protection Strategy (AEPS). The AEPS Declaration on Protection of the Arctic Environment was signed by the eight circumpolar nations in June 1991. The priorities for the monitoring phase of AMAP have been established and this phase was initiated in 1993 through individual country implementation plans. The goals and objectives of the Northern Contaminants Program are directly compatible with those of AMAP and studies carried out under the NCP comprise Canada's implementation plan for AMAP. Cooperative contaminant studies through a bilateral agreement are planned with Russia to aid its contribution to AMAP. Collectively, the AMAP programs will help delineate how contaminants are reaching the circumpolar Arctic, and will assist in the identification of sources.

12) The United Nations Economic Commission for Europe (UNECE) Task Force on Persistent Organic Pollutants under the Convention on Long-Range Transboundary Air Pollution (LRTAP) is mandated to draft by 1994 the elements of a possible protocol for the international control of persistent organic pollutants. Canada is playing a key role in this initiative and our contributions relating to quantifying atmospheric transport, and to formulating control and abatement strategies for adoption in the protocol are being funded by the AES/Northern Contaminants Program.

As part of the program's evaluation of results and progress of research, a workshop is being held November 23-24, 1993 in Winnipeg at the Freshwater Institute of Fisheries and Oceans Canada, for participants of the Northern Contaminants Program to present their research findings for the 1992/93 fiscal year, along with preliminary results from projects funded for 1993/94.

This report outlines the findings of research and monitoring studies carried out under the AES/Northern Contaminants Program during 1992/93.

SOURCES, PATHWAYS AND SINKS

ATMOSPHERIC EMISSIONS OF TOXIC SUBSTANCES

PROJECT LEADER: E.C. Voldner, Atmospheric Environment Service, Environment Canada, Downsview, Ontario

PROJECT TEAM: Y.-F. Li and E.C. Voldner

OBJECTIVE

On a global scale, determine sources and atmospheric emission of acidifying and toxic pollutants, such as SO_x, NO_x, volatile organic compounds (VOC), metals and pesticides, as the basis for modelling activities and international action.

RATIONALE

Estimation of contaminant source data (emissions) on a global scale is required for input to models (refer to Pudykiewicz 1992) for the determination of source regions that may have a significant impact on sensitive areas such as the Arctic and the Great Lakes regions. Establishment of source/receptor relationships and relative contributions of sources in other countries to receptors in Canada will allow assessment of control strategy scenarios and hence aid Canada in the development of international control policies. Evaluation of historical and future trends in emissions are important for understanding and predicting response time of various components of the ecosystem to changes in the atmospheric load.

DESCRIPTION

Phase I

Acidifying Species and Metals

The initial effort focuses on SO_x, NO_x, VOC and lead. Information on anthropogenic emissions has been sought from organizations or programs throughout the world or estimated through surrogate methods such as emission factors and statistical information on fuel use, production capacity, population, etc. A computerized database will be created and emissions well be gridded for input to models. The emission inventories will be updated as the information base improves.

Pesticides

Initial effort involved the compilation of information and the creation of a computerized database of historical, present and predicted global usage or sales of persistent pesticides such as Aldrin, Dieldrin, Chlordane, DDT, Endosulfan, Endrin, HCH, Lindane, Heptachlor and Toxaphene.

The information sought included registration status; amount of use/emission; region of use/emission; mode and time of application; physical and chemical properties of the active ingredient as well as of the technical mixture; and representative concentrations in soils. The information is being obtained through literature surveys and contact with international agencies and researchers.

Phase II

Phase II consists of the determination of time-dependent, spatially gridded emission inventories of the specified compounds for input to the hemispherical/global model. Based on the information obtained in Phase I, regionally representative crop and soil data, as well as climatological hourly meteorological data, a time-dependent numerical model will be executed to quantify the emissions. The latter model was developed by the Atmospheric Environment Service for predicting the volatilization of pesticides and other toxic materials from vegetated soils. Emission estimates will be compared with limited measurements.

Phase III

Phase III will involve, if warranted, an expansion to the northern hemisphere of a North American study on emissions of current use pesticides, being conducted under the Great Lakes Water Quality Program.

ACTIVITIES IN 1991/92

In 1991/92, efforts were essentially devoted to the search for and retrieval of basic information as well as further development of the air/soil exchange model. The progress was slower than expected due to difficulties in obtaining information, deficiencies in information obtained, and delay in funding and subsequent placement of contracts.

Data for SO_x, NO_x, VOC and Pb for various regions were compiled and quality controlled. First estimates of annual emission inventories of SO_x, NO_x and VOCs were prepared.

Battelle Research Centre in Geneva, Switzerland was contracted to provide information on organochlorine usage in the western hemisphere, while the International Registry for Potentially Toxic Chemicals (IRPTC) of the United Nations Environmental Program arranged contracts with local experts in the former Soviet Union, the Baltic Republics, the former East Germany and the People's Republic of China. A literature search and contacts with researchers were initiated. Analysis of data was initiated.

ACTIVITIES IN 1992/93

Due to changing priorities, work on SO_x, NO_x, VOC and Pb wound down, since there would be no future support under the Northern Contaminants Program. The effort will continue,

however, through support from other national and international programs. Phase III regarding North American pesticides received partial funding.

A computerized database including software for analysis and display of data was established (as presented at the Database Workshop, Burlington, February 1993), which is part of the embryonic Canadian Global Emissions Inventory Center.

Phase I

Compilation of data continued but emphasis shifted from SO_x and NO_x to persistent pesticides from IRPTC/UNEP, Pesticide Action Network, and records from individual countries. Contracts with the former East Bloc countries experienced significant delays due to the political and economic situation in these countries. Only hard copies of the information were received, necessitating manual entry in the database. Only data for toxaphene has been reviewed.

The data vary in quality, and show large spatial and temporal gaps. When conflicting data were found for a period and region, the data "deemed of best quality" were used. This subjective judgement was based on the method by which the data were obtained, comparison with other regions, registration permits, etc. No interpolation in space was attempted, while temporal interpolation was based on auxiliary information such as production, registration, and trends in other regions. Obviously, this introduces a great deal of uncertainty in the reported amounts.

RESULTS

Figure 1 (Voldner and Li 1993) shows the registration status of toxaphene. The figure indicates where and when the use of toxaphene was banned, where it is not registered, where it is severely restricted, and regions for which no information was found. Thus toxaphene was banned in the United States in 1986 and in 1990 in the former East Germany, has not been registered in Canada since 1983, may still be used in Mexico and for certain applications in former Soviet Union, while information for Africa is essentially lacking.

Figure 2 provides the amount which has been accounted for in the search, which totals about 450 000 tonnes. Accumulated interpolated total use for the period 1970 to present is shown in Figure 3. The accumulated total from 1970 to the present is 670 000 tonnes. The amount used in a country is indicated by colour code and shown for the whole country rather than attributed to a specific region. This can be very deceiving. Thus usage in the United States was primarily concentrated in the southern cotton belt, with no known usage in Alaska. The largest amount of use appears to have been in the United States, the former Soviet Union, and Central America. This may in part be true, but could be an artifact since more detailed information may show other regional use to be high.

CONCLUSION

Data on present and historical uses of persistent organochlorines such as toxaphene are difficult to obtain and uncertain. The reported information should therefore be viewed with caution.

It appears that many countries do not keep records on pesticides, while in other countries such information is confidential. A limited number of countries report to the Food and Agriculture Organization (FAO). Commercial databases such as assembled by the Battelle Europe are proprietary and costly to access. The IRPTC/UNEP do not routinely compile information, however, information in the former East Bloc has been obtained through them. Such service, however, is not free.

To improve the database of toxaphene and pesticides in general, international cooperation is required. As a step in this direction a project on organochlorines has been established under the Global Emissions Inventory Activity of the International Geosphere Biosphere Program. This avenue will be pursued in 1993/94.

Phase II

Required data have been received and processed for input to the air/soil exchange model and emissions estimates are being computed. As an example, Figure 4 shows the temporal pattern of lindane emission rates following uniform incorporation to a depth of 10 cm in soil on Julian day 91 in year 1 and subsequent tilling on day 456 in year two. The base year is 1989 and the location is in southern Ontario. The secondary peak due to tilling is clearly shown. Differences in volatilization of various pesticides are illustrated in Figure 5. The pesticides are identified in Table 1 (reference, Scholtz *et al.* 1993).

Gridded estimates of emissions will be computed in 1993/94.

Phase III

Progress continued under the Great Lakes Water Quality Progress on the assembly of information on current use pesticides.

In addition to the technical aspects of these various activities, a great deal of time in 1992/93 was spent in negotiating internationally the direction of a number of emissions-related programs and in emphasizing the fundamental importance of work of this nature in understanding pollutant movement on a global basis.

CONCLUSION AND FUTURE DIRECTIONS

The task of compiling global emission inventories is,as described in this report, fraught with problems. We have now investigated most "obvious" sources of information and have completed much of the contract work required to establish preliminary data.

In 1993/94 effort will be focused on consolidating the information already obtained and on fulling in gaps as far as possible. Continued presence in the international arena on this issue will be necessary to ensure that we have the most recent information and that international efforts continue to emphasize the importance of this type of data.

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Scholtz, M.T., C.F. Slama and E.C. Voldner. 1993. Pesticide emission factors from agricultural soils. Presented at the AWMA Conference, Denver, Colorado, June 1993.

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Table 1. Pesticides considered in this report. The numbers refer to those plotted in Figure 5.

Pesticide No.	Name	Pesticide No.	Name
2	2,4-D	61	DISULFOTON
6	ALACHLOR	62	DIURON
7	ALDRIN	64	ENDOSULFAN
10	ATRAZINE	66	ENDRIN
17	BIFENOX	69	ETHOPROP
18	BROMACIL	81	HEPTACHLOR
23	CAPTAN	82	HEXACHLOROBENZENE
24	CARBARYL	87	HEXAZINONE
25	CARBOFURAN	91	LACTOFEN
30	CHLOROPICRIN	92	LINDANE
33	CHLORPROPHAM	100	METHAZOLE
34	CHLORSULFURON	102	METHYL PARATHION
36	CLOMAZONE	104	METOLACHLOR
38	CYANAZINE	107	MIREX
46	DALAPON	117	PARATHION
49	DIAZINON	122	PICLORAM
50	DICAMBA	125	PROMETRYN
54	DIELDRIN	139	THIOBENCARB
55	DIETHATYL ETHYL	140	TOXAPHENE
57	DINOSEB	144	TRIFLURALIN
59	DIPROPETRYN		

Figure 1. Restriction Status for Toxaphene Surveyed in 1992.

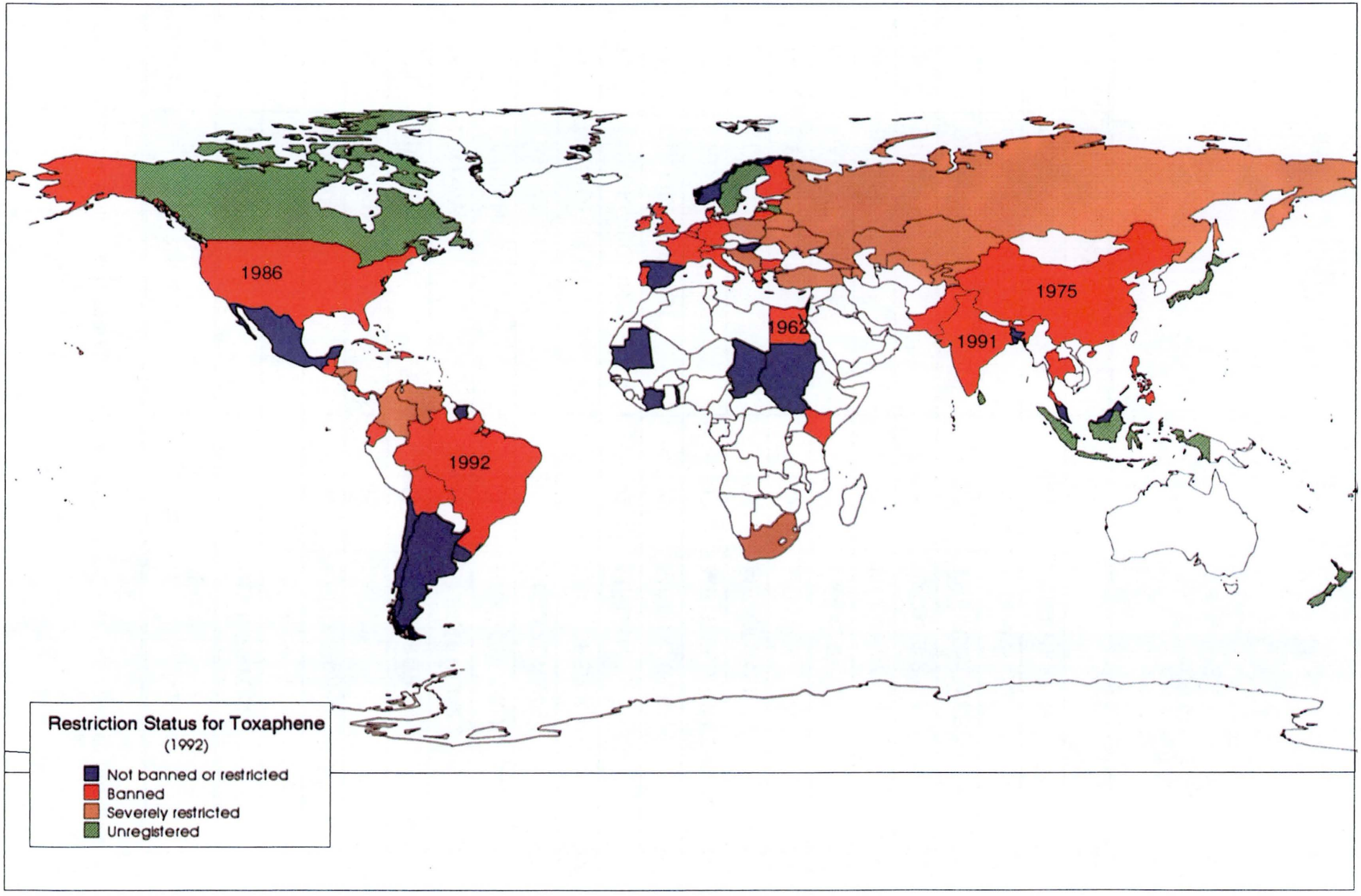


Figure 2.

Total Toxaphene Usage

(Accounted For)

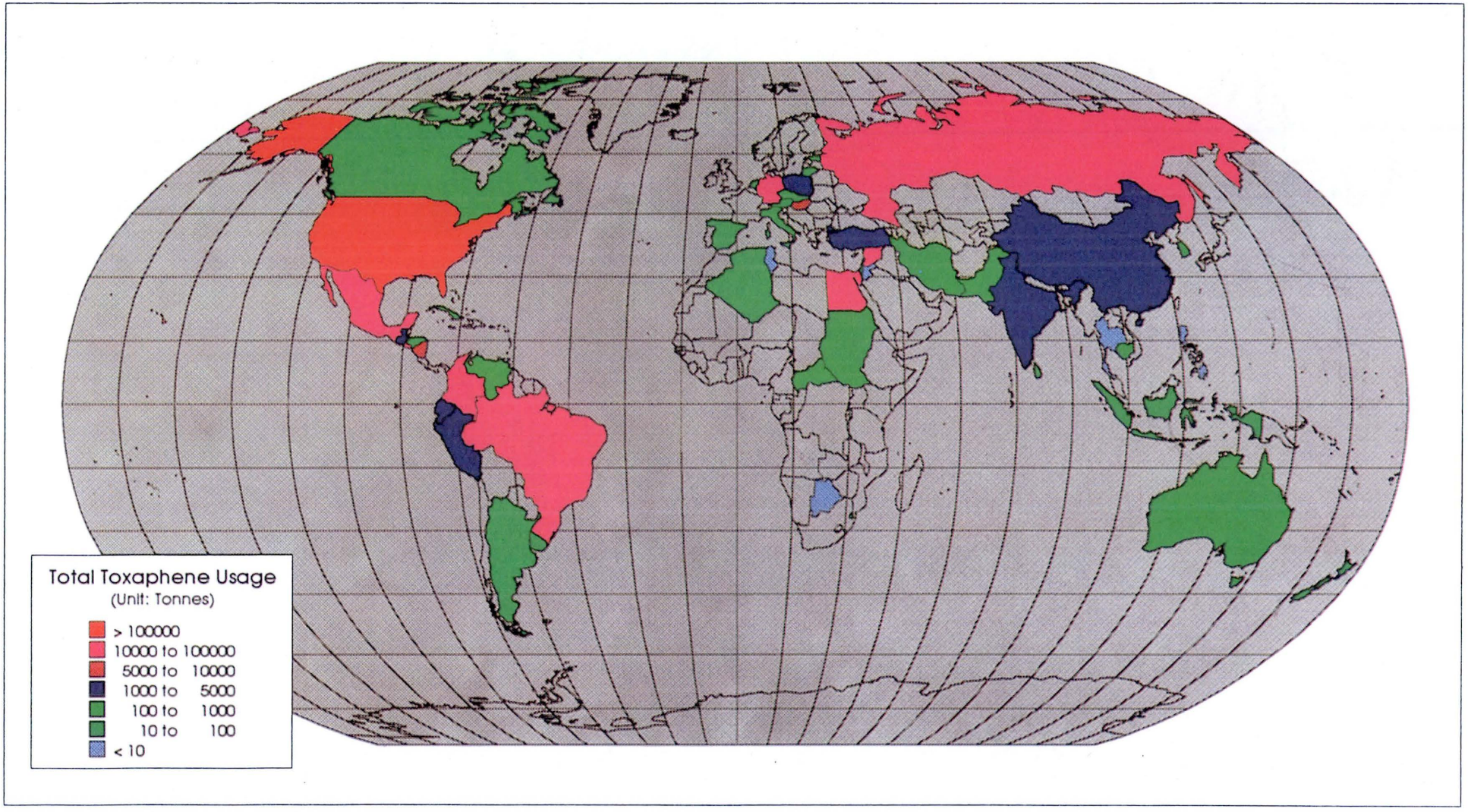
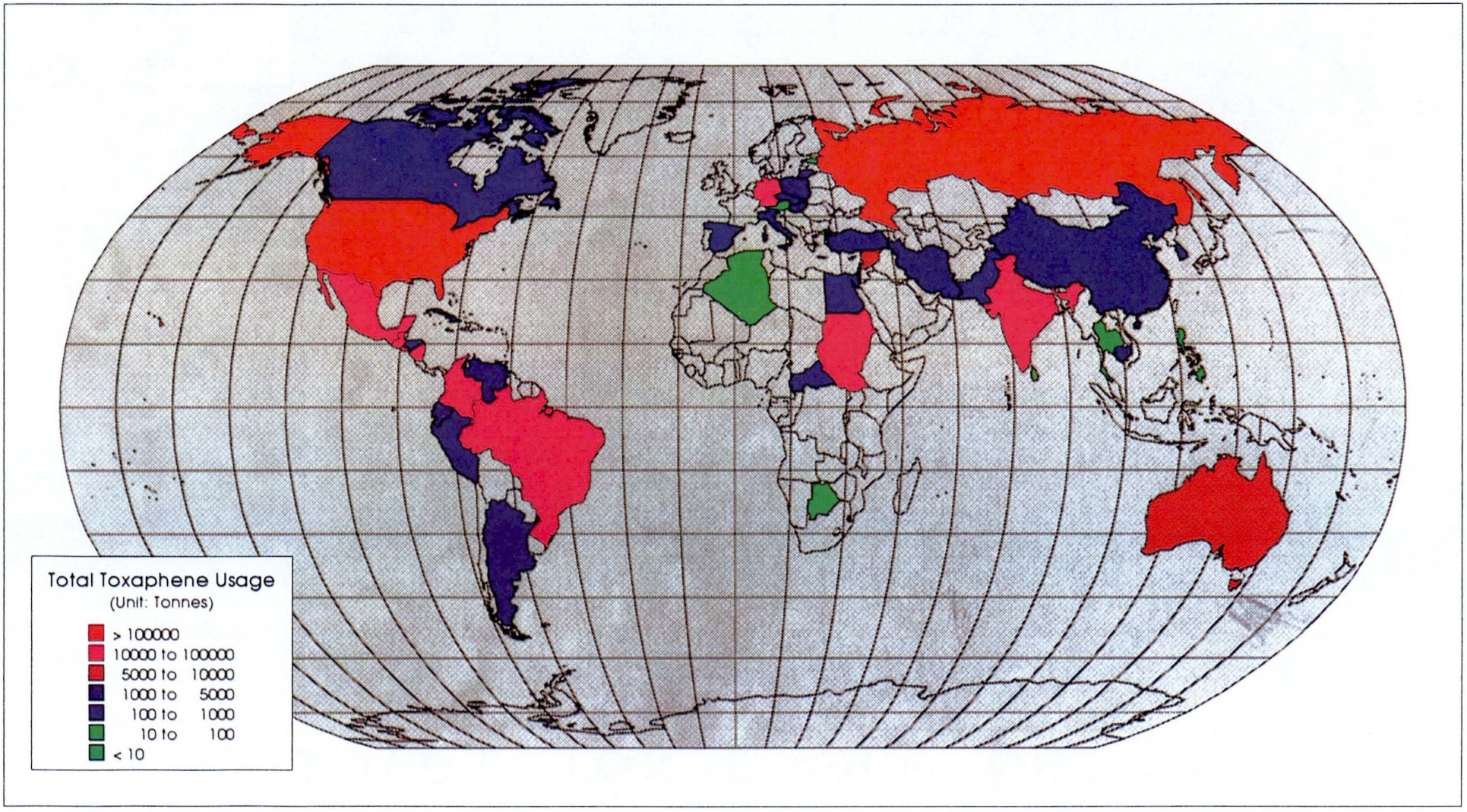


Figure 3.

Total Toxaphene Usage

(1970-1992, Interpolated)



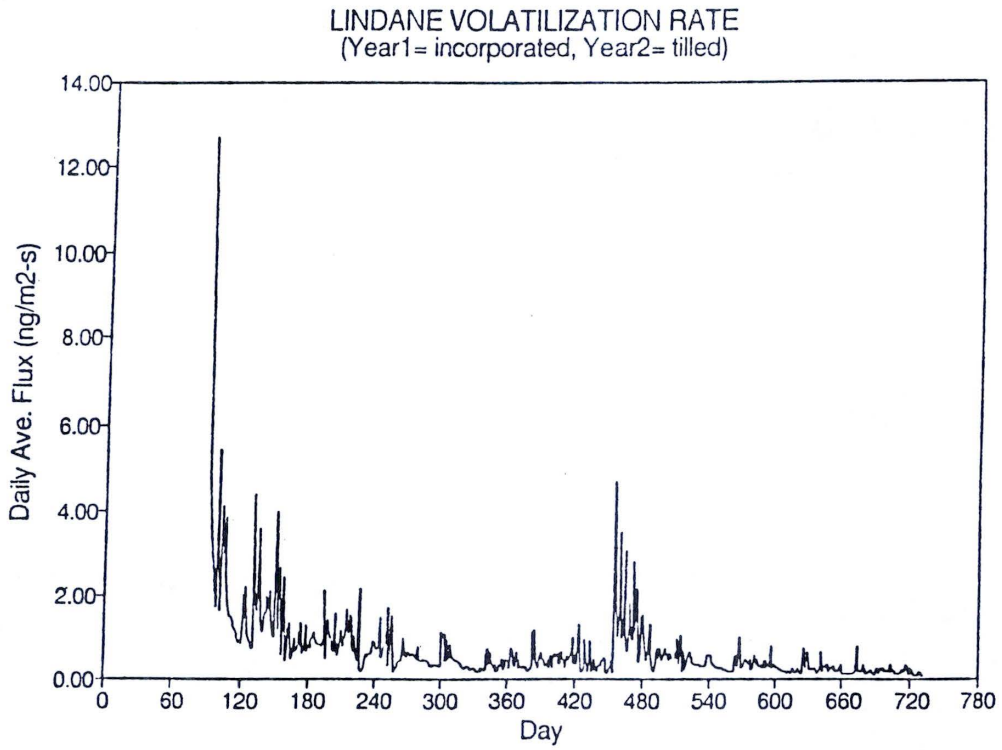


FIGURE 4. Temporal pattern of lindane emission rate following uniform incorporation to a depth of 10 cm. in soil on day 91 in year 1 and subsequent tilling on day 456 in year 2.

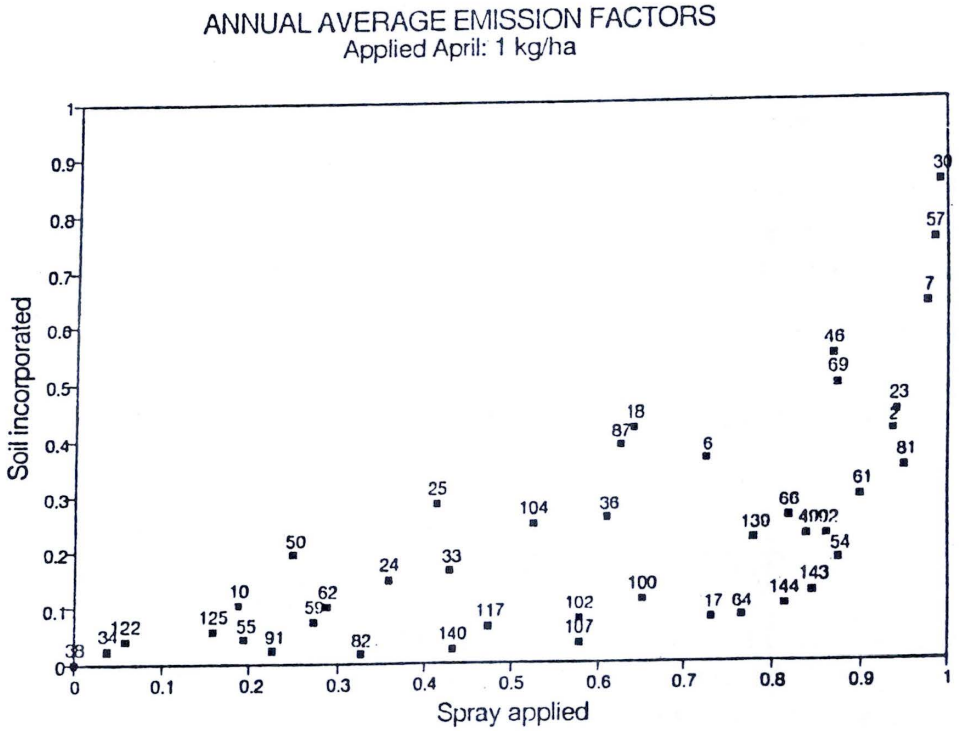


FIGURE 5. Annual average pesticide emission factors for Toronto meteorology: Comparison between factors for spray applied and soil incorporated pesticide (plotted numbers refer to the pesticide list in Table 1).

TRANSPORT OF PERSISTENT ORGANIC POLLUTANTS WITH SPECIAL EMPHASIS ON ARCTIC REGIONS

PROJECT LEADER: J. Pudykiewicz, Atmospheric Environment Service, Environment Canada, Downsview, Ontario

PROJECT TEAM: J. Pudykiewicz, A. Sirois, E. Voldner and T. Bidleman

OBJECTIVE

To carry out large scale modelling to simulate the atmospheric transport of persistent toxic chemicals to the Arctic. To develop source/receptor relationships where possible.

DESCRIPTION

Since many of the persistent toxic chemical species found in the Arctic ecosystem are not used in the north, it is postulated that other sources, perhaps global in extent, are having an impact on the Arctic through atmospheric transport. Measurements of levels of selected organochlorines and polycyclic aromatic hydrocarbons are being carried out under another project (Barrie this report). Also, identification of major sources of these chemicals on a global scale is being done (Voldner this report). A numerical model which simulates weather patterns that carry toxic chemicals will be used to link sources to Arctic ecosystems.

ACTIVITIES IN 1992/93

1. A global spectral weather model has been modified to handle chemical species. In addition, modifications have been made to better handle clouds and precipitation as well as to handle major natural sources of pollution, such as volcanoes. A paper summarizing this work is in preparation.

2. A sophisticated software package was purchased to aid in visualization of model output. It was implemented on our computer equipment and one of our staff established expertise in using it.

CONCLUSIONS AND FUTURE DIRECTIONS

A suitable model framework for Arctic applications has been established. Work in 1993/94 will focus on selecting and performing specific model applications. Funding will provide programming support to allow many model runs. This project is dependent on the 1993/94 project entitled "Modelling Global Scale Transport of HCHs: Review and Preparation of Supporting Data" for input data for the model.

LONG-RANGE TRANSPORT OF CONTAMINANTS TO THE CANADIAN BASIN

PROJECT LEADER/PROJECT TEAM: R.W. Macdonald, F.A. McLaughlin
Institute of Ocean Sciences, Sidney B.C.

OBJECTIVES

1. To measure organochlorines at a time series site in the Canada Basin to obtain the large-scale signature for the Arctic Ocean interior.

2. To collect a profile for dissolved organochlorines (e.g. HCH, Chlordane) to augment the Arctic Organochlorine database (mainly the Ice Island dataset).

3. To measure organochlorines in the Husky Lake - Liverpool Bay system to determine quantities and their movement through various compartments (water, zooplankton, fish, sediments).

DESCRIPTION

During 1992/93, the focus has been on two sites:

One site is a station in the Canada Basin (3300 m - see Station A, Fig. 1). Here we are establishing water-column contaminant profiles in the context of the regional oceanography. These data then lead naturally into calibrating box models of the upper Arctic Ocean. We have developed an oceanographic time series at this station since 1987. In 1992, we collected a profile for HCHs (and other contaminants) which can be compared with data from 1986 at the Ice Island (Hargrave *et al.* 1988). With future measurements at the Basin site these data can be used to determine trends with time for HCH and other contaminants. The contaminant dataset is considerably strengthened by supporting water property measurements which can be used to determine time scales for the water (e.g. tritium) and time scales of gas exchange with the atmosphere (freons). Further, we have developed water-mass analyses to determine sources for the water and how they distribute themselves within the Arctic Interior (e.g. surface water, water from the Bering Sea or Pacific Ocean, water from the Atlantic Ocean). Moored, sequential sediment traps have been collecting data on sedimenting particles, currents, and water-mass properties; these will give us estimates of the importance and timing of particle fluxes to the movement of sequestered contaminants.

The other site incorporates the Husky Lakes - Liverpool Bay estuary. From the point of view of its oceanography, this region has for the most part been ignored (Evans and Grainger 1980). Nevertheless, this estuary has many advantages as a site for a systems study focusing on contaminants. The system consists of several basins that connect to one another by narrow passages and ultimately connect to the Arctic Ocean through Liverpool Bay. As such, it lends itself to physical and geochemical modelling (e.g. Engqvist and Omstedt 1992). Up to the

present, we have not studied the transport of contaminants through an arctic estuary. This estuary would be an ideal location to do such a study. Due to the relatively large area of the drainage basin, it is likely that atmospherically transported contaminants will tend to be focused into this estuary which also provides a significant resource for native foods. It is worth noting that high concentrations of organochlorines were found in belugas trapped by ice in the inner lakes (Muir *et al.* 1990, 1992) although these levels may in part have been induced by starvation.

PRELIMINARY RESULTS

At the time of writing, the dataset is not complete and our interpretations are preliminary.

Canada Basin

We were able to recover the sequential sediment trap mooring placed at the Basin site in 1990, and we recovered the 1991 mooring (placed at Stn 44) and redeployed the mooring at the Basin site. These moorings have given us particle flux records for 2 years. We sampled the water for oceanographic parameters (e.g. salinity, nutrients, freons, tritium) and for contaminants (HCHs and other pesticides using liquid/liquid extraction). Additionally, two moorings with 5 Infiltrex or Seastar *in situ* pump samplers were deployed to sample the top 250 m of the water column. Figure 2 shows our preliminary data for total HCH at the site along with the Ice Island data taken from Hargrave *et al.* (1988, 1989). The total HCH profiles are similar at the two stations; our data support the Ice Island observations and confirm that the distribution of HCH in the Arctic Interior is spatially uniform. Our data suggest an increase in total HCH, especially deeper in the water column (100-200 m). Integrated burdens are slightly larger than at the Ice Island (880 $\mu g/m^2$ vs. 650 $\mu g/m^2$). If however, as suggested by Bidleman *et al.* (1989) (and see Hargrave *et al.* 1989), we adjust the in-situ pump data for about 70% recovery, the Ice Island integrated burden becomes 930 $\mu g/m^2$ which is not significantly different from our data. The question of trends for HCH is still open and requires further profiles, probably at about 5 year intervals.

Husky Lakes - Liverpool Bay

During April-May, 1992, we sampled this system from the ice for water, particulate, sediment, zooplankton, fish, and ice cores. Measurements were made for organochlorines (pesticides, PCB congeners, coplanar PCBs, toxaphene) and PAHs. The dataset is large and we have not had sufficient time to examine it carefully. Table 1 briefly summarizes concentration ranges for some key compound classes. As might be expected, the highest concentrations of organochlorines were found in fish tissue (trout). We collected only 2 samples; further samples are clearly required to determine whether there are species or locations that are particularly subject to contamination.

UTILIZATION OF RESULTS/DISCUSSION/CONCLUSIONS

We have found a discrepancy between *in situ* pump samples (XAD columns, back extracted) and bottle samples (liquid/liquid extraction); XAD concentrations are lower and sometimes not reproducible. Both methods have been and will continue to be used by investigators for the determination of HCHs. The advantage of the pumps is that by concentrating organochlorines from a large volume of water they allow additional pesticides to be determined. To measure trends and spatial distributions, it is critical that we ground truth the XAD resin columns *in situ* pumps with bottle samples (liquid/liquid extraction). In conjunction with Axys Group Ltd., we are now evaluating both methods in local waters.

Expected project completion date: March 31, 1997

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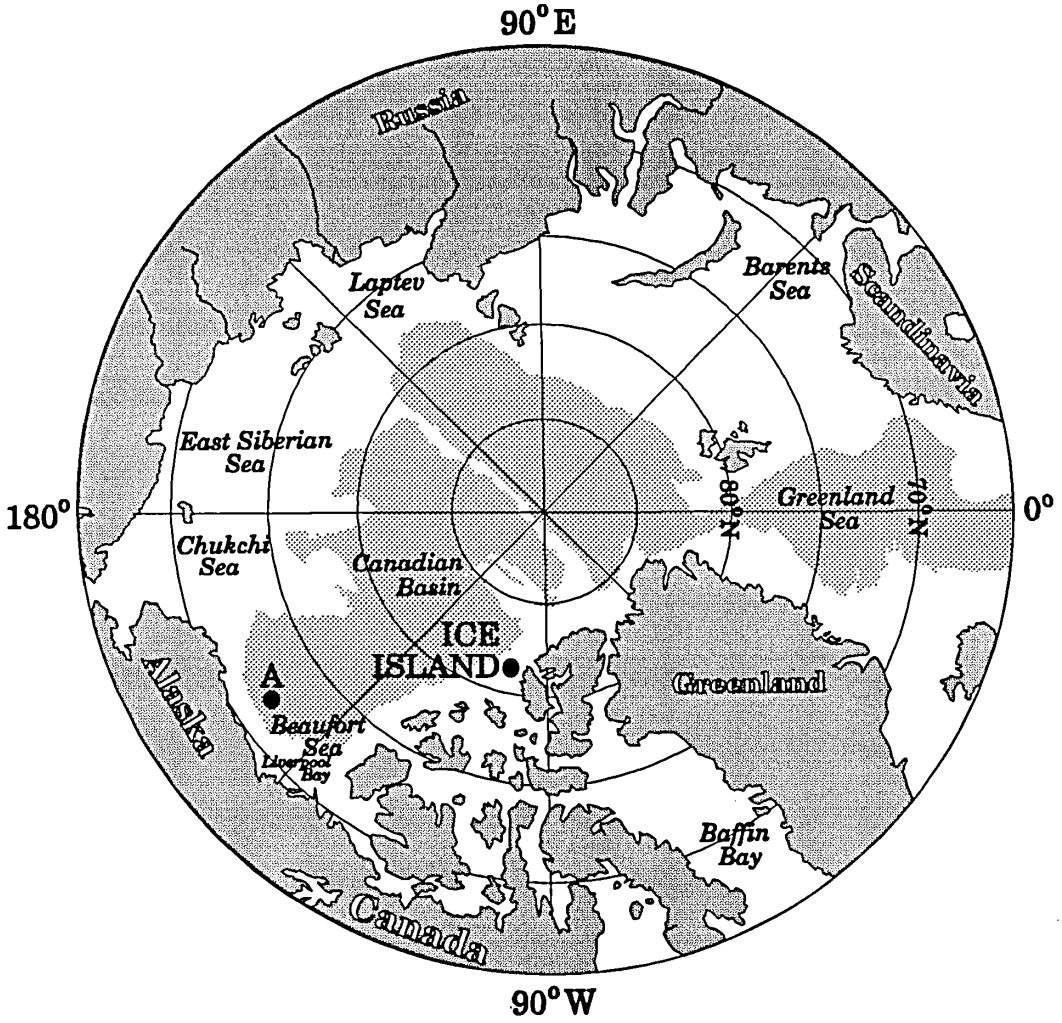

Figure 1. Circumpolar map indicating the locations of Ice Island and Station A.

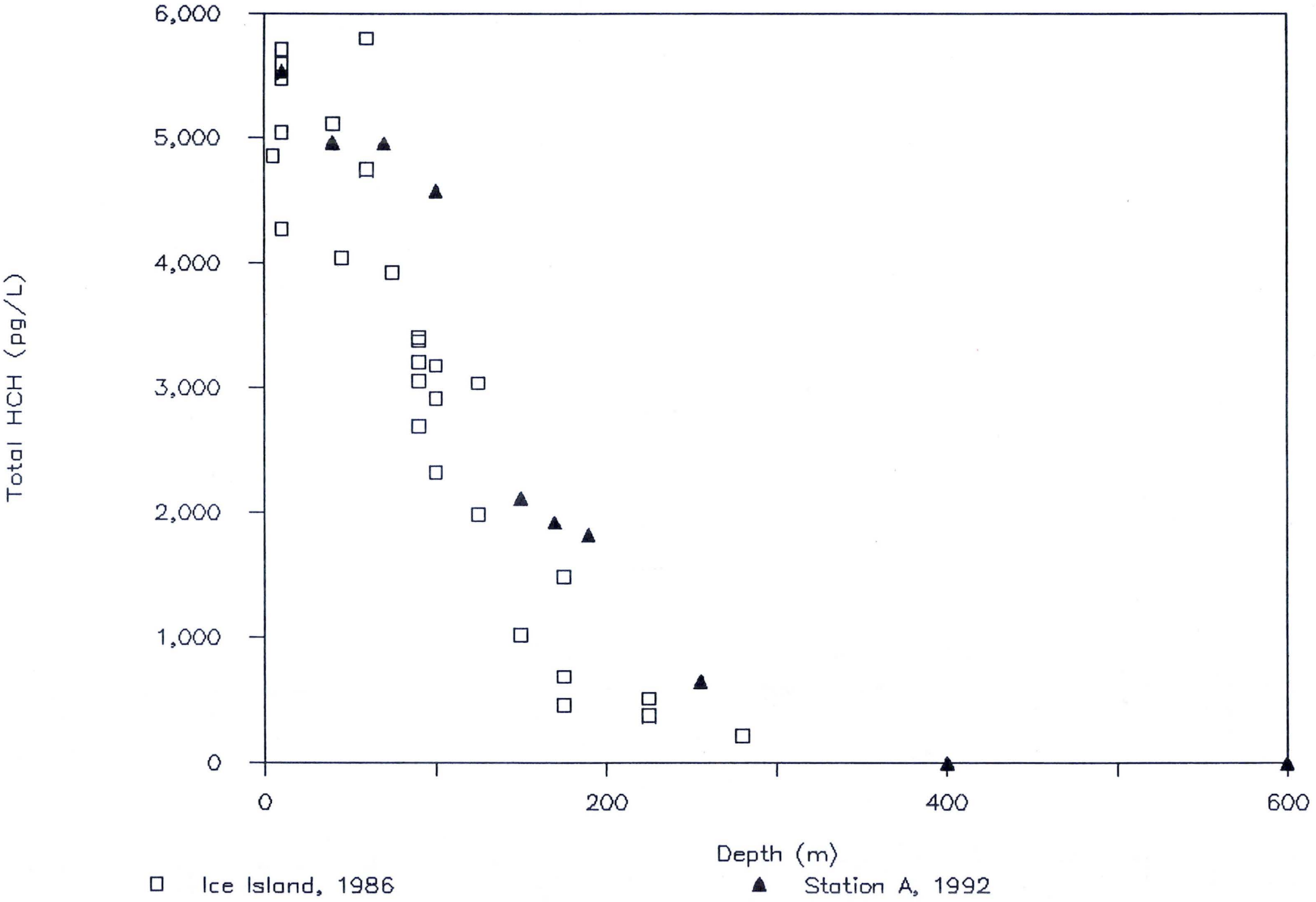

Figure 2. Total HCH concentrations in ocean water at various depths at Station A (this study) and at Ice Island (Hargrave *et al.* 1988, 1989).

Table 1. Organochlorine concentrations in water, particulate, sediment, zooplankton and fish collected from the Husky Lakes - Liverpool Bay system in April-May, 1992.

COMPOUND	DISSOLVED pg/l	PARTICULATE pg/l	SEDIMENT pg/g	TISSUE pg/g liver-tissue	ZOOPLANKTON pg/g
HCB	9-74	0.34-0.73	300-1100	2.3-5.5	1.9
a-HCH	39-909	0.38-2.57	130-1400	1400-2900	300
1254/1260	2-14	0.78-6.9	30-340	9200-24000	11000
TOXAPHENE	ND-70	ND	ND-1800	29000-72000	9300
105 2,3,3',4,4'-PCB	0.05-0.43	0.03-0.26	ND-3	120-250	92
153 2,2',4,4',5,5'-HCB	0.1-1.27	0.03-1.97	4.2-19	1300-3000	1400
138 2,2',3,4,4',5'HCB	0.11-1.27	0.07-0.62	2.7-31	840-2200	730
77 3,3',4,4'TCB				1.1-5.6	1.1
126 3,3',4,4',5PCB				1.1-2.7	1
169 3,3',4,4',5,5'HCB				4.1-8.7	ND
TCDD			1.4-38	ND	ND
O8CDD			ND-5.4	NDR	ND
TCDF			ND-1.1	ND-0.1	1

ORGANOCHLORINES AND POLYCYCLIC AROMATIC HYDROCARBONS IN THE ARCTIC ATMOSPHERE

PROJECT LEADER: L.A. Barrie, Atmospheric Environment Service, Environment Canada, Downsview, Ontario

PROJECT TEAM: L. Barrie, R. Bailey, B. Billeck, T. Bidleman, K. Brice, P. Fellin, B. Grift, L. Lockhart, J. Kovalick, D. Muir and D. Toom

OBJECTIVES

To measure the occurrence of selected organochlorines and polycyclic aromatic hydrocarbon compounds in the Arctic atmosphere for a period of two years thereby, providing insight into environmental transport, removal, transformation and surface exchange processes, as well as data for the development of realistic environmental pathways models.

INTRODUCTION

As a result of a comprehensive review of the sources, occurrence and pathways of northern contaminants (Barrie *et al.* 1992), it was apparent that little information existed on the occurrence in the atmosphere of organochlorines (OCs) and polycyclic aromatic hydrocarbons (PAHs). Such information is necessary if an understanding of their sources, pathways and impacts on the Arctic environment is to be achieved.

There were only a few observations of the atmospheric abundance of these compounds attained through short-term (6-8 weeks) intensive field studies scattered over a few years and locations.

ACTIVITIES IN 1991/92

Upon receipt of funds in August 1991, a contract was let to Concord Environmental in Toronto to build on their research, supported by the Atmospheric Environment Service in 1990/91 that produced an optimal instrumental package for air sampling in the Arctic on a routine basis. A high volume sampling system had been developed that had a 10 μm diameter particle cutoff, the capability of dividing the compounds of interest sampled at 1 cubic meter per minute into particulate and gaseous fractions and depositing them on pre-cleaned filters that can subsequently be chemically analyzed in a central laboratory. What remained to be done was the preparation of such a system for the Arctic, the development of sample handling and collection procedures, analytical testing of the sample preparation procedure and preparation of a suitable field measurements facility and protocol. By early 1992, everything was prepared for the initiation of a sample test program. A laboratory facility at Alert on northern Ellesmere Island was prepared for the installation of the sampler on January 20, 1992. Since that time, weekly samples have been taken and shipped to Concorde Scientific in Toronto where they have been extracted into hexane solvent. The extracts have been analyzed for at least 18 PAHs and 90 OCs

by D. Muir's laboratory at the Freshwater Institute of Canada. K. Brice of the Atmospheric Environment Service will provide an independent check on the analysis through exchange of split extracts.

ACTIVITIES IN 1992/93

Three major accomplishments in this year were:

1. The successful operation of routine air sampling at Alert (Fig. 1) for the whole year that was marked by the presentation of a paper on "Atmospheric Toxaphene Observations at Alert" at an international workshop on Analytical and Environmental Chemistry of Toxaphene, held in Burlington, Ontario in February 1993 by L. Barrie on behalf of the group. This paper will be submitted to the journal Chemosphere as part of the workshop proceedings. Figure 2a shows preliminary results for toxaphene air concentrations in the first six months of 1992 and compares them to previous observations made in 1988 at Alert. Figure 2b shows a diagnostic parameter, the fraction of total toxaphene on the second of two polyurethane foam plugs that qualifies the integrity of a sample.

   Routine sampling requires considerable effort in sample preparation, shipping to and from the station, sample extraction, and sample information accounting. A documented sampling protocol was prepared and a sample tracking system instituted that is illustrated in Figure 3. In addition, chemical analysis of approximately 200 extracts per year for 14 PAHs and 60 OCs is very difficult. This is being capably accomplished by the analytical laboratories at the Freshwater Institute of Canada in Winnipeg.

2. The selection, installation and successful operation to year end of a second site at Tagish, Yukon (Fig. 1) by Concord Environmental and local Yukon authorities led by M. Palmer of the Yukon Contaminants Committee. Sampling began in December 1992. Analysis results will be available in the next fiscal year.

3. The instrumentation of a third site in the Russian Arctic at Dunay Island (Fig. 1) by P. Fellin of Concord Environmental in cooperation with L. Barrie and Russian scientists in February 1992. This part was largely funded by External Affairs Canada.

   Both Canadian sites are co-located with routine weekly snowfall measurements of D. Gregor of the National Water Research Institute in Burlington, Ontario.

CONCLUSIONS AND FUTURE DIRECTIONS

A robust Arctic air sampler for organochlorines and polycyclic aromatic hydrocarbons has been developed and demonstrated to function under Arctic conditions. The first routine measurements in the High Arctic at Alert were conducted throughout 1992 and preliminary results on toxaphene presented.

Two other sites, one in the Yukon and one in the Russian Arctic, were established. A third site in the south eastern Canadian Arctic will be selected and established in cooperation with local authorities in 1993/94. Co-located with this will be a snow sampler operated by D. Gregor.

Reports on the results for the first year of sampling at Alert 1992 and for the first 4 months of sampling at Tagish will be ready in draft by September 1993 and will be reported at an International Conference on Northern Contaminants in Iceland October 1993. A documented network sampling and analysis protocol will also be available in September 1993 now that procedures have stabilized and are practised at several sites.

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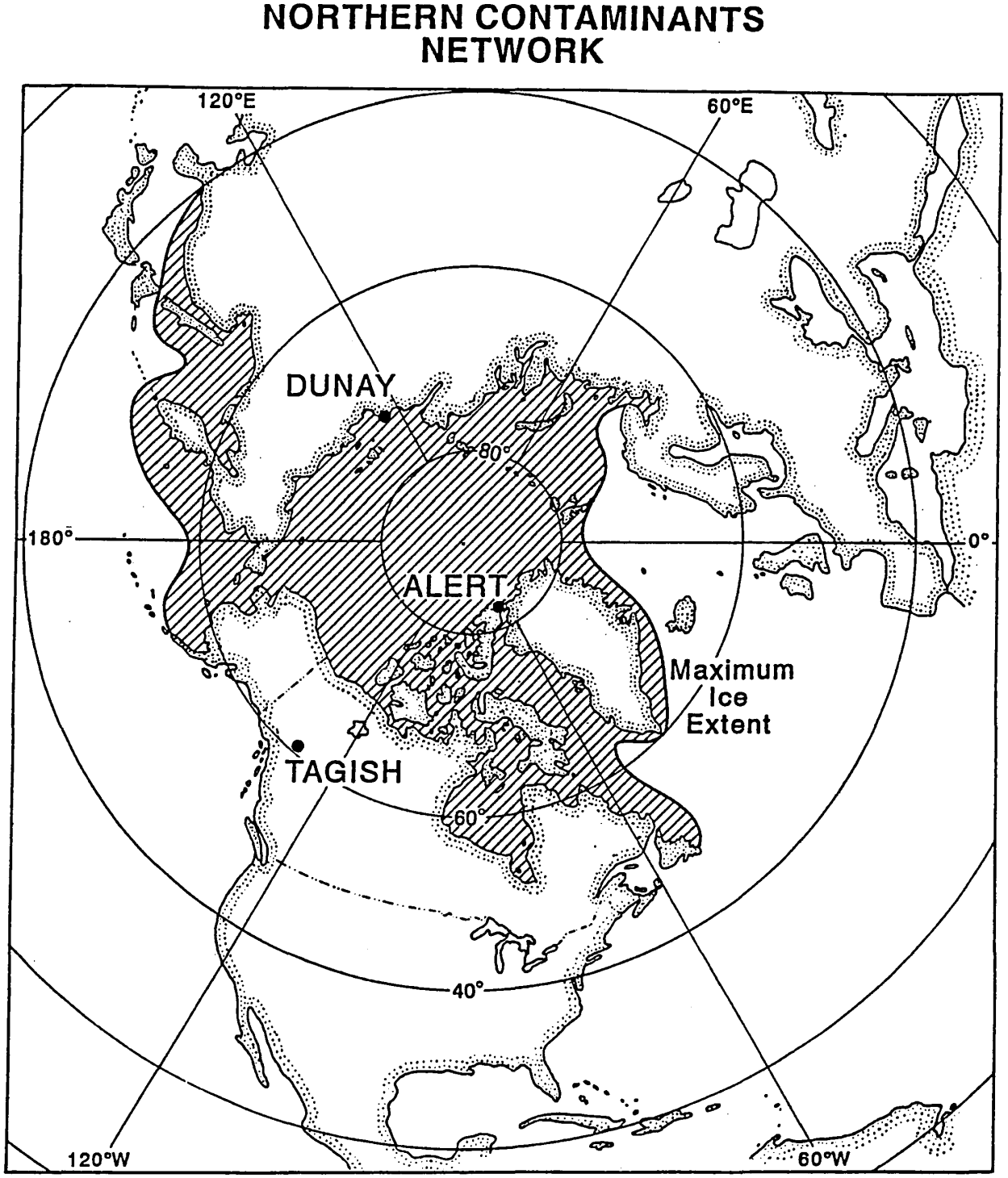


Figure 1. Location of atmospheric toxics samplers in operation in May 1993 in the Arctic under the Northern Contaminants Program.

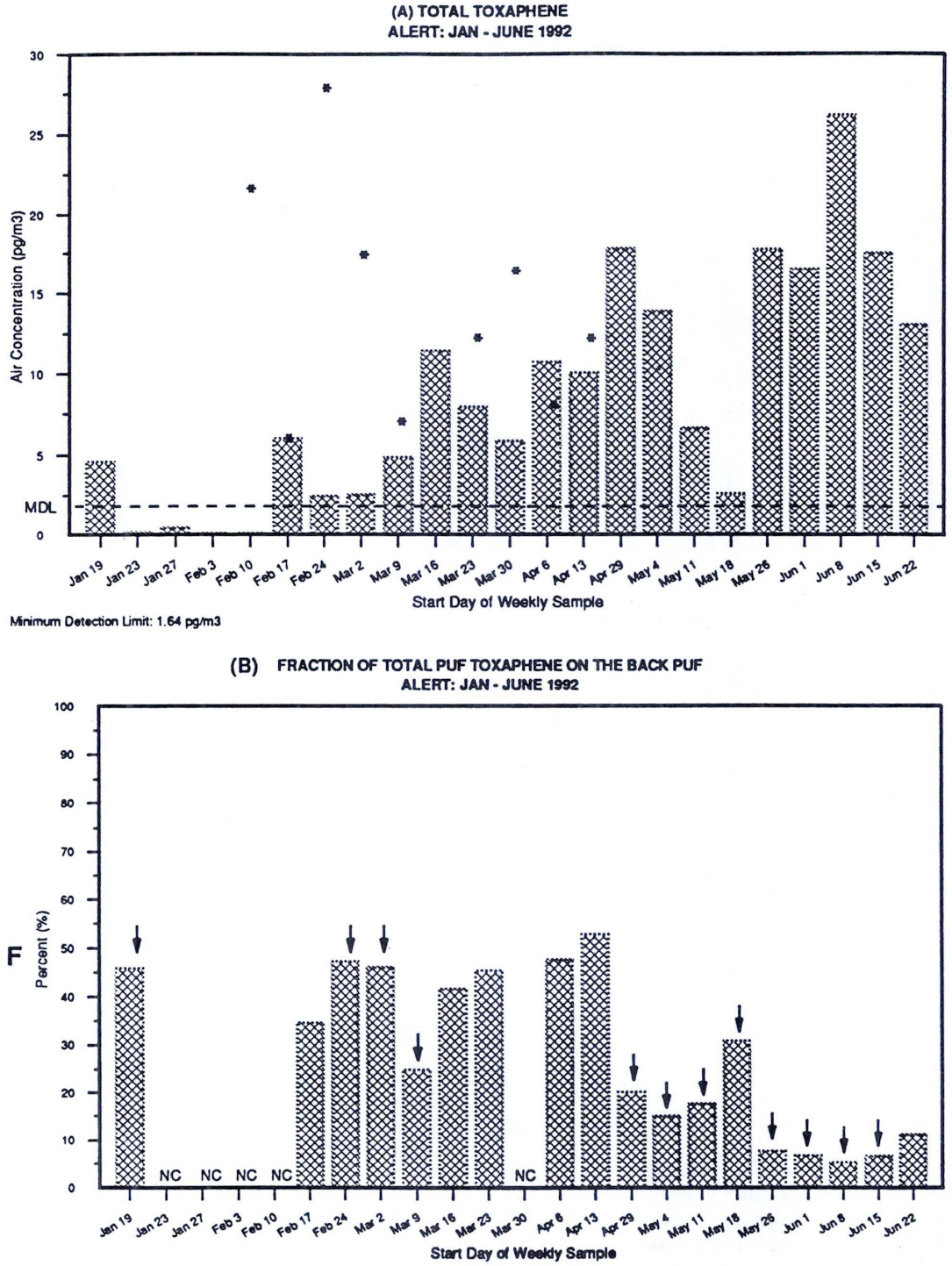


Figure 2. Preliminary results of total toxaphene concentrations in at Alert in the first half of 1992.

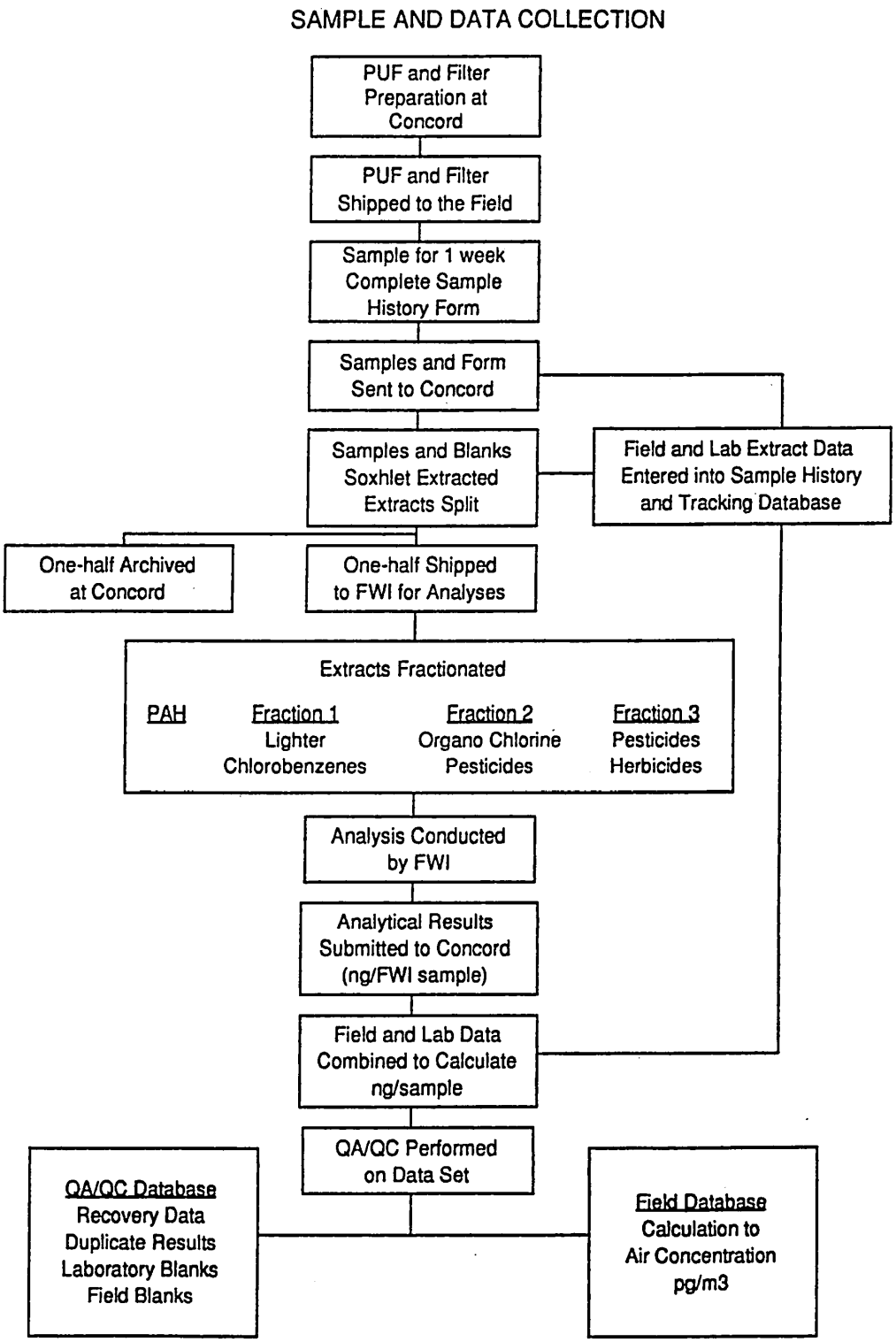


Figure 3. Flow chart of the sample and data collection process in the atmospheric contaminants monitoring program for PAHs and organochlorines.

CANADIAN GLOBAL EMISSIONS FEASIBILITY STUDY - PRESENTATION OF CONCEPT TO CAMRAQ

PROJECT LEADER: A. McMillan, Atmospheric Environment Service, Environment Canada, Downsview, Ontario

PROJECT TEAM: A. McMillan, T. Sholtz and E. Voldner

OBJECTIVE

To promote the concept of a Canadian Global Emissions Inventory Center.

DESCRIPTION

This project is a "project of opportunity" brought forward for modest funding in the fall of 1992. It is part of a much larger effort.

ACTIVITIES IN 1992/93

A contract was given to T. Sholtz of ORTECH to prepare a presentation of the concept of a Canadian Global Emissions Inventory Center and to present it to the Consortium on Advanced Modelling of Regional Air Quality (CAMRAQ).

The presentation was very well received by the members of CAMRAQ including the United States Environmental Protection Agency and the Electrical Power Research Institute (EPRI) in the United States. Dr. Sholtz followed up with a letter to the members of CAMRAQ asking for their comments. A response letter was received from A. Hansen of EPRI who indicated that EPRI is interested in further supporting such an initiative once it is more clearly defined.

CONCLUSION AND FUTURE DIRECTIONS

The response to the concept of a Canadian Global Emissions Inventory has been overwhelmingly positive. In 1993/94, the proposal will be refined and funding partners pursued.

CURRENT CONTAMINANT DEPOSITION MEASUREMENTS IN PRECIPITATION

PROJECT LEADER: D. J. Gregor, National Water Research Institute (NWRI)
Environment Canada, Burlington, Ontario

PROJECT TEAM: S. McDonald, D. Ross, C. Brumwell, NWT Programs, Inland Waters Directorate (IWD), Yellowknife
G. Whitley, M. Palmer, Indian and Northern Affairs (INAC), Whitehorse
Atmospheric Environment Service (Winnipeg, Mould Bay, Eureka, Alert, Downsview, Resolute Bay)
M. Alaee. A. Peters, C. Teixeira, N. Jones, NWRI, Environment Canada

OBJECTIVES

1. To quantify annual deposition of trace organic contaminants to the Canadian north through the establishment of a multi-station sampling network.

2. To develop methods capable of assessing seasonal and event deposition of contaminants at selected intensive sampling sites co-located with several of the annual deposition stations.

3. To determine spatial variability in contaminant deposition and assess short-term trends.

DESCRIPTION

The network of annual snowpack sampling locations was initiated in part in 1990 and continued in 1991 and 1992 (see Table 1). Samples are collected annually through arrangements with Inland Waters Directorate (NWT), Indian and Northern Affairs Canada (NWT and Yukon), Government of NWT and Atmospheric Environment Service, Environment Canada (selected weather stations). Sample collection begins in March of each year in the Yukon and southern Mackenzie and gradually proceeds to the east and north with completion in the High Arctic in May. Samples are carefully collected into specially constructed aluminum containers and sealed until melted and extracted in the north. While samples are collected by a number of people, all samples are extracted and analysed by NWRI. Detailed field sheets are filled out at each site. A rigid protocol for preparation of sample containers and sampling equipment in the field has been developed. A total of 10 spikes are added to each sample beginning with a field spike and continuing through all aspects of extraction and clean-up in order to measure any losses in the sample between the time of collection and analyses.

In addition to the annual snowpack samples, large volume snow collectors were operated at Eureka and Mould Bay from October, 1990 to March, 1991 and again during the winter of 1991/92. A third snow collector, complete with a remote automated weather station was installed near Whitehorse in December, 1991. The fourth and fifth snow collectors located at Whitepass

on the B.C./Alaska border and at Tagish in the Yukon were installed in the fall of 1992 for operation during the 1992/93 winter. All bulk snow samples and collector samples for the winter of 1991/92 have been analyzed for OCs and PCBs and extracts have been retained for PAH analysis.

The broad spatial coverage that we are achieving would not be possible without the extensive support of IWD-NWT Programs office in Yellowknife, INAC district offices and the ongoing support of Polar Continental Shelf Project (PCSP), Resolute Bay.

ACTIVITIES DURING 1992/93

All samples have been analysed for OCs and PCBs and are awaiting PAH analyses. Samples from the five snow collectors for the winter of 1992/93 are presently being processed. Sampling of bulk snow samples in the spring of 1992 was very successful except in the Yukon where an unseasonably warm March resulted in snowmelt prior to the planned sampling time.

RESULTS

The emphasis here will be on the snow collectors as a thorough review of the snowpack sampling program is planned for later this year. Daily deposition data for HCHs and PCBs for the Mould Bay collector were reported last year. However, the 1991/92 season is the second full season for the collector at this site. The concentration data for Mould Bay during this season are shown for PCB congener groups and for the HCHs in Figure 1. The daily depositions have not been calculated as yet as the daily precipitation data have not yet been received. This information is necessary to first of all correct the snow collector for its "catch" efficiency relative to the standard AES Nipher gauge and second, to determine the actual daily flux.

These concentration data for Mould Bay nevertheless show several things of interest. First, as noted last year, PCB is dominated by the lower chlorinated compounds (generally fewer than seven or eight chlorines) with the more chlorinated PCBs tending to occur later in the season. This is likely explained by the fact that the heavier compounds are more probably particle associated and therefore are deposited closer to the source. Second, peak concentrations occurred in late January and February which seems to coincide with the pattern of the previous year. There is evidence from the HCH data that concentrations for these compounds also peak in mid-winter but perhaps a little earlier than the PCBs. Throughout most of the year the α:γ ratio is about 2:1 except for the last week of October, 1991 when γ-HCH was very high. This value is being rechecked.

This is really the first year of data for Eureka as limited snowfall in the previous year limited the collection program to only a few samples (Fig. 2). PCB concentrations at Eureka are less variable than at Mould Bay (note the difference in scale) especially during the late January/February time period. Nonetheless, like Mould Bay, the lower chlorinated PCBs are dominant with a tendency for the higher chlorinated PCBs to occur only in the latter part of the season. HCH concentrations are generally comparable between the two sites but the ratio of α:γ

tended to be higher. There is some evidence that γ-HCH is likely to be more common early in the season. At both sites, β-HCH contributes only a small component to Σ-HCH.

Other compounds are also present in the fresh snow but at much lower concentrations than the HCHs and PCBs. These compounds include tri-, tetra-, penta- and hexachlorobenzenes; trans-nonachlor, heptachlor epoxide, chlordane, dieldrin, endosulfan and low levels of DDT and metabolites.

UTILIZATION OF RESULTS

The large volume snow collector data will serve to evaluate the ability of the snowpack samples to characterize annual deposition to arctic sites. As well, timing of deposition of contaminants and thus linkages to major air masses and source trajectories may be possible. While the design and efficiency of the collectors is still being assessed, it is possible that these collectors will be accepted in other areas of the Arctic with low snowfall and thus can be employed as part of the circumpolar Arctic Monitoring and Assessment Programme. Snowfall collectors for high snowfall areas are also being tested at the northern British Columbia/Alaska site. These data will also be essential in calibrating and verifying atmospheric deposition models under development.

Table 1. List of stations sampled in NWT and Yukon snow survey.

	1990	1991	1992	1993
Western and Central Arctic				
Peel River above Fort Mcpherson (67° 13'N 134° 57'W)	X		X	#
Nahanni River near Nahanni Butte	X		X	#
Great Bear River (65° 8'N 123°31'W)	X			
Great Bear Lake at Hornby Bay (66° 36'N 117° 37'W)		X	X	#
Ellice River near mouth (67° 42.5'N 104° 8.5'W)	X	X	X	#
Akasta River	X	X	X	#
Thonokied River (WRB site discontinued)			X	
Cameron River below Reid Lake (62° 29'N 113° 31'W)	X	X	X	#
Kakisa River	X		X	#
Mackenzie River near Fort Good Hope		X	X	#
Snare Rapids				#C
Keewatin Arctic				
Baker Lake (VOR Lake) (64° 18'N 96° 5'W)	X	X	X	#C
Yathkyed Lake (62° 42'N 98° 18'W)]	X		X	#
Hayes River (67° 32'N 94° 5'W)	X	X	X	#
Back River below Deep Rose Lake (66° 5'N 96° 30'W)	X		X	#
Brown River (66° 2'N 91° 50'W)	X		X	#
Thelon River above Thelon Bluffs (64° 32'N 101° 24'W)	X	X	X	#
Lorillard River (64° 15'N 90° 26'W)		X	X	#
Seal Hole			X	#
High Arctic				
Stanwell Fletcher Lake, Somerset Island	X	X	X	#
Lake Hazen, Ellesmere Island		X		
Agassiz Ice Cap, Ellesmere Island (80° 40'N 73° 30'W)	X	X	X	#
Unnamed lake near Resolute Bay (74° 42'N 94° 54'W)		X	X[1]	
Mould Bay, Prince Patrick Island (76°15'N 119° 16'W)	XC*	XC	XC	#C
Eureka, Ellesmere Island (80°00'N 86°36'W)	XC	XC	#C	#
Alert, Ellesmere Island	X		C	#C
Lady Melville Lake (68°38'N 92°30'W)	X	X	X	#
Chartrand Lake (near Spence Bay)	X	X	X	#
Penny Ice Cap, Baffin Island			X	#
Iqaluit				#C
Yukon				
(see parallel project submitted by Yukon Contaminants Committee)				

* Indicates that a large volume snow collector was operated in addition to the collection of snowpack samples.

Proposed for 1993 along with Yukon sites proposed in parallel study.

[1] This site has been replaced by the Amituk Lake site (about 50 km to the east) which is reported as part of that study.

Note: a number of sites were missed in 1991 due to a lack of funds at the end of FY 1990/91 (sampling had to be done in March in the west) as well as to weather problems in the Keewatin. Also, stations such as Lake Hazen and Alert are collected only as opportunities permit due to the high cost.

Figure 1a. Time series of α-, β- and γ-HCH concentrations in snow from the Mould Bay snow collector from October 1, 1991 to March 2, 1992.

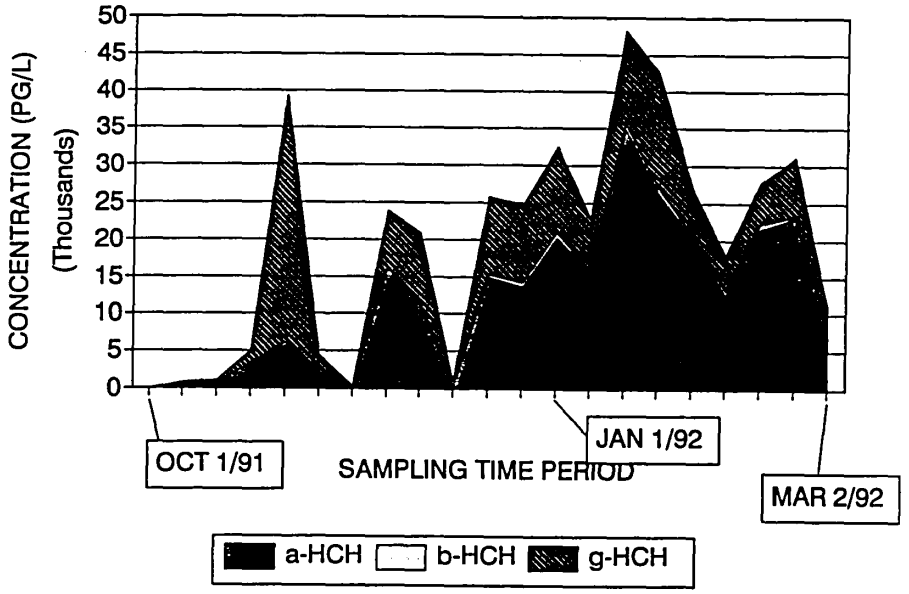


Figure 1b. Time series of groups of PCB congener concentrations in snow from the Mould Bay snow collector from October 1, 1991 to March 2, 1992.

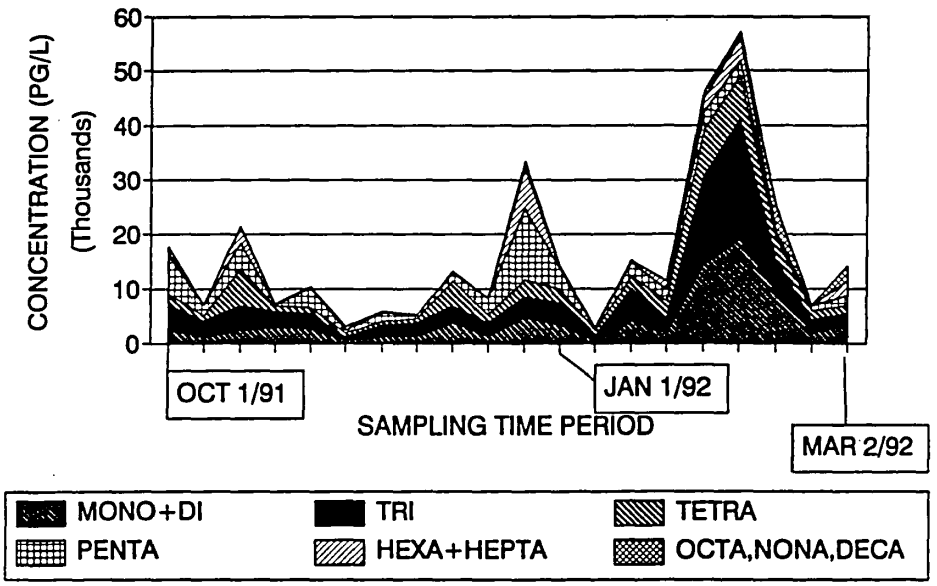


Figure 2a. Time series of α-, β- and γ-HCH concentrations in snow from the Eureka snow collector from October 1, 1991 to March 2, 1992.

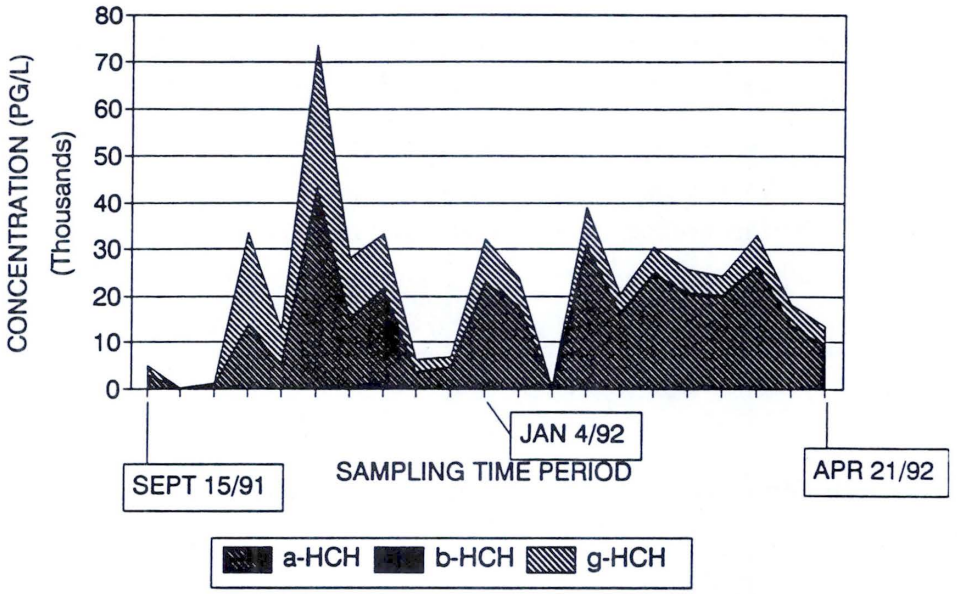
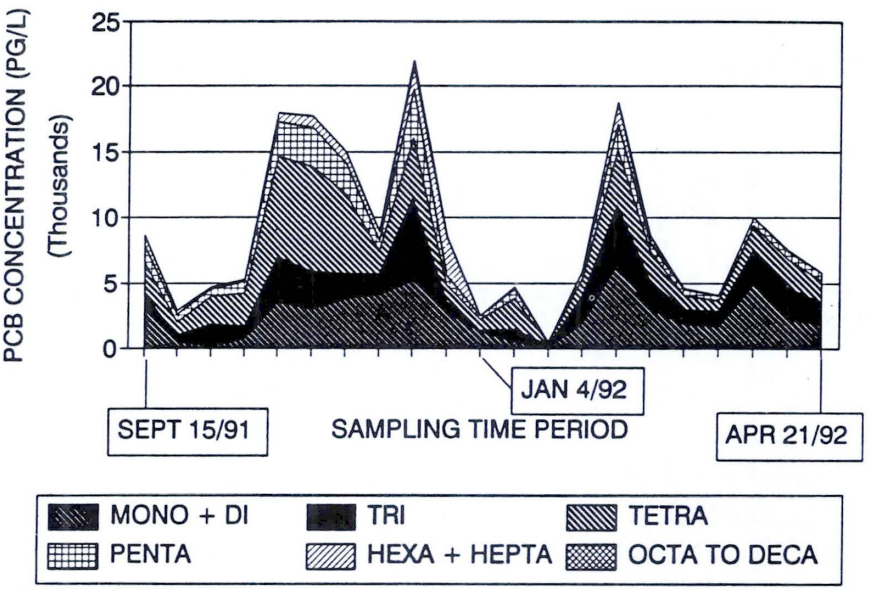


Figure 2b. Time series of groups of PCB congener concentrations in snow from the Eureka snow collector from October 1, 1991 to March 2, 1992.



CONTAMINANT DEPOSITION PATTERNS WITHIN THE YUKON

PROJECT LEADER: Yukon Technical Committee on Contaminants in Northern Ecosystems and Native Diets (Contact: Mark Palmer, Chair)

PROJECT TEAM: 1) D. Gregor, National Water Research Institute, Environment Canada
2) Indian and Northern Affairs Canada
3) Environmental Protection, Environment Canada

OBJECTIVES

Short-term:

1. To determine the deposition of organochlorine contaminants into the Yukon through the establishment of a comprehensive snow monitoring network.

2. To determine the difference in contaminant loadings from the different air masses influencing the Yukon.

3. To determine spatial variability in contaminant loadings and assess short term trends.

Long-term:

1. To establish an ongoing snow chemistry monitoring program to assess contaminant loadings in the Yukon.

2. To establish a snow chemistry database for use in determining long-term trends and for estimating contaminant loadings.

DESCRIPTION/RESULTS

Recent burbot liver results, as well as lake trout muscle results from several lakes located within the Yukon indicate detectable levels of organochlorine pesticides. In response to the elevated toxaphene levels, Health and Welfare issued a public health advisory for Lake Laberge recommending that consumption of lake trout flesh be limited and that no burbot livers be consumed. This information has all but shut down the commercial fishery in the area.

Varying levels of organochlorine contaminants have been found in fish collected throughout the Yukon. The source and pathway of these contaminants is unclear at the present time. The establishment of a snow quality monitoring network will assist in determining pathways and loading into the Yukon. It will also assist in determining the varied effects of the different air masses which influence the Yukon.

In cooperation with D. Gregor, NWRI, the Yukon snow monitoring network was expanded from one full time monitoring site and one snowpack to include three full time monitoring sites and 5 snowpack sampling sites. The locations of the sites are indicated in Figure 1.

The full time snow collectors were visited weekly during the winter months. The samples were extracted at Yukon College, Whitehorse and the extracts were sent to the NWRI laboratory in Burlington for analysis. Results from this year's sampling program have not been synthesized.

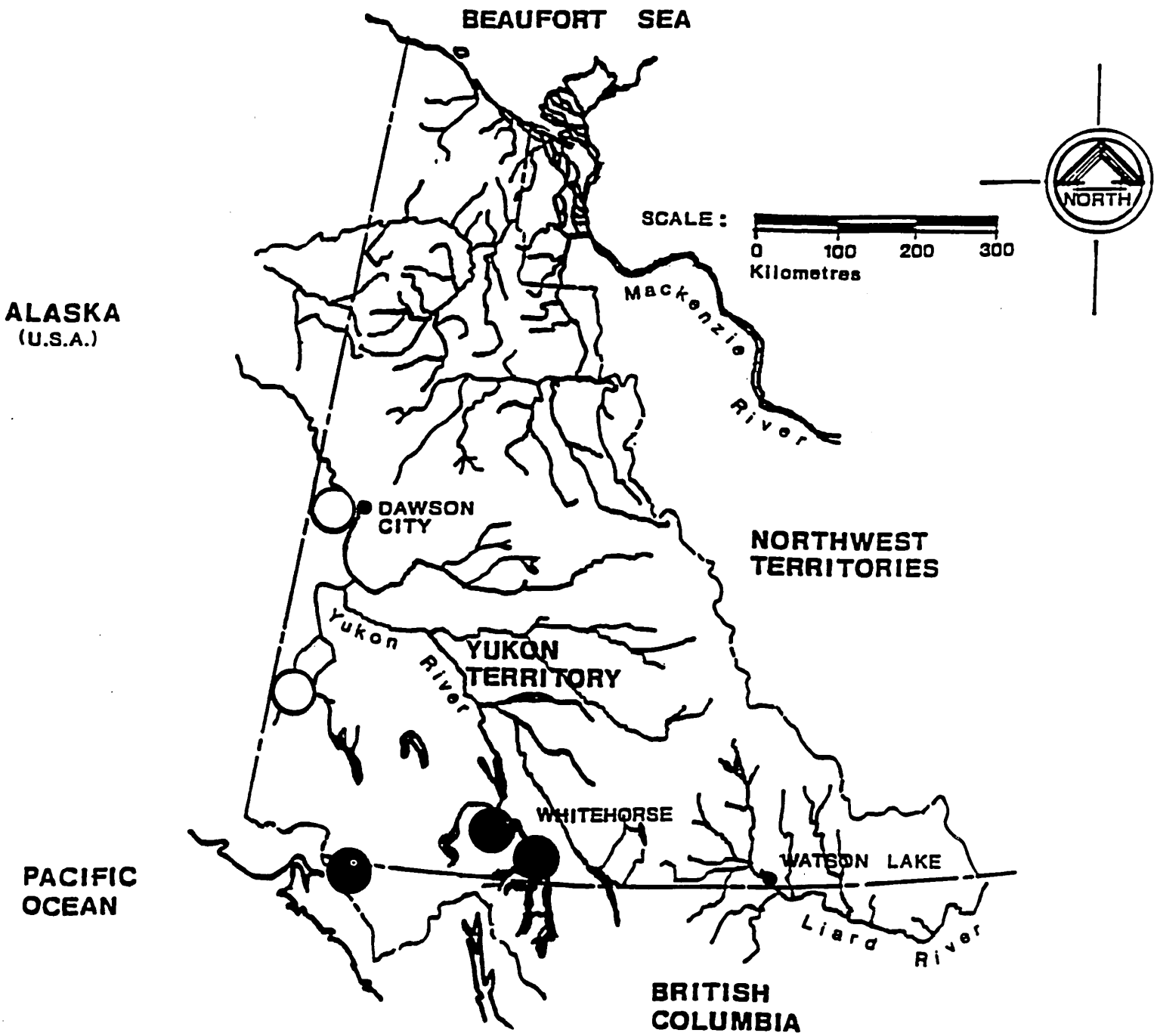
UTILIZATION OF RESULTS

Results from the snow quality network will be used to characterise the current deposition of contaminants into the Yukon. In addition, it is hoped that back trajectories of major air masses will be used to identify the sources of some of the contaminants transported to the north and as a result assist in explaining the elevated levels of contaminants detected in fish from selected Yukon lakes.

The large volume snow collectors will be used to evaluate the effectiveness of using the snowpack samples to characterize the annual deposition of contaminants into the Yukon.

Expected project completion date: March 31, 1997; data reports will be produced annually.

Figure 1. Snow sampling sites in Yukon Territory 1992/93.



● Large Volume Snow Collector Site/ Snowpack Sampling Site

○ Snowpack Sampling Site

THE HISTORICAL RECORD OF PERSISTENT ORGANIC POLLUTANTS AND TRACE METALS IN GLACIAL SNOW AND ICE

PROJECT LEADER: D. J. Gregor, National Water Research Institute, Environment Canada, Burlington, Ontario

PROJECT TEAM: R. M. Koerner, Energy Mines and Resources, Ottawa
A. Peters, M. Alaee, N. Jones, C. Teixeira, J. Nriagu and G. Lawson, National Water Research Institute, Environment Canada
N. Doubleday, Department of Biology, Queen's University, Kingston, Ontario

OBJECTIVES

Short-term:

1. To quantify annual deposition and historical residue trends of trace organic contaminants (including a standard pesticide/PCB suite and toxaphene) deposited through long-range atmospheric transport in an arctic ice cap.

2. To quantify annual deposition and historical residue trends of trace metals deposited through long-range atmospheric transport in an arctic ice cap.

3. To investigate the temporal trend of polycyclic aromatic hydrocarbons (PAHs) and elemental carbon (EC) transported and deposited on an arctic ice cap through long-range atmospheric transport.

4. To compare findings from Canadian research with comparable information from ice caps within Russia.

DESCRIPTION

The initial work on Agassiz Ice Cap to investigate trends of contaminants stored in the glacial snow and ice was undertaken in 1986 and 1987. This work has been published in Gregor (1990), Barrie *et al.* (1992), Gregor (1991a), Gregor (1991b) and Gregor *et al.* (in press). Subsequent to this intensive study we have been maintaining an annual watch on deposition to the ice cap so that we can continue the profile. Efforts have been made to extend the snow pit record beyond the original 1970 date but these results are questionable due to the fact that the pit has been kept open for more than five years. As well, sampling and analytical methods have improved and the target list of compounds have expanded over this time. This new effort will extend the parameter list to match current research activities in the north, will ensure consistent sampling throughout the entire profile and permit annual profiling of contaminant residues from the recent past to the present, thereby greatly extending the current record.

The Agassiz Ice Cap, at an elevation of about 1800 m and at 80°N is believed to be an excellent location to study the deposition of atmospheric contaminants. Its northerly location means that contaminant data should characterize the deposition of material transported from Eurasia across the Arctic Ocean. Furthermore, its location means that there will be comparability between contaminant patterns presently being measured at Alert by means of high volume air samplers and weekly snow collectors. Thus the detailed record available for Alert, can be related to the long-term record of residues from the ice cap. These data will provide the most detailed long-term temporal record of contaminants in the Canadian Arctic. As the methods are comparable to all other snow sampling methods, the long-term data will be generally comparable to the annual spatial data collected across the NWT under a separate project.

ACTIVITIES IN 1992/93

In the spring of 1993, with the support of the Polar Continental Shelf Project (PCSP), two visits were made to the Agassiz Ice Cap on north central Ellesmere Island (80°40'N, 73°30'W).

The first visit in mid-March was intended to complete all of the sampling; however, the extremely cold weather at the time (-40°C and below) and associated equipment problems delayed the progress of the work requiring a second visit in May, 1993. In preparation for the March field expedition, over 5000 Kg of freight were moved to the Ice Cap initially by scheduled jet service to Resolute Bay, chartered turboprop service to Eureka and finally Twin Otter aircraft provided by PCSP and equipped with wheel/skis to ferry the equipment onto the ice cap. While it had been intended to undertake this work in April or May, the PCSP aircraft support was limited to March so work had to be completed to the extent possible prior to March 31, 1993. The extreme weather conditions, and limited light at this time of the year required additional equipment, fuel and personnel. The increased cost was more than offset by the aircraft support provided by PCSP at no cost to the study.

Snow samples representing a thirty year period of contaminant deposition to the Agassiz Ice Cap were collected, melted and extracted on site. To sample each annual layer, a pit was excavated to a total depth of over 8 m. Samples were taken horizontally from the pit so as to minimize contamination and exchange with the atmosphere. These samples will provide the longest detailed record of contaminant deposition in the Arctic presently available for OCs/PCBs, PAHs and toxaphene. All work in the snow pit was undertaken using battery power to minimize contamination due to fossil fuels.

It had been planned to continue the sampling beyond the early 1960 time period through the use of core samples. Unfortunately, coring was not successful during either of the two trips due to equipment problems inherent in the hand operated corer in hard packed firn and ice. If funding is available next year, we propose to work cooperatively with R. Koerner, Energy, Mines and Resources, and obtain core samples using his new electro-mechanical corer which was successfully tested at the same site in May, 1993. These core samples will allow us to extend the record of OC/PCB and toxaphene deposition to pre-production periods on about five year intervals. We will also attempt to extend the record for PAHs back to 100+ years if contamination from the corer can be controlled.

Trace metals were sampled in a special pit nearby the main pit but in which special precautions were made to avoid contamination from metals. Multiple seasonal samples were taken to investigate seasonal deposition rates.

RESULTS

Including replicates, a total of 41 samples were collected for OC/PCB and PAH analyses. An additional 35 samples were collected specifically for toxaphene. Additional samples were collected for trace metals and elemental carbon. The analyses of these samples will be undertaken during 1993.

UTILIZATION OF RESULTS

To date there is inadequate information regarding current contaminant deposition in arctic ecosystems as well as details with respect to the trends of many of these contaminants. Ice caps have been utilized to measure changes in global tropospheric chemistry over long time periods, specifically with respect to climate change (e.g. CO_2), inorganic pollutants resulting from global industrialization, and the effect of naturally occurring large scale phenomena (e.g. volcanoes). The only known work to date on organochlorines in arctic ice caps is that of Stengle *et al.* 1973 and various publications by this author (Gregor 1990, Gregor 1991a,b, Barrie *et al.* 1992; and Gregor *et al.* (in press)). Stengle investigated only the pesticide DDT and did not determine its presence above the limit of detection of the time. Gregor's work has reported on 17 years of contaminant residues in the Agassiz Ice Cap on north central Ellesmere Island between 1970 and 1987. Samples have continued to be taken on this Ice Cap but from a pit originally opened in 1987 and during which time sampling and analytical procedures have changed. In addition, no long-term historical record exists for toxaphene as this was not a priority contaminant when this work was begun. Thus, these data will provide for the first time a thirty year record of contaminant deposition to the arctic environment which will be useful in i) assessing trends seen in other media; ii) understanding the future risk of contaminants in the north to the ecosystem, including native diets; and iii) evaluating models.

Expected project completion date: 1996/97

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RIVERINE INPUTS OF CONTAMINANTS TO THE ARCTIC MARINE ENVIRONMENT

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PROJECT TEAM: D. Gregor, A. Peters, M. Alaee, NWRI, Burlington
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OBJECTIVES

Short-term:

1. To estimate and characterize the total contaminant load delivered by major river systems to the arctic marine environment.

2. To characterize the source of the total contaminant burden of the selected rivers with respect to source through the use of biogeochemical markers.

3. To investigate seasonal variations of the total contaminant burden of arctic river systems and assess the controlling biogeochemical processes.

Long-term:

1. To investigate and quantify the processes and rates of contaminant transport and transformation in northern riverine systems.

2. To assess whether existing models of estuarine dynamics and transport developed for temperate systems are applicable in arctic estuarine systems.

3. To refine existing temperate riverine system contaminant delivery models or if necessary develop new ones for application to arctic rivers.

4. To develop an understanding of the biogeochemical dynamics of contaminants in northern systems at the freshwater/marine interface.

DESCRIPTION

The focus of the Arctic Environmental Strategy is the evaluation of contaminant transport to the marine ecosystem, uptake in the bio-system, and possible effects on top predators (i.e. humans). Northward flowing rivers, which drain 10^7 km^2 of northern Asia, northern Europe and North

America, are thought to be major conduits to the Arctic Ocean of contaminants originating from point sources and/or atmospheric deposition to the terrestrial ecosystem. Most of the major rivers are found within Europe and Asia. The Mackenzie River is the only major north-flowing river in North America. While the amount of freshwater flowing into the Arctic Ocean is approximately equivalent to only 1% of the inflow of water entering through Fram Strait, rivers have an important effect on the oceanography of the region. Freshwater inflows contribute to the stable, less dense, more productive surface layer, and adjacent to river mouths, introduce turbid water and establish horizontal density gradients which drive ocean currents. The processes controlling the types of contaminants, as well as the timing and rate of their provenance to the marine environment by the river systems, likely differ from those in temperate climates. Similarly, the impact on the coastal, productive receiving waters must also be assessed to fully appreciate the importance of the riverine delivery of contaminants.

Information on contaminant loadings to the Arctic Ocean from any northward flowing river is limited. Data for the Mackenzie River (1985 and 1986) collected by NWRI are essentially restricted to hydrocarbons and metals and these data are too limited to estimate seasonal loadings or to assess aquatic system effects. The Institute of Ocean Sciences collected dissolved and suspended and bottom sediment samples in the vicinity of the mouth of the Mackenzie River in 1987. These samples have not been analyzed for organochlorines, and as they were neither collected nor stored for this purpose, are of questionable use at this time. Hence neither study provided data useful to assess seasonal patterns, contaminant speciation, bio-availability, etc.

While the water quality monitoring component of the AES will contribute basic data on contaminant levels in northward flowing Canadian rivers, there is a need for more extensive information related to their fate and effects, both within the river system itself and particularly at the freshwater/seawater interface. As well, while the Mackenzie River is the largest North American tributary of the Arctic Ocean, there is a need to undertake similar research in river systems that drain other biogeographical zones (e.g. tundra) and inflow to other marine environments (e.g. estuarine versus deltaic). The information necessary to address these complete questions can only be obtained through intensive research projects effectively extending existing research data and anticipated monitoring data.

ACTIVITIES IN 1992/93

Element 1: Mackenzie River

The Mackenzie River was sampled on three occasions in the summer of 1992: in June, July and August. Sampling was undertaken close to Inuvik (East channel), Aklavik (West channel) and Arctic Red River (main channel) to provide samples of river water, suspended sediment and filterable solids. In addition, water temperature, conductivity and pH were measured at all sites and samples were also collected for the determination of major cations and anions. Two large volume samples (200 L) were also obtained at Inuvik specifically for the analysis of toxaphene. All samples are currently undergoing analysis at NWRI, where they are fractionated and analyzed first for the basic analytical scheme which includes organochlorine pesticides, PCBs and priority PAHs. The more intensive mass spectrometry work will be conducted as the second

phase of the analyses. This phase will include the geochemical markers, alkyl-PAH homologues and alkane and fatty acid distributions. However, it is most efficient to delay this work, until the initial data have been thoroughly evaluated.

Element 2: Reconnaissance Sampling of Major Northward Flowing Rivers

Although the Mackenzie River is the largest Canadian river tributary to the Arctic Ocean, there are other significant river systems which are draining basins entirely different from that of the Mackenzie. In particular, river systems such as the Back and Hayes, drain large areas of tundra which are likely to deliver contaminants from snowmelt that differ to those from large portions of the Mackenzie valley which is treed and mountainous. Second, the eastern Arctic may be subject to different loading rates and contaminant mixtures than the Mackenzie valley (contaminant deposition patterns and mixtures will be assessed through the spatial snow surveys as outlined under the project entitled "Current Contamination Deposition Measurements in Precipitation").

To initially compare these river systems, a baseline survey of 11 rivers was conducted in August 1992. The same sampling protocol applied to the Mackenzie River was employed, with the exception that owing to the very low filterable sediment load at all of these sites, samples of water and filterable solids only were collected. These samples are currently undergoing the same analytical process as described for the Mackenzie River samples, above. Large volume samples (120 L) were obtained at 4 sites for the analysis of toxaphene.

RESULTS

To date, only the dissolved phase samples have been fully analysed for organochlorines (OCs). The raw data currently available show that for total PCBs and other OCs, most samples have values below detection limits when corrected for blank values. This is an encouraging indication but to a certain extent this is also to be expected when considering the remoteness of the sites and the fact that these are hydrophobic compounds that exhibit a relatively high degree of particulate affinity. A better indication of site contamination will be available from the particulate phase samples.

The exceptions to the above are the data for lindane and DDT. Values for these compounds at each site are presented in Table 1. Even with these limited data, there is an indication of a difference between the Mackenzie River sites and the other rivers. Values of dissolved α-HCH (both are measured on filtered samples) in the arctic rivers are more than ten times higher than the Mackenzie River samples except for the Hayes, Ellice, Anderson and Quoich Rivers. Concentrations of the γ-HCH are commonly, but not consistently, higher in the eastern rivers than in the Mackenzie system. Ongoing work will investigate and attempt to explain the significance of these differences. It should be noted that although the Mackenzie samples were collected in June and the other arctic rivers were sampled in July, there is some degree of comparability as the Mackenzie system melts earlier than the more easterly rivers.

CONCLUSION AND UTILIZATION OF RESULTS

A comprehensive analysis of the data at the present time is limited by the current extent of the dataset. Meaningful conclusions with respect to contaminant loadings and sources will be possible once the data for contaminant concentrations in the other phases (suspended sediment and solids) are available, and also when data for other contaminants (ie. PAHs) and classes of compounds (ie. alkanes and fatty acids) are available. This will be possible in the near future.

Expected project completion date: March 31, 1997

Partners: Contaminants, Waste Management and Water Quantity/Quality components of AES
Inland Waters Directorate, Western & Northern Region, Yellowknife
Government of Northwest Territories
Water Resources, Indian and Northern Affairs Canada

Table 1. Hexachlorocyclohexane (HCH) and DDT ($pg.l^{-1}$) in water samples from June 1992 for the Mackenzie River and from July, 1992 for the other rivers. All samples are from filtered water samples

Compound	Mack. East	Mack. West	Mack. Main	Copper-mine	Dubawnt	Back	Burn-side	Kazan	Lori-llard	Quoich	Thelon	Hayes	Ellice	Ander son
α-HCH	--	82	66	1330	1399	1014	1118	626	1192	140	819	316	220	--
β-HCH	--	--	--	--	--	--	--	--	--	--	--	--	--	--
γ-HCH (lindane)	8	26	32	148	178	267	215	125	283	--	198	139	--	54
Σ-HCH	8	108	98	1478	1577	1281	1333	751	1475	140	1017	455	220	54
(α:γ ratio)	NA	3:1	2:1	9:1	9:1	4:1	5:1	5:1	4:1	NA	4:1	2:1	NA	NA
o,p-DDT	2	--	--	--	--	--	--	--	--	--	--	--	--	--
p,p-DDT	--	29	--	78	--	--	--	--	--	--	--	--	--	--

Instrumental detection limit: 500 $fg.\mu l^{-1}$

--: not detected

N.B. All values are blank corrected

DEPOSITIONAL TRENDS - LAKE AND MARINE SEDIMENTS

PROJECT LEADER: W.L. Lockhart, Fisheries and Oceans Canada, Central and Arctic Region, Freshwater Institute, Winnipeg, Manitoba

PROJECT TEAM: P. Wilkinson, D. Metner ; E. Slavacek (contractor); B.N. Billeck, (PAHs and hydrocarbons); D. Muir, N. Grift, A. Yarechewski, C. Ford ; R. Hunt, R. Wagemann. Samples are also being supplied to H. Kling for algal morphology studies, and to J. Smol and N. Doubleday (Queen's University) for studies of carbon particles.

OBJECTIVES

To obtain sediment core samples from a grid of arctic headwater lakes in sufficient quantities to permit determinations of layer ages using radionuclide concentrations and down-core profiles of contaminating polycyclic aromatic hydrocarbons, organochlorines, and metals. In addition, sediments from marine locations are taken in instances where the availability of ships and equipment permit. The aim is to generate contaminant input data over a grid of locations throughout the Canadian Arctic and to set those inputs in the context of inputs over the last century and longer. Initial work was on a north/south transect, from the U.S. border to Northern Ellesmere Island, and work is now underway in the western Arctic to initiate an interlocking east/west transect.

INTRODUCTION

The primary source of several contaminants in arctic fish and wildlife is aerial deposition (LRTAP) of contaminants originating from southern latitudes. This program measures the rates at which contaminants are being supplied to arctic lakes and marine sediments and determines whether the current inputs are more or less severe than those which have occurred over the last century and longer. Through the linkage to the subproject on contaminant trends in freshwater and marine biota we can determine how contaminants in the fish are related to current inputs, and hence predict future contamination under various loadings. The marine component was added with the collection of several cores from Hudson Bay during a cruise of C.S.S. Hudson in August, 1992.

The international Arctic Monitoring and Assessment Programme (AMAP) has identified lake and marine sediments as key components for monitoring spatial variations in contaminants on a circumpolar basis. However, the spatial resolution required by AMAP for good circumpolar coverage of lake sediments will require at minimum 15-20 sites in the Canadian Arctic. Additional sampling, possibly by use of KB corers, is planned to try to meet AMAP's 1995 deadline.

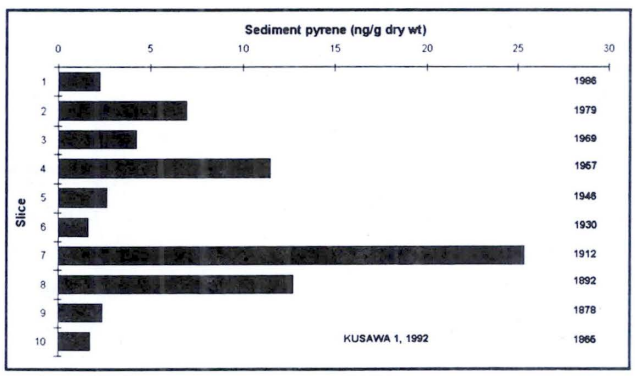
METHODS

Cores are collected from depositional zones, sliced at the point of collection into thin slices, and brought back to the laboratory where they are analyzed separately for radionuclides (mainly lead-210, radium-226, cesium-137, beryllium-7) to estimate mixing and, if mixing is not serious, to estimate time intervals during which slices were deposited. The slices are also analyzed for sediment mass and a series of chemical contaminants, notably organochlorines, several metals and polycyclic aromatic hydrocarbons. Taken together these assays allow the calculation of the rates of current and historical net inputs of each contaminant to the sediments.

RESULTS

Field sampling

Field activities during 1992/93 included a summer trip to Lake Belot, N.W.T. and participation in the cruise of C.S.S. Hudson in southern Hudson Bay. The summer trip to Lac Belot was required because data from some previous collection sites indicate that core locations were not in satisfactory depositional zones. Core sampling in arctic lakes generally is hampered by a lack of bathymetric data on lakes, so that location of sedimentation areas has been a matter of luck. With the development of hand-held global positioning systems, we can now determine core sites very accurately, but it is still necessary to know the bathymetry of the basin to choose likely settling zones. Since there was no previous information on the bathymetry of Lac Belot, the summer trip was arranged to take advantage of an offer by C. Gubala (U.S. EPA contractor) to visit the lake and make a bathymetric map using a newly developed portable combination of GPS positioning and sonar depth sounding. This unit can be operated from a small boat operated at high speed, and so production of a bathymetric map of even a large lake like Lac Belot can be done quickly. The bathymetric map of Lac Belot (Fig. 1) is the first ever prepared for that lake, and it allowed us to select core sites based on basin morphology and proceed directly to them by snowmobile (using GPS navigation) on the coring expedition in April, 1993.



The second field activity in 1992 was the participation in the cruise of C.S.S. Hudson to southeastern Hudson Bay during August, 1992. This was a geophysical expedition organized by the Atlantic Geosciences Centre and from it we obtained 10 cores from near-shore areas of Hudson Bay at the locations shown in Figure 2.

Sediment Samples

Over the life of this project, and through cooperative ventures with a number of other individuals and projects, we have accumulated cores or dredge samples from several locations. A complete listing of those sediments until April, 1993 is appended.

Only a few representative sets of data can be shown in a summary report of this type. For example, we recently reported high levels of mercury in arctic char from Amituk Lake on Cornwallis Island, and so the core profile of mercury from that lake is of interest. Figure 3 (top) shows the down-core profile of lead-210 in the upper slices of sediment from Amituk lake together with the mercury content of slices from the same core. The lead-210 profile shows almost ideal exponential decay for the first four slices, and then an anomalous event followed by another period of near ideal decline. (There was excess lead-210 only in slices 1-9, and so deeper slices cannot be dated by this isotope). Based on the lead-210 counts, the time span covered by the top 9 slices is from 1870 to the present. The dates are used to estimate the sedimentation rate (389 g $m^{-2}yr^{-1}$). Estimating the ages of older slices is by extrapolation, but the mercury profile (Fig. 3, bottom) obviously spans several hundred years.

There is a clear increase in mercury in the top three slices (since the early 1960s), and there is the suggestion of a smaller increase from slice 9 upwards (since 1870)(Fig. 3, bottom). The mean mercury content for slices 10 and deeper is 15.7 ng g^{-1}, as compared with over 70 in the top slice, and a mean of 28.2 for slices 4-9. If slices 10 and deeper are taken as a true geological background, then the average loading prior to the mid 1800s was about 6.1 μg $m^{-2}yr^{-1}$ (15.7 ng g^{-1} x 389 g $m^{-2}yr^{-1}$) while the current loading is about 28.4 μg $m^{-2}yr^{-1}$ (73 ng g^{-1} x 389 g $m^{-2}yr^{-1}$).

One reason for the high levels of mercury in fish from Amituk lake is clearly their age; the fish from our samples there ranged up to 30 years old. However, the sediment profile in Figure 3 suggests increasing inputs over at least the last 30 years, and probably for more than a century, and these increasing inputs may have provided increased amounts of mercury to the fish. There are two questions which often come up in interpreting core profiles for mercury. The first is whether core profiles with clear increases near the top as seen in Figure 3 actually represent increased loadings rather than some unknown process which might create such a profile without increases in loading. The second question is whether increasing inputs of mercury from aerial deposition actually results in increased mercury in the fish. In some cases, this appears to be the case (Johnson 1987), but there is still too little evidence to be sure. All the lake sediment profiles we have from eastern arctic lakes where the lead-210 profile indicates little or no mixing have shown striking increases in mercury in the uppermost slices, which we interpret as accelerating loadings over the last century or longer.

Lake Kusawa was sampled late in 1992 as a "control" for Lake Laberge. The lead-210 profile shown in Figure 4 (top) indicates little mixing and allows the assignment of dates back to the mid 1800s as shown. The sedimentation rate calculated is 227 g $m^{-2}yr^{-1}$. Organochlorine analyses are available for the top five slices, and representative results are shown for ΣPCB in the middle part of Figure 4. The PCB profile indicates increasing loadings with PCBs during the 1940s, with peak loadings during the 1950s and 1960s, followed by a striking recent decline. This is

generally consistent with the history and use of PCBs. Figure 4 also shows a profile for pyrene, a typical combustion-related hydrocarbon. In this instance the profile is quite unusual with a large peak in slices 7 and 8 dating from the late 1800s to some time between 1912 and 1930. Other than those two slices, the profile is similar to that for Hawk Lake (Saqvaqjuac) reported earlier and that from other North American sites. The reason for the striking peak in slices 7 and 8 is unknown; it is present in a number of other PAHs as well. There is nothing comparable in the Lake Laberge profile for pyrene. The analyses of the PAHs are replicate GC/MS injections of single extracts. The replicates agreed well, the these slices are being re-extracted in an effort to confirm them.

Preliminary analysis of core 10 from Hudson Bay has been completed for lead-210 and mercury. The lead-210 and cesium-137 profiles suggest mixing in at least the top 5 slices (5 cm), and the mercury profile is uniform throughout the whole core. The source of these sediments is presumed to be largely from erosion in the Grande Baleine River system, and failure to find a mercury gradient may reflect simple mixing or it may mean that the sediments have in fact not changed over time. The sediments showed evidence of deep burrows by polychaetes and so biological mixing processes may have obscured any historical profile.

CONCLUSIONS

The sediment cores give excellent records of inputs of contaminants to arctic lakes when coring sites with little mixing have been found. The scarcity of bathymetric information on arctic lakes has made location of such sites problematic until now. The availability of portable GPS/sonar equipment and software has now made it possible to produce high quality bathymetric information in a very short time, and that information is critical to the selection of good sites for coring. At unmixed sites we have evidence of increasing inputs of some contaminants (e.g. mercury), declining inputs of some (e.g. ΣPCBs), and variable inputs of others (e.g. PAHs). For a given contaminant we can see geographic trends in current and historical loadings. The measures of contaminant loadings offer the means to compare among sites at the present time and over the past 100-150 years. Since they provide information over a relatively long period, the cores allow the separation of natural inputs of contaminants from those typically originating with human activities. Given that the sources are generally from aerial deposition to the watershed, the core histories also provide a sound basis for extrapolation to future loadings.

Partners: Logistical support by the Polar Continental Shelf Program and by the Atmospheric Environment Service, Environment Canada; collaboration with J. Ford and C. Gubala (US EPA, Corvallis).

REFERENCE

Johnson, M.G. 1987. Trace element loadings to sediments of 14 Ontario lakes and correlations with concentration in fish. Can. J. Fish. Aquat. Sci. 44: 3-13.

Figure 1. Bathymetric map of Lac Belot, N.W.T. 1992.

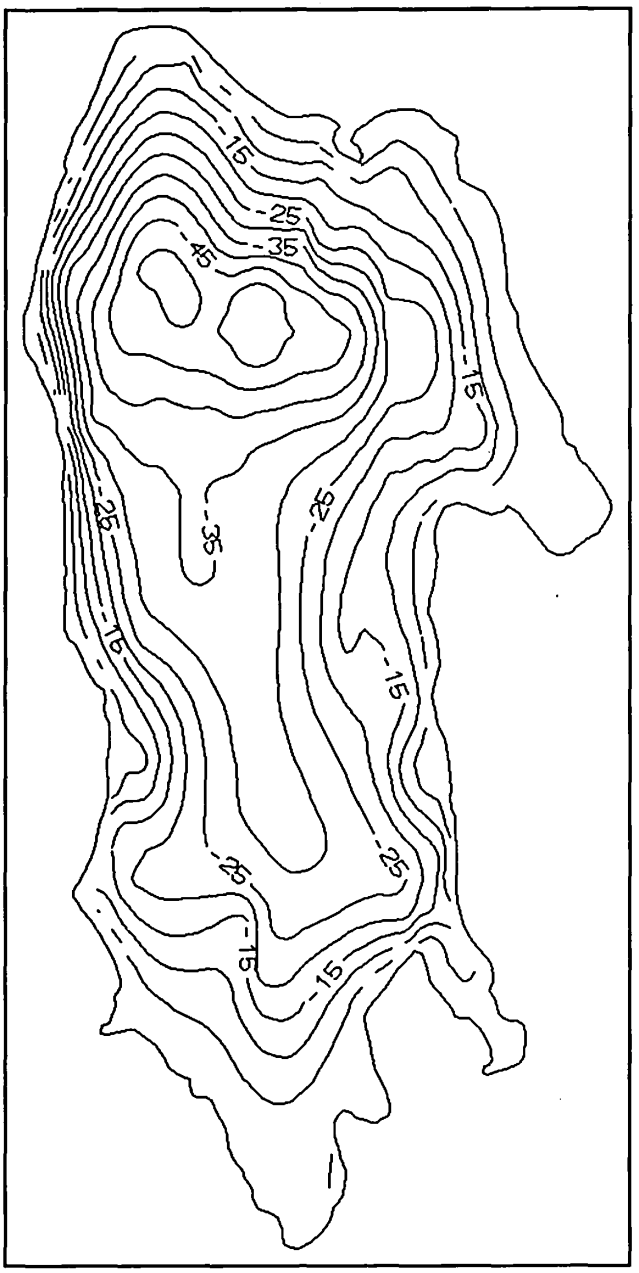


Figure 2. Locations of cores taken from Hudson Bay, August 1992.

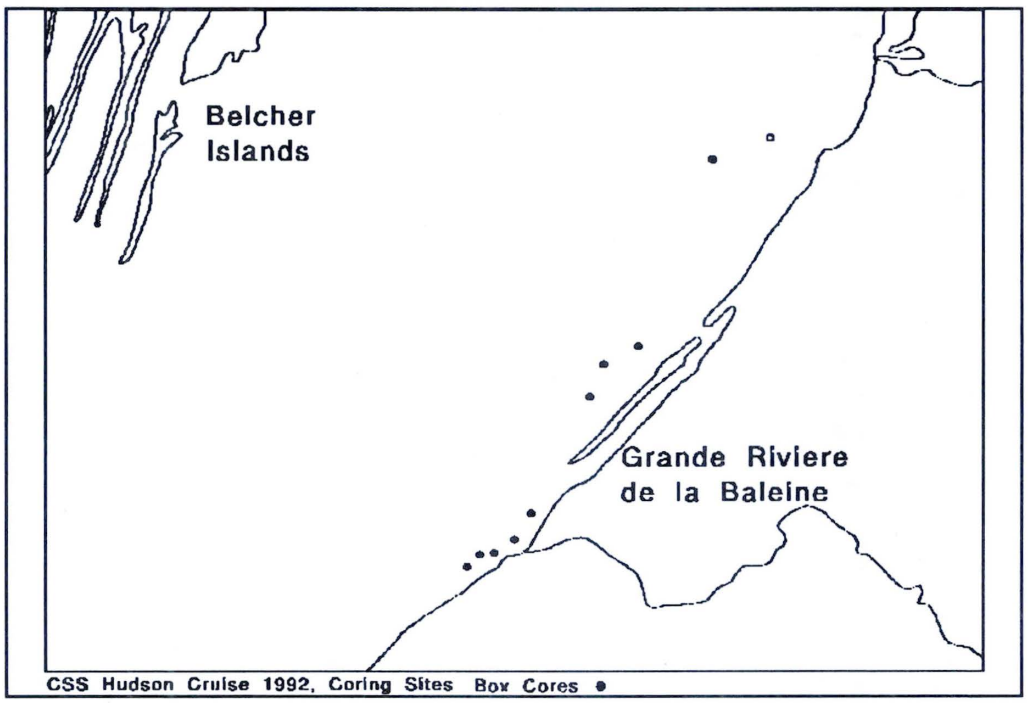


Figure 3. Sediment Lead-210 and mercury profiles for Amituk Lake, Cornwallis Island, N.W.T.

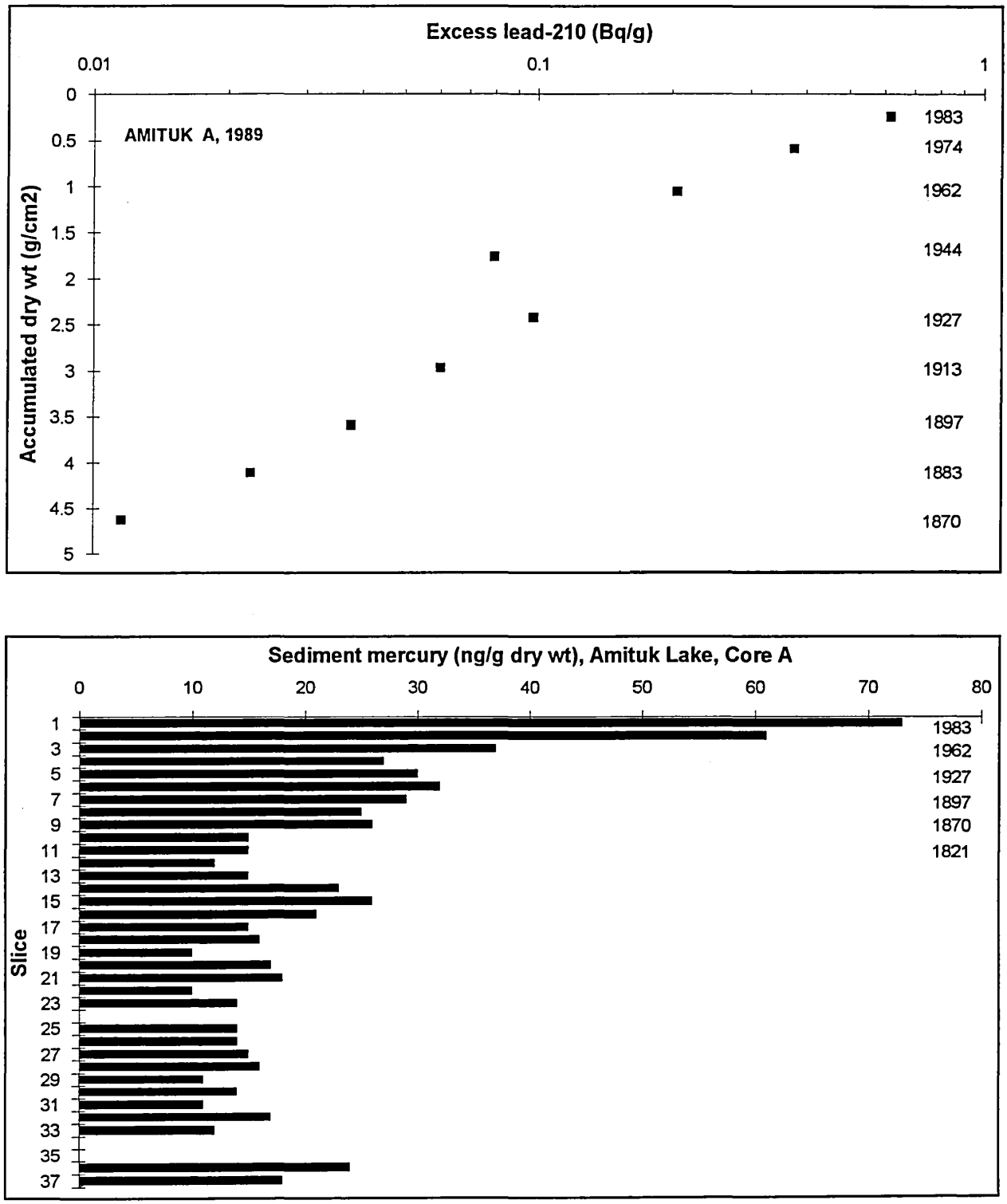


Figure 4. Sediment lead-210, ΣPCB, and pyrene from Lake Kusawa, Yukon Territory, 1992.

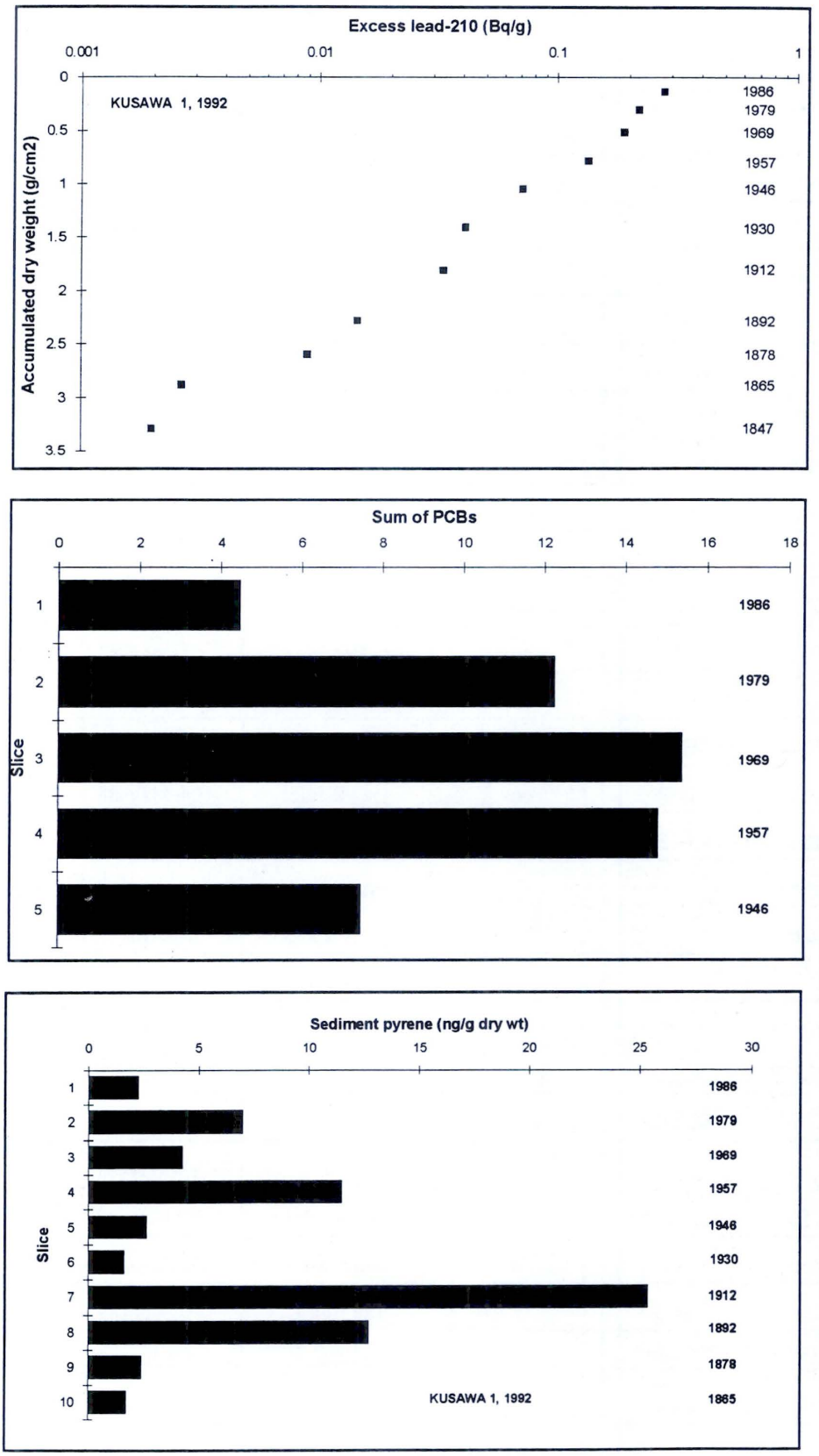


Table 1. List of sediment samples and cores available.

Sample Number	Core Number	Core Type	Location	Date	Lats and Longs
8701001-8701030	A to Q	15 dredges	ELAL382	20-Mar-87	
8701038-8701047	R to V	5 dredge	ELAL382	18-Jun-87	
8701031-8701037			Devon Is	27-Mar-87	
8701048			Upper Carp L	27-Jul-87	
8701049-8701063	ARS1-32	15 ekman	Athabasca R	07-Aug-87	
8701064	3BS066F		Baker L	30-Oct-87	
8701065	3BS065F		Schultz L	30-Oct-87	
8802001-8802032	1	box	ELA L382	03-Mar-88	
8802001-8802032	1	box	ELA L382	03-Mar-88	
8802038-8802062	FLA	box	Far L.	30-Apr-88	
8802063-8802087	FLB	box	Far L.	01-May-88	
8802088-8802112	HLA	box	Hawk L.	02-May-88	
8802113-8802134	HLB	box	Hawk L.	03-May-88	
8802135-8802159	HIA	box	Hawk Inlet	04-May-88	
8802160-8802184	HIB	box	Hawk Inlet	04-May-88	
8802257-8802268			Whatever L.	14-Aug-88	62.673N/97.039W
8902001-8902026		box	ELA L375	10-Apr-89	
?		KB	ELA L375	10-Apr-89	
8902037	control	box bottom	ELA L111	08-Apr-89	
8902238	auger	exhaust	ELAL111	08-Apr-89	
8902071-8902107	SLKBA	KB-10	Sophia L.	28-May-89	
8902108-8902145	SLKBB	KB-10	Sophia L.	28-May-89	
8902146-8902156	SLBCA	box	Sophia L.	30-May-89	
8902157-8902165	SLBCB	box	Sophia L.	31-May-89	
8902166-8902210	ALKBA	KB-10	Amiktuk L.	01-Jun-89	
8902211-8902236	ALKBB	KB-10	Amiktuk L.	03-Jun-89	
9002046-9002060	HAZKBA	KB-10	Hazen L.	12-May-90	

Table 1 Cont'd

9002061-9002078	HAZKBB	KB-10	Hazen L.	13-May-90	
9002079-9002090	HAZABC	box	Hazen L.	14-May-90	
9002091-9002095	HAZBBC	box	Hazen L.	14-May-90	
9002096-9002101	HAZCBC	box	Hazen L.	15-May-90	
9002125-9002129		5 dredges	Red River	15-Jun-90	
9102021-9102045	BLKBA	KB-10	Buchanan L.	18-May-91	
9102046-9102080	BLKBB	KB-10	Buchanan L.	20-May-91	
9102081-9102120	BLKBC	KB-16	Buchanan L.	21-May-91	
9102131-9102132	NSR	2 dredges	N.Sask.River	04-Oct-90	
9102133	0-10 cm	dredge	Sydney L.	15-Sep-91	
9102134	0-10 cm	dredge	Green L.	15-Sep-91	
9202001-9202021	LAB1	KB-16	Lac Laberge	10-Mar-92	
9202022-9202049	LAB2	KB-16	Lac Laberge	11-Mar-92	
9202050-9202075	LAB3	KB-16	Lac Laberge	11-Mar-92	
9202076-9202100	LAB4	KB-16	Lac Laberge	12-Mar-92	
9202101-9202129	KUS1	KB-16	Kusawa L.	16-Mar-92	
9202130-9202147	KUS2	box	Kusawa L.	16-Mar-92	
9202148-9202229		1.7 meter	Clear Lake	15-May-92	
9202230-9202254	NIP1	KB-10	L. Nipigon	10-Apr-92	
9202255-9202283	NIP2	KB-10	L. Nipigon	10-Apr-92	
9202284-9202314	NIP3	KB-10	L. Nipigon	11-Apr-92	
9202315-9202344	NIP4	KB-10	L. Nipigon	11-Apr-92	
9202346-9202355	soil	profile	L. Hazen	15-Jun-92	
9202701-9202729	Core #1	KB-10	Hudson Bay	15-Aug-92	55.288N/77.823W
9202730-9202766	Core #2	KB-10	Hudson Bay	16-Aug-92	55.336N/77.768W
9272767-9202801	Core #3	KB-10	Hudson Bay	17-Aug-92	55.239N/77.983W
9202802-9202837	Core #4	KB-10	Hudson Bay	18-Aug-92	55.260N/77.939W
9202838-9202870	Core #5	KB-10	Hudson Bay	18-Aug-92	55.263N/77.891W
9202871-9202910	Core #6	KB-10	Hudson Bay	19-Aug-92	55.629N/77.519W
9202911-9202946	Core #7	KB-10	Hudson Bay	23-Aug-92	56.091N/76.938W

Table 1 Cont'd

9202947-9202980	Core #8	KB-10	Hudson Bay	24-Aug-92	55.358N/77.720W
9202981-9202999	Core #9	KB-10	Hudson Bay	26-Aug-92	55.358N/77.719W
9202443-9202453	Core #9	KB-10	Hudson Bay	26-Aug-92	55.358N/77.719W
9202406-9202442	Core #10	KB-10	Hudson Bay	29-Aug-92	55.546N/78.747W
9302111-9302146	FOX1	KB-16	Fox Lake	10-Mar-93	61.226N/135.487W
9302147-9302178	FOX2	KB-16	Fox Lake	12-Mar-93	61.213N/135.440W
9302179-9302220	LAT1	KB-16	Little Atlin L.	15-Mar-93	60.271N/133.991W
9302221-9302262	LAT2	KB-10	Little Atlin L.	17-Mar-93	60.255N/133.944W
9302263-9302305	BEL4	KB-10	Lac Belot	14-Apr-93	66.881N/126.311W
9302306-9302348	BEL3	KB-10	Lac Belot	15-Apr-93	66.903N/126.324W
9302349-9302393	BEL2	KB-10	Lac Belot	15-Apr-93	66.963N/126.342W
9302394-9302439	BEL1	KB-10	Lac Belot	16-Apr-93	66.953N/126.294W
9302440-9302459	BELBC	box	Lac Belot	18-Apr-93	66.953N/126.296W
9302460-9302501	COL1	KB-16	Colville L.	19-Apr-93	67.064N/125.995W
9302502-9302547	COL2	KB-16	Colville L.	19-Apr-93	67.129N/125.921W

PROCESSES AND FLUXES OF CONTAMINANTS IN AQUATIC SYSTEMS

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OBJECTIVES

Short-term:

1. To establish an experimental basin in the High Arctic for the purpose of determining a detailed hydrologic and contaminant budget for the basin.

2. To measure in an intensive manner within the basin the hydrology and the flux of organic contaminants and specific ions especially during the winter accumulation and snowmelt seasons.

3. To investigate and quantify the major processes affecting contaminant transformation, transport and fate in each basin in order to provide predictive capability for basin mass balances.

4. To develop and calibrate contaminant transport models utilizing the results from this basin study and to compare the results among comparable basin studies.

Long-term:

1. To quantify mass balances of contaminants and selected inorganic substances for selected arctic watersheds in the Canadian north.

2. To investigate and quantify the key abiotic processes controlling contaminant fate and dynamics in arctic freshwater systems.

3. To utilize the study results from all basins studied in the Arctic for model development and calibration in order to estimate contaminant transport and flux in larger northern aquatic systems.

DESCRIPTION

The Amituk Lake study began in 1992 under the auspices of the Arctic Environmental Strategy (AES) of the Green Plan. The objective of the multidisciplinary research activities was to document the distribution, pathways and sinks of inorganic and organic contaminants in an arctic snowpack and in the meltwaters as they move through a terrestrial lake basin. By recording the occurrence and movement of these chemicals in a freshwater system, a better understanding of the migration of contaminants to the marine environment can be attained. Furthermore, scientific information from a relatively small basin such as Amituk Lake will be used in model development and calibration for the behaviour of contaminants in larger northern aquatic ecosystems.

Amituk Lake is located on the eastern coast of Cornwallis Island at latitude 75° 02' 57" and longitude 93° 45' 51". The basin is underlain by Ordovician and Silurian carbonate rocks. The overall basin is approximately 26 km^2 and contains 6 small watersheds (Fig. 1) of which Gorge, Cave and Mud Creek account for 78% of the drainage area contributing to Amituk Lake (Table 1). The climate on Cornwallis Island is typical of High Arctic regions with an annual mean temperature of -16.6°C and total annual precipitation of 131.4 mm reported at Resolute Bay (Environment Canada, 1990). Snowmelt begins in mid- to late-June and peaks during the first two weeks in July. Streamflow ceases in mid- to late-August with freeze-up.

ACTIVITIES IN 1992/93

The study basin was visited in May, 1992 when a field laboratory facility was added to the thirty foot accommodation trailer moved there in March, 1992 and an extensive snow quantity survey of the basin was undertaken along with ground reconnaissance. Snow samples were collected both on the ice surfaces of Amituk Lake and a small lake on the western side of the basin. Two snow pits were dug in the valleys of Rock and Mud Creeks to provide a comparable snow chemistry sample in areas of high snow accumulation. These pits were covered and snow sampling continued at these sites during the snowmelt season. Also installed at this time was an automatic, year round, weather station. The weather station will be used to provide real time information of wind, temperature, snow accumulation and especially incoming and outgoing solar radiation as base information to subsequent modelling efforts.

A field party returned to the site in late June to undertake the snowmelt measurements which included snow and snowmelt chemistry, lake water chemistry, stream discharge measurements into and from the lake and surface water chemistry. This measurement program continued through to the middle of July, 1992.

Please note that this lake has previously been studied in part (lake sediments and fish) by Fisheries and Oceans Canada (DFO), Winnipeg under an earlier AES project. As well, NWRI undertook some preliminary water and snow chemistry work on this lake in conjunction with the DFO study. These earlier snow and water data for Amituk Lake are presented in Semkin and Gregor (1992).

RESULTS

The analyses of 1992 snow and water samples for organic contaminants has been completed. However, we have not had an opportunity to thoroughly review and integrate these data with the water balance data at this time. Therefore the emphasis this year will be on the water and ion budget of the system.

Snowpack

The snowpack at the Amituk Lake was very heterogeneous with respect to both water and chemical content. As a result of strong northerly winds continuously redistributing the snow cover, stream valleys and protected areas often contained deep drifts. Measurements taken in the valleys of Rock and Mud Creeks showed snow depths of 1.9 and 2.6 m respectively during a pre-melt survey whereas the flatter, windswept areas recorded depths averaging 0.6 m. The chemical composition of the snowpack was influenced by basin geology and meteorological conditions. At the western end of Amituk Lake, relatively clean snow in the drainage area of Gorge Creek displayed an average pH of 6.37. The cation pool was equally shared by Ca^{2+}, Mg^{2+} and Na^{+} ions; alkalinity (HCO_3^-) and Cl^- dominated the anions (Table 2). In the rest of the watershed, the snowpack exhibited a strong alkaline enrichment. Here, fine limestone/dolomite dust derived from the ridges surrounding the lake and in the sharply dissected stream valleys was picked up and disseminated across the central and eastern part of the basin. "Dirty" snow was characterized by a higher pH (7.33) and levels of Ca^{2+} and alkalinity ten times that found in the cleaner snow (Table 2).

Amituk Lake

A bathymetric survey was conducted through the ice on Amituk Lake to define the morphometry of the lake basin (Fig. 2). The lake area was calculated to be 37.84 ha and the mean and maximum depths 19.4 and 43.0 m, respectively (Table 3). Ice thickness on the first day of sampling, June 20, 1992, was measured at 2.38 m. Temperature profiling revealed a reverse stratification in Amituk Lake with water temperatures approaching 0°C beneath the ice cover and warming to 2.0°C at depth (Fig. 3). Not until the final survey on July 8, 1992 did the temperature of surface water increase above 2.0°C with the influx of relatively warm stream water (Mud Creek recorded a temperature of 14.8°C on July 7, 1992), higher air temperatures and increased solar radiation.

Chemical weathering of limestone and dolomite in the Amituk Lake basin was reflected in the high concentrations of Ca^{2+}, Mg^{2+} and alkalinity in the water column (Table 4). On an equivalent basis, Ca^{2+} plus Mg^{2+} and alkalinity constituted an average 95% and 90%, respectively, of the cation and anion content of lake water.

Stream Hydrology and Chemistry

Instantaneous discharges were measured at least daily at the six influent streams and at the outflow from Amituk Lake. Flow was first detected on June 26, 1992 in Mud and Camp Creeks which, because of the southern exposure, was several days ahead of Gorge Creek (Fig. 4). The

relative discharge rates parallel the distribution of drainage areas (Table 1) with Gorge Creek accounting for over 53% of the total inflow to Amituk Lake during the study period (Table 5). In terms of major ion chemistry, the streamwater composition was also indicative of limestone weathering in the watershed with Ca^{2+} and alkalinity (HCO_3^-) dominating the ionic content at all sites (Table 6). Gorge Creek contained somewhat lower levels of Ca^{2+}, alkalinity, K^+, SO_4^{2-}, and SiO_2, suggesting decreased weathering rates in this basin.

Water and Chemical Budgets

Streamwater flows into Amituk Lake were summed and compared to the lake outflow (Fig. 5). Based on three daily discharge measurements at the outlet, the outflow averaged 24.86 m^3/sec and ranged from 24.00 to 26.33 m^3/sec over the period June 26 to July 9, 1992 compared to a cumulative inflow of 23.81 m^3/sec. This translated as 2.065 x 10^6 m^3 measured water inflow and as 2.148 x 10^6 m^3 gauged outflow from Amituk Lake. The difference could be attributed to water storage in the lake proper and to water input from ungauged areas in the basin (which account for 9% of the total drainage area upstream of the outflow station).

Estimates of the water content of the pre-melt snowpack were difficult as a result of the irregular distribution and variable depth of the snow cover; however, by using time-lapse photography of the snowpack from fixed vantage points in the basin and field investigations, a value of 112.5 mm of snow water equivalent (SWE) was calculated for the overall basin. For a watershed area of 26.57 km^2, this meant that 2.989 x $10^6 m^3$ of snow water equivalent (less ablation losses) were initially available for surface run-off and infiltration.

Combining the estimated snowpack water content and the measured inflows and outflow with the water chemistry, a material budget was calculated for Amituk Lake (Table 7). Because of the uncertainty in delineating the distribution of "clean" and "dirty" snow, a range of possible chemical loadings in the parent snowpack is presented.

UTILIZATION OF RESULTS

The measurement of contaminants in snow in the spring has not been verified as a true estimate of the quantity of contaminants delivered to the freshwater ecosystem as a result of snowmelt. The processes that control the timing of contaminant release to melt waters and the efficiency of the release relative to the total deposition must be quantified before contaminant burdens to surface waters can be estimated. Once these contaminants have been released into the surface waters it is subsequently essential to be able to quantify the movement of these contaminants through the various compartments of the systems (e.g. lakes) so that losses and permanent and temporary sinks can be determined. It is ideal to conduct this work at a number of basins to assess the spatial variability of the abiotic processes (latitudinally as well as geomorphologically) but essential to continue the work for a number of years at least one basin to assess annual variability due to meteorological conditions. To date, this is the only northern aquatic system being studied in detail for contaminant mass balances. Information from this system will be essential in calibrating deposition and fate models as well as providing a better general understanding of the fate of contaminants in high arctic watersheds.

Measurements of total contaminant transport from major river systems to the marine environment have been recognized in the arctic contaminant research strategy as being essential to estimate total contaminant burden to the Arctic Ocean. However, this fails to recognize the importance of contaminant flux within freshwater aquatic systems as well as the contaminant flux from other smaller river systems to potentially critical marine environments with restricted circulation. The study of these small basins and the quantification of the basin mass balances and the development of contaminant transport models will greatly assist in addressing this information gap.

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Semkin, R. and D. Gregor. 1992. Processes and fluxes of contaminants in aquatic systems. Pp. 38-41 in: Synopsis of Research Conducted Under the 1991/92 Northern Contaminants Program, J.L. Murray and R.G. Shearer (eds.). Environmental Studies No. 68, Indian and Northern Affairs Canada. 10 Wellington Street, Hull, Quebec K1A 0H4.

Table 1. Drainage Patterns in the Amituk Lake Basin

Drainage Unit		Area(km2)
Amituk Lake		0.48
Direct to Lake		2.13
Direct to Outflow		0.31
East Creek		1.95
Rock Creek		1.08
Cave Creek		4.69
Gorge Creek		10.29
Mud Creek		5.22
Camp Creek		0.44
Total		26.57

Table 2. Chemistry of the Pre-Melt Snowpack, Amituk Lake, 1992

		Clean Snow	Dirty Snow	Dirty/ Snow
Snow Depth(m)		0.553	0.743	1.3
SWE(mm)		149.5	298.2	2.0
Density		0.27	0.39	1.4
Conductivity(uS/cm)		8.4	25.8	3.1
H(ueq/L)		0.4277	0.0473	0.1
Ca(ueq/l)		15.4	157.1	10.2
Mg(ueq/L)		13.8	26.3	1.9
Na(ueq/L)		12.3	21.4	1.7
K(ueq/L)		1.07	2.74	2.6
NH4-N(ueq/L)		0.25	0.52	2.1
Alkalinity(ueq/L)		20.1	192.6	9.6
SO4(ueq/L)		4.09	6.63	1.6
Cl(ueq/L)		19.5	35.9	1.8
NO3-N(ueq/L)		2.27	2.89	1.3
SiO2(mg/L)		0.003	0.024	8.0
DOC(mg/L)		0.075	0.406	5.4
DIC(mg/L)		0.328	2.13	6.5
TOC(mg/L)		N.D.	0.277	-

Table 3. Morphometry of Amituk Lake

Area(ha)		37.84
Volume(m3)		7336296
Mean Depth(m)		19.4
Max.Depth(m)		43.0
Shoreline Length(m)		4.05
Shoreline Development		22.07
Volume Development		1.35

Table 4. Amituk Lake Chemistry July 5, 1992
Ice Thickness 1.9 m

Water Depth(m)	25 C Cond uS/cm	pH	Ca mg/L	Mg mg/L	Na mg/L	K mg/L	NH4-N mg/L	ALK meq/L	SO4 mg/L	NO3-N mg/L	Cl mg/L	SiO2 mg/L	DOC mg/L	DIC mg/L	TOC mg/L
1.0	8.0	6.38	0.42	0.04	0.00	0.02	0.0003	0.03330	0.01	0.000	0.078	0.000	0.121	0.6	0.049
2.0	7.5	6.48	0.38	0.04	0.01	0.01	0.0003	0.01670	0.00	0.000	0.073	0.000	0.100	0.5	0.006
2.5	108.0	8.14	15.18	3.59	1.23	0.14	0.0010	0.97590	1.02	0.020	2.630	0.163	0.200	12.4	0.177
3.0	110.0	8.14	16.71	3.56	1.29	0.14	0.0005	0.99850	1.10	0.019	2.760	0.172	0.262	12.8	0.175
4.0	125.0	8.15	18.23	3.77	1.45	0.17	0.0010	1.15140	1.47	0.022	3.220	0.199	0.259	14.6	0.103
5.0	138.0	8.17	19.21	4.05	1.59	0.18	0.0005	1.26570	1.66	0.023	3.710	0.207	0.246	16.0	0.088
6.0	152.0	8.21	22.72	4.35	1.74	0.21	0.0007	1.39380	1.93	0.025	4.140	0.238	0.192	17.6	0.064
7.0	153.0	8.20	22.74	4.39	1.81	0.21	0.0007	1.39030	1.95	0.026	4.150	0.238	0.193	17.6	0.053
8.0	154.0	8.19	21.49	4.41	1.80	0.21	0.0002	1.39290	1.96	0.032	4.140	0.238	0.192	17.8	0.051
9.0	153.0	8.19	22.25	4.36	1.83	0.22	0.0002	1.39310	1.93	0.028	4.140	0.239	0.168	17.7	0.056
15.0	153.0	8.19	22.64	4.47	1.78	0.22	0.0056	1.39890	1.98	0.027	4.170	0.234	0.121	17.7	0.085
21.0	150.0	8.23	21.76	4.33	1.79	0.21	0.0201	1.37570	1.92	0.014	4.060	0.270	0.456	17.5	0.008
25.0	151.0	8.21	23.30	4.38	1.78	0.22	0.0273	1.38410	1.95	0.017	4.100	0.247	0.083	17.6	0.089
30.0	149.0	8.22	22.05	4.14	1.80	0.21	0.0164	1.36640	1.92	0.020	3.950	0.316	0.046	17.5	0.092
35.0	149.0	8.20	22.61	4.17	1.78	0.22	0.0051	1.37460	2.02	0.044	3.910	0.392	0.017	17.5	0.110
42.0	185.0	8.02	29.25	5.28	2.19	0.23	0.0035	1.75090	2.28	0.177	4.860	2.870	0.017	22.4	0.035

Table 5. Discharge Summary of Streams Influent to Amituk Lake, June26-July9, 1992

Stream		Discharge (m3/sec)
East		1.3641
Rock		1.6367
Cave		4.4382
Gorge		12.5233
Mud		3.6778
Camp		0.1693

Table 6. Volume-Weighted Concentrations In Stream Water 1992

Site	Units	H	Ca	Mg	Na	K	NH4-N	ALK	SO4	Cl	SiO2	DOC	DIC	TOC
Rock Creek	(meq/L)	7.7485E-06	0.8822	0.2024	0.0362	0.0056	0.0001	1.0391	0.0187	0.0376				
	(mg/L)	8.11	17.68	2.46	0.83	0.22	0.0014		0.90	1.332	0.506	0.305	13.02	0.195
Cave Creek	(meq/L)	8.3466E-06	0.8542	0.1998	0.0471	0.0065	0.0002	0.9978	0.0296	0.0634				
	(mg/L)	8.08	17.12	2.43	1.08	0.26	0.0030		1.42	2.246	0.311	0.354	12.47	0.184
Gorge Creek	(meq/L)	1.0151E-05	0.5322	0.2238	0.0338	0.0019	0.0001	0.7030	0.0085	0.0475				
	(mg/L)	7.99	10.66	2.72	0.78	0.08	0.0017		0.41	1.685	0.054	0.276	8.96	0.194
Mud Creek	(meq/L)	6.6827E-06	1.0133	0.2152	0.0496	0.0062	0.0002	1.1649	0.0224	0.0553				
	(mg/L)	8.18	20.31	2.61	1.14	0.24	0.0027		1.07	1.959	0.462	0.524	14.57	0.365
Camp Creek	(meq/L)	7.2838E-06	0.9115	0.2613	0.0494	0.0060	0.0001	1.0922	0.0360	0.0625				
	(mg/L)	8.14	18.26	3.17	1.14	0.24	0.0020		1.73	2.216	0.519	0.823	13.81	0.513
East Creek	(meq/L)	6.1614E-06	1.0048	0.2098	0.0390	0.0057	0.0001	1.1552	0.0219	0.0391				
	(mg/L)	8.21	20.13	2.55	0.90	0.22	0.0017		1.05	1.388	0.520	0.356	14.49	0.256
Amituk Lake	(meq/L)	6.9146E-06	0.8993	0.2734	0.0586	0.0047	0.0001	1.0925	0.0240	0.0808				
Outflow	(mg/L)	8.16	18.02	3.32	1.35	0.18	0.0018		1.15	2.866	0.221	0.263	13.76	0.195

Table 7. Material Budget for Amituk Lake, June 26-July 9, 1992

Constituent		Clean Snow	Dirty Snow	Influent Streams	Lake Outflow
H2O (m3x10^6)		2.989	2.989	2.065	2.146
H (keq)		1.278	0.141	0.018	0.015
Ca (keq)		45.9	470	1490	1932
Mg (keq)		41.3	78.7	446	587
Na (keq)		36.7	63.9	81.1	126
K (keq)		3.2	8.2	8.2	10.1
NH4-N (keq)		0.76	1.5	0.31	0.27
Alk (keq)		60.1	576	1822	2347
SO4 (keq)		12.2	19.8	33.6	51.5
NO3-N (keq)		6.8	8.6	3.9	4.3
Cl (keq)		58.2	107	104	174
SiO2 (kg)		8.4	71.7	469	475
DOC (kg)		225	1213	701	566
DIC (kg)		980	6372	22980	29566

Figure 1. Drainage areas in the Amituk Lake basin

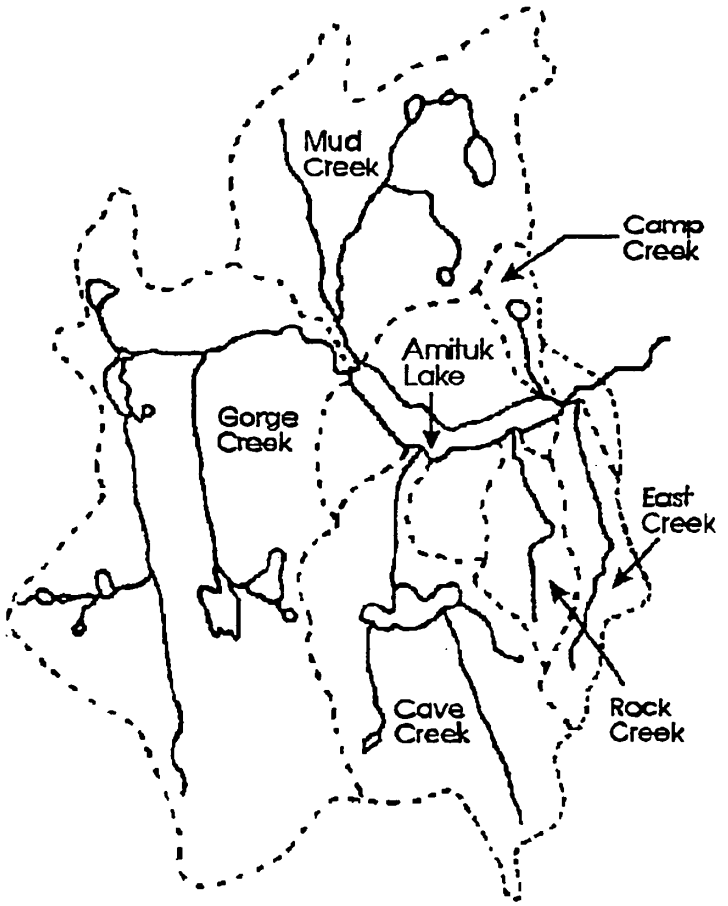


Figure 2. Morphometry of Amituk Lake

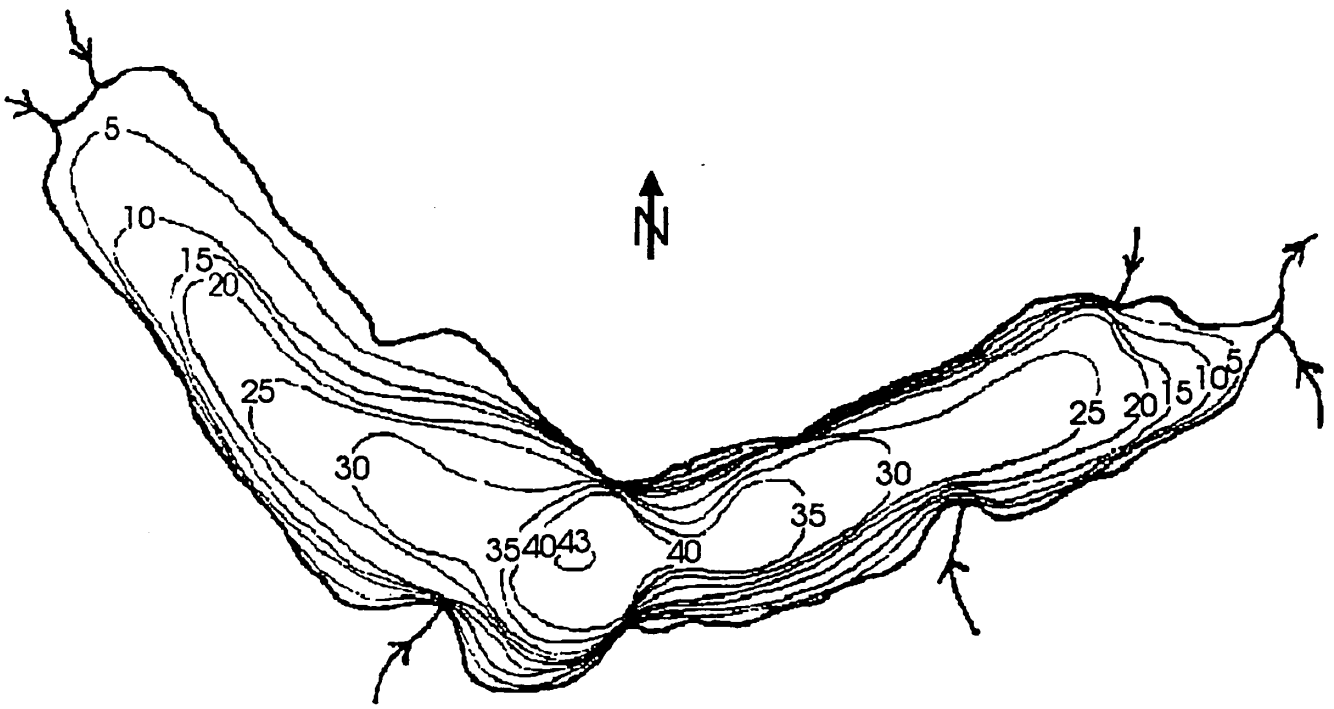


Fig 3. Temperature-Depth Profiles Amituk Lake 1992

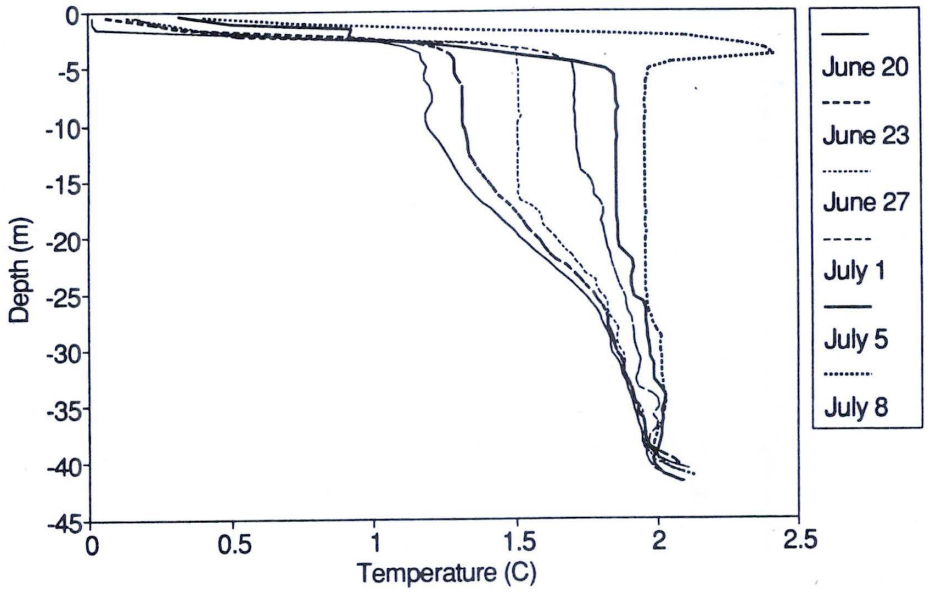


Fig 4. Discharge Summary of Streams Influent to Amituk Lake

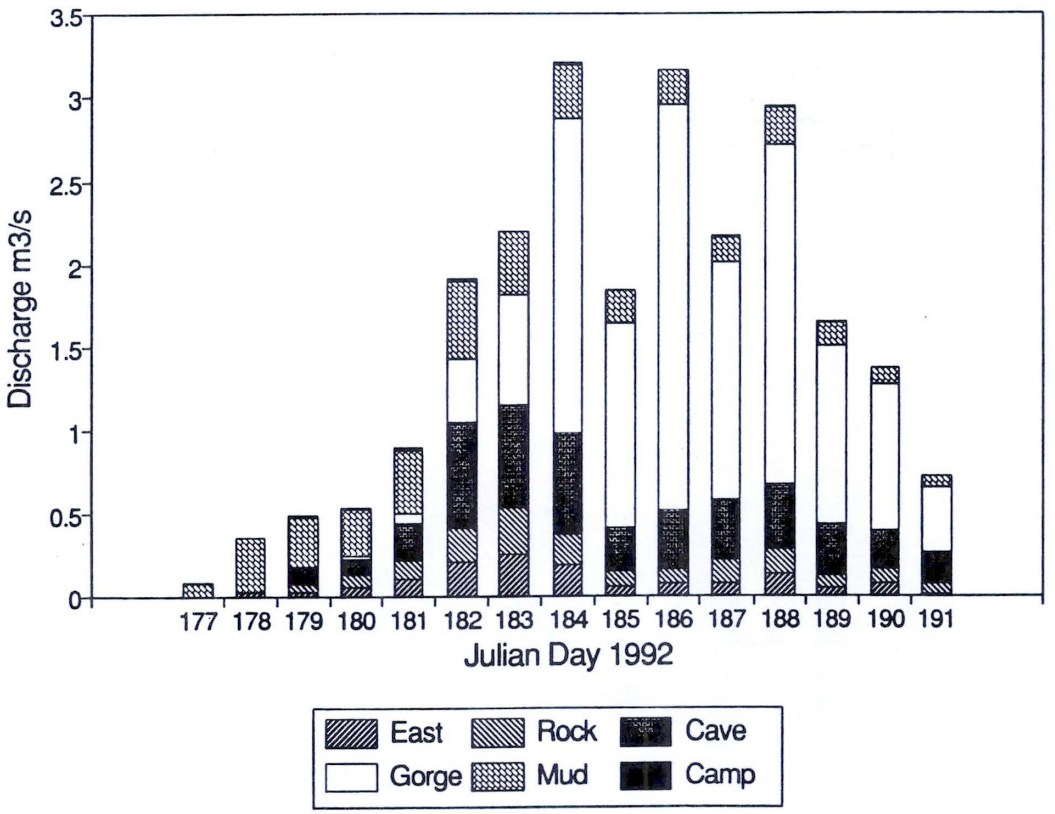
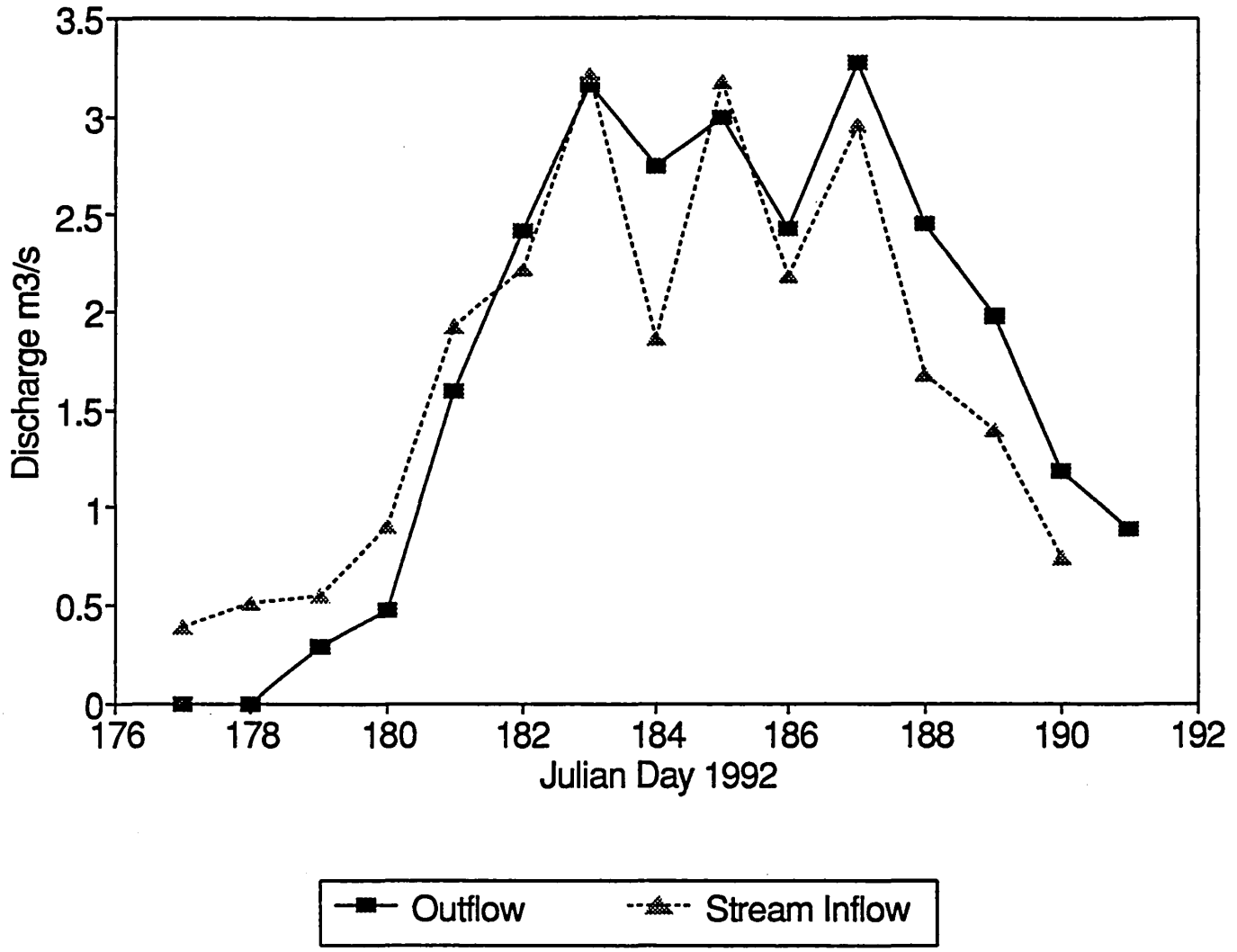


Fig 5. Comparison of Lake Outflow with Cumulative Stream Inflow



BASE FLOW WATER QUALITY AND CONTAMINANT LEVELS IN THE UPPER YUKON RIVER SYSTEM

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STUDY RATIONALE

Previous work has shown that fish in Lake Laberge contain toxaphene, PCBs and DDTs at levels higher than would be expected from long-range atmospheric deposition, based on comparison with analyses of fish from the Mackenzie River and other locations in Canada. Toxaphene is the pesticide of most concern, as it was also elevated in samples from Atlin Lake, and as the levels of toxaphene in fish in these two lakes resulted in health advisories to consumers. This study was developed to complement other studies being undertaken to define sources, pathways and sinks of contaminants in the upper Yukon River system.

OBJECTIVES

1. To describe water quality, including nutrients, metals, organochlorine pesticides and PCBs, under base flow conditions in the upper Yukon River drainage basin, including Lake Laberge.

2. To determine, through comparing water quality at points along the system at a time when river flow is dominated by groundwater, if significant contaminated groundwater inputs exist.

3. To describe water quality in surface waters potentially influenced by seepage from the abandoned Range Road Dump, and determine if the dump influences Yukon River water quality.

DESCRIPTION

Samples were collected in March, 1992 at the following stations (see Figure 1 for location of study area; stations are numbered from upstream to downstream:

1 Little Atlin Lake (flows into Atlin Lake)
2A Atlin Lake (depth: 1 m)

2B Atlin Lake (depth: 135 m)
3 Atlin River (flowing from Atlin Lake to Tagish Lake)
4 Nares River (flowing from Bennett Lake to Tagish Lake)
5 Tagish River (flowing from Tagish Lake to Marsh Lake)
6 Yukon River downstream of Marsh Lake
7 Schwatka Lake (reservoir in the Yukon River at Whitehorse)
8 Yukon River at Range Road Dump (2 samples combined)
9 McIntyre Creek backwater at Range Road Dump (the dump is located at the confluence of McIntyre Creek and the Yukon River)
10 Range Road Dump Pond (large pond at toe of dump, near McIntyre Creek)
11 Yukon River upstream of Takhini River confluence
12 Kusawa Lake (on the Takhini River drainage)
13 Takhini River near confluence with Yukon River
14A Lake Laberge (downstream of Takhini River confluence) (depth: 1 m)
14B Lake Laberge (depth: 128 m)

Samples collected for inorganic analyses were field filtered as required, preserved and shipped to the Conservation and Protection Laboratory, Environment Canada, North Vancouver, B.C., for analysis of total and dissolved metals, nutrients, major ions, and residues. Large volume samples (40 to 80 L) for organochlorine and pesticide analyses were extracted in Whitehorse and the crude extract was sent to the NWRI laboratory in Burlington for cleanup and fractionation. Fractions were analyzed using dual column GC/ECD. Details of field and laboratory methods, including numbers of replicates, and quality assurance/quality control procedures and results, will be provided in the report.

RESULTS

General Water Quality and Metals

Lake and river water samples were characterized by low levels of filterable and non-filterable residues, low hardness and alkalinity, and low nutrient levels (Table 1). Water was slightly alkaline throughout the system. Metal levels in the lake and river samples were generally very low, with few marked differences among the water courses (Table 2). With low levels of non-filterable residues, high proportions of the metals detected in water samples were in the dissolved form. Little Atlin Lake water quality differed significantly from other sites, with markedly higher levels of filterable residues, alkalinity, hardness, dissolved phosphorus, nitrogen, barium, and strontium, and with slightly higher sulphate, silicon, sodium and chloride. Little Atlin water quality would be influenced by drainage from the limestone formation along the northeast of this lake. The Takhini and Tagish Rivers were more turbid than other lake and river sites, and the Takhini River water was also higher in ammonia, sodium, silicon, aluminum, iron and titanium. Kusawa Lake had much lower concentrations of filterable residues, alkalinity and hardness than other sites.

The backwater of McIntyre Creek and the Range Road Dump Pond (Stations 9 and 10) differed from the lake and river sites, with much higher levels of filterable residues, colour, turbidity,

alkalinity, hardness, nitrogen, phosphorus, sulphate, sodium, chloride, and silicon, as well as several trace metals. The two stations were dissimilar in water quality, with the pond having higher levels of most chemical species. Water at both sites was alkaline, but the pond had the lowest pH measured (7.68), while the creek backwater had the highest (8.27). The most notably elevated metal was manganese at the dump pond. The more toxic metals (such as lead, mercury, copper and cadmium) were at low levels at both sites. Evaporation and relative contributions of groundwater and surface drainage to the pond and creek backwater, as well as seepage influenced by the abandoned dump, could affect water quality at these sites.

Organochlorines

Results of toxaphene analyses were considered ambiguous. While toxaphene was not detected using NCI/MS, quality assurance/quality control procedures showed poor recovery on spiked blanks. Methods have been improved, and toxaphene will be measured in 1993/94 water samples.

Relatively few organochlorines were detected (see Table 3). Only 1,2,4-TCB, HCB, α-HCH, γ-HCH (lindane), and total PCBs were found at all or most sites. Other compounds detected at several sites were: γ-chlordane, penta-CB, heptachlor epoxide and β-HCH. Aldrin, heptachlor and *o,p*-DDD were detected at one or two lake and river sites as well as at the dump pond (Station 10), while dieldrin, α-chlordane, 1,3,5-TCB, *p,p*-DDD, *p,p*-DDT and *p,p*-DDE were found only in the McIntyre Creek backwater and/or the dump pond. 1,2,3,4-TTCB was just above the detection limit at two lake sites. The following compounds were below detection in all samples (and thus were omitted from Table 3): nonachlor; 1,4-DBB; endosulfan, endosulfan I and II; methoxychlor; *o,p*-DDE; and *o,p*-DDT. See Table 4 for full names and uses of the organochlorine compounds listed above.

Figure 2 provides a graphical summary of the distribution of classes of organochlorine compounds in the samples (samples at surface and at depth for Little Atlin and Laberge lakes are averaged in this graph). Total HCH was the dominant class of organochlorine compound in most lake and river samples. HCHs were highest in Atlin Lake and the Yukon River upstream of the Takhini River, and were relatively low in the dump pond and McIntyre Creek samples, in the Nares and Takhini Rivers, and in Little Atlin Lake. PCBs were most abundant in water from the Takhini River and the dump pond. TCBs were found at similar levels at all sites. DDTs were detected only in the dump pond, the backwater of McIntyre Creek, the Takhini River and Lake Laberge.

As well as being higher in PCBs and DDTs than other sites, the dump pond contained several compounds not found in the other samples. These compounds could have originated from atmospheric deposition and then have become more concentrated in this pond. Contaminants accumulated in sediments may be more soluble in the stagnant, anoxic pond than in other waters sampled. The dump pond and any seepage from the dump to the Yukon River or to the much smaller McIntyre Creek did not have a measurable impact on Yukon River water quality (comparing Stations 7, 8 and 11).

A current source of PCBs and DDTs to Lake Laberge may be the Takhini River (based on results of the one sample from Station 13). Higher levels of these contaminants in the Takhini River may be related to the higher turbidity of this river, as these compounds may be transported as part of the particulate phase. It should be noted that PCBs in the Lake Laberge samples were not significantly higher than in samples from the Yukon River upstream (five Laberge samples ranged from 331 to 1068 pg/L, while PCBs in Stations 6, 7, 8 and 11 ranged from 81 to 1087 pg/L). DDTs were detected in only one of the five Laberge samples. It is clear that larger sample sizes would be required to determine if there are significant differences in PCBs and DDTs among these sites. Further sampling of organochlorines in the dissolved and particulate phases in the Takhini and Yukon River systems is being undertaken in 1993/94.

CONCLUSIONS AND UTILIZATION OF RESULTS

1. Base flow water quality is fairly consistent throughout the system, with low levels of dissolved solids, and no evidence of metals enrichment from human sources. Total nitrogen approximately doubles from Atlin Lake to Lake Laberge. Little Atlin Lake water quality differs from the rest of the system, being harder and higher in nutrients.

2. While water from a pond which would be influenced by seepage from the Range Road Dump does show signs of contamination, there was no measurable impact on Yukon River water quality from the abandoned dump.

3. There is no evidence from this dataset to suggest other current local sources of contaminants to the Yukon River and Lake Laberge.

4. These data will be compared with analyses of organochlorine compounds in snow in the study area when results are available; the data have been made available to researchers examining levels of contaminants in sediment and in aquatic biota in the Yukon.

5. More extensive water and suspended sediment sampling being carried out in 1993/94 in the study area is based on this program.

Expected project completion date: Report to be completed by August 15, 1993.

Table 1. Selected Water Quality Data.

LOCATION	turbidity FTU	Ca+Mg hardness mg/L	filterable residue mg/L	total nitrogen ug/L	total phosphor. ug/L	sulphate mg/L	chloride mg/L	sodium mg/L
Stn. 1	0.17	135	190	260	7	8.7	0.9	5.3
Stn. 2A	0.18	42	60	50	<2	4.8	<0.2	1.2
Stn. 2B	0.38	42	60	30	6	5.0	<0.2	1.1
Stn. 3	0.11	44	60	30	<2	5.9	<0.2	1.3
Stn. 5	1.8	34	50	50	4	2.9	0.3	1.3
Stn. 6	0.20	43	60	50	3	3.9	1	1.4
Stn. 7	0.28	44	60	30	3	6.1	0.2	1.5
Stn.8	0.50	46	60	50	10	5.4	<0.2	1.5
Stn. 9	3.3	111	160	180	33	8.1	0.7	4.6
Stn. 10	0.85	400	740	1300	27	59.2	87.9	82.3
Stn. 11	0.65	47	67	40	6	4.5	0.4	1.6
Stn. 12	0.16	10	20	60	2	1.6	0.3	1.0
Stn. 13	1.8	45	70	110	6	3.6	0.5	2.8
Stn.14A	0.19	43	60	110	5	5.1	0.4	1.6
Stn.14B	0.06	42	60	100	4	3.7	0.4	1.6
Control	0.06	<0.4	<10	40	<2	<0.5	<0.2	<0.1

Table 2. Trace Metals

2-A. Similar levels at all sites

Below or close to detection limits (det. limits in ug/L)	Low levels (total metal range in ug/L)
Ag (0.6), Be (1), Co (6), Cr (6), Cu(0.6), Mo (10), Ni (20), Sb (60), Se (60), Sn (60), V (10), Zn (2)	As (3.6 to 10.8), Hg (0.05 to 0.51), Pb (<0.6 to 2.1)

2-B. below detection or low levels at most sites; some higher levels.

Metal (detection limit or range of most measurements, in ug/L)	Sites with higher levels (mean concentration for site in ug/L)
Al (<60)	Stn. 6 (180); Stn. 10 (1040); Stn. 13 (230)
B (<10)	Stn 10 (250)
Ba (<10 to 32)	Stn. 1 (70); Stn 9 (42); Stn. 10 (157)
Cd (<0.1)	Stn . 9 (1.2)
Fe (27 to 109)	Stn. 9 (853); Stn. 10 (716); Stn. 11 (249)
Mn (<1 to 25)	Stn. 10 (1140)
Sr (40 to 92)	Stn. 1 (167); Stn. 9 (193); Stn. 10 (480)
Ti (<2)	Stn. 9 (60); Stn. 13 (14)

Table 3. Organochlorines detected in water samples (picograms/L)*.

	Stn1	Stn2 A	Stn2 B	Stn3	Stn4	Stn5	Stn6	Stn7	Stn8	Stn9	Stn10	Stn11	Stn12	Stn13	Stn14 A	Stn14 B
DIELDRIN											85					
α-CHLORD											170					
γ-CHLORD	27					25					227				89	51
PENTA-CB	14	25			25			26		27	26		24	28		17
ALDRIN	27										227					17
135-TCB											57					
124-TCB	169	398	569	427	223	331	191	438	251	191	85	107	584	363	368	468
HCB	14	25	25	25	25	25		77	25	27	26	27	24	28	25	25
HEPTACHLOR																34
HEPT. EPOX	14		74		74		163		201		365		122	307	216	34
α-HCH	531	2935	2649	2261	620	1858	1168	1005	1457	462	482	2842	1387	670	1903	1002
β-HCH		25	74	101							85		97		38	34
γ HCH	169	796	842	678	248	534	353	335	402	272	142	965	511	251	558	476
1234-TTCB								26								9
p,p-DDD										136	340					
o,p-DDD										27	368			56		26
p,p-DDE										163	57					
p,p-DDT										136						
sDDT										462	765			56		26
sPCB	329	50	274	50	670	484	1087	361	151	543	4676	81	316	3130	774	756

* 1 picogram (pg) = 1/1000 nanogram (ng) = 1/1,000,000 microgram (ug)

Notes: 1) Stations 1, 14A and 14B are means of replicate samples; 2) Blank cells indicate compound was below detection limit.

Table 4. Names and uses of organochlorine compounds.

ABBREVIATED NAME	FULL NAME	USE
DIELDRIN	dieldrin	insecticide
α-and γ-CHLORD	alpha and gamma chlordane	isomers of chlordane, a fumigant
PENTA-CB	pentachlorobenzene	fungicide
ALDRIN	hexachlorohexahydro dimethanonaphthalene	insecticide
135- and 124-TCB	1,3,5- and 1,2,4- trichlorobenzene	insecticides
HCB	hexachlorobenzene	fungicide
HEPTACHLOR	heptachlor	insecticide
HEPT. EPOX.	heptachlor epoxide	derivative of heptachlor
α-, β- and γ-HCH	alpha, beta and gamma hexachlorocyclohexanes	insecticides (γ HCH is lindane)
1234-TTCB	1,2,3,4-tetrachlorobenzene	insecticide
p,p- and *o,p*-DDD	para, para'-and ortho, para'-dichlorodiphenyl dichloroethane	metabolites of DDT
p,p- and *o,p*-DDE	para, para '- and ortho, para'-dichlorodiphenyl dichloroethylene	metabolites of DDT
p,p- and *o,p*-DDT	para, para '- and ortho, para'-dichlorodiphenyl trichloroethane	contact insecticide
sDDT	sum of DDDs, DDEs and DDTs	insecticide (DDTs) and metabolites (DDDs, DDEs)
sPCB	sum of polychlorinated biphenyls	dielectric fluid, other industrial uses
METHOXYCHLOR	para, para'-methoxychlor	insecticide
1,4-DBB	1,4-dibromobenzene	solvent for oils
ENDOSULFAN	endosulfan	insecticide
ENDOSULFAN I AND II	gamma and alpha endosulfan	isomers of endosulfan
NONACHLOR	trans - nonachlor	insecticide

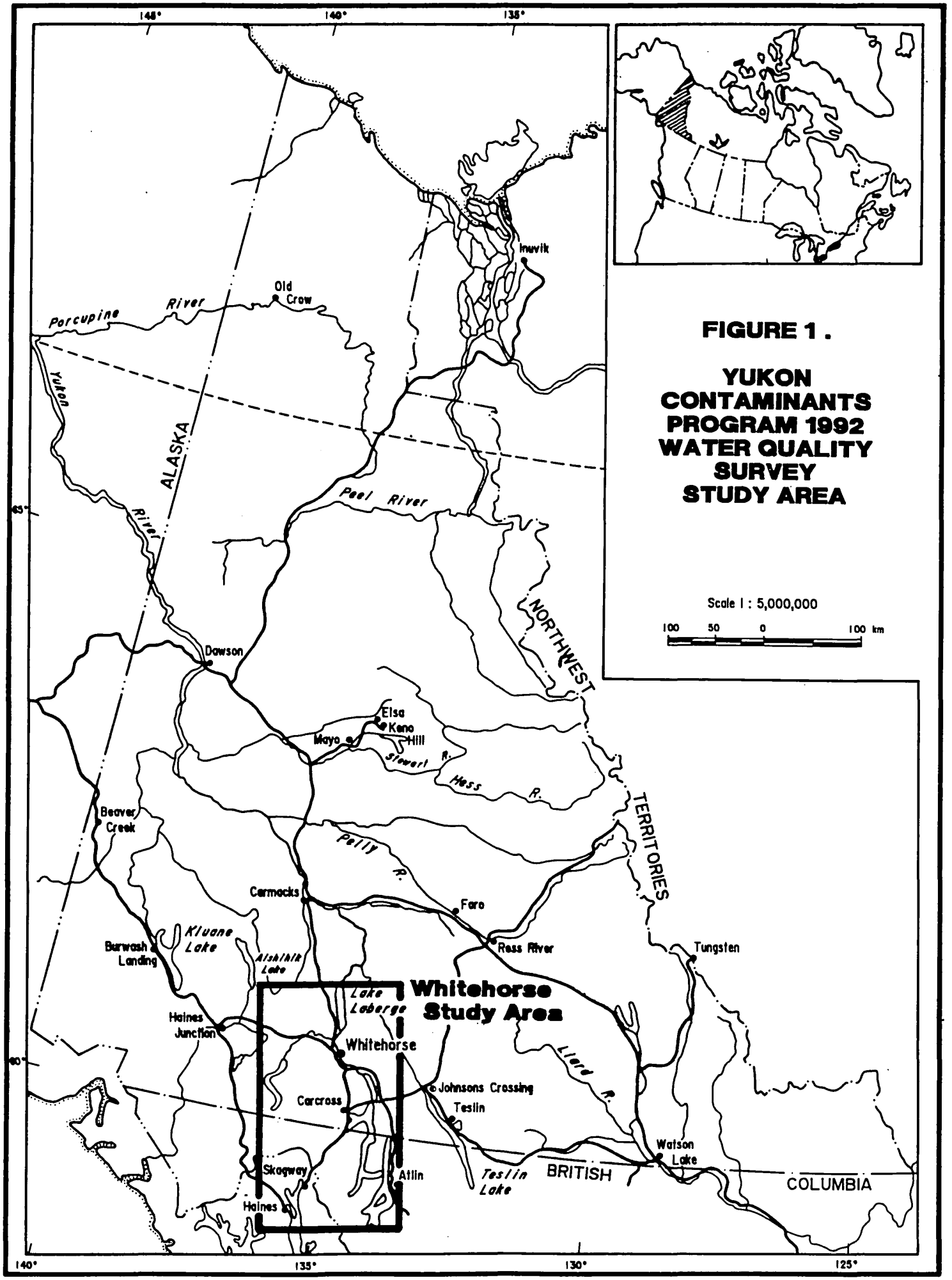
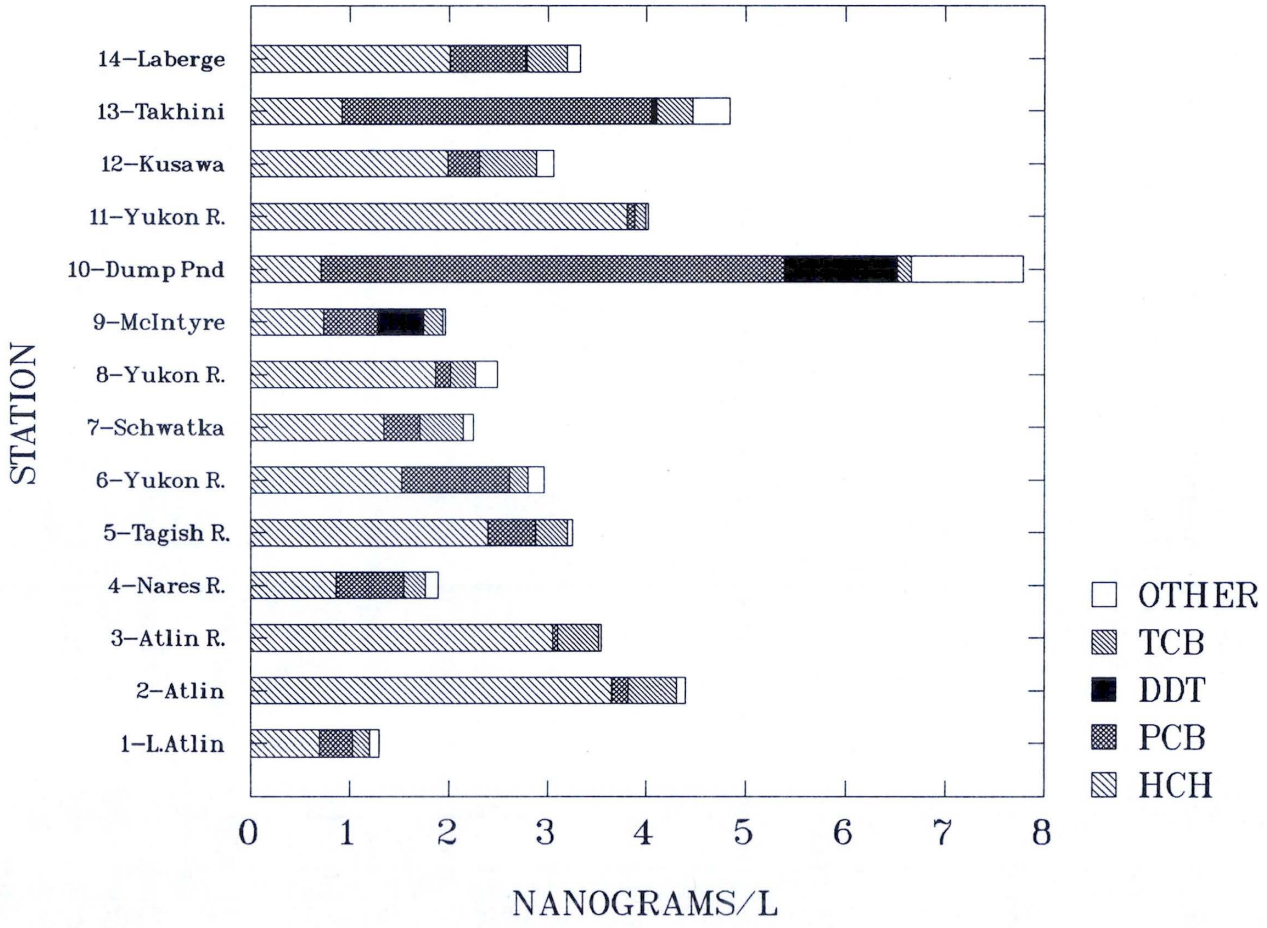


FIGURE 1.

YUKON CONTAMINANTS PROGRAM 1992 WATER QUALITY SURVEY STUDY AREA

FIG. 2. ORGANOCHLORINES IN WATER



HISTORICAL/ARCHIVAL RESEARCH INTO THE USE OF CONTAMINANTS IN THE YUKON

PROJECT LEADER: Yukon Technical Committee on Contaminants in Northern Ecosystems and Native Diets (Contact: Mark Palmer, Chair)

PROJECT TEAM: 1) Fisheries and Oceans Canada
2) Council for Yukon Indians
3) Environmental Protection, Environment Canada
4) Indian and Northern Affairs Canada
5) Yukon College

OBJECTIVES

Short-term:

1. To document the transport, use, and disposal of contaminants into and within the Yukon Territory.

2. To provide a record of oral accounts of the use and disposal of contaminants within the Yukon Territory.

Long-term:

1. To form a database concerning past contaminant use in the Yukon Territory.

2. The database will be invaluable in the planning of future contaminant projects and programs pertaining to both public and ecosystem health.

3. The database will be invaluable in the planning and assessment of government and private sector projects, programs, and policies.

DESCRIPTION/RESULTS

There is limited archival documentation of the transport, use, and disposal of contaminants into and inside the Yukon. Recent sampling of fish in the headwater lakes of the Yukon River determined that there were significant levels of selected contaminants in certain fish in all lakes for which analysis has been completed. Levels encountered indicate strongly that there are, or were, local sources of contamination. An inventory of transport, use, and disposal of contaminants is therefore considered essential. Archival documentation is expected to be incomplete, and interviews with those persons employed or otherwise active in sectors or industries where contaminants were known or likely to be used will be necessary.

The Yukon Contaminants Committee contracted a local firm (Laberge Environmental Services) to carry out an investigation into the use and transport of contaminants in the Yukon. The contract was carried out in cooperation with the Waste Clean-up Section of the Arctic Environmental Strategy.

The project was divided into three tasks as listed below:

1. Oral Interviews

Native elders and persons expected to be knowledgeable on the use, transport and disposal of contaminants in the Yukon between 1910 to 1980; with particular emphasis on the period 1940 to 1970. Elders from 9 of the 14 First Nations in the Yukon were interviewed as well as one First Nation from Northern British Columbia. Five First Nations were unable to participate in the study due to other commitments.

In addition, 25 non-native interviews were carried out with retired employees or individuals who would be able to provide information on the past use and disposal of contaminants in the Yukon.

2. Literature Archival Search

This was intended to provide background and follow-up information on issues identified throughout the interview process. Information was obtained through sources such as government files, Yukon Archives, corporate files, municipal records and old newspapers.

3. Arctic Environmental Strategy Waste Site Inventory

The three key objectives were to:

- determine if waste sites are abandoned;
- obtain relevant information on past owners/users; and
- determine past use of contaminants on the site.

UTILIZATION OF RESULTS

The final report will be used to assist in trying to explain the elevated levels of contaminants (such as PCBs) in southern Yukon lakes. It will also be used in the design of future monitoring programs within the Yukon.

Project completion date: The final report, entitled "The Use, Disposal and Transportation of Selected Contaminants in the Yukon, has been completed.

ECOSYSTEM CONTAMINANT UPTAKE AND EFFECTS

SOURCES AND SINKS OF ORGANOCHLORINES IN THE ARCTIC MARINE FOOD WEB

PROJECT LEADER: B. Hargrave, Fisheries and Oceans Canada

PROJECT TEAM: B. Hargrave, G. Phillips, W. Vass, and G. Harding, R. Conover (Fisheries and Oceans, Scotia Fundy Region, Dartmouth), H. Welch (Fisheries and Oceans, Central and Arctic Region, Winnipeg)

OBJECTIVES

1. To quantify the long-range atmospheric and marine transport of organic contaminants and their incorporation into lower trophic level organisms of the marine food web in the Arctic Ocean.

2. To provide baseline measurements of major semi-volatile organics (chlorinated pesticides, PCBs) in the Canadian high Arctic Ocean environment by sampling seawater (dissolved and particulate phases), plankton, benthos and fish.

3. To assess the relative importance of atmospheric versus oceanic input of these contaminants to Arctic Ocean biota by seasonal measurements.

4. To evaluate the bioconcentration of these compounds for comparison with data from more southern latitude ocean sites to assess input of organochlorines to food webs utilized as food by native populations.

DESCRIPTION

Although chlorinated hydrocarbon pesticides (OCs) and polychlorinated biphenyls (PCBs) are produced and used primarily in temperate and tropical latitudes, long-range atmospheric transport (Oehme and Ottar 1984, Barrie 1986, Patton *et al.* 1989), as well as surface ocean currents and river drainage, have introduced these compounds into the Arctic and other ocean basins. The semi-volatile nature of most of these compounds and their resistance to photolysis or biodegradation has resulted in a global distribution with OC pesticide residues and PCBs detectable in marine plankton from a wide variety of oceanic areas (Harding 1986).

Many OCs have a low water and high lipid solubility. Bioaccumulation (the partitioning of compounds between an aqueous phase and tissues of organisms) should be greatest in small-bodied aquatic organisms where equilibrium conditions are established between internal lipid pools and external ambient OC concentrations (Bidleman *et al.* 1989). In most aquatic animals, however, uptake of these organic contaminants occurs through food ingestion.

Bioconcentration (calculated as the ratio of concentrations for predators over their prey) occurs in food webs where predators store the ingested compounds in lipids with slow rates of excretion or metabolism (Woodwell *et al.* 1971, Harding 1986). Bioconcentration should be favoured in arctic marine ecosystems where there is a summer maxima of lipid synthesis and storage.

While the present distribution and mechanisms for transfer of OCs between various compartments of the arctic marine environment and food web are poorly understood, it is known that concentrations of organic matter (particulate and dissolved) in the Arctic Ocean are extremely low (Gordon and Cranford 1985). Since there is no photosynthesis during months of darkness, marine organisms must store energy-rich lipids to survive. The high fat content of tissues used for lipid synthesis and storage make these organisms more likely to accumulate various OCs that have a high fat solubility. This in turn may result in a greater bioconcentration of these contaminants by higher trophic levels than occurs in more temperate regions. The dynamics of accumulation and storage of potentially toxic OCs has human relevance in the Canadian Arctic where native people rely on marine mammals and fish for a large portion of their protein and caloric intake.

Published observations of the distribution of OCs in the Arctic Ocean are restricted primarily to measurements of concentrations in the atmosphere and tissues from large mammals (Muir *et al.* 1992). There is little published data for organisms from lower trophic levels of the marine food web. Studies carried out from the Canadian Ice Island (1986-89) and currently underway in Barrow Strait off Resolute Bay (1993) provide samples for measurements of OC pesticides and PCBs in various lower trophic level marine organisms in the Arctic Ocean. Samples collected from an ice camp on seasonal ice and from the Canadian Ice Island Station have been analyzed and results summarized in recent publications (Hargrave *et al.* 1992; Muir *et al.* 1992). The distribution and relative concentrations of various OCs and PCBs in air, snow, ice and water phases sampled from the Ice Island were reported earlier (Hargrave *et al.* 1988). OC pesticides and PCBs detected in these physical compartments were also present in particulate matter melted from the lower ice surface, various size classes of zooplankton, pelagic and benthic amphipods and liver tissue from the glacial eelpout. Preferential uptake of specific compounds by different crustacean species results in lipid concentrations of OCs in fauna from lower trophic levels as great as those measured in seal blubber and polar bear liver.

ACTIVITIES IN 1992/93

A 12-month over-wintering sampling program for organochlorines was initiated using DFO's facilities at South Camp, Resolute Bay, NWT. The project began with the first sampling trip in January 1993 and has continued since then with monthly sample collections of seawater, planktonic and benthic crustaceans and fish. Collections of samples of OC determinations are co-ordinated with DFO studies to measure biomass and productivity of plankton under the ice and in the water column. Observations of hydrographic variables (temperature, salinity, dissolved nutrients), chlorophyll *a*, zooplankton biomass and species composition are made at monthly or more frequent intervals.

The start-up date for work in Barrow Strait was advanced to January 1993 from the planned date of September 1992 to allow food, laboratory supplies and equipment to be shipped to Resolute Bay by sea-lift. This also ensured that ice was sufficiently thick to allow establishment of a research base-camp on the ice. Monthly sampling of organochlorines in seawater and lower trophic level organisms commenced in January 1993 using a 1 m^2 ice hole covered by a heated sampling hut placed approximately 6 km off Cape Martyr in Barrow Strait. A local Inuit technician (P. Amarualik) has been hired and trained to assist with sampling and sample processing. Sampling collection will continue at monthly intervals until December 1993.

Several publications appeared during the year:

Barrie, L.A., D. Gregor, B. Hargrave, R. Lake, D. Muir, R. Shearer, B. Tracey and T. Bidleman. 1992. Arctic contaminants: sources, occurrence and pathways. Sci. of the Total Environ. 122: 1-74.

Hargrave, B.T., G.C. Harding, W.P. Vass, P.E. Erickson, B.R. Fowler and V. Scott. 1992. Organochlorine pesticides and polychlorinated biphenyls in the Arctic Ocean food web. Arch. Environ. Contam. Toxicol. 22: 41-54.

Muir, D.C.G., R. Wagemann, B.T. Hargrave, D.J. Thomas, D.B. Peakall and R.J. Norstrom. 1992. Arctic marine ecosystem contamination. Sci. of the Total Environ. 122: 75-134.

Phillips, G.A. and B.T. Hargrave. 1992. Intercalibration of organochlorine residues in biota. Can. Data Report Fish. Aquat. Sci. 879, 65 pp.

A contribution to the Toxaphene Workshop held in Burlington Ontario in January 1993 has been submitted for publication as a primary paper in Chemosphere:

Hargrave, B.T., D.C.G. Muir and T.F. Bidleman. Toxaphene in amphipods and zooplankton from the Arctic Ocean. Submitted to Chemosphere.

RESULTS

a) **Seasonal sampling** is underway in Barrow Strait as planned. Due to the delayed start-up date, results of quantitative analyses from the first samples are not yet available. A contract for analyses of OCs in samples collected in January 1993 was prepared in March 1993.

b) **Distribution of OCs** in lower trophic level organisms from the Arctic Ocean collected between 1986 and 1989 from the Canadian Ice Island was summarized in Hargrave *et al.* (1992) and Muir *et al.* (1992). Polychlorinated biphenyls (PCBs), polychlorinated camphenes (PCCs) and isomers of DDT and DDE were the predominant organochlorine hydrocarbons measured in epontic particulate matter, zooplankton, pelagic and benthic amphipods and liver tissue from a glacial eelpout collected at 79°N off Axel Heiberg and

Ellef Ringnes Islands and 85°N over the Alpha Ridge in the Arctic Ocean. Chlordane, dieldrin and other cyclodienes and hexachlorocyclohexane isomers were present at lower concentrations. Levels on a dry weight basis in plankton of various sizes <65 μm to 2 mm were similar to those in epontic particulate matter, but on a lipid weight, concentrations in smaller sized plankton were from two to five times higher.

Organochlorines in amphipods and liver from the glacial eelpout *Lycodes frigidus* exceeded levels in zooplankton by up to an order of magnitude. Large benthic lysianassid amphipods (*Tmetonyx cicada, Anonyx nugax* and *Eurythenes gryllus)* accumulated higher concentrations than small species (*Onisimus* spp. and *Andaniexis* spp.) or the under-ice gammaridean amphipod (*Gammarus wilkitzkii*).

No significant differences in organochlorine concentrations were measured in benthic amphipods collected at different times of the year from the Ice Island. A more complete seasonal collection of samples in Barrow Strait during 1993 will allow a more complete evaluation of seasonal changes. However, concentrations of OCs in lipids of small zooplankton (<63 μm) collected from the Ice Island decreased between May and August. Large zooplankton (>500 μm) collected in August, dominated by adult copepods (*Calanus hyperboreus* and *Calanus glacialis*) and ctenophores (*Oikopleura vanhoeffeni* and *Beroe cucumis*), contained concentrations of α-HCH, chlordane isomers and other cyclodienes that were two to four times higher than levels in May. Biomagnification factors calculated for presumed predator-prey links in the food web varied over two orders of magnitude for different organochlorines. Ratios between epontic particulate matter and plankton were generally lower (1-10) than values derived for trophic links between amphipods, fish and marine mammals (10-100). Results from the 1993 study in the much more productive area of Barrow Strait can be compared with those from more oligotrophic waters of the Arctic Ocean sampled from the Ice Island.

c) **Interlaboratory OC calibration study** carried out in 1990 was published during the year (Phillips and Hargrave 1992). Biological samples (zooplankton and amphipods from the Arctic Ocean) were analyzed for 18 different OCs and PCBs by four different laboratories. Results for seven major OC compounds or groups were tabulated and compared between pairs of laboratories. Toxaphene measurements which were carried out as part of the intercalibration study provided data that was included in the primary paper (Hargrave *et al.* 1993) submitted for publication in the proceedings of the Toxaphene Workshop in Burlington, January 1993.

CONCLUSIONS AND UTILIZATION OF RESULTS

Five classes of compounds (hexachlorocyclohexanes, cyclodienes, isomers of DDT and its metabolites and congeners of PCBs and toxaphene (PCCs)) have been detected in under-ice epontic particulate matter and tissue samples of marine biota from lower trophic levels of the Arctic Ocean (Table 1). PCCs, PCBs, isomers of DDT and DDE, chlordane, dieldrin, alpha-endosulfan, HCB and α-HCH were present in quantifiable levels. Traces of β- and γ-HCH and the cyclodienes aldrin, endrin, heptachlor, heptachlor epoxide, methoxychlor and

mirex were present but small sample sizes for epontic particles and various zooplankton size fractions did not allow concentrations to be determined.

The OCs in samples of epontic particles, pelagic and benthic crustaceans are also present in the Arctic atmosphere, particulate and dissolved fractions of snow, ice melt water and seawater samples collected from the Ice Island between 1986 and 1988 (Hargrave *et al.* 1988). The data show that small-bodied organisms which are short-lived generally have a lower lipid content for storage of OCs than larger sized animals. The large storage potential for OCs in lipids of fish, seal and mammal tissues, where lipid content may amount to < 50% of tissue fresh weight contrasts average values for lipid expressed as a percentage of wet weight in benthic amphipods (6.4%), zooplankton (3.1%) and epontic particles (< 1%) melted from the under-surface of the ice.

Plans for 1993/94 include the completion of the seasonal sampling in Barrow Strait to determine the fate of OCs added to the Arctic Ocean during the melt period in July and August with subsequent changes during ice formation and freeze-up in late 1993. Seasonal differences in bioaccumulation of OCs are expected due to changes in marine food web productivity as the source of primary production shifts from under-ice epontic algae to phytoplankton during the open water period in Barrow Strait between July and September. The sampling program in Barrow Strait provides the first opportunity for seasonal measurements of the major OCs in the Canadian arctic marine environment. The study will provide additional information for mass budget calculations to assess the relative importance of atmospheric vs. ocean sources.

Expected project completion date: March 31, 1997

Partners: DOE staff (T. Bidleman and L. Barrie (AES) and D. Gregor (IWD)) have provided advice and collaboration on measurements of toxaphene and interpretation of atmospheric and runoff sources of other OCs.

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Table 1. Concentrations on a dry weight basis of various organochlorines (single measures, means (n=4 to 27) with one standard deviation (sd) or ranges for 2-3 determinations) in epontic particulate matter and tissues of biota from lower trophic levels in the Arctic Ocean. nm indicates that the compound was not measured.

Compound	Epontic particles	Zooplankton			Amphipods[1]			*Lycodes frigidus* liver (ng/g dry wt)
		Size fraction (μm)			(ng/g dry wt.)			
		<63	125-250	500	PL	BL	EG	
PCCs	nm[2]	nm	12-39[3]	nm	nm	1085[3] (912)	4182 (252)	nm
PCBs	<2-36	35 (24)	60 (43)	20 (12)	73	1142 (461)	5334 (3107)	1290
s-DDT[4]	<2-46	30 (16)	31 (24)	20 (10)	56	394 (98)	4811 (2472)	811
s-CLD[4]	<1-18	26 (11)	15 (2)	19 (13)	73	173 (84)	642 (513)	32
DIELD	<1-7	10 (4)	6 (4)	4 (<1)	34	14 (3)	32 (40)	7
α-HCH	<1-2	21 (3)	24 (8)	16 (13)	73	31 (9)	21 (18)	3
HCB	<1-8	5 (<1)	10 (2)	3 (1)	29	33 (14)	37 (20)	34

[1] Amphipod species: PL - *Gammarus wilkitzkii*
BL - *Anonyx nugax, Andaniexis spp., Onisimus spp., Tmetonyx cicada*
EG - *Eurythenes gryllus*

[2] nm indicates that the compound was not measured

[3] reported in Bidleman *et al.* (1989)

[4] s-CHL=sum of *cis*+*trans*-Chlordane + Oxychlordane + *cis*-Nonachlor + Heptachlor epoxide;
s-DDT=sum of *p,p'*-DDT + *o,p',p,p'*-DDE

SPATIAL AND TEMPORAL TRENDS OF ORGANOCHLORINES IN ARCTIC MARINE MAMMALS

PROJECT LEADER: D. Muir, Fisheries and Oceans Canada, Freshwater Institute, Winnipeg, Manitoba

PROJECT TEAM: D. Muir, M. Segstro, B. Grift, C. Ford, R. Stewart, S. Innes and G. Stern

OBJECTIVES

1. To determine temporal and spatial trends in PCBs and other organochlorines in arctic marine mammals on a circumpolar basis, with special emphasis on beluga whales, ringed seals and walrus.

2. To provide data for use in surveys of dietary contamination by circumpolar countries and for use by the Arctic Monitoring and Assessment Programme (AMAP).

DESCRIPTION

People in arctic coastal communities consume marine mammals as part of their traditional diets. Information is needed to evaluate current risks of exposure to PCBs via consumption of skin, blubber and other tissues. Surveys have shown the presence of a wide range of organochlorine contaminants in arctic marine mammals throughout the Arctic. There is relatively complete data on organochlorine levels in ringed seals and beluga in the Canadian Arctic (especially for N. Quebec, W. Hudson Bay, Baffin Island, and the southern part of the Arctic archipelago) for samples collected in the mid-1980's. There is little information on geographic trends in other pinnipeds, e.g. Bearded, Harp and Harbour seals and walrus. Information is also limited on variations of contaminants with age and sex because of small sample sizes analysed from most locations, and on temporal trends at all locations except Holman Island where R. Addison has reported results beginning in 1972. This project is a combination of two from 1991/92 - Spatial Trends in Ringed seals and Walrus and Circumpolar Survey of PCBs in Beluga. Both had similar objectives so they were combined for 1992/93.

The specific objectives for 1992/93 were (1) to follow up findings of high PCBs and organochlorine pesticide residues in samples of walrus blubber from eastern Hudson Bay by obtaining more samples from northern Québec, (2) to begin additional studies on ringed seals from Arviat and other locations to examine variations with age and sex, (3) to determine PCBs and organochlorine pesticide residues in samples of beluga kidney and mattak from western Greenland animals (in collaboration with M.-P. Heide-Jørgensen and R. Dietz, Greenland Fisheries Research Institute, Copenhagen), (4) to continue attempts to collect beluga samples on a circumpolar basis, and (5) to complete the isolation of individual PCC

components from beluga blubber in order to identify them and develop a better method of quantifying toxaphene.

ACTIVITIES IN 1992/93

Samples

Kidney and muktuk samples from beluga previously received from R. Dietz (Greenland Fisheries Research Institute) were analysed for OC contaminants. Ringed seal samples were collected from Resolute (B. Welch, DFO), Pangnirtung (R. Stewart) and Arviat (R. Mulders, NWT Natural Resources, R. Stewart and I. Stirling, Canadian Wildlife Service). Twenty-two ringed seals from Arviat, collected in fall of 1991, were analysed. Samples of narwhal and beluga blubber were obtained from the hunt at Creswell Bay (Somerset Island) with the help of B. Welch and R. Stewart from those animals that were also analysed for hepatic mixed function oxidase enzyme activity. Nine beluga whale blubber samples from Norton Sound were obtained from S. Wise (National Institute of Standards and Technology (NIST), Gaithersburg, Maryland) as part of an interlab calibration exercise. Walrus and ringed seals were obtained by hunters at Inukjuak with the help of biologists with Makivik Corporation, Kuujjuak.

Methods

All animals were aged by counting growth layers (reference). Samples of blubber were analysed for 90 PCB congeners and 40 other organochlorine (OC) compounds (see attached list Table 1).

The extraction and separation of analytes from lipid coextractives were the same as those described in Muir *et al.* (1990). Blubber extracts were chromatographed on a GC-ECD with a 60m x 0.25 mm DB-5 column using H_2 carrier gas. GC conditions are described in previous studies (Muir *et al.* 1988). Total PCB (ΣPCB) was the sum of all congeners. Total chlordane (ΣCHLOR) was the sum of all chlordane-related compounds including heptachlor epoxide while total DDT (ΣDDT) was the sum of 4,4'- and 2,4'-DDE, -DDD, -DDT isomers. Polychlorinated camphenes (PCCs) were quantified using a single response factor based on 19 peaks in the standard (obtained from USEPA repository, Cincinnati, Ohio).

The two major toxaphene components were isolated from a 2 kg sample of beluga blubber from the Western Hudson Bay population (Stern *et al.* 1992). Briefly: blubber was blended with hexane and the lipid separated from toxaphene and other OCs by dialysis in "layflat" polyethylene tubing. Toxaphene components were separated from PCBs by chromatography on Florisil columns and further separation was made by reverse-phase HPLC. Identities of the major peaks were confirmed by mass spectrometry and by ^{1}H-NMR. GC-electron capture negative ion mass spectrometry (ECNIMS) was performed on a VG-7070E-HF double focusing mass spectrometer and ^{1}H NMR spectra were recorded with a Bruker AMX500 spectrometer, operating at a frequency of 500 MHz.

Quality Assurance

Internal standard recoveries (aldrin and octachloro-naphthalene) were uniformly greater than 90%. The Cod liver standard reference material (SRM-1588) from NIST (Gaithersburg, MD) was used as a laboratory control sample for major organochlorine pesticides and PCB congeners. The laboratory continued to take part in an ICES (International Council for Exploration of the Sea) interlaboratory comparison of PCB congeners in seal blubber and in an intercomparison with NIST on whale blubber.

RESULTS

Almost all of the 130 organochlorine compounds listed in Table 1 could be detected in whale and seal blubber except as noted in the footnote.

The analysis of 5 additional walrus blubber samples from Inukjuak confirmed the elevated levels in animals from this location reported last year (Muir 1992). Levels of ΣCHLOR, ΣDDT and ΣPCB in walrus were similar to those reported in narwhal, beluga and polar bear fat and much higher than reported by Born *et al.* (1981) in blubber of walrus from western Greenland or in walrus we previously analysed from Igloolik, Hall Beach, Akulivik and Loks Land (Muir 1992). Our tentative conclusion, backed up by analysis of stable isotopes of N and C in walrus, seal and clams, is that these walrus are feeding on ringed seals for a significant portion of their diet. Consumption of seal by walrus has been observed but it has been thought to be limited to a small number, perhaps 10%, of rogue animals. In discussions with walrus experts at a recent workshop held in Winnipeg, it was suggested that the higher proportion of seal consumption in the Inukjuak animals may be learned behavior.

Narwhal and beluga from Creswell Bay in 1991 had similar levels and relative proportions of ΣDDT, ΣPCB and toxaphene to those we observed in previous studies of samples collected in the period 1982 to 1986 (Muir *et al.* 1992). Toxaphene was the major organochlorine present in narwhal and beluga blubber (Table 2). The major organochlorines were present at similar concentrations in beluga from Norton Sound (Bering Sea) to those in beluga from Creswell Bay. Levels of all 5 major OC groups were similar to those we reported previously in Beaufort Sea beluga (Muir *et al.* 1990).

During 1992 we completed the identification of the two major toxaphene components in beluga (and narwhal) blubber. T2 was confirmed by ^{1}H NMR and MS as 2-exo,3-endo,5-exo,6-endo,8,8,10,10-octachlorobornane and T12 as 2-exo,3-endo,5-exo,6-endo,8,8,9,10,10-nonachlorobornane. T2 and T12 were identical structurally except for the presence of the additional chlorine on the bridge methyl group. Further details on the identification are given in Stern *et al.* (1992).

Table 1. List of PCBs and other organochlorine compounds determined in marine mammal blubber samples by DFO Winnipeg.

PCBs	# Cl	PCBs	# Cl	PCBs	# Cl	Other OCs	Group
3[1]	1	56/60	4/4	178/129	7/6	1245-Cl_4benzene	CBz
4/10[1]	2	91	5	175	7	1234-Cl_4benzene	CBz
7[1]	2	84/89	5	187	7	Cl_5-benzene	CBz
6[1]	2	101	5	183	7	Cl_6-benzene	CBz
8/5	2	99	5	128	6	α-HCH	HCH
19	3	83	5	185	7	β-HCH	HCH
18	3	97	5	174	7	γ-HCH	HCH
17	3	87	5	177	7	Chlordane "C"	Chlordane
24/27	3	85	5	171	7	heptachlor	Chlordane
16/32	3	136	6	156	6	Cl_8-sytrene	
26	3	110	5	201/157	8/6	Chlordane-C1B	Chlordane
25	3	82	5	172/197	7/8	Chlordane-C2	Chlordane
31	3	151	6	180	7	Chlordane-C3	Chlordane
28	3	144/135	6	193	8	Chlordane-C5	Chlordane
33	3	149	6	191	8	Nonachlor-III	Chlordane
22	3	118	5	200	9	oxychlordane	Chlordane
45	4	134	6	170	8	t-chlordane	Chlordane
46	4	114	5	190	8	c-chlordane	Chlordane
52	4	131	6	198	8	t-nonachlor	Chlordane
49	4	146	6	199	8	c-nonachlor	Chlordane
47	4	153	6	196/203	8/8	hept. epoxide	Chlordane
48	4	132	6	189	8	dieldrin	
44	4	105	5	208	9	endrin	
42	4	141	6	195	8	p,p'-DDE	DDT
41/71	4	130	6	207	9	p,p'-DDD	DDT
64	4	141	6	194	8	p,p'-DDT	DDT
40	4	130/176	6/7	205	8	o,p'-DDE	DDT
74	4	179	7	206	9	o,p'-DDD	DDT
70/76	4	137	6	209	10	o,p'-DDT	DDT
66	4	138	6			mirex	
95	5	158	7	Other OCs		photomirex	
				toxaphene[2]		Cl_5-anisole	
				α-endosulfan[1]		methoxychlor[1]	

[1]These compounds were not detected in marine mammal tissues.
[2] 19 chlorobornane peaks were monitored, generally T2 and T12 predominated.

Table 2. Concentrations (arithmetic means ± SD) of major organochlorine groups in arctic marine mammal tissues analysed during 1992/93.

Species/Location	Tissue	Sex	N	ΣHCH	ΣCHLOR	ΣDDT	ΣPCB	toxaphene
Beluga								
Nuussuaq	Kidney	M	10	12±7.3	120±42	160±42	169±60	454±219
		F	10	4.8±2.7	53±21	53±23	75±38	203±115
Nuussuaq	Muktuk	M	10	27±15	322±175	396±258	323±265	1350±704
		F	10	31±29	384±242	462±293	411±178	1680±1080
Creswell Bay	Blubber	M	2	247	2720	2520	3540	9060
		F	1	75	1120	1240	1730	4090
Norton Sound	Blubber	M&F	9	348±72	2020±429	2610±895	3830±1000	4180±823
Narwhal								
Creswell Bay	Blubber	M	8	85±17	1990±381	3230±871	3200±765	7930±2410
Walrus								
Inukjuak	Blubber	M	4	267±149	6290±4240	4610±3160	11460±8450	3490±2440
		F	9	214±114	2750±1780	2160±924	4760±2370	1450±954
Ringed seal								
Arviat	Blubber	M	13	325±132	1500±1110	1430±970	1760±1200	1100±2970
		F	9	280±214	641±390	656±310	846±310	181±78

ΣPCBs, toxaphene, ΣDDT, and Σchlordane were the major organochlorine contaminants found in beluga kidney and mattak from western Greenland (Table 2). This was expected because these compounds were also the most prominent in blubber of the same animals. Toxaphene was present at the highest concentrations of all organochlorines ranging (in males) from 454 ng/g (wet wt) in kidney to 1350 ng/g (wet wt) in mattak. Concentrations of all major organochlorine groups in mattak from females were identical to those of males (Table 2), but in kidney females had about 2-fold lower levels than males (Table 1). Males and females under 4.5 years of age also had identical levels of organochlorines in blubber so the differences between sexes for levels in kidney was unexpected. It may be due to differences in lipid content of the kidney. Extractable lipids averaged 2.7% in females and 4.2% in males.

Some of the variation in concentrations of the organochlorines in kidney and mattak was due to age. Significant (at $p < 0.05$) declines were observed in ΣPCB and ΣDDT levels in kidney with age in females (DDT vs age, $r^2 = 0.47$; PCB vs age, $r^2 = 0.42$). The same trend was also observed in males but only for the 0-2 yr age group. The declines probably reflect clearance of ΣDDT and ΣPCBs following elevated exposure of the young animals via nursing and growth dilution. Statistically significant declines were not observed for toxaphene levels in kidney nor for any organochlorine in mattak.

CONCLUSIONS AND UTILIZATION OF RESULTS

Results of the project during 1992/93 have added to the limited information available on organochlorine contaminants in kidney and mattak. The comparison with other mattak samples from Broughton Island suggests that organochlorine levels in mattak will vary directly with lipid content. The results also indicate that concentrations in mattak and kidney can be predicted from levels in blubber if the content of neutral lipids (i.e. hexane extractable) in the tissues is known. The analysis narwhal blubber gave us the opportunity to look at temporal trends in this species for the first time (over a 9 year period). Levels of major OCs do not appear to have declined significantly in males. But further assessment, taking into account possible differences in age, is needed. Ongoing collections of beluga from SE Baffin Island and ringed seals from Resolute and Western Hudson Bay will make possible additional temporal trend studies. Future work for this project include (1) analysis of marine mammal samples from the Barents and White Sea, provided that contacts can be maintained with Russian scientists, (2) analysis of additional seals from Arviat to thoroughly examine variations with age, blubber thickness and sex in one population, (3) further temporal and spatial trend studies to include seals from Resolute, Grise Fiord and possibly Eureka, (4) further interpretation of results of organochlorines in Greenland beluga tissues and in walrus via preparation of papers on these topics, and (5) isolation of additional toxaphene components from seal blubber.

Expected project completion date: March 31, 1997

Partners: Inuit Circumpolar Conference (Nuuk, Greenland), Greenland Fisheries Research Institute, NOAA (Anchorage, Alaska), Makivik Corp., Kuujjuak Qc., National Institute of Standards and Technology (Gaithersburg Va), Collaboration with Murmansk Institute of Marine Biology and other Soviet scientists.

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LONG-TERM TRENDS IN ORGANOCHLORINE RESIDUES IN EASTERN AND WESTERN ARCTIC SEAL BLUBBER

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PROJECT TEAM: R.F. Addison, T.G. Smith (Pacific Biological Station, Fisheries and Oceans Canada, Nanaimo, BC)

OBJECTIVES

In general, to measure trends in levels of PCBs, the DDT-group and other organochlorines from the early 1970's to the present in Arctic seals.

RESULTS

Between 1991 and 1993 a suite of Western Arctic ringed seal blubber samples were analysed which were collected in 1989 from a Holman Island population that has been studied extensively by Fisheries and Oceans Canada. Some blubber samples from a 1981 sample of the same population were re-analysed to ensure that results obtained by different analysts using more modern methods were comparable. The results have confirmed previous conclusions that PCB residue concentrations seem to be declining fairly rapidly in this population, whereas DDT-group concentrations are declining only slowly. Thus, between 1972 and 1989, PCB concentrations declined by about 80%, whereas *p,p'*-DDE declined by only about 40%. Since in east coast seals, *p,p'*-DDE has declined much more rapidly than PCBs, and since PCBs have declined about equally rapidly in both Arctic and east coast seals, there seems to be a continuing supply of DDT-group residues to the western Arctic. HCB concentrations in these seals have declined over the interval 1981-1989, but α-HCH (the major HCH component) concentrations have stayed constant. A manuscript summarising these data has been submitted for publication in "Arctic" and is now under review. Tables 1 and 2 (attached) from that report summarise the main dataset from the study.

In addition, a further suite of male and female ringed seal blubber samples (accompanied by relevant data for age, sex and condition) was taken in the spring of 1992, by contract with the Holman Island Eskimo Co-operative.

An opportunity arose to examine organochlorine residue concentrations in harp seals (*Phoca groenlandica*) from northern Labrador. Comparison of these data with those from harp seals taken in the Gulf of St. Lawrence suggest that most of the organochlorine residue accumulation by these animals occurs during migration and feeding in the Gulf of St. Lawrence, rather than in northern Labrador waters.

UTILIZATION OF RESULTS

These results and conclusions will be valuable to agencies concerned with public health in the Arctic, particularly in the context of providing advice about the consumption of subsistence foods. They will also be useful to agencies dealing with environmental regulation, since they confirm that imposing bans on the release of PCBs, for example, to the environment can lead to an eventual reduction in concentrations of these chemicals accumulated by biota.

Expected project completion date: Assuming more trend analyses are required to substantiate these preliminary results, 1996/97.

Table 1. Organochlorine residue concentrations in blubber, age and blubber thickness in female ringed seals (*Phoca hispida*) sampled at Holman, NWT, in 1972, 1981 and 1989.

Variable	1972 (n=19)		1981 (n=15)		1989 (n=14)	
p,p-DDE (μg/g wet wt.)	0.323 ±0.111	a	0.159 ±0.114	b	0.192 ±0.056	b
p,p-DDT (μg/g wet wt.)	0.447 ±0.254	a	0.323 ±0.189	a	0.081 ±0.028	b
ΣDDT (μg/g wet wt.)	0.770 ±0.311	a	0.482 ±0.213	a	0.273 ±0.075	b
p,p-DDE (as % ΣDDT)	44.8 ±17.4	a	33.9 ±13.7	a	70.5 ±9.33	b
PCB (μg/g wet wt.)	2.71 ±1.21	a	0.858 ±0.374	b	0.397 ±0.166	c
Age (y)	10.9 ±8.94	a	9.60 ±7.23	a	8.07 ±4.46	a
Blubber thickness (cm)	3.19 ±1.29	a	4.55 ±0.73	b	3.56 ±1.42	a

Data are shown as mean ± s.d. Data in the same row but in different columns followed by the same letter do not differ significantly by t-test ($p > 0.05$).

Table 2. Organochlorine residue concentrations in blubber, age and blubber thickness in male ringed seals (*Phoca hispida*) sampled at Holman, NWT, in 1972, 1981 and 1989.

Variable	1972 (n=19)		1981 (n=15)		1989 (n=14)	
p,p-DDE (μg/g wet wt.)	0.735 ±0.380	a	0.458 ±0.373	ab	0.386 ±0.325	b
p,p-DDT (μg/g wet wt.)	0.779 ±0.339	a	0.637 ±0.498	a	0.184 ±0.212	b
ΣDDT (μg/g wet wt.)	1.51 ±0.685	a	1.10 ±0.786	a	0.570 ±0.531	b
p,p-DDE (as % ΣDDT)	47.6 ±10.1	a	43.0 ±11.1	a	70.6 ±9.61	b
PCB (μg/g wet wt.)	5.46 ±1.99	a	1.89 ±1.11	b	0.606 ±0.365	c
Age (y)	11.2 ±6.69	a	8.88 ±5.41	a	8.63 ±6.79	a
Blubber thickness (cm)	3.12 ±0.780	a	4.48 ±0.682	b	2.90 ±1.25	a

Data are shown as mean ± s.d. Data in the same row but in different columns followed by the same letter do not differ significantly by t-test ($p > 0.05$).

CO-PLANAR PCBs IN ARCTIC MARINE MAMMALS AND FISH

PROJECT LEADER: D. Muir, Fisheries and Oceans Canada, Freshwater Institute, Winnipeg, Manitoba

PROJECT TEAM: D. Muir, C. Ford, B. Rosenberg, and B. Grift

OBJECTIVES

1. To provide geographic and temporal information on toxic PCB congeners and chlorinated dioxins/furans.

2. To compare arctic results with mid-latitude levels of co-planar PCBs.

3. To provide a linkage to biomarker studies in the same fish and marine mammals and to carry out assessments of dietary contamination.

DESCRIPTION

PCB congeners with 3,4,3',4'-chlorine substitution are the most biologically active and are referred to as toxic "co-planar" or "non-ortho" PCBs. They lack chlorine substituents in the 2 and 6 (or ortho) positions and can therefore assume a planar configuration. These congeners are isostereomers of 2,3,7,8-TCDD and have a similar mode of action; induction of hepatic mixed function oxidase (MFO) enzymes, immunotoxicity, teratogenicity and embryotoxicity (Safe 1990). The toxicity of Aroclor mixtures (commercial PCB formulations) is thought to be due almost entirely to these co-planar PCBs (Kannan *et al.* 1988).

Another group having a single chlorine in the 2- position is referred to as "mono-ortho" PCBs. Some of these compounds also have MFO enzyme induction potencies which are similar to those of the co-planar molecules. Toxic equivalent factors (TEFs) for non-ortho congeners, which are a measure of the biological potency relative to 2,3,7,8-TCDD, range from 0.1 for PCB-126 (3,3',4,4',5-pentachlorobiphenyl) to 0.01 for PCB-77 (3,3',4,4'-tetrachloro-biphenyl) (Safe 1990). Mono-ortho PCBs have been assigned TEFs of 0.001 (Safe 1990). These TEFs were used to calculate the contribution of co-planar PCBs to total toxic equivalents in arctic tissue samples.

Previous studies (Muir and Ford 1990, Muir 1992) have shown that co-planar PCBs and mono-ortho PCBs, are present in fatty tissues of ringed seal, walrus, narwhal and polar bear as well as arctic char. Chlorinated dioxins and furans are also present. However, coplanar PCBs account for most of the "TCDD equivalents" in arctic diet samples using the TEFs of Safe (1990). The objectives of this work for 1992/93 were to broaden the limited database by determining non-ortho PCBs, as well as chlorinated dioxins and furans, in additional fish and in whale and seal tissues to develop a larger database than is available at present, and to provide support for biomarker studies in fish and marine mammals.

ACTIVITIES IN 1992/93

Samples

Samples of walrus blubber were obtained from Inukjuak, Québec. Narwhal and beluga blubber were obtained from Creswell Bay (Somerset Island). Arctic char, that had previously been analysed for ortho-substituted PCBs, were obtained from Buchanan Lake and Hazen Lake, NWT. Lake trout and burbot liver were obtained from Lake Laberge and Kusawa Lake in the Yukon as part of the project entitled "Food Chain Accumulation, Biochemical Effects and Sediment Contamination in Lake Laberge and Other Yukon Lakes". Pooled samples of lake trout, burbot liver and walrus fat (from Inukjuak and Akulivik, Québec) were submitted for analysis of chlorinated dioxins and furans (PCDD/Fs).

Analytical Methods

A detailed description of the analytical procedure for non-ortho PCBs was prepared during 1992 and accepted for publication in Chemosphere (Ford *et al.* 1993). Briefly: fish tissues were extracted with hexane:dichloromethane (1:1), lipid removed with automated gel permeation chromatography. Fish extracts were split into two portions, one for determination of non-ortho PCBs and the other for determination of all other organochlorines. Extracts were then chromatographed on a silica-gel column to remove additional lipid coextractives then subjected to carbon-column chromatography to isolate the planar PCBs. Marine mammal blubber extracts were chromatographed on a silica gel column to separate PCBs (hexane elution) from most organochlorine pesticides. The hexane eluate was then subjected to chromatography on carbon to isolate non-ortho PCBs and were analysed for all other PCBs. The carbon column eluates were then analysed by GC-MS analysis and non-ortho PCBs quantified with the aid of ^{13}C-internal standards for PCB-77, 126 and 169 (3,3',4,4',5,5'-hexachlorobiphenyl). Mono-ortho PCBs (congeners 105, 118, 114, 156) in discarded fractions from the carbon column were determined by GC-ECD. Pooled samples of lake trout, burbot liver, and walrus blubber were also analysed for chlorinated dioxins/furans (PCDD/PCDFs) by Fisheries and Ocean's Ultratrace laboratory in Burlington, Ontario.

Quality Assurance

The National Institute of Standards and Technology (NIST) cod liver oil SRM 1588 was used as an internal control sample. An interlab comparison of coplanars in cod liver oil SRM 1588 was made with AXYS Analytical Labs (Sidney, BC) and with T. Metcalfe (Trent University).

RESULTS

An interlaboratory study showed relatively good agreement between the two labs on the analysis of the three non-ortho PCBs in cod liver oil (Table 1). No certified values for the three coplanar congenérs have been determined by NIST. The sample nevertheless serves as a useful intra laboratory control as well as enabling comparison with other labs.

Walrus from northern Quebec had a distinctive pattern of co-planar PCBs characterized by relatively high PCB 126, low PCB 77 and non-detectable PCB 169. (Table 2). This pattern is similar to what we have observed in ringed seal blubber (Ford *et al.* 1993, Muir and Ford 1990). The proportion of ΣPCB for each coplanar congener in walrus was similar to beluga and narwhal but lower than found for ringed seals (Muir 1992). Only PCDDs were detected in pooled walrus blubber from Inukjuak and Akulivik; PCDFs were uniformly non-detectable (<0.5 ng/kg) except for OCDF and H_7CDFs. The latter congeners were present at trace amounts in Akulivik walrus blubber (0.5 ng/kg) and 3-fold higher levels in the Inukjuak sample (1.55 ng/kg). Levels of 2,3,7,8-TCDD in the Akulivik sample were higher than at Inukjuak (7.3 vs 2.3 ng/kg) as were penta- and hexachlorodioxins. The higher levels of PCDDs in walrus blubber from Akulivik contrasted with the PCB levels which were generally much higher in the Inukjuak animals. Walrus appear to have the capability to metabolize PCDD/Fs, as do other marine mammals, but the congener pattern differs from what has been observed for seals and whales (Norstrom *et al.* 1990).

Narwhal and beluga from Creswell Bay had similar mean levels of PCB 77, 126 and 169 (Table 2) to those we have observed previously in animals from Pond Inlet which are likely part of the same population (Muir and Ford 1990). We found higher levels of PCB 169 in beluga from Husky Lakes which may have been related to the emaciated condition of the animals which were stranded in the ice (Muir 1992). Levels in these narwhal blubber samples, which were collected in 1991, were similar to those observed in samples collected in 1982, suggesting no significant decline over a 9 year period. But further assessment of age and PCB levels is required to confirm this observation.

Low levels of coplanar PCBs were detected in landlocked char from Buchanan and Hazen Lakes (Table 2). These low ng/kg concentrations were similar to those we have observed previously in sea run char (Muir 1992). The pattern of co-planar congeners differed between lakes; PCB 77 was undetectable (<5 ng/kg) in samples of char from Hazen Lake but was about 10-fold higher than PCB 126 in samples from Buchanan Lake. In general we have found PCB 77 to be the major coplanar congener in char (Muir 1992) so the results from Hazen are difficult to explain. PCB 126 and 169 represented about the same proportion of ΣPCB in char in both lakes.

Levels of coplanar PCBs in lake trout from Lake Laberge were higher than those from Kusawa Lake by about 10-fold (in males) as was expected given higher levels of ΣPCB(Table 2). Concentrations of coplanar congeners are about 10x lower in Lake Laberge lake trout than in lake trout we have analysed from Lake Ontario, but similar to levels in lake trout we have analysed from Lakes Erie and Huron (Muir and Ford, unpublished data). The proportions of coplanar congeners in ΣPCB were similar to what has been observed in char and lake trout from the Great Lakes. PCDD/Fs were detected at low ng/kg levels in pooled samples of lake trout and burbot liver from Lake Laberge. The most prominent congener was OCDD which was present at 7.5 ng/kg (fresh wt) in lake trout muscle from Laberge and 12.8 ng/kg in Kusawa. No tetra-, penta- , or hexachloro-CDDs were detected in lake trout from the two lakes (detection limit <0.5 ng/kg), but burbot liver from Lake Laberge contained low ng/kg levels of 1,2,3,6,7,8-HxCDD, HpCDD and OCDD. The most prominent PCDF congener in the three pooled samples, 2,3,7,8-TCDF, was present at 3.3 ng/kg in lake trout muscle from Laberge

but was undetectable in Kusawa (<2.2 ng/kg). TCDF was present at 28 ng/kg in burbot liver. This TCDF concentration is within the range observed in burbot liver in the Slave River (Whittle 1992).

Table 1. Interlaboratory comparison of non-ortho PCBs in NIST cod liver oil SRM 1588

Laboratory	Concentration (ng/kg±2SD) of each non-ortho PCB congener		
	77	126	169
DFO Winnipeg	1112±104	1083±118	176±66
AXYS Analytical Labs	1420±140	1400±150	227±36

CONCLUSIONS AND UTILIZATION OF RESULTS

The results indicate that non-ortho and mono-ortho PCBs contribute a substantial proportion of TECs in arctic marine mammal and fish samples. Landlocked char has similar low levels of non-ortho substituted (coplanar) PCBs to those we observed in sea run char (Muir 1992). But given the variability observed in levels of organochlorines observed in char and other fishes it seems prudent to examine additional samples. This is evident from the relatively elevated levels of planar PCBs in lake trout muscle and burbot liver from Lake Laberge. The proportion of co-planar PCBs, although quite variable among species, seems to be relatively constant within genus (for e.g. among *Salvelinus* sp.). In the coming year we intend to analyse coplanar PCBs in landlocked char from Amituk Lake because of elevated levels of PCBs and toxaphene in those fish. Co-planar PCBs will also be determined in fishes from Great Slave Lake where no analyses have been carried out for these analytes previously. Interesting results in 1991/92 showed relationships between planar PCBs and EROD in whale livers. Further work will also be carried out on PCDD/Fs in selected samples. The project will also forward results to Health and Welfare Canada for assessment of dietary contamination.

Table 2. Mean concentrations (±SD) of non-ortho PCBs and percent of ΣPCB in samples of arctic biota

Species/location	Sex	N	Concentration (ng/kg wet wt)			Percent of total PCB congeners		
			77	126	169	77	126	169
Marine mammal blubber								
Walrus - Inukjuak	M	2	51	897	<5	0.0005	0.0069	<0.0001
	F	6	27±22	207±105	<5	0.0009	0.0052	<0.0001
Narwhal - Creswell Bay	M	8	88.2±41.1	182±46.5	31.6±11.9	0.0032	0.0060	0.0011
Beluga - Creswell Bay	M	2	99.7	188	133	0.0028	0.0051	0.0041
	F	1	26.9	23.3	16.5	0.0016	0.0013	0.0010
Arctic char muscle								
Buchanan Lake	M	5	6.2±11.0	1.2±2.1	1.0±2.2	0.118	0.019	0.013
	F	5	2.7±2.2	<1.0	<1.0	0.048	<0.001	<0.001
Hazen Lake	M	1	<1	17.8	6.9	<0.001	0.036	0.014
	F	6	<1	19.2±4.4	7.0±2.0	<0.001	0.039	0.014
Lake trout muscle								
Lake Laberge	M	4	74.6±39.4	186±220	64.5±54.333	0.006	0.016	0.005
	F	2	31.5	61.4	56.4	0.031	0.060	0.055
Kusawa Lake	M	3	8.3±2.2	14.5±5.6	45.0±10.2	0.073	0.024	0.072
	F	2	7.8	24.9	52.9	0.014	0.046	0.098
Burbot liver								
Lake Laberge	M	3	396±96.6	1510±888	1410±906	0.013	0.051	0.048
	F	4	441±234	476±119	431±40	0.070	0.076	0.066

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LEAD AND OTHER HEAVY METALS IN TISSUES OF WALRUS AND RINGED SEALS

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PROJECT TEAM: R. Wagemann, M. Kingsley, D. Savoie, (contractor), E. Trebacz (contractor)

OBJECTIVES

1. To determine spatial and temporal trends of toxic metals (lead, cadmium, mercury) in tissues of ringed seals and other arctic marine mammals.

2. To provide a database for estimating dietary contamination with toxic metals in ringed seals and other arctic marine mammals.

3. To provide a database of methylmercury in ringed seals and other arctic marine mammals exhibiting high total mercury concentrations.

DESCRIPTION

This project is aimed at providing information on metals in marine mammals across the Arctic as a basis for determining spatial variability, deducing temporal trends by comparison with past and future data, and providing a basis for dietary calculations. Past information is very spotty for ringed seals and narwhal, but a much better database (total metals) now exists for belugas. This report addresses the information gap for ringed seals. High mercury levels have been reported in ringed seals, bearded seals, and belugas from the western Arctic (Smith and Armstrong 1975; Wagemann *et al.* 1990). In muscle, mercury levels significantly exceeded the Federal Guideline of 0.5 ppm (wet weight basis) in both ringed seals and belugas from this area, and other tissues were similarly elevated in mercury. The marine mammals from the western Arctic appear to have much higher concentrations of mercury in their tissues than those from the eastern Arctic. On the other hand, metals such as cadmium and lead appear to be higher in the eastern Arctic than the western Arctic. Lead was reported higher in belugas from Hudson Bay than from the Mackenzie Delta, and was very high in dolphins off the eastern Atlantic coast (Muir *et al.* 1988) and in some ringed seals from the Strathcona Sound area in the Northwest Territories (Wagemann 1989); the latter were affected by mining activities in that area. Cadmium in belugas from the eastern Arctic was higher than in belugas from the Mackenzie Delta, and cadmium in narwhal from the vicinity of Pond Inlet, NWT was higher than in any other group of marine mammals (Wagemann *et al.* 1983). However, a spatial distribution database for metals in arctic marine mammals which would allow rigorous statistical geographic trend analysis is as yet incomplete for ringed seals and will require additional ringed seal data from the mid-Arctic (vicinity of King William Island and Cornwallis Island). Two years ago we reported on data for metals in ringed seals from the eastern Arctic and Hudson Bay (Wagemann

1991). This report deals primarily with metals in ringed seals from the western Arctic (Sachs Harbour), and is a continuation of the effort to produce a spatial distribution database of metals in ringed seals from across the Canadian Arctic. The data on ringed seals from Sachs Harbour reported here comprise a substantial dataset (>100 animals) for dietary intake calculations for the western Arctic as well as being a bench-mark for comparison with past and future studies to determine temporal trends and ecosystem health in this area. Nevertheless, some data for ringed seals from the central Arctic are still required to complete the survey across the Arctic, and very little data is available for narwhal.

ACTIVITIES IN 1992/93

Samples

Liver, kidney and muscle tissues from well over 100 ringed seals from Sachs Harbour were obtained through M. Kingsley and colleagues. A further 26 are now available from Resolute Bay through B. Welch. Narwhal tissue samples (9 animals to date) from the vicinity of Pond Inlet were obtained from D. Blair (Fish and Marine Management, Freshwater Institute, Winnipeg), with prospects of obtaining more animals this summer. It is hoped that additional samples of ringed seals from Spence Bay or King William Island in the central Arctic can be obtained this summer to complete the spatial Arctic database for ringed seals.

Methods

Ringed seal tissues (liver, kidney, muscle) were analyzed (some in duplicate) for lead, cadmium, mercury, selenium, zinc and copper, involving well over 2500 analyses. Total mercury was determined by cold-vapour flameless/AA, and other metals by direct-current plasma emission spectrometry or graphite furnace AA, depending on tissue concentration. Prior to analysis, lead, and cadmium in muscle required pre-concentration by complexing (diethyldithiocarbamate) and extraction (butyl acetate).

Quality Assurance

Certified reference materials (Bovine Liver, NBS; DOLT, NRC; DORM, NRC) were used with every set of digests (~40) as a check on accuracy. High-purity reagents were used. Water for working standards and reagents was triply distilled in a quartz still, and reagent-grade acids were re-distilled in a Teflon still. Results were inspected, and seeming outliers were reanalysed using a fresh sample. This laboratory has participated in an international inter-laboratory trace metal analysis comparative study (Wagemann and Armstrong 1988) with a good outcome.

RESULTS

Ringed seals were collected at Sachs Harbour in 1987 (57 animals) and 1988 (60 animals). The mean ages of the 1987 animals (7.0±2.6 years) and the 1998 animals (6.0±4.3 years) were not significantly different and their age distribution was nearly normal (Fig. 1). Toxic metal tissue

concentrations varied greatly among individual animals irrespective of year of collection. Mean metal concentrations were not significantly different between those two years for any of the metals. Ages and metal results were therefore combined for both years (Table 1) and were used as a combined set in further analyses. Mean metal concentrations (dry weight basis) and standard deviations can be converted to a wet weight basis by multiplying with the following factors: 0.322 for liver, 0.241 for kidney, and 0.307 for muscle.

Mercury

The ringed seals in this survey (Sachs Harbour 1987/88) were relatively young (7/6 years), but had, nevertheless, on average, a relatively high mercury concentration in the liver (Table 1) which was significantly higher than in any of the ringed seals from the eastern Arctic and Hudson Bay (Fig. 2). A geographic trend of a decreasing concentration from northwest toward southeast may be present, but samples from the central Arctic are needed to complete the picture. For comparison, previously published results for mercury in ringed seals (Smith and Armstrong 1975) taken near Holman on western Victoria Island, NWT in 1972 and 1973 are also shown in Figure 2. Ages of animals were obviously determined in the earlier survey since a correlation between age and mercury was reported, but ages were not reported. Significantly, in the present survey total mercury in liver and age were found to be correlated in females only (Table 2), similar to the finding in the earlier survey, but also between age and total mercury in the kidney in both males and females, a tissue not analyzed in the earlier survey. The means of total mercury in the liver, 107 ± 115 μg/g dry wt (34.4 ± 37.0 μg/g wet wt) in the present survey, and 85 ± 93 μg/g dry wt (27.5 ± 30.1 μg/g wet wt) in the earlier survey, were comparable and not significantly different in view of the large variation in mercury concentration among individual animals in both surveys (0.72-636 μg/g, dry wt, in this survey). In the present survey, the mean mercury concentration in muscle was 1.43 μg/g, dry wt (0.45 μg/g wet wt) and in the earlier survey 2.35 μg/g dry wt (0.72 μg/g wet wt). The Federal Guideline (0.5 μg/g wet wt) was exceeded for muscle tissue in 33 % of the animals from Sachs Harbour. Of the three tissues analyzed, the highest concentration of total mercury was found in the liver, which was approximately 12 times higher than in the kidney, and 75 times higher than in the muscle. Because of the large inter-animal variability in both surveys these means for any of the tissues were not significantly different in the two surveys. Tentatively, we conclude that no change occurred in the mercury burden in the intervening 15-16 years in ringed seals in the western Arctic. This conclusion is conditional, however, on a more rigorous comparison correcting for any age differences by comparing regression slopes of mercury on age in the two surveys; this was, unfortunately, not possible at the present time. The authors of the earlier survey will be contacted to see if the age information from their survey is still available.

Lead

The mean lead concentration in liver of the Sachs Harbour ringed seals was approximately twice as high as in ringed seals from the eastern Arctic and Hudson Bay, (Table 1). The spatial trend for lead in liver of ringed seals appears to be similar to that for mercury, decreasing toward southeast (Fig. 3) which is different from that reported for belugas (Wagemann *et al.* 1990). However, statistically ($\alpha = 0.05$) the Sachs Harbour group mean was significantly different only from the Wakeham Bay group mean (0.04 ppm, Fig. 3), which was the only other group

comprising a substantial number of animals (33). Most other groups from the Hudson Bay area were small in size and their means did not differ from the Sachs Harbour mean largely for that reason. The Sachs Harbour animals had lead concentrations in their liver in the range 0.007-0.789 μg/g dry wt (0.002-0.25 μg/g wet wt), the maximum concentration being very comparable to the maximum reported for adult male ringed seals from the Baltic Sea taken in 1988 (0.11-0.27 μg/g wet wt) (Frank *et al.* 1992). Lead in any of the three tissues was not correlated with age, and was only correlated positively in muscle of males with cadmium and zinc, and with selenium and zinc, in muscle of females. Lead, zinc, and cadmium commonly coexist in sulfidic ore deposits, and their correlation in tissues may reflect a common source. In liver of males lead was correlated negatively with cadmium. Unlike mercury and cadmium, lead in muscle did not differ greatly from that in liver. It is generally the case that lead concentrations are comparable in different soft tissues. Kidney was not analyzed for lead.

Cadmium

Sachs Harbour ringed seals had a much higher concentration range of cadmium, 0.3-49.5 μg/g dry wt (0.10-15.9 μg/g wet wt) in liver and 0.74-363 μg/g dry wt (0.12-87.5 μg/g wet wt) in kidney, than the Baltic Sea ringed seals, 0.15-0.860 μg/g wet wt in liver and 1.36-5.41 μg/g wet wt in kidney (Frank *et al.* 1992). As in other marine mammals, average cadmium concentration was considerably higher in the kidney (90.9 μg/g dry wt, 21.7 μg/g wet wt) than the liver (19.5 μg/g liver, dry wt, 6.23 μg/g wet wt). The Sachs Harbour animals also had considerably higher concentrations of cadmium in both liver and kidney than belugas from the western Arctic (Mackenzie Delta, Wagemann *et al.* 1990), and ringed seals from Admiralty Inlet (Wagemann 1989). However, compared to the groups of ringed seals from the eastern Arctic and Hudson Bay (Table 1), Sachs Harbour ringed seals had considerably less cadmium in the liver. Kidney was not available from Hudson Bay ringed seals. The spatial trend for cadmium in ringed seals appears to be approximately opposite to that for lead and mercury, namely increasing from west to east and south east. (Fig. 4), in line with previous findings in marine mammals of the Canadian Arctic. For example, cadmium in the liver of polar bears (Norstrom *et al.* 1986) was reported to increase from west to east. In Sachs Harbour seals cadmium was strongly positively correlated with age in all three tissues, with selenium in liver, and with copper and zinc in kidney (Table 2). The latter correlation most likely reflects the common occurrence of these three metals in metallothionein.

Selenium

Selenium in ringed seals from Sachs Harbour was, on average, 48.0 μg/g dry wt (15.5 μg/g wet wt) in liver and 1.68 μg/g dry wt (0.52 wet wt) in muscle, approximately the same as in Baltic Sea adult ringed seals (19 μg/g, median in liver, wet wt) but much higher than in ringed seals from Hudson Bay (19-26 μg/g dry wt), which undoubtedly reflects the higher concentration of mercury in Sachs Harbour seals than Hudson Bay seals. Selenium in Sachs Harbour seals was also significantly higher than in walrus liver 9.61 μg/g dry wt (2.87 μg/g wet wt) and muscle 12.5 μg/g dry wt (3.33 μg/g wet wt) (Wagemann *et al.* 1993). Although selenium is an essential element, it is clear that tissue concentrations can vary considerably depending on the species and exposure to mercury, and is seemingly not as well controlled homeostatically as zinc. The concentration of selenium in Sachs Harbour ringed seals in both males and females was very

closely tied to that of mercury through the highly positive correlation with mercury in liver (Table 2), as in other marine mammals.

Zinc and Copper

These metals were measured in liver, kidney and muscle of Sachs Harbour seals, and in liver of seals from the Hudson Bay area. Since zinc is an essential element it was expected not to differ significantly among the different groups of ringed seals; it did not, and the levels in liver were quite comparable among the different groups. Copper, being also an essential element, was expected to behave similar to zinc, however, the mean copper concentration in liver of Sachs Harbour ringed seals was consistently lower than in Hudson Bay ringed seals. The significance of this difference is not clear, except that it may indicate more copper being available to the ringed seals at the eastern locations.

CONCLUSIONS AND UTILIZATION OF RESULTS

Sachs Harbour ringed seals had high mercury levels, particularly in the liver. The present data confirmed the similarly high values obtained in a survey 15 years ago in the same general area. It is tentatively concluded that there was no statistical difference in the mean concentration of mercury in the liver between this and the earlier survey. Although total mercury was much lower in muscle than in liver or kidney, the levels in muscle, nevertheless, exceeded the Federal Guideline (0.5 ppm) in one third of the animals surveyed. These results have implications for the assessment of dietary exposure via consumption of ringed seal tissues, particularly liver. In females total mercury in liver was highly correlated with age, older females having more mercury. A comparison with other geographic areas showed that mercury in ringed seals was higher in the western Arctic than the eastern Arctic. Lead was also higher in Sachs Harbour ringed seals than in those from eastern Canada, and was approximately the same as in ringed seals from the Baltic Sea. Cadmium in Sachs Harbour ringed seals was lower than in ringed seals from eastern Canada in accordance with the general trend of increasing cadmium in marine mammals from west to east in the Canadian Arctic. In all three tissues, cadmium accumulated increasingly with age. Additional work will be done on ringed seals from the central Arctic to complete the survey on the spatial distribution of metals in ringed seals across the Canadian Arctic. Work will also be undertaken to determine methylmercury in Sachs Harbour ringed seals to further delineate the implications for dietary exposure to mercury. Additionally, work will also be initiated on metal contaminants in narwhal to confirm or refute the extremely high cadmium levels found in 1978/79, the only sample of narwhals investigated to date.

Expected project completion date: March 31, 1996

Partners: Makivik Research Corp., Kuujuak, Quebec; Inuit of NWT; Fisheries and Oceans Canada (M. Kingsley).

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Table 1. Mean (± SD) concentrations (μg/g, dry wt), number of samples, and ranges of metals in tissues of ringed seals from Sachs Harbour and various locations in Hudson Bay.

Tissue	Pb	Cd	Hg	Se	Cu	Zn	Location
Liver	0.098±0.125 115 0.007-0.79	19.5±9.9 86 0.3-49.5	107±115 117 0.72-636	48.0±41.4 117 4.04-203.5	32.8±21.5 117 9.8-129	129±19 117 85.1-223	Sachs Harbour
	0.024±0.006 2 0.02-0.03	36.8±14.3 2 26.7-46.9	65.5±30.7 2 43.8-87.2	27.5±3.7 2 24.9-30.1	44.7±5.1 2 41.1-48.3	143±7 2 137-148	Salluit
	0.038±0.026 33 0.005-0.128	22.8±27.6 33 0.74-127.4	15.0±18.4 33 1.2-69.1	11.0±8.6 33 3.5-35.2	49.0±23.8 33 11.9-103	139±25 33 82.7-198	Wakeham Bay
	0.055±0.031 9 0.023-0.114	32.6±46.5 9 2.8-148.6	35.7±28.1 9 9.8-92.7	19.1±11.2 9 7.1-45.6	50.9±40.5 9 16.7-129	153±35 9 103-226	George River
	0.050±0.029 10 0.017-0.098	51.8±35.0 10 7.3-110.7	16.2±9.3 10 2.19-28.39	17.9±5.9 10 8.1-24.6	56.2±35.1 10 27.8-152	170±21 10 150-207	Inukjuak
Kidney	---[1]	90.9±62.5 117 0.74-363	8.7±5.4 118 1.1-29.8	---	26.0±8.5 115 13.5-54.8	158±40 115 92-321	Sachs Harbour
Muscle	0.064±0.049 86 0.007-0.198	0.14±0.15 99 0.0-0.89	1.43±0.99 106 0.16-6.59	1.68±0.47 97 0.81-2.96	3.75±0.55 100 2.4-5.2	94±18 100 57.5-141	Sachs Harbour

[1] --- not analysed

Table 2. Associations among metals, and gender, and between age and metals in liver, kidney, and muscle of ringed seals from Sachs Harbour (1987, 1988 combined) at $0.90 \leq P < 0.95$, $0.95 \leq P < 0.99$ (underlined), and $0.99 \leq P$ (doubly underlined).

Dependent Variable		Independent Variables		
		Liver	Kidney	Muscle
Common to males and females	Se	+Age, +Hg	---[1]	---
	Cd	+Zn	+Cu, +Zn	---
	Zn	+Cu	---	---
	Hg	---	+Age	-Cu
Females only	Hg	-Age	---	+Cd
	Cd	+Age, +Se	+Age	+Age
	Cu	---	-Age	-Age
	Se	---	---	+Pb, +Zn
Males only	Cd	-Pb	---	+Pb
	Zn	---	+Hg, +Cu	+Pb
	Cu	---	-Age	+Se
	Se	---	---	Age, +Hg

[1] --- no associations found at the indicated confidence level.

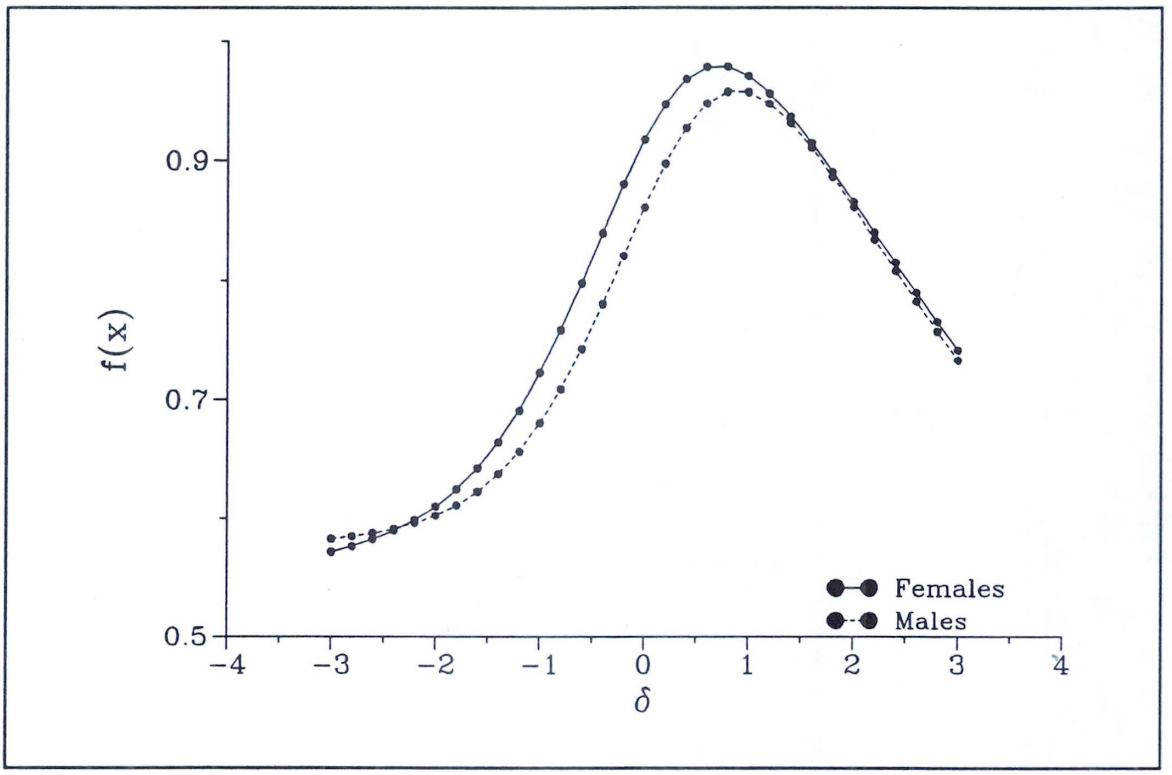
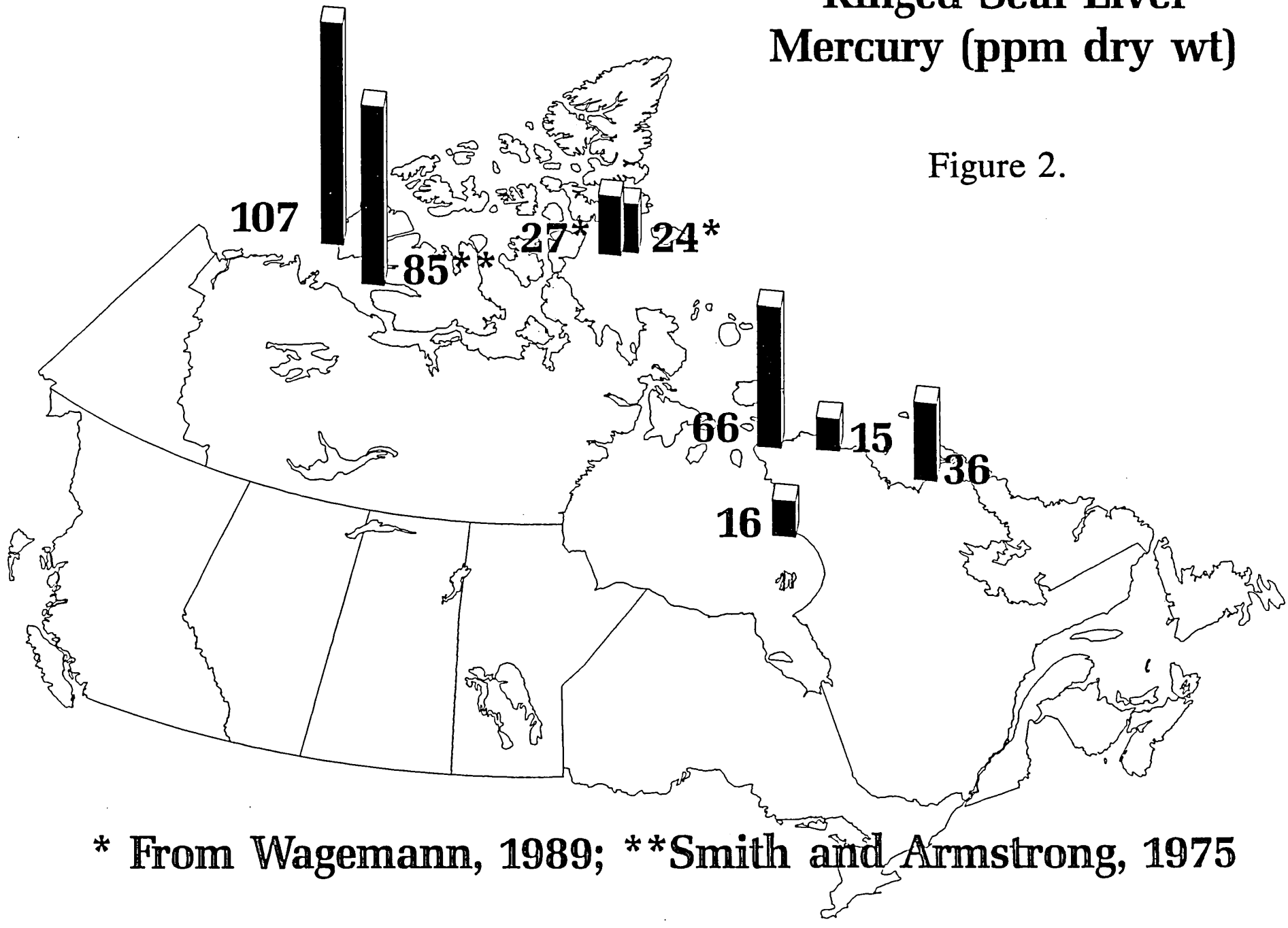


Figure 1. Normal distribution of male and female Sachs Harbour ringed seals; samples from both years, 1987 and 1988, combined.

Ringed Seal Liver Mercury (ppm dry wt)

Figure 2.



* From Wagemann, 1989; **Smith and Armstrong, 1975

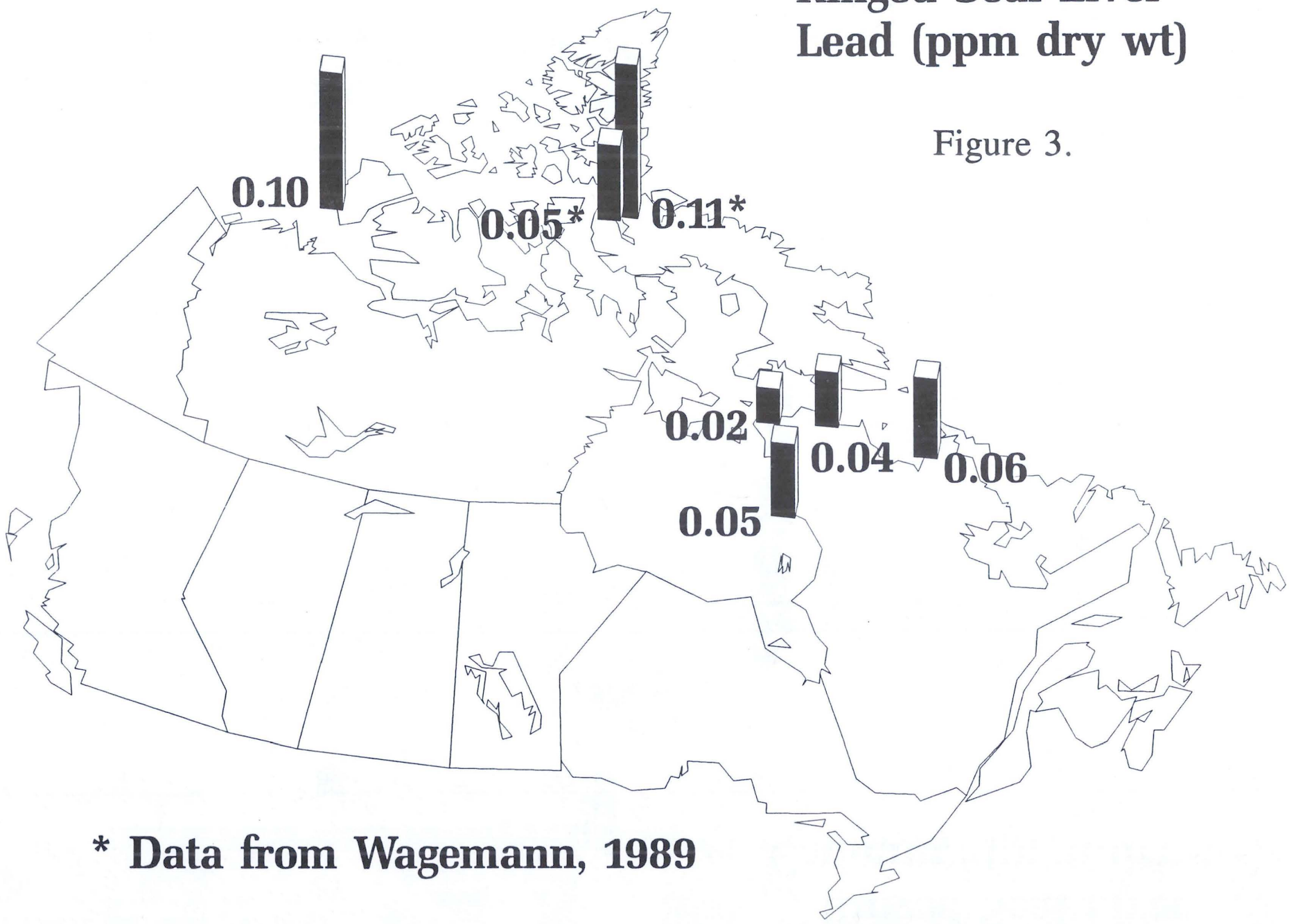
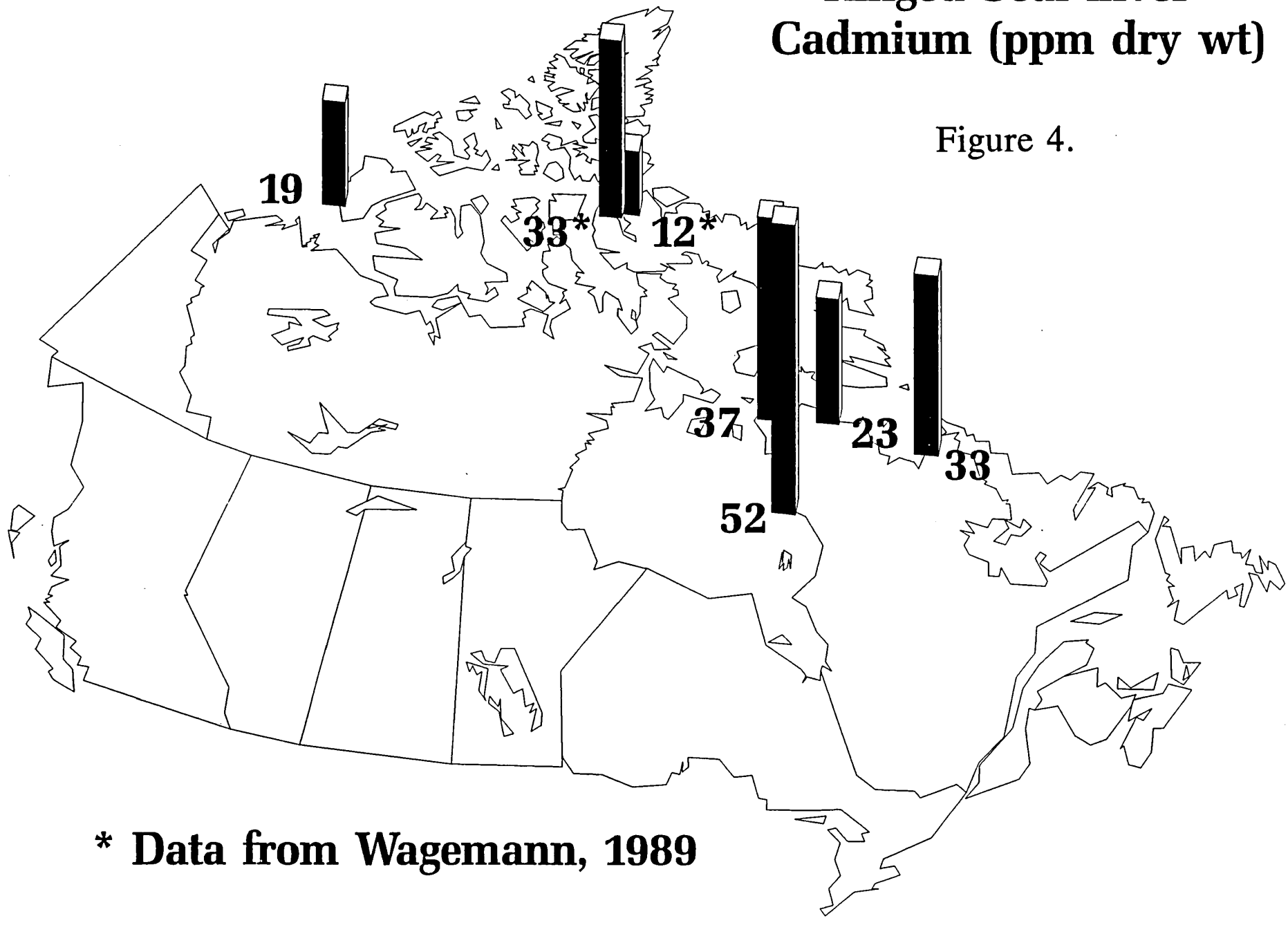

Ringed Seal Liver
Lead (ppm dry wt)
0.10
0.05*
0.11*
0.02
0.04
0.06
0.05
* Data from Wagemann, 1989

Figure 3.

Ringed Seal Liver Cadmium (ppm dry wt)

Figure 4.



*** Data from Wagemann, 1989**

MODELLING AND EVALUATION OF CONTAMINANT ACCUMULATION AND EFFECTS IN MARINE MAMMALS

PROJECT LEADER: M. Kingsley, Fisheries and Oceans Canada, Mont-Joli, Québec

PROJECT TEAM: B. Hickie, Environmental and Resource Studies Program, Trent University, Peterborough, Ontario

OBJECTIVES

1. To develop contaminant accumulation models for Arctic marine mammals.

2. To understand contaminant pathways.

3. To provide a framework for directing contaminant monitoring programs concerned with marine mammals of significance to the diet of native peoples.

DESCRIPTION

Marine mammals are long-lived top-level predators with high maintenance-energy requirements. They therefore have the capability of both concentrating and accumulating persistent contaminants (including fat-soluble organochlorines and some metals) from background and local sources. In many parts of the Canadian Arctic, they are esteemed food species and form a significant component of human diets. The potential consequences of this situation include unacceptable levels of contaminant intake among people consuming traditional diets in Arctic coastal communities. Monitoring programs are essential for assessing possible direct toxic effects and for assessing the human health implications of contaminants in Arctic marine mammals. It is important that we develop an understanding of the sources, pathways, and factors controlling rates of contaminant accumulation by marine mammals if we wish to relate ecosystem contaminant levels and loadings with those in marine mammal species.

Energetics-based contaminant accumulation models provide the basis for understanding and quantifying the importance of factors such as age, sex, reproductive effort, growth and diet on contaminant concentrations at the level of the individual animal for beluga, narwhal, ringed seal, and walrus. Development and refinement of models of contaminant accumulation will aid in identifying data gaps, directing contaminant sampling programs, interpreting data from monitoring programs in terms of spatial and temporal trends, and in relating contaminant levels in marine mammals to those in other components of their food web.

ACTIVITIES IN 1992/93

The Arctic species selected were: beluga, narwhal, ringed seal, and walrus. Among Arctic marine mammals, these species are the most important as dietary components of Canadian Arctic people, and between them cover a spectrum of different feeding niches. Literature reviews were conducted to assemble the biological and contaminant data required to develop the models and to identify data gaps which may limit model development or use. Approximately 600 references have been identified and inspected. About 200 of these have been reviewed in detail. Sufficient information on the life-history, growth and development, energetics and diet was found to permit the full development of models for ringed seals and beluga. The basic model code was revised for these species. Major revisions were required for the ringed seal model to reflect their reliance on blubber as an energy reserve during lactation and during the summer moult. Changes in blubber reserves can lead to significant variation in organochlorine concentrations in the blubber and in other tissues. Comparisons of organochlorine levels to define spatial or temporal trends in this species could be more revealing if they were based on total body burden rather than on concentration.

Because no reliable and practical age determination technique exists for the narwhal, there is no age-specific data, and the ability to build realistic models of any age-related process for this species, including the pharmacokinetics of contaminant accumulation, is limited by this restriction. However, a tentative model has been proposed for the narwhal which is based on its similarity to the beluga. Development of a model for walrus is presently restricted by the general lack of published data on this species, particularly for the Atlantic subspecies. We are presently seeking the cooperation of biologists with active research programs who may be able to provide the data required to complete the walrus model.

Data have been assembled on the past and present levels of contaminants in the food web in the St Lawrence. While information is not plentiful, what there is shows that contaminant levels in estuarine fish, now low, have changed little over the last 15 years. In contrast, contaminant levels in migrating American eels are now much lower than they were even 10 years ago; eels are also less numerous than they were. Contaminant trends in migrating eels are similar to the trends found for many fish species in Lake Ontario, suggesting that contaminant levels in eels will now remain stable or decline only slowly in the future. Modelling applications completed include evaluation of the probable past contribution (about 40%) of migrating eels to the contaminant burdens found in beluga, and comparisons of contaminant levels predicted by modelling with actual levels found in 20 selected recovered carcasses of various ages and both sexes.

RESULTS

Modelling of contaminant transfers within ecosystems forms a necessary part of the framework of investigation and interpretation of data, just as modelling of transfers in physical systems is a necessary part of the investigation of those transport mechanisms. The *development* of these biological models provides mutual cross-checking of current understanding in several related fields of ecological study: trophic relationships within the marine ecosystems, energetics and

energy transfer between and within organisms, and contaminant pharmacokinetics and partitioning between tissues. The models themselves serve as tools for the interpretation of present and future data on contaminant burdens in mammals and their prey, and for the prediction of trajectories of contaminant burdens at the population level.

Modelling at the population level has been developed to the point of using contaminant trends in prey species as forecasting devices to quantitatively predict future trends in contaminant levels of beluga populations.

The transfer of contaminants from mother to nursling is an important pathway of contaminant clearance for females, and is critical in defining the contaminant burdens in progeny. The model for beluga and narwhal suggests that females may lose as much as 40 percent of their PCB burden while nursing their first calf, while female ringed seals may lose as much as 60 percent. These estimates are based on simple predictions of contaminant partitioning between maternal blubber and milk, since to date no data are available to define the partitioning more precisely. Contaminant analysis of matched maternal blubber and milk samples are required to define this pathway, and will be useful in explaining why males and females differ little in the levels of contaminants such as Mirex.

CONCLUSIONS AND UTILISATION OF RESULTS

It appears from tests of all the models that the ability to apply them is at present limited by the absolute scarcity of data on contaminant levels in the prey of all four species. Further development of sampling and analysis programmes directed to the lower trophic levels in Arctic food webs appears essential to the investigation of the sources of contaminants found in marine mammals. Key prey species for which there are few or no data include; Arctic cod (*Boreogadus saida*), Greenland halibut (*Reinhardtius hippoglossoides*), *Parathemisto* spp., and bivalves (*Mya truncata*, *Serripes groenlandicus*).

Expected project completion date: March 31, 1997

Partners: Arctic marine mammal scientists have made available information on food habits and life history variables; contaminant chemists have provided access to contaminant data.

CONTAMINANT TRENDS IN POLAR BEARS

PROJECT LEADER: R.J. Norstrom, Environment Canada.

PRINCIPAL INVESTIGATORS AND SUPPORTING AGENCIES:

M. Simon, M. Mulvihill, Environment Canada
R. Letcher, J. Zhu, Carleton University
I. Stirling, Canadian Wildlife Service and University of Alberta
M. Ramsay, University of Saskatchewan
M. Taylor, Government of Northwest Territories
M. Oehme, Norwegian Air Research Institute

OBJECTIVES

Short-term:

Analyze data on circumpolar geographical distribution of persistent organochlorine contaminants in the Polar Bear. Determine effects of age, sex, nutritional status, and sampling time on analysis of long-term trends. Determine structures of unidentified contaminants.

Long-term:

Determine spatial and temporal trends of persistent and toxic organochlorine chemicals in a species at the top of the arctic marine ecosystem food web as an indication of the time constants and effectiveness of global controls on these chemicals.

RATIONALE

It has been suggested that the Arctic Ocean may become one of the principal global sinks of organochlorine compounds such as PCBs, DDT, chlordane and dieldrin because of the tendency for net atmospheric transport from warmer to colder areas. This may have some basis in fact, since recent estimates indicate the net flux of PCBs from the atmosphere to the ocean surface is greater at higher latitudes (Iwata *et al.* 1993). Organochlorines have been detected in arctic air, snow, ice, water and biota (Barrie *et al.* 1992, Muir *et al.* 1992). Knowledge of the circumpolar distribution and temporal trends in concentrations of organochlorines is therefore important in determining the sources and potential significance of these contaminants to arctic marine and maritime wildlife and the humans that consume them. It has already been demonstrated that levels of PCBs are about three to five times higher in breast milk of Inuit from northern Québec than in the southern Caucasian population (Dewailly *et al.* 1989, 1992).

The polar bear (Ursus maritimus) is an excellent candidate biomonitor for organochlorines. Polar bears are the principal mammalian predators at the top of the arctic marine food chain,

and are distributed widely throughout the Arctic and Subarctic circumpolar regions. Several studies have been published identifying organochlorines and their metabolites in polar bear tissues, studying their bioaccumulation in the food web and determining geographical distribution of levels in fat and liver. Most of the studies have been in the Canadian Arctic, but there is some information for Alaska, Greenland and Svalbard bears (Clausen *et al.* 1974, Bowes and Jonkel 1975, Lentfer 1976, Edelstam *et al.* 1981, Norstrom *et al.* 1988, Muir *et al.* 1988, Norstrom *et al.* 1990, Letcher *et al.* 1992, Jarman *et al.* 1992, Norheim *et al.* 1992, Bergman *et al.* 1993).

RESULTS

A proposal for a circumpolar survey of contamination in polar bear fat was presented by the Canadian Wildlife Service at the International Union on Conservation of Nature Polar Bear Specialists Group in Sochi, USSR in October 1988. The Department of Renewable Resources of the NWT (M. Taylor, Yellowknife), the University of Saskatchewan (M. Ramsay, Saskatoon), the University of Alberta (I. Stirling, Edmonton), the Kuujjuaq Research Centre (S. Olpinski, Kuujjuaq, Quebec), the Norsk Polarinstitut (Ø. Wiig, Oslo), the Greenland Fisheries Research Institute (E. Born, Copenhagen) and the United States Fish and Wildlife Service (S. Schliebe, Anchorage) agreed to collaborate in the survey. Although the former U.S.S.R. agreed to participate, no samples were actually obtained. Unfortunately, the survey is therefore only hemispheric. Approximately 700 samples of fat from hunted bears and subcutaneous biopsies from research programs were collected from across Canada, Alaska, Greenland and Svalbard in 1989-1991. These have been analysed by the Canadian Wildlife Service for over 20 different PCB congeners and organochlorines. The summary statistical analysis and brief discussion of the data are presented in this report. A manuscript for publication in the scientific literature, based on these findings, with all of the participants as co- authors, is in preparation.

Samples were placed into 17 geographical groups based on the likelihood of similar exposure to contaminants through a common food supply and common range. In the case of Canadian bears, these groups were modified from the Polar Bear Management Zones used by the Department of Renewable Resources of the NWT.

For purposes of statistical analysis, bears were put into three sex categories, male (M), solitary females (F) and females with cubs (FC). Levels of PCBs and chlordane in fat biopsy samples from western Hudson Bay showed a tendency to decrease from young-of-the-year cubs to approximately 5 years. Trends in levels with age were much less pronounced in mature adults. Therefore, all statistical comparisons among areas were made with results from bears older than five years. In the case of multiple-compound classes, the data were summed to give the total for the class: s-DDT (DDT+DDE), s-Chlordane (mainly the metabolite oxychlordane and nonachlor-III), and s-PCB (sum of 20 congeners) prior to statistical analysis. Details on the identity of the various individual organochlorine compounds and most of the chemical methodology has been reported previously (Norstrom *et al.* 1988, Muir *et al.* 1988).

The dataset was cleaned up by removing records for which some critical data (either biological or chemical) was missing. There were 300 bears in the final adult dataset. Statistical analyses

were done using the SAS GLM procedure. To obtain normality, the data were log transformed (square root for dieldrin). Where it was significant, sex (PCB and s-Chlordane) and sex by age interactions (PCBs) were eliminated by standardizing the data to solitary females for purposes of comparing geographical distribution free of these effects. Overall, females have 42% higher levels of s-Chlordane than males ($p<0.05$). Females with cubs also have 32% higher levels of s-DDT than males. There is no significant difference between the two groups of females for any organochlorine. The pattern is quite different for s-PCBs than for s-Chlordane and s-DDT. Males have 50% higher levels than females ($p<0.05$). Ranges in concentrations were: s-Chlordane, 1.5-6.2 mg/kg; dieldrin, 0.08-0.49 mg/kg; s-DDT, 0.09-0.68 mg/kg; s-PCB, 1.7-15.7 mg/kg.

DISCUSSION

The geographical distribution of geometric mean total residue levels in the four major classes, standardized to solitary females, are shown in Figure 1. In general, geometric mean levels tended to increase gradually from west to east and south (Alaska-Svalbard-Hudson Bay), as was found in the Northwest Territories of Canada in a previous study (Norstrom *et al.*, 1988). This trend was most pronounced for s-DDT and dieldrin and is consistent with long-range atmospheric transport of contaminants from the North American continental land mass, which is expected to contribute more to the "background" levels in the eastern and southern Arctic west of Greenland. The situation is more complex east of Greenland, since European/Eurasian sources also play a role. Polar bears from the Bering and Chukchi Seas have lower levels of most organochlorine compounds than those from the Greenland and Norwegian Seas, indicating that Asian sources contribute less to arctic contamination than other areas. However, the relatively uniform levels indicate that these organochlorine compounds, especially s-Chlordane, are well distributed in the Arctic marine ecosystem of the western hemisphere, perhaps even close to equilibrium with global atmospheric cycling (Iwata *et al.* 1993). With few exceptions, the geographical variation for each chemical group is within a factor of two or three throughout the North American Arctic, and is not statistically significant among areas (data not presented).

The most pronounced geographical variation was the significantly higher level of s-PCB (11-16 mg/kg) in eastern Greenland, Svalbard and M'Clure Strait bears than in the other 14 areas, which averaged 3.7 ± 1.6 mg/kg. The Greenland and Svalbard results probably indicate a substantial European/Eurasian input of PCBs which is not strongly influencing the rest of the hemisphere. The high levels of s-PCB and also s-DDT in the M'Clure Strait area appear to be anomalous. There is a possibility that these levels are a reflection of a higher flux to this part of the Canadian Arctic, resembling that at Svalbard, perhaps from transpolar circulation during the winter. Transpolar circulation has been proposed as a mechanism for the higher levels of polychlorinated dibenzodioxins found in polar bears and ringed seal from nearby areas (Norstrom *et al.* 1990). There is also the possibility that some of the geographical variation in levels is a reflection of differences in feeding ecology of the bears, or the seals they eat. Livers and hair of bears from the M'Clure Strait/Viscount Melville Sound area had the highest level of mercury in the Arctic (Braune *et al.* 1991, Renzoni and Norstrom 1990), which was thought to be at least partly related to feeding ecology of ringed seal. Bears from southeastern Hudson Bay had the highest levels of s-Chlordane and s-DDT. Because this group of bears are farther

south than any other (ca. 57°N), they are in an area which is frequently in the trajectory of air movements from northeastern North America. A higher input of these pesticides from the North American continental land mass is therefore indicated in southeast Hudson Bay.

Temporal trends are impossible to establish with any certainty using existing data because of the complexity of biological and ecological effects on levels of contaminants in fat noted above, and inconsistent protocols. Only pooled fat samples (but individual liver samples) were analyzed from previous time periods, and there was no attempt to control sex or age makeup of the samples (Norstrom *et al.* 1988). The previous data for polar bear fat are therefore equivalent to arithmetic mean levels, rather than geometric means as presented in Figure 1. Ignoring these factors, the levels of all organochlorines may have gone down slightly between 1982 and 1991-92. Reanalysis of individual samples from the Canadian Wildlife Service Specimen Bank is underway to establish statistically valid temporal trends.

Expected project completion date: Newly collected and archived samples will be analysed to answer specific questions, such as reasons for high levels of many organochlorines in the Melville Island area, and to fill in data gaps, at least to the end of the AES Greenplan in 1997. Hopefully another broad-scale survey can be conducted around the year 2000 to establish long-term trends.

ACKNOWLEDGEMENTS

Many individuals have contributed to this work besides the principal investigators mentioned on page 1. Foremost among these are the hunters who provided most of the samples, Mary Simon and Michael Mulvihill, who were in charge of the chemical analysis; Brian Malone, who set up the database and performed the statistical analysis; Wendy Calvert, Andy Derocher, Dennis Andriashek and John Lee who were involved in obtaining and summarizing biological and geographical information for the Canadian Samples. Funding for chemical and statistical analysis was obtained through the Arctic Environmental Strategy Program, Indian and Northern Affairs Canada.

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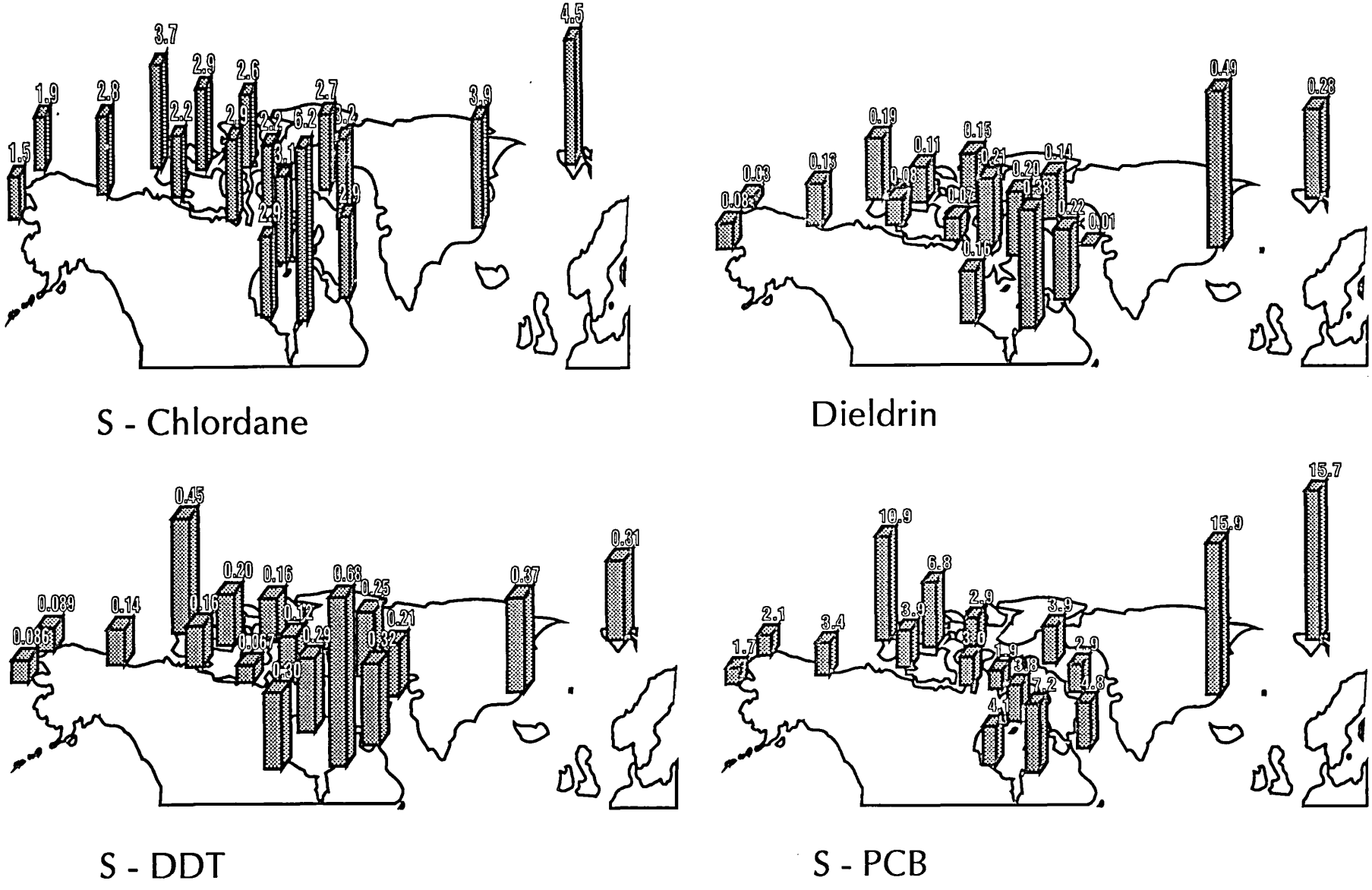

Figure 1. Geographical distribution of geometric mean levels (mg/Kg) of total organochlorine residue classes: S-Chlordane, Dieldrin, S-DDT and S-PCB in Polar Bear fat, standardized to solitary adult females to remove the effect of age, sex and sex-age interactions.

ASSESSMENT OF ARCTIC ECOSYSTEM STRESS: EFFECTS ON POLAR BEARS

PROJECT LEADER: R.J. Norstrom, Environment Canada.

PRINCIPAL INVESTIGATORS AND SUPPORTING AGENCIES:

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D. Muir, Fisheries and Oceans Canada
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Inuit hunters
I. Stirling, M. Simon, M. Mulvihill, S. Kennedy, S. Trudeau, G. Sans-Cartier, Environment Canada

OBJECTIVES

Short-term:

Determine identity, concentrations and tissue distribution of potentially toxic sulphur-containing metabolites of halogenated aromatic compounds and their precursors in the polar bear and other arctic marine mammals; determine presence or absence of specific pathologies associated with these contaminants; determine cytochrome P450 monooxygenase enzyme activity levels in polar bear tissues and compare to Western Blot analysis of P450 proteins; determine EROD- and porphyrin-inducing capability of fractionated tissue extracts in chick hepatocyte bioassay; determine organic/inorganic mercury ratios and heavy metal/metallothionein correlations in polar bear tissues; determine the effect of season and adipose tissue size on contaminant levels in adipose tissue and milk and correlate levels of contaminants in females and cubs.

Long-term:

Determine the effects, at the individual and population level, of persistent toxic organochlorine chemicals, their metabolites and heavy metals in the polar bear; collaborate with DFO in assessing effects on other marine mammals at the top of the arctic marine ecosystem food web; determine the potential for exposure of the human population to persistent PCB and DDT metabolites through ingestion of wild foods.

RATIONALE

Estimates of number of polar bear cubs per potentially reproducing female have shown steady declines in Hudson Bay throughout the 1970s (Stirling pers. comm). There are several possible ecologically-based factors which may reduce fertility, such as poor nutritional status as result of poor hunting due to unsuitable ice conditions much of the year. Persistent organochlorines may also be a contributing factor. Utilization of fat reserves causes relocation of these compounds from adipose tissue, therefore during times of nutritional stress, organochlorine toxicity might be much more expressed than under more favourable conditions. It is probable that pregnant females, fetuses and cubs are most at risk for toxic effects of organochlorines because of mobilization from fat to target organs in the female and fetus during gestation, and to the cubs via milk for the first year or two after birth (Norstrom 1990).

Levels of PCBs, DDT and chlordanes are highest in Hudson Bay bears, reaching several ppm levels in adipose tissue and liver (Norstrom *et al.* 1988). PCBs and DDT are the suspect compounds in the reproductive failure of grey and ringed seals in the Baltic Sea (Helle *et al.* 1976). Mink are also known to be highly sensitive to PCB-mediated reproductive failure (Jensen *et al.* 1977). These animals share with the polar bear the strategy of delayed implantation of fertilized ova, which is a potential element in sensitivity to reproductive failure.

The significance of these results to polar bear ecotoxicology is difficult to determine. Mean PCB levels in fat are generally four to ten times below those which are associated with reproductive effects in most wild mammals such as mink and grey seal (Olsson 1987). However, some individual bears have PCB levels (20-30 mg/kg in lipid) which are approaching the level of concern *if* polar bears are sensitive species.

In collaboration with the University of Stockholm we have recently identified methylsulphone (MSF) metabolites of PCBs and DDE in Polar bear tissues (Bergman *et al.* 1993). About 15 MSFs were identified in liver and fat of both species. Polar bear liver contained a relatively large concentration, ca. 1 ppm in lipid, of 2- and 3-MSF-DDE, but these compounds were not found in adipose tissue. Total MSF-PCBs had a concentration of ca. 1 ppm in adipose tissue, about 5% of total PCBs. Large amounts of tris-(4-chlorophenyl)-methanol were also present. We have recently identified this compound as a global contaminant (Jarman *et al.* 1992). DDE methylsulphones have been implicated in an endocrine disease syndrome (hyperadrenocorticism) in Baltic grey seal which led to osteoporosis, possible immune deficiencies and reproductive failure (Bergman *et al.* 1986; Brandt *et al.* 1992). MSF-DDE is a potent metabolic activator in adrenal cortex (Brandt *et al.* 1990). MSF-PCBs have been identified in whales, fish, oysters and clams (Haraguchi *et al.* 1989). This may indicate formation by the organism, or accumulation of MSF-PCBs in the environment. MSF-PCBs have been shown to accumulate in other species known to be sensitive to PCB-mediated reproductive failure, such as otter (Haraguchi *et al.* 1990).

Studies on humans exposed to high levels of PCB in rice oil (Yusho disease), have indicated the presence of ppb levels of MSF-PCBs 15 years after exposure (Haraguchi *et al.* 1986). An occupationally PCB-exposed mother had 0.8 ppm of MSF-PCBs in breast milk the first 5 months after parturition, which declined to about 0.2 ppm 16 months after parturition (Yoshida *et al.*

1979). Concentrations of MSF-PCBs were constantly 5% of the total PCB concentration in milk. The breast milk of mothers eating arctic marine mammals may contain MSF-PCBs and DDE from endogenous metabolism as well as possible accumulation from marine mammal blubber.

The presence, tissue distribution and toxicity of MSF-PCBs and MSF-DDE in the polar bear and other arctic species, and the significance to human health, needs to be investigated.

RESULTS

This project forms the major portion of the Ph.D. research of R. Letcher, Centre for Analytical and Environmental Chemistry, Carleton University.

Previously developed analytical methods (Haraguchi *et al.* 1986) were adapted to the determination of MSF-PCBs and MSF-DDE in five archived polar bear liver and fat samples from Western Hudson Bay in 1985. The identities and concentrations of the MSF-PCBs, -DDEs and their parent compounds are presented in Table 1. DDE and total PCB concentrations are also given for comparison. Identities and levels of total MSF-PCBs and were similar to those found previously (Bergman *et al.* 1993), approximately 1.5 μg/g lipid in liver and 0.4 μg/g lipid in fat. The average ratio of s-MSF-PCBs in liver to fat was 3.6 $\pm$ 0.94. This ratio is about twice that of s-PCBs, 1.67 $\pm$ 0.78, and DDE, 2.11 $\pm$ 0.97, and intermediate to the liver/fat ratio of epoxide type compounds such as dieldrin and oxychlordane, which is generally greater than 10 (Norstrom *et al.* 1988). Levels of 2- and 3-MSF-DDE levels were high in liver, 0.3 μg/g lipid, but were undetectable in adipose tissue.

The results of this research were presented at the DIOXIN'92 conference in Tampere, Finland (Letcher *et al.* 1992).

A more rapid multi-residue method for the analysis of PCBs and their methylsulphone (MSF) and hydroxy (OH) metabolites was developed and validated for a variety of tissues, including polar bear fat and liver, fish and eggs. This research was conducted in collaboration with Å. Bergman, University of Stockholm, and K. Haraguchi of Daiichi College of Pharmacy, Fukuoka, Japan, who supplied authentic MSF-PCB and MSF-DDE standards. A publication on the method is in preparation.

As part of a study of the half-life of the drug, Telazol, used to immobilize polar bears for research purposes, fresh samples of tissues (kidney, liver, muscle, fat and gonads) and blood from 4 adult male bears were obtained by M. Ramsay (University of Saskatchewan) from the Resolute area in the NWT in cooperation with Inuit hunters as part of their authorized quota. These samples will be analysed this year for PCBs, MSF-PCBs and other organochlorines using the newly developed method.

Liver specimens from these four bears were preserved at liquid nitrogen temperatures and analyzed by immunoblot techniques (Western Blots) by S. Bandiera (University of British Columbia) and M. Hahn (Woods Hole, Massachusetts) to characterize the cytochrome P450

enzyme system of the polar bear. The immunoblot technique is a surrogate for direct measurement of enzyme activity, and is not as sensitive to tissue preservation conditions as enzyme activity. Several liver samples archived in the CWS Specimen Bank at -40° were also analysed to test if cytochrome P450s could be recognized in the samples preserved under conditions which would destroy enzyme activity. Relative amounts of cross-reactive P450 proteins were compared with total PCB levels (Table 2).

The results can be summarized as follows: Polar bear liver contains orthologues of scup cytochrome P450 1A1, rat epoxide hydrolase and rat cytochrome P450 1A1 and 1A2, 2B1 or 2B2, 2C11 and 3A, but not 2C7 or 2C13. Cytochrome P450 1A is induced by TCDD and non-ortho PCBs, indicating that the polar bear is exposed to this class of compounds. Cytochrome P450 2B was also induced, probably by PCBs other than the non-ortho PCBs. Some of the homogenized liver samples frozen at -40° for 10 years retained a significant amount of P450 protein which cross-reacted with scup and rat P450 1A1. P450 protein levels may correlate with PCB concentration, but may also reflect the degree of P450 protein deactivation in the samples. In fresh livers kept at -196°, Cytochrome P450 2B1 levels seemed significantly higher than in the -40° stored samples, possibly because of recent exposure to the drug, Telazol. Correlation of P450 1A and 2B levels with PCB concentrations is not ruled out, but the sample size was too small to define the relationship. The immunoblot results were presented at the 7th International Symposium on Responses of Marine Organisms to Pollutants in Göteborg, Sweden (April 20-22, 1993).

DISCUSSION

Relatively non-specific, selective retention of MSF-PCBs was found in polar bear liver. Persistent polar compounds may be retained in polar bear liver because of the presence of a large number of lipid-containing stellate cells, which are specialized in vitamin A storage (Leighton *et al.* 1986). This may also be the case for MSF-DDE, or perhaps high affinity of MSF-DDEs for binding in tissues such as adrenal cortex (Brandt *et al.* 1990) may compete for storage in adipose tissue, causing the highly selective retention of MSF-DDE in liver.

Levels of parent PCBs in polar bear tissues were only occasionally above the detection limit, and were at least 5-10 times lower than their MSF-metabolites. No MSF metabolites of major PCBs in polar bear tissues were detected. The ratio of total MSF-PCBs to remaining unmetabolized total PCBs was 0.12 in liver and 0.04 in fat. This implies that the MSF-PCBs identified in polar bears have similar persistence to unmetabolized PCBs. All of the MSFs identified had either 25- or 236- chlorine substitution and 3- or 4-MSF substitution on one ring, indicating that a requirement for formation (or perhaps whole-body retention) of MSF metabolites is *meta-para* epoxidation. The absence of MSF metabolites of other metabolizable PCBs may indicate that MSFs are not formed or that they are formed and further metabolized or easily excreted. PCB-52 (25-2'5'-TeCB) is major in seals but has two free *meta-para* positions and may form the *bis*-methylsulfone. The ratio of MSF-DDEs to DDE in liver was 0.17. This probably indicates that DDE is not as rapidly metabolized as the MSF-PCB precursors, but yields a fairly high percentage of MSF metabolites. The presence of the 2- MSF-DDE is unusual. Normally only 3-MSF-DDE is found (Jensen *et al.* 1977, Bergman *et al.* 1993).

The fractional composition of MSF-PCBs in bear tissues was compared to that the fractional composition of the parent PCBs (excluding all other congeners) in ringed seals blubber (Fig. 1) It appeared that PCB-49 (25-2'4'-TeCB) and PCB-87 (25-2'3'4'-PnCB) gave a higher proportion of MSF metabolites than the other PCB congeners, assuming that the MSF-PCBs are not themselves accumulated from the diet. Further studies will address the possibility of bioaccumulation of MSF-PCBs and -DDEs from the diet.

The significance of the relatively high levels of MSF-PCBs in polar bear tissues is unknown, and requires further investigation. Clearly, there is also the potential for humans to be exposed to this class of compounds through ingestion of ringed seal blubber.

Polar bear fat, liver and milk samples in the CWS specimen bank will be analyzed by R. Letcher for the presence of methylsulfone metabolites and their precursors. Samples of other species indicative of high trophic level food chains in the Arctic, such as beluga, narwhal and walrus, and lower trophic levels will also be studied in collaboration with D. Muir, DFO.

In order to understand the potential effect of these contaminants on polar bears, it is necessary to have better information on basic biochemistry of the polar bear as it relates to xenobiotic chemicals, especially the hepatic cytochrome P450 enzyme system responsible for metabolising xenobiotics. We are not aware of any previous studies on P450 enzyme activities in ursids. Although we have found strong indirect evidence of high CYP1A1 (TCDD- inducible) activity in some individual bears through immunoblot analysis (Western Blots) of P450 proteins in polar bear liver, so far we have not measured enzyme activity directly.

Lack of bioaccumulation of normally recalcitrant PCBs and DDE, and the presence of relatively high levels of methylsulfone metabolites of these compounds in polar bear liver is an indication of unusual types/activities of hepatic P450 enzymes. There are two features of polar bear metabolism which may be correlated to this unusual activity: high concentration in liver of stellate cells which are rich in lipid and specialized in (presumably protective) storage of vitamin A, and facultative fasting. Stellate cells are the suspected cause of the highly selective accumulation of intermediate polarity organochlorine metabolites, such as oxychlordane and methylsulfones, in liver. The role of stellate cells in cytochrome P450-mediated metabolism has not been investigated. Facultative fasting is invoked in times of food shortage or inclement weather. It is a metabolic status comparable to hibernation, in which protein is conserved by recycling of urea. There may be a profound effect of this metabolic state on phase one metabolism (CYP450 oxidation) and phase 2 metabolism (conjugation reactions leading to methylsulfones) which affects the ultimate toxicity of chlorinated organics.

Controlled storage studies of immunoreactive P450 protein stability are necessary. A larger sample size of fresh liver samples with a wider range of PCB concentrations and control samples without Telazol exposure are needed to test hypotheses of P450 enzyme induction. Direct measurements of enzyme activity in fresh samples are needed to correlate with immunoblot assay results. This should now be possible because the Telazol study was repeated in April, 1993, and tissue samples from 9 bears were obtained. Because polar bears cannot be killed in Canada except by Inuit hunters, the Telazol study provided an extraordinary opportunity to obtain tissue samples in a controlled fashion minutes after death. It is very difficult to obtain usable samples

for many types of analysis from Polar bears killed by hunters in the usual way because of the large time delays between death and sampling.

Expected project completion date: The first phase of this research, completion of the short term objectives, will be complete by March, 1995. It is anticipated that followup research will be required based on the results of the first phase.

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Table 1. Mean and standard deviation of levels (ng/g lipid) and ratios of methylsulfone (MSF) -PCBs and -DDE, and tris-(p-chlorophenyl)-methanol in polar bear liver and fat (n=5).

	Liver		Fat		
	mean	SD	mean	SD	
3-MSF-25-2'4'-TeCB	151	107	35	13	ng/g
4-MSF-25-2'4'-TeCB	177	93	39	18	
3-MSF-25-2'4'5'-PnCB	185	89	81	32	
4-MSF-25-2'4'5'-PnCB	104	41	70	31	
4-MSF-236-3'4'-PnCB	17	8	9	6	
3-MSF-25-2'3'4'-PnCB	229	111	31	12	
4-MSF-25-2'3'4'-PnCB	116	56	57	25	
4-MSF-25-2'3'5'6'-HxCB	101	59	6	3	
4-MSF-236-2'4'5'-HxCB	240	69	63	31	
3-MSF-236-2'3'4'-HxCB	38	16	4	1	
4-MSF-236-2'3'4'-HxCB	15	6	9	4	
3-MSF-25-2'3'4'5'-HxCB	30	14	7	4	
4-MSF-25-2'3'4'5'-HxCB	15	5	15	7	
(3-MSF-236-2'3'4'5'-HpCB)[1]	29	11	<0.01	<0.01	
(4-MSF-236-2'3'4'5'-HpCB)[1]	20	15	<0.01	<0.01	
s-MSF-PCBs	1467	628	424	174	ng/g
s-PCBs	16133	9854	11247	5518	ng/g
Ratio s-MSF-PCB/s-PCBs	0.1157	0.0738	0.0412	0.0112	
2-MSF-DDE	152	91	<0.01	<0.01	ng/g
3-MSF-DDE	152	59	<0.01	<0.01	
s-MSF-DDEs	303	143	<0.01	<0.01	ng/g
DDE	*1621	*690	646	115	ng/g
Ratio s-MSF-DDEs/DDE	*0.1713	*0.0318			
tris-(p-Cl-phenyl)-MeOH	666	232	0.01	<0.01	ng/g

[1] Tentative chlorine substitution *n=4

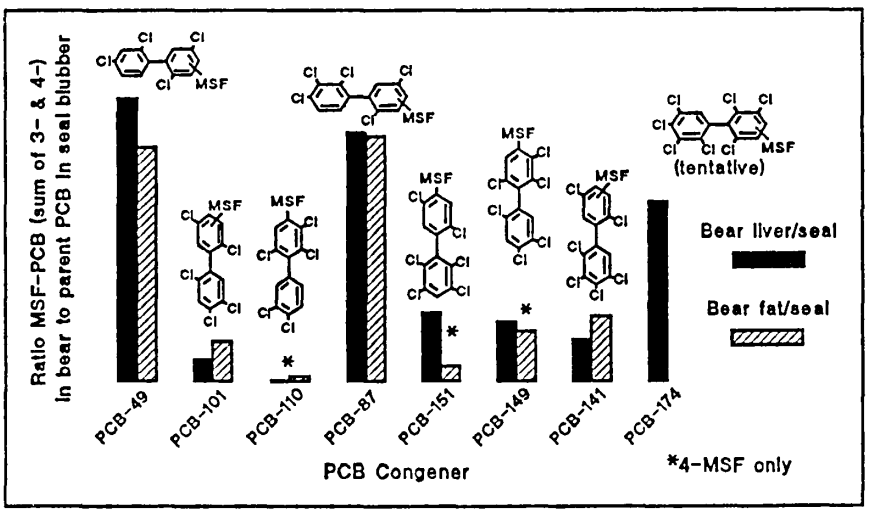

Figure 1. Average Fractional makeup of MSF-PCB congeners in polar bear fat and liver.

Table 2. Sample description, S-PCB levels and relative amounts of immunoreactive protein in polar bear livers determined by densiometric analysis of immunoblots probed with mouse anti-scup P450 1A1 (antibody 1-12-3), mouse anti-rat P450 1A1 and rabbit anti-rat P450 2B1. Note that the amount or microsomal protein is not comparable between samples 1-8 and A-D in the mouse anti-rat P450 1A1 and rabbit anti-rat P450 2B1 immunoblots because samples 1-8 contained cytosolic plus microsomal protein, whereas microsomes were isolated for samples A-D. Liver samples 1-8 were homogenates stored at -40 degrees celcius for 9-10 years before analysis. Samples A-D were fresh and stored at -196 degrees celcius until analysed. Bears A-D had been immobilized with Telazol 28-121 hours prior to death.

Polar Bear	Location	Telazol dose (mg/Kg)	Hours from Telazol dose	Sex	Age	% Lipid	S-PCBs (ug/g lipid)	nmol P450/mg protein	Relative amount of cross-reactive P450 protein in immunoblot assay		
									mouse anti-scup 1A1	mouse anti-rat 1A1	rabbit anti-rat 2B1 +2B2
1	Barrow Strait	---	---	F	11	2.6	27.9	n.d.	<2.0	<0.01	0.089
2	Barrow Strait	---	---	M	4	2.4	36.8	n.d.	2.5	0.057	0.046
3	Barrow Strait	---	---	M	9	3.7	73.6	n.d.	10.6(23.4)*	0.139	0.497
4	Larsen Sound	---	---	F	3	1.6	23.0	n.d.	2.4	0.055	0.022
5	Beaufort Sea	---	---	F	4	11.6	6.6	n.d.	=<2.0	<0.01	<0.01
6	McLure Strait	---	---	M	18	14.3	25.4	n.d.	3.6	<0.01	<0.01
7	Amundsen Gulf	---	---	M	14	8.0	5.9	n.d.	=<2.0	<0.01	<0.01
8	Larsen Sound	---	---	M	8	5.9	8.5	n.d.	<<2.0	<0.01	<0.01
A	Barrow Strait	13.1	121	M	5	4	65.7	1.98	---	0.274	1.689
B	Barrow Strait	8.7	28	M	9	4	32.7	1.01	---	0.149	1.074
C	Barrow Strait	6.5	53	M	22	4	57.6	1.18	---	0.311	1.093
D	Barrow Strait	8.4	53	M	11	4	35.6	1.09	---	0.260	0.908
UNTREATED RAT	---	---	---	---	---	---	---	---	---	0.024	0.761
3-MC TREATED RAT	---	---	---	---	---	---	---	---	---	1.246	0.296
PB TREATED RAT	---	---	---	---	---	---	---	---	---	0.004	13.286

(including second band)*

1 - pmol P450E equivalent per mg microsomal protein.
2 - densitometric analysis of immunoblot probed with mouse anti-rat P450 1A1 IgG (monoclonal at 1ug/mL).
3 - densitometric analysis of immunoblot probed with rabbit anti-rat P450 1B1 (polyclonal at 3ug/mL).

POTENTIAL FOR EFFECTS ON REPRODUCTION, CARCINOGENESIS, MUTAGENESIS AND TERATOGENESIS IN ARCTIC MAMMALS: STATUS OF BIOMARKERS IN ARCTIC SEALS AND WHALES

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PROJECT TEAM: A. Mathieu, L. Fancey and A. Rahimtula, Memorial University

OBJECTIVES

To determine if the levels of contaminants in the Arctic are sufficiently high to engender concerns about reproductive failure and carcinogenesis, mutagenesis, and teratogenesis in marine mammals - the species at highest risk. To assess the present situation and provide a baseline on the critical biological indices that will be of paramount importance for a monitoring/assessment strategy for the Arctic.

DESCRIPTION

Sublethal stressors can elicit physiological, immunological and pathological changes in vertebrates - changes which can lead to cancer, and impairment of growth and reproductive functions. Responses to stressors are preceded by changes at the biochemical/molecular level and such changes can be used as either biomarkers or early warning indicators of impending problems.

Is there a cause for concern for contaminant mediated diseases in arctic animals, such as marine mammals which bioaccumulate inordinate amounts of persistent organochlorines? Investigations have revealed a dramatic decrease in reproductive capacity in ringed, grey and harbour seals in the Baltic (Helle and Stenman 1991). Also, recent studies have demonstrated a high frequency of cancer and other pathological changes in beluga whales from the Gulf of St. Lawrence (Martineau *et al.* 1988).

It is proposed to investigate the status of three important markers of biological damage in beluga whales, seals and walrus from Arctic and Labrador waters. Where possible, such as for belugas, harbour seals and ringed seals, comparisons will be made in reference to animals from more heavily contaminated areas, namely the Gulf of St. Lawrence and the Baltic and Wadden Seas.

Given the known effects of contaminants on marine mammals from the North and Baltic Seas and the Gulf of St. Lawrence, the biomarkers identified as being important include those for reproduction (e.g. hormone receptor numbers), cancer/mutagenesis (e.g. DNA-adducts, DNA oxidation and ras-oncoproteins) and general pathology (e.g. porphyria).

Reproductive Effect Biomarkers

Steroid receptors

The reproductive effects seen in seals from the North Sea are believed to be caused by the high level of organochlorines in their diet. Basic studies indicate that the molecular basis for reproductive effects may be at the hormone receptor level, which is decreased in animals exposed to organochlorines such as polychlorinated biphenyls (Romkes *et al.* 1987, Korach *et al.* 1988).

Vitamin A

Vitamin A is a critical vitamin for the maintenance of epithelial cells and as such is directly involved in modulating viral and bacterial infections, reproductive disorders, and other pathologies. Exposure of animals to elevated concentrations of chlorinated organics such as PCBs can result in depression of Vitamin A levels. Studies carried out by Dutch scientists suggest that the lethal viral infections and reproductive disorders in seals and other marine mammal populations in the Baltic, North and Wadden Seas are linked to Vitamin A and thyroid hormone imbalances. It is postulated that such imbalances (Brouwer *et al.* 1989) can be affected through consumption of fish having elevated concentrations of chlorinated hydrocarbons.

Cancer/Mutagenesis Biomarkers

Sophisticated techniques are presently available for measuring pollutant mediated DNA damage. DNA damage can occur directly through covalent bonding of contaminants (or contaminant by-products) to selective nucleotides forming so-called DNA adducts. Alternatively, damage can occur indirectly through the formation of powerful oxidizing radicals which degrade nucleotide bases in DNA in a more or less characteristic manner. Organic compounds are primarily involved in DNA adduct formation while both heavy metals and organic compounds can be involved in radical formation.

Both chemical mediated DNA adduction and DNA oxidation are linked with organism carcinogenesis, mutagenesis and teratogenesis. (In this regard it is worth noting that DNA damage is most often emphasized but a theoretical basis exists for damage to other cellular macromolecules). Techniques for measuring both DNA adduction and radical produced DNA damage have been established in a Fisheries and Oceans lab in St. John's, Newfoundland in the past 3 years.

Ras-oncoprotein

Many reports have shown that interaction of a variety of chemical carcinogens with DNA can lead to specific mutations resulting in oncogene activation (Zarbl *et al.* 1985). Results obtained with carcinogen-induced tumours and transgenic mice have indicated that ras-oncogene can participate in the initiation of carcinogenesis (Stowers *et al.* 1987). The overexpression of total proto-oncoproteins such as ras and myc have been implicated in the pathogenesis of a number of neoplastic conditions (Rayter *et al.* 1989).

Porphyria

Porphyrins are building blocks for heme proteins, the most commonly known being oxygen binding hemoglobins, cytochromes and muscle myoglobins. Heme biosynthesis is carefully regulated and levels of porphyrins and their precursors are normally quite low. Disturbances in heme metabolism can lead to the build-up of excessive levels of various porphyrins, a condition clinically defined as porphyria. Porphyrins can be hereditary but are also expressed under various disease conditions or chemical insult. In this regard it is of particular interest that evidence has been obtained for porphyria in some birds in the Great Lakes (Fox *et al.* 1988).

ACTIVITIES IN 1992/93

DNA Adducts

These studies include exploring for "bulky" aromatic type adducts in DNA by the P-32 postlabelling assay or specific tetrols by HPLC/fluorometric techniques. In 1991/92 we reported on the relatively high concentrations of "bulky" adducts in the muscle and brain of seals from waters around Newfoundland and Labrador. We now believe that these adducts are primarily natural in origin but additional samples of beluga tissues were obtained from the Arctic in 1992/93 and will be analyzed in 1993/94.

This year considerable effort was directed towards exploring for specific benzo(a)pyrene adducts (released as tetrols) in the DNA of beluga from the Arctic and the Gulf of St. Lawrence. Investigators from the Oak Ridge National Laboratory have produced evidence for such adducts in the DNA of beluga from the Gulf of St. Lawrence. This discovery drew considerable attention, including in the national press. In spite of considerable effort, we have not been able to confirm the findings of the Oak Ridge group. The DNA studies being negative, we speculated on the potential for finding adducts associated with protein. Exploratory studies were carried out with mice and it was observed that protein was a suitable target for adducts in animals exposed to benzo(a)pyrene (Fig. 1). However, as for DNA, we have been unable to find evidence for adducts in the protein of beluga whales, either from the St. Lawrence or the Arctic. A sample chromatogram is presented in Figure 2. None of the peaks reliably correspond to standards. It is also worth noting that the peaks on the chromatogram represent tetrol equivalents in the 5 picogram range, indicating that if any peaks were bona fide tetrols, their absolute amounts would be very low.

These findings are significant in that they bring into question earlier studies about benzo(a)pyrene adducts in beluga whales.

Ras-oncoproteins

An historical method was established for the detection of ras-oncoproteins by fluorescence microscopy.

Vitamin A

In 1991/92 we obtained evidence for altered Vitamin A levels in beluga from the Gulf of St. Lawrence. These studies were carried out by fluorometry and this year a more specific technique namely HPLC/fluorometry was applied to measure retinol, the biologically active form of Vitamin A, as well as the storage form (fatty acid conjugates) in liver tissues. Both retinol and retinol palmitate were markedly lowered or depleted in 4 out of 6 whales from the Gulf of St. Lawrence (Figs. 3 and 4). Since it is likely that the whales from the Gulf of St. Lawrence were dead for longer periods before sample preservation, it was important to address Vitamin A stability in unpreserved tissues. Accordingly, exploratory studies were carried out on the stability of the vitamin in decomposing beef liver. It was of interest to note that Vitamin A levels were not markedly altered in beef liver left to decompose for 2 - 3 days at room temperature, indicating that the lower vitamin levels found in whales from the Gulf are unlikely a sampling artifact.

Altogether, strong evidence has now been provided indicating that Vitamin A status may be an important health effects response to assess in connection with chemical insult in marine mammals. Also, at this time, chemically mediated Vitamin A depletion may be seriously affecting the health of beluga whales in the Gulf of St. Lawrence. If this is the case, it is of major ecotoxicological importance.

Porphyrins

Total porphyrins as measured by fluorometry (in protoporphyrin equivalents) were shown to be elevated in some liver samples of beluga from the Gulf of St. Lawrence in comparison with the Arctic (Fig. 5). This indicates the possibility of chemically mediated porphyria in whales from the St. Lawrence.

One of the most interesting observations made in 1992/93 was in association with a brown chromophore in liver tissues. Five of the six whales from the Gulf of St. Lawrence had elevated levels of this pigment (Fig. 6). We are presently pursuing the possibility of this pigment being a degraded heme and a possible indicator of liver injury associated with blood coagulation.

CONCLUSIONS AND FUTURE DIRECTIONS

Evidence has been produced indicating that a reasonable case can be made for use of biomarkers in assessing the health of marine mammals. Investigations will continue on beluga in 1993/94 and will be expanded to include studies on stress proteins. It is also hoped to obtain suitable samples of either harbour seals or ringed seals from "contaminated" and reference sites.

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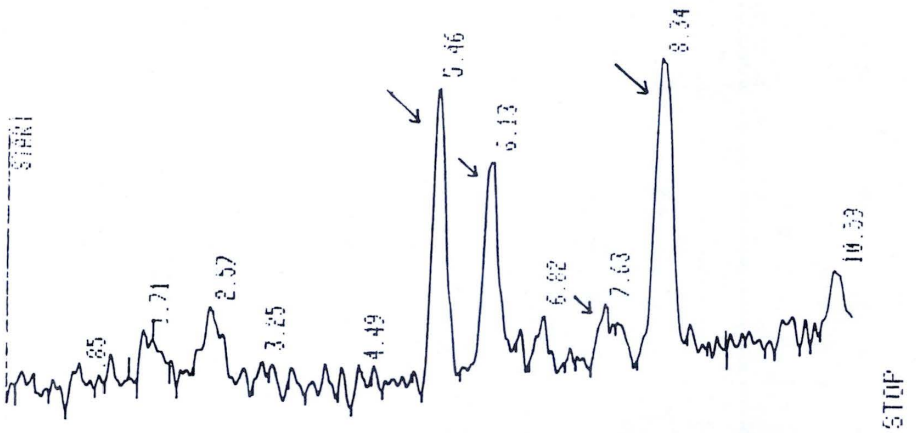


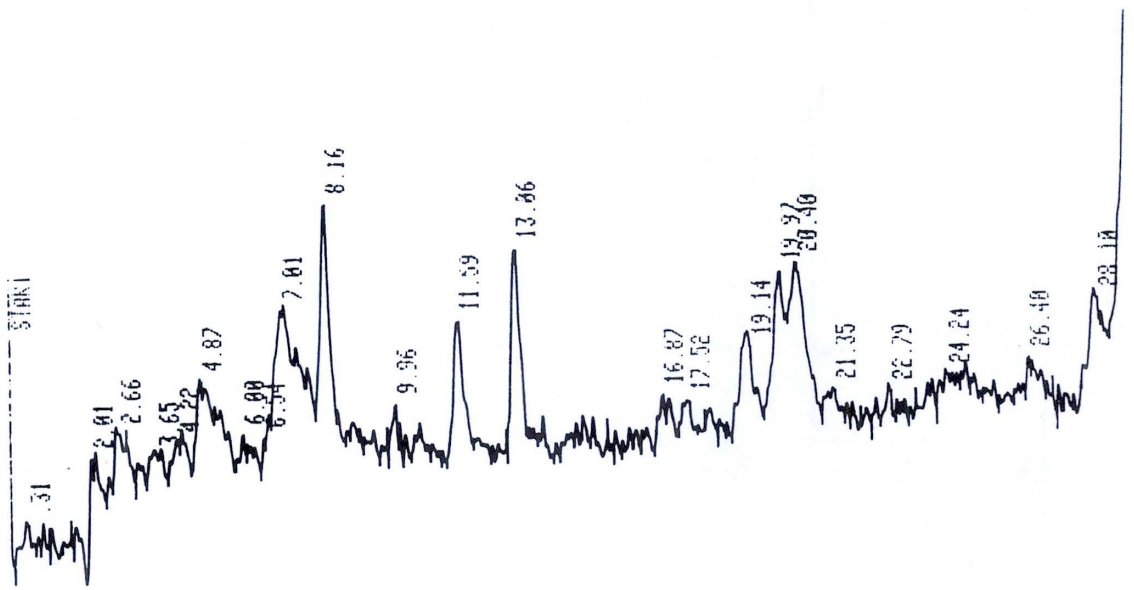
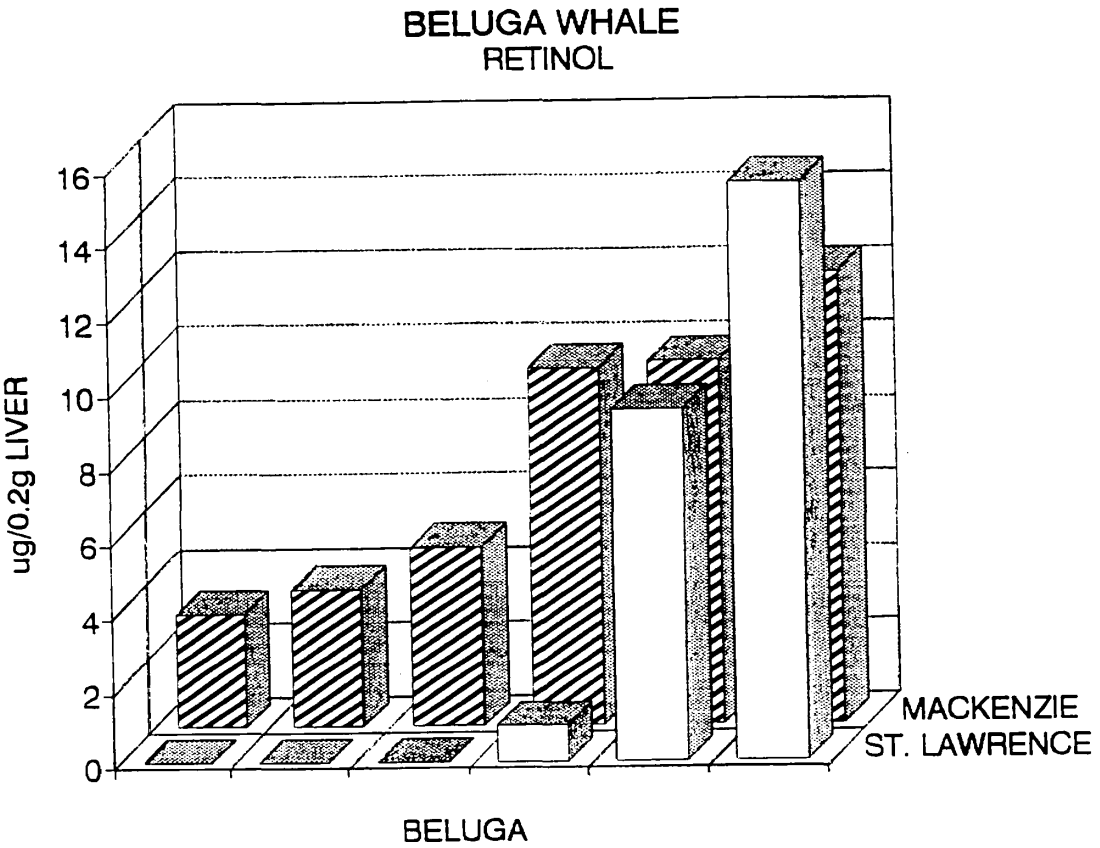
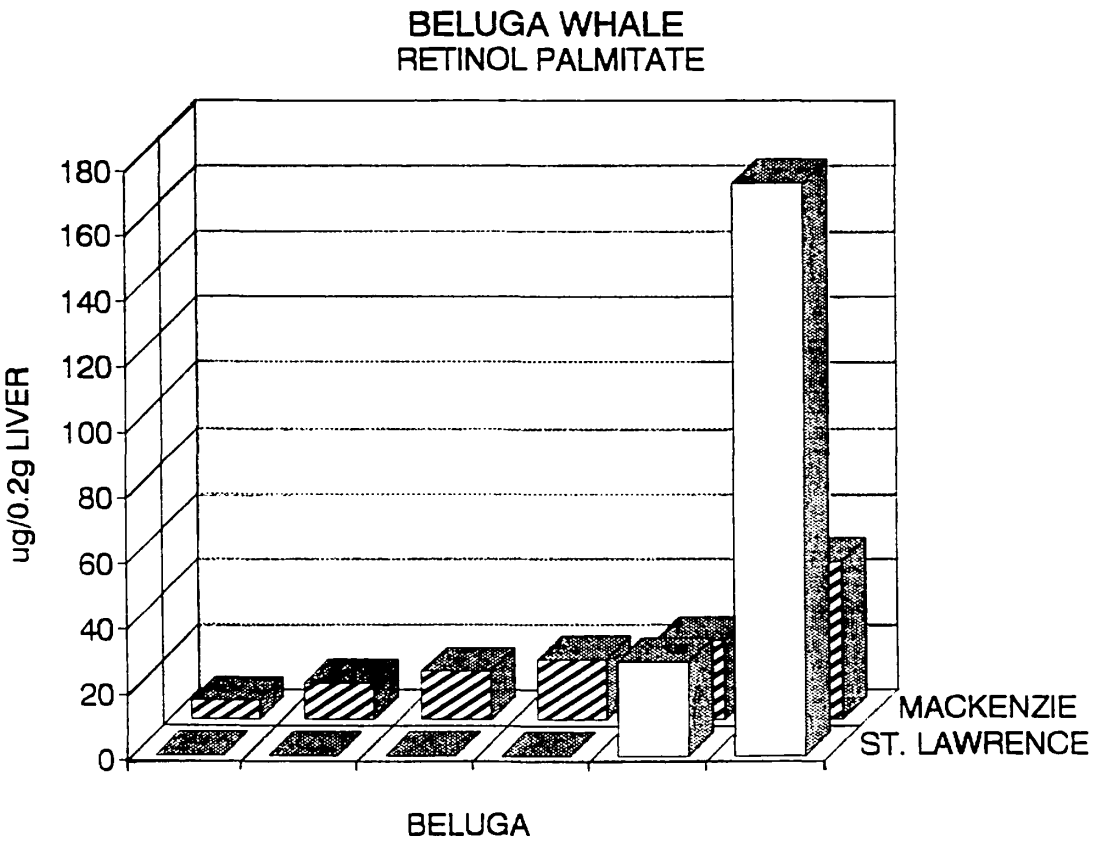


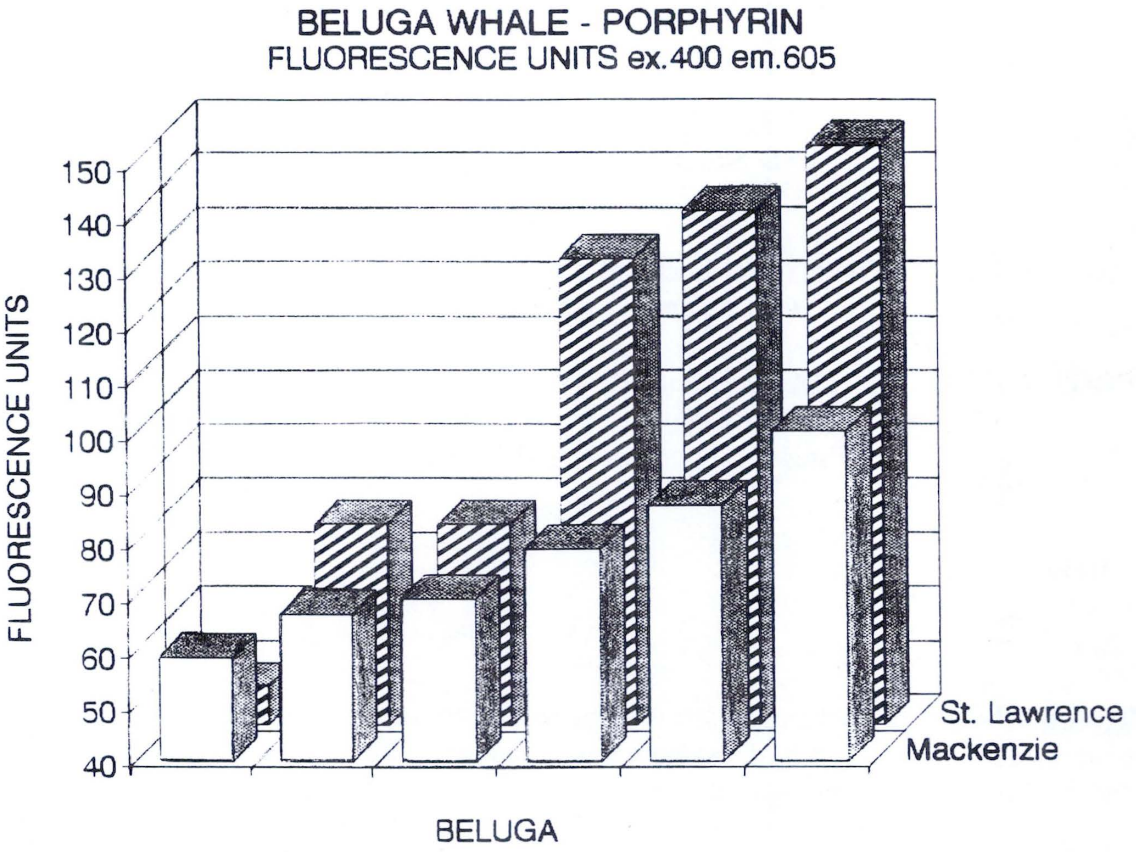
Figure 1 (top legend): HPLC profile of an hydrolysate of liver proteins from rats injected with benzo(a)pyrene. The 4 peaks noted by arrows correspond to the 4 tetrol standards. Peaks approximate 5 picograms of standards.

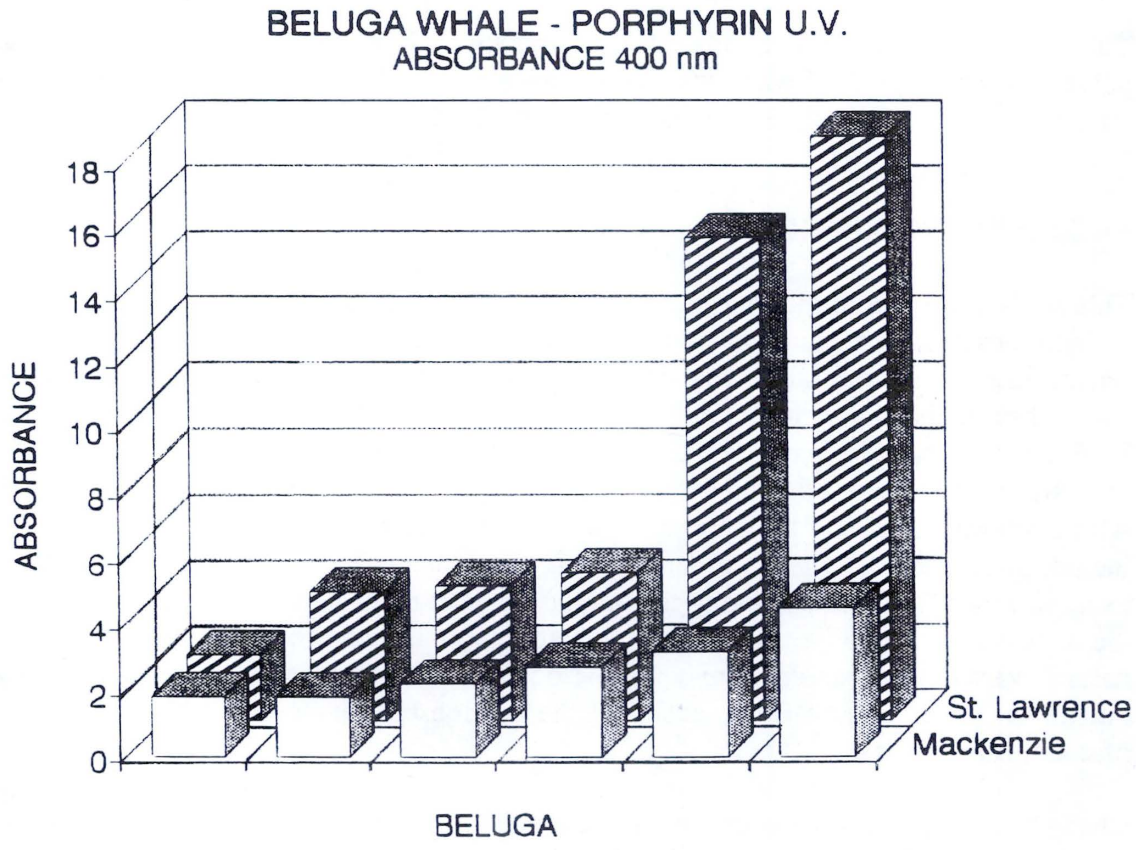
Figure 2 (bottom legend): HPLC profile of an hydrolysate of liver proteins from a beluga whale. None of the resolved peaks correspond to benzo(a)pyrene tetrol standards. The solvent system used here was different from that used for Figure 1, but the system used in Figure 1 as well as other solvent systems also gave negative results.





Retinol (Figure 3, top legend) and retinol palmitate (Figure 4, bottom legend) levels in liver tissues of beluga whales from the Gulf of St. Lawrence and the MacKenzie Delta.





Porphyrin in protoporphyrin equivalents (Figure 5, top legend) and an unknown "porphyrin" referred to as a brown chromophore (Figure 6, bottom legend) in liver tissues of beluga whales from the Gulf of St. Lawrence and the MacKenzie Delta.

BIOMARKERS AND STRESS EFFECTS IN ARCTIC MARINE MAMMALS

PROJECT LEADERS: W.L. Lockhart and R.E.A. Stewart, Fisheries and Oceans Canada, Central and Arctic Region, Freshwater Institute, Winnipeg, Manitoba

PROJECT TEAM: D.C.G. Muir, R. Wagemann, D.A. Metner, R.W. Danell, D. Kenny (contractor), C. Ford (contractor), J.Garlich-Miller (contractor), B. Stewart (contractor) and B. Thomson

OBJECTIVES

Short-term

To define levels of biochemical stress responses in marine mammals as functions of contaminant concentrations measured in the same animals, and to define the influences of normal biological and habitat variables (eg. sex, age, size, etc.) on the ranges of bioindicator values.

Long-term

To understand the implications of contaminants (PCBs, chlorinated dioxins and furans, polycyclic aromatic hydrocarbons, heavy metals) for the health of individual arctic marine mammals, using biomarkers (mixed-function oxidases, steroid hormones).

PROJECT DESCRIPTION

This study examines arctic marine mammals for indicators of biological responses (biomarkers or bioindicators) to contaminants. The bioindicators selected for these animals have established sensitivities (in several laboratory animals and in some experiments with marine mammals) to some of the same contaminants measured in arctic marine mammals and habitat, notably several PCB congeners, chlorinated dioxins and furans and polycyclic aromatic hydrocarbons. The biomarker being used is the microsomal monooxygenase system since it has been associated both with exposures to planar aromatic compounds and with a cascade of other biological responses including reproduction and immune system functions. The study is in two parts. Part one examines the relationships among the biomarkers and other natural biological measures such as the annual reproductive cycle. It is anticipated that the biomarkers themselves will show some natural variation depending on other biological circumstances (e.g. reproductive condition, nutritional status, etc). Part two examines the relationships among the biomarkers and measures of chemical residues.

Changes in populations do occur, but there is currently no clear methodology to establish whether contaminants have or do not have a role in these changes. Cause/effect relationships cannot be demonstrated or refuted without knowing and demonstrating steps between exposure and ecological change. Examination of the animals for indicators of subtle responses known to

be associated with the contaminants found in them is the best way to confirm or refute the hypothesis that trace contaminants are acting biologically on them. Establishing the natural range and variation in the biomarker measures is essential to the interpretation of these measures. Samples of ringed seals are collected specifically for this study (with sub-samples of tissues being supplied to other projects)in order to quantify the range and variation within this species over a yearly biological cycle. Samples of other species are obtained as opportunities permit with the intent of building up a grid of normal biomarker values for marine mammals throughout the Canadian Arctic, and beyond if cooperative studies can be arranged.

FIELD ACTIVITIES IN 1992/93

In order to do the biochemical analyses, fresh tissue samples must be fast-frozen at very low temperatures (on dry ice or liquid nitrogen). This creates obvious logistical problems when operating in remote locations since dry ice must be shipped to the location and returned to the laboratory before it sublimes and the samples thaw. During 1992, two collections of ringed seals were obtained successfully from near Arviat, and analyses of the first of these have been completed. Collections planned for Pangnirtung and Foxe Basin were less successful, partly because of unseasonable weather and partly because of difficult arctic shipping schedules. Some samples were obtained as planned but thawed before arrival at the laboratory, and some samples arrived safely. Late in the fiscal year the sampling difficulties in Pangnirtung were overcome and regular shipments of samples became available. Sampling for the seasonal variation component of the program was underway until the end of the fiscal year. The intent remains to examine seasonal and spatial variation in the bioindicator responses extending through one annual cycle; this would have required sampling through until the fall of 1993. Tissues from Arviat were supplied by the Government of the Northwest Territories and Canadian Wildlife Service. Those from Pangnirtung were being collected by a native hunter (contracted through the Hunters and Trappers Association) trained in the type of sampling required. This project also supplied samples for a number of other types of analyses (organochlorines, metals, metallothioneins). It would have been the first systematic examination of the influences of seasonal, biological, and habitat variables on these biochemical stress indicators in marine mammals. Along with the systematic seasonal studies at Pangnirtung, other populations and species were being sampled at lower intensity to form a grid of biomarker values over as much as possible of the Canadian Arctic. With the cancellation of the project, samples collected but not yet analyzed are being retained for analysis if the project is restored.

LABORATORY ANALYSES DURING 1992/93

The May/June collections of ringed seals from Arviat were analyzed for liver microsomal ethoxyresorufin-O-deethylase (EROD) and aryl hydrocarbon hydroxylase (AHH) activities. Cytochrome P-450 difference spectra were also obtained for most samples. Aminopyrene-N-demethylase activities have been determined for some of the samples, as a representative phenobarbital-type activity. With the cancellation of the project, the samples on hand but not yet analyzed have been stored at -60°C in the hope that some future opportunity will permit their analysis. We have conducted a long-term storage study on liver from seals collected from Sable

Island in 1984 which indicates little deterioration in activity over several years under these storage conditions.

RESULTS

The analysis of the spring Arviat collection has given us the first opportunity to examine the relationships between biological variables (size, sex) and enzymatic activities since it is the first collection of a large enough group of animals from the same time and place. Somewhat surprisingly, the sexes were virtually identical in both EROD and AHH activities (Table 1). Residues of inducing contaminants like PCBs are frequently higher in male marine mammals than in females, and levels of these enzyme activities are also frequently higher in males. However, there was no indication of increased enzyme activities in the males from this collection.

Table 1. EROD and AHH values for ringed seals from Arviat collected in May/June, 1992. (These seals breed in April/May).

Activity	Males Mean	SD	N	Females Mean	SD	N
EROD	0.160	0.163	42	0.141	0.147	32
AHH	0.081	0.062	39	0.084	0.066	28

A valuable feature of this collection was the record of exact times elapsed between the death of each animal and the fast freezing of its liver. Unfortunately, these times were often quite long. When working in the laboratory it is normal practice to take and fast-freeze liver tissue within a few seconds of the death of the animals. However, when working with large animals in the wild, there are unavoidable delays in recovering and dissecting animals after they are shot. The accurate records of this time interval between death and freezing have allowed us to evaluate the importance of this interval for the first time. Analysis of covariance using PROC GLM of SAS incorporating log length, log weight, age, sex, and delay between death and freezing indicated that only the time delay influenced EROD activity significantly. There were weak negative correlations between these time intervals and the enzymatic activities. For EROD the correlation coefficient was -0.39 ($N=70$, $p=0.0009$) and for AHH it was -0.51 ($N=64$, $p=0.0001$), suggesting some loss in activity during the interval. Inspecting the plots in Figures 1 and 2 suggests that little change occurred within the first 2-3 hours. Clearly, it would be desirable to reduce or eliminate this post-mortem loss, but it is often unavoidable when sampling large animals under field conditions. The mean time interval was about the same for males (9.9 hr) as for females (8.3 hr).

In view of the apparent effects of delayed freezing on subsequent measurement of EROD or AHH activity, the means were recalculated on the smaller group of 8 males and 8 females for which the time interval was less than 1 hour (Table 2). Taking only these values and applying

the analysis of covariance, age ($p=0.0018$ for EROD; $p=0.01$ for AHH) and sex ($p=0.01$ for EROD; $p=0.05$ for AHH) influenced the enzymatic activities, but the time between death and freezing did not.

Table 2. Mean EROD and AHH activities in ringed seals from Arviat in May/June, 1992, for which the interval between death and freezing was less than 1 hour. (These seals breed in April/May).

Activity	Males Mean	SD	N	Females Mean	SD	N
EROD	0.335	0.189	8	0.267	0.175	8
AHH	0.157	0.069	8	0.137	0.056	8

A smaller group of twelve ringed seals was also obtained from Arviat late in October, 1992. Analyses of these seals are not complete yet. The mean EROD value for the six females in this group was 0.095 while that for the six males was 0.077, both somewhat lower than the May/June values. These samples had lower average times between death and sampling than the spring samples, the longest interval being 5.75 hours. Since the spring values are probably biased low by the preponderance of samples with long pre-freezing intervals, the difference between spring and fall activities may be more striking than the means suggest. Only 3 females and 1 male met the arbitrary criterion of a death-to-freezing time under 1 hour, and for these the respective EROD means were 0.117 ($\pm$ 0.039) and 0.174, also substantially lower than the May/June values. These preliminary data show about two-fold differences in activities between May/June and October for this population. The activities were higher in the breeding season; in fish the activities are often reported to be lower during the spawning season. In May/June the ringed seals would be pupping, mating and moulting; in October they would be feeding but doing nothing obvious beyond that.

Twenty-one of the spring Arviat seals have been analyzed for PCB levels in blubber, and there were no statistical linkages suggested between the ΣPCB and the EROD or AHH activities. The times between death and freezing for the 21 individuals ranged from 2.5 to 25 hours, and so none met the arbitrary criterion of one hour or less. It will be recalled that such a relationship was demonstrated in the Husky Lakes beluga whales reported last year. However those whales had been starving for some months before being sampled and had lost significant quantities (about 200 kg each) of blubber. (The beluga whales were almost all sampled within the 1 hour criterion). The Arviat seals had not been undergoing any starvation period. The working hypothesis remains that the effects of PCBs and other planar contaminants with inducing activities would show up when blubber is mobilized for metabolic use such as in a year with limited food availability. In "good" years, the PCBs or other fat-soluble contaminants would remain in the blubber and not act toxicologically on the seals. Blubber is also mobilized during lactation, but we have not seen an indication that this results in any induction of enzyme activity in nursing mothers. It seems probable that blubber mobilized for milk production results in a

direct transfer of both calories and contaminants to the nursing pup, with little or no impact on the mother. However, when blubber is mobilized as a calorie source for the adult, it appears that the contaminants exert their effects.

Late in the fiscal year our first samples of ringed seals were obtained from Pangnirtung. Earlier shipments were lost due to the difficulties of shipping dry ice to and from northern communities. These seals were taken between January 29 and February 13, 1993 and they are the freshest samples obtained to date, all with death-to-freezing time intervals of less than 10 minutes. The values obtained are listed in Table 3.

Table 3. EROD and AHH values for ringed seals from Pangnirtung collected in January/February, 1993. (Breeding season approaching)

Activity	Males Mean	SD	N	Females Mean	SD	N
EROD	0.200	0.089	29	0.186	0.100	23
AHH	0.133	0.040	30	0.128	0.052	29

Comparing the Arviat seals taken in May/June of 1992 and the Pangnirtung seals from January/February of 1993, the enzymatic activities are quite similar. The Pangnirtung samples have slightly higher mean values which probably reflects their short death-to-freezing time period. They also show somewhat lower variation as judged by the standard deviations as proportions of the means, and this would be consistent with reduction of a sampling artifact. The sampling methods employed in Pangnirtung, with the short death-to-freezing interval are clearly preferable. Even here, with the improved sampling methods, the males and females were essentially the same for both EROD and AHH activities. When the Pangnirtung seals are compared with the restricted group of Arviat seals with short death-to-freezing intervals, the Pangnirtung seals are substantially lower.

Six walrus were also obtained from Foxe Basin. Again, other samples were obtained but only six arrived in Winnipeg in a condition suitable for analysis. Although sample numbers were very small, the values shown in Table 4 suggest that males exceeded females by about a factor of two.

Table 4. EROD and AHH values for walrus from Foxe Basin collected on July 28/29, 1992.

Activity	Males Mean	SD	N	Females Mean	SD	N
EROD	0.224	0.270	4	0.097	0.074	2
AHH	0.113	0.129	4	0.045	0.023	2

Samples of the beluga whales reported in 1992/93 were sent to J. Stegeman at Woods Hole, for some additional analyses. As a first step in the joint study of these animals, the results for EROD and AHH analyses by both laboratories are shown in Figures 3 and 4, respectively (unpublished Woods Hole data from R. White, May 6, 1993). The AHH activities by the two laboratories agree well, but the Winnipeg EROD activities were consistently higher than those from Woods Hole. With both activities the two sets of results correlated closely, with r values of +0.96 for EROD and +0.94 for AHH. With a given set of samples, then, both laboratories would reach the same conclusions although they would do so on the basis of different EROD values. The basis for the difference in EROD activities is currently unknown. Additional comparisons are underway.

CONCLUSIONS

The findings from this project suggest a number of conclusions:

1. Most species exhibit a sexual difference in MFO enzyme activities, at least at some times of the year; these were detected in ringed seals from Arviat in May/June (1992) but not in those from Pangnirtung in January/February (1993).

2. Ringed seals have seasonal differences in MFO activities.

3. The effects of MFO inducers stored in blubber may become apparent when blubber is mobilized as a calorie source. That is, the importance of these contaminants to the animals may be apparent only in a season of limited food availability or fasting. In this event, we should expect to find linkages to processes of reproduction and disease resistance.

FUTURE ACTIVITY

With the cancellation of this program, the only future activity remaining is the analysis of the final Pangnirtung samples and reporting of results through conferences, articles or other means as may be appropriate. If the reviewers opt to renew the project, then the continuity of the Pangnirtung collections has now been lost and it will have to be reestablished by starting the collections over. In this event, we shall be able to examine not only the seasonal variation but also the year to year variation by comparing new collections with the existing ones reported here and those awaiting analyses.

In view of the indication of post-mortem changes in these activities using conventional catalytic activities to perform the assays, it would be useful to adopt assays based on antibody reactions since these might be more resistant to post-mortem changes. However, to our knowledge, there is not yet a suitable antibody preparation available commercially and so this remains a longer term goal. In the interim, future collections should be done in ways to keep the death-to-freezing interval under an hour, as was the case with the Pangnirtung samples.

Reports

Some aspects of the results of these studies have been presented as follows:

W. L. Lockhart, D.A. Metner, D.C.G. Muir and R.E.A. Stewart. 1992. Mixed function oxygenase catalytic activities and PCB congener residues in a group of beluga whales subject to starvation in the western Canadian Arctic. Presented at the Seventh International Bioindicators Symposium and Workshop on Environmental Health, Kuopio, Finland, September 28 - October 3.

W. L. Lockhart, D.A. Metner, D.C.G. Muir and R.E.A. Stewart. 1992. Hepatic microsomal oxygenase activities and PCB congener residues in a group of beluga whales subject to starvation in the western Canadian Arctic. Presented at the 19th Aquatic Toxicity Workshop, Edmonton, October 4-7.

R.D. White, M.E. Hahn, J.J. Stegeman and W.L. Lockhart. 1992. Catalytic and immunochemical analysis of cytochrome P450 in the beluga whale. Presented at the 13th annual meeting, Society of Environmental Toxicology and Chemistry. Cincinnati, Nov. 8-12.

W.L. Lockhart, D.C.G. Muir, R. Wagemann, G.J. Brunskill and T. Savinova. 1993. Methods for the assessment of the effects of chemicals in cold climates. R.A. Tardiff (ed.). Book Chapter, UNEP, Scientific Committee on Problems in the Environment (SCOPE), Scientific Group on Methodologies for the Safety Evaluation of Chemicals. J. Wiley & Sons, In Press.

W.L. Lockhart. 1993. Approaches to the evaluation and regulation of chemical contaminants in the Canadian Arctic. Presented at The International Nordic Symposium on Chemicals in the Arctic Boreal Environment, Helsinki, May 12-14.

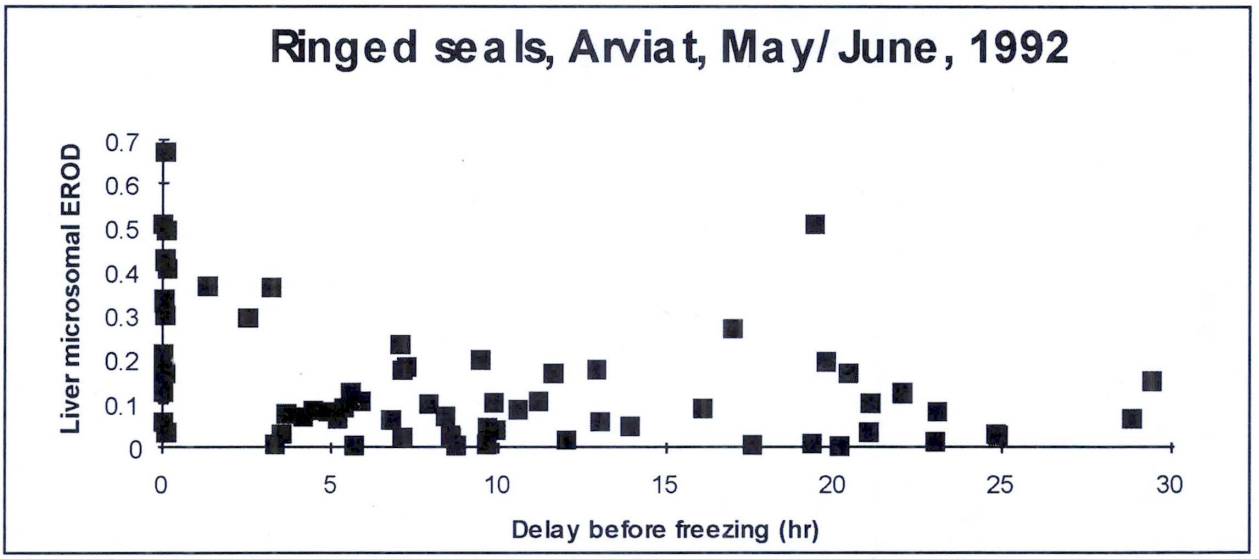


Figure 1 Liver microsomal EROD activities as a function of the time between death and fast freezing of liver from ringed seals, Arviat, NWT, May/June, 1992.

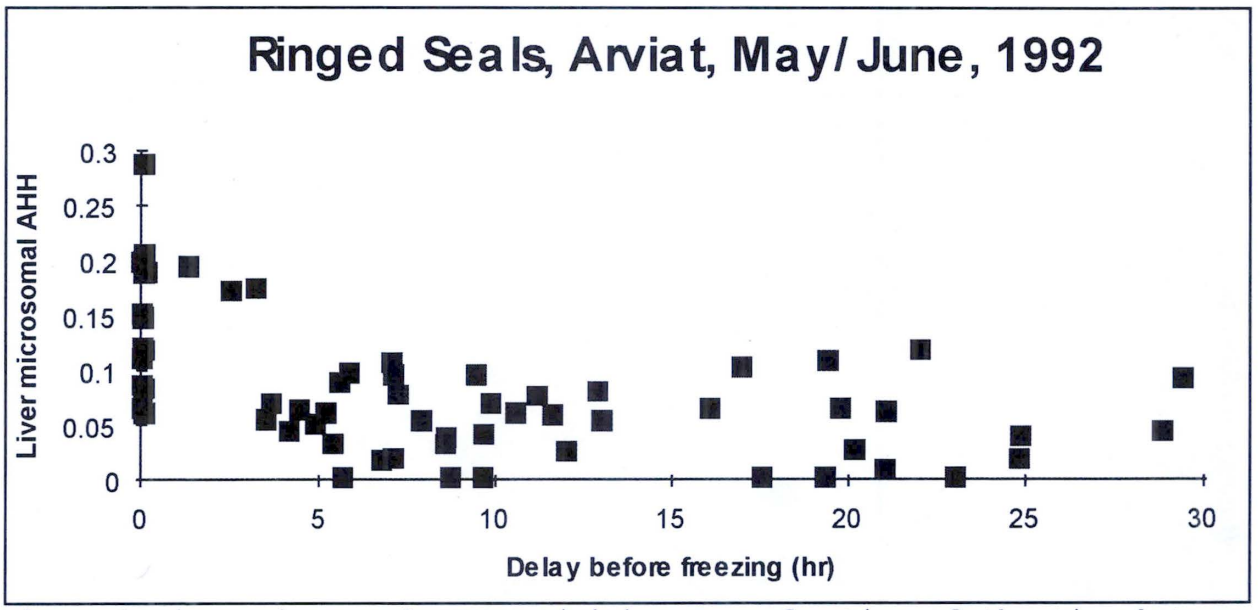


Figure 2 Liver microsomal AHH activities as a function of the time between death and fast-freezing the liver of ringed seals from Arviat, NWT, May/June, 1992.

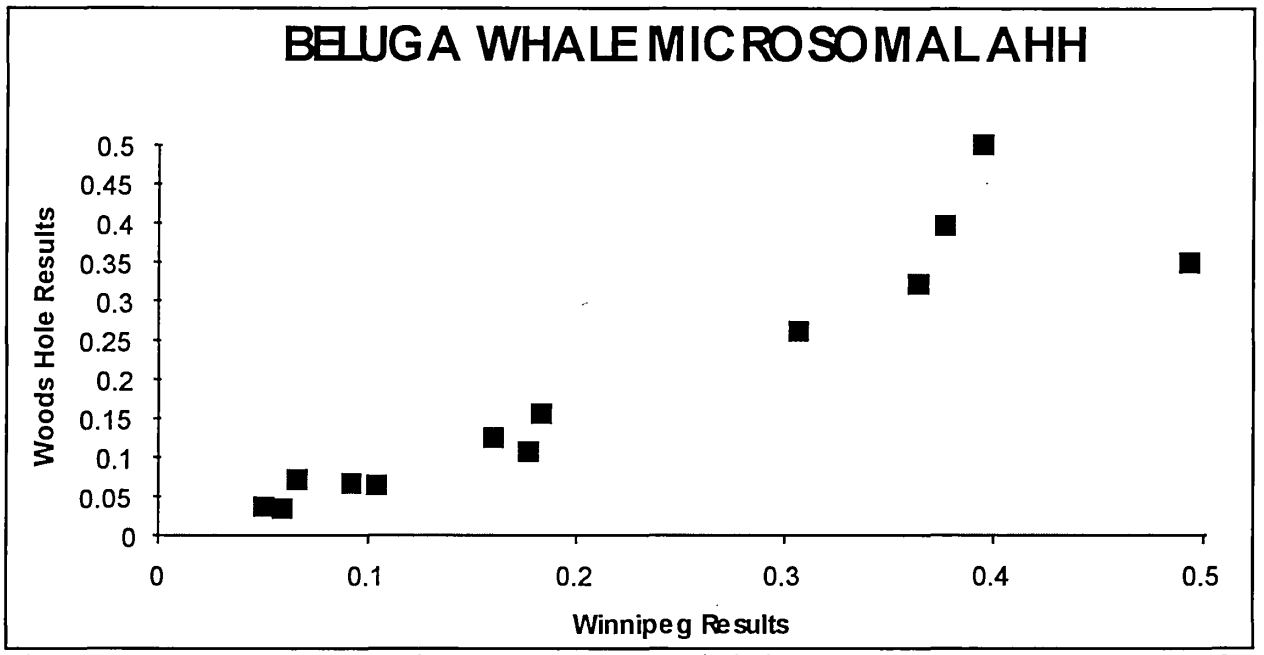


Figure 3. Beluga whale microsomal AHH activities: results from analyses by Woods Hole Oceanographic Institution (R. White) and Freshwater Institute (D. Metner).

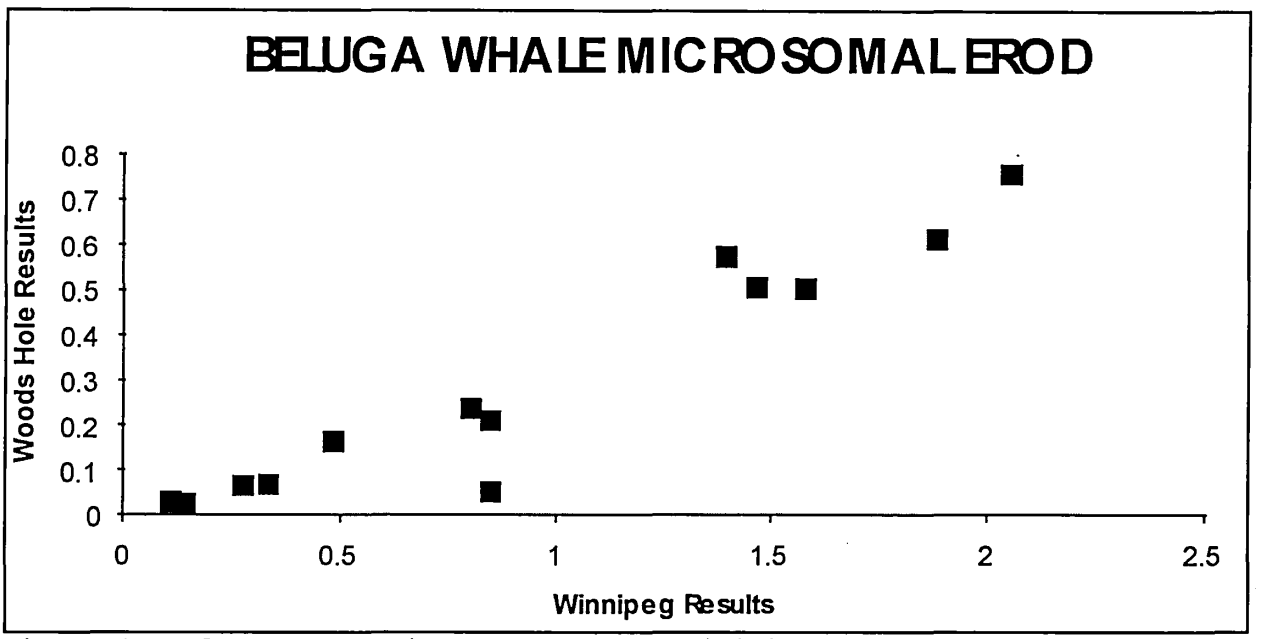


Figure 4. Beluga whale microsomal EROD activities: results from analyses by Woods Hole Oceanographic Institution (R. White) and Freshwater Institute (D. Metner).

EVALUATION OF BIOCHEMICAL METHODS TO ASSESS IMMUNOSUPPRESSION IN SEALS BY PCBS

PROJECT LEADER: R.F. Addison, Institute of Ocean Sciences, Fisheries and Oceans Canada, Sidney, British Columbia

PROJECT TEAM: P.S. Ross, Dalhousie University, Halifax and National Institute for Public Health and Environmental Protection (RIVM), Bilthoven, The Netherlands.

OBJECTIVES

To assess the possible immunosuppressive effects of PCBs in seals by relating PCB concentrations in serum to levels of thymulin.

DESCRIPTION

Thymulin (facteur thymique sérique: FTS) is an oligopeptide which is believed to control the maturation of the T-lymphocytes which play an important role in immune function. Experimental evidence shows that high burdens of organochlorine (OC) compounds may reduce immune function in various species of animals, but there is so far no firm evidence that immune function is compromised by OC concentrations in animals from the wild. This project attempts to apply a newly-developed Enzyme Linked Immuno-Sorbent Assay (ELISA) to measurements of thymulin in serum of grey seals from Sable Island, and to correlate the thymulin data with serum OC concentrations, which we know from previous work (Addison and Brodie 1987) to be related to body burdens. Grey seals were chosen since they have relatively high OC concentrations; if the approach is successful in the grey seals, it will be applied to Arctic ringed seals.

RESULTS

During 1992/93 an ELISA method was developed for thymulin. This was based on rabbit anti-thymulin (prepared by harvesting the anti-serum from rabbits treated with thymulin (Sigma)) linked to a standard colorimetric assay. Preliminary studies on rat serum have shown that thymulin can apparently be detected in samples of 5 ml serum.

During January 1993, approximately twenty grey seals were sampled on Sable Island, N.S. Serum was prepared on Sable Island and was immediately frozen on dry ice and shipped to the Bedford Institute of Oceanography (BIO) in Dartmouth, Nova Scotia. Samples were divided and one half was retained and analysed at BIO for OC residues. The other half was shipped to the Netherlands for thymulin analysis.

Results of the OC analyses show that residue concentrations are in the same range as those reported previously from this population of seals. Thymulin analyses are still being carried out. Once they are completed, the two datasets will be correlated.

UTILIZATION OF RESULTS

A significant correlation (either positive or negative) between OC residue and thymulin concentrations in the Sable Island grey seals would be extremely important in our general assessment of the environmental significance of OC contamination. On the other hand, the absence of any significant correlation, particularly in a seal population containing high OC burdens, would suggest that Arctic ringed seals (which have much lower OC burdens than the grey seals) are unlikely to have their immune systems compromised, and by extension, would be unlikely to suffer any adverse effects from current OC concentrations.

Expected project completion date: Mid 1993 for this phase of the work. If a significant correlation is established between thymulin and OC concentrations in grey seals, a similar study on Arctic ringed seals will be planned.

REFERENCE

Addison, R.F. and P.F. Brodie. 1987. Transfer of organochlorine residues from blubber through the circulatory system to milk in the lactating grey seal *Halichoerus grypus*. Can. J. Fish. Aquat. Sci. 44: 782-786.

METALLOTHIONEIN IN ARCTIC MARINE MAMMALS AND FISH

PROJECT LEADER: R. Wagemann, Fisheries and Oceans Canada, Freshwater Institute, Winnipeg, Manitoba

PROJECT TEAM: R. Wagemann, G. Boila, L. Wesson, D. Savoie, (contractor), L.W. Lockhart

OBJECTIVES

1. To determine the natural ranges of metallothionein in tissues of ringed seals, harp seals, belugas and narwhal from the Canadian Arctic.

2. To determine the sub-cellular distribution of metals (Cd, Cu, Zn, Hg) in Arctic marine mammals.

3. To determine the influence of animal age on metallothionein, and the relationship between metal concentration in tissues and metallothionein.

DESCRIPTION

The most frequent type of measurement made for toxic metals (Hg, Cd, Pb) in tissues is the total concentration of the metal. However, metals exist in tissues in different forms (chemical species) which have different toxicities. Mercury, for example, can be present as inorganic mercury and as methylmercury, the latter being much more toxic than inorganic mercury. The total metal concentration alone is therefore not a good measure of the potential toxicity of the metal. Measurements of cadmium in tissues of wild animals have, to date, been made largely only as total cadmium. It is known that cadmium is present in tissues in different forms by association with different proteins in the cytosol. The different protein-cadmium compounds have different toxicities. The cadmium associated with a low-molecular weight protein called "thionein", or "metallothionein" when saturated with metals, is much less toxic to organisms than the cadmium associated with higher molecular weight proteins. Induction of metallothionein is a protective cellular response to some metals, specifically to cadmium; when this protective capacity is exceeded toxicity occurs. As in the case of mercury, the total cadmium concentration alone does not indicate well its toxicity to organisms. The measurement of metallothionein in conjunction with total cadmium is a much better toxicity indicator than total cadmium, and the additional measurement of the fraction of cadmium associated with high-molecular weight proteins is a further improvement in the knowledge of the in-tissue toxicity of cadmium. Metallothionein (MT) is a low-molecular-weight, ubiquitous, protein that occurs naturally at back-ground levels in most organisms. Synthesis of metallothionein in tissues of organisms to levels higher-than-background is induced when organisms are exposed to stressors in their environment such as elevated levels of heavy metals (Cd, Zn, Cu, Hg) and other substances, and to a lesser extent diseases, nutritional status and stress as mediated by

glucocorticoid hormones. Synthesis of metallothionein takes place primarily in the liver and kidney in proportion to the stressor. The role of metallothionein in the protection of organisms from metal toxicity, its potential for environmental monitoring, as well as the induction of its synthesis, primarily in response to metal stressors has been studied extensively. Metallothionein is potentially a useful biomarker because its synthesis in organisms provides an integrated response to stressors, however, its normal concentration range in tissues of wild marine mammals is largely unknown and needs to be established. This project is concerned with providing information on background concentration ranges of metallothionein in marine mammal tissues, determining the influence of variables such as age and metal concentration in tissues on metallothionein levels, and determining the fraction of cadmium associated with high-molecular weight proteins in liver and kidney cytosol. In addition to providing an integrative response to stressors, metallothionein responds specifically to cadmium, is fairly quick and easy to measure, and is a much better indicator of toxicity in conjunction with total cadmium than total cadmium alone. Narwhal are known to have some of the highest cadmium concentrations in liver and kidney of marine mammals. We intend to measure metallothionein in tissues of narwhal and other marine mammals to determine background concentration ranges in the different species, and to establish whether or not the relatively high cadmium burden carried by some animals is largely in the form of metallothionein and therefore non-toxic, and to determine the fraction of cadmium bound to higher molecular weight proteins and therefore posing a potential danger to the animals.

ACTIVITIES IN 1992/93

Samples

Liver samples from 30 harp seals (15 mother-pup pairs) were used in this project. It had been established in the field which pup belonged to which mother. This unique set of samples was obtained through L. Lockhart, Freshwater Institute. The animals were harvested in March 1984 near the Magdalene Islands in the Gulf of St. Lawrence in conjunction with another project. Ages of pups and mothers were known from previous work on these animals (Wagemann *et al.* 1988). Tissue samples of narwhal from the vicinity of Pond Inlet, and ringed seals and belugas from across the Arctic are available for future analyses.

Methods

Harp seal liver tissue was homogenized in 0.9% saline solution (containing 5 mM dithiothreitol), centrifuged (100 000 g, 1 h), and the pellet and supernatant (cytosol) and original homogenate were analyzed for total cadmium, mercury, zinc and copper. The cytosol was further fractionated by gel permeation chromatography to separate high and low molecular weight proteins and these fractions were also analyzed for metals. A sub-sample of the cytosol was heated for 5 min. at 95° C, centrifuged as before and analyzed for metallothionein by differential pulsed polarography. Total mercury was determined by cold-vapour flameless/AA, and other metals by direct-current plasma emission spectrometry, flame AA, and graphite furnace AA, depending on concentration.

Quality Assurance

Purified rabbit liver metallothionein (Sigma Chemical Co.) was used as a reference material for metallothionein determinations. Metallothionein was also measured by the ^{203}Hg saturation method using the same reference material. Although some differences between the two methods were encountered, depending on the pre-analysis processes employed, the overall conclusions derived from results by the two methods were in agreement. The two methods had been subjected in the past to rigorous comparative assessment (Wagemann *et al.* 1993). Certified reference materials (Bovine Liver, NBS; DOLT-1, NRC; DORM-1, NRC) were used to check the accuracy of total metal determinations. High-purity reagents were used. Water for working standards and reagents was triply distilled in a quartz still, and reagent-grade acids were re-distilled in a Teflon still.

RESULTS

In a previous report (Wagemann 1992) results for only a fraction of the number of animals available were reported. In this report results of analysis of liver samples from all available harp seal mothers and their pups (30 animals) are reported for metallothionein, and for metals in the different tissue compartments. The average metallothionein concentration in the liver of mothers was 300 μg/g. Most individuals had concentrations within 1 standard deviation ($\pm$ 100 μg/g) of the mean (Fig. 1), with the exception of two outstanding high values 500-600 μg/g. The average metallothionein concentration in the liver of harp seal pups was 500 μg/g. As for mothers, most individual values for pups were within one standard deviation ($\pm$ 100 μg/g) of the mean (Fig. 2), with one exception (1100 μg/g). Examination of the metal concentration in tissues of the individuals showed that the two outstanding results (Fig. 1), were for animals with the highest concentration of cadmium and zinc respectively, and the outlier in Figure 2, belonged to an animal with the highest copper concentration. The cause of these high metallothionein values was very likely a consequence of the high metal concentrations in these individuals. The means for pups and mothers were significantly different (α=0.05). Metallothionein concentrations in mothers and in pups were positively correlated (P=0.096).

The metallothionein concentration in the liver was not significantly correlated with the age of animals, neither in adults nor pups. Similarly, none of the metals in the liver were individually correlated with metallothionein in pups or mothers. (Cadmium was not detectable in the liver of pups). In a previous report (Wagemann 1992) a possible non-linear correlation between age and metallothionein in mothers was depicted but this was based on too few results.

In adult harp seals the cadmium in the liver was found largely in the cytosol (approximately 80%), and < 20% in the solid cellular material i.e. the "pellet". The distribution of copper (Fig. 3) was similar to that of cadmium, but mercury was distributed differently; only approximately 20% of the total mercury was in the cytosol and 80% was in the solid cellular material (pellet). Some of the mercury associated with solid cellular material was undoubtedly present as inorganic mercuric selenide, known to be present in the liver as amorphous vesicular particles (Pelletier 1985, Wagemann and Stewart 1993) thereby reducing mercury toxicity; however, mercuric selenide as such was not measured during this investigation. In pups, zinc

was distributed approximately equally between cytosol and solid cellular material, copper was distributed similar to that in adults, but mercury was distributed differently from that in adults (Fig. 4). Most mercury in pup livers was found in the cytosol (approximately 80%), and only ~20% was associated with solid cellular material. This type of distribution may be related to the low selenium concentration in the liver of pups (Wagemann *et al.* 1988). Consequently, pups would have a much lower mercuric selenide concentration in the liver than adults and a proportionately lower degree of protection by selenium against mercury toxicity than adults.

Gel permeation chromatographs for copper (Fig. 5), cadmium (Fig. 6) and zinc (Fig. 7), show the association of metals with high-molecular weight proteins (first two peaks for copper; first small peak for cadmium; first three peaks for zinc) and with metallothionein (large doublet peaks for cadmium and copper, small doublet peaks for zinc). Two iso-metallothioneins (two forms of metallothionein commonly present in mammalian tissues) appear nearly resolved (doublet peaks for metallothionein), particularly in Figure 6. Cadmium was mainly associated with metallothioneins in liver cytosol of adult harp seals, >90% (Fig. 8), and very little with high-molecular weight proteins (~5%). This indicates that cadmium in liver is almost entirely sequestered by metallothionein and has a greatly reduced potential for toxicity. No cadmium (below detection) was found in the liver of pup harp seals.

CONCLUSIONS AND UTILIZATION OF RESULTS

The metallothionein background concentration in the liver of adult, lactating harp seals was in the range of 200-400 μg/g, and in pups (300-500 μg/g). Cadmium was not detected in the liver of pups, and the metallothionein in pups contained only copper and zinc. Approximately 80% of the total cadmium in the liver of adult harp seals was in the cytosol, of which more than 90% was associated with metallothionein, and only 5% with high molecular weight proteins, indicating that most of the cadmium in the liver was in a relatively nontoxic form. Unlike copper and cadmium, mercury in adults was largely associated with solid cellular material, probably mostly in the form of mercuric selenide deposited in vesicles, affording protection against mercury toxicity. In pups, on the other hand, most of the mercury (~80%) was in the cytosol where it could be relatively more toxic to these animals than in adults. Correlations between metallothionein and age and between metallothionein and metal concentrations in tissues were not significant, however, three animals with outstandingly high metallothionein concentrations had outstandingly high metal levels. These results have implications for the assessment of the toxicity of cadmium and mercury to animals. They provide a better assessment of toxicity than do total metal concentrations alone.

Work will be undertaken to determine background metallothionein concentrations and sub-cellular metal distributions in tissues of narwhal, beluga and ringed seals. The results will have implications for the correct assessment of the toxicity of metals. Contractual arrangements have been made to obtain samples of narwhal and other marine mammals in 1993.

Expected project completion date: March 31, 1996

Partners: A private laboratory (Methodologies Consultations Limited, Gloucester, Ontario) was involved in the methods assessment work.

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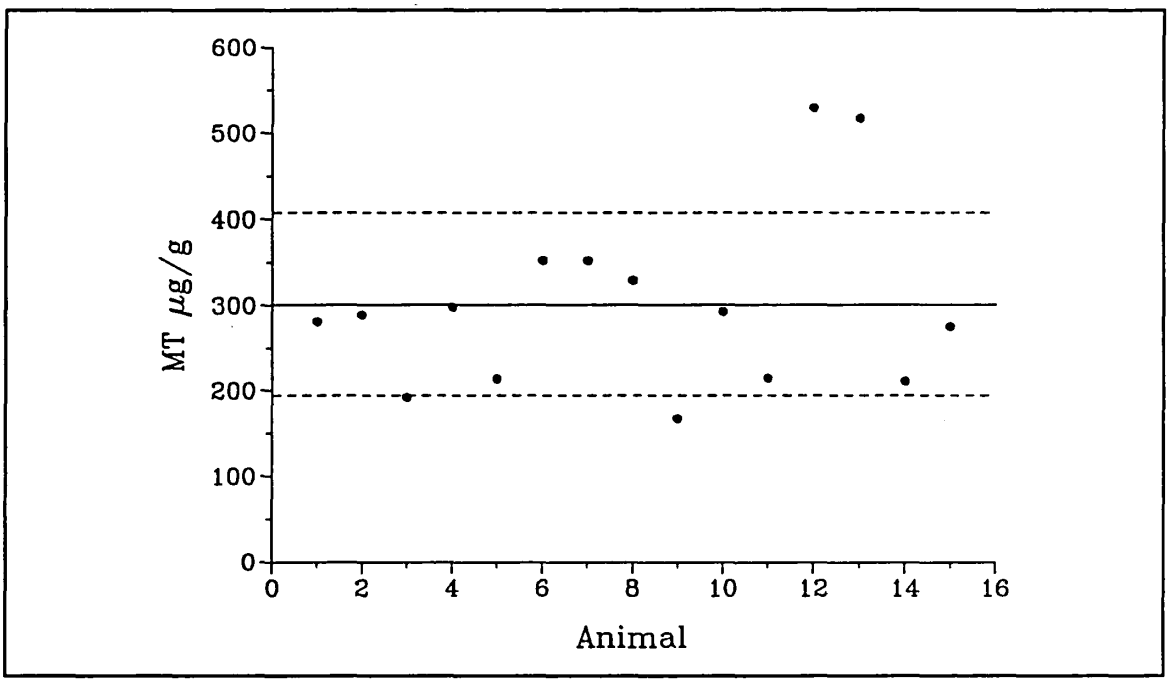


Figure 1. Average ± 1 standard deviation, and concentrations in individual animals (μg/g) of metallothionein (MT) in the liver of harpseal mothers.

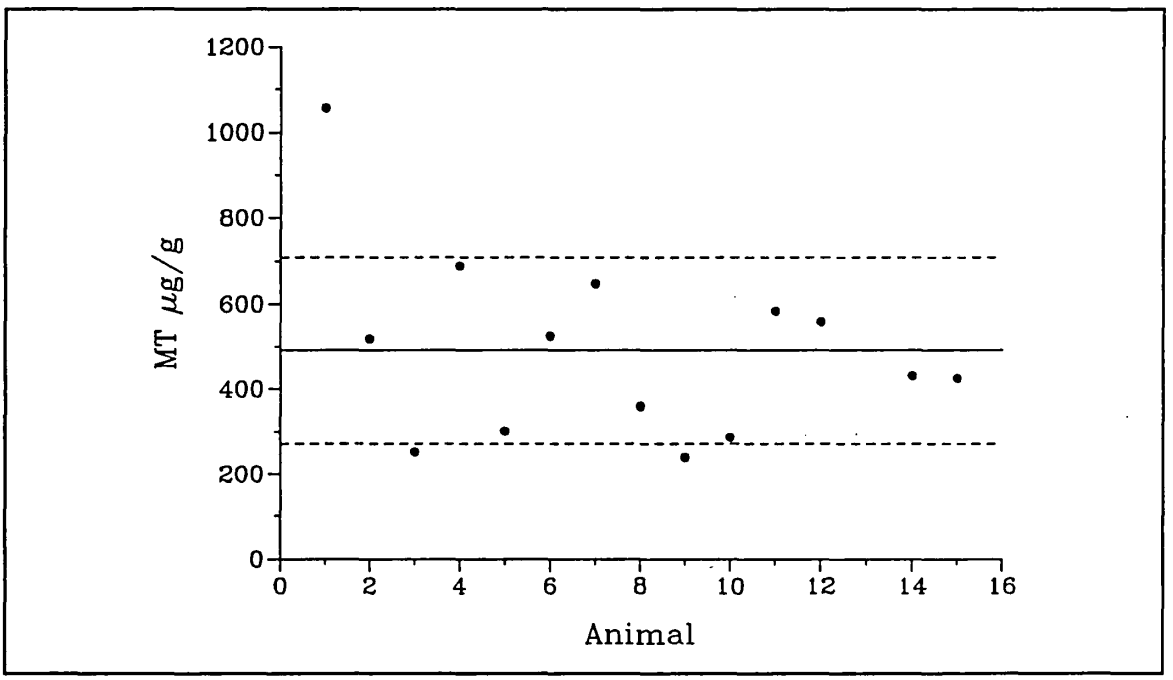


Figure 2. Average ± 1 standard deviation, and concentrations in individual animals (μg/g) of metallothionein (MT) in the liver of harp seal pups.

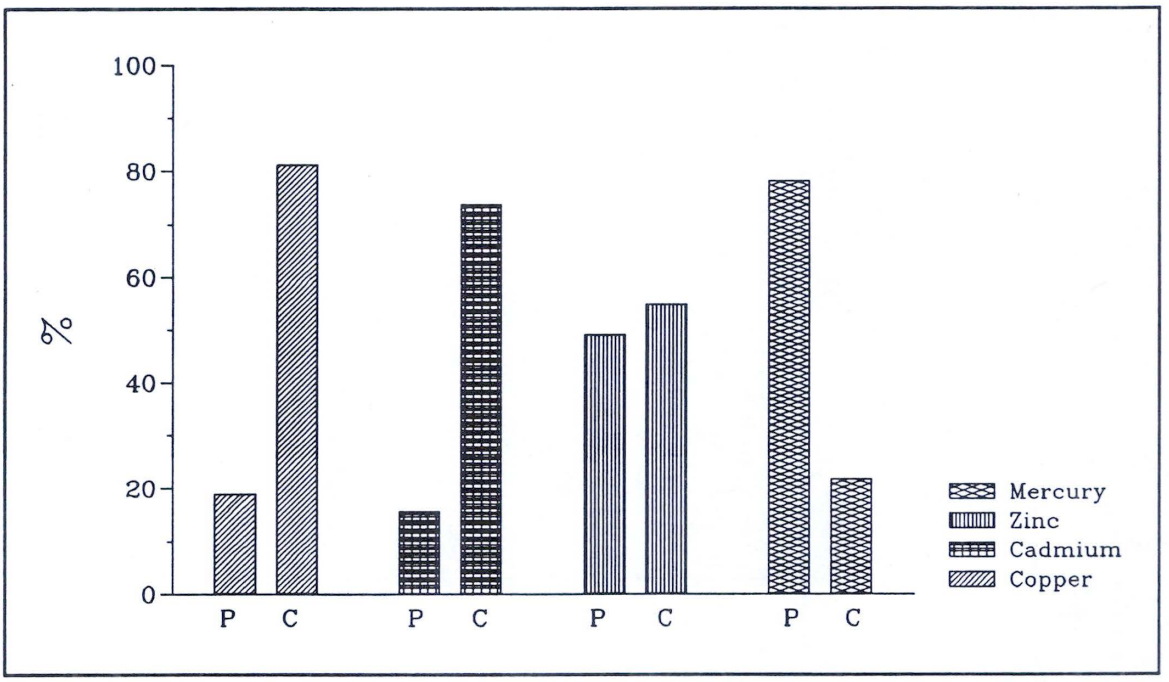


Figure 3. Distribution (percent) of total metals in the liver between solid cellular material (P) and cytosol (C) of harp seal mothers.

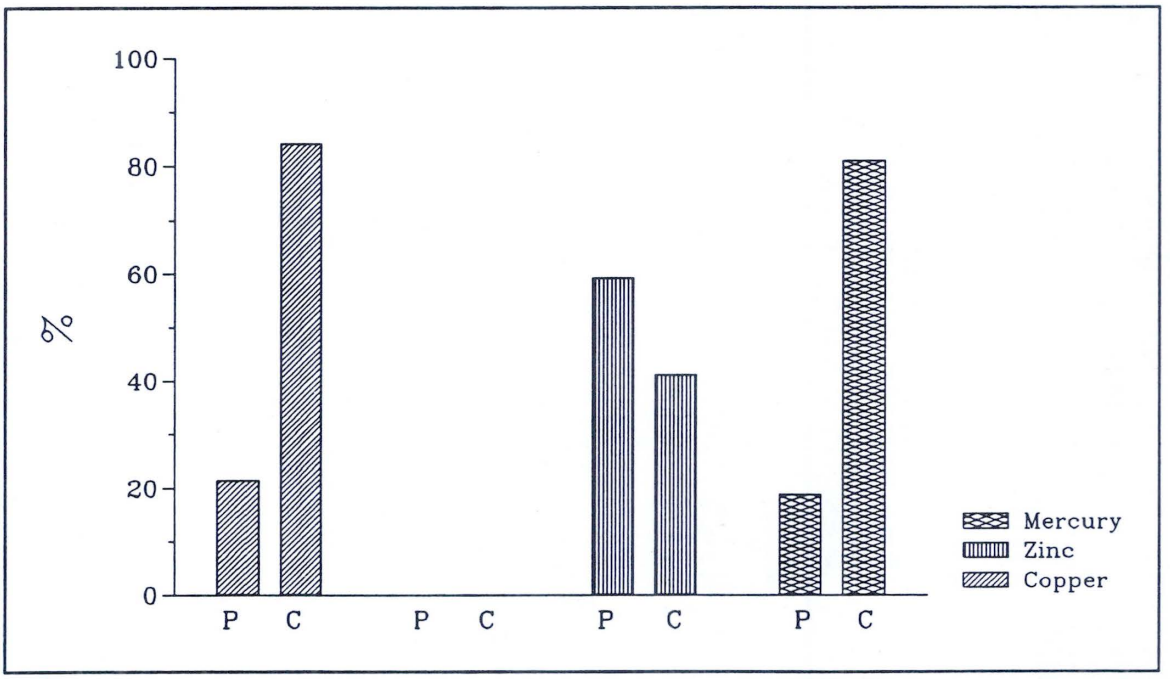


Figure 4. Distribution (percent) of total metals in the liver between solid cellular material and cytosol of harp seal pups.

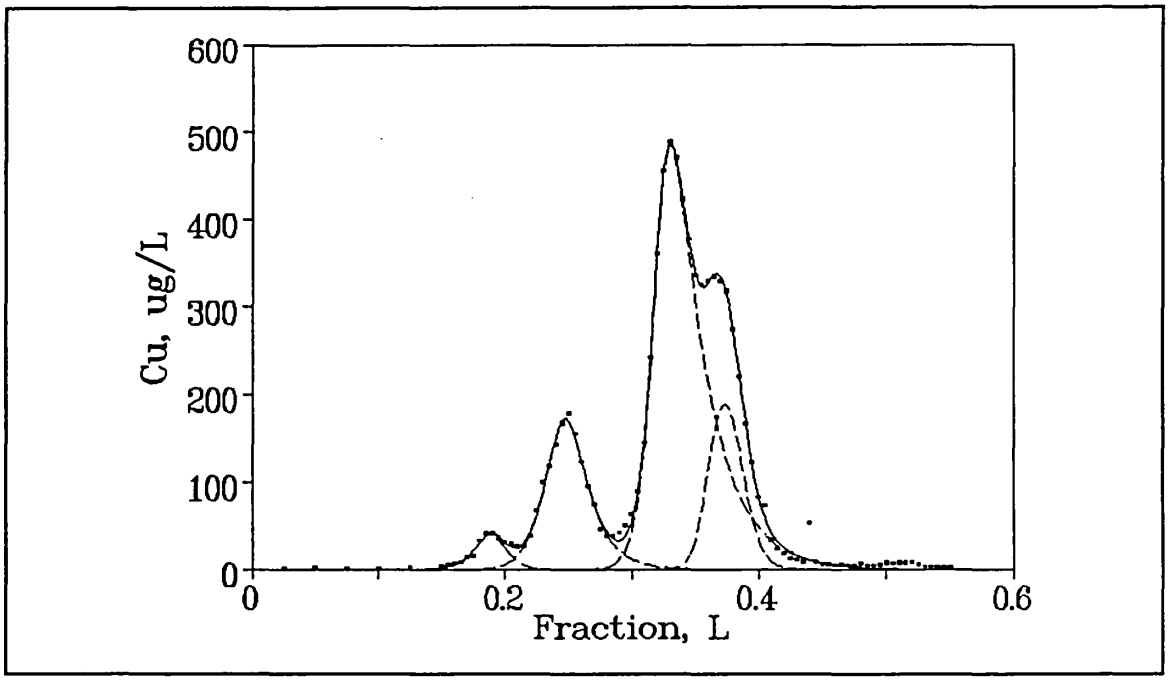


Figure 5. Gel permeation chromatographic copper profile of mother harp seal liver cytosol. Large peak and shoulder are metallothionein, the first two are high-molec. wt proteins.

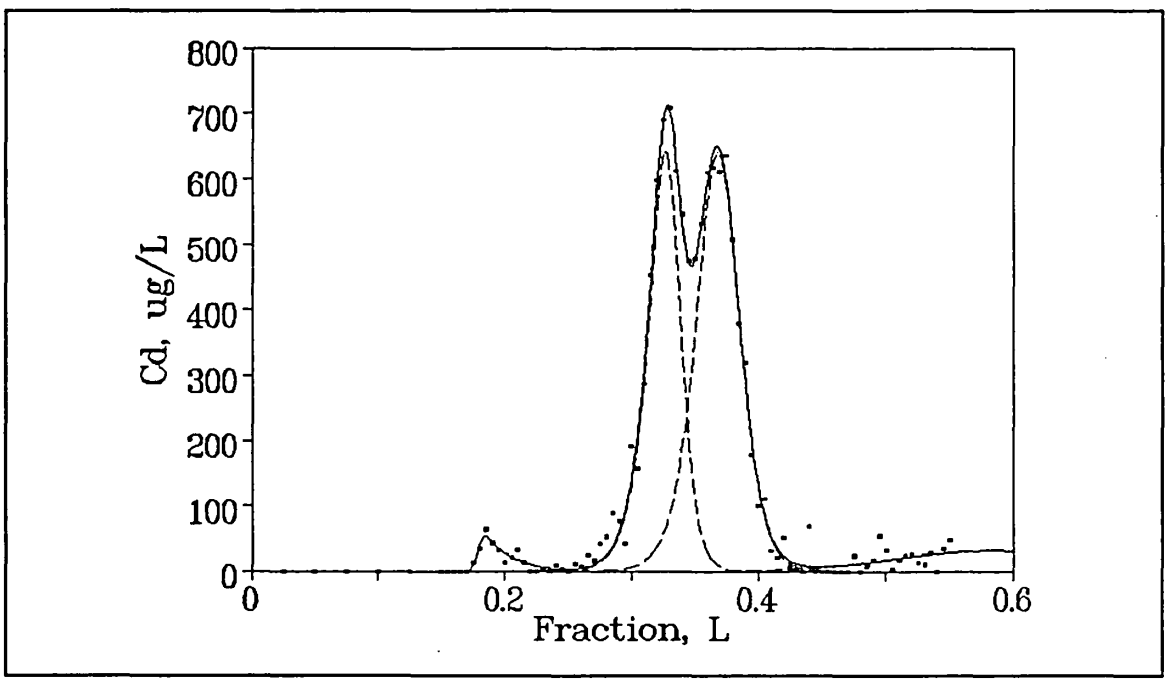


Figure 6. Gel permeation chromatographic cadmium profile of mother harp seal liver cytosol. Large, double-peaks are metallothionein.

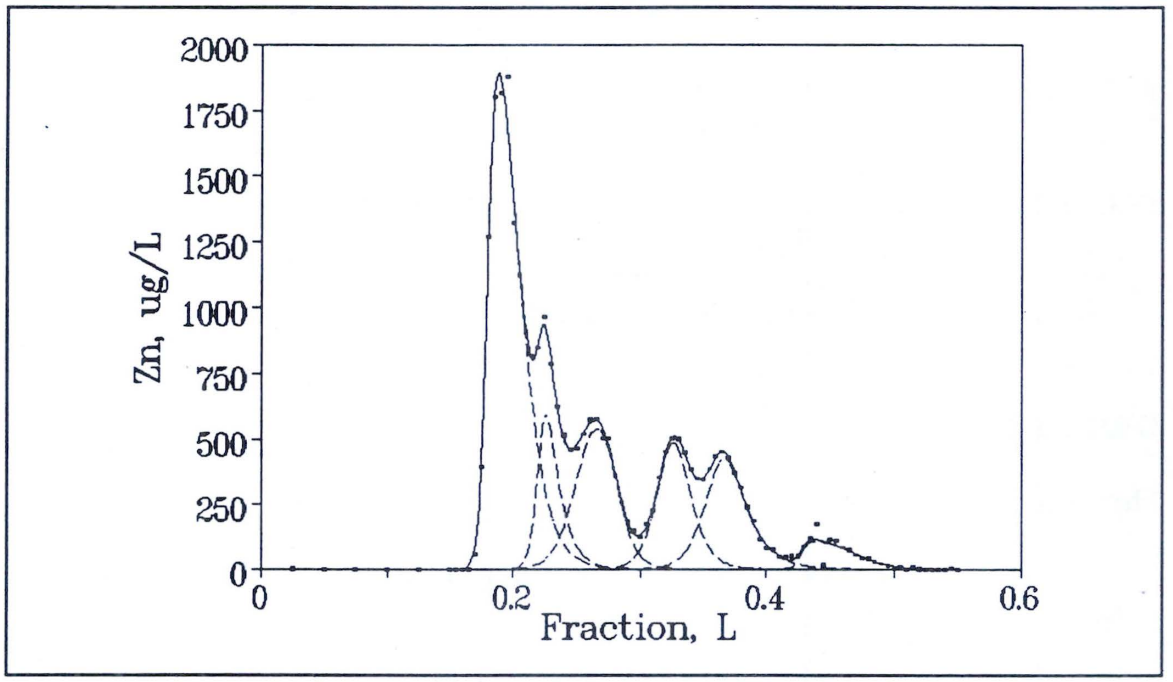


Figure 7. Gel permeation chromatographic zinc profile of mother harp seal liver cytosol. The 1st, 2nd and 3rd peaks (unresolved) are high-mol. wt proteins, the 4th and 5th peaks are metallothionein.

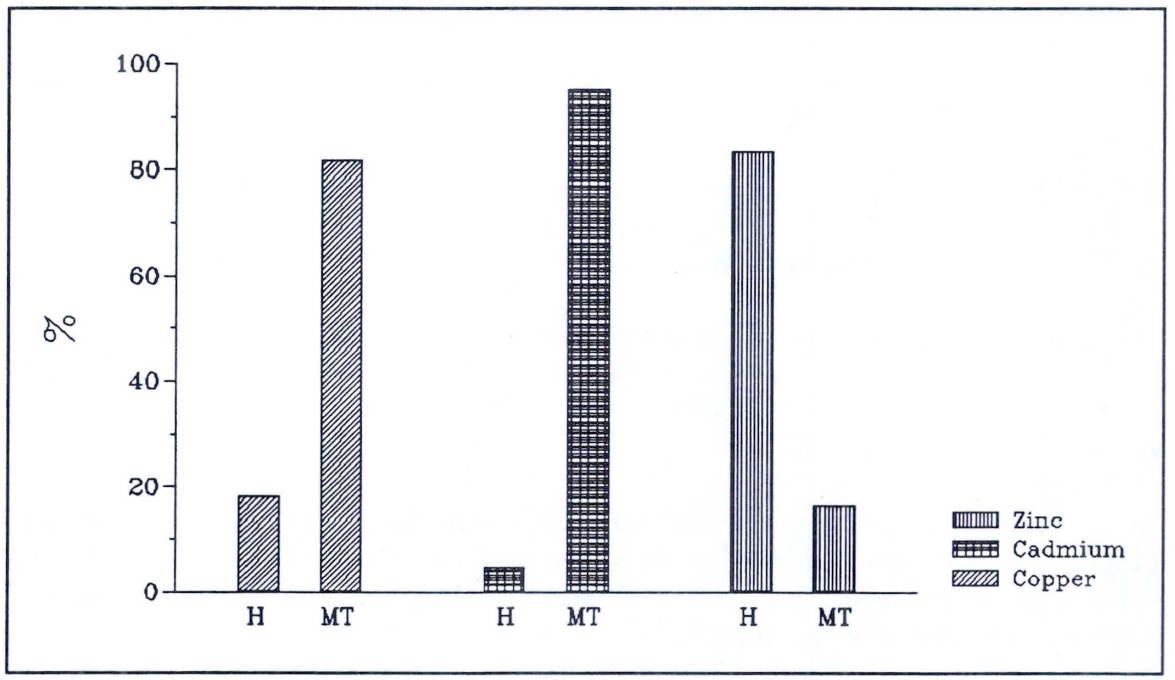


Figure 8. Distribution of metals in liver cytosol between high-molecular weight proteins (H) and metallothionein (MT) in harp seal mothers.

SOURCES, PATHWAYS AND LEVELS OF CONTAMINANTS IN FISH FROM YUKON WATERS

PROJECT LEADER: Yukon Technical Committee on Contaminants in Northern Ecosystems and Native Diets (Contact: Mark Palmer, Chair)

PROJECT TEAM:
1) Government of Yukon Fisheries
2) Fisheries and Oceans Canada
3) Indian and Northern Affairs Canada
4) Environmental Protection, Environment Canada

OBJECTIVES

Short-term:

1. To collect information to complement that gathered in the 1991 season by sampling additional lakes.

2. To collect information to confirm that gathered in the 1991 season to address concerns raised by health advisories based on existing data.

3. To determine spatial variability in contaminant loadings, and to assess short term trends.

Long-term:

1. To investigate the sources, processes and rates of contaminant deposition and transport into and within the waters of the Yukon.

2. To determine levels of contaminants for use in long-term trend analysis.

3. To develop additional monitoring on levels of organic contaminants within the Yukon.

4. To provide additional information for use in updating health advisories.

DESCRIPTION/RESULTS

Recent burbot liver and lake trout flesh samples from headwater lakes in the Yukon River system (Tagish, Laberge, etc.) indicated elevated levels of organochlorine pesticides. In response to elevated toxaphene levels, Health and Welfare Canada issued a public health advisory on Lake Laberge. The advisory recommended that consumption of lake trout flesh be limited, and that burbot livers not be consumed. This has affected the various fisheries on the lake, and generated considerable concern from residents who used the fisheries resources of the lake.

There is as yet insufficient information to determine the source(s) of the contamination. Additional information is essential to understand the current situation, and to use that understanding to determine the source(s) of the contamination. Contaminant levels vary considerably from lake to lake within the Yukon and as a result residents are requesting that all lakes used for subsistence basis be tested.

Eleven lakes were sampled in 1992/93 between June and September. The names and locations of the lakes sampled are given in Figure 1. The target species and tissues were lake trout flesh and stomachs, whitefish flesh and stomachs and burbot livers. The sample locations and species were selected on two basic criteria: i) their use as a subsistence fishery, and ii) to provide additional information to determine the source(s) of the contaminants.

The results from this year's program are being verified and will be published in a data report in the fall of 1993.

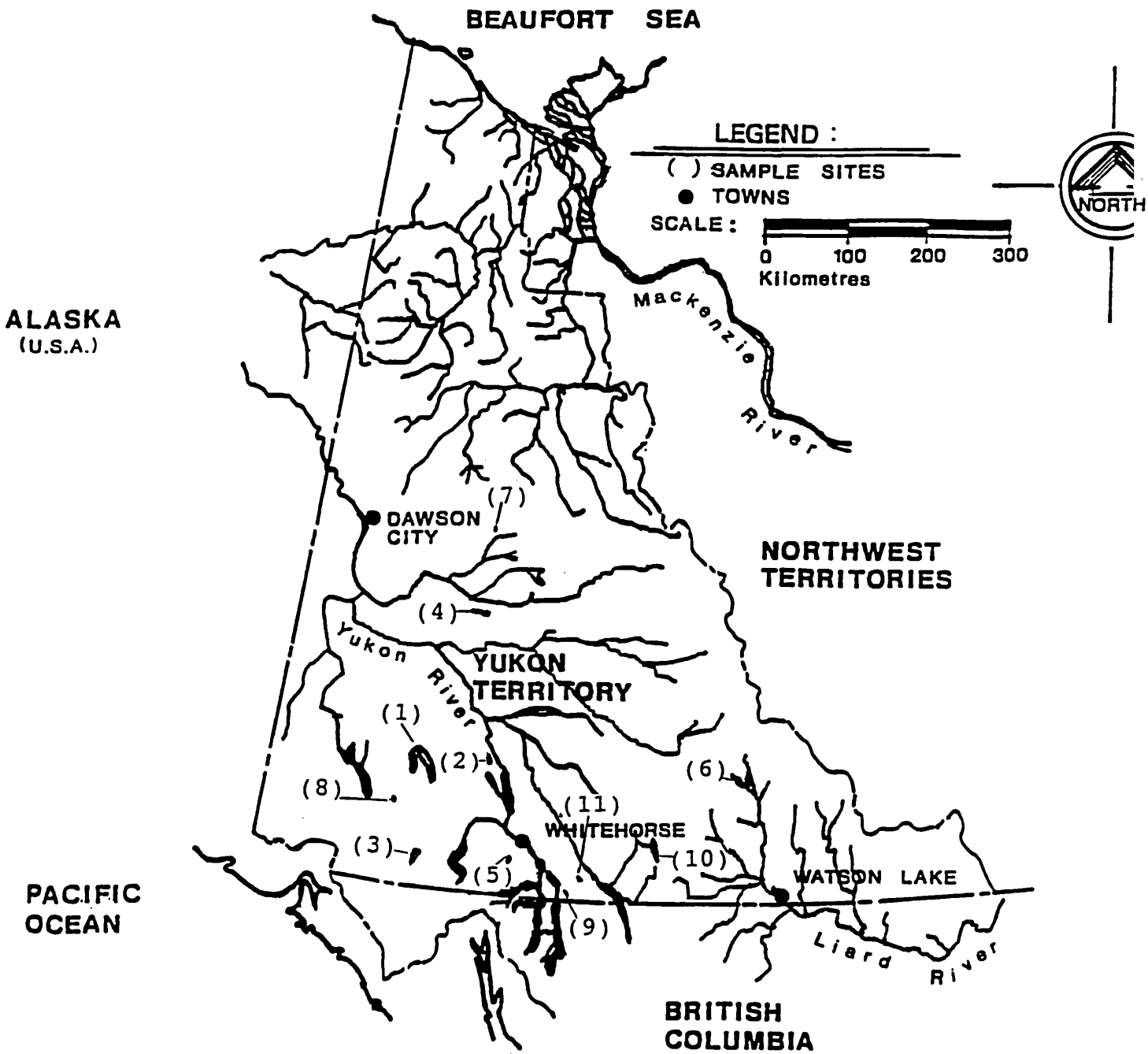
UTILIZATION OF RESULTS/DISCUSSION/CONCLUSION

As outlined, one of the main goals of this program is to determine the levels of contaminants in fish from lakes used to support subsistence lifestyles. As soon as the information has been verified it is submitted to Health and Welfare Canada for a health assessment to determine if the fish is safe to consume.

This information along with other information gathered under several other projects will be used to try and determine the source(s) of contaminants in Yukon fish. A workshop to address this topic is scheduled to take place in Whitehorse, Yukon later this year.

Expected project completion date: Due to the unexplainable variation of contaminant levels between lakes, it is anticipated that the project will continue as resources permit.

Figure 1. Sampling site locations, Yukon Territory.



LOCATIONS

(1) Aishihik Lake

(2) Braeburn Lake

(3) Dezadeash Lake

(4) Ethel Lake

(5) Fish Lake

(6) Frances Lake

(7) Hansen Lake

(8) Kloo Lake

(9) Little Atlin

(10) Quiet Lake

(11) Squanga Lake

CONTAMINANT TRENDS IN FRESHWATER BIOTA

PROJECT LEADERS: D. Muir and W.L. Lockhart, Fisheries and Oceans Canada, Central and Arctic Region, Winnipeg, Manitoba

PROJECT TEAM: D. Muir, B. Grift, D. Metner, B. Billeck, L.. Lockhart, B. Rosenberg, S. Mohammed, R. Wagemann and R. Hunt

OBJECTIVES

1. To determine temporal and spatial trends in PCBs, other organochlorines and PAHs in fish from lakes and rivers in NWT and N. Quebec.

2. To provide information on contaminants to evaluate current risks of exposure to PCBs, PAHs and chlorophenols via fish consumption.

DESCRIPTION

The initial objectives of the project (1989-91) were to determine organic contaminant levels in fishes from remote lakes in northern Québec and N.W.T., first monitored in 1970 by Reinke *et al.* (1972) and Riseborough and Berger (1971), in order to examine the general temporal trends in DDT, dieldrin and PCBs as well as investigate the presence of additional contaminants not reported in the early work. Reviews of contaminant data in freshwater fish from Arctic and subarctic Canada (Lockhart *et al.* 1992; Muir *et al.* 1990) have indicated that information on the levels and geographic variation of PCBs and related organochlorines (OCs) and polyaromatic hydrocarbons (PAHs) and heavy metals (except perhaps for Hg) is limited while data on temporal trends is nonexistent. We have placed most of the emphasis on organochlorine chemicals such as PCBs and toxaphene, rather than PAHs, because the OCs are known to biomagnify in aquatic food chains. Other agencies are measuring heavy metals (or at least Hg) in fish, for e.g. Fisheries and Oceans Canada (DFO) Fish Inspection, so not all fish that are analysed in this study for OCs are also analysed for metals. Although the main focus of the study is on piscivorous fishes, (such as burbot, lake trout and arctic char), because of interest in biomagnification of PCBs, DDT-compounds and other OC pesticides, there is also a need to examine contamination of fishes feeding at lower trophic levels, such as whitefish, which are also of dietary importance to native people in the NWT, the Yukon and N. Québec. The long-term objective is to provide a baseline for contaminant levels at a level of detail sufficient to give northern residents useful information on contaminant levels in major species and in a wide range of lakes.

ACTIVITIES IN 1992/93

Samples

Fish samples were collected by DFO Winnipeg personnel or by DFO biologists in the NWT (B. Ferguson, Inuvik) and S. Harbicht and A. Wilson (Yellowknife) from a wide range of locations in the NWT. Species were the same as those from previous years, broad whitefish (*Coregonus nasus*), land-locked char (*Salvelinus alpinus*), burbot (*Lota lota*) and lake trout (*Salvelinus namaycush*). Also collected (but not analysed during 1992/93) were Arctic cod (*Boreogadus saida*) at Resolute (by B. Welch, DFO) and marine clams from southeast Hudson Bay (by L. Lockhart). Sampling locations are given in Table 1. Samples of dorsal muscle and skin were analysed, with the exception of burbot for which liver was analysed. Sex, weight and age (by using otoliths) were determined for all samples. Generally equal numbers of male and female samples were analysed and samples were selected to give 3 to 6 specimens from similar age classes for each species.

Analysis

Samples of whole fish or muscle were analysed for organochlorines [PCB congeners and other organochlorine contaminants (toxaphene, PCCs), chlordane (CHLOR), and the DDT group]. A total of 130 individual OC compounds were determined. A complete list of OC analytes is given in our report on OCs in marine mammals (this volume). Methods of extraction and GC analysis were identical to those described by Muir *et al.* (1990). In brief: muscle/skin samples were homogenized by grinding with dry ice. The homogenate (20 g) was Soxhlet extracted with hexane: dichloromethane (1:1). Internal standards of aldrin and octachloronaphthalene (OCN) were added at the extraction step. Lipid was removed by automated gel permeation chromatography. Extracts were then chromatographed
on a Florisil column to separate PCBs, p,p'-DDE and trans-nonachlor (hexane eluate) from most chlorinated bornanes (toxaphene), chlordanes and DDT-related compounds. Florisil eluates were then analysed by capillary gas chromatography with electron capture detection using a 60 m x 0.25 mm id DB-5 column with H_2 carrier gas. Confirmation of PCBs was carried out by GC-mass spectrometry using a HP5971MSD while chlorinated bornanes were confirmed by electron-capture negative ion mass spectrometry.

Heavy metals (Hg, Cd, Pb, As) were determined in fish muscle by atomic absorption (AA) spectrophotometry (Hendzel and Jamieson 1976, Vijan and Wood 1974). Hg was determined by hot block digestion followed by cold vapour AA. Cu, Zn were determined by air-acetylene flame atomic absorption (Varian SpectrAA-20) with deuterium background correction, after nitric acid digestion. Cd and Pb were determined by Zeeman background corrected graphite furnace AA spectrophotometer (Hitachi model Z8200) after sulfuric acid digestion.

Quality Assurance

Recovery of internal standards was checked in each sample and samples with low recoveries (generally $<60\%$) were reextracted. Blank samples were run approximately every 10 samples to check contamination of reagents and glassware. During 1992, the laboratory participated in

the International Council for the Exploration of the Seas (ICES) interlab comparison program, in a CAPCO (National Water Research Institute) fish check samples program for analysis of PCB congeners, and in a National Institute of Standards and Technology (NIST) program to certify additional PCB congeners in the cod liver oil SRM 1588. An interlab comparison of the SRM 1588 was also made with AXYS Analytical Services Ltd. (Sidney, BC).

RESULTS

Fishes from 10 lakes in NWT were analysed during 1992/93 for a suite of about 130 individual organochlorines (OCs) (PCB congeners, 4 chlorobenzenes, organochlorine pesticides). In addition heavy metals were determined in Arctic char (muscle and liver) from Hazen and Amituk Lakes. As has been observed previously, toxaphene was the major OC contaminant in all fish with the exception of char from Char Lake at Resolute where PCBs and DDT predominated (Table 1). Char lake is located near the airport at Resolute and could have been contaminated by past disposal of PCBs and use of DDT. This is the first lake in which we have observed such a pattern of PCB/DDT/Toxaphene.

We added significantly to the database on contaminants in whitefish this year with analysis of samples from 4 lakes in the Mackenzie delta, from Great Slave Lake, and from Gordon Lake (northeast of Yellowknife). As we have observed in other lakes (Muir and Lockhart 1992) whitefish generally have low levels of OCs. Highest concentrations of OCs were found in whitefish from Great Slave Lake. These fish also had high lipid content compared to those from the other locations. On a lipid weight basis, OC concentrations in whitefish from Gordon Lake were similar to levels in the same species from Great Slave Lake, while broad whitefish from the Mackenzie delta region had lower levels. Lipid weight comparisons remove the effect of lipid content which usually co-varies with OC concentrations in fish.

Lake trout had higher levels of OCs than whitefish in Gordon Lake by a factor of about 2, as expected for piscivores. Comparing all *Salvelinus* in 5 lakes, the highest levels of OCs were found in samples from Amituk Lake (Table 1). The elevated levels of OCs at Amituk confirms our previous results (Muir and Lockhart 1992) from 9 samples collected in 1990 from Amituk. Char from Amituk have particularly elevated levels of toxaphene and PCBs, levels which are comparable to those observed in Lake trout in Lake Laberge, when compared on a lipid basis (Table 2). The reason for the elevated OCs at Amituk Lake is not known but given that all major groups are high it must be due to food chain characteristics of the lake.

Burbot liver from Great Slave Lake contained relatively high levels of OCs in comparison with whitefish, with toxaphene predominating. Burbot liver from Trout Lake had about 3-fold lower concentrations of PCBs and toxaphene than those from Great Slave Lake although average lipid content of the liver was higher in the former group. The reasons for the higher levels in Great Slave burbot may be due to the greater importance of fish in the burbot diet. Levels of toxaphene and PCBs in burbot captured in the lake (near Fort Resolution) are similar to those determined in burbot from the Slave River (Whittle 1992) which suggests that they may be part of the same population. ΣDDT and PCB concentrations in Great Slave Lake burbot are about 10-times lower than in Lake Laberge burbot liver when compared on a lipid weight basis, while

toxaphene levels are about 5-fold lower (Table 2). Concentrations for the major OCs in burbot liver are similar to those we observed in our previous studies of Mackenzie River burbot (Muir *et al.* 1990).

Relatively high Hg levels (i.e. exceeding 0.5 μg/g fresh wt) were found in liver and muscle of landlocked char from Amituk Lake. These results from samples collected in 1992 confirm previous findings from samples collected in 1989. The reason for the elevated levels of Hg are not known but they parallel high levels of PCBs and toxaphene. In contrast, Hg in muscle of Hazen Lake char averaged 0.18 μg/g (fresh wt).

CONCLUSIONS AND UTILIZATION OF RESULTS

The results further confirm the remarkable lake to lake variation of organochlorines and metals in Arctic lakes. Food chain length and trophic status (i.e. eutrophic versus oligotrophic) of lakes are known to influence levels of organochlorines in fish (Rasmussen *et al.* 1990). Further analyses are required to sort out the influence of these variables on organochlorine levels in remote Arctic lakes. Feeding preferences are the most likely explanation for the higher levels of OCs in Great Slave Lake fish than in those in nearby lakes. Although OC levels are relatively low compared to Lake Laberge, further study of Great Slave Lake seems advisable to characterize variations within and among species because of the importance of commercial and subsistence fisheries in this lake. This will be accomplished in part by a new project by M. Evans (National Hydrology Research Institute, Saskatoon) and by analysis of additional samples collected by DFO and the Metis Nation. Further sample collections are planned for other major lakes in NWT in cooperation with DFO Fish Inspection, including Baker Lake. Additional work will be done in the Mackenzie Delta area in cooperation with Fisheries Joint Management Committee. Char and lake trout from lakes in the delta and from Banks Island, obtained from J. Reist (DFO Winnipeg) will also be analysed. Arctic cod from Resolute (collected in 1992), clams from Hudson Bay, and whitefish from Noril'sk region of Russia will also be analysed. To date sampling has concentrated on Arctic char, whitefish species and burbot. As the emphasis of the project switches to lakes of central NWT we are planning to analyse more lake trout, burbot and lake whitefish.

Expected project completion date: March 31, 1997

Partners: DFO Fish Inspection (M. Hendzel), J. Reist, K. Chang-Kue, B. Bond, (DFO Winnipeg), S. Harbicht, N. Wilson (DFO Yellowknife), B. Ferguson (DFO Inuvik), Fisheries Joint Management (Inuvik), B. Carpenter (Metis Nation, Yellowknife).

Table 1. Mean concentrations (ng/g ± SD wet wt) of major organochlorines in fish from NWT lakes.

Lake/species/tissue	Region	Sex	N	Lipid %	ΣHCH	ΣCHLOR	ΣDDT	ΣPCB	Toxaphene
A. Broad whitefish muscle									
L100	Mackenzie delta	M&F	4	10.8 ± 2.35	2.6 ± 2.3	2.9 ± 2.5	2.5 ± 2.2	5.4 ± 5.1	16.4 ± 14.5
Travaillant Lake	Mackenzie delta	M&F	4	2.6 ± 0.5	0.46 ± 0.11	0.67 ± 0.11	0.31 ± 0.05	1.9 ± 0.69	3.4 ± 0.66
Kugaluk Lake	Mackenzie delta	M&F	9	3.8 ± 1.6	1.0 ± 0.63	1.5 ± 0.71	0.61 ± 0.34	6.1 ± 4.1	12.5 ± 11.9
Campbell Lake	Mackenzie delta	M&F	3	1.9 ± 1.7	2.4 ± 3.1	1.8 ± 2.2	1.2 ± 1.5	2.6 ± 1.2	5.2 ± 0.95
B. Lake whitefish muscle									
Great Slave Lake	Ft. Resolution	M&F	4	18.8 ± 4.0	6.1 ± 2.3	25.9 ± 4.6	8.3 ± 2.0	22.2 ± 3.5	154 ± 18.6
Gordon Lake	Yellowknife	M&F	5	2.7 ± 2.5	0.92 ± 0.91	5.5 ± 3.7	3.7 ± 2.8	7.9 ± 4.4	23.8 ± 17.2
C. Burbot liver									
Great Slave Lake	Ft. Resolution	M&F	6	35.0 ± 8.0	10.2 ± 1.9	114 ± 44.0	57.0 ± 22.0	178 ± 72.9	758 ± 282
Trout lake	Ft. Simpson	M&F	6	40.3 ± 12.1	13.7 ± 4.7	33.3 ± 12.2	19.8 ± 7.2	50.1 ± 9.7	152 ± 65.9
D. Lake trout muscle									
Trout Lake	Ft. Simpson	M&F	9	4.2 ± 1.9	2.5 ± 2.6	10.5 ± 10.3	5.3 ± 4.7	13.8 ± 9.0	44.0 ± 55.0
Gordon Lake	Yellowknife	M&F	6	3.0 ± 1.3	0.68 ± 0.34	7.9 ± 6.3	5.4 ± 5.0	16.3 ± 12.6	35.7 ± 36.8
E. Arctic char muscle									
Buchanan Lake	Axel Heiberg	M	5	7.5 ± 4.4	3.2 ± 2.1	10.3 ± 3.2	3.7 ± 0.9	7.3 ± 1.8	27.9 ± 9.8
		F	5	7.0 ± 1.4	2.7 ± 1.0	9.3 ± 3.6	3.6 ± 2.2	6.4 ± 3.6	27.0 ± 14.2
Amituk lake	Cornwallis Is	M	6	3.9 ± 1.8	1.7 ± 0.86	43.8 ± 10.3	31.2 ± 14.9	71.0 ± 48.6	252 ± 77.9
		F	6	4.9 ± 2.5	2.4 ± 1.3	50.9 ± 32.8	33.2 ± 16.9	74.1 ± 35.0	314 ± 223
Char Lake	Resolute	M&F	5	4.3 ± 1.9	2.6 ± 1.4	17.8 ± 7.9	114 ± 52.9	289 ± 118	73.2 ± 37.0

Table 2. Comparison of organochlorine levels in Arctic fishes on a lipid weight basis (ng/g lipid).

Location/species	Sex	N	ΣDDT	ΣPCB	Toxaphene
Whitefish muscle					
Lake Laberge (DFO 1992)	F	6	13±6	44±52	119±59
	M	4	13±9	38±6	100±30
Fisherman Lake	M+F	4	31±6	672±122	190±43
Great Slave Lake	M&F	4	44±6	119±16	826±73
Gordon Lake	M&F	5	172±170	406±213	918±395
Burbot liver					
Lake Laberge (DFO 1992)	M	3	16700±4900	7780±1250	11400±2360
	F	3	7290±3370	2670±1300	5940±2110
Great Slave Lake	M&F	6	179±105	568±367	2380±1400
Trout Lake	M&F	6	51±16	136±36	383±129
Lake trout					
Lake Laberge	M	3	13900±18970	14300±21760	8860±10900
	F	3	9070±12070	7550±10170	5930±4470
Gordon Lake	M&F	6	202±144	670±482	1100±833
Amituk Lake	M	6	1870±3150	4880±9080	9210±8130
	F	6	1680±2750	4320±7650	7120±3320

Table 3. Concentrations of heavy metals in Arctic char tissues (μg/g fresh weight).

Lake	Region	Tissue	N	Concentration (μg/g fresh weight)		
				Cd	Cu	Zn
Hazen Lake	N. Ellesmere Is.	Muscle	45	0.0015±0.0020	0.471±0.127	4.44±0.62
Amituk Lake	Cornwallis Is.	Muscle	15	0.0076±0.0102	0.400±0.066	5.91±2.21
		Liver	16	0.349±0.352	7.01±7.90	32.3±7.00
				Pb	Hg	Se
Hazen Lake	N. Ellesmere Is.	Muscle	45	0.002±0.002	0.181±0.093	0.902±0.403
Amituk Lake	Cornwallis Is.	Muscle	27[1]	-	0.567±0.597	0.846±0.189
		Liver	24[2]	-	1.24±1.55	3.89±1.90

[1] N=27 for Hg and 11 for Se.
[2] N=24 for Hg and 14 for Se.

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THE BIOMAGNIFICATION OF ORGANOCHLORINES THROUGH THE FOOD WEB OF LAKE LABERGE AND OTHER YUKON LAKES

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OBJECTIVES

1. To collect benthic and planktonic invertebrates, and piscivorous and forage fishes from Laberge, Fox and Kusawa Lakes, Yukon Territory (YT) and analyse these samples for organochlorines (including PCBs, DDT and toxaphene).

2. To characterize the food webs of these lakes through the use of stable isotopes of carbon and nitrogen, and fish stomach contents, in order to quantify and qualify the biomagnification of organochlorines through the biota.

3. To understand the among lake differences in organochlorine concentrations in fishes from Yukon lakes.

RATIONALE

Lake Laberge, situated 30 km downstream from Whitehorse on the Yukon River system, has been widely used for commercial, sport and native subsistence fisheries for over a century. Analyses of lake trout, burbot and lake whitefish from Laberge in 1991 revealed that the levels of PCBs, DDT and toxaphene (chlorinated bornanes, CHB) are up to 30 times higher than those found in the same species from other regional lakes (Palmer 1992, Muir *et al.* 1992, Kidd *et al.* in press). The presence of high concentrations of contaminants in fishes from this lake has led to the closure of all fisheries on the lake and has caused significant concern among native populations due to the loss of this resource.

It is currently hypothesized that overfishing and increased nutrient input from Whitehorse sewage has changed the food web structure in Lake Laberge, a factor implicated in higher concentrations of contaminants in fishes (Rasmussen *et al.* 1990). Netting surveys indicate that lake trout from Laberge are high in lipid, solely piscivorous and are reduced in numbers unlike the trout from several other large lakes in the area. The community structure of Lake Laberge is also different; the percentage of burbot and suckers is high and the abundance of lake trout and whitefish is low (N. degraff, Yukon Government, Department of Renewable Resources, pers. comm.). This

study will determine the relationship between food web structure and the biomagnification of organochlorines through the biota from Laberge, Fox and Kusawa Lakes. The latter two lakes have been chosen for comparison to Laberge due to their known low levels of persistent contaminants in the fishes.

Many biomagnification studies have used food webs that have been defined using fish stomach content analyses. Accurate predator-prey relationships are very difficult to determine from stomach contents alone because of the problems in identifying the organisms, and the daily and seasonal variation in the feeding patterns of fishes. Many fishes also feed at two or more trophic levels making it difficult to assign them a discrete position in the food web.

Measurements of naturally occurring carbon and nitrogen isotopes in biotic tissues have provided an alternative method for determining food web structure. Tissue stable isotope ratios provide an accurate representation of what is assimilated into an organism from its diet, information that cannot be determined from stomach content analyses. Nitrogen isotopes are enriched during metabolic processes and increase in a constant manner from one trophic level to the next, whereas ratios of carbon do not change and can be used to identify food sources (Hesslein *et al.* 1991). These isotopes are being used to define food web structures, and to qualify and quantify the flux of organochlorines through the biota in these lakes.

ACTIVITIES IN 1992/93

Several species of benthic and planktonic invertebrates, and piscivorous and forage fish were collected from Laberge, Fox and Kusawa Lakes over the summer of 1992. All fishes were measured, weighed and sexed, and stomach contents, muscle samples and aging tissues were removed. Stomach contents were identified when possible and predominant invertebrates were sorted and frozen for future organochlorine analyses.

This past fall and winter, samples were analysed for organochlorines using the methods described in Muir *et al.* (1990). Lake, round, and broad whitefish, longnose sucker, northern pike and cisco were analysed from Laberge, Fox and Kusawa Lakes both individually and as composite samples. Some burbot and lake trout were analysed from these lakes for comparative purposes but the data are not included in this report. The invertebrates analysed over the past year were those most prevalent in fish stomach contents and those collected in sufficient numbers to allow for analyses. Samples of chironomids from benthic grabs and fish stomachs were analysed to determine if both methods of collection result in comparable levels of organochlorines in the biota.

Several species of fish and invertebrates have been analysed for carbon and nitrogen stable isotopes from Laberge, Fox and Kusawa Lakes using the methods described in Hesslein *et al.* (1991). Food web structures are being defined based on these results and future laboratory work. This information will allow us to quantify and quality the flux of organochlorines through the biota in these lakes using the available contaminant data.

RESULTS AND DISCUSSION

Concentrations of total chlordane, DDT, PCBs and toxaphene in invertebrate and fish samples presently analysed from Laberge, Fox and Kusawa Lakes are presented in Tables 1 and 2.

Preliminary invertebrate analyses indicate that organochlorine concentrations in Laberge samples are generally slightly higher or comparable to those found in Fox and Kusawa Lakes. Higher concentrations of contaminants in tricopterans from Fox Lake may be due to higher lipid levels. Concentrations of organochlorines in chironomids from fish stomachs are consistently lower than those samples collected from benthic grabs on Lake Laberge, indicating that contaminant levels in stomach contents are not representative of the true levels in biota. Other invertebrate samples are currently being collected and analysed to improve our understanding of among and within lake variability.

In general, the concentrations of organochlorines in lake and round whitefish, and longnose sucker from Laberge are higher than those found in Fox and Kusawa Lakes. These results are consistent with previous studies on the concentrations of organochlorines in lake trout and burbot from these lakes (Palmer 1992, Muir *et al.* 1992, Kidd *et al.* in press). Additional fish samples are being collected and analysed individually to improve sample sizes and clarify the inter-lake differences. Results from these analyses will also be used to determine if any relationships exist between organochlorine concentrations and age, length, weight or percent lipid of the fishes.

The results of this study will allow us to compare the biomagnification processes in these subarctic lakes to previous studies conducted on the Great Lakes. It is not known if contaminants are biomagnified through the biota of Yukon lakes in a manner comparable to warmer, more productive lakes. Data will be used to validate existing food web models, or to design one specifically for Yukon lakes. This model will be critical in predicting future trends of organochlorine contamination and the subsequent impacts on commercial, sport and native subsistence fisheries.

Expected project completion date: September, 1995

Additional sources of funding:

Arctic Institute of North America - Lorraine Allison Scholarship
Canadian Circumpolar Institute - Boreal Alberta Research Grant
DIAND - Northern Scientific Training Program Grant
Northern Research Institute Fellowship

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Table 1. Concentrations of chlordane, DDT, PCBs and toxaphene in invertebrates from Laberge, Fox and Kusawa Lakes, Yukon Territory (mean, ng/g wet weight).

Site	Sample	N	% Lipid	ΣChlordane	ΣDDT	ΣPCB	ΣToxaphene[1]
Laberge	Zooplankton	1	1.0	0.9	2.4	4.6	9.6
	Tricopterans (Limnephilidae)	2	2.3	0.5	1.2	2.1	2.2
	Molluscs						
	Fossaria sp.	2	0.9	0.5	0.6	1.8	1.3
	Gyraulus sp.	1[2]	0.2	0.3	1.9	2.5	2.2
	Chironomids						
	(benthic grabs)	3	2.5	2.0	27.0	10.9	7.3
	(fish stomachs)	3	3.4	0.9	12.7	6.1	1.6
Fox	Zooplankton	1	2.6	0.6	0.6	1.1	2.5
	Amphipods						
	Gammarus sp.	1	0.5	0.4	1.0	3.7	1.6
	Tricopterans (Limnephilidae)	1	12.4	7.4	24.7	26.2	18.9
Kusawa	Zooplankton	1	2.7	1.1	0.6	3.2	7.9
	Molluscs						
	Fossaria sp.	1	0.4	0.5	0.2	0.5	2.2
	Gyraulus sp.	1[2]	1.7	0.6	0.3	1.2	6.6

[1]Toxaphene concentrations calculated using a single response factor.
[2]Analyses included shells.

Table 2. Concentrations of chlordane, DDT, PCBs and toxaphene in fishes from Laberge, Fox and Kusawa Lakes, Yukon Territory (mean and range, ng/g wet weight).

Site	Species	N	% Lipid	ΣChlordane	ΣDDT	ΣPCB	ΣToxaphene[1]
Laberge	Lake Whitefish	9	2.4 (0.8-3.4)	3.4 (0.3-9.5)	20.4 (0.5-52.2)	16.4 (1.0-44.2)	18.0 (1.0-49.5)
	Round Whitefish	6	2.2 (1.5-3.2)	2.9 (0.8-5.7)	20.8 (7.0-33.8)	15.8 (6.0-28.6)	24.0 (3.9-40.5)
	Broad Whitefish	2	1.3	2.3	9.2	9.8	4.4
	Cisco	3	6.5 (5.1-7.7)	9.2 (7.3-11.4)	31.2 (29.3-32.7)	24.8 (23.5-27.4)	83.0 (64.6-107.5)
	Northern Pike	1	0.6	1.9	10.7	8.7	13.1
	Longnose Sucker	3	0.8 (0.5-1.0)	2.7 (1.2-4.2)	24.1 (18.8-29.0)	21.3 (18.2-22.9)	17.8 (12.2-23.4)
Fox	Lake Whitefish	1	-	0.5	3.0	4.5	3.1
Kusawa	Lake Whitefish	1	0.7	0.8	0.4	2.8	4.6
	Round Whitefish	1	0.9	1.4	5.8	6.2	6.1
	Longnose Sucker	1	1.2	1.9	11.9	19.1	5.9

[1] Toxaphene concentrations calculated using a single response factor.

FOOD CHAIN ACCUMULATION AND BIOLOGICAL EFFECTS OF ORGANOCHLORINES IN FISH FROM LAKE LABERGE AND OTHER YUKON LAKES

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PROJECT TEAM: D. Muir, L. Lockhart, D. Metner, B. Billeck, P. Wilkinson, R. Danell, T. Kenny, B. Grift, C. Ford, B. Rosenberg

OBJECTIVES

1. To measure biochemical stress indicators (e.g. hepatic mixed function oxidase enzyme activity) and concentrations of organochlorines (PCBs, toxaphene, etc.) in tissues of burbot, lake trout and whitefish of Lake Laberge and other lakes.

2. To determine temporal trends in organochlorines (OCs) and PAHs via analysis of archived fish and via dated sediment cores.

3. To contribute to studies of food chain contamination of organochlorines in Yukon Lakes.

INTRODUCTION

During 1990/91, samples of burbot liver, as well as muscle of lake trout and whitefish from Lake Laberge were shown to contain high levels of organochlorine pesticides (toxaphene, DDT) and polychlorinated biphenyls (PCBs). The objectives of this project are to examine biomarkers for PCBs and toxaphene, to determine temporal trends in organochlorine levels via retrospective analysis, and to determine food chain contamination in Lake Laberge. The elevated levels of toxaphene, DDT and PCBs in fishes from Laberge suggest that there could be biological effects such as reproductive failure, similar to those observed in the Great Lakes. One measure of exposure to toxicants is hepatic mixed function oxidase (MFO) activity which has been shown, when used as an *in vitro* assay, to be correlated with concentrations of planar PCBs and dioxins in Great Lakes fish.

Lake sediments are being used to examine temporal trends in the deposition of particle reactive contaminants (OCs, PAH, heavy metals) in lakes in NWT. They are being employed in this study to examine the possibility that Lake Laberge and other lakes have been contaminated by spills of these chemicals as well as by atmospheric deposition. Temporal trend studies are also supplemented by analysis of archived whitefish available from J. Clayton at the Freshwater Institute.

MATERIALS AND METHODS

Sample Collection

Fish samples were collected March 10-22, 1992, from Lake Laberge and Kusawa Lake (control site) by Fisheries and Oceans Canada (DFO) personnel (D. Metner, P. Wilkinson, B. Billeck, R. Danell) assisted by Ph.D. student K. Kidd (University of Alberta) in cooperation with biologists from the federal Departments of Environment and Fisheries and Oceans in the Yukon, and the Yukon Government. Livers were obtained from lake trout, burbot and whitefish, quick frozen on dry ice, and stored at -80°C in Winnipeg until assayed for MFO activity. Weight, length and sex of the fish were recorded. Otoliths were taken for aging. Samples for OC analysis were stored on ice in the field, frozen in Whitehorse and shipped frozen to Winnipeg where they were stored at -40°C until analysis.

Sediments were collected from Lake Laberge and Kusawa Lake in March 1992. A large (16 cm diam.) KB corer and a box corer (30 x 30 cm) were used to collect sediments at the deepest points in Laberge and Kusawa Lakes. Cores were sliced at 0.5 cm intervals and the sediment placed in "WhirlPak" bags. Further details of sediment sampling are given in Lockhart (1993, this volume). On a second sampling trip during March 1993 we obtained additional cores from Fox and Little Atlin Lakes.

Sample Analysis

OCs in fish tissues: Procedures for the analysis of fish muscle and liver samples are described in this volume by Muir and Lockhart (1993) and further details on the extraction and cleanup steps are given by Muir *et al.* (1990). Analysis of planar PCBs and chlorinated dioxins/furans in fish tissues is described by Muir (1993, this volume).

Sediments: Sediments were freeze-dried in their sample bags. Dried sediment was assayed for ^{210}Pb, ^{226}Ra, ^{137}Cs and ^{7}Be and the profile of radionuclides was used to date each slice. Sediment (10g) was extracted with dichloromethane (DCM) in a Soxhlet apparatus using glass thimbles with sintered glass frits. Internal standards of deuterated PAHs were added at the extraction step. The DCM extract was split 1:1 for determination of PAHs and OCs. To recover PAHs the extract was chromatographed on a silica column (topped with 1 cm alumina) and eluted with hexane (to recover alkanes) followed by hexane:DCM (1:1) for 2 to 6 ring PAHs. OCs were isolated by chromatography on Florisil (Muir *et al.* 1990).

Capillary gas chromatographic analysis of PCBs, toxaphene, DDT and other organochlorines was carried out on a 60 m x 0.25 mm DB-5 column (0.25 um film thickness) under conditions described by Muir *et al.* (1990). Organochlorines were quantified by use of external standards of individual PCB congeners, DDT-group, chlordane-related compounds, and hexachlorocyclohexanes and chlorobenzenes. Toxaphene was quantified using a single response factor based on the area of 19 peaks in the standard. PAHs were quantified by capillary GC-mass spectrometry (HP 5980-5970 MSD) using an internal standard technique. Sixteen unsubstituted "priority pollutant" PAHs, plus alkylated naphthalenes and phenanthrenes were measured. The biogenic compounds, perylene and retene were also determined.

MFO activity: Fish livers were homogenized, centrifuged to obtain a microsomal pellet and assayed for two cytochrome-P-450 related catalytic activities, ethoxyresorufin-O-deethylase (EROD) and aryl hydrocarbon hydroxylase (AHH) as well as cytochrome P-450 levels.

Quality Assurance

An interlaboratory study was conducted with AXYS Analytical Labs (Sidney, BC) on the analysis of organochlorines in burbot liver. The laboratories also participated in interlab check sample programs on PAHs in sediments and on organochlorines in sediments and fish tissues during 1992/93.

RESULTS

Interlab Comparison

There was reasonable agreement between the two laboratories on total (ortho-substituted) PCBs and total chlorobenzenes (s-CBz) (within 10% for lake trout muscle), and for toxaphene, and total chlordane (s-CHLOR) (within 20-50%), but poor agreement for total DDT group (s-DDT), dieldrin, and s-HCH (Table 1). Results were compared on a lipid weight basis because lipid contents obtained by each lab differed, possibly as a result of changes during storage and due to the use of different extract solvents. Previous interlab comparisons with Atlin Lake samples showed better agreement between the two labs for the same analytes (Table 1).

PCB and Organochlorine Pesticide Concentrations in Fishes

Concentrations of major organochlorines in lake trout and burbot liver from Lake Laberge (Table 2) were similar to those observed in previous studies (Eamer 1991). Variability among individual fish is high, with standard deviations of 50 to >200%. Some of the variability is accounted for by sex. In burbot liver, for example, all organochlorine concentrations (especially PCBs and DDT-related compounds) were two-times higher in males than in females. Burbot spawn in the spring months and most of the female burbot were in spawning condition. They may therefore have transferred lipid associated contaminants from liver to egg mass. Differences in organochlorine levels were also observed between male and female lake trout in Lake Laberge but these were largely accounted for by elevated levels in one fish. Differences in lipid content could also
account for variability among individual samples, however, lipid normalization did not greatly reduce the standard deviations. Although concentrations of major organochlorines in lake trout from Kusawa Lake were lower than those in Laberge, by a factor of 5 to more than 10-fold (in the case of s-DDT), lipid normalized concentrations were similar because of lower lipid content of fish from Kusawa. Further discussion will await information on age and stable isotope results in these fish.

Table 1. Interlaboratory comparison of organochlorine levels in lake trout muscle and burbot liver homogenates by DFO Winnipeg and by Axys Labs (Sidney, BC).

			Means (ng/g lipid wt basis)			
Location	Species	N	ΣPCB	ΣDDT	ΣCHLOR	Toxaphene[1]
Lake Laberge - DFO	L. Trout	3	2010	2830	440	2840
Lake Laberge - Axys	L. Trout	3	1770	6800	518	4370
Lake Laberge - DFO	Burbot	1	4110	6870	810	2840
Lake Laberge - Axys	Burbot	1	5630	16500	865	4370
Atlin Lake - DFO	Burbot	6	530	210	450	2830
Atlin Lake - Axys[2]	Burbot	6	400	300	430	3120

[1] Toxaphene quantified with a single response factor.
[2] Results from a report by Axys Analytical Services Ltd. to Indian and Northern Affairs Canada, December 1991.

The DDT-related compounds (*p,p'*-DDE, -DDD,-DDT and *o,p'*-DDE,-DDD and -DDT) were the major organochlorine compounds observed in both species from Lake Laberge with (arithmetic) mean wet weight concentrations of 938 ng/g in lake trout muscle and 3020 ng/g in burbot liver (Table 2). As has been noted in previous analyses of Lake Laberge fish these s-DDT levels exceed concentrations in the same species from NWT by 4 to greater than 10-times which suggests local sources rather than an atmospheric pathway of contamination. The 56-fold higher levels of s-DDT in lake trout from Lake Laberge than from Kusawa Lake (12-fold for males on a lipid wt basis) is consistent with the local source hypothesis for Laberge.

PCB levels in Lake Laberge burbot liver and lake trout were also elevated, with means ranging from 820 ng/g (wet wt) in lake trout to 1380 ng/g in burbot liver. Kusawa Lake fish had 15-fold lower ΣPCB levels although they were comparable in weight (mean 1399 g in Kusawa versus 1632 g in Laberge). PCB levels in Kusawa Lake lake trout were similar to those observed in lake trout from NWT lakes (Muir and Lockhart 1993).

Toxaphene levels in Lake Laberge burbot liver were within the range reported previously for pooled samples (Eamer 1991). Toxaphene in lake trout muscle from Lake Laberge was 6-times higher than in the same species in Kusawa Lake. However, lipid normalized concentrations of toxaphene were similar in both lakes due to the lower lipid content of the trout in Kusawa. These toxaphene results are calculated with a single response factor which is a similar approach to that used by AXYS labs (Table 1) in determining toxaphene by negative-chemical ionization MS. Other organochlorine pesticides such as HCH, chlordane and dieldrin were also present at similar concentrations in Kusawa and Lake Laberge lake trout lipid (Table 3).

Lake whitefish collected in 1974 had 2-fold higher concentrations of ΣDDT, 4.5 fold lower PCBs and 2-fold lower toxaphene than those from 1992 (Table 2). Lipid content of the 1974 animals was higher (4.5%) than the 1992 group but the differences were still apparent after lipid normalization. One sample from 1992 had elevated PCB levels compared to the 5 others but omitting it from the mean still resulted in higher levels in 1992 than 1974 (by 3.5 fold).

PAH, PCB and Organochlorine Pesticide Concentrations in Lake Sediments

Profiles of ΣDDT, ΣPCB, toxaphene and total PAHs (minus perylene and retene) in Core 3 from Lake Laberge are shown in Fig. 1. This core had a single sedimentation rate of 650±31 g $m^{-2}yr^{-1}$ with no disruptions in the ^{210}Pb profile. Analyses of the sediments for OCs have only been completed back to slices dated to 1976. Concentrations of organochlorines in top slices of this core are similar to what has been observed in lakes in the High Arctic (e.g. Hazen Lake and are generally lower than levels in northwestern Ontario lakes (Lockhart 1992). Results for PAHs (total of 17 priority pollutants minus retene and perylene) in core 3 are available back to 1894 (Fig. 1). The PAH concentrations are quite constant for the past 50 years and show greater fluctuations between 1900 and 1940. It would be interesting to know whether these fluctuations correspond to an addition of lampblack to the ice to enhance melting, a practice which was carried out earlier in the century. Concentrations of PAHs are lower than those found in northwestern Ontario and similar to what we have found in other Arctic lakes (Lockhart 1992).

MFO Activity

EROD activity in lake trout from Lake Laberge and Kusawa Lake was similar (Table 3). Higher levels of EROD were found in burbot liver. Both burbot and whitefish were in spawning condition, a factor which is known to influence MFO activity. EROD and AHH activity in lake trout is similar to what we have reported for lake trout from the Saqvaqjuac area (63° N). ERODs in male lake trout from Laberge are about 2-fold higher than we found in Buchanan Lake char but similar to levels observed in char from Amituk Lake (Lockhart 1992b). There appeared to be no correlation between total TCDD equivalents (due to planar PCBs; see Muir 1993) in char muscle and burbot liver based on the analysis of a subset of the fish reported in Table 3.

Table 2. Organochlorine contaminants in lake trout, lake whitefish and burbot from Yukon Lakes determined by DFO Winnipeg during 1992/93.

Lake	Year	Species	Tissue	N	Sex		Concentration (ng/g wet wt) %Lipid	ΣCBz	ΣHCH	ΣCHLOR	ΣDDT	ΣPCB	Toxa-phene	Dieldrin
Laberge	1992	L. Trout	Muscle	4	M	Mean	8.7	4.3	4.3	85.8	1307	1177	725	1.8
						SD	3.4	1.1	1.8	68.5	1332	1219	612	0.8
Laberge	1992	L. Trout	Muscle	2	F		7.8	2.8	4.4	36.8	199	103	227	1.2
Laberge	1992	Burbot	Liver	3	M	Mean	38.4	31.7	35.0	654	6310	2983	4283	14.1
						SD	6.2	10.4	4.8	23.7	1464	638	276	5.8
Laberge	1992	Burbot	Liver	3	F	Mean	30.7	17.2	27.3	218	1908	693	1631	7.5
						SD	14.8	3.0	3.4	20.8	242	128	199	2.3
Laberge	1992	Whitefish	Muscle	6	M+F	Mean	2.5	1.3	1.7	10.9	155	279	43.8	0.3
						SD	1.0	0.4	0.6	5.7	186	468	21.0	0.2
Laberge	1974	Whitefish	Muscle	10	M+F	Mean	4.5	2.4	11.4	6.3	389	60.9	22.3	2.1
						SD	1.2	0.9	4.6	3.2	463	39.9	12.1	0.6
Kusawa	1992	L. Trout	Muscle	6	M	Mean	2.2	1.1	1.5	15.7	29.6	69.6	106	0.5
						SD	2.2	1.1	1.5	4.5	28.9	25.6	53.0	0.4
Kusawa	1992	L. Trout	Muscle	6	F	Mean	1.3	1.2	0.9	18.9	58.7	101	135	0.4
						SD	0.9	0.7	0.6	12.2	95.0	119	103	0.2

[1] Toxaphene quantified with a single response factor (on a lipid normalized basis).

Table 3. Mixed function oxidase enzyme activity in liver from fishes in Lake Laberge and Kusawa Lake.

Location	Species	Sex	N	EROD	BAP	TEC (pg/g)
Laberge	Lk. trout	M	5	0.0130±0.0040	0.0442±0.0247	47±42
		F	2	0.0085	0.0350	11
Laberge	Burbot	M	10	0.0095±0.0040	0.0077±0.0040	255±130
		F	10	0.0041±0.0020	0.0061±0.0036	83±14
Laberge	Whitefish	F	6	0.0132±0.0061	0.0373±0.0142	
Kusawa	Lk. trout	M	12	0.0226±0.0139	0.0547±0.0234	
		F	8	0.0134±0.0107	0.0359±0.0185	

CONCLUSIONS

The organochlorine contaminants in individual lake trout and burbot samples from Lake Laberge and Kusawa Lake were characterized by large variations among individuals of the same size class, which could not be entirely accounted for by sex or lipid differences. It may reflect differences in feeding and in growth rates which additional information on age and stable isotopes in these individuals will help clarify. The similarity of the lipid-based results for most organochlorines, except s-DDT and s-PCBs, in lake trout from Kusawa and Laberge suggests that pathways of accumulation are similar in both lakes, i.e. via atmospheric inputs. However, additional data are needed because there are potential errors in calculating lipid based results when lipid content is only around 1% and sample size is small.

The analysis of sediment cores from Lake Laberge is incomplete at the present time but Core 3 appears to have preserved an excellent record (no disturbance of the ^{210}Pb profile) of contaminant inputs. Concentrations of OCs and PAHs are not high by southern Canadian standards but fluxes (concentrations x sedimentation rates) of 1-2 $\mu g\ m^{-2}yr^{-1}$ are similar to those in northwestern Ontario and higher than we have found in small arctic headwater lakes in NWT.

Despite high levels of PCBs in burbot liver and lake trout muscle, relative to most other arctic lakes, MFO activities are not elevated in fishes from Lake Laberge. However, burbot and whitefish were in spawning condition during the sampling in March 1992. Further sampling in the fall would be more appropriate for these species and would allow comparison with other data.

Future plans include the analysis of additional cores from Laberge, Fox and Atlin Lakes and the continuation of sediment sampling from other lakes to more thoroughly characterize inputs of hydrophobic organics. Additional fish from Lake Laberge will also be analysed for MFO and planar PCBs to determine if a correlation exists.

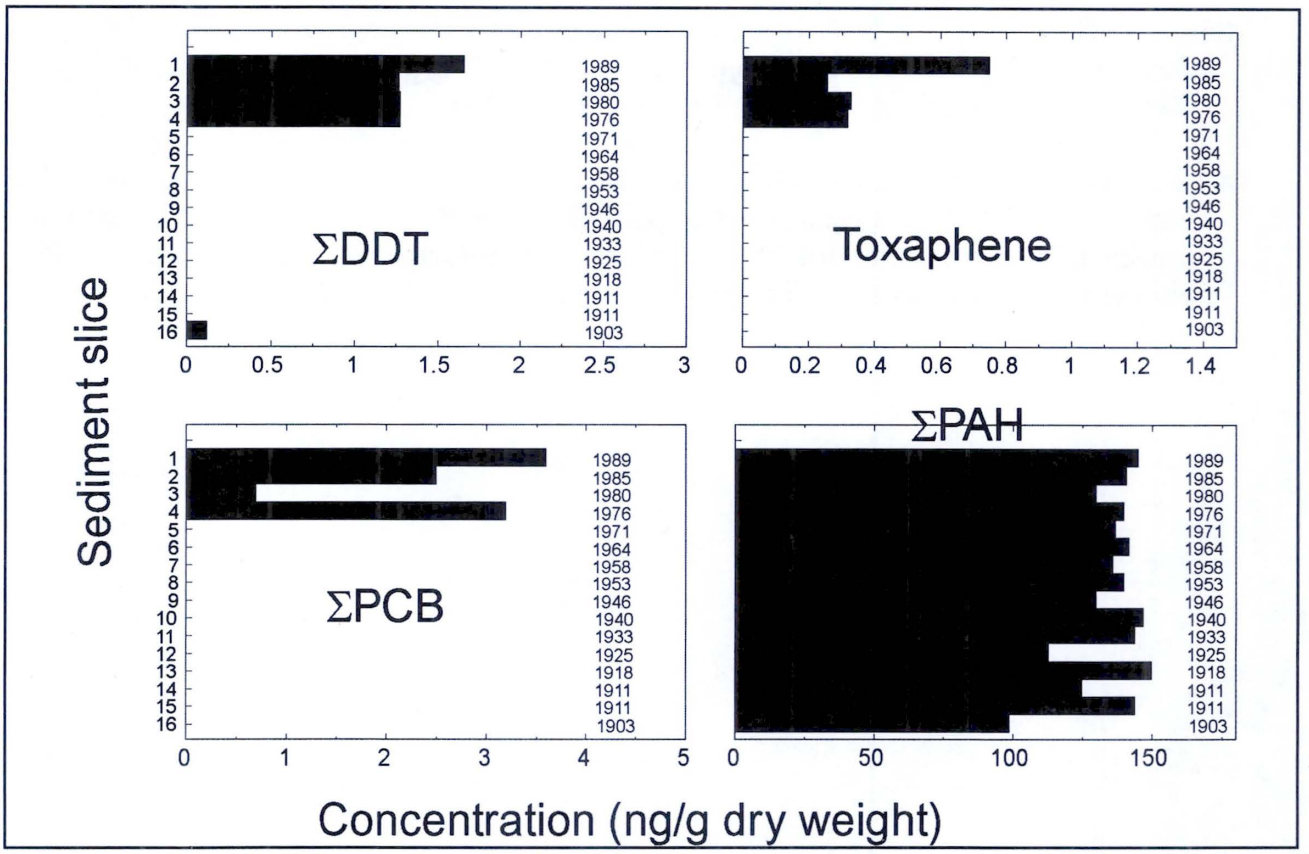

Figure 1. Profiles of ΣDDT, toxaphene,ΣPCB and ΣPAHs (minus perylene + retene) in Lake Laberge Core 3.

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MODELLING INORGANIC AND ORGANIC CONTAMINANTS IN ARCTIC LAKES AND FOOD CHAINS

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OBJECTIVES

1. To estimate the fate and transport of inorganic and organic contaminants in Arctic freshwater lakes and bioaccumulation in lake food chains, and to identify factors controlling chemical movement and exposure to biota.

2. To identify long-term and transient sinks of contaminants and the time response to changes in contaminant inputs.

3. To examine spatial and temporal variability in contaminant dynamics in lakes and determine lakes vulnerable to high contaminant levels.

DESCRIPTION

Efforts have been and are underway to measure contaminant levels in fish from lakes covering a wide geographic range in order to provide consumption guidelines and baseline knowledge in remote areas (e.g., Muir *et al.*). In addition, contaminant concentrations in lake sediments are being surveyed to shed light on temporal and geographic patterns of contaminant deposition (e.g., Lockhart *et al.*). To rationalize these efforts it would be useful to target lakes in which fish are vulnerable to high contaminant concentrations. To use sediment profiles as chronological records of chemical deposition it is necessary to link lake concentrations, amounts and deposition or loading. Mechanistic mass balance models have proven useful for establishing these linkages (e.g., Diamond *et al.* 1992, Mackay 1989) and can be used to identify factors leading to elevated contaminant levels.

Previous work on Arctic lakes has focused on understanding the hydrologic and thermal regimes, nutrient dynamics and primary productivity (e.g., Schindler *et al.* 1974, Rigler 1975, 1978, Welch and Bergmann 1985, Bergmann and Welch 1985). This research provides an invaluable basis upon which to build an understanding of contaminant dynamics. The current Amituk Lake study of Gregor will provide the basis for an in-depth understanding of contaminant dynamics in this system. What is needed, however, is to extend this understanding to other lakes and to build a general understanding of the behaviour of contaminants in Arctic lakes and food chains.

ACTIVITIES IN 1991/92

Firstly, we have developed a mass balance model of chemical fate and transport using fugacity/ aquivalence as equilibrium criteria (Mackay 1991, Mackay and Diamond 1989) and based on the QWASI model (Quantitative Water Air Sediment Interaction, Mackay *et al.* 1983, Figure 1). The QWASI model, which was developed for temperate lakes, was adapted to simulate Arctic lakes as follows:

1. ice is assumed to cover the lake for 10 months per year,
2. a lake-specific fraction of the meltwater (and hence chemical) enters and mixes with the water column, with the remainder flowing over the lake surface and leaving through the outflow, and
3. during summer the average water temperature is assumed to be 2°C.

The model considers the two month ice-free period during which steady-state conditions are assumed to apply (this is a simplifying assumption which, it is recognized, is questionable). Consequently, we assume steady-state inputs of chemical from an inflow, of which only a fraction mixes with the lake.

Secondly, the model was calibrated and partially tested using data on phosphorus budgets in Char Lake (Cornwallis Island, 74°N, 94°W, de March 1975, 1978, Schindler *et al.* 1974) and Toolik Lake (Alaska, 68°N, 149°W, Whalen and Cornwell 1985), these being the only lakes and chemical for which there were sufficient data. Good agreement was achieved between model estimates and observed values using flow-through fractions of 0.9 and 0.5 for Char and Toolik Lakes, respectively, i.e., 90% and 50% of phosphorus flowed directly through the lake and did not enter and mix with the water column. Clearly knowledge of this fraction is critical to estimating total chemical dynamics but is difficult to obtain since this fraction is lake-specific and its determination requires extensive limnological and chemical characterization.

Thirdly, the model was used to consider organochlorine dynamics. Physical-chemical properties of chemicals at 4°C were estimated using the Doucette-Andren equation for solubility (Doucette and Andren 1988) and the Antione-Claperon equation to estimate vapour pressure (Hinkley *et al.* 1990). Since a complete dataset is unavailable for any one lake, we assembled a "composite" picture of steady-state dynamics in a lake similar to Char Lake using the following data:

1. air concentrations from Alert (82°N, Patton *et al.* 1989)
2. organochlorine concentrations in snowmelt entering Amituk Lake (75°N, 95°W, Gregor unpubl. data), and Ice Island (81°N, 97°W, Gregor 1991), and
3. Char Lake limnology (Kalff 1970, de March 1975, 1978, Rigler 1975, 1978, Schindler *et al.* 1974).

We were limited to considering DDT, chlordane and HCH by the availability of snowmelt concentrations (Gregor, unpubl. data, Gregor 1991). DDT was considered as the total of p,p'-DDT, o,p'-DDT, p,p'-TDE and p,p'-DDE. Chlordane was assumed to consist of methoxychlor, heptachlor, heptachlor epoxide, α-chlordane and γ-chlordane. Total HCH was obtained by summing results for α-HCH and γ-HCH.

Model results were compared with organochlorine concentrations measured in sediment (Muir and Lockhart 1992).

Fourthly, a food chain model was developed based on the work of Gobas (1992). The model uses chemical concentrations (phosphorus) in water and sediment generated by the lake model. A typical freshwater food chain is very short, consisting of benthos (mostly chironomids) and Arctic char (*Salvelinus alpinus*). The food chain in some lakes has another trophic level of juvenile char that are consumed by older fish. The model assumes that benthos are in equilibrium with sediment. Fish take up chemicals from water through gill ventilation and from food through ingestion. Chemicals are lost by gill ventilation, egestion (in faeces) and metabolism. Chemical concentrations are diluted by growth. Rates of food intake and egestion at 2°C, gill ventilation and elimination, and growth at 10°C were estimated using the relationships of Gobas (1992). Rates of chemical metabolism in char are not available in the literature but have been assumed to be very slow.

Values specific to the food chain model are as follows:

benthic lipid fraction	0.1 (Gobas, pers. comm.)
char lipid fraction	0.033 (Muir and Lockhart, 1992)
average weight of char (13 yrs old)	0.3 kg (MacCallum and Regier 1984)

RESULTS

Measured and estimated phosphorus budgets agreed relatively well (Table 1). This agreement was achieved by manipulating the flow-through fraction (0.9 and 0.5 for Char and Toolik Lakes, respectively), and the particulate to dissolved partition coefficient, K_D (L/kg) (K_D for inorganics must be obtained from measured values whereas for organics it is predicted from K_{OW}). The results indicate that over 75% of phosphorus entered the lakes from snowmelt with the remainder entering from direct atmospheric deposition. Phosphorus retention in the lakes was relatively low at 10% and 34% for Char and Toolik Lakes, respectively. The remaining portion was exported.

Estimated organochlorine concentrations in sediment and char are within an order of magnitude of measured values (from Amituk and Hazen Lakes, respectively, Muir and Lockhart 1992) with the exception of chlordane (Table 2). Estimated concentrations in char are, however, about five times lower than average values reported by Muir and Lockhart (1992) for Amituk Lake. The latter measurements show considerable variability and appear to be anomalously high compared with other Arctic lakes sampled. The relatively consistent underestimation of char concentrations by the food chain model may be due to using bioenergetic relationships developed for temperate fish at 10°C rather than the 2°C average of Arctic lakes. In general though, the correspondence between measured and estimated concentrations is considered reasonable since model estimates represent a composite picture with air and inflow concentrations, and limnological characteristics coming from different locations and lakes, which are also different from the lakes from which the measured values have been taken.

The results of the lake model are illustrated in Figure 2 for DDT. Over 99% of DDT enters from snowmelt with direct atmospheric contributions being negligible. Over 90% of DDT is exported with less than 10% being buried; transformation rates are too slow for this process to be important. Loss or gain of chemical through air-water exchange is also negligible because of cold temperatures and the low volatility of DDT. Thus, these results, which are similar to those for HCH and chlordane, indicate that the organochlorines behave similarly to a conservative inorganic such as phosphorus, i.e., because of low temperatures, rates of volatilization and transformation are minimal leaving advection (inflow and outflow), and sediment/water exchange (resulting in burial) as the main transport mechanisms.

Figure 3 illustrates the results obtained from a mass balance of DDT for fish. Over 85% of chemical enters through food (chironomids) with the balance entering across the gills. About half is lost via gill ventilation, almost 40% via egestion and less than 5% is lost through growth dilution and metabolism. These results are consistent with the behaviour of a highly hydrophobic chemical but dilution is minimized by slow growth.

The results of a sensitivity analysis indicate that estimated fish and sediment concentrations are insensitive to doubling or halving chemical vapour pressure or solubility (Fig. 4). Estimated concentrations in biota are, not surprisingly, dependent on lipid fractions in both trophic groups (Fig. 5).

DISCUSSION AND USE OF RESULTS

Model results, which are within an order of magnitude of measured values, suggest that organochlorines in water and sediment approach equilibrium. This is illustrated in Figure 6 which shows the similarity of water and sediment aquivalences (an indication of equilibrium status). This suggests that chemical concentrations within the lake can be approximated by organic carbon partitioning as estimated by K_{ow}, particularly for chemicals with low K_{ow}. Chemicals in air have significantly lower aquivalences indicating that chemicals will strive to diffuse from water to air. However, diffusion (volatilization) is restricted by low temperatures and the brief ice-free season. Aquivalences in fish vary from being equal to (HCH) to three times greater (DDT) than water aquivalences, the difference being related to K_{ow}. For lower K_{ow} chemicals such as HCH (log K_{ow} of 3.8) it appears that fish can be predicted by partitioning between lipid and water, and that chemical is not appreciably bioaccumulating through food or biomagnifying in this simple food chain. For chlordane (log K_{ow} of 5.4) and DDT (log K_{ow} of 6), bioaccumulation and biomagnification moderately elevate fish concentrations beyond that expected by partitioning alone. Fish concentrations also may be somewhat elevated by slow growth rates, the longevity of fish and slow metabolic rates.

As noted above, lakes are largely conduits for chemicals, with most chemical flowing through the lake and progressively smaller fractions of inputs mixing with the water column and being retained in sediment. These results suggest that chemical concentrations in the lake and biota depend on the hydrologic regime, or specifically, the flow-through fraction which controls chemical loading.

FUTURE WORK

In 1993/94 we plan to continue developing and testing the lake and food chain models, and extend the model to treat additional chemicals (e.g., HCB, PCBs, toxaphene, cadmium). Data will be used from Amituk Lake as they become available. As part of the 1993 Amituk Lake field studies supervised by Gregor, Hilary Freitas, a M.A.Sc. candidate under my supervision, will conduct a detailed study of particle dynamics in the lake. She will also sample Char Lake sediment to obtain organochlorine concentrations so that the model can be well-tested for this system.

Presently, H. Freitas and R. Laposa (an NSERC summer student), who have done much of the model development and testing, are developing an unsteady-state version of the lake model that will treat variations in ice cover and water flow over summer. H. Freitas has extended the steady-state version of the model to treat multiple, interconverting species such as mercury (Diamond *et al.* 1992). Again, we are awaiting data with which to test the model.

Efforts are currently underway to examine chemical dynamics in Hawk Lake (63°N 90°W) at Saqvaqjuac using the data of Welch (1985) and Welch and Legault (1986). Additional lakes will be considered depending on data availability, e.g., lakes around Schefferville, Quebec and Baker Lake (the latter are being sampled this summer by Muir and co-workers).

Expected project completion date: March 31, 1997

Partners: Contaminant chemistry data in water, sediment and fish provided by Gregor (DOE, NWRI) and Muir (DFO, Freshwater Institute). Air concentration data provided by Bidleman and Barrie (DOE, AES). Additional information to be obtained from Welch (DOE, Freshwater Institute), Lockhart (DOE, Freshwater Institute) and possibly Lean (DOE, NWRI).

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Table 1. Calibration using Phosphorus: a comparison of observed values and model estimates.

	Rates of Phosphorus Movement (kg/yr)			
	Toolik Lake		Char Lake	
	Observed[1]	Estimated	Observed[2]	Estimated
Inflow P[3]	51.9	42.5	4.54	5.04
D[4]	150.4	152.3	2.28	2.28
Outflow P	57.0	44.4	4.54	5.34
D	93.4	105.1	1.70	3.57
Burial	78.8	77.4	1.41	0.99

[1]Whalen and Cornwell (1985).
[2]Schindler *et al.* (1974b).
[3]P: Particulate
[4]D: Dissolved

Table 2. Lipid-normalized fish concentrations: a comparison of model predictions and measured chemical concentrations.

Chemical Modelled	Fish Concentration (ng Chemical/g lipid)		Inflow Concentration Used in Model (ng/L)
	Measured[1]	Model Prediction	
ΣHCH	59±7	24.0	3.8[2]
ΣHCH	59±7	1.53	0.226[3]
ΣHCH	59±7	3.20	0.482[4]
ΣHCH	59±7	4.30	0.680[5]
DDT	438±331	523.4	0.96[2]
DDT	438±331	382.1	0.698[6]
DDT	438±331	126.3	0.230[7]
Chlor	1040±669	476.9	1.76[2]
Chlor	1040±669	29.3	0.108[7]
Chlor	1040±669	36.4	0.134[6]
	Sediment Concentration (ng chemical/g dry weight)		
	Measured[8]	Predicted	
ΣDDT	2	4.30	0.96[2]

1. Muir and Lockhart (1992), Hazen Lake.
2. Gregor, unpubl. data from Amituk Lake.
3. Gregor and Gummer (Table 8.9), Ice Island *In:* Gregor 1991.
4. Arctic Labs (Table 8.9), Ice Island *In:* Gregor 1991.
5. University of South Carolina (Table 8.9), Ice Island *In:* Gregor 1991.
6. Arctic Labs (Table 8.8), Ice Island *In:* Gregor 1991.
7. Gregor and Gummer (Table 8.8), Ice Island *In:* Gregor 1991.
8. Muir and Lockhart (1992), Amituk Lake.

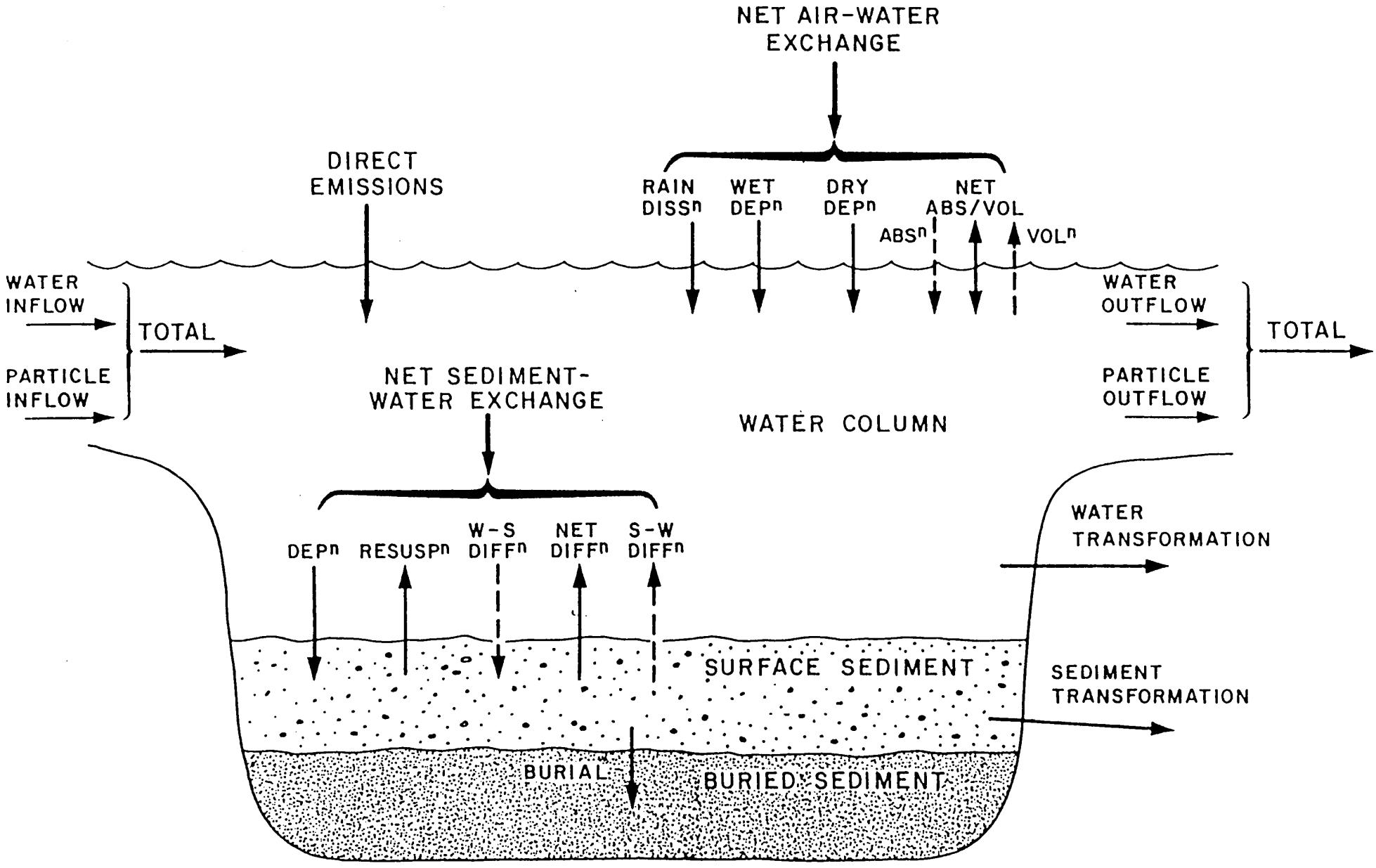

Figure 1. Processes in QWASI lake model.

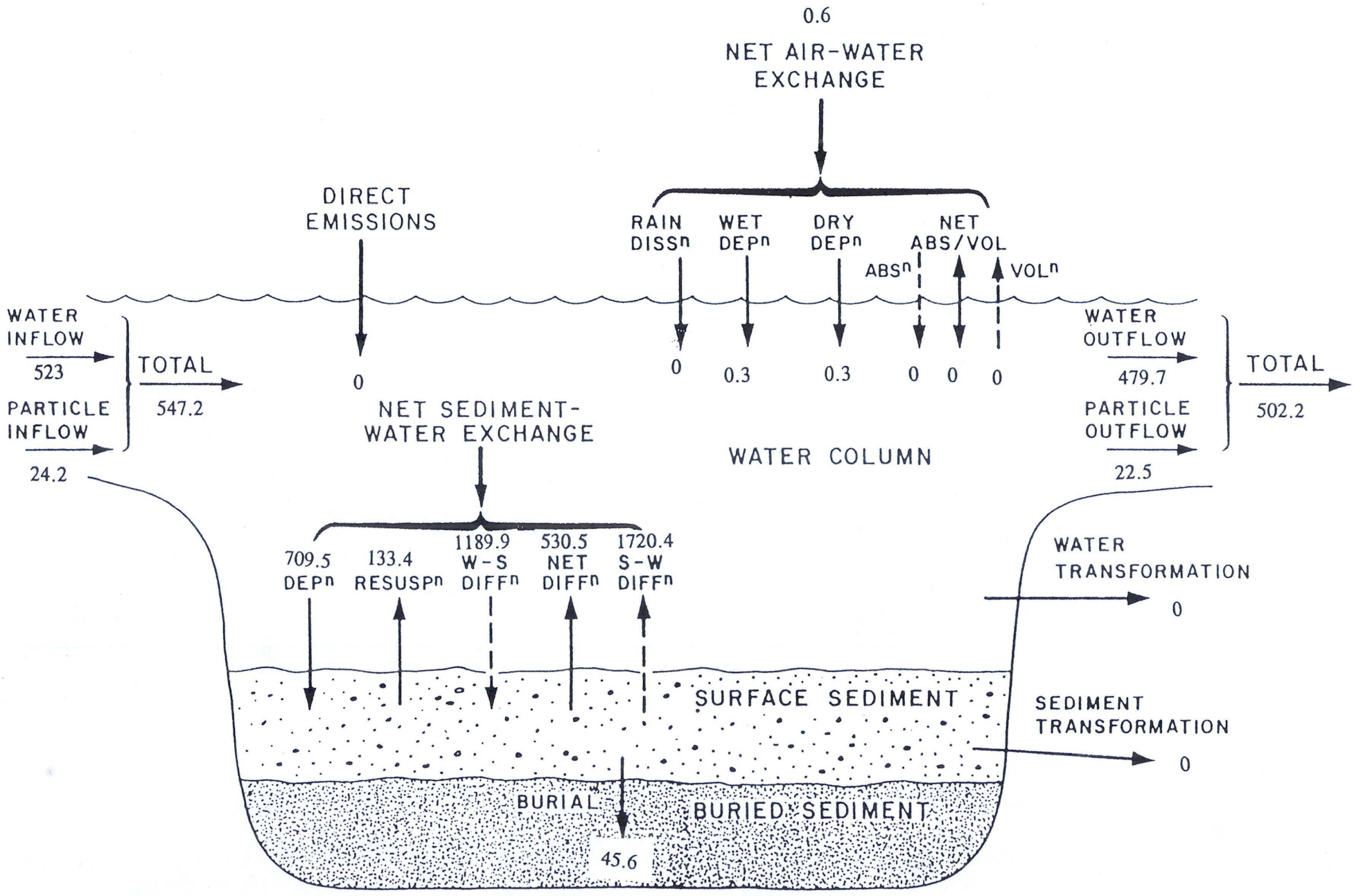

Figure 2. Rates of movement (mg/y) of DDT as estimated by lake model.

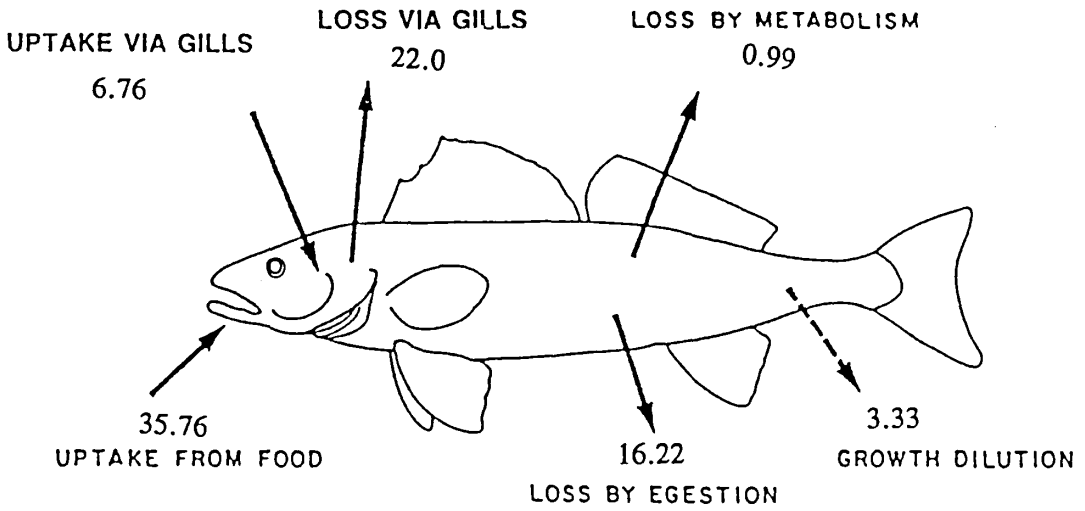

Figure 3. Rates of bioaccumulation processes (ng/d) for DDT in Arctic char.

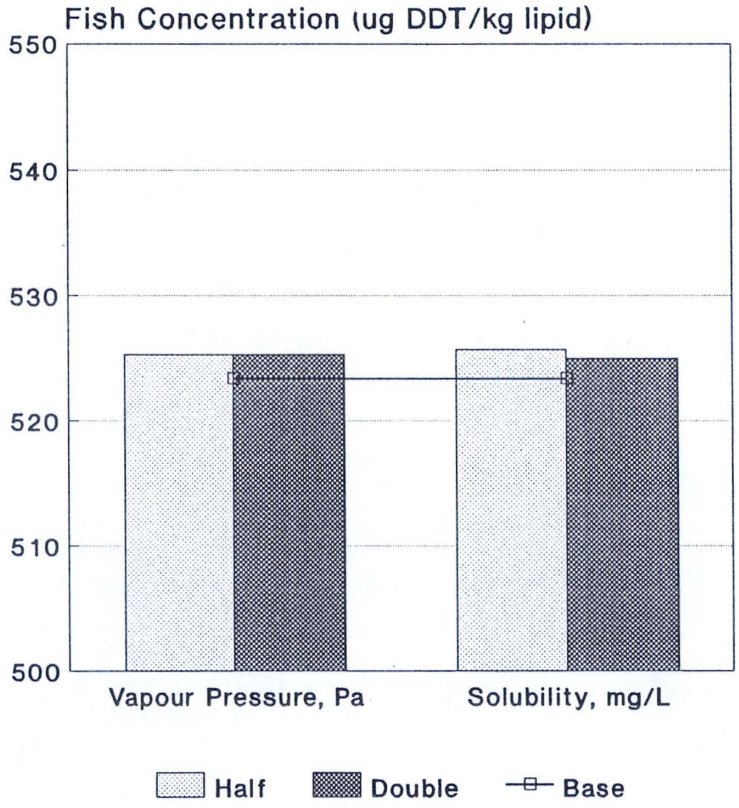

Figure 4. Sensitivity of lake model to changes in physical-chemical properties.

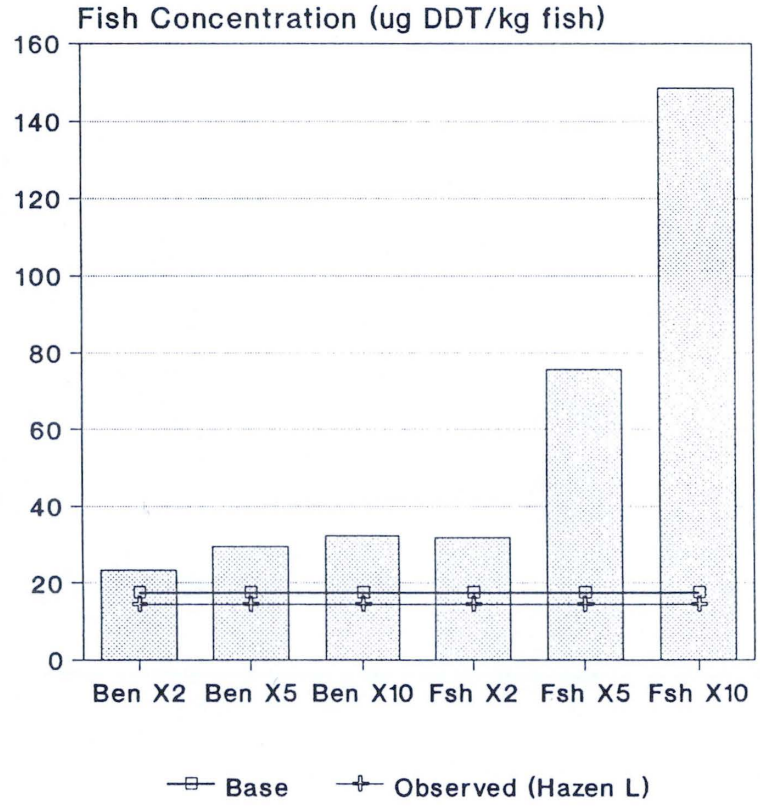

Figure 5. Sensitivity of food chain model to benthic & fish lipid fractions.

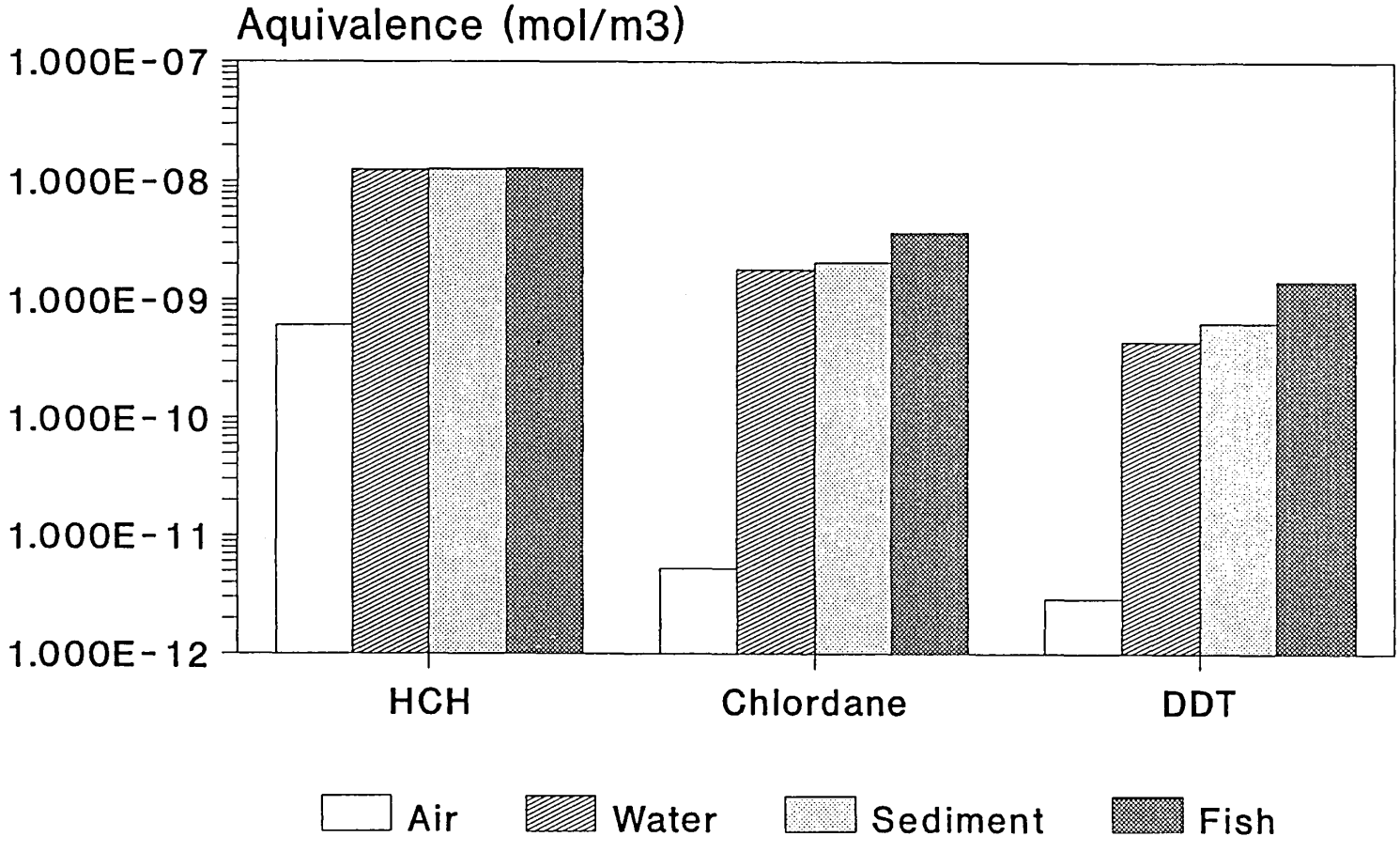

Figure 6. Comparison of aquivalences in air, water, sediment and fish.

TRENDS AND EFFECTS OF ENVIRONMENTAL CONTAMINANTS IN ARCTIC SEABIRDS, WATERFOWL, AND OTHER WILDLIFE. STUDY I. CONTAMINANTS IN WATERFOWL: NATIVE HARVEST IN LABRADOR

PROJECT LEADER: B.M. Braune, National Wildlife Research Centre, Canadian Wildlife Service, Environment Canada, Hull, Québec

SUPPORTING AGENCIES: Indian and Northern Affairs Canada
Health and Welfare Canada

CONTRACTOR: Labrador Inuit Association, Nain, Labrador, organized and supervised all collections.

OBJECTIVES

To address Inuit concerns in Labrador with regard to contaminant levels in their food through collection and chemical analysis of the game birds and eggs hunted and eaten by those people. The chemical data for the collected birds and eggs will be submitted to Health and Welfare Canada so that the risk to human health of eating those birds may be assessed, and so that consumption advisories may be issued, if necessary.

RESULTS

As part of the survey of contaminants in wild foods (waterfowl/game birds) currently being carried out by the Canadian Wildlife Service, 22 collections of waterfowl were made from 13 sites across the Northwest and Yukon Territories between 1988 and 1990 (Braune 1991). Results of organochlorine and metal analyses (Tables 2 and 4) are currently being evaluated by Health and Welfare Canada for risks to human health from eating these waterfowl.

During 1991 and 1992, a total of 50 collections of birds and 6 collections of eggs were received from six native communities in Nunavik, northern Québec, plus a collection each of Canada and Lesser Snow Geese from Arviat (Eskimo Point) and a collection of eggs from Digges Island (see Tables 1 to 4). Collections from Labrador started in 1992. To date, a total of 6 collections of birds and 2 collections of eggs have been received from 4 sites in northern Labrador. The Labrador Inuit Association has assured us that they will continue to arrange for collections to be made during 1993. This level of agreement worked well in the case of Makivik Corporation which managed during 1992 to almost double the number of collections made during the first year (1991) of collections in northern Québec.

All tissue preparation was carried out at the National Wildlife Research Centre, Hull, Québec. Pooled samples of egg contents and breast muscle are being analyzed for organochlorines under contract by the Great Lakes Institute, University of Windsor. Mercury, cadmium and lead are being analyzed by the National Wildlife Research Centre laboratories, and selenium and arsenic

are being analyzed under contract by Fenwick Laboratories Limited, Halifax, N.S. All contract laboratories meet the quality assurance standards set by the Laboratory Services Section at the National Wildlife Research Centre. Where adequate tissue was available, samples of breast muscle were sent to the Bureau of Radiation and Medical Devices, Health and Welfare Canada, for radiocesium analyses. As well, the gastro-intestinal tracts are being sent to the University of Québec at Rimouski as part of a waterfowl feeding study, morphometric information is being sent to the University of Québec for a second study on morphology of diving birds, and samples are being sent to the Norwegian Institute for Nature Research in support of an international project on metals in Willow Ptarmigan.

We do not yet have any results for the metal analyses for 1992 samples. Highest metal levels measured in 1991 eggs were as follows: arsenic - Common Eider eggs from Kangiqsualuujjuaq (0.27 mg/kg ww); lead - all values below detection limits; selenium - Thick-billed Murre eggs from Digges Island (0.70 - 0.84 mg/kg ww); mercury - Black Guillemot eggs from Kangiqsualuujjuaq (0.34 - 0.68 mg/kg ww) (Table 3).

The highest levels measured in breast muscle samples to date are as follows: cadmium - King Eider from Holman Island and Thick-billed Murres from northern Québec (0.40 - 0.55 mg/kg ww); arsenic - Herring Gulls and Thick-billed Murres from northern Quebec (1.52 and 1.22 mg/kg ww, respectively); selenium - King Eiders from Holman Island (1.10 - 1.90 mg/kg ww) and Common Eiders from northern Quebec (1.22 mg/kg ww); mercury - Red-breasted Mergansers and Common Loons from northern Québec (1.23 and 1.04 mg/kg ww, respectively) (Table 4). It is interesting that the higher levels of cadmium and selenium found in breast muscle occurred in diving species such as eiders and murres associated with northern marine waters. Those levels were the highest recorded in this survey to date in any species in Canada, north or south of 60°N. Lead levels will have to be scrutinized carefully since most of the birds were collected using lead shot. The measured lead levels may, therefore, represent fragments of lead shot in the tissue in addition to biologically incorporated lead.

Of the organochlorine results received to date for eggs (Table 1), *trans*-chlordane was not detected (<0.001 mg/kg ww) in any of the samples. The highest levels of DDE, mirex, oxychlordane and PCBs (as Aroclor 1254:1260, 1:1) were found in Herring Gull eggs from Kangiqsualuujjuaq in northern Québec (0.40-1.52, 0.003-0.027, 0.040-0.058, 3.3-10.1 mg/kg ww, respectively) whereas the highest hexachlorobenzene level was found in Thick-billed Murre eggs from Digges Island (0.058-0.077 mg/kg ww). The values for Herring Gull eggs are similar to recent levels found in eggs from some of the colonies in the Great Lakes, and the levels found in the murre eggs are similar to levels found in Thick-billed Murre eggs collected from Prince Leopold Island in 1988. Only DDE was found to any extent in Canada Goose eggs collected from Labrador (0.034 mg/kg ww) in 1992 and that value is comparable to the DDE level found in Canada Goose eggs collected from the George River area of northern Québec. The contaminant profile found in the Common Eider eggs collected from Labrador in 1992 is also similar to that found in Common Eider eggs collected from the George River area in 1991 and 1992 (Table 1).

Among the samples of breast muscle, the highest levels of DDE and PCBs (as Aroclor 1254:1260) were found in Red-breasted Merganser from Kangiqsualuujjuaq (0.943 mg DDE/kg,

3.44 mg PCB/kg), Herring Gulls from Kuujjuaraapik (0.687 mg DDE/kg, 6.28 mg PCB/kg), Common Loons from Inukjuaq (0.453 mg DDE/kg, 3.25 mg PCB/kg), and Glaucous Gulls from Inukjuaq (0.345 mg DDE/kg, 17.29 mg PCB/kg) and Kuujjuarapik (3.0 mg DDE/kg, 6.1 mg PCB/kg) (Table 2). Levels in Common Loons and Red-breasted Mergansers are similar to levels found in breast muscle of those species collected from northern Ontario lakes during 1988-90.

Results for the 1992 collections from Labrador to date show no detectable levels (<0.001 mg/kg ww) of tetra- and penta-chlorobenzenes, γ-HCH, and *trans*- and *cis*-chlordanes in breast muscle (Table 2). Of the 6 species from Labrador for which results are available, the Black-legged Kittiwake had the most significant contaminant profile (Table 2).

UTILIZATION OF RESULTS

All chemical data from the birds collected from the Nunavik Region in 1992 as well as the collections from Labrador will be submitted to Health and Welfare Canada in fall 1993 for evaluation of risk to human health of eating those birds and eggs. Recommendations from Health and Welfare Canada for the 1988-90 dataset submitted in 1991 will be released in early fall 1993. Results from the 1993-94 survey being conducted in the Northwest Territories should serve to greatly expand the residue database particularly with respect to geographical representation and allow for some interesting inter-specific and geographical comparisons to be made. These data are being submitted on an annual basis to the Arctic Contaminants Database managed by the Département de Santé Communautaire du Centre Hospitalier de l'Université Laval. As well, the data will be submitted as part of the Canadian database to the international Arctic Monitoring and Assessment Program.

Expected project completion date: 1995

TABLE 1. LEVELS OF ORGANOCHLORINES IN EGGS OF WATERFOWL FROM NORTHERN CANADA - 1988-92

Wet Wt. (mg/kg or ppm)

I.D.	Tissue	Species	Age	Sex	N	Date mo yr	Latitude	Longitude	Prov	Location
55773*	EGG	Canada Goose	---	---	4	6 92	5628	06130	Labr	SE Nain
55778*	EGG	Common Eider	---	---	4	7 92	5627	06144	Labr	SE Nain
49494	EGG+	Thick-billed Murre	---	---	5	7 91	6235	07750	NWT	Digges Island
49492	EGG+	Thick-billed Murre	---	---	1	7 91	6235	07750	NWT	Digges Island
49489	EGG+	Thick-billed Murre	---	---	1	7 91	6235	07750	NWT	Digges Island
49493	EGG+	Thick-billed Murre	---	---	1	7 91	6235	07750	NWT	Digges Island
49491	EGG+	Thick-billed Murre	---	---	1	7 91	6235	07750	NWT	Digges Island
49490	EGG+	Thick-billed Murre	---	---	1	7 91	6235	07750	NWT	Digges Island
50096	EGG+	Black Guillemot	---	---	5	7 91	5842	06557	Que	Kangiqsualuujjuaq (George River)
50094	EGG+	Black Guillemot	---	---	1	7 91	5842	06557	Que	Kangiqsualuujjuaq (George River)
50091	EGG+	Black Guillemot	---	---	1	7 91	5842	06557	Que	Kangiqsualuujjuaq (George River)
50092	EGG+	Black Guillemot	---	---	1	7 91	5842	06557	Que	Kangiqsualuujjuaq (George River)
50093	EGG+	Black Guillemot	---	---	1	7 91	5842	06557	Que	Kangiqsualuujjuaq (George River)
50095	EGG+	Black Guillemot	---	---	1	7 91	5842	06557	Que	Kangiqsualuujjuaq (George River)
50102	EGG	Canada Goose	---	---	1	7 91	5842	06557	Que	Kangiqsualuujjuaq (George River)
57352**	EGG	Canada Goose	---	---	4	Spr 92	4921	06759	Que	George River
50107	EGG	Common Eider	---	---	4	7 91	5842	06557	Que	Kangiqsualuujjuaq (George River)
57353**	EGG	Common Eider	---	---	1	Spr 92	4921	06759	Que	George River
50113	EGG+	Herring Gull	---	---	5	6 91	5842	06557	Que	Kangiqsualuujjuaq (George River)
50108	EGG+	Herring Gull	---	---	1	6 91	5842	06557	Que	Kangiqsualuujjuaq (George River)
50109	EGG+	Herring Gull	---	---	1	6 91	5842	06557	Que	Kangiqsualuujjuaq (George River)
50110	EGG+	Herring Gull	---	---	1	6 91	5842	06557	Que	Kangiqsualuujjuaq (George River)
50111	EGG+	Herring Gull	---	---	1	6 91	5842	06557	Que	Kangiqsualuujjuaq (George River)
50112	EGG+	Herring Gull	---	---	1	6 91	5842	06557	Que	Kangiqsualuujjuaq (George River)

ND < 0.001 mg/kg wet wt.

Im - Immature; Ad - Adult; F - Female; M - Male; U - Unknown

* average of 2 replicates; where there was an "ND" value and a positive value, the positive value was used instead of an average.

** Dieldrin results unreliable due to analytical problems

\+ The following are individual analyses of pools: EGG - 50108-50112 = 50113; 50091-50095 = 50096; 49489-49493 = 49494;

Wet Wt. (mg/kg or ppm)

I.D.	Tissue	Species	% Lipid	% Water	1,2,4,5 -T4CB	1,2,3,4 -T4CB	QCB	HCB	a-HCH	b-HCH	g-HCH	OCS	oxy- chlordane	trans- chlordane	cis- chlordane	trans- nonachlor	cis- nonachlor
55773*	EGG	Canada Goose	14.64	70.5	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
55778*	EGG	Common Eider	18.78	64.1	0.001	ND	0.001	0.009	ND	0.005	0.002	ND	0.006	ND	ND	0.003	0.001
49494	EGG+	Thick-billed Murre	15.06	72.0	0.002	0.002	0.004	0.084	0.004	0.005	ND	0.009	0.026	ND	0.001	0.004	0.007
49492	EGG+	Thick-billed Murre	14.63	71.5	0.001	ND	0.003	0.060	0.001	0.005	ND	0.004	0.026	ND	ND	0.001	0.003
49489	EGG+	Thick-billed Murre	12.93	73.3	0.002	0.003	0.004	0.058	0.007	0.006	0.001	0.007	0.031	ND	0.002	0.004	0.007
49493	EGG+	Thick-billed Murre	13.00	73.0	0.002	0.001	0.003	0.060	0.003	0.006	0.001	0.007	0.035	ND	0.002	0.005	0.008
49491	EGG+	Thick-billed Murre	16.54	71.9	0.002	0.002	0.003	0.059	0.010	0.005	0.001	0.007	0.028	ND	0.002	0.004	0.007
49490	EGG+	Thick-billed Murre	11.94	74.1	0.002	0.001	0.002	0.077	0.006	0.007	0.001	0.009	0.034	ND	0.002	0.005	0.010
50096	EGG+	Black Guillemot	11.44	73.8	ND	0.001	0.002	0.036	0.002	0.004	ND	0.005	0.020	ND	0.001	0.021	0.013
50094	EGG+	Black Guillemot	11.82	73.3	ND	0.001	0.002	0.032	0.002	0.006	0.001	0.005	0.025	ND	0.001	0.010	0.022
50091	EGG+	Black Guillemot	11.69	72.3	ND	ND	ND	0.017	0.002	0.004	ND	0.003	0.018	ND	ND	0.005	0.011
50092	EGG+	Black Guillemot	10.75	74.2	ND	ND	ND	0.016	0.003	0.004	ND	0.003	0.017	ND	ND	0.005	0.010
50093	EGG+	Black Guillemot	9.61	74.7	ND	ND	0.001	0.020	0.002	0.004	ND	0.003	0.023	ND	ND	0.006	0.013
50095	EGG+	Black Guillemot	11.44	73.7	ND	0.002	0.002	0.047	0.005	0.003	ND	0.006	0.022	ND	0.004	0.038	0.018
50102	EGG	Canada Goose	19.81	65.1	ND	ND	0.001	0.007	ND	0.002	ND	ND	0.010	ND	ND	0.007	ND
57352**	EGG	Canada Goose	17.26	70.0	ND	ND	ND	0.002	ND	ND	ND	ND	0.003	ND	ND	0.001	0.001
50107	EGG	Common Eider	18.50	67.0	ND	ND	0.001	0.007	ND	0.001	ND	ND	0.005	ND	ND	0.003	ND
57353**	EGG	Common Eider	20.65	66.0	ND	ND	0.001	0.009	ND	ND	ND	ND	0.006	ND	ND	0.005	ND
50113	EGG+	Herring Gull	10.19	75.9	ND	ND	0.002	0.029	ND	0.002	ND	0.004	0.048	ND	0.001	0.057	0.009
50108	EGG+	Herring Gull	11.56	75.8	ND	ND	0.002	0.051	ND	0.003	ND	0.005	0.040	ND	0.002	0.078	0.012
50109	EGG+	Herring Gull	9.70	76.0	ND	0.001	0.001	0.021	ND	0.001	ND	0.005	0.043	ND	0.006	0.045	0.011
50110	EGG+	Herring Gull	9.63	76.7	ND	0.001	0.001	0.025	ND	0.002	ND	0.005	0.045	ND	0.003	0.039	0.012
50111	EGG+	Herring Gull	9.39	75.8	0.002	0.001	0.002	0.020	ND	0.002	ND	0.003	0.040	ND	0.003	0.040	0.009
50112	EGG+	Herring Gull	10.27	76.4	ND	ND	0.001	0.028	ND	0.003	ND	0.004	0.058	ND	0.002	0.057	0.005

Wet Wt. (mg/kg or ppm)

I.D.	Tissue	Species	pp'-DDE	pp'-DDD	pp'-DDT	Photo-mirex	Mirex	HE	Dieldrin	Aroclor 1254:1260
55773*	EGG	Canada Goose	0.034	ND	0.006	ND	ND	0.001	ND	ND
55778*	EGG	Common Eider	0.015	ND	ND	0.002	ND	0.004	0.017	0.082
49494	EGG+	Thick-billed Murre	0.281	0.002	0.008	ND	0.006	0.007	0.014	0.393
49492	EGG+	Thick-billed Murre	0.152	0.001	0.003	ND	0.001	0.006	0.012	0.278
49489	EGG+	Thick-billed Murre	0.165	0.001	0.011	ND	0.002	0.012	0.030	0.394
49493	EGG+	Thick-billed Murre	0.232	0.002	0.011	ND	0.002	0.010	0.023	0.483
49491	EGG+	Thick-billed Murre	0.145	0.002	0.011	ND	0.002	0.011	0.024	0.372
49490	EGG+	Thick-billed Murre	0.249	0.002	0.015	0.002	0.003	0.010	0.020	0.269
50096	EGG+	Black Guillemot	0.086	ND	ND	ND	0.003	0.014	0.029	0.470
50094	EGG+	Black Guillemot	0.057	0.002	0.023	ND	0.002	0.018	0.018	0.260
50091	EGG+	Black Guillemot	0.041	ND	0.012	0.001	0.002	0.013	0.009	0.220
50092	EGG+	Black Guillemot	0.039	ND	0.013	0.001	0.002	0.012	0.009	0.214
50093	EGG+	Black Guillemot	0.035	ND	0.014	0.001	0.002	0.017	0.013	0.201
50095	EGG+	Black Guillemot	0.076	0.002	0.024	ND	0.003	0.027	0.022	0.340
50102	EGG	Canada Goose	0.017	ND	ND	ND	ND	0.005	0.014	0.085
57352**	EGG	Canada Goose	0.033	0.003	0.008	ND	ND	0.002	---	0.006
50107	EGG	Common Eider	0.011	ND	ND	ND	ND	0.005	0.009	0.059
57353**	EGG	Common Eider	0.010	ND	0.001	ND	ND	0.002	---	0.055
50113	EGG+	Herring Gull	0.931	0.001	ND	0.066	0.016	0.017	0.022	6.271
50108	EGG+	Herring Gull	0.689	ND	0.010	ND	0.008	0.028	0.020	6.137
50109	EGG+	Herring Gull	0.402	0.002	0.019	ND	0.003	0.018	0.047	3.279
50110	EGG+	Herring Gull	1.522	0.003	0.023	ND	0.027	0.023	0.019	8.616
50111	EGG+	Herring Gull	0.529	ND	0.009	ND	0.006	0.018	0.022	3.745
50112	EGG+	Herring Gull	1.360	ND	0.005	ND	0.027	0.029	0.013	10.067

TABLE 2. LEVELS OF ORGANOCHLORINES IN BREAST MUSCLE OF WATERFOWL FROM NORTHERN CANADA - 1988-92

Wet Wt. (mg/kg or ppm)

I.D.	Tissue	Species	Age	Sex	N	Date mo	Date yr	Latitude	Longitude	Prov	Location
57110**	BM	Black Duck	6Ad	???	6	9	92	5410	05830	Labr	Stagg I Brook/Snook's Cove
57080**	BM	Black-legged Kittiwake	8Ad	2F,6M	8	9	92	5410	05830	Labr	2 mi from Rigolet
55554	BM	Canada Goose	8Ad	2F,6M	8	6	92	5717	06154	Labr	Tasiujak Bay area
57091**	BM	Common Eider	8Ad	???	8	9	92	5410	05830	Labr	30 mi from Rigolet
57099**	BM	Common Merganser	2Ad,5Im	3F,4M	7	9	92	5410	05830	Labr	Stagg I Brook
57103**	BM	Green-winged Teal	3Ad	2F,1M	3	9	92	5410	05830	Labr	Snook's Cove, Rigolet
50165	BM	Black Guillemot	5Ad	5M	5	7/8	91	5827	07805	Que	Inukjuaq
50179	BM	Black Guillemot	5Ad	4F,1M	5	7	91	5842	06557	Que	Kangiqsualuujjuaq (George River)
51769	BM	Black Scoter	5Ad	5F	5	9	91	5827	07805	Que	Inukjuaq
55743	BM	Black Scoter	2Ad	2M	2	???	92	5525	07715	Que	Kuujjuarapik
51978	BM+	Canada Goose	2Im	2M	2	9	91	5827	07805	Que	Inukjuaq
51979	BM+	Canada Goose	2Ad	2F	2	9	91	5827	07805	Que	Inukjuaq
51810	BM+	Canada Goose	2Ad,2Im	2F,2M	4	9	91	5827	07805	Que	Inukjuaq
50258	BM	Canada Goose	4Ad,1Im	4F,1M	5	7	91	5842	06557	Que	Kangiqsualuujjuaq (George River)
56193	BM	Canada Goose	1Ad	1M	1	10	92	6136	07158	Que	Kangiqsujuaq
55671	BM	Canada Goose	???	???	3	8	92	5525	07715	Que	Kuujjuarapik
56200	BM	Canada Goose	6Ad	4F,2M	6	10	92	6214	07538	Que	Salluit
51805	BM	Canada Goose	1Im	1M	1	6-8	91	6214	07538	Que	Salluit
55497	BM	Canada Goose	???	???	1	Spr	92	6214	07538	Que	Salluit
51783	BM	Common Eider	3Ad	3F	3	9/10	91	5827	07805	Que	Inukjuaq
50172	BM	Common Eider	6Ad	5F,1M	6	7	91	5842	06557	Que	Kangiqsualuujjuaq (George River)
51789	BM	Common Eider	3Ad	1F,2M	3	6-8	91	6214	07538	Que	Salluit
51845	BM	Common Loon	3Im	1F,2U	3	10	91	5827	07805	Que	Inukjuaq
50252	BM	Common Loon	1Ad	1M	1	8	91	5827	07805	Que	Inukjuaq
50192	BM	Common Loon	3Ad	2F,1M	3	7	91	5842	06557	Que	Kangiqsualuujjuaq (George River)
56012*	BM	Common Loon	5Ad	4F,1M	5	5-7	92	5525	07715	Que	Kuujjuarapik
51848	BM	Common Loon	2Ad	1F,1M	2	6-8	91	6214	07538	Que	Salluit
56312	BM	Common Loon	???	???	3	10	92	6214	07538	Que	Salluit
51568	BM+	Glaucous Gull	1Im	1F	1	9	91	5827	07805	Que	Inukjuaq (garbage dump)
51569	BM+	Glaucous Gull	1Ad	1F	1	9	91	5827	07805	Que	Inukjuaq (garbage dump)
51570	BM+	Glaucous Gull	1Im	1F	1	9	91	5827	07805	Que	Inukjuaq (garbage dump)
51571	BM+	Glaucous Gull	1Ad,2Im	3F	3	9	91	5827	07805	Que	Inukjuaq (garbage dump)
55768*	BM	Glaucous Gull	???	???	2	7/8	92	5525	07715	Que	Kuujjuarapik
51574	BM+	Herring Gull	1Ad,1Im	2F	2	9	91	5827	07805	Que	Inukjuaq (garbage dump)
51572	BM+	Herring Gull	1Im	1F	1	9	91	5827	07805	Que	Inukjuaq (garbage dump)
51573	BM+	Herring Gull	1Ad	1F	1	9	91	5827	07805	Que	Inukjuaq (garbage dump)
50528	BM	Herring Gull	3Ad	2F,1M	3	8	91	5529	07731	Que	Kuujjuaraapik
55496	BM	Herring Gull	3Ad	1F,2M	3	Spr	92	6214	07538	Que	Salluit

Wet Wt. (mg/kg or ppm)

I.D.	Tissue	Species	Age	Sex	N	Date mo	Date yr	Latitude	Longitude	Prov	Location
51790	BM+	King Eider	1Ad,1Im	1F,1M	2	9/10	91	5827	07805	Que	Inukjuaq
51785	BM+	King Eider	1Ad	1F	1	10	91	5827	07805	Que	Inukjuaq
51780	BM+	King Eider	1Im	1M	1	9	91	5827	07805	Que	Inukjuaq
51784	BM	King Eider	1Ad	1M	1	6-8	91	6214	07538	Que	Salluit
51798	BM+	Lesser Snow Goose	1Ad,1Im	1F,1M	2	9	91	5827	07805	Que	Inukjuaq
51796	BM+	Lesser Snow Goose	1Ad	1F	1	9	91	5827	07805	Que	Inukjuaq
51797	BM+	Lesser Snow Goose	1Im	1M	1	9	91	5827	07805	Que	Inukjuaq
51795	BM	Lesser Snow Goose	4Ad	1F,3M	4	6	91	6214	07538	Que	Salluit
55334	BM	Snow Goose	5Ad	4F,1M	5	6	92	6224	07755	Que	Ivujivik
55672	BM	Snow Goose	???	???	1	8	92	6136	07158	Que	Kangiqsujuaq
55339	BM	Snow Goose	4Ad	2F,2M	4	6	92	6136	07158	Que	Kangiqsujuaq
55673	BM	Snow Goose	???	???	1	8	92	5529	07731	Que	Kuujjuarapik
56073	BM	Snow Goose	8Ad,1Im	3F,6M	9	10	92	6214	07538	Que	Salluit
51523	BM+	Oldsquaw	1Ad,4Im	4F,1M	5	10	91	5827	07805	Que	Inukjuaq
51521	BM+	Oldsquaw	1Ad	1F	1	10	91	5827	07805	Que	Inukjuaq
51980	BM+	Oldsquaw	4Im	3F,1M	4	10	91	5827	07805	Que	Inukjuaq
50357	BM	Oldsquaw	5Ad	3F,2M	5	7	91	5842	06557	Que	Kangiqsualuujjuaq (George River)
56232	BM	Oldsquaw	2Ad	3M	3	10	92	6214	07538	Que	Salluit
56479	BM	Red-breasted Merganser	2Im	2F	2	Summer	92	4921	06759	Que	George River
51804	BM+	Red-breasted Merganser	4Ad,1Im	4F,1M	5	9	91	5827	07805	Que	Inukjuaq
51981	BM+	Red-breasted Merganser	4Ad	4F	4	9	91	5827	07805	Que	Inukjuaq
51803	BM+	Red-breasted Merganser	1Im	1M	1	9	91	5827	07805	Que	Inukjuaq
50259	BM	Red-breasted Merganser	1Ad	1M	1	6	91	5842	06557	Que	Kangiqsualuujjuaq (George River)
55746	BM	Red-breasted Merganser	2Ad	2F	2	???	92	5529	07731	Que	Kuujjuarapik
50195	BM	Red-throated Loon	2Ad	2M	2	6/7	91	5842	06557	Que	Kangiqsualuujjuaq (George River)
55517	BM	Rock Ptarmiagn	4Ad	2F,2M	4	Spr	92	6214	07538	Que	Salluit
52145	BM	Rock Ptarmigan	2Ad,1Im	1F,2M	3	12	91	5842	06557	Que	George River
52147	BM	Rock Ptarmigan	1Ad	1F	1	8	91	5842	06557	Que	George River
55747	BM	Surf Scoter	???	???	1	???	92	5529	07731	Que	Kuujjuarapik
49500	BM	Thick-billed Murre	5Ad	3F,2M	5	7	91	6224	07755	Que	Ivujivik
51778	BM	Thick-billed Murre	4Ad	1F,3M	4	6-8	91	6214	07538	Que	Salluit
52146	BM	Willow Ptarmigan	1Im	1M	1	8	91	5842	06557	Que	George River
56233	BM	Willow Ptarmigan	1Ad	1M	1	10	92	6214	07538	Que	Salluit
49246	BM	Canada Goose	1Ad,1Im	1F,1M	2	5/6	91	6107	09403	NWT	Arviat (Eskimo Point)
46654	BM	Canada Goose	9Im,1U	9F,1M	10	Spr	90	5632	07914	NWT	Sanikiluaq
44791	BM	Canada Goose	10Ad	3F,6M,1U	10	5/6	90	6932	09332	NWT	Spence Bay

Wet Wt. (mg/kg or ppm)

I.D.	Tissue	Species	Age	Sex	N	Date mo	yr	Latitude	Longitude	Prov	Location
43280	BM	Common Eider	3Ad	3F	3	1	90	6510	06530	NWT	Cumberland Sound area
43279	BM	Common Eider	5Im	3F,2M	5	1	90	6510	06530	NWT	Cumberland Sound area
40634	BM	Common Eider	2Ad	2F	2	11	88	5624	07919	NWT	Sanikiluaq
40281	BM+	Common Eider	5Im,5Ad	1F,9M	10	11	88	5624	07919	NWT	Sanikiluaq
40870	BM+	Common Eider	5Ad	5M	5	11	88	5624	07919	NWT	Sanikiluaq
40871	BM+	Common Eider	5Im	1F,4M	5	11	88	5624	07919	NWT	Sanikiluaq
42669	BM+	King Eider	11Ad	3F,8M	11	5	89	7044	17744	NWT	Holman Island
43112	BM+	King Eider	8Ad	8M	8	5	89	7044	11744	NWT	Holman Island
43111	BM+	King Eider	3Ad	3F	3	5	89	7044	11744	NWT	Holman Island
49331	BM+	Lesser Snow Goose	1Im	1F	1	5/6	91	6107	09403	NWT	Arviat (Eskimo Point)
49333	BM+	Lesser Snow Goose	7Ad	7F	7	5/6	91	6107	09403	NWT	Arviat (Eskimo Point)
49332	BM+	Lesser Snow Goose	7Ad	7M	7	5/6	91	6107	09403	NWT	Arviat (Eskimo Point)
41793	BM+	Lesser Snow Goose	10Ad	4F,6M	10	6	89	6107	09403	NWT	Arviat (Eskimo Point)
43107	BM+	Lesser Snow Goose	4Ad	4F	4	6	89	6107	09403	NWT	Arviat (Eskimo Point)
43108	BM+	Lesser Snow Goose	6Ad	6M	6	6	89	6107	09403	NWT	Arviat (Eskimo Point)
42634	BM+	Lesser Snow Goose	6Im,4Ad	4F,5M,1U	10	5	89	6927	13302	NWT	Tuktoyaktuk
43109	BM+	Lesser Snow Goose	4Ad	2F,2M	4	5	89	6927	13302	NWT	Tuktoyaktuk
43110	BM+	Lesser Snow Goose	6Im	2F,3M,1U	6	5	89	6927	13302	NWT	Tuktoyaktuk
39916	BM	Mallard	7Im,1Ad,2U	6F,4M	10	9/10	88	6244	11544	NWT	Stagg River
41747	BM	Common Goldeneye	6U	3F,3M	6	10	89	6043	13503	Yukon	17 km SE Whitehorse
45882	BM	Lesser Scaup	8Ad	8M	8	5/6	90	6735	13950	Yukon	Old Crow
45979	BM	Lesser Scaup	10Ad	2F,8M	10	5	90	6043	13503	Yukon	Whitehorse
40417	BM*	Lesser Scaup	1Im	1F	1	9	88	6043	13503	Yukon	Yukon River (17 mi SE Whitehorse)
43276	BM	Mallard	8Ad	2F,6M	8	5	88	6111	13512	Yukon	Shallow Bay, Lake Laberge (N of Whitehorse)
40427	BM	Mallard	4Im,4Ad	7F,1M	8	9/10	88	6111	13512	Yukon	Yukon River/Lake Laberge (N of Whitehorse)
45942	BM	Northern Pintail	5Ad	5M	5	5/6	90	6735	13950	Yukon	Old Crow River
41859	BM*	Northern Pintail	1Im,11Ad	2F,10M	12	5	88	6111	13512	Yukon	Shallow Bay, Lake Laberge (N of Whitehorse)
45888	BM	Surf Scoter	5Ad	1F,4M	5	5/6	90	6735	13950	Yukon	Old Crow
45988	BM	Surf Scoter	4Ad	1F,3M	4	5	90	6043	13503	Yukon	Whitehorse
46019	BM	White-winged Scoter	7Ad	1F,6M	7	5/6	90	6725-35	13950-14100	Yukon	Old Crow/Porcupine River
46015	BM	White-winged Scoter	6Ad	6M	6	5	90	6043	13503	Yukon	Whitehorse

ND < 0.001 mg/kg wet wt. BM - Breast Muscle; Im - Immature; Ad - Adult; F - Female; M - Male; U - Unknown

* average of 2 replicates; where there was an "ND" value and a positive value, the positive value was used instead of an average.

** Dieldrin results unreliable due to analytical problems

\+ 51981 & 51803 are sub-pools of 51804; 40870 & 40871 are sub-pools of 40281; 43111 & 43112 are sub-pools of 42669; 43107 & 43108 are sub-pools of 41793; 43109 & 43110 are sub-pools of 42634; 51978 & 51979 are sub-pools of 51810; 51521 & 51980 are sub-pools of 51523; 51981 & 51803 are sub-pools of 51804.

\+ The following are individual analyses of pools: 51796 & 51797 = 51798; 51780 & 51785 = 51790; 51572 & 51573 = 51574; 51568-51570 = 51571.

\+ The following are related pools: Lesser Snow Geese - 49331, 49332 & 49333.

Wet Wt. (mg/kg or ppm)

I.D.	Tissue	Species	% Lipid	% Water	1,2,4,5 -T4CB	1,2,3,4 -T4CB	QCB	HCB	a-HCH	b-HCH	g-HCH	OCS	oxy- chlordane	trans- chlordane	cis- chlordane	trans- nonachlor	cis- nonachlor
57110**	BM	Black Duck	3.28	71.4	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
57080**	BM	Black-legged Kittiwake	7.59	69.4	ND	ND	ND	0.013	0.001	0.001	ND	0.002	0.014	ND	ND	0.005	0.002
55554	BM	Canada Goose	3.15	71.5	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
57091**	BM	Common Eider	2.20	77.5	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
57099**	BM	Common Merganser	2.67	73.7	ND	ND	ND	0.002	ND	ND	ND	ND	0.002	ND	ND	ND	ND
57103**	BM	Green-winged Teal	3.00	72.3	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
50165	BM	Black Guillemot	2.81	71.0	ND	ND	ND	0.016	0.001	ND	ND	0.003	0.006	ND	ND	0.004	0.006
50179	BM	Black Guillemot	2.61	70.8	ND	ND	ND	0.019	0.001	0.001	ND	0.003	0.008	ND	ND	0.005	0.005
51769	BM	Black Scoter	5.25	72.0	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
55743	BM	Black Scoter	3.71	71.7	ND	ND	ND	0.002	ND	ND	ND	ND	0.003	ND	ND	ND	ND
51978	BM+	Canada Goose	2.99	74.0	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
51979	BM+	Canada Goose	3.77	73.2	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
51810	BM+	Canada Goose	4.05	72.9	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
50258	BM	Canada Goose	3.10	71.3	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
56193	BM	Canada Goose	2.06	73.6	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
55671	BM	Canada Goose	2.36	72.6	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
56200	BM	Canada Goose	3.71	71.8	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
51805	BM	Canada Goose	3.75	73.9	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
55497	BM	Canada Goose	4.63	71.4	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
51783	BM	Common Eider	2.04	77.2	ND	ND	ND	0.001	ND	ND	ND	ND	ND	ND	ND	ND	ND
50172	BM	Common Eider	2.62	72.0	ND	ND	ND	0.003	ND	ND	ND	ND	0.002	ND	ND	ND	ND
51789	BM	Common Eider	2.52	70.7	ND	ND	ND	0.005	0.001	ND	ND	ND	0.003	ND	ND	ND	ND
51845	BM	Common Loon	6.07	69.9	ND	ND	ND	0.018	ND	ND	ND	ND	ND	ND	ND	0.001	ND
50252	BM	Common Loon	3.67	70.1	ND	ND	0.001	0.016	ND	ND	ND	0.003	0.019	ND	0.002	0.057	0.019
50192	BM	Common Loon	3.22	71.0	ND	ND	ND	0.008	ND	ND	ND	0.002	0.006	ND	ND	0.019	0.005
56012*	BM	Common Loon	4.29	70.1	ND	ND	ND	0.006	0.001	ND	ND	0.000	0.005	ND	0.002	0.020	0.005
51848	BM	Common Loon	5.87	66.4	ND	ND	ND	0.119	ND	0.001	ND	0.002	0.008	ND	0.002	0.009	0.003
56312	BM	Common Loon	3.38	70.5	ND	ND	ND	0.013	ND	ND	ND	ND	0.010	ND	ND	0.008	0.002
51568	BM+	Glaucous Gull	6.07	71.1	ND	0.003	0.002	0.042	ND	0.003	ND	0.006	0.159	ND	0.001	0.054	0.012
51569	BM+	Glaucous Gull	7.66	69.3	ND	ND	ND	0.013	ND	0.001	ND	0.003	0.055	ND	ND	0.004	ND
51570	BM+	Glaucous Gull	5.01	72.0	0.002	ND	0.001	0.014	0.002	0.003	ND	0.003	0.163	ND	ND	0.028	0.004
51571	BM+	Glaucous Gull	6.61	70.6	ND	ND	0.001	0.023	ND	0.003	ND	0.003	0.117	ND	ND	0.023	0.003
55768*	BM	Glaucous Gull	4.91	71.2	0.002	0.002	0.004	0.119	0.010	0.001	0.010	ND	0.341	0.004	0.002	0.146	0.026
51574	BM+	Herring Gull	6.17	70.4	ND	ND	ND	0.015	ND	0.002	ND	0.002	0.066	ND	ND	0.011	0.002
51572	BM+	Herring Gull	4.70	72.0	ND	ND	ND	0.021	ND	ND	ND	0.021	0.042	ND	ND	0.015	0.003
51573	BM+	Herring Gull	6.11	70.4	ND	ND	ND	0.010	ND	0.002	ND	0.002	0.105	ND	ND	0.006	ND
50528	BM	Herring Gull	2.88	74.0	ND	ND	0.001	0.026	ND	0.002	ND	0.002	0.083	ND	ND	0.011	0.003
55496	BM	Herring Gull	7.07	71.3	0.002	ND	0.001	0.015	0.003	0.002	0.003	ND	0.077	ND	ND	0.017	0.003

Wet Wt. (mg/kg or ppm)

I.D.	Tissue	Species	% Lipid	% Water	1,2,4,5 -T4CB	1,2,3,4 -T4CB	QCB	HCB	a-HCH	b-HCH	g-HCH	OCS	oxy- chlordane	trans- chlordane	cis- chlordane	trans- nonachlor	cis- nonachlor
51790	BM+	King Eider	3.90	72.7	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
51785	BM+	King Eider	3.41	74.0	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
51780	BM+	King Eider	3.59	72.1	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
51784	BM	King Eider	1.85	72.4	ND	ND	0.001	0.014	ND	0.003	ND	ND	0.007	ND	ND	0.002	ND
51798	BM+	Lesser Snow Goose	3.76	72.6	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
51796	BM+	Lesser Snow Goose	4.17	73.1	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
51797	BM+	Lesser Snow Goose	3.62	72.5	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
51795	BM	Lesser Snow Goose	3.45	71.4	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
55334	BM	Snow Goose	3.41	72.3	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
55672	BM	Snow Goose	2.59	73.5	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
55339	BM	Snow Goose	3.15	71.5	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
55673	BM	Snow Goose	5.10	70.7	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
56073	BM	Snow Goose	3.92	72.2	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
51523	BM+	Oldsquaw	3.03	72.6	ND	ND	ND	0.001	0.001	ND	ND	ND	ND	ND	ND	ND	ND
51521	BM+	Oldsquaw	3.04	75.1	ND	ND	ND	0.003	0.002	0.001	ND	ND	0.003	ND	ND	0.001	ND
51980	BM+	Oldsquaw	2.79	73.4	ND	ND	ND	ND	0.002	ND	ND	ND	ND	ND	ND	ND	ND
50357	BM	Oldsquaw	3.32	72.0	ND	ND	ND	0.006	0.006	ND	ND	ND	0.009	ND	ND	0.003	ND
56232	BM	Oldsquaw	3.44	71.7	ND	ND	ND	0.005	ND	0.010	0.001	ND	0.007	ND	ND	0.001	ND
56479	BM	Red-breasted Merganser	2.15	76.8	ND	ND	ND	0.001	ND	ND	ND	ND	ND	ND	ND	ND	ND
51804	BM+	Red-breasted Merganser	4.03	71.8	ND	ND	ND	0.002	ND	ND	ND	ND	0.001	ND	ND	ND	ND
51981	BM+	Red-breasted Merganser	3.73	71.9	ND	ND	ND	0.001	ND	ND	ND	ND	0.001	ND	ND	ND	ND
51803	BM+	Red-breasted Merganser	3.91	72.1	ND	ND	ND	0.002	ND	ND	ND	ND	0.001	ND	ND	ND	ND
50259	BM	Red-breasted Merganser	3.52	71.0	ND	ND	ND	0.011	ND	ND	ND	0.003	0.025	ND	ND	ND	ND
55746	BM	Red-breasted Merganser	4.08	70.7	ND	ND	0.001	0.010	0.005	0.001	0.001	ND	0.015	ND	ND	0.005	0.005
50195	BM	Red-throated Loon	6.34	69.3	ND	ND	0.001	0.014	ND	ND	ND	0.003	0.012	ND	0.001	0.050	0.008
55517	BM	Rock Ptarmiagn	3.11	71.5	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
52145	BM	Rock Ptarmigan	2.87	70.2	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
52147	BM	Rock Ptarmigan	2.16	71.7	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
55747	BM	Surf Scoter	2.77	72.4	ND	ND	ND	0.004	ND	ND	ND	ND	0.004	ND	ND	ND	ND
49500	BM	Thick-billed Murre	2.96	71.0	ND	ND	0.001	0.023	ND	0.001	ND	0.002	0.007	ND	ND	ND	ND
51778	BM	Thick-billed Murre	3.29	69.7	ND	ND	ND	0.016	0.001	0.001	ND	0.002	0.007	ND	ND	ND	ND
52146	BM	Willow Ptarmigan	1.95	71.8	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
56233	BM	Willow Ptarmigan	2.80	72.0	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
49246	BM	Canada Goose	3.00	72.7	ND	ND	ND	ND	ND	ND	ND	ND	0.001	ND	ND	ND	ND
46654	BM	Canada Goose	1.55	77.2	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
44791	BM	Canada Goose	4.93	68.7	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND

Wet Wt. (mg/kg or ppm)

I.D.	Tissue	Species	% Lipid	% Water	1,2,4,5 -T4CB	1,2,3,4 -T4CB	QCB	HCB	a-HCH	b-HCH	g-HCH	OCS	oxy- chlordane	trans- chlordane	cis- chlordane	trans- nonachlor	cis- nonachlor
43280	BM	Common Eider	3.28	72.0	ND	ND	ND	0.003	ND	ND	ND	ND	ND	ND	ND	0.001	ND
43279	BM	Common Eider	3.47	71.2	ND	ND	ND	0.005	ND	ND	ND	ND	ND	ND	0.001	0.001	0.001
40634	BM	Common Eider	3.31	71.7	ND	ND	ND	0.006	ND	ND	ND	ND	0.005	ND	ND	0.003	ND
40281	BM+	Common Eider	2.66	69.8	ND	ND	ND	0.002	ND	ND	ND	ND	0.002	ND	ND	0.001	ND
40870	BM+	Common Eider	3.35	70.0	ND	ND	ND	0.002	ND	ND	ND	ND	0.002	ND	ND	0.001	ND
40871	BM+	Common Eider	3.07	70.8	ND	ND	ND	0.001	ND	ND	ND	ND	ND	ND	ND	0.001	ND
42669	BM+	King Eider	4.36	68.9	ND	ND	ND	0.003	ND	0.001	ND	ND	0.002	ND	0.001	ND	ND
43112	BM+	King Eider	4.30	68.6	ND	ND	ND	0.003	ND	0.001	ND	ND	0.001	ND	ND	ND	ND
43111	BM+	King Eider	4.32	68.7	ND	ND	ND	0.002	ND	0.001	ND	ND	0.002	ND	ND	0.001	ND
49331	BM+	Lesser Snow Goose	5.09	68.5	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
49333	BM+	Lesser Snow Goose	5.25	68.8	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
49332	BM+	Lesser Snow Goose	5.16	68.5	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
41793	BM+	Lesser Snow Goose	4.02	71.4	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	0.001	ND	ND
43107	BM+	Lesser Snow Goose	4.58	70.8	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	0.001	ND	ND
43108	BM+	Lesser Snow Goose	3.86	71.3	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
42634	BM+	Lesser Snow Goose	4.05	70.8	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	0.002	ND	ND
43109	BM+	Lesser Snow Goose	4.12	70.7	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
43110	BM+	Lesser Snow Goose	4.36	69.7	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	0.001	ND	ND
39916	BM	Mallard	1.94	71.6	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
41747	BM	Common Goldeneye	3.69	70.3	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
45882	BM	Lesser Scaup	3.14	70.9	ND	ND	ND	0.002	ND	ND	ND	ND	0.002	ND	ND	ND	ND
45979	BM	Lesser Scaup	3.78	70.6	ND	ND	ND	0.002	ND	ND	ND	ND	0.001	ND	ND	ND	ND
40417	BM*	Lesser Scaup	2.44	70.5	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
43276	BM	Mallard	2.69	70.0	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
40427	BM	Mallard	2.08	70.4	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	0.001	ND	ND
45942	BM	Northern Pintail	2.03	72.2	ND	ND	ND	0.002	ND	ND	ND	ND	0.003	ND	ND	ND	ND
41859	BM*	Northern Pintail	4.31	69.1	ND	ND	ND	0.011	ND	ND	ND	ND	ND	ND	ND	ND	ND
45888	BM	Surf Scoter	3.02	70.8	ND	ND	ND	0.003	ND	0.001	ND	ND	ND	ND	ND	ND	ND
45988	BM	Surf Scoter	4.77	70.9	ND	ND	ND	0.005	ND	0.001	ND	ND	0.001	ND	ND	ND	ND
46019	BM	White-winged Scoter	5.04	69.4	ND	ND	ND	0.003	ND	0.001	ND	ND	ND	ND	ND	0.001	ND
46015	BM	White-winged Scoter	4.49	69.2	ND	ND	ND	0.002	ND	0.001	ND	ND	ND	ND	ND	ND	ND

Wet Wt. (mg/kg or ppm)

I.D.	Tissue	Species	pp'-DDE	pp'-DDD	pp'-DDT	Photo-mirex	Mirex	HE	Dieldrin	Aroclor 1254:1260
57110**	BM	Black Duck	0.002	ND	ND	ND	ND	ND	---	0.004
57080**	BM	Black-legged Kittiwake	0.037	ND	0.010	0.003	0.005	0.004	---	0.924
55554	BM	Canada Goose	0.003	0.002	ND	ND	ND	ND	0.001	0.003
57091**	BM	Common Eider	ND	ND	ND	ND	ND	ND	---	0.004
57099**	BM	Common Merganser	0.011	ND	ND	ND	ND	ND	---	0.094
57103**	BM	Green-winged Teal	0.001	ND	ND	ND	ND	ND	---	0.004
50165	BM	Black Guillemot	0.029	ND	0.006	ND	0.001	0.004	0.003	0.112
50179	BM	Black Guillemot	0.022	ND	0.007	ND	ND	0.005	0.005	0.095
51769	BM	Black Scoter	ND	ND	ND	ND	ND	ND	ND	0.003
55743	BM	Black Scoter	0.066	0.001	0.002	0.004	0.001	0.003	ND	0.315
51978	BM+	Canada Goose	ND	ND	ND	ND	ND	ND	ND	ND
51979	BM+	Canada Goose	ND	ND	ND	ND	ND	ND	ND	ND
51810	BM+	Canada Goose	ND	ND	ND	ND	ND	ND	ND	ND
50258	BM	Canada Goose	0.004	ND	ND	ND	ND	ND	ND	0.004
56193	BM	Canada Goose	ND	ND	ND	ND	ND	ND	ND	ND
55671	BM	Canada Goose	0.001	ND	ND	ND	ND	ND	ND	0.002
56200	BM	Canada Goose	ND	ND	ND	ND	ND	ND	ND	ND
51805	BM	Canada Goose	ND	ND	ND	ND	ND	ND	ND	ND
55497	BM	Canada Goose	0.001	ND	ND	ND	ND	ND	ND	0.001
51783	BM	Common Eider	ND	ND	ND	ND	ND	ND	ND	0.003
50172	BM	Common Eider	0.006	ND	0.003	ND	ND	ND	ND	0.030
51789	BM	Common Eider	0.005	ND	0.001	ND	ND	0.001	ND	0.019
51845	BM	Common Loon	0.002	ND	ND	ND	ND	ND	0.002	0.020
50252	BM	Common Loon	0.453	0.006	0.005	0.032	0.004	0.007	0.033	3.254
50192	BM	Common Loon	0.241	0.005	0.003	0.018	0.029	0.003	0.020	1.144
56012*	BM	Common Loon	0.139	0.004	0.009	0.011	0.003	0.005	0.013	0.966
51848	BM	Common Loon	0.069	0.002	0.002	ND	0.001	0.007	0.055	0.810
56312	BM	Common Loon	0.035	ND	0.001	ND	0.003	0.003	0.006	0.237
51568	BM+	Glaucous Gull	0.345	ND	0.038	ND	0.010	0.045	0.158	17.289
51569	BM+	Glaucous Gull	0.605	ND	0.003	ND	0.011	0.008	0.005	1.865
51570	BM+	Glaucous Gull	0.497	0.006	0.014	ND	0.008	0.027	0.017	1.511
51571	BM+	Glaucous Gull	0.670	0.002	0.004	0.016	0.009	0.002	0.004	1.748
55768*	BM	Glaucous Gull	2.966	0.023	0.055	0.054	0.062	0.084	0.003	6.066
51574	BM+	Herring Gull	0.432	ND	0.002	0.024	0.001	0.013	0.008	2.425
51572	BM+	Herring Gull	0.366	0.002	0.010	ND	0.004	0.010	0.008	1.367
51573	BM+	Herring Gull	0.608	ND	0.002	ND	0.018	0.020	0.006	2.848
50528	BM	Herring Gull	0.687	0.002	0.009	0.061	0.023	0.015	0.011	6.281
55496	BM	Herring Gull	0.722	0.004	0.040	0.034	0.015	0.013	0.046	3.052

Wet Wt. (mg/kg or ppm)

I.D.	Tissue	Species	pp'-DDE	pp'-DDD	pp'-DDT	Photo-mirex	Mirex	HE	Dieldrin	Aroclor 1254:1260
51790	BM+	King Eider	ND	ND	ND	ND	ND	ND	0.001	0.002
51785	BM+	King Eider	ND	ND	ND	ND	ND	0.001	ND	0.002
51780	BM+	King Eider	ND	ND	ND	ND	ND	ND	ND	0.003
51784	BM	King Eider	0.014	ND	0.003	0.001	ND	0.002	ND	0.053
51798	BM+	Lesser Snow Goose	0.001	ND	ND	ND	ND	ND	ND	ND
51796	BM+	Lesser Snow Goose	ND	ND	ND	ND	ND	ND	ND	ND
51797	BM+	Lesser Snow Goose	ND	ND	ND	ND	ND	ND	ND	ND
51795	BM	Lesser Snow Goose	ND	ND	ND	ND	ND	ND	ND	ND
55334	BM	Snow Goose	0.002	ND	ND	ND	ND	ND	0.002	0.002
55672	BM	Snow Goose	0.004	0.004	ND	ND	ND	ND	0.001	0.002
55339	BM	Snow Goose	0.005	ND	0.002	ND	ND	ND	0.003	ND
55673	BM	Snow Goose	0.004	ND	ND	ND	ND	ND	0.001	ND
56073	BM	Snow Goose	0.005	ND	ND	ND	ND	ND	ND	0.001
51523	BM+	Oldsquaw	0.005	ND	ND	ND	ND	ND	0.001	0.003
51521	BM+	Oldsquaw	0.018	ND	0.001	ND	ND	0.004	0.003	0.105
51980	BM+	Oldsquaw	ND	ND	ND	ND	ND	ND	ND	0.006
50357	BM	Oldsquaw	0.048	ND	ND	0.004	0.001	ND	0.006	0.225
56232	BM	Oldsquaw	0.091	ND	0.001	ND	0.002	0.005	ND	0.227
56479	BM	Red-breasted Merganser	0.003	ND	ND	ND	ND	ND	ND	0.025
51804	BM+	Red-breasted Merganser	0.006	ND	ND	ND	ND	ND	ND	0.032
51981	BM+	Red-breasted Merganser	0.006	ND	0.001	ND	ND	0.002	0.002	0.028
51803	BM+	Red-breasted Merganser	0.004	ND	ND	ND	ND	ND	ND	0.021
50259	BM	Red-breasted Merganser	0.943	0.002	0.005	0.083	0.185	0.003	0.003	3.439
55746	BM	Red-breasted Merganser	0.215	0.019	0.017	0.019	0.034	0.012	0.048	0.858
50195	BM	Red-throated Loon	0.287	0.005	0.006	0.022	0.003	0.007	0.031	1.875
55517	BM	Rock Ptarmiagn	ND	ND	ND	ND	ND	ND	ND	0.003
52145	BM	Rock Ptarmigan	ND	ND	ND	ND	ND	ND	ND	0.002
52147	BM	Rock Ptarmigan	ND	ND	ND	ND	ND	ND	ND	0.005
55747	BM	Surf Scoter	0.063	ND	0.001	0.005	0.003	0.002	0.002	0.258
49500	BM	Thick-billed Murre	0.080	ND	ND	ND	ND	ND	0.002	0.138
51778	BM	Thick-billed Murre	0.040	ND	ND	ND	ND	0.001	0.003	0.070
52146	BM	Willow Ptarmigan	ND	ND	ND	ND	ND	ND	ND	0.006
56233	BM	Willow Ptarmigan	ND	ND	ND	ND	ND	ND	ND	ND
49246	BM	Canada Goose	0.001	ND	ND	ND	ND	0.001	ND	0.004
46654	BM	Canada Goose	ND	ND	ND	ND	ND	ND	ND	ND
44791	BM	Canada Goose	ND	ND	ND	ND	ND	0.003	ND	0.001

Wet Wt. (mg/kg or ppm)

I.D.	Tissue	Species	pp'-DDE	pp'-DDD	pp'-DDT	Photo-mirex	Mirex	HE	Dieldrin	Aroclor 1254:1260
43280	BM	Common Eider	0.003	ND	ND	ND	ND	ND	0.002	0.008
43279	BM	Common Eider	0.001	ND	ND	ND	ND	ND	ND	0.008
40634	BM	Common Eider	0.017	ND	ND	ND	0.001	0.003	0.004	0.047
40281	BM+	Common Eider	0.005	ND	ND	ND	ND	ND	0.003	0.013
40870	BM+	Common Eider	0.005	ND	ND	ND	ND	ND	0.001	0.014
40871	BM+	Common Eider	0.003	ND	ND	ND	ND	ND	0.002	0.009
42669	BM+	King Eider	0.003	ND	ND	ND	ND	ND	0.001	0.010
43112	BM+	King Eider	0.003	ND	ND	ND	ND	ND	0.001	0.010
43111	BM+	King Eider	0.004	ND	ND	ND	ND	ND	0.001	0.007
49331	BM+	Lesser Snow Goose	0.002	ND	ND	ND	ND	ND	ND	0.002
49333	BM+	Lesser Snow Goose	ND	ND	ND	ND	ND	ND	0.001	0.002
49332	BM+	Lesser Snow Goose	0.001	ND	ND	ND	ND	ND	0.001	0.002
41793	BM+	Lesser Snow Goose	0.001	ND	ND	ND	ND	ND	0.001	0.002
43107	BM+	Lesser Snow Goose	ND	ND	ND	ND	ND	ND	0.001	ND
43108	BM+	Lesser Snow Goose	0.001	ND	ND	ND	ND	0.001	0.001	ND
42634	BM+	Lesser Snow Goose	0.001	ND	ND	ND	ND	0.001	ND	ND
43109	BM+	Lesser Snow Goose	ND	ND	ND	ND	ND	0.001	ND	ND
43110	BM+	Lesser Snow Goose	0.001	ND	ND	ND	ND	0.003	0.001	ND
39916	BM	Mallard	ND	ND	ND	ND	ND	ND	ND	ND
41747	BM	Common Goldeneye	0.004	ND	ND	ND	ND	ND	ND	0.003
45882	BM	Lesser Scaup	0.051	0.002	ND	ND	ND	0.001	ND	0.151
45979	BM	Lesser Scaup	0.058	0.001	ND	ND	ND	ND	ND	0.119
40417	BM*	Lesser Scaup	0.091	ND	ND	ND	ND	ND	ND	0.003
43276	BM	Mallard	0.012	ND	ND	ND	ND	0.004	0.006	0.006
40427	BM	Mallard	0.001	ND	ND	ND	ND	ND	ND	ND
45942	BM	Northern Pintail	0.068	0.002	ND	ND	ND	0.001	ND	0.129
41859	BM*	Northern Pintail	0.035	0.002	ND	ND	ND	0.001	0.001	0.003
45888	BM	Surf Scoter	0.005	ND	ND	ND	ND	ND	ND	0.017
45988	BM	Surf Scoter	0.125	ND	ND	ND	ND	ND	ND	0.183
46019	BM	White-winged Scoter	0.004	ND	ND	ND	ND	ND	ND	0.013
46015	BM	White-winged Scoter	0.007	ND	ND	ND	ND	ND	ND	0.011

TABLE 3. METAL LEVELS IN EGGS OF WATERFOWL COLLECTED FROM NORTHERN CANADA, 1988-91

Wet Wt. (mg/kg or ppm)

I.D.	Tiss	Species	Age	Sex	N	Date mo	Date yr	Latitude	Longitude	Prov/ Terr	Location	Cd	As	Pb	Se	Hg*
49489	EGG+	Thick-billed Murre	---	---	1	7	91	6235	07750	NWT	Digges Island		0.15	<0.03	0.81	0.34
49490	EGG+	Thick-billed Murre	---	---	1	7	91	6235	07750	NWT	Digges Island		0.13	<0.03	0.70	0.19
49491	EGG+	Thick-billed Murre	---	---	1	7	91	6235	07750	NWT	Digges Island		0.18	<0.04	0.84	0.18
49492	EGG+	Thick-billed Murre	---	---	1	7	91	6235	07750	NWT	Digges Island		0.16	<0.03	0.82	0.32
49493	EGG+	Thick-billed Murre	---	---	1	7	91	6235	07750	NWT	Digges Island		0.18	<0.04	0.71	0.22
49494	EGG+	Thick-billed Murre	---	---	5	7	91	6235	07750	NWT	Digges Island		0.16	<0.03	0.81	0.23
50091	EGG+	Black Guillemot	---	---	1	7	91	5842	06557	Que	Kangiqsualuujjuaq (George River)		0.07	<0.04	0.70	0.68
50092	EGG+	Black Guillemot	---	---	1	7	91	5842	06557	Que	Kangiqsualuujjuaq (George River)		0.05	<0.04	0.57	0.51
50093	EGG+	Black Guillemot	---	---	1	7	91	5842	06557	Que	Kangiqsualuujjuaq (George River)		0.07	<0.04	0.66	0.47
50094	EGG+	Black Guillemot	---	---	1	7	91	5842	06557	Que	Kangiqsualuujjuaq (George River)		0.08	<0.03	0.70	0.54
50095	EGG+	Black Guillemot	---	---	1	7	91	5842	06557	Que	Kangiqsualuujjuaq (George River)		0.07	<0.04	0.55	0.34
50096	EGG+	Black Guillemot	---	---	5	7	91	5842	06557	Que	Kangiqsualuujjuaq (George River)		0.07		0.66	
50102	EGG	Canada Goose	---	---	1	7	91	5842	06557	Que	Kangiqsualuujjuaq (George River)		0.17		0.70	
50107	EGG	Common Eider	---	---	4	7	91	5842	06557	Que	Kangiqsualuujjuaq (George River)		0.27		0.79	
50108	EGG+	Herring Gull	---	---	1	6	91	5842	06557	Que	Kangiqsualuujjuaq (George River)		0.08	<0.03	0.54	0.20
50109	EGG+	Herring Gull	---	---	1	6	91	5842	06557	Que	Kangiqsualuujjuaq (George River)		0.15	<0.04	0.53	0.19
50110	EGG+	Herring Gull	---	---	1	6	91	5842	06557	Que	Kangiqsualuujjuaq (George River)		0.13	<0.02	0.66	0.17
50111	EGG+	Herring Gull	---	---	1	6	91	5842	06557	Que	Kangiqsualuujjuaq (George River)		0.05	<0.04	0.44	0.19
50112	EGG+	Herring Gull	---	---	1	6	91	5842	06557	Que	Kangiqsualuujjuaq (George River)		0.13	<0.04	0.50	0.15
50113	EGG+	Herring Gull	---	---	5	6	91	5842	06557	Que	Kangiqsualuujjuaq (George River)		0.10	<0.04	0.52	0.22

* Total Mercury

All 1991 values measured on dry wt. basis & converted to wet wt. values
Blanks indicate not analyzed.

EGG - Egg; Ad - Adult; Im - Immature; F - Female; M - Male; U - Unknown

+ The following are individual analyses of pools: EGG - 50108-50112 = 50113; 50091-50095 = 50096; 49489-49493 = 49494;

TABLE 4. METAL LEVELS IN BREAST MUSCLE OF WATERFOWL COLLECTED FROM NORTHERN CANADA, 1988-91

Wet Wt. (mg/kg or ppm)

I.D.	Tiss	Species	Age	Sex	N	Date mo	Date yr	Latitude	Longitude	Prov/ Terr	Location	Cd	As	Pb	Se	Hg*
41747	BM	Common Goldeneye	mixed	mixed	6	10	89			Yukon	Yukon R/Whitehorse	<0.01	<0.10	0.20	0.50	0.10
40417	BM	Lesser Scaup	immat	female	1	9	88			Yukon	Yukon River	0.01	<0.10	<0.05	<0.10	0.17
45882	BM	Lesser Scaup	8Ad	8M	8	5/6	90	6735	13950	Yukon	Old Crow	<0.29		0.13		<0.08
45979	BM	Lesser Scaup	10Ad	2F,8M	10	5	90	6043	13503	Yukon	Whitehorse	<0.22		<0.07		<0.07
40427	BM	Mallard	mixed	mixed	8	9/10	88	6111	13512	Yukon	Yukon R./L. Laberge	0.01	0.11	0.05	0.21	0.12
43276	BM	Mallard	adult	mixed	8	5	88			Yukon	Yukon R/Whitehorse	0.08	<0.10	<0.10	0.40	<0.05
41859	BM	Northern Pintail	mixed	mixed	12	5	88			Yukon	Yukon R/Whitehorse	<0.01	<0.10	0.50	0.40	<0.05
45942	BM	Northern Pintail	5Ad	5M	5	5/6	90	6735	13950	Yukon	Old Crow R	<0.20		0.14		<0.08
45888	BM	Surf Scoter	5Ad	1F,4M	5	5/6	90	6735	13950	Yukon	Old Crow	<0.21		<0.07		0.15
45988	BM	Surf Scoter	4Ad	1F,3M	4	5	90	6043	13503	Yukon	Whitehorse	<0.21		<0.07		0.17
46015	BM	White-winged Scoter	6Ad	6M	6	5	90	6043	13503	Yukon	Whitehorse	0.30		<0.08		0.11
46019	BM	White-winged Scoter	7Ad	1F,6M	7	5/6	90	6725-35	13950-14100	Yukon	Old Crow/Porcupine R	<0.22		<0.08		0.11
44791	BM	Canada Goose	10Ad	3F,6M,1U	10	5/6	90	6932	09332	NWT	Spence Bay	<0.23		0.20		<0.08
46654	BM	Canada Goose	9Im,1U	9F,1M	10	Spr	90	5632	07914	NWT	Sanikiluaq	<0.23		3.73		<0.09
43279	BM	Common Eider	immat	mixed	5	1	90			NWT	Pangnirtung	0.03	0.10	<0.10	0.40	0.11
43280	BM	Common Eider	adult	female	3	1	90			NWT	Pangnirtung	0.11	<0.10	<0.10	0.70	0.09
40870	BM	Common Eider	adult	male	5	11	88			NWT	Sanikiluaq	0.07	<0.10	0.29	0.32	<0.05
40871	BM	Common Eider	immat	male	5	11	88			NWT	Sanikiluaq	0.02	0.13	0.41	0.78	<0.05
40634	BM	Common Eider	adult	female	2	11	88			NWT	Sanikiluaq	0.06	0.13	<0.05	1.06	0.10
43111	BM+	King Eider	adult	female	3	5	89			NWT	Holman Island	0.28	0.20	1.00	1.10	0.15
43278	BM+	King Eider	adult	male	8	5	89			NWT	Holman Island	0.51	<0.10	0.30	1.90	0.15
42669	BM+	King Eider	adult	mixed	11	5	89			NWT	Holman Island	0.40	<0.10	0.20	1.60	0.19
43108	BM+	Lesser Snow Goose	adult	male	6	6	89			NWT	Eskimo Point	0.03	<0.10	12.30	0.20	<0.05
43107	BM+	Lesser Snow Goose	adult	female	4	6	89			NWT	Eskimo Point	0.02	0.40	0.20	0.20	<0.05
41793	BM+	Lesser Snow Goose	adult	mixed	10	6	89			NWT	Eskimo Point	0.05	<0.10	12.30	0.20	<0.05
43109	BM+	Lesser Snow Goose	adult	mixed	4	5	89			NWT	Tuktoyaktuk	0.01	<0.10	0.30	0.20	<0.05
43110	BM+	Lesser Snow Goose	immat	mixed	6	5	89			NWT	Tuktoyaktuk	0.03	<0.10	0.30	0.50	<0.05
42634	BM+	Lesser Snow Goose	mixed	mixed	10	5	89			NWT	Tuktoyaktuk	0.14	<0.10	0.20	0.20	<0.05
39916	BM	Mallard	mixed	mixed	10	9/10	88			NWT	Stagg River	0.02	<0.10	0.21	<0.10	0.12
49246	BM	Canada Goose	1Ad,1Im	1F,1M	2	5/6	91	6107	09403	NWT	Arviat (Eskimo Point)		0.01		0.59	
49331	BM+	Lesser Snow Goose	1Im	1F	1	5/6	91	6107	09403	NWT	Arviat (Eskimo Point)	<0.11	0.01	<0.05	0.61	<0.08
49332	BM+	Lesser Snow Goose	7Ad	7M	7	5/6	91	6107	09403	NWT	Arviat (Eskimo Point)	<0.11	0.01	<0.04	0.68	<0.07
49333	BM+	Lesser Snow Goose	7Ad	7F	7	5/6	91	6107	09403	NWT	Arviat (Eskimo Point)	<0.11	0.01	<0.05	0.66	<0.10

Wet Wt. (mg/kg or ppm)

I.D.	Tiss	Species	Age	Sex	N	Date mo	Date yr	Latitude	Longitude	Prov/ Terr	Location	Cd	As	Pb	Se	Hg*
50165	BM	Black Guillemot	5Ad	5M	5	7/8	91	5827	07805	Que	Inukjuaq	0.27	0.58	0.08	0.91	0.29
50179	BM	Black Guillemot	5Ad	4F,1M	5	7	91	5842	06557	Que	Kangiqsualuujjuaq (George River)	0.36	0.41	0.10	1.02	0.32
51769	BM	Black Scoter	5Ad	5F	5	9	91	5827	07805	Que	Inukjuaq	<0.11	0.10	0.08	0.79	0.10
51979	BM+	Canada Goose	2Ad	2F	2	9	91	5827	07805	Que	Inukjuaq	<0.11	0.003	0.58	0.04	<0.07
51810	BM+	Canada Goose	2Ad,2Im	2F,2M	4	9	91	5827	07805	Que	Inukjuaq		0.004	0.08	0.06	<0.06
51978	BM+	Canada Goose	2Im	2M	2	9	91	5827	07805	Que	Inukjuaq	<0.11	0.003	0.07	0.06	<0.04
50258	BM	Canada Goose	4Ad,1Im	4F,1M	5	7	91	5842	06557	Que	Kangiqsualuujjuaq (George River)	<0.11	0.01	<0.05	0.15	<0.08
51805	BM	Canada Goose	1Im	1M	1	6-8	91	6214	07538	Que	Salluit		0.004	0.07	0.05	<0.13
51783	BM	Common Eider	3Ad	3F	3	9/10	91	5827	07805	Que	Inukjuaq	<0.11	0.18	0.08	0.58	<0.05
50172	BM	Common Eider	6Ad	5F,1M	6	7	91	5842	06557	Que	Kangiqsualuujjuaq (George River)	0.18	0.26	0.21	1.22	0.23
51789	BM	Common Eider	3Ad	1F,2M	3	6-8	91	6214	07538	Que	Salluit	0.18	0.33	0.06	0.85	0.26
51845	BM	Common Loon	3Im	1F,2U	3	10	91	5827	07805	Que	Inukjuaq		0.24	<0.05	0.59	0.90
50252	BM	Common Loon	1Ad	1M	1	8	91	5827	07805	Que	Inukjuaq	<0.11	0.42	<0.05	1.16	0.83
50192	BM	Common Loon	3Ad	2F,1M	3	7	91	5842	06557	Que	Kangiqsualuujjuaq (George River)	<0.11	0.46	<0.05	0.91	0.68
51848	BM	Common Loon	2Ad	1F,1M	2	6-8	91	6214	07538	Que	Salluit		0.41	<0.05	0.69	1.04
51568	BM+	Glaucous Gull	1Im	1F	1	9	91	5827	07805	Que	Inukjuaq (at garbage dump)	<0.11	0.13	<0.06	0.35	0.57
51569	BM+	Glaucous Gull	1Ad	1F	1	9	91	5827	07805	Que	Inukjuaq (at garbage dump)	<0.11	0.01	<0.06	0.25	0.14
51570	BM+	Glaucous Gull	1Im	1F	1	9	91	5827	07805	Que	Inukjuaq (at garbage dump)	<0.11	0.08	0.09	0.45	0.16
51571	BM+	Glaucous Gull	1Ad,2Im	3F	3	9	91	5827	07805	Que	Inukjuaq (at garbage dump)	<0.11	0.06	<0.05	0.32	0.21
50528	BM	Herring Gull	3Ad	2F,1M	3	8	91	5529	07731	Que	Kuujjuaraapik	<0.11	1.52	<0.06	0.94	0.50
51572	BM+	Herring Gull	1Im	1F	1	9	91	5827	07805	Que	Inukjuaq (at garbage dump)	<0.11	0.10	<0.06	0.45	0.21
51573	BM+	Herring Gull	1Ad	1F	1	9	91	5827	07805	Que	Inukjuaq (at garbage dump)	<0.11	0.01	0.71	0.40	0.33
51574	BM+	Herring Gull	1Ad,1Im	2F	2	9	91	5827	07805	Que	Inukjuaq (at garbage dump)	<0.11	0.06	0.24	0.45	0.31
51785	BM+	King Eider	1Ad	1F	1	10	91	5827	07805	Que	Inukjuaq	<0.11	0.18	0.44	0.59	0.25
51780	BM+	King Eider	1Im	1M	1	9	91	5827	07805	Que	Inukjuaq	<0.11	0.22	<0.05	0.64	0.16
51790	BM+	King Eider	1Ad,1Im	1F,1M	2	9/10	91	5827	07805	Que	Inukjuaq	<0.11	0.19	0.09	0.63	0.21
51784	BM	King Eider	1Ad	1M	1	6-8	91	6214	07538	Que	Salluit	0.20	0.17	<0.05	0.91	0.24
51797	BM+	Lesser? Snow Goose	1Im	1M	1	9	91	5827	07805	Que	Inukjuaq	<0.11	0.002	1.90	0.03	<0.05
51798	BM+	Lesser? Snow Goose	1Ad,1Im	1F,1M	2	9	91	5827	07805	Que	Inukjuaq		0.003	<0.05	0.06	<0.09
51796	BM+	Lesser? Snow Goose	1Ad	1F	1	9	91	5827	07805	Que	Inukjuaq	<0.11	0.002	<0.06	0.03	<0.12
51795	BM	Lesser? Snow Goose	4Ad	1F,3M	4	6	91	6214	07538	Que	Salluit	<0.11	0.003	0.08	0.51	<0.07

Wet Wt. (mg/kg or ppm)

I.D.	Tiss	Species	Age	Sex	N	Date mo	Date yr	Latitude	Longitude	Prov/ Terr	Location	Cd	As	Pb	Se	Hg*
51980	BM+	Oldsquaw	4Im	3F,1M	4	10	91	5827	07805	Que	Inukjuaq	<0.11	0.15	0.10	0.49	0.16
51523	BM+	Oldsquaw	1Ad,4Im	4F,1M	5	10	91	5827	07805	Que	Inukjuaq	<0.11	0.15	0.08	0.45	0.13
51521	BM+	Oldsquaw	1Ad	1F	1	10	91	5827	07805	Que	Inukjuaq	<0.11	0.19	<0.05	0.57	0.08
50357	BM	Oldsquaw	5Ad	3F,2M	5	7	91	5842	06557	Que	Kangiqsualuujjuaq (George River)	<0.11	0.09	<0.05	0.82	0.24
51981	BM+	Red-breasted Merganser	4Ad	4F	4	9	91	5827	07805	Que	Inukjuaq	<0.11	0.07	0.30	0.41	0.56
51803	BM+	Red-breasted Merganser	1Im	1M	1	9	91	5827	07805	Que	Inukjuaq		0.10	<0.05	0.48	0.50
51804	BM+	Red-breasted Merganser	4Ad,1Im	4F,1M	5	9	91	5827	07805	Que	Inukjuaq		0.08	0.07	0.49	0.51
50259	BM	Red-breasted Merganser	1Ad	1M	1	6	91	5842	06557	Que	Kangiqsualuujjuaq (George River)	<0.11	0.09	<0.06	0.65	1.23
50195	BM	Red-throated Loon	2Ad	2M	2	6/7	91	5842	06557	Que	Kangiqsualuujjuaq (George River)	<0.11	0.26	<0.06	0.85	0.30
52145	BM	Rock Ptarmigan	2Ad,1Im	1F,2M	3	12	91	5842	06557	Que	George River	<0.11	0.003	0.83	0.18	<0.10
52147	BM	Rock Ptarmigan	1Ad	1F	1	8	91	5842	06557	Que	George River		0.002		0.13	
51778	BM*	Thick-billed Murre	4Ad	1F,3M	4	6-8	91	6214	07538	Que	Salluit	0.55	0.90	0.06	0.98	0.45
49500	BM	Thick-billed Murre	5Ad	3F,2M	5	7	91	6224	07755	Que	Ivujivik	0.41	1.22	0.07	1.00	0.29
52146	BM**	Willow Ptarmigan	1Im	1M	1	8	91	5842	06557	Que	George River	<0.11	0.02	10.75	0.09	<0.08

* Total Mercury

All 1991 values measured on dry wt. basis & converted to wet wt. values
Blanks indicate not analyzed.

* average of 2 replicate analyses calculated for Pb, Hg & Cd
** average of 3 replicate analyses calculated for Pb

BM - Breast Muscle; Ad - Adult; Im - Immature; F - Female; M - Male; U - Unknown

\+ The following are related pools: Brant - 53382, 53383 & 53384; Canada Goose - 52227 & 52395;
Mallard - 52447 & 52479; Northern Pintail - 52480, 52508, 52507 & 52506; Lesser? Snow Goose - 52814, 52815 & 52816;
Lesser Snow Goose - 49331, 49332 & 49333.

\+ The following are individual analyses of pools: BM - 51796 & 51797 = 51798; 51780 & 51785 = 51790; 51572 & 51573 = 51574; 51568-51570 = 51571.

\+ 51978 & 51979 are sub-pools of 51810; 51521 & 51980 are sub-pools of 51523; 51981 & 51803 are sub-pools of 51804.

\+ Lesser Snow Geese - 43107 & 43108 are sub-pools of 41793; 43109 & 43110 are sub-pools of 42634.
\+ King Eider - 43111 & 43278 are sub-pools of 42669.

TRENDS AND EFFECTS OF ENVIRONMENTAL CONTAMINANTS IN ARCTIC SEABIRDS, WATERFOWL, AND OTHER WILDLIFE. STUDY III. CONTAMINANTS IN AQUATIC BIRDS OF GREAT SLAVE LAKE

PROJECT LEADER: B.M. Braune, National Wildlife Research Centre, Canadian Wildlife Service, Environment Canada, Hull, Québec

OBJECTIVES

To collect eggs and prefledged chicks of four species of aquatic birds representative of different trophic levels from the Slave River delta (Great Slave Lake) and the North Arm of Great Slave Lake in order to obtain baseline data on current levels of contaminants taken up locally from a potentially contaminated site due to pulp and paper mill effluent and a control site on the same lake.

RESULTS

Eggs for 3 of the 4 species requested and chicks for 2 species were received from the North Arm of Great Slave Lake (control site). Eggs were collected from the Slave River Delta by GNWT personnel but went missing from the Fort Resolution office before they could be sent to the National Wildlife Research Centre for analysis. No chicks were collected from the Slave River Delta.

Samples were analyzed on a pooled basis. Residue data available to date (Table 1) indicate that eggs of Red-breasted Merganser had the highest levels of almost all contaminants analyzed compared with eggs of Greater Scaup and Herring Gulls. Levels of contaminants found in whole body analyses of chicks were considerably less than the levels found in the eggs suggesting that contaminant accumulation through local food sources is relatively low compared with the contaminant load transferred from the adult female to the egg.

UTILIZATION OF RESULTS

We are awaiting the analytical results for the second collection of chicks to confirm the comparison made for Herring Gull eggs and chicks. The results obtained from this study will be used as baseline data for any future studies to determine the contaminant impact, if any, of the development of pulp and paper mill operations on the river systems feeding into Great Slave Lake.

Expected project completion date: 1993

TABLE 1. LEVELS OF ORGANOCHLORINES IN AQUATIC BIRDS FROM NORTH ARM OF GREAT SLAVE LAKE

I.D.	Tissue	Species	N	Date mo	yr	% Water	% Lipid	1,2,4,5 -T4CB	1,2,3,4 -T4CB	QCB	HCB
55910	Egg	Red-breasted Merganser	2	Spring	92	65.6	18.20	ND	ND	0.002	0.036
55921	Egg	Greater Scaup	10	Spring	92	62.2	19.10	ND	ND	0.002	0.011
55932	Egg	Herring Gull	10	Spring	92	75.1	8.40	ND	ND	ND	0.022
57587	Whole Body	Herring Gull	10	8	92	70.2	9.00	ND	ND	ND	0.004

Wet Wt. (mg/kg or ppm)

I.D.	Tissue	Species	a-HCH	b-HCH	g-HCH	OCS	oxy-chlordane	trans-chlordane	cis-chlordane	trans-nonachlor	cis-nonachlor
55910	Egg	Red-breasted Merganser	0.001	0.002	ND	0.018	0.047	ND	0.005	0.018	0.014
55921	Egg	Greater Scaup	0.002	ND	0.001	0.004	0.019	ND	0.001	0.006	0.002
55932	Egg	Herring Gull	ND	0.007	ND	0.002	0.020	ND	ND	0.007	0.004
57587	Whole Body	Herring Gull	ND	ND	ND	ND	0.004	ND	ND	0.003	0.001

Wet Wt. (mg/kg or ppm)

I.D.	Tissue	Species	pp'-DDE	pp'-DDD	pp'-DDT	Photo-mirex	Mirex	HE	Dieldrin	Aroclor 1254:1260
55910	Egg	Red-breasted Merganser	0.655	0.003	0.017	0.004	0.004	0.032	0.050	2.869
55921	Egg	Greater Scaup	0.218	0.001	0.003	ND	0.001	0.005	0.011	1.633
55932	Egg	Herring Gull	0.613	ND	0.001	0.003	0.005	0.007	0.007	0.883
57587	Whole Body	Herring Gull	0.034	ND	ND	ND	ND	0.001	0.001	0.095

ND < 0.001 mg/kg wet wt.

CONTAMINANTS IN THE FINLAYSON CARIBOU HERD, ROSS RIVER, YUKON

PROJECT LEADER: L. Mychasiw, Renewable Resources, Yukon Territorial Government

PROJECT TEAM: M. Gamberg (Gamberg Consulting, Watson Lake, Yukon)
R. Florkiewicz (Renewable Resources, Yukon Territorial Government)

OBJECTIVES

1. To document background levels of heavy metals, radionuclides and organic contaminants in woodland caribou in the Yukon.

2. To identify potential contaminant related health problems in the caribou population.

3. To identify potential health risks to humans from the consumption of caribou meat and organs.

4. To develop recommendations for long-term monitoring and additional studies of organic, inorganic and radioactive contaminants in caribou in the Yukon.

DESCRIPTION

High levels of radioactive cesium have been found in caribou from the Northwest Territories (Taylor *et al.* 1988), and high levels of cadmium in Quebec caribou have prompted health advisory warnings by provincial authorities (Crete *et al.* 1989). While high levels of cadmium have also been found in barren-ground caribou from the Porcupine Herd in the northern part of the Yukon Territory (Gamberg and Scheuhammer in press), other contaminants, including cesium, lead, mercury, PCBs, dioxins and organochlorines had not been measured in Yukon caribou. Even cadmium had not been measured in woodland caribou in the Yukon Territory. These contaminants present potential health risks to the caribou population, and to those who consume the meat and organs from these animals. The Finlayson caribou herd is heavily used as a food source by local natives, who are concerned about contaminant levels in their natural diet.

ACTIVITIES IN 1991/92

A broad range of metals were measured in liver, kidney, muscle and bone from thirty-one woodland caribou taken from the Finlayson Caribou Herd near Ross River, Yukon. Tissues were analyzed using the inductively-coupled plasma with mass spectroscopy by Fenwick Laboratories, Halifax, N.S. Eight liver and eight kidney samples were also analyzed for methyl mercury using cold vapour atomic absorption spectroscopy at the National Wildlife Research Center in Hull, Quebec. Muscle tissue was analyzed for cesium^{137} by the Institute

of Arctic Biology at the University of Alaska, Fairbanks using a Nuclear Data (ND-1100) Spectrometer coupled with a NaI (TL) detector. Fat tissue from the same animals were pooled into five composite samples which were analyzed for organochlorines and PCBs (as Aroclors) using GC/MS and GC/ECD, and for dioxins and furans using high resolution gas chromatography with high resolution mass spectrometric detection. This analysis was performed by Axys Analytical Services, Sidney, B.C.

RESULTS

Cesium^{137} and metal concentrations in Finlayson caribou liver, kidney, muscle and bone were within expected ranges with the following exceptions. Kidneys had an average of 0.85 μg/g total mercury (wet weight), which was considerably higher than muskoxen from the Arctic (Salisbury *et al.* 1992). However, all eight kidneys and livers analyzed for methyl mercury had < 0.1 μg/g (wet weight), indicating that the mercury present was in the relatively non-toxic inorganic form. Bone had unusually high levels of barium (447 μg/g wet weight), which although of interest, is not of toxicological concern. Cadmium levels in livers and kidneys from Finlayson caribou were very high. In one instance kidney cadmium reached 141.1 μg/g (wet weight), which may be indicative of renal damage in the animal, although this was not evaluated. Cadmium in both organs was positively correlated with age, and this must be considered when comparing animals or herds. Taking this into consideration, cadmium levels in the Finlayson herd were considerably higher than in other caribou from the Canadian Arctic and Norway (Table 1).

Most pesticides tested in fat tissue from the Finlayson caribou, including PCBs measured as Aroclors, were not detectable. HCB and the HCH isomers were found, but the levels were low compared with caribou from the Northwest Territories (Muir *et al.* 1988; Thomas and Hamilton 1988).

The pattern of dioxin and furan deposition in fat tissue from the Finlayson caribou was unusual (Table 2). It was remarkably similar to patterns seen in air (Eitzer and Hites 1989), which may indicate an atmospheric vector and ultimately a combustion-related source. This pattern is not usually seen in animals and may be the result either of very high intake or contamination of the samples prior to or during analysis.

CONCLUSIONS AND UTILIZATION OF RESULTS

In general, caribou from the Arctic have similar levels of cadmium (Gamberg and Scheuhammer in press; Crete *et al.* 1989; Froslie *et al.* 1986), indicating contamination through atmospheric long-range transport. The elevated levels of cadmium, as well as mercury and barium in the Finlayson caribou herd may indicate a local source of contaminants. A large lead zinc mine at Faro, operational since 1969, and an abandoned barium mine in Macmillan Pass are both close to the study area and may be responsible for some of the contamination. Alternatively, the naturally high levels of minerals in the native

rock that attract mining activity, may be responsible for high contaminant levels in the caribou.

Further study in this area is ongoing under a new Arctic Environmental Strategy (AES) study "Contaminants in Woodland Caribou". The Finlayson herd will be sampled again, and compared with another herd from the same area and a control herd, to determine the extent of the high cadmium levels in caribou from the Territory. Lichens, soil and rock from all three areas will also be analyzed to determine the avenue of contamination, and radioisotope ratios will be used to attempt to identify a source.

The presence and patterns of dioxins and furans in the Finlayson caribou are not currently well understood. They will continue to be studied and compared with patterns in caribou from the Northwest Territories in cooperation with an AES project being carried out in 1993/94 by C. Hebert (National Wildlife Research Center, Canadian Wildlife Service, Hull, Quebec).

Results of all contaminant analyses have been submitted to Health and Welfare Canada for human health risk assessments.

When original results were received from the service laboratories, it was clear there was a potential human health hazard, at least from cadmium in caribou livers and kidneys. Meetings were held with the Ross River Dena Band, and subsequently with the Kaska Tribal Council to inform them of the situation. Additional meetings are planned pending the completion of the human health risk assessments by Health and Welfare Canada.

A final report on the project has been prepared. Publication of results in a scientific journal will be delayed until completion of the 1993/94 caribou project.

Expected Project completion date: March 31, 1993

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Table 1. Mean cadmium concentrations in (N) livers and kidneys from Arctic caribou (μg/g wet wt.).

Age (yr)	Norway[1]	Yukon/NWT[2]	Finlayson[3]
Kidney			
≤0.5	—	2.9 (12)	8.2 (1)
0.5-1.5	2.6 (35)	5.1 (16)	—
1.5-3.5	4.4 (24)	5.4 (25)	20.4 (9)
3.5-5.5	6.5 (29)	8.5 (16)	26.5 (10)
5.5-8.5	12.8 (19)	11.0 (9)	47.5 (9)
>8.5	19.8 (4)	19.0 (8)	47.3 (3)
Liver			
≤0.5	—	0.5 (12)	2.0 (1)
0.5-1.5	0.7 (48)	0.9 (17)	—
1.5-3.5	1.0 (39)	1.1 (33)	3.2 (9)
3.5-5.5	1.2 (43)	1.6 (34)	3.0 (10)
5.5-8.5	1.8 (29)	1.8 (23)	6.1 (9)
>8.5	1.8 (10)	2.2 (22)	4.7 (3)

[1] Data from Froslie *et al.* 1986 (whole liver, whole kidney)
[2] Data from Gamberg and Scheuhammer 1993 (whole liver, renal cortex)
[3] Data from this study (whole liver, whole kidney)

Table 2. Mean dioxin and furan concentrations in fat tissue from Finlayson caribou (pg/g). N=5 pooled samples, each of 4-5 individuals.

	X[1]	SD	Range
Dioxins			
T_4CDD - Total	ND	—	ND - ND
2,3,7,8	ND	—	ND - ND
P_5CDD - Total	ND	—	ND - 0.6
1,2,3,7,8	ND	—	ND - ND
H_6CDD - Total	124.8	80.5	34 - 270
1,2,3,4,7,8	ND	—	ND - <0.3
1,2,3,6,7,8	8.0	5.4	ND - 18
1,2,3,7,8,9	0.6	0.3	ND - 1.1
H_7 -Total	846	538.2	230 - 1800
1,2,3,4,6,7,8	225.8	145.3	69 - 490
O_8CDD - Total	2100	1410.0	700 - 4700
Furans			
T_4CDF - TotaL	ND	—	ND - 5.5
2,3,7,8	ND	—	ND - 1.4
P_5CDF -Total	10.12	14.5	1.0 - 39
1,2,3,7,8	ND	—	ND - 0.4
2,3,4,7,8	ND	—	ND - 0.5
H_6CDF - Total	44.36	34.9	9.8 - 110
1,2,3,4,7,8	ND	—	ND - 1.3
1,2,3,6,7,8	ND	—	ND - 0.5
2,3,4,6,7,8	ND	—	ND - 0.7
1,2,3,7,8,9	ND	—	ND - ND
H_7CDF - Total	62.4	37.8	22.0 - 130
1,2,3,4,6,7,8	22.2	13.6	8.0 - 47
1,2,3,4,7,8,9	ND	—	ND - ND
O_8CDF	10.64	6.6	3.0 - 22

[1]ND = non-detectable

IDENTIFICATION OF BASELINE LEVELS AND SPATIAL TRENDS OF ORGANOCHLORINE, HEAVY METAL AND RADIONUCLIDE CONTAMINANTS IN CARIBOU (*Rangifer tarandus*)

PROJECT LEADER: B. Elkin, Department of Renewable Resources, Government of Northwest Territories (GNWT)

PROJECT TEAM: Local Hunters
Renewable Resource Officers

OBJECTIVES

1. To assess the exposure of free-ranging caribou in the Northwest Territories to organochlorine, heavy metal and radionuclide contaminants.

2. To establish baseline levels of organochlorine, heavy metal and radionuclide contaminants in several caribou tissues.

3. To determine spatial trends of these contaminants in caribou herds across the NWT.

4. To identify specific contaminants or geographical locations that warrant further study in caribou.

5. To provide baseline contaminant data that will serve as the basis for ongoing monitoring of temporal trends of specific contaminants in caribou.

DESCRIPTION

Information on levels and spatial variation of contaminant exposure in caribou and other terrestrial wildlife species in Canadian arctic and subarctic regions are extremely limited, and data on temporal trends are nonexistent. The scarcity of metal or organic residue data for terrestrial mammals has been identified as one of the major data gaps in arctic contaminant research. The limited analyses that have been conducted on terrestrial species have indicated that a variety of contaminants are present, including cadmium, and warrant more comprehensive studies to establish baseline levels in species such as caribou.

Caribou are strict herbivores that have a winter diet consisting primarily of lichen. This simple food chain makes caribou a good species for monitoring changes and spatial trends in terrestrial ecosystem contamination. Lichens have a long-lived surface and accumulate atmospheric contaminants in a non-selective manner, resulting in a contaminant load similar to atmospheric input through long-range transport. The defined ranges and distribution of caribou herds across the NWT also make it an ideal species for the examination of spatial trends of contaminant deposition in the terrestrial ecosystem.

Caribou are a major component of the traditional diet in inland and coastal communities across the NWT. Baseline data on contaminant levels in caribou tissues are needed to direct and support the evaluation of human exposure to contaminants through the consumption of caribou as a country food. The very limited data that currently exist for contaminants in terrestrial species are generally insufficient to provide information on contaminant loads in caribou tissues used as country food.

This study will provide important baseline levels and spatial trends of organochlorine, heavy metal and radionuclide contaminants in caribou from 10 major herds across the NWT. Spatial trends in contaminant residues will contribute to the understanding of contaminant deposition within the terrestrial ecosystem. This baseline data will provide important information on current contaminant levels and serve as a basis for ongoing monitoring of temporal trends of specific compounds of concern. The survey of contaminant residues may also identify specific compounds or locations that warrant more detailed study.

RESULTS

Field caribou collections were conducted on Southampton Island in 1991/92, and at Lake Harbour, Cape Dorset, Arviat and the Bathurst herd in 1992/93. Ten male and 10 female caribou were collected at each site, with equal representation among age classes. Muscle, liver, kidney and fat samples were collected from each animal for contaminant analysis. Teeth were collected for aging purposes, and a variety of reproductive, biological and morphometric measurements were taken.

Tissue samples from 10 of the caribou collected at each site were analyzed in 1992/93 for organochlorine and heavy metal contaminants at the Great Lakes Institute in Windsor, Ontario. The suite of contaminants assessed were a spectrum of 63 organic chemicals, including 43 PCB congeners and 20 pesticides, toxaphene, and 10 metals. Muscle samples of 10 caribou from each location were also analyzed for 7 radionuclides at AECL Whiteshell Laboratories in Pinawa, Manitoba. Samples from the remaining 10 caribou from each site were banked for future use. The tissue residue analysis from these collections was completed in 1992/93, and the organochlorine and heavy metal dataset has been forwarded to Health & Welfare Canada for human health risk assessment. The radionuclide data is currently being evaluated, and results are not presented here.

The contaminant residue dataset from the first 5 caribou herds sampled was completed in 1992/93. An initial review of the data has taken place, and a more intensive evaluation and interpretation is currently underway. Results of data from Arviat, Cape Dorset, Lake Harbour and Southampton Island are presented here; results from the Bathurst herd are still being reviewed. Overall, contaminant levels in NWT caribou were low in comparison to arctic marine mammals, and similar to limited analyses previously conducted on terrestrial herbivores in the Canadian Arctic. Caribou from Cape Dorset had significantly lower body fat and tissue lipid content than the other 3 locations.

Although a wide range of organochlorine compounds were detected, many of the pesticides and PCB congeners were found at low levels. Residue levels of the predominant organochlorine compounds as well as several biologically important contaminants are given in Table 1. In general, organochlorine compounds were significantly higher in caribou from Cape Dorset and Lake Harbour on southern Baffin Island than in caribou from Southampton Island and the mainland community of Arviat. Total PCB residues ranged from a mean of 6.8 μg/kg in Arviat caribou to 31.0 μg/kg in Lake Harbour animals. PCB congener #153 was bioaccumulated to the greatest extent by caribou. Moderate levels of HCB, α-HCH, and oxychlordane were also detected. HCB residues ranged from a mean of 33.7 μg/kg (wet weight) in fat of Southampton Island caribou to 107.6 μg/kg in Lake Harbour animals. The levels and patterns of organochlorine residues detected were similar to those from limited analyses previously conducted on Baffin Island and Fort Good Hope caribou.

Heavy metal burdens were obtained for 10 metals. Metal residues were also comparatively low with the exception of cadmium and mercury, which had community means of 14.1-33.9 μg/g and 1.3-2.9 μg/g, respectively, in kidney tissue (Table 2). Cadmium levels in kidney were an order of magnitude higher than the levels found in liver tissue. Cadmium levels detected in this study were similar to recent findings in caribou in the NWT and Yukon, and are comparable to those found in caribou from Norway and northern Quebec.

DISCUSSION

Long-range atmospheric transport appears to be the primary source of the contaminants detected, and the air-plant-animal contaminant pathway is the most likely route of contaminant deposition into the terrestrial food chain. The comparatively low levels of contaminants detected, coupled with stable or expanding populations in the herds tested, suggest little or no effects on caribou population health as a result of these contaminants. The initial results of this work have been provided to the NWT Policy Advisory Committee on Arctic Contaminants, and the complete organochlorine and heavy metal dataset has been forwarded to GNWT Health and Health and Welfare Canada for interpretation of human health significance through a culturally appropriate health risk assessment. Spatial trends in contaminant exposure in caribou herds across the NWT will be further defined with ongoing collections and analyses.

Expected project completion date: Field caribou collections will be completed in 1993/94, and contaminant analyses will be completed in 1994/95. Data evaluation and interpretation, human health risk assessment, and community consultation will occur until 1995/96.

Table 1. Mean (SD) levels (wet weight) of selected organochlorine contaminants (μg/kg [ppb]) from fat of caribou collected from four herds in the NWT, 1991-93.[a,b,c]

Community	% lipid	Oxychlor	HCB	α-HCH	1245-TCB[d]	QCB[e]	DDE	PCB118	PCB153	PCB138	Aroclor[f]
Arviat	78.6	0.69 (0.35)	43.66 (6.22)	19.85 (7.67)	0.88 (0.20)	0.84 (0.13)	0.35 (0.36)	0.57 (0.08)	0.64 (0.08)	0.50 (0.11)	6.80 (1.41)
Cape Dorset	48.0	1.91 (0.84)	59.45 (24.99)	20.82 (9.58)	3.25 (2.47)	1.77 (0.96)	1.06 (1.40)	1.44 (0.65)	2.13 (0.93)	1.73 (0.69)	23.35 (9.27)
Lake Harbour	83.5	1.40 (0.84)	107.67 (14.80)	21.20 (7.77)	1.68 (0.63)	1.79 (0.21)	1.15 (0.93)	2.26 (0.24)	3.35 (0.65)	2.16 (0.44)	31.02 (8.97)
Southampton Island	85.6.	0.72 (0.46)	33.72 (9.89)	21.86 (8.88)	0.68 (0.16)	0.694 (0.18)	0.59 (0.28)	0.68 (0.27)	1.43 (0.63)	0.853 (0.33)	11.52 (4.40)

[a] n = 10 from each community.
[b] PCB126, PCB169 and PCB185 not detected (ND) in any samples.
[c] Samples below detection limit were assigned one half the limit (0.01 ppb) for calculation of means.
[d]TCB: Tetrachlorobenzene.
[e]QCB: Pentachlorobenzene.
[f] Aroclor 1242:1254:1260 is equivalent to Total PCB.

Table 2. Mean levels (wet weight) of cadmium (Cd) and mercury (Hg) (μg/g [ppm]) from kidneys and livers of caribou collected from four herds in the NWT, 1991-93.[a]

Community	Kidney		Liver	
	Cd	Hg	Cd	Hg
Arviat	33.86	2.93	3.70	0.90
Cape Dorset	14.06	1.25	2.24	0.38
Lake Harbour	31.96	2.57	4.37	0.58
Southampton Island	18.79	2.22	n.a.	n.a.

[a] n = 10 from each community.
n.a. = samples not available for analysis.

ORGANOCHLORINE, HEAVY METAL AND RADIONUCLIDE CONTAMINANT TRANSFER THROUGH THE LICHEN-CARIBOU-WOLF FOOD CHAIN

PROJECT LEADER: B. Elkin, Department of Renewable Resources, Government of Northwest Territories

PROJECT TEAM: Local Hunters and Trappers
Local Renewable Resource Officers

OBJECTIVES

1. To assess the exposure of the lichen-caribou-wolf food chain in the Northwest Territories (NWT) to organochlorine, heavy metal and radionuclide contaminants.

2. To establish baseline levels of organochlorine, heavy metal and radionuclide contaminants in lichen, several caribou tissues, and several wolf tissues.

3. To determine spatial variation of contaminant levels in the lichen-caribou-wolf food chain among three locations in the NWT.

4. To model the transfer of these contaminants through the lichen-caribou-wolf food chain.

5. To derive transfer coefficients for specific contaminants between trophic levels in the food chain.

6. To identify specific contaminants or geographical locations that warrant further study at any of the three trophic levels.

DESCRIPTION

Information on levels and spatial variation of contaminant exposure in caribou (*Rangifer tarandus*) and other terrestrial wildlife species in Canadian arctic and subarctic regions are extremely limited, and data on temporal trends are nonexistent. The scarcity of metal or organic residue data for terrestrial mammals has been identified as one of the major data gaps in arctic contaminant research. The limited work that has been done has indicated the presence of a number of contaminants of concern, including cadmium, HCH, HCB, toxaphene, PCBs, DDT, chlordane-related compounds, ^{137}Cs and ^{210}Po. These findings warrant more comprehensive studies to establish baseline levels in terrestrial species.

All indications suggest a direct air-plant-animal pathway of contaminant transfer in the terrestrial food chain. Lichens are an important component of the arctic ecosystem, and accumulate contaminants more readily than other vascular plants because of their large surface area, longevity, and ability to bind heavy metals. Upon deposition, airborne

contaminants are accumulated and retained by lichen, thereby entering the food chain and potentially accumulating in herbivores and their predators. Caribou are strict herbivores that have a winter diet made up primarily of lichen. In arctic areas of the NWT, caribou has been shown to be the predominant food item of wolves (*Canis lupus*). This short and simple food chain provides an excellent opportunity to model and quantify the transfer of contaminants through three trophic levels to the top of the chain.

The defined ranges and distribution of caribou herds across the NWT also allows for an examination of spatial variation in contaminant exposure or food chain dynamics. Samples will be collected near Yellowknife (Bathurst herd), Cambridge Bay (Victoria Island herd), and Inuvik (Bluenose herd).

The presence and persistence of contaminants in the arctic terrestrial ecosystem is of concern because of the lichen-caribou-human food chain. Caribou are a major component of the traditional diet in inland and coastal communities across the NWT. Examination of the lichen-caribou-wolf food chain will provide basic information on the transfer of contaminants through three trophic levels, and may provide some insight into contaminant exposure in other top trophic level species.

This study will provide baseline data on organochlorine, heavy metal and radionuclide contaminant levels in all three trophic levels of the food chain. This data will be used to evaluate the transfer of specific contaminants through the lichen-caribou-wolf food chain. Transfer coefficients will be determined for specific contaminants, and the occurrence of biomagnification will be assessed. The data will be examined for any variation in contaminant levels or transfer based on animal age, sex, condition or geographical location.

RESULTS AND DISCUSSION

This study was initiated in 1992/93, and lichen and caribou collections were completed near Yellowknife (Bathurst herd). Wolf carcasses were collected by local hunters at Fort Reliance and submitted frozen for sampling and analysis. For both caribou and wolves, 10 male and 10 female animals were collected, with equal representation of age classes. Half of the samples were collected for immediate analysis, and half were banked pending results of the initial testing. Muscle, liver, kidney and fat were collected for contaminant analysis. Stomach contents were collected from the caribou and wolves for diet analysis. Teeth were collected for aging purposes, and a variety of biological and morphometric measurements were taken. Samples of three common lichen species (*Cladina mitis*, *Cladina rangiferina*, *Cetraria nivalis*) that are important in the caribou diet were selected for analysis. Lichen samples were collected at three different locations on the caribou range, with four subsites sampled at each location.

Lichen, caribou and wolf samples were analyzed in 1992/93 for organochlorine and heavy metal contaminants at the Great Lakes Institute in Windsor, Ontario. The suite of contaminants assessed were a spectrum of persistent organic chemicals (including 43 PCB congeners, 20 pesticides and toxaphene) and 10 metals. Organochlorine analyses were

conducted on individual fat, liver and muscle samples to evaluate tissue distribution. Pooled liver samples were used for toxaphene analysis. Whole lichen and muscle samples of caribou and wolves were also analyzed for radionuclides at AECL Whiteshell Laboratories in Pinawa, Manitoba. Individual samples were tested for ^{137}Cs, ^{40}K, ^{235}U, ^{226}Ra, ^{106}Ru, as well as the Chernobyl-related products ^{134}Cs and ^{60}Co.

The tissue residue analyses from these collections were completed in 1992/93, and the contaminant residue dataset is still being reviewed and evaluated at this time. Lichen, caribou and wolf collections are scheduled for Victoria Island and Inuvik in 1993/94 and 1994/95.

Expected project completion date: Field collections of lichen, caribou and wolves as well as contaminant analysis will be completed in 1994/95. Data evaluation and interpretation, human health risk assessment, and community consultation will occur until 1995/96.

IDENTIFICATION OF BASELINE LEVELS AND REPRODUCTIVE EFFECTS OF ORGANOCHLORINE AND HEAVY METAL CONTAMINANTS IN MINK (*Mustela vison*)

PROJECT LEADERS: K.G. Poole and B. Elkin, Department of Renewable Resources, Government of Northwest Territories (GNWT)

PROJECT TEAM: Local Trappers
Renewable Resource Officers

OBJECTIVES

1. To assess the exposure of wild mink to organochlorine and heavy metal contaminants in the western Northwest Territories.

2. To determine baseline levels of organochlorine and heavy metal contaminants in several mink tissues.

3. To identify spatial and temporal trends of these contaminants in mink along the Mackenzie, Slave and Liard drainage systems.

4. To evaluate the potential biological effects of contaminants on mink reproduction.

5. To determine the potential sources (via the prey base) of contaminants found in mink.

6. To evaluate mink as a sensitive indicator species to monitor environmental contaminants and ecosystem health.

DESCRIPTION

Mink (*Mustela vison*) are a top trophic level species that readily bioaccumulate environmental pollutants such as polychlorinated biphenyls (PCBs), DDT and methylmercury. Small mammals and fish form the greatest component of mink diet in most areas (Eagle and Whitman 1987), thus the species is exposed to contaminants from both the aquatic and the terrestrial food webs. Mink are extremely vulnerable to organochlorine contaminants, and are known to experience reproductive failure as a result of eating fish contaminated with relatively low levels of PCBs (reviewed in Ringer 1981, Eisler 1986). This unique sensitivity can result in population effects at low levels of environmental contaminants (Wren 1991). As such, mink may provide a sensitive indicator to assess short and long-term trends in environmental contaminants and ecosystem health.

A number of organochlorine and heavy metal contaminants have been identified in freshwater fish in the Mackenzie River, providing a potential source of contaminants for mink (Muir *et al.* 1989a, 1989b). Studies on fish at Fort Good Hope and Colville Lake have detected the presence

of PCBs, toxaphene and chlordane, as well as HCH, chlorobenzene, dieldrin and DDT (Kuhnlein 1991). The heavy metals copper, nickel, cadmium, mercury, selenium and zinc have also been identified. Furans, dioxins and toxaphene have also been identified in fish in low but detectable levels in the NWT portion of the Slave River and nearby lakes (K. Robertson, pers. comm.).

RESULTS

In 1991/92, 510 mink carcasses were collected from 36 trappers around Inuvik, Fort Good Hope, and Fort Rae. Tissue samples from 20 mink collected at each site were analyzed for organochlorine and heavy metal contaminants at the Great Lakes Institute in Windsor, Ontario. The suite of contaminants assessed were a spectrum of 63 organic chemicals, including 43 PCB congeners, dioxins/furans, and 10 metals. Stomach contents were collected for diet analysis, teeth were collected for aging purposes, and a variety of biological and morphometric measurements were taken. A number of possible reproductive measures were examined and determined to be ineffective in mink. The complete contaminant residue dataset from the 1991/92 sampling program was finalized and evaluated during 1992/93. A progress report detailing the 1991/92 collection is available (Poole and Elkin 1992).

In 1992/93, carcass collections were attempted in five communities. Ninety-eight mink were collected from Inuvik for a second year as part of a multi-year evaluation of temporal trends. Single year collections were planned for Fort Liard and Fort Simpson (Liard drainage), and Fort Smith and Fort Resolution (Slave drainage) in order to examine geographical variation in contaminant exposure. Unfortunately, we were only able to collect 6 carcasses from Fort Smith, 3 from Fort Liard and 1 from Fort Simpson, a result of a dismal fur harvest year across the North. Tissue samples also were taken from marten (*Martes americana*) collected in the same area as the Fort Good Hope mink. Examination of contaminant burdens in this sympatric Mustelid, which has a diet similar to mink with the exception of fish, may further elucidate the source of the contaminants detected in mink. Snowshoe hares (*Lepus americanus*), one of the major prey species of mink, were also collected at Inuvik and Fort Good Hope. Samples were sent for contaminant analysis, and the results are currently being examined.

Overall, contaminant levels in NWT mink were low in comparison with other mink studies in North America. Many of the pesticides and PCB congeners detected were found at very low levels, with less toxic compounds predominating. Residue levels of the predominant organochlorine compounds as well as several biologically important contaminants are given in Table 1. Total PCB residues ranged from a mean of 8.9 μg/kg in the livers of Inuvik mink to 95.5 μg/kg in mink from Fort Rae. In comparison, mink from Lake Ontario had total PCB levels which ranged from 34-1800 μg/kg (D. Weseloh, unpubl. data cited in Wren 1991), while PCB levels in mink from New York ranged from 30-7900 μg/kg (Foley *et al.* 1988). There were no significant differences in organochlorine contaminant burdens between the sexes, with body size, or with age. There was a general trend of decreasing contaminant levels with increasing latitude; Fort Rae mink had slightly higher residue levels than mink from Fort Good Hope, and Inuvik mink were significantly lower than other communities.

Heavy metal burdens were obtained for all 10 metals. Metal residues were comparatively low with the exception of mercury, which was at moderate levels (community means of 1.0-3.0 μg/g in liver samples). Mean levels of mercury and cadmium by community are given in Table 2. Again, there was a general trend of diminishing heavy metal burden the further north the samples originated. Statistically there were no differences in cadmium levels among sites, but mercury levels in Fort Rae and Fort Good Hope mink were significantly higher than those found in Inuvik. Cadmium burdens in females were higher than males ($p = 0.028$).

Evaluation of the 510 mink collected in 1991/92 indicated that over 60% of the harvest in all areas was composed of juveniles, and 62% were males. By conventional assessment, these figures suggest light to moderate harvesting pressure from healthy populations. Inuvik mink were significantly larger than mink from the Fort Good Hope area, which were in turn larger than Fort Rae mink. Fort Rae mink were significantly less fat than mink from the other two communities.

DISCUSSION

The population indices, coupled with comparatively low levels of contaminants, suggest little or no effects on mink reproduction or population health as a result of these contaminants. Spatial and temporal trends in contaminants in mink and selected prey species will be further defined with ongoing collections and analyses. At present, long-range atmospheric transport appears to be the primary source of the contaminants detected. Inuvik mink, trapped in the Mackenzie Delta, were the only animals in the collections in direct contact with the Mackenzie River and potential water-borne pollutants being flushed down-stream from the South. Burdens in Inuvik mink were generally one-tenth the levels found in mink from the other two communities, which were taken far off the river. Thus, long-range aquatic transport from southern sources of pollution is at present not likely a major source of the contaminants found in mink.

Expected project completion date: The determination of baseline contaminant levels and spatial/temporal trends in mink and major prey species will be completed at the end of 1994/95. The second part of the study examining the physiological impact of contaminants on reproduction is currently under review, and will begin in 1994/95 if appropriate methodology is in place.

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Table 1. Mean (SD) levels (wet weight) of selected organochlorine contaminants (μg/kg [ppb]) from livers of mink taken near three communities in the western NWT, 1991-92.[a,b,c]

Community	% lipid	Oxychlor	DDE	DDT	Dieldrin	PCB118	PCB153	PCB138	PCB126	PCB189	Aroclor[d]
Fort Rae	6.9	3.3 (2.87)	6.2 (9.30)	0.03 (0.10)	0.6 (1.52)	1.5 (1.91)	6.2 (7.83)	7.1 (9.89)	ND	0.2 (0.54)	95.5 (113.62)
Fort Good Hope	7.2	3.5 (3.68)	5.3 (7.73)	ND	0.4 (0.46)	1.0 (1.58)	7.1 (19.09)	5.8 (13.75)	0.02 (0.06)	0.2 (0.87)	78.0 (185.84)
Inuvik	5.2	1.4 (2.18)	0.8 (0.63)	ND	0.7 (2.03)	0.1 (0.11)	0.6 (0.62)	0.7 (0.70)	ND	ND	8.9 (9.41)

[a] n = 20 from each community.
[b] PCB77 and PCB169 not detected (ND) in any samples.
[c] Samples below detection limit were assigned one half the limit (0.01 ppb) for calculation of means.
[d] Aroclor 1242:1254:1260 is equivalent to Total PCB.

Table 2. Mean (SD) levels (wet weight) of cadmium (Cd) and mercury (Hg) (μg/g [ppm]) from kidneys and livers of mink taken near three communities in the western NWT, 1991-92.[a,b]

Community	Kidney		Liver	
	Cd	Hg	Cd	Hg
Fort Rae	1.0 (1.58)	1.7 (0.92)	0.4 (0.57)	3.0 (2.45)
Fort Good Hope	0.9 (0.86)	1.2 (0.66)	0.2 (0.17)	2.2 (1.28)
Inuvik	0.8 (1.23)	0.7 (0.36)	0.3 (0.39)	1.0 (0.58)

[a] n = 20 from each community.
[b] Outliers removed (liver n = 3, kidney n = 1).

ARCTIC SPECIMEN BANKING

PROJECT LEADER: B. Wakeford, National Wildlife Research Centre, Canadian Wildlife Service, Environment Canada, Hull, Quebec

OBJECTIVES

1. To identify existing facilities for Arctic specimens/tissues, complete with a description of the facilities and estimate of capacity.

2. To compile a list of Arctic sample holdings to be published in a catalogue.

3. To document how data management is being handled by the different facilities.

RESULTS

The objectives were to be met through the issuance of a contract. The contract was let to the UMA engineering company in November 1992, and the final report was delivered in March 1993.

The report contained the details of a survey of some 100 facilities in Canada and the USA which contained biological specimens from the arctic region. The information was entered into a computer database which is searchable using a commonly available software (dBase IV). The contractor also supplied the database printed in a manner which is suitable for compilation in a catalogue.

UTILIZATION OF RESULTS

The survey was extremely successful and interest in the final report has been expressed from many of the participants.

The contractor's report was circulated to individuals in Parks Canada and in the Departments of Fisheries and Oceans, Health and Welfare, and Indian and Northern Affairs, whose organisations were involved in specimen banking, for comments on the outline of the final catalogue.

Comments received will be considered during the drafting of the Arctic Specimen Catalogue which will be completed in 1993/94. The catalogue will be drafted in the form of a Technical Report with an initial print run of about 500 copies. It is expected that it will receive widespread distribution in North America and Europe.

Expected completion date: The catalogue is expected to be printed in final form by October 1, 1993.

NORTHERN AQUATIC FOOD CHAIN CONTAMINATION DATABASE CONTINUED DEVELOPMENT

PROJECT LEADER: H. Careau, Public Health Centre (Québec region), Centre Hospitalier de l'Université Laval (CHUL)

PROJECT TEAM: H. Careau, É. Dewailly, P. Ayotte

OBJECTIVES

1. To organize and proceed with updating the database, on an on-going basis, by gathering and integrating new data.

2. To develop a more user-friendly version of the database so that it can be used by researchers; this includes its translation in both French and English.

3. To promote and use the database as a research tool.

DESCRIPTION

The initial development of the database started in late summer of 1990. In the early stages of this project, which reviews contamination data of organisms in the aquatic food chain and of humans from northern regions, it became clear that the processing of large amounts of data was going to be a large task. The production of a database was the option chosen to store the information gathered through literary review and collection of unpublished data directly from researchers and organizations. It was soon found that such a database was needed for encompassing a growing area of research in one media.

The system supporting the software 4th DIMENSION is Macintosh. This was decided for reasons of program flexibility, user-friendliness and possible interaction with any geographical information system (GIS). Briefly, the database consists of mean contamination levels of any contaminant in organisms of the aquatic food chain and humans of the regions covering the Yukon and Northwest Territories, the hydrographic basins of the Hudson and James Bays, Northern Québec, Labrador, and Greenland. More than thirty variables are defined to characterize a mean concentration level (Fig. 1). Availability of data is a question of obtainability.

Because of the possibilities of usage and the growing demand for such a database, it was in the interest of everyone to pursue updating and further development of the Northern Aquatic Food Chain Contamination Database. Indeed, the interest of researchers was evident and requests for information have already been received.

ACTIVITIES IN 1992/93

In June of 1992 a report depicting the "State of Contamination of Northern Canada and Greenland" (French version) was completed. The basis of the study was human contamination by major organochlorines and metals as a result of aquatic food chain contamination. The database was the primary key which provided and managed the multi-source data on which the report was based. In June of 1993, the English version of this report became available.

One of the major undertakings for 1992/93 has been to make the database more user-friendly. We have also looked into making the database bilingual. Steps are being taken to produce an instruction manual which will be available in the fall of 1993.

The gathering of data continued throughout 1992/93, although both the collecting and the input of new data has been slowed down so as to permit consistency while testing the developing versions. Presently, over 16 500 record sheets make up the database and some have yet to be entered. There are more than 80 contaminants divided into three groups: organochlorines, metals, and other types. This last category includes hydrocarbons, radioactive products and any other compound which might not fit into the first two categories. More than one hundred species are grouped into several categories: humans, marine mammals, aquatic birds, terrestrial mammal (polar bear), fishes, and invertebrates. More than 30 different types of tissue have been catalogued. Revision of the data and the database before distribution was also a priority.

ACTIVITIES IN 1993/94

Distribution and methods of distribution of the database will be important aspects of this project, as well as gathering and input of new data. A strategy of data collection from researchers and organizations will have to be established to ensure proper coverage of new research.

Furthermore, with the assistance of Health and Welfare Canada the possible compatibility of this database with DOS (IBM-PC and compatible) has been investigated. The database has been tested on a PC, using a program which allows the PC to emulate the Macintosh system. Testing will continue when the development of user-friendliness, a user manual, and revision of data have been completed.

UTILIZATION AND CONCLUSION

The potential of the database will probably be better explored when access is increased. The ways in which the database will be utilized will depend on research needs. If many people use the database as either a reference or research tool and input of new data runs smoothly, it can only benefit those who are interested in northern research.

Partners: Hydro-Québec; Government of Northwest Health Department

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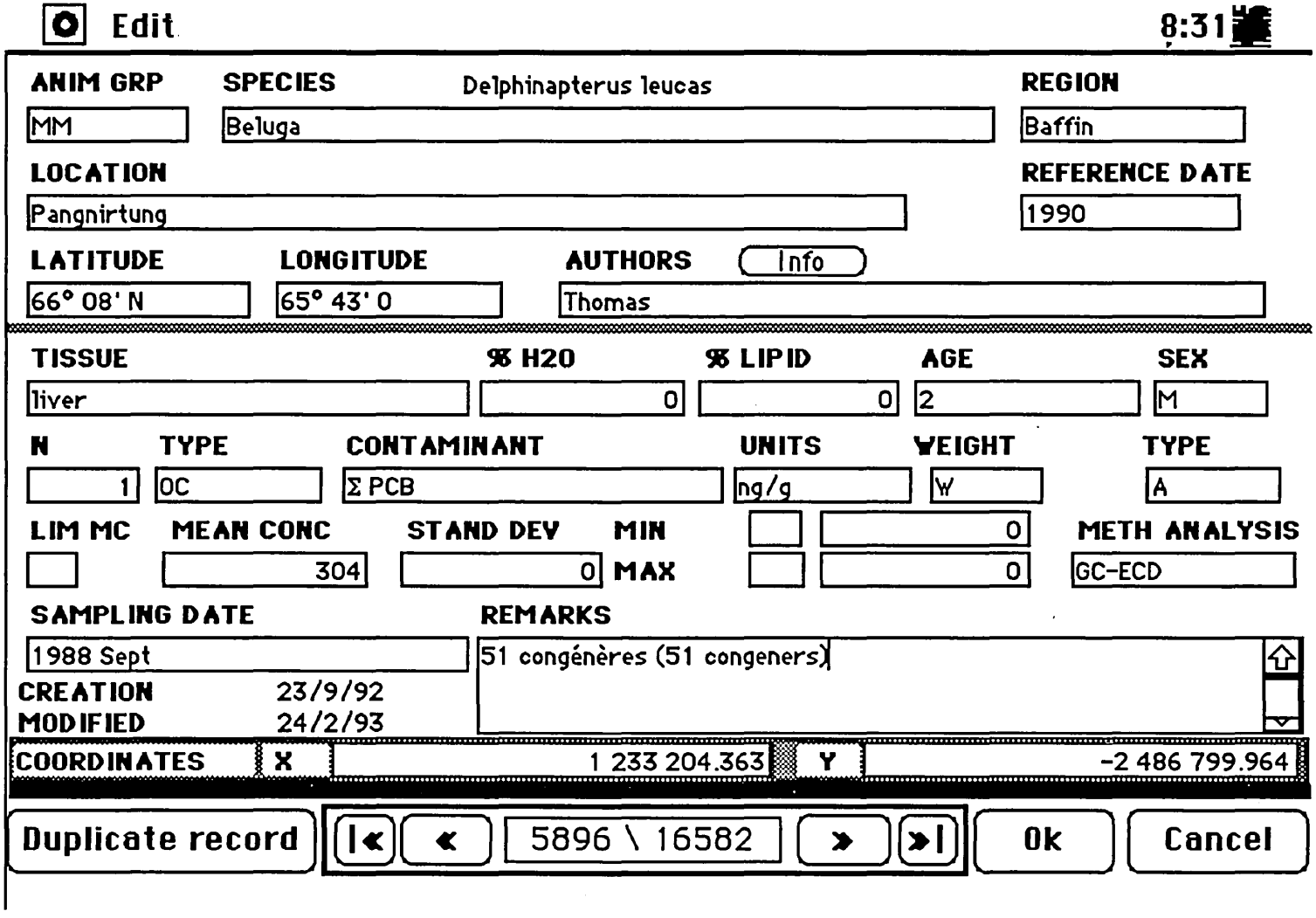

Figure 1 Example of a database record

HUMAN HEALTH

SANTÉ QUÉBEC INUIT STUDY: ENVIRONMENTAL ASPECTS

PROJECT LEADER: É. Dewailly, Laval University (CHUL)

PROJECT TEAM: É. Dewailly, S. Bruneau, L. Sauvé, P. Ayotte, C. Laliberté, J.P. Weber, A. Corriveau, L. Ferron, D. Audet, G. Lebel, B. Holub

OBJECTIVES

1. Risk Analysis

1.1 To establish a profile of organochlorine and heavy metal contamination in a representative sample of the Nunavik population.

1.2 To determine the role of personal factors (age, sex, lifestyle: tobacco, diet...) and environmental factors (geographical variation in country food contamination) associated with this human contamination.

2. Benefit Analysis

2.1 To determine the effect of seafood consumption on polyunsaturated fatty acid (PUFA) and selenium plasma levels.

2.2 To evaluate the associations between PUFA and the lipidogram.

3. Risk-benefit Analysis

DESCRIPTION

Since 1987, we have evaluated the contamination of milk from Inuit women of Arctic Québec. Organochlorine levels, including PCBs, coplanar PCBs, PCDDs and PCDFs, HCB and 10 chlorinated pesticides, were measured and used as exposure variables in a cohort study (Dewailly *et al.* 1989, Dewailly *et al.* 1992, Dewailly *et al.* In press(a), Dewailly *et al.* In press(b)) These studies were therefore limited to organochlorines and to a specific age group, i.e. women aged between 18 and 30 yrs. No data were available on mercury and lead exposure. Cadmium was measured in blood and urine of 142 hunters from Kuujjuaq (Benedetti *et al* In prep.). For organochlorines the need to assess adult exposure of both men and women from all ages was evident.

The Québec Health Inuit Survey which took place between September and November 1992 included the evaluation of the biological exposure of the general Arctic Québec population to organochlorines and heavy metals. Out of the 1790 randomly selected persons aged between 18 and 74 years, 1567 agreed to participate in the survey (87.5%). Among them, 498 had a

medical examination and a venous blood puncture for biochemical and toxicological analysis (glucose-insulin, phospholipid [PUFA], lipidogram, plasma concentration of heavy metals and organochlorines). External controls were performed for 10% of the population.

Heavy metals included lead, methylmercury and total mercury, cadmium, and selenium. The organochlorines measured were aldrin, β-BHC, α- and δ-chlordane, *cis*-nonachlor, heptachlor epoxide, hexachlorobenzene, mirex, oxychlordane and *trans*-nonachlor. PCB determination included Aroclor 1260 and 14 PCB congeners (IUPAC No. 28, 52, 99, 101, 105, 118, 128, 138, 153, 156, 170, 180, 183, 187). In addition, 30 pools of 10 plasma samples will be analysed for coplanar PCBs (IUPAC No. 77, 126, 169) and 2,3,7,8 chloro-substituted PCDDs and PCDFs.

ACTIVITIES IN 1992/93

- elaboration of the final protocol
- data collection: questionnaires, medical examination, blood sampling
- as of March 31 1993, we completed:
  - 498 analyses of Cd, Hg and Pb
  - 46 analyses of Se
  - 167 analyses of organochlorines
  - 498 analyses of biochemical parameters
- PUFA analyses are on-going

RESULTS

Because most of the biological variables had a log-normal distribution, we present preliminary results using geometric means. Mean concentrations are presented for the 467 Inuit persons (defined as born in Nunavik and with Inuktitut as their mother tongue) and for the 31 persons considered as non-native with another mother tongue (however half of them were born in Nunavik). When available, comparative data obtained in other populations are presented.

Heavy Metals

Cadmium

The mean level of cadmium in blood samples was 32.4 nmol/L (26.3 for non-natives and 32.9 for natives). Mean levels were 32.5 and 33.1 nmol/L for the 186 Inuit men and the 281 Inuit women, respectively. No trend was observed according to age. Cadmium was poorly correlated with other food chain contaminants. Further epidemiological analyses are on-going to determine the effect of tobacco and diet on cadmium blood levels. Table 1 presents some comparative data.

Table 1. Comparative data on blood cadmium (nmol/L) in Québec.

Cadmium Surveys	n[a]	Xg[b]	CI[c] 95%
1. Québec Health Inuit Survey (this study)	467	32.9	(30.4-35.6)
2. Kuujjuaq study[d]	142	31.9	(26.1-39.0)
3. Québec City study[d]	212	24.9	(21.1-29.5)
4. Lower North Shore study[d]	200	9.9	(8.0-12.3)

[a] n: number of participants [b] X: geometric mean
[c] CI: Confidence Interval (95%) [d] Benedetti *et al.* (In prep.)

Lead

The mean blood lead concentration was 0.437 μmol/L for the Inuit and 0.286 μmol/L for the non-natives ($P < 0.05$). Among natives, men had greater levels (0.521 μmol/L) than women (0.389 μmol/L) ($P < 0.05$). An increasing statistically significant trend was observed with age. Lead blood concentrations were 0.292, 0.449, 0.559 and 0.571 for the 15-24, 25-44, 45-64 and 65+ age groups, respectively. Lead was highly correlated with other food chain contaminants.

Mercury

The geometric mean plasma level for mercury was 88.6 nmol/L among natives and 18.0 nmol/L among non-natives. There were no differences between men and women. A strong association was observed between blood mercury and age, as shown in Table 2. Mercury was highly correlated with most PCB congeners, chlorinated pesticides, lead and selenium (data not shown).

Table 2. Mean (geometric) concentration of mercury (nmol/L) in blood samples from Inuit men and women, for various age groups.

Age	N	Men (n= 186)		Women (n= 280)	
		Xg	CI 95%	Xg	CI 95%
18-24	107	50.4	39.2 - 64.8	50.8	43.3 - 59.6
24-44	210	85.2	73.2 - 99.2	80.8	72.2 - 90.5
45-64	124	159.2	138.2 - 183.4	134.5	118.4 - 152.7
65+	25	200.5	141.6 - 284.0	131.0	131.0 - 183.4
Total	466	95.0	84.5 - 106.7	84.6	77.7 - 92.0

Selenium

The mean plasma concentration for selenium was 1.91 μmol/L with slight variations according to sex and age. The variation of plasma selenium concentration in this population was small (CI 95%: 1.79 - 2.03 μmol/L). Selenium was correlated with other food chain contaminants including mercury (see Fig. 1)

Organochlorines

PCBs

When expressed as Aroclor 1260, the geometric mean plasma concentration calculated for the first 167 samples was 22.1 μg/L (arithmetic mean: 33.7 μg/L). The concentration in men (n= 63) was 30.8 μg/L and 18.1 μg/L in women (n= 104). The mean levels of the 14 PCB congeners are presented in Table 3. The most persistent congeners, i.e. 138, 153 and 180, represent 22, 35 and 200%, respectively of the sum of the 14 PCB congeners (7480 ng/L).

Table 3: Mean (geometric) concentrations of various PCB congeners in plasma samples from the Arctic Québec Inuit population.

IUPAC No.	Geometric mean (ng/L)	CI 95%
28	14	12 - 15
52	93	81 - 106
99	607	524 - 703
101	82	72 - 92
105	89	76 - 103
118	381	327 - 441
128	22	20 - 25
138	1616	1388- 1881
153	2624	2247 - 3064
156	206	176 - 241
170	495	422 - 580
180	1502	1283 - 1759
183	168	146 - 193
187	581	506 - 668

Other organochlorines

Table 4 presents the geometric mean concentrations for 13 chlorinated compounds in plasma samples. Most of these pesticides were highly correlated with PCBs, Hg, Pb and Se concentrations. The relationship between PCBs (Aroclor 1260) and *p,p'*-DDE is presented in Figure 2. Figure 3 depicts the relationship between Aroclor 1260 and mercury.

ONGOING WORK

Organochlorine analyses for the remaining 331 plasma samples are on-going. Subsequently, 30 pooled samples will be analysed for coplanar PCBs and PCDD/PCDFs. PUFA analyses will be completed by the end of June 1993. Further work will include epidemiologic analyses using the information gathered from the health and dietary questionnaires, as well as the toxicological and biochemical data. The final report will be available by the end of December 1993. A tour of the communities to communicate the results is scheduled for February 1994.

Table 4: Mean (geometric) concentration of various organochlorine compounds (ng/L) in plasma samples from the Arctic Québec Inuit population.

Organochlorine	Geometric mean (ng/L)	CI 95% (ng/L)
aldrin	11	10 - 11
α-chlordane	12	11 - 13
δ-chlordane	28	24 - 32
BHC	156	138 - 177
cis-nonachlor	292	251 - 340
p,p'-DDE	8406	7212 - 9799
p,p'-DDT	283	245 - 326
dieldrin	175	150 - 205
heptachlor epoxide	209	177 - 246
HCB	1308	1139 - 1503
mirex	217	183 - 258
oxychlordane	1252	1037 - 1513
trans-nonachlor	1628	1371 - 1934

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Fig. 1: Quebec Health Inuit Survey
Correlation between blood Hg and plasma Se

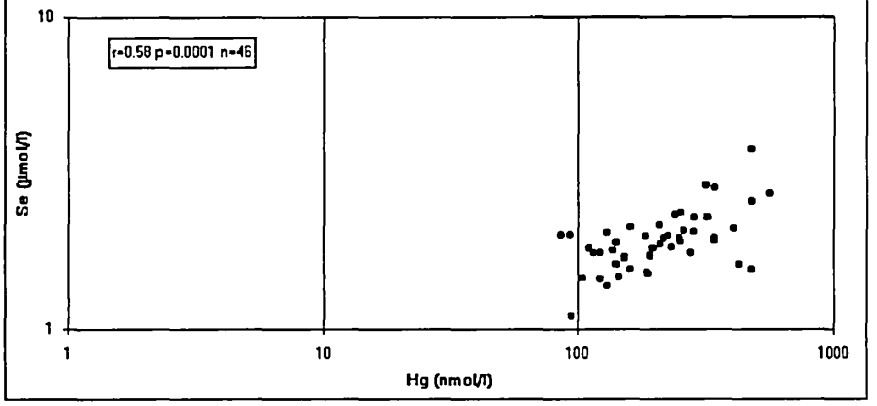


Fig. 2: Quebec Health Inuit Survey:
Correlation between blood Aroclor 1260 and blood pp'-DDE

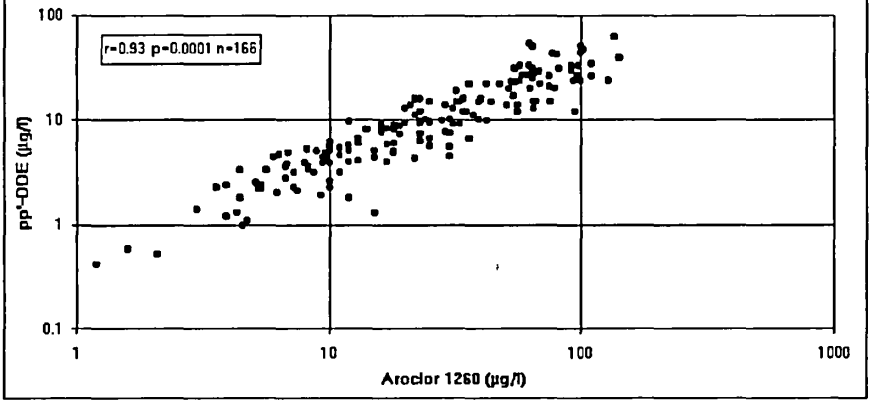
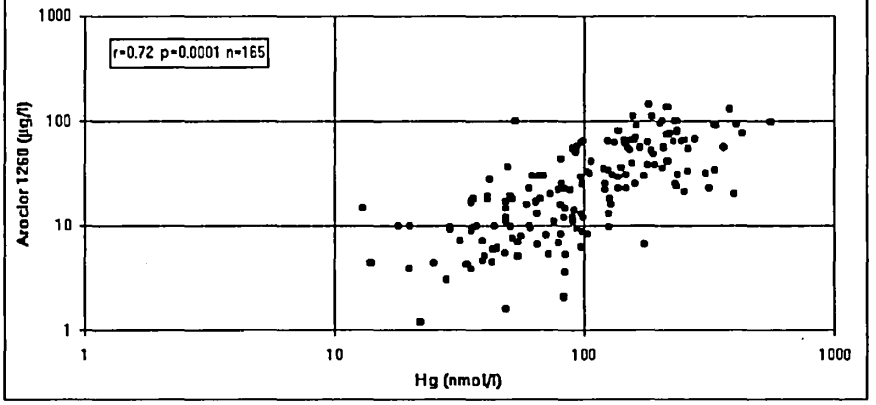


Fig. 3: Quebec Health Inuit Survey:
Correlation between blood Hg and blood Aroclor 1260



CORD BLOOD STUDY - NUNAVIK

PROJECT LEADER: É. Dewailly, Laval University (CHUL)

PROJECT TEAM: É. Dewailly, S. Bruneau, A. Corriveau, J. Walker, R. Nuttall, J.P. Weber, P. Ayotte.

OBJECTIVES

1. To assess the exposure of Northern neonates to selected organochlorines, heavy metals and radionuclides in at least two regions of Arctic Canada.

2. To compare the levels measured in different regions of Arctic Canada and in comparison groups from Southern Canada; identify communities where additional research should be implemented.

3. To evaluate temporal trends during a five-year program.

4. To determine a public health level of action according to blood concentrations found in these populations and the known toxicological properties of the contaminants identified from the scientific literature.

DESCRIPTION

Early work conducted in Baffin Island and Arctic Québec has demonstrated that because of their traditional dietary habits, Inuit people are exposed to an unusually high quantity of contaminants, mainly heavy metals and organochlorines.

This human exposure could be responsible for chronic health effects principally in the most sensitive groups: fetus and breast-fed babies. In the Great Lakes area, results reported in 1983 by Jacobson and coworkers has indicated that newborns of women who had eaten large quantities of contaminated fish had smaller birth size and a delay in maturation of motor abilities, cognitive function and poorer visual recognition memory at age one. At age four, researchers reported recently that deficits in body size persisted and indicators of poorer cognitive performance continued to be present and associated with *in utero* exposure, as measured by cord blood PCB levels. For these reasons, a routine screening program was started to determine newborn exposure to the contaminants present in the food chain. This project is also considered as a pilot study for the AMAP subprogram on human health.

Every mother delivering in Kuujjuaq and Povungnituk is asked to participate in the program after receiving proper information. Blood is collected at birth from the umbilical cord. Heavy metals (lead, mercury) and selenium are measured in whole blood. For blood samples containing more than 100 nmol/L of total mercury, inorganic mercury is also measured. Organochlorines include 14 PCB congeners (IUPAC 28, 52, 99, 101, 105, 118, 128, 138, 153,

Organochlorines include 14 PCB congeners (IUPAC 28, 52, 99, 101, 105, 118, 128, 138, 153, 156, 170, 180, 183, 187) and pesticides (aldrin, β-HCH, α- and δ-chlordane, *cis*-nonachlor, *p,p'*-DDE, *p,p'*-DDT, dieldrin, heptachlor epoxide, hexachlorobenzene, mirex, oxychlordane and *trans*-nonachlor). Detection limits are 0.03 μg/L for *p,p'*-DDT and β-HCH, 0.1 μg/L for heptachlor epoxide and dieldrin and 0.02 μg/L for other toxicants.

Total blood lipid is also measured in order to adjust the organochlorine concentration on a lipid basis (total lipids are usually 50 % lower in fetal blood than in maternal blood). Thyroid function tests are on-going.

PRELIMINARY RESULTS

All procedures (sampling, shipping and analyses) have now been tested and no major changes in the protocol were needed.

The following concentrations are expressed per litre of whole blood or plasma, since lipid measurements are not yet available. By the end of April 1993, only 8 samples from Ungava Bay had been analyzed for all contaminants. Hence, the following results need to be interpreted with caution.

Heavy Metals

Lead

The mean concentration in whole cord blood was 0.27 μmol/L (range 0.12-0.55 μmol/L). We compared this preliminary mean level to those measured in two recent studies. In the Québec City area, 430 cord blood samples were analyzed in 1990 (Rhainds and Levallois, In press). The mean concentration was 0.09 μmol/L (CI 95 % 0.09-0.10). In 1990, a study conducted in Toronto on 95 newborns revealed a similar mean concentration of 0.08 μmol/L (Koren *et al.* 1990). The intervention level of the Centre for Disease Control (CDC) is 0.48 μmol/L.

Mercury

The mean concentration of total mercury was 73.8 nmol/L. This level is about twice the level of 40 nmol/L found during the Québec Health Inuit Survey among women aged 18 to 25 years from Ungava Bay. A similar difference between cord blood Hg level and that of the mother has been reported previously. The concentration of mercury in cord blood was assessed in a fishing population of the Faroe Islands (Granjean *et al.* 1992). The median concentration was 121 nmol/L (n=997).

Selenium

The levels measured in the first 8 cord blood samples are particularly high (mean concentration equal to 4.2 μmol/L), when compared to the median concentration of 1.4 μmol/L (n= 1020) reported in the Faroe Island study (Granjean *et al.* 1992). The mean selenium concentration in

fetal blood was 3.1 μmol/L in West Greenland and 1.86 μmol/L in East Greenland, with a fetal/maternal concentration ratio of 0.88 (Hansen 1988). Our preliminary results show a fetal/maternal ratio of 2.2. These surprisingly elevated concentrations of selenium in cord blood must be confirmed by further sample analyses. Plasma analyses may be preferable to whole blood analyses for selenium determination.

Organochlorines

Mean PCB concentration measured in whole cord plasma was 2.6 μg/L (range 1.0-3.8 μg/L). These relatively low levels compared to the concentration reported for the adult population (35 μg/L) could be due to:

- the low lipid content of fetal plasma (50% lower than that of the mother)
- the relatively low concentration of organochlorines in the plasma of young women from Ungava Bay ($\approx$ 8 μg/L).

Accordingly, in reports to come, results will be presented on a lipid basis. The PCB congener pattern observed in cord blood samples is similar to that of adults. The congeners with the highest neurotoxic potential (IUPAC 28, 52) were only detected in two samples. For pesticides, aldrin, α- and δ-chlordane, dieldrin and heptachlor epoxide were never detected (detection limit 0.1-0.02 μg/L).

ONGOING WORK

In 1993/94 we will continue this program in the Kativik region. NWT and Labrador results will probably be available during 1993. It will then be possible to evaluate the spatial trends of human contamination throughout the Canadian Arctic.

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HEALTH RISK ASSESSMENT FOR NEWBORNS EXPOSED TO ORGANOCHLORINE COMPOUNDS THROUGH BREAST-FEEDING

PROJECT LEADER: P. Ayotte, Public Health Centre (Québec region), Centre Hospitalier de l'Université Laval (CHUL)

PROJECT TEAM: P. Ayotte and É. Dewailly, Centre Hospitalier de l'Université Laval

OBJECTIVES

1. To estimate the intake of contaminants for Inuit newborns through breast-feeding.

2. To conduct a health risk assessment associated with this exposure.

3. To provide an exposure assessment for the cohort of Inuit newborns followed prospectively to monitor their development and identify early symptoms of toxicity.

DESCRIPTION

The Community Health Center of the "Centre Hospitalier de l'Université Laval" has been involved since 1988 in the health assessment of an Inuit population exposed to various xenobiotics through their country foods. In a recent study, mothers breast-feeding their babies born between July 1989 and July 1990 were asked to submit a milk sample for organochlorine compound analysis. Results for 105 breast milk analyses showed high PCB levels, as previously reported for this Inuit population living in Arctic Québec (Dewailly *et al.* 1989). The mean concentration was 2.9 mg/kg lipid, a level five-fold greater than that measured in breast milk samples of women living in the southern part of the province (0.52 mg/kg lipid) (Dewailly *et al.* 1991). For chlorinated pesticides, milk levels found among Inuit women were between three to five times greater than those of the reference population. Differences between the two groups were less important for PCDD and PCDF levels (Dewailly *et al.* 1991). Mean daily intakes of the various organochlorine contaminants measured in breast milk, were calculated for a typical newborn drinking 120 ml of milk/kg-body weight. Reference doses proposed by the United States Environmental Protection Agency (USEPA 1992) are exceeded during the breast-feeding period for chlordane (3.7x), dieldrin (2.8x), DDE (8x), heptachlor epoxide (6.8x), PCBs (10x), PCDD/PCDF (80.6x) and coplanar PCBs (145x).

It could be concluded on the basis of this preliminary risk assessment that exposure to contaminants found in breast milk may pose a significant health risk for the newborns of this Inuit population. However, this crude risk analysis is not satisfactory, since risk estimates must be weighed against the benefits of this practice for the infant's health, which results in decreased postneonatal morbidity. Hence we initiated a more elaborate risk assessment using a toxicokinetic model, Monte-Carlo simulations and various toxicological endpoints.

ACTIVITIES 1992/93

We focused on the most toxic group of substances present in the breast milk: the planar halogenated polycyclic aromatic hydrocarbons (PHPAH), which includes dioxins, furans and non-ortho coplanar PCBs. Information on the PHPAH content of breast milk was available for 30 mothers. Toxic equivalency factors developed for PHPAHs (Safe 1990) were used to convert each PHPAH concentration into 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) toxic equivalents (TEQs). Mean PHPAH concentration in breast milk was 39 ng TEQ/kg lipid (range: 20-108). Mean body weight at birth of the 30 newborns was 3.36 kg (range: 2.26-4.22); 18 were males and 12 females. The average breast-feeding period was 39 weeks (range: 4-60).

Using the measured breast milk levels and data on milk consumption by newborns, we wished to estimate the PHPAH concentrations in the liver, the adipose tissues and in the whole body of the newborn at different times during breast-feeding. To realize this goal, we used a toxicokinetic model developed by Carrier (1991). This physiologically-based toxicokinetic model developed for PHPAHs takes into account the non linear elimination kinetics of these compounds observed in various species. This is the result of liver enzyme induction produced by the binding of PHPAH to the Ah receptor. The increased synthesis of cytochrome P450IA2, which displays a strong affinity for TCDD, is apparently responsible for the increased capacity of the liver to sequester dioxin-like substances seen as the body burden increases. Liver uptake is saturable and is described mathematically by a Michaelis-Menten equation developed for receptor-driven biological effects. The model was modified to accommodate varying exposure doses through time. This feature was needed in the present study, since PHPAH doses received by the newborn will vary according to: 1) the breast milk concentration, which declines as breast-feeding goes on, breast-milk secretion being an effective route of elimination for PHPAH; 2) the quantity of milk consumed by the newborn which varies with age and sex; 3) the body weight increase of the newborn with age. In order to take into account the variability of the various parameters entering into the model, a stochastic treatment was used with the help of Monte-Carlo simulations, when solving the equation system. A total of 300 simulations were performed.

RESULTS

Results of the 300 simulations conducted from birth to age two follows a similar trend in all three compartments: the concentration rises rapidly during the first few months of breast feeding and then reaches a plateau around 12-13 months. The concentrations observed in the various compartments at this time are presented in Table 1. The mean liver and adipose concentrations predicted are about the same, while the mean whole body concentration is four times smaller than those calculated in fat and liver compartments. The 95 percentile concentration in the liver is 1.7 times greater than that in the adipose tissues, reflecting the disproportionate accumulation of PHPAH in the liver with increasing body burden (Carrier 1991).

Table 1. Mean and 95 percentile concentrations in liver, adipose tissue and whole body, estimated following 300 simulations of newborn Inuit body concentrations during breast-feeding.

Body compartment	Mean concentration (ng TEQ/kg)	95%-ile concentration (ng TEQ/kg)
Liver	144	435
Adipose tissue	136	250
Whole body	36	75

In order to put into perspective the exposure during the breast-feeding period, additional simulations were performed up to age 50. These were conducted using the same model and assuming a constant exposure of 7.7 pg/kg/day following weaning. Figures 1 and 2 present the variation of whole body concentration with age for two extreme cases selected out of the 30. Figure 1 presents the case of a newborn with a small PHPAH exposure through breast-feeding (both low PHPAH concentration in milk and short breast-feeding duration), while figure 2 illustrates the data for a highly exposed newborn. While in the first case, PHPAH exposure through breast-feeding does not significantly influence the body burden variation with age, PHPAH exposure through breast-feeding alters noticeably the PHPAH body burden in a highly exposed newborn. In the latter case, the body concentration reaches 80 ng TEQ/kg following breast-feeding and decreases after weaning to attain at age 30 a level similar to that observed for the individual who received a low dose through breast-feeding.

DISCUSSION/CONCLUSIONS/FUTURE DIRECTIONS

Studies on cancer and reproduction in rats identified a No Observed Adverse Effects Level (NOAEL) of 1 ng/kg/day for TCDD, which translates according to the model of Carrier (1991) into a whole body concentration equal to 77 ng/kg. According to our simulations, the vast majority of Inuit newborn (more than 95%) would not be expected to accumulate a body concentration exceeding 77 ng/kg. In extreme cases, where both the PHPAH milk concentration is elevated and the breast-feeding duration is long, temporary excursion over the NOAEL might be encountered (see Fig. 1). However, this would occur only during a few months in early childhood. Levels encountered thereafter would decrease to reach by age 30 the mean body concentration expected in the Inuit population (15 ng/kg). In as much as the risk in the animal studies was a function of the area under the body concentration/time curve and that this holds true in humans, even in these extreme cases, the chronic exposure dose would be below those associated with chronic health effects that occurred in laboratory animals.

Recently, Mably *et al.* (1992) reported adverse effects on the reproductive function of the adult male rat induced by *in utero* and lactational exposure to TCDD. A single 64 ng/kg dose to the mother during gestation was sufficient to depress by 40% the sperm content of the cauda epididymis compared to controls. The mean body concentration in Inuit women during child-

bearing age was estimated at 15 ng TEQ/kg in the present study, a level 4 times smaller than the 64 ng/kg Lowest Observed Adverse Effects Level (LOAEL). In our simulations, the adult dose was assumed to be constant (7.7 pg/kg/day). The highest concentration in milk lipids among the 30 Inuit women was 108 ng TEQ/kg, which would correspond approximately to a 25 ng/kg body concentration. This concentration is again lower than the LOAEL, but the margin of safety is very small if at all present.

In addition to inducing carcinogenic effects and adverse effects on reproduction and development, immunotoxicity has been observed in all animal species tested with TCDD, including primates (NRC, 1992). We recently observed in a cohort study of Inuit newborns followed up to age two that those "highly" exposed to organochlorines through breast-feeding (newborns breast-fed with a milk containing an organochlorine concentration greater than the median concentration for the group) had a higher risk of contracting otitis at 12 months than the newborns who were never breast-fed (Dewailly *et al.* manuscript in preparation). A lower CD4/CD8 T-cell ratio was observed at six and 12 months for the high-exposure group (PCB concentration in mother's milk greater than the median value) compared to the group never breast-fed. PHPAHs are known to alter cellular immunity, with a reduction of the CD4/CD8 T-cell ratio being a main feature. Although differences were modest and globally the CD4/CD8 ratios were within a normal range, the apparent negative association between organochlorine exposure and CD4/CD8 deserve further analysis. In order to refine the exposure variable, the newborn body concentration at the time of the clinical investigation will be estimated using the model developed in the present work. This may reduce the misclassification of exposure and facilitate the identification of immune system dysfunction possibly associated with organochlorine exposure in the Inuit newborn cohort. Other simulations will also be conducted for other PCBs and chlorinated pesticides.

Expected project completion date: July 31, 1993

Partners: G. Carrier (Université de Montréal): toxicokinetic modelling; J.-P. Trépanier (Ministère de l'Environnement du Québec): exposure modelling and computer simulations.

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Figure 1. Body burden and body concentration of PHPAH (TEQs) in an Inuit newborn receiving a low dose through breast-feeding

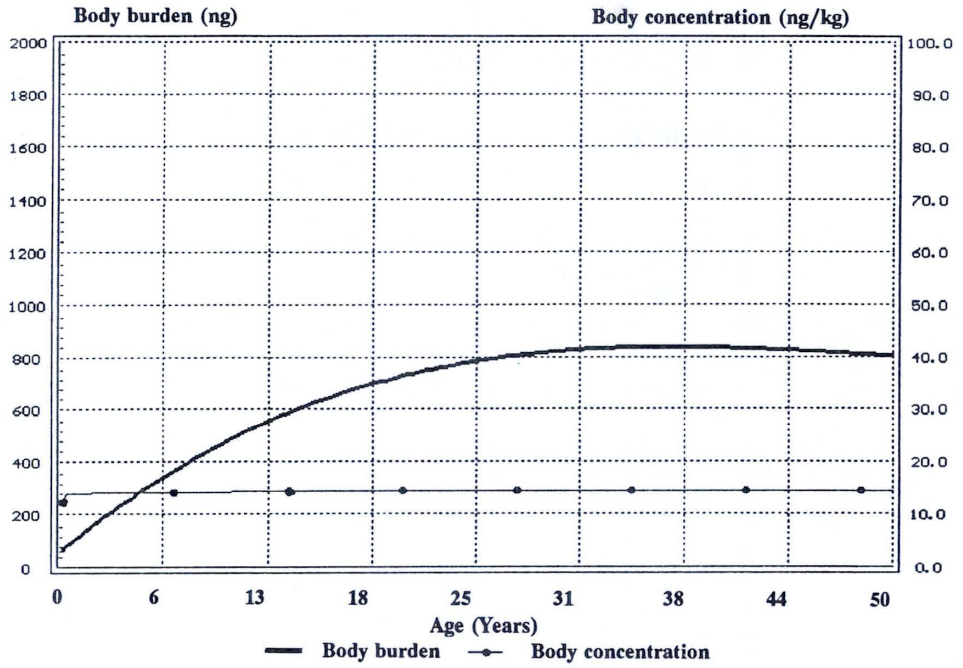
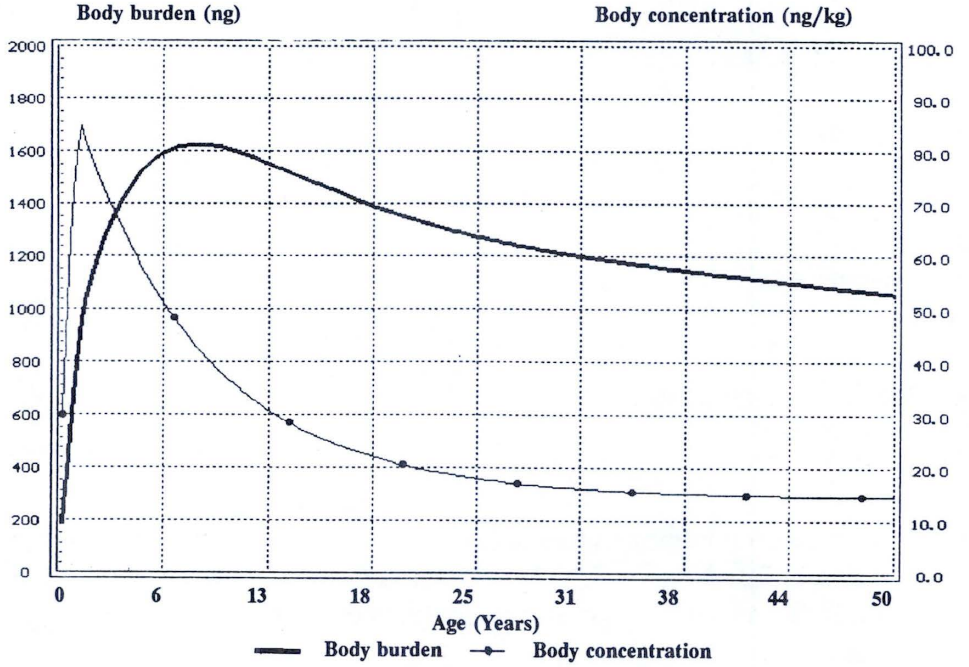


Figure 2. Body burden and body concentration of PHPAH (TEQs) in an Inuit newborn "highly" exposed through breast-feeding



HUMAN CONTAMINANT TRENDS IN ARCTIC CANADA

PROJECT LEADER: J.B. Walker, Health Dept., Government of Northwest Territories

PROJECT TEAM: R. Nuttall, Mackenzie and Kitikmeot Health Regions
R. Allen, Baffin Health Region
J. Christensen, Mackenzie Health Region

OBJECTIVES

1. To assess the exposure of NWT northern neonates and their mothers to selected organochlorine, heavy metal and radionuclide contaminants in at least 2 regions, by expanding existing human health baseline monitoring.

2. To develop preliminary methodologies for assessing variance in contaminant intake in selected NWT populations.

RESULTS

Phase I - Regional Consultations

Due to the nature of contaminant monitoring in humans, extensive and ongoing consultations are necessary. Activities in this project during the period under review in the Mackenzie and Kitikmeot Regions (Fig. 1) consisted principally of community consultation, as well as protocol development.

These consultation activities covered the following communities and groups:

Mackenzie Region

a) Community Meetings

Lac La Martre
Rae Edzo
Rae Lakes
Fort Liard
Fort Resolution
Fort Simpson

Meetings in these communities were held with Health Centre staff and with members of the Hamlet and/or Band Council. The problem of contaminants in the Arctic ecosystem was discussed, as well as our specific plans for measuring and monitoring contaminants in the Region.

b) Medical Staff - Stanton Yellowknife Hospital

Since much of the sampling will take place at this Regional Hospital, the Medical Staff was briefed on our contaminant activities, and subsequently the Medical Staff approved the sampling program we are planning to undertake at the hospital.

c) Nursing Conference

At the two annual nursing conferences for Health Centre nurses in Mackenzie Region the issue of contaminants was formally presented, and our sampling and monitoring plans were discussed. A need was identified for an in-service workshop for health workers who will be directly involved in the project, before sampling begins.

Kitikmeot Region

a) Community Meetings

Holman
Coppermine
Cambridge Bay
Pelly Bay

Meetings in these communities were held with Health Centre staff and with members of the Hamlet Council to discuss the contaminant issue and program.

b) Regional Workshop

A Contaminants Workshop for the Kitikmeot Region was scheduled for March 1993 to be held in Pelly Bay. Due to adverse weather conditions, it was necessary to reschedule the workshop for June in Taloyoak. The workshop will focus on members of the Kitikmeot Health Board and the Kitikmeot Inuit Association. The workshop, while tailored for the Kitikmeot Region, will parallel a similar workshop previously held in Hay River.

A one and a half day Northern Health Consultation Workshop was held in Yellowknife in January 1993, to build on issues identified at the Environmental Contaminants Workshop, Hay River, March 1992. The Health workshop was attended by representatives from the Inuit Tapirisat of Canada, Dene Nation, Metis Nation-NWT, Inuvialuit Regional Corporation, Health and Welfare Canada, Mackenzie and Kitikmeot Health Regions, and Government of Northwest Territories (GNWT) Health. Consultation strategies were developed and reviewed to further address the importance of community consultation and networking. General contaminants-related information, activities and initiatives were also addressed, including definition and extent of invasiveness, types of research, community education needs, and the importance of linking various monitoring activities.

Translation of Materials (re: Broughton Island PCB Studies) into Inuktitut

In the Baffin Health Region, consultation activities have focused mainly on fulfilling requests from the Baffin Regional Health Board and residents of Broughton Island for an Inuktitut translation of information about the Broughton Island Study.

Two separate items (as described below) were translated. These items will be for distribution to: a) members of the Baffin Regional Health Board, b) identified individuals and groups in the community of Broughton Island, and c) Baffin Regional Inuit Association.

Translated Items:

1. The Hansard (titled: Consideration of the Matter of Arctic Contaminants) from the NWT Legislative Assembly on March 10, 1989.

   The discussion that occurred in the Legislative Assembly addressed (in lay person's terminology) the issues and questions re: PCB and related nutritional studies in Broughton Island. Because the terminology and concepts were expressed for the lay person, the Hansard proved to be informative and easy to translate.

2. Summary of the results of PCB and related nutritional studies. The summary was titled: "Inuit Traditional Foods and Diet: Dietary Studies in Broughton Island - 1985-1988".

   The four page summary was drafted by the Baffin Regional Medical Health Officer. It was based on a review of study results and previous summaries. The draft was submitted to the Community Health Department (Yellowknife) and others, where recommendations were made concerning wording and format. The summary was then translated.

Early in 1993/94 these translated items will be presented to the Baffin Regional Health Board, and subsequently taken to Broughton Island for distribution and discussion. The President of the Baffin Regional Inuit Association, and representatives from the Baffin Regional Health Board and GNWT Health will be travelling to Broughton Island to participate in these discussions.

Phase II - Methodological Development

As part of the Yellowknife Bay Study, Mackenzie Health Region is developing a methodology to determine variance in food use in Dettah and N'Dilo. These two communities have raised concern about contaminants in fish and water, and information about their food use is important to help answer their questions. While a larger food consumption proposal had been submitted by GNWT Health for 92/93, funding was delayed, and subsequently not received. It was agreed, however, that the Yellowknife Bay food consumption work should begin, and that subsequent similar work would benefit from this project. Methodologies from similar studies were reviewed; initial contact made and meetings held with Chiefs of Dettah and N'dilo, other community representatives, the Dene Nation and the Metis Nation-NWT. Discussion was also

initiated about collaborating with others involved in related work, particularly McGill University's Centre for Nutrition and Environment of Indigenous People.

Additionally, a human milk literature review was completed, focusing on NWT northerners, and it is anticipated that collaboration with Health and Welfare Canada, and others, may occur in 1993/94 in this area.

UTILIZATION OF RESULTS

In the Mackenzie and Kitikmeot regions, consultation will continue and will lead into monitoring activities once protocol details are finalized. Work plans are in the process of being completed. The work in these regions will benefit other regions in the NWT as their contaminants-related activities continue to evolve.

Expected project completion date: March 31, 1997

Figure 1.

NORTHWEST TERRITORIES HEALTH CARE SYSTEM

REGIONAL HEALTH BOARDS/HOSPITALS

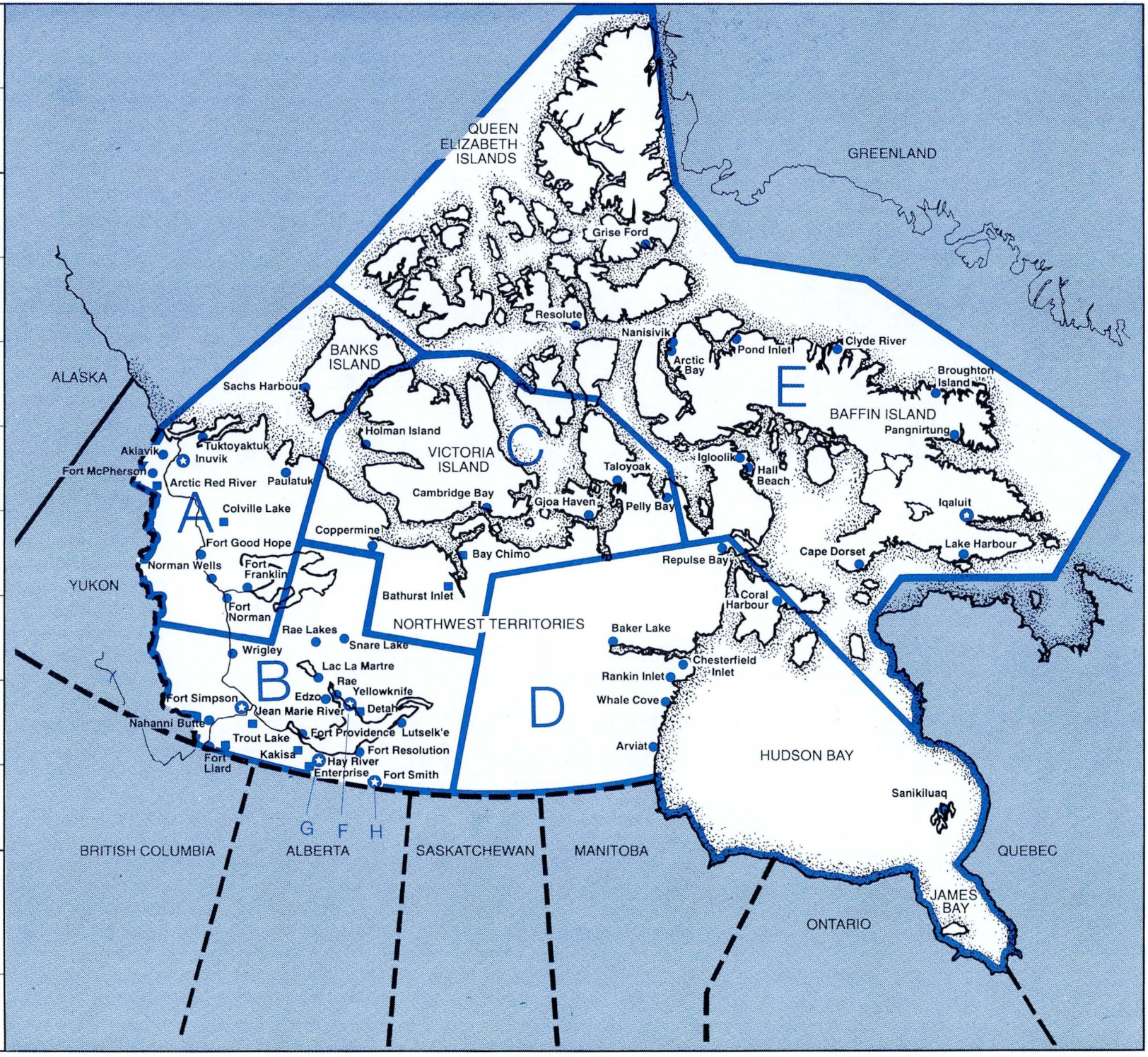
A	INUVIK REGIONAL HEALTH BOARD
B**	MACKENZIE REGIONAL HEALTH SERVICE
C	KITIKMEOT HEALTH BOARD
D	KEEWATIN REGIONAL HEALTH BOARD
E	BAFFIN REGIONAL HEALTH BOARD
F*	STANTON YELLOWKNIFE HOSPITAL
G	H.H. WILLIAMS MEMORIAL HOSPITAL
H	FORT SMITH HEALTH CENTRE

* Stanton Yellowknife Hospital is the Western Arctic referral centre A,B,C,F,G,H,

** Hay River Public Health Services are provided by Mackenzie Regional Health Services

*** Boundary lines reflect the area served by Regional Hospital/Health Boards

- HOSPITALS
- HEALTH CENTRES
- HEALTH STATIONS



HUMAN CONTAMINANT TRENDS IN THE NORTHWEST TERRITORIES

PROJECT LEADER: A. Gilman, Health Canada

OBJECTIVES

1. To provide timely and up-to-date information and advice to the Government of the Northwest Territories (GNWT) and Northerners on multimedia exposure trends, toxicological research, epidemiology studies, human biomarkers, tissue sampling/storage and consultation strategies.

2. To ensure international Arctic initiatives adequately address human health issues and are informed of current human impacts associated with exposure to environmental contaminants.

3. To ensure well coordinated "health" initiatives under the Green Plan/Arctic Environmental Strategy and the Health Canada Action Plan for Health and Environment Arctic Initiative.

ACTIVITIES/RESULTS

Northern Health Consultation Workshop (Objective #1)

Prepared (under contract) two resource documents for workshop participants: a) "Types of Health-based projects", and b) "Generic consultation strategies" and convened stakeholder workshop in Yellowknife.

Other activities with GNWT (Objective #1)

Met with GNWT staff in June 1992 to further discuss program activities, including planning for consultation workshop. Participated in GNWT Health Policy Committee meeting, GNWT Technical Committee meeting January 1993.

AMAP Human Health Program (Objective #2)

Led Canadian delegation to AMAP Human Health meeting, Nuuk, Greenland. Prepared document on AMAP Human Health Monitoring Program (jointly with Denmark). Attended follow-up meeting of Canadian AMAP program (in Toronto). The Human Health Monitoring Document has established international priorities and essential elements in carrying out monitoring of human levels of environmental contaminants and research on health effects associated with environmental contaminants in the Arctic.

AES Management (Objective #3)

Participated in AES Technical Committee meetings for project reviews (Yellowknife, March 1993). Participated in Science Managers Committee Meeting (Ottawa, October 1992; Whitehorse, April 1993).

NHW Arctic Initiative Coordinator (Objective #3)

Formed group and participated in 3 coordination meetings of Health Canada staff in Medical Services Branch, Food Directorate and Environment Health Directorate.

UTILIZATION OF RESULTS

Results of the Northern Health Consultation Workshop are measured in clarification of the roles of science and stakeholders in community based research, reduced public fear and increased trust.

Expected project completion date: These activities are complete but represent segments of ongoing program activities. Other elements are inserted to meet emerging needs. Coordination work and work on AMAP will be ongoing.

CENTRE FOR NUTRITION AND THE ENVIRONMENT OF INDIGENOUS PEOPLES

PROJECT LEADER: Harriet V. Kuhnlein, McGill University

PROJECT TEAM:

T.A. Johns, L. Chan, O. Receveur (McGill University); B. Erasmus (Dene Nation);
R. Kuptana, M. Demmer (Inuit Tapirisat of Canada); M. Grey, C. Reimer (Inuit Circumpolar Conference); J. Gingell, E. Schultz (Council for Yukon Indians);
B. Carpenter (Métis Nation, NWT); R. Jock, A. Kewayosh (Assembly of First Nations);
R. Soueida, J. Yousif (McGill University).

OBJECTIVES AND ACTIVITIES

1. The most commanding objectives and activities regard start-up activities of the Centre.

   a) The Governing Board of the Centre was created by the Dene Nation, Inuit Tapirisat of Canada, Inuit Circumpolar Conference, Council for Yukon Indians, Métis Nation (NWT) and the Assembly of First Nations. Representatives from the host community (Mohawk Council of Kahnawake) and the Arctic Environmental Strategy (AES) (E. Berthelet, NWT Health Department) also contribute to the Board. The Board works with the CINE Executive and staff and various levels of McGill Administration. In 1992/93, the Board and the CINE executive met 16 times (three times in person and 13 times by tele-conference calls).

   The role of the Governing Board is to provide direction to the Centre in establishing and achieving its objectives, priorities and plans and to provide the major link between the Centre and the aboriginal communities it serves. In this capacity, the Governing Board provides support to various activities of the Centre; approves budgets, workplans and reports; addresses community involvement and information in research programs; and assists in the development of CINE communications.

   b) Terms of reference for the Board and an overview of CINE activities were created. A mission statement for the Centre was developed as follows: "In concert with Indigenous Peoples, CINE will undertake community-based research and education related to traditional food systems. The empirical knowledge of the environment inherent in indigenous societies will be incorporated in all of its efforts." Board discussions on CINE scope, and community participation processes have been held. A Centre statement for brochures was developed, as well as a logo and stationery design.

c) The building renovation, planning and execution proceeded throughout the year, with start of demolition January 1, 1993. Completion of the renovations, anticipated for summer 1993, will yield a 6000 ft.^{2} facility incorporating meeting and reception areas, staff offices, an equipped multi-functional laboratory, and classroom space. Equipment was ordered.

d) Staff recruitment was a major time commitment for CINE. An administrative assistant (Angela Deer) with several years experience in the executive offices of Dominion Bridge in Montreal was hired. Jafer Yousif was recruited as a half-time statistical/computer assistant; Jafer is a Ph.D. student in the Department of Statistics at the University of Montreal. Rula Soueida remains on staff with her wealth of experience in dietary data management and traditional food sampling and analysis. Two other research assistants are being recruited, and several student assistants have been doing various tasks on an hourly basis.

   CINE staff and Board Members spent quite a lot of time recruiting and interviewing 6 candidates for two CINE professorial positions. Dr. Laurie Chan has accepted one of the positions, coming from a postdoctoral position at the Dept. of Pathology, Faculty of Medicine, at the University of Western Ontario. Laurie is a zoologist and marine biologist with experience in the metabolism and toxicity of environmental contaminants and the effects of trace element nutritional deficiencies. He has done research with cadmium, mercury, copper, lead, and zinc in various species, including humans. Laurie has an impressive array of laboratory expertise in metal and organochlorine analyses. Dr. Olivier Receveur also accepted our offer. He was previously with the Dept. of Nutritional Sciences at the University of California at Berkeley, where he has completed his Ph. D.; also from Berkeley, he has an MPH in Epidemiology and Biostatistics and a B.S. in Clinical Dietetics. Olivier has spent several years in cross-cultural work, and speaks 5 languages; he has research experience with various anthropometric, socioeconomic, dietary and health indices and health risk assessment from Kenya, Egypt, Mexico and with migrant Mexican-American farmworkers in California.

e) Extensive communications have taken place with the Aboriginal organizations in their networks. Media requests from several avenues have been honoured (radio, newspaper, TV). Visits to both Arctic College and Yukon College have been made to begin discussions on an agreement for staff exchange and potential activities in the college campuses. Communications have also begun in the fund-raising offices of McGill to identify additional funds to augment the AES contribution agreement for educational activities.

2. CINE developed a project proposal for evaluating variations in traditional food use in 28 Dene/Métis communities. This project will be implemented in 1993/94. In addition, staff have been oriented to, and have continued data management from earlier work in native communities. The Governing Board has provided direction for consideration of additional research activities.

3. Substantial time has been spent on the process for institutional recognition of CINE as a McGill Centre. Documents and presentations were made to (1) The Faculty of Agricultural and Environmental Sciences, (2) the subcommittee on Centres of the Executive Committee of the Faculty of Graduate Studies and Research, (3) the Executive Committee of the Faculty of Graduate Studies and Research, and (4) the Graduate Faculty Council. The process will be continued in the next year. Links with interested McGill staff and other researchers in Canada have been initiated to develop the interdisciplinary focus of CINE on topics of traditional food systems, nutrition and the environment of Indigenous Peoples.

4. The results of the creation of CINE have been useful for presentations and discussions in several contexts: (1) Canadian Polar Commission round tables on (a) contaminants in the North, and (b) health and social issues; (2) the Dene Nations Chiefs' Assembly; (3) the NHRDP review panel on Aboriginal diabetes; (4) the Cree Trappers Association Annual Meeting; (5) proposals submitted to the Eco-Research program of the Tri-Council program for universities; (6) to Quebec Minister of Indian Affairs, Christos Sirros; and (7) to the Royal Commission on Aboriginal Peoples Round Table on Health and Social Issues.

EDUCATION, COMMUNICATIONS AND INTERNATIONAL INITIATIVES

PROMOTING RESPONSIBLE RESEARCH IN INUIT COMMUNITIES: PRIVACY, CONSENT AND OTHER LEGAL AND ETHICAL ISSUES RELATING TO RESEARCH

PROJECT LEADER: M. Demmer, Inuit Tapirisat of Canada

PROJECT TEAM: M. Demmer and E.L. Oscapella (contractor)

DESCRIPTION

Members of Inuit communities have expressed concern about their lack of control over research conducted in their communities. A report was commissioned by the Inuit Tapirisat of Canada (ITC) which outlines relevant legal and ethical rules relating to research so that Inuit communities will better understand their rights when dealing with researchers. The report also suggests strategies for enhancing control over research in Northern communities.

The report sets out several definitions of "research". It concludes that there is no universally accepted definition, and that the Inuit are largely free to adopt as broad a definition as they wish.

The report then touches on the complex web of laws and ethical guidelines that protect the privacy of individuals from intrusions by others, including researchers. These include international law, constitutional privacy protections such as the Charter of Rights, privacy (data protection) legislation, other legislation containing confidentiality provisions, codes of professional conduct and research guidelines set by universities and research bodies. It explains that, together, these can offer substantial privacy protection by limiting how information about individuals can be collected, used and disclosed.

The report notes, however, that these protections are often watered down by exceptions. For example, provincial legislation often permits hospitals to disclose patient information to researchers, even without the consent of the patient. Other legislation may allow personal information kept by various organizations to be disclosed to researchers without the individual's consent.

In other situations information about a person may be used by researchers only if the individual consents. One concern of the ITC is that individuals are not being given adequate information with which to decide whether to consent to information about them being used for research.

Several organizations have set out what they consider to be the minimum conditions for obtaining informed consent. These include the Inuit Circumpolar Conference, the Association of Canadian Universities for Northern Studies, the Medical Research Council of Canada, the Canada Council, the University of Toronto, and the Council for International Organizations of Medical Sciences. The report reproduces sections of these guidelines as examples of how informed consent requirements could be structured.

The report also touches on other ethical duties that arise in research. These include the following:

to do research that benefits individuals or communities;

to report research results to the research subjects;

to minimize harm to communities that may flow from doing research and releasing results;

to involve community members in the design and ethical review of research programs; and

to be sensitive to the wishes of different cultures.

The report concludes that the law does offer some protection to individual privacy, but that gaps remain in the law and in some ethical guidelines. These leave the Inuit vulnerable to exploitation. The report recommends several measures to reduce the chances of being exploited:

a) Educating about rights. The Inuit, like others in Canada, likely know little about their legal and ethical rights to privacy and the extent of their rights to control research involving them. Even researchers may not be fully aware of their obligations. Education about these issues - both of the Inuit and researchers - is important.

b) Designing appropriate consent forms. Some current forms designed to secure the consent of individuals give little information about the proposed research. Those who consent may do so with incomplete understanding of the research. To remedy this, the ITC could design "generic" consent forms that identify the detailed categories of information for researchers to provide when looking for research subjects. Researchers would be responsible for filling in the specific information required. The report describes in detail the categories of information that should go into such forms.

c) Communicating concerns to potential researchers. Some researchers simply may not be aware of the concerns of the Inuit about intrusive research. The ITC could prepare a document explaining these concerns. It could circulate the document to universities and other research bodies and funding institutions.

d) Obtaining copies of research guidelines applying to specific researchers. Researchers are often subject to ethical guidelines. Inuit communities would benefit from knowing what guidelines the researchers must follow. The community can then verify that the research guidelines, which often address issues such as privacy and consent, are being followed, or hold the researchers accountable if they are not.

e) Developing comprehensive research guidelines for the North. Existing guidelines for research in the North are a good first step, but they are somewhat general. Comprehensive guidelines are needed.

f) Pressing for changes to the law. Some laws permit access to personal information for research without the consent of the individual. Some of these laws may be too broad in light of today's changing perceptions about the value of privacy. The ITC could consider pressing governments for amendments.

g) Using the law for remedies. In some cases, researchers may violate ethical and even legal obligations. In extraordinary situations, where they show open disregard for the law, affected individuals or communities might consider legal action. This could range from civil suits to seeking prosecutions for violations of medical records legislation. However, legal action should be taken only in extraordinary situations, to avoid discouraging other researchers from doing necessary research in Inuit communities.

INFORMATION AND EDUCATION ON CONTAMINANTS IN THE YUKON

PROJECT LEADER: Yukon Technical Committee on Contaminants in Northern Ecosystems and Native Diets (Contact: Mark Palmer, Chair)

PROJECT TEAM:
1) Yukon College
2) Environment Canada
3) Council for Yukon Indians
4) Health and Welfare Canada
5) Indian and Northern Affairs Canada

OBJECTIVES

Short-term:

To make technical information on contaminants available to Yukoners through a library collection and through fact sheets.

Long-term:

1. To "demystify" contaminants issues by providing information relevant to the Yukon in an accessible form.

2. To improve communications and local decision-making on contaminants issues in the Yukon.

DESCRIPTION/RESULTS

Local decision-making and local control of communications on contaminants issues have been identified as important goals of the Yukon component of the Contaminants Program of the Arctic Environmental Strategy. It is critical that decision-makers, those involved in communications, and the general public, become better informed about contaminants and issues surrounding contaminants.

It is currently very difficult for Yukoners to obtain scientific information on contaminants. No public library in the Yukon can provide information on contaminants, and information in government libraries and offices is not easily accessible. There are few publications available which provide information on contaminants and related issues in the Yukon in non-technical language.

The Yukon Contaminants Committee in cooperation with the AES Waste Clean-up Program identified four contaminants or groups of contaminants that were of most concern to local residents. The contaminants to be addressed under this project were:

1. **PCBs** (due to elevated levels found in fish from southern Yukon lakes and identification of several sites with contaminated soils).

2. **DDT** (due to elevated levels in southern Yukon lakes and the widespread use during the construction of the Alaska highway for pest control).

3. **Other Organochlorines** (due to the detection of many organochlorines, such as PCBs, DDT and toxaphene, in the fish and wildlife of the north as well as in other components of the environment).

4. **Petroleum Products** (a large percentage of the waste sites in the Yukon are comprised of abandoned fuel drums. This along with local spills make this an important issue).

It should be noted that the Committee is already working on two fact sheets on its own, for toxaphene and zinc.

Over 200 articles from scientific journals, etc. and 40-50 books were catalogued and placed in the Yukon College library. This information is now available for general use.

Draft copies of the fact sheets have been completed and are being reviewed by the Yukon Contaminants Committee. The final versions should be ready for publication shortly.

UTILIZATION OF RESULTS/DISCUSSION/CONCLUSION

The Yukon Contaminants Committee has received positive feedback and support for this program. Yukon residents including First Nations and educational institutions are eager to get the completed fact sheets. Copies have also been requested from various territorial/provincial and federal departments.

CYI will coordinate the reproduction/translation of the fact sheets for the individual First Nations in the Yukon.

Additional contaminants of concern have already been identified through public consultation efforts (such as cadmium and trace metals in general).

Expected project completion date: It is anticipated that this will be an ongoing project for the duration of the Arctic Environmental Strategy. New fact sheets will be produced each year addressing emerging concerns.

THE UNITED NATIONS ECONOMIC COMMISSION FOR EUROPE TASK FORCE ON PERSISTENT ORGANIC POLLUTANTS

Persistent organic compounds with a high potential chronic toxicity (such as PCBs, dioxins and toxaphene) have been found to occur in surprisingly elevated concentrations in biota at the top of the Arctic food chain, particularly in many of the favoured dietary items of northern indigenous people. Some of these substances (such as PCBs) have been found in blood and breast milk from Inuit residents of Broughton Island (N.W.T.) and northern Quebec at concentrations which exceed levels recommended by National Health and Welfare. Similarly, in the Yukon, it has been necessary for Territorial health officials to issue dietary health advisories concerning the consumption of some species of fish due to tissue concentrations of the pesticide toxaphene. The situation has led to general concern relating to human health and ecosystem implications.

Many of these chemicals are prohibited (or have never been widely used) in Canada (e.g. toxaphene). They are carried to the Arctic primarily in the atmosphere, after being released to the environment at mid and low latitudes. Therefore a key component of Canada's strategy to deal with this issue has been to pursue the development of international controls on emissions and discharges to the environment.

To achieve our goal, we have focused on the Convention on Long-Range Transboundary Air Pollution (LRTAP), administered under the United Nations Economic Commission for Europe (UN-ECE). This instrument is attractive on two counts. First, the UN-ECE membership includes all east and west European nations, together with the United States, Russia and Canada. It therefore captures a significant proportion of the northern hemisphere's users and manufacturers of persistent organics. Second, the Convention has developed a proven track record in achieving consensus and action in the challenging arena of acid rain generating pollutants within the fifty-member countries.

Following proposals from Canada, the UN-ECE established a special Task Force in November 1991 to report on the magnitude of the persistent organic pollutant problem. The Task Force (co-chaired by Canada and Sweden), subsequently prepared a report stating that the situation is of sufficient concern that development of the elements of a protocol for the control of these substances should be immediately initiated under the LRTAP Convention. In November 1991, the Executive Body to the LRTAP Convention responded by modifying the mandate of the Task Force to provide by 1994, the basis for elements of a possible protocol. At a meeting hosted by Canada in May 1992, the Task Force agreed upon a Work Plan to achieve this goal.

The key components of the Work Plan are as follows:

1. preparation of a schedule of priority substances to be covered by the protocol (led by the United Kingdom);

2. establishment of emission and discharge inventories upon which compliance with a protocol would be based (led by Canada and the United States);

3. development of monitoring and modelling capability to evaluate the effectiveness of proposed control strategies (led by Sweden);

4. evaluation and recommendations relating to optimal control technologies and strategies for principle pollutant sources:

   a) stationary industry (led by Germany and the Netherlands)
   b) transportation (led by the United States)
   c) pesticides (led by Canada and Germany); and

5. preparation of a report for the Executive Body to the LRTAP Convention in November 1994, to provide the basis for a protocol (led by Canada and Sweden).

Germany hosted the third meeting of the Task Force (May 16-20, 1993) at which time progress on the five Work Plan components was evaluated. Adjustments were made to the Work Plan to ensure that all supporting work is completed before the week of December 13, 1993, when a small group of experts led by Canada and Sweden will prepare a preliminary draft final report from the Task Force to the Executive Body to the LRTAP Convention. The final report containing the basic elements for a protocol will be approved at a meeting in Den Hague, February 21-25, 1994, after which it will be submitted to the UN-ECE Secretariat for consideration by the Executive Body in November 1994. At that time, it will be decided whether or not to begin the preparation of a legal protocol to control persistent organics under the LRTAP Convention.

Progress has been rapid. Much of the momentum can be attributed to the Arctic Environmental Protection Strategy (AEPS) adopted by ministers from the eight Arctic countries in Rovaniemi (Finland) in June 1991, and which included a strong commitment to support the work of the UN-ECE Task Force. At their second meeting in Nuuk, Greenland on September 16, 1993, the ministers agreed to support the development of a protocol under the UN-ECE LRTAP Convention.

ARCTIC MONITORING AND ASSESSMENT PROGRAMME

At Rovaniemi, Finland in June 1991, ministers from all eight Arctic countries signed a declaration adopting the Arctic Environmental Protection Strategy (AEPS). The AEPS sets out the shared environmental objectives of the eight countries, and identifies the principles which should guide their implementation. It reviews the nature of Arctic environmental problems, assesses the existing institutional and legislative arrangements for dealing with them, and finally, identifies specific actions which should be implemented. One of these actions commits the eight governments to establish a comprehensive Arctic Monitoring and Assessment Programme (AMAP).

The mandate of AMAP is to monitor the levels of, and assess the effects of, anthropogenic pollutants in all components of the Arctic environment. It was envisaged that AMAP should as far as possible be built upon existing national and international programs, which are to be developed into a circumpolar framework. An initial priority is to be given to persistent organic pollutants, selected heavy metals, and radionuclides.

To implement AMAP, a Task Force has been established comprised of delegates from each of the participating nations, observing countries (Germany, Netherlands, and the United Kingdom), and organizations of indigenous people (Inuit Circumpolar Conference, Nordic Saami Council and the Russian Association of Small Peoples of the North). Finland has been elected to Chair the Task Force, and Canada to Vice-chair. Norway has provided a Secretariat, based in Oslo.

The first meeting of the Task Force was held in December 1991 (at Tromso, Norway) where a 1992 work plan was established. Under the plan, lead countries took responsibility to develop detailed sub-programmes as follows; Atmosphere (Canada), Marine Environment (Norway), Terrestrial Environment (Sweden), Freshwater (Finland), Human Health (Denmark/Greenland) and Remote Sensing and Modelling (United States).

At the second meeting of the Task Force, (hosted by Canada in Toronto, November 30-December 4, 1992), the draft plan for AMAP (as represented by the sum of the sub-programmes) was approved in principle for implementation in 1993. Countries have now submitted to the Secretariat their detailed national implementation plans and AMAP has entered into the data collection phase.

Coordination of Canada's implementation plan is being achieved by using the interdepartmental and intergovernmental Technical and Science Managers Committees established to coordinate delivery of the Arctic Environmental Strategy (Green Plan). Implementation is being greatly facilitated by the fact that the goals and objectives of AMAP are directly compatible with those of the Arctic Environmental Strategy (AES).

Canada is assisting Russia in meeting its commitments to the AMAP (e.g. the atmospheric station at Dunay Island). Canadian and Russian officials have agreed to delay the renegotiation of the northern contaminants component of the Canada-Russia Program on Scientific and Technical Cooperation in the Arctic and the North until October 1993. This will enable both

countries to carefully evaluate the 1993 AMAP implementation plans, in order to determine the most critical activities in need of additional support.

Output from AMAP will take the form of comprehensive "State of the Arctic Environment" reports for the eight circumpolar ministers, the first of which will be produced in 1996/97. However, at the ministerial meeting in Nuuk, Greenland (September 16, 1993), AMAP provided an interim presentation on emerging priority issues relating to pollution of the Arctic. The report includes an appraisal of new information dealing with persistent organics, heavy metals, radioactivity, Arctic haze, Arctic stratospheric ozone depletion, climate change, and the incorporation of traditional indigenous knowledge into AMAP activities. The report pays particular attention to the rapidly accumulating information concerning the nuclear waste dumping practices of the former Soviet Union in the Arctic. At Nuuk, Ministers reaffirmed their commitment to AMAP and strengthened the institutional arrangements made to address radionuclide contamination.