Health Parameters of Three Terrestrial Wildlife Species Harvested by Innu Hunters in Labrador, Canada, in Relation to Tissue Concentrations of Environmental Contaminants

A Thesis

Submitted to the Graduate Faculty

in Partial Fulfilment of the Requirements

for the Degree of

Master of Science

in the Department of Pathology and Microbiology

Faculty of Veterinary Medicine

University of Prince Edward Island

Charlottetown, P.E.I.

December, 2004

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Your file Votre référence
ISBN: 0-494-10352-3
Our file Notre référence
ISBN: 0-494-10352-3

SIGNATURE PAGE(S)

Not numbered in thesis

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ABSTRACT

Hunting, eating and sharing of wildlife for subsistence purposes continue to be important activities in many northern Canadian communities, including the Innu communities of Sheshatshit and Utshimassit (now Natuashish) in Labrador. The Innu are First Nations people formerly known as Naskapi-Montagnais. There are 13 Innu communities in Québec and Labrador. The Innu Nation is the governing body that represents the collective interests of the approximately 1,700 Innu residents of Sheshatshit and Utshimassit.

It has been shown that environmental contaminants such as metals and organochlorines (OC) have the potential to cause a wide range of negative health effects in numerous animal species. Efforts have been made to measure these and other contaminants in a variety of wild animals in northern Canada, however, a knowledge gap remains concerning the biological effects that these contaminants have on free-ranging wildlife species. Overall, little attention has been focused on contaminants in terrestrial animals of Labrador and the related concerns of the Innu communities.

The general purpose of this study was to assess, in collaboration with Innu hunters and their families, the health of important wildlife species harvested by Innu hunters in Labrador in relation to tissue concentrations of environmental contaminants measured in the animals.

Specific objectives included: identifying, through harvest surveys, wildlife species important to the community members; assessing the overall health of each species by measuring selected health parameters in animals killed during regular seasonal hunts; determining tissue concentrations of a variety of common environmental contaminants in the animals; and assessing the relationship between contaminant concentrations and health parameters for each species, using multiple regression analyses.

The focus of the thesis was on the levels and potential health effects of contaminants in wildlife species. The risks of consumption of these animals to people are not addressed here, although the data collected could be used by human health researchers to contribute towards a risk assessment of country food consumption for the Labrador Innu communities.

Based in part on the results of the harvest surveys, four species were chosen for the subsequent objectives of the thesis: caribou, porcupine, Canada goose and black bear. Because of limited access to carcasses of black bears during the study period, work was confined to the remaining three species.

Most caribou, porcupines and Canada geese examined appeared to be healthy, based on an assessment of their fat reserves and gross and, to a lesser extent, microscopic examination of their internal organs. Furthermore, populations of helminth parasites and concentrations of metal and OC contaminants measured in these animals did not appear to negatively affect their health.

Caribou and porcupines had very low concentrations of most contaminants in their tissues. As expected, cadmium levels were elevated in kidneys of older

caribou (geometric mean: 6.5 µg/g wet weight; range [1.5 - 44.0]). Elevated cadmium levels and decreased selenium levels in caribou were associated with an increased abundance of *Fascioloides magna* (large American liver fluke); however, this association may, in part, have been related to age.

Porcupines killed in the spring were thinner than those killed in the fall and had a greater abundance of intestinal nematodes. Although some associations were found between contaminant concentrations and health parameters in these animals, it is unlikely that these associations are biologically significant as metal and OC contaminants were generally near or below minimum detection limits.

Concentrations of contaminants found in Canada geese were generally similar to those reported in other studies and were generally lower than those found in other waterfowl species (geometric means in fat for some OCs were: sumDDT, 277 ng/g wet weight; dieldrin, 21.4 ng/g wet weight; and sumPCB,17.6 ng/g wet weight). Higher levels of most contaminants, particularly OCs, were found in geese collected in the spring as compared to those collected in the fall. This seasonal difference in contaminant levels was likely due to a number of related factors including: higher amounts of contaminants in the overwintering areas, differences in age distribution between seasons, and distribution and dynamics of lipid stores in the birds. Although some positive associations between parasitological parameters and OC concentrations were found, results were inconsistent, and there is insufficient evidence to suggest a causal relationship.

In general, environmental contaminants do not appear to be affecting the health of caribou, porcupines and Canada geese in Labrador, however, several aspects of this study may warrant further investigation. For caribou, further studies would benefit from focusing on older animals, which tend to have higher levels of cadmium and greater prevalence and abundance of *F. magna.* For Canada geese, further investigations should be targeted at spring geese. Health effects from contaminant exposure would be more likely to occur in the spring or summer, when organochlorine contaminants are at higher levels and fat reserves containing OCs are being mobilized. For porcupines, the health impacts of the high intestinal parasite burdens found in these animals may warrant further investigation.

Over the long term, the collection and examination of animals with observed abnormalities during the regular seasonal hunt would lead to a greater understanding of the health of these and other species hunted by Innu people in Labrador. This type of long-term collaboration could help to further address the concerns expressed by Innu people regarding the overall health of the land and animals in a manner that would be relevant and meaningful to all participants.

ACKNOWLEDGMENTS

I would like to gratefully recognize the funding for the research provided through the Toxic Substances Research Initiative and Northern Ecosystems Initiative as well as the financial support I received in the form of a Fellowship from the Canadian Institutes of Health Research.

I am also very grateful to and would like to acknowledge and thank the participating elders, hunters and family members for their many invaluable contributions to the study.
In Sheshatshit: Sebastien Benuen, Gregory Mark, Joseph Mark, Kenny Rich, Frances Rich, Christine Rich, Greg Rich, Lorraine Rich, Peter Penashue, Mary Ann Penashue, Louie Penashue, Mary-Adele Penashue, Joseph Penahue, Basile Penashue, Angela Penashue, Jimmy Rich, Etienne Pastiwet, Joseph Rich, John Selma, Mina Selma, Andy Budden, Annette Budden, Calvin Hurley, Anniett Nuna, Apenam Pone, Margaret Rich, Simeo Rich, Patrick Andrew, Terry Andrew, Roger Blowes, Catherine Penashue, George Nuna, Kathleen Nuna, Philomena Abraham, Sebastien Penunsi, Philomena Penunsi, Joseph Pokue, Elizabeth Penashue, Francis Penashue, Simon Michel, Mary Ann Michel, Sebastien Pastichi, Ben Michel, Janet Michel, Michel Jack.
In Utshimassit: David Nui, John Nui, Joachim Nui Sr., Jim Nui, Clarence Nui, Mark Nui, Daniel Poker, Edward Mistinapeo, William Katshinak, Charles James Pasteen, John Murphy and Sebastien Piwas.

I would also like to thank Innu Nation past presidents Peter Penashue and David Nuke for their support of the study as well as current and former Innu Nation staff members, in particular Sebastien Piwas, David Nui, Jim Nui, Damien Benuen, Sam Mistenapeo, George Gregoire and Kevin Head in Utshimassit and Catherine Penashue, Nancy Nuna, Etien Andrew, Jack Selma, Guy Playfair and Jay Forsyth in Sheshatshit for their assistance with many logistical aspects of the study in the communities. Many thanks also to Etien Pone and John-Baptiste Malek for giving me a glimpse of the living history of *nutshimit*.

Many thanks to Larry Innes for his work and support in the initial stages of the study as well as periodically throughout the last 4 years - even from afar.

I am most grateful to and appreciative of co-researcher Basile Penashue for his trust, his patience with my countless questions and his dedication to the study from the outset. Many thanks also to Angela and their children for their tolerating his many absences during sample collection, *Tshinaskamitin.*

At the AVC, I would like to thank my supervisors Drs. Pierre-Yves Daoust and John Vanleeuwen for being open-minded and supportive in their guidance, and my supervisory committee for their direction and patience: Drs. Basile Ikede, Andy Tasker and Scott McBurney. Thanks also to Scott McBurney and Pierre-

Yves Daoust for their assistance with sample collection in the field and to Neil Burgess with Environment Canada for his input and expertise.

Thanks to Dr. Gary Conboy for the parasite identifications, to Greg Dobbin for his tremendous help in the lab and beyond, to Melissa Mrosiak for help with the caribou samples, to Diagnostic Services Staff including Jim Carlsen, Leonard Doucette, to the Diagnostic Toxicology Laboratory and to Henrik Stryhn for help with statistical analysis.

I am also indebted to Andrea Bourque, Laura Rogers and Maria Forzan for their friendship throughout the years.

Finally and most importantly, thanks to Scott for his tremendous and unwavering support and to Connor and Ian for taking it all in stride.

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DISCLAIMER

Toxicological analyses of samples collected for this study were completed either through the Diagnostic Toxicology Laboratory at the Atlantic Veterinary College or under contract at two laboratories outside the Atlantic Veterinary College. Analyses for several organochlorine compounds including pesticides and PCBs were completed at the Environmental Quality Laboratory, Environment Canada, Moncton, New Brunswick, Canada, and analyses for dioxins, furans and coplanar PCBs were completed at Axys Analytical Services Ltd., Sidney, British Columbia, Canada.

1. GENERAL INTRODUCTION

1.1 Introduction

Contaminants such as metals, organochlorines and radionuclides in northern ecosystems have been the focus of recent research programs in Canada and worldwide (AMAP, 2002; CACAR II, 2003). Compounds that are entirely anthropogenic, such as organochlorine pesticides and polychlorinated biphenyls (PCBs), are present in relatively pristine northern environments, often geographically distant from the site where these contaminants are produced or utilized. In part due to anthropogenic activities, tissue concentrations of naturally occurring metals, such as mercury and cadmium, have been found in various fish and wildlife species at relatively elevated levels (Braune, Muir et al., 1999). It has been shown experimentally that contaminants such as metals and organochlorines have the potential to cause a wide range of negative health effects in numerous animal species (Muir et al., 1997). Efforts have been made to measure these and other environmental contaminants in a variety of wildlife such as caribou and waterfowl in the North (eg. Elkin and Bethke, 1995; Langlois and Langis, 1995; Braune, Malone et al., 1999), however, a knowledge gap remains concerning the potential biological effects of these contaminants in free-ranging species (AMAP, 2002; CACAR II, 2003).

Aboriginal peoples in the North have an interest in contaminants that stems from their intimate relationship with the environment. Hunting, eating and sharing of wildlife for subsistence purposes continue to be important activities in many northern communities (Gilman et al., 1997). The consumption of wildlife,

or country foods, is an important route of exposure to environmental contaminants for these people (Gilman et al., 1997). Thus, there is concern regarding the possible health effects in people who are exposed to these contaminants.

The potential impact of these compounds on the health of wildlife is also a concern of Aboriginal Northerners[1]. Questions have been raised regarding possible links between environmental contaminants and health problems in species commonly hunted in the North (eg. Pellerin and Grondin, 1998). Concern for the health of animals and the environment has been expressed by residents of the Innu communities of Sheshatshit and Utshimassit (Davis Inlet) (now Natuashish) in Labrador (see map, Figure 1.1), particularly in relation to environmental issues associated with mining (Innu Nation, 1996), hydrological developments (Innu Nation, 2000) and environmental contaminants (Innes, 1998).

The Innu are First Nations people formally referred to as Naskapi-Montagnais. *Nitassinan* (the Innu homeland) encompasses a large portion of the Québec-Labrador peninsula (Figure 1.1). Today, eleven communities on the North Shore of the St. Lawrence River in Québec and two communities in Labrador are home to approximately 15,000 Innu people. The Innu Nation

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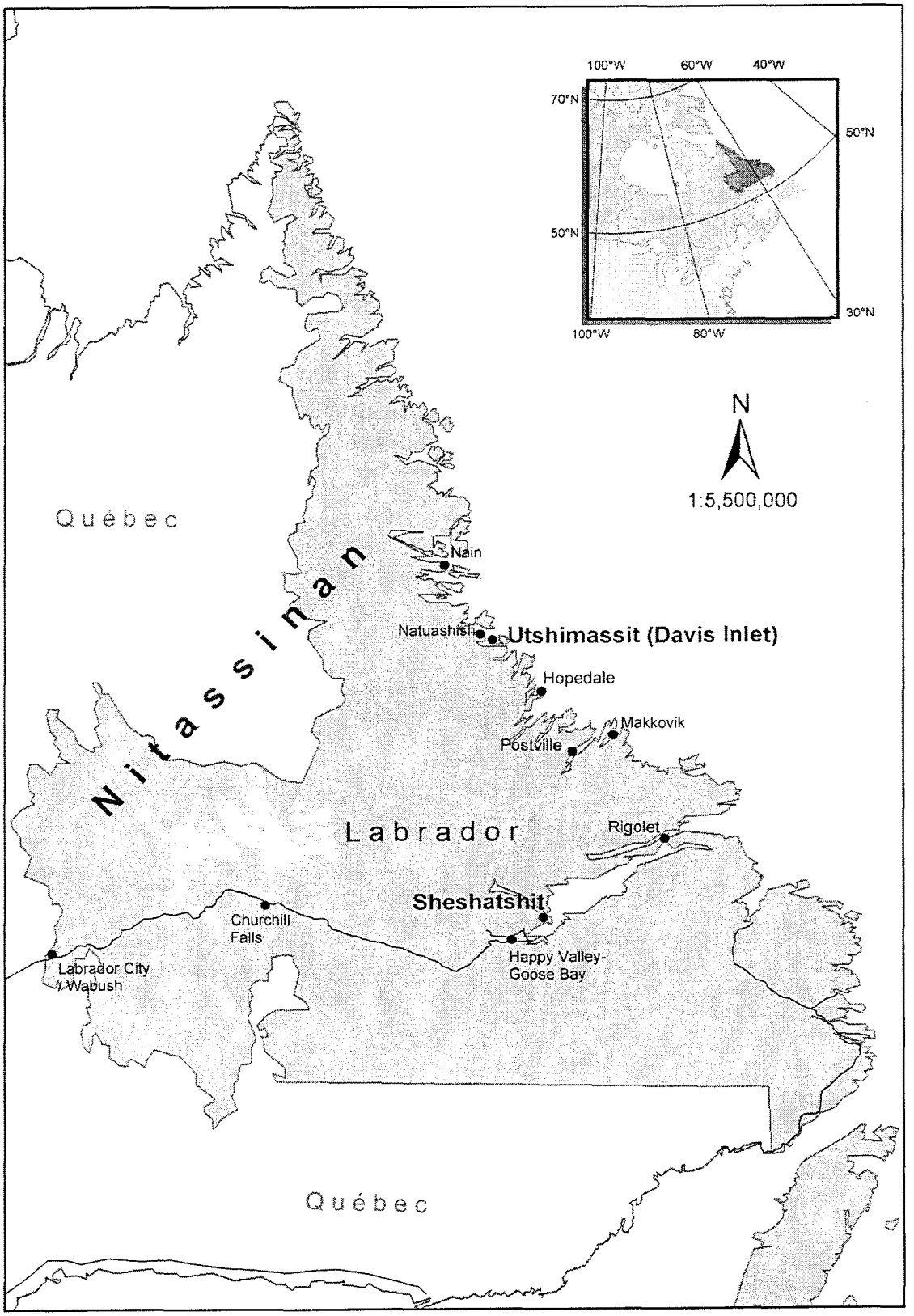
1 For the purposes of this thesis, Aboriginal Northerners refer to the descendents of original inhabitants of Northern Canada including First Nations and Inuit peoples. In Labrador this includes both Innu and Inuit peoples. Inuit and Innu are culturally distinct groups. Innu are referred to in more detail in the text. Inuit are descended from prehistoric Thule. In Labrador there are approximately 4,000 Inuit most of whom reside in five communities on the North coast: Nain, Hopedale, Postville, Makkovik and Rigolet. Inuit in Labrador are represented by the Labrador Inuit Association.

Figure 1.1 Map of Labrador



represents the collective interests and rights of Innu residents of Sheshatshit (approximately 1100 people) and Natuashish (approximately 700 people) (formerly Utshimassit). The language of the Innu people, Innu-aimun, is spoken by most, if not all Innu in Sheshatshit and Natuashish.

In addition to the nutritional and economic value of hunting and eating country foods, there are important cultural, social and spiritual dimensions to the relationship between many Aboriginal Northerners and the animals that share their environment. For Innu people of Labrador, hunting and all of the activities that surround it are "important and distinctive features of the Innu identity" (Innes, 1998). This intimate relationship with the land and animals places the issue of environmental contaminants in the North into a cultural context that cannot be ignored. Collaboration between aboriginal communities and outside scientists can help to ensure that community concerns, as well as scientific goals, are addressed in studies on the effects of environmental contaminants.

Although, in the last decade, a considerable amount has been learned about environmental contaminants in northern Canada through directed studies, little attention has been focused on terrestrial animals in Labrador and the concerns of the Innu communities.

1.2 Background information

In 1997, a pilot study was initiated by the Innu Nation, in collaboration with researchers from Atlantic Veterinary College, that was designed to assess the health and environmental contaminant levels of mammals and birds harvested by Innu hunters in remote camps in Labrador (Innes, 1998). This Innu Nation

Country Food Contaminant Study arose in part as a response to concerns expressed by a number of Innu elders and hunters regarding physiological changes in individual animals and populations of various species (Innes, 1998). Elsewhere in northern Canada and other circumpolar countries, a series of scientific studies was being conducted relating to the presence of environmental contaminants in northern ecosystems (CACAR, 1997; AMAP, 1998). Although data were accumulating with regard to environmental contaminant levels within ecosystems across northern Canada, little was known about their potential effects on the health of terrestrial wildlife in Labrador, and the safety and quality of country food in that region (Innes, 1998). Innu harvesters have also raised concerns regarding the health of local wildlife populations in relation to large-scale industrial developments that have taken place in traditional hunting areas, such as low-level military flights, hydroelectric projects and mining (Innes, 1998).

Drawing heavily on the information presented in the report by Innes (1998) the results and conclusions of this pilot study are outlined below. In 2001, a report based on the results of this pilot study was written in plain language for distribution to the Innu participants in the study and is provided in Appendix C. Both the original report (Innes, 1998) and the plain language summary interpreted the contaminant levels based on recommended guidelines for consumption of contaminants for people.

Sampling of animals harvested by Innu hunters was carried out by project researchers and Innu co-researchers and took place at four locations. In spring 1997, eight different species (four brook trout, three northern pike, three spruce

grouse, two surf scoters, one black scoter, one Canada goose, one muskrat and one porcupine) were sampled at two outpost camps. In October and November 1997, tissues from nine caribou harvested in western Labrador were sampled. Tissue samples were analysed for metals and persistent organic pollutants.

High levels of organochlorine and metal contaminants were found in some animals, however, the small number of samples analysed did not permit general conclusions to be drawn about the range and distribution of contaminant levels in the general population of wildlife in the Innu territory. Two key areas of focus for future studies were identified: "the quantification of contaminant levels and the assessment of spatial variation and trends in key species within the Innu harvesting area" and "the quantification of the Innu subsistence harvest and consumption patterns for both communities" (Innes, 1998).

The specific recommendations arising from the study included: identification of the range and distribution of mercury levels in fish throughout Labrador; quantification of organochlorine contaminants in migratory waterfowl such as Canada geese and scoters; identification of baseline contaminant data in all species important to Innu people; and for caribou, consideration of radionuclide contaminants and comparative analysis of contaminants in samples from other herds (Red Wine and Mealy Mountain). This thesis addresses a number of these recommendations.

1.3 Objectives of the thesis

The general purpose of this thesis was to assess, in collaboration with Innu hunters and their families, the health of important wildlife species harvested by

Innu hunters in Labrador in relation to tissue levels of environmental contaminants in the animals. This was accomplished through four specific objectives.

The first objective was to identify, through harvest surveys, which wildlife species were important, in terms of the edible weight of country food, to the community members of Sheshatshit and Utshimassit, Labrador. Based on the results of the harvest surveys, as well as historical harvest survey data and consultative meetings with community members regarding their experiences with wildlife, four species were chosen for study in relation to the subsequent objectives of the thesis.

The second objective was to assess, through collaboration with hunters from each community, the overall health of each of the four species by measuring selected health parameters in animals killed during regular seasonal hunts.

The third objective was to determine, in the four species, the tissue levels of a variety of common environmental contaminants.

The final objective was to assess the relationship between contaminant levels and health parameters for each species using multiple regression analysis techniques.

The focus of the thesis was on the levels and potential health effects of contaminants in wildlife species. The risks of consumption of these animals to people are not addressed here, although the data collected could be used by

human health researchers to contribute towards a risk assessment of country food consumption for the Labrador Innu communities.

1.4 Structure of the thesis

The first chapter of the thesis serves as a general introduction to the issue of environmental contaminants in northern ecosystems. The results of a pilot study conducted in the year prior to the initiation of the present research are presented. In a review of the relevant literature, particular reference is given to the perspective of Aboriginal people in northern communities and the potential health effects of environmental contaminants in various free-ranging northern wildlife species.

In the second chapter, the methods and results of harvest surveys are presented, as well as some discussion relating to the manner in which the species were subsequently chosen for detailed health and contaminant study. The following three chapters are devoted to the results of the research conducted on the health and contaminant levels in three of the species selected for detailed study: caribou, porcupine and Canada goose. Each of these three chapters is organized in a similar manner.

The final chapter is an overall discussion of the results of the research in relation to the scientific context, as well as a consideration of their implications at the regional and community levels.

1.5 Literature review

The following literature review presents an overview of the issue of environmental contaminants in free-ranging wildlife species including some

considerations of local Aboriginal perspectives. A brief overview of health parameters in wildlife follows and the remainder of the literature review summarizes the relevant literature relating to the known biological effects of exposure to metals, organochlorine pesticides and PCBs in free-ranging avian and mammalian wildlife with particular emphasis on northern species. A wide range of biological effects have been reported in wildlife, many, but not all of which are presented here. Some of the effects discussed were not addressed in the thesis (eg. reproductive effects) but are presented here to demonstrate the variety of effects reported in the literature.

1.5.1 Aboriginal perspective

The intimate relationship between many Aboriginal Northerners and the environment is well recognized. Time spent in the country hunting wildlife is important for many Aboriginal people and affirms both traditional and present day cultural values. Although much has changed in recent years for Aboriginal peoples with the introduction of European society, and its different social, economic and spiritual structures, for many Aboriginal Northerners, this intimacy with the environment remains.

Benefits of hunting, eating and sharing country foods identified by these people are numerous and span many different aspects of individual, family and community health. When asked about benefits of country foods, Inuit from Nunavik (northern Québec) and Labrador affirmed their central role in all aspects of well-being and in the maintenance of social and cultural values (Furgal et al., 1999). Some examples of the listed benefits of hunting and consuming country

foods included: a good source of nutrition; important economically; important for physical activity; help to reinforce bonds within and between communities; traditional importance; important to passing on knowledge to younger generations; and value as medicine. The identification of environmental contaminants in northern areas has sparked a flurry of research activity across the North, and they have now been identified in all ecological compartments, including air, snow, water, soil, many different animal species, as well as people (CACAR, 1997). Although the levels in the North are, in general, much lower than in most southern areas, some contaminants have been shown to accumulate in certain northern species to levels that may be injurious to their health (AMAP, 2002).

In addition to identifying levels, pathways and trends of these contaminants in northern ecosystems, more recent research efforts have included: identifying biological effects in wildlife and people; improving communication of research findings to the residents of these regions; and influencing policy decisions regarding the use of environmental contaminants at the national and international levels (AMAP, 2002; CACARII, 2003).

The potential impacts of environmental contaminants on northern ecosystems are varied. For example, the concerns over the potential health risks to people from exposure to contaminants through the consumption of country foods, must be measured against the potential health risks (at the individual and community levels) of avoiding country foods and the many benefits

they provide. Currently, there is little certainty regarding the health effects of eating, or choosing not to eat, country foods (CACARII, 2003).

Although ongoing research programs in the North have documented tissue contaminant levels in a wide variety of wildlife species, in some cases at elevated levels, an important knowledge gap remains concerning their biological effects in these animals at both the individual and population levels (CACARII, 2003).

Environmental issues that affect the integrity of ecosystems, such as those that may impact on wildlife health, strike at the core of the intimate relationship that many residents of northern communities share with the environment. Other local and global issues, such as local industrial developments and global climate change, are also worrisome to those with a close connection to the land. For example, community consultations by the Innu Nation in Sheshatshit and Utshimassit, Labrador, on mining activities and hydrological developments on Innu lands identified a number of concerns raised by Innu of all ages regarding the potential impacts of these activities on the land and animals in the area (Innu Nation, 1996, 2000).

Within environmental issues such as these, the interests of outside researchers and Aboriginal peoples in the North often overlap. Western scientists have long had an interest in various aspects of the biology and ecology of northern ecosystems and have been conducting studies on the animals and environment in the North for decades. In the past, typically, there was little acknowledgement by outside researchers of the perspectives or knowledge base of local aboriginal peoples. If this knowledge was recognized and reported, it

was rarely credited to Aboriginal peoples (Usher, 2000). It is only relatively recently that concepts such as Traditional Ecological Knowledge (TEK) and collaborative research practices have been acknowledged and incorporated into environmental studies in the North.

The scope of TEK is vast and reflects aboriginal individual and community knowledge. TEK has been defined as: "the body of knowledge, practice and belief, handed down through generations by cultural transmission, about the relationships of living beings (including humans) with one another and with their environment" (Berkes, 1998). Although the recognition and incorporation of TEK into traditional western science studies is gaining more widespread acceptance and has become a policy requirement in many areas of northern Canada (Usher, 2000), there has been considerable discussion surrounding its implementation (Wenzel, 1999; Usher 2000). In spite of the challenges in joining the two types of knowledge, there is little disagreement over its necessity.

Depending on the user group and the intended application, TEK may be abstracted and potentially manipulated as it is removed from the original framework (Kuhn and Duerden, 1996). In collecting and incorporating TEK, concerns of both Aboriginal peoples and non-Aboriginal researchers include its misuse and misinterpretation as it is extracted from the cultural context. Consequently, researchers have an ethical responsibility to insure that TEK is interpreted appropriately (Wenzel, 1999).

In addition to the recognition of TEK, increasingly, research conducted in the North is becoming more culturally relevant, consultative and collaborative.

The realization that issues that impact the environment cannot be considered in isolation of the local communities is becoming more widespread. In most areas of research, it is considered unethical to undertake studies in northern regions without consultation and consent of local Aboriginal communities (eg. Furgal et al., 1995; Innu Nation, 1999) and in many jurisdictions, such consultations are required. Collaborative research allows for the participation of community members in all aspects of the planning and implementation of studies so that results obtained are meaningful at both the scientific and local levels.

With regard to environmental contaminants (an issue with relevance to Aboriginal Northerners at many different levels) it is being increasingly recognized that local involvement in all aspects of research into this area is essential. Miscommunication and lack of involvement in the past have led to disastrous results for the communities affected and, in some cases, for the relationship between aboriginal communities and outside researchers (Usher et al., 1995). Within the context of northern communities, all perspectives relating to this issue (eg. ecological, cultural, social) are deeply interrelated and cannot be considered in isolation of each other. In the wider research community, this is being recognized, and efforts to concurrently address many different aspects of environmental contamination in northern areas are being made by research groups in Canada and elsewhere, with a large degree of local involvement (CACAR, 1997; AMAP, 1998, 2002; CACARII, 2003).

1.5.2 Effects of environmental contaminants on wildlife health

1.5.2.1 Overview of environmental contaminants

Environmental contaminants were identified in the North as early as the 1970s (CACAR, 1997). Since that time, compounds such as organochlorines, metals and radionuclides have been found throughout northern ecosystems, often geographically distant from the site where these contaminants are produced (AMAP, 2002; CACAR II, 2003). Many organochlorine compounds are globally ubiquitous. In general, they are very persistent and are recycled and redistributed throughout the globe primarily through long-range atmospheric transport. These compounds are lipophilic, readily transferred up the food chain, and will biomagnify in higher trophic levels.

Many organochlorine compounds have been restricted or banned in Canada and other northern countries due to their persistence, bioaccumulation, potential for long-range transport and adverse effects. These include industrial chemicals and by-products such as polychlorinated biphenyls (PCBs), polychlorinated dibenzo-*p*-dioxins (dioxins or PCDDs), polychlorinated dibenzofurans (furans or PCDFs) and hexachlorobenzene (HCB). A variety of organochlorine pesticides are also included: aldrin, dieldrin, endrin, chlordane, heptachlor, DDT, mirex, hexachlorocyclohexane (HCH), and toxaphene (AMAP, 2002). Most of these compounds have been banned from use or production in North America since the 1970s, however, due to their persistence and, in the case of PCBs their presence in old equipment, they are still circulating in the environment.

In marine ecosystems, predators such as polar bears (Norstrom et al., 1998), beluga whales (Muir et al., 1996) and sea otters (Bacon et al., 1998) have been shown to accumulate significantly elevated levels of these organochlorine contaminants which can vary with their geographical location and their feeding habits.

Both terrestrial and marine avian species have been shown to accumulate elevated levels of OCs. Seabirds that feed primarily by scavenging, such as great skuas and glaucous and great black-backed gulls, tend to accumulate high levels of a number of organochlorine contaminants, as do fish-eating birds such as kittiwakes (Henriksen et al., 1996; deMarch et al., 1998; Sagerup et al., 2000). In the past, due to elevated levels of DDE, birds of prey such as peregrine falcons and bald eagles have suffered population declines (Blus, 1996).

Another group of environmental contaminants that are of concern in northern ecosystems are metals. In contrast to the organochlorines, metals such as mercury, cadmium and lead are naturally occurring elements. These metals have no known biological function and can be toxic in small quantities. Mercury and cadmium are found naturally in rocks and sediments. However, human activities such as mining and smelting, and burning waste and fossil fuels, mobilize these metals making them available to be taken up by living organisms. These metals bioaccumulate, and in some cases, are present in northern ecosystems at high levels. Global lead emissions have decreased in recent years, largely due to bans on leaded gasoline in many countries, however, some

local sources, such as mines and lead shot used for hunting, may impact some local wildlife populations (eg. Hoffman et al., 2000).

Elevated mercury levels have been found in liver and kidneys of marine mammals such as ringed seals and beluga whales across northern Canada (Muir et al., 1997). Piscivorous birds (common loons and common mergansers) from eastern Canada (Scheuhammer et al., 1998) and predatory freshwater fish (eg. lake trout, northern pike) in many northern lakes (Muir et al., 1997) have also been found to accumulate high levels of mercury.

Elevated levels of cadmium have been found consistently in ungulate populations throughout North America (Scanlon et al., 1986; Brazil and Ferguson, 1989; Crête et al., 1989; Gamberg and Scheuhammer, 1994; Elkin and Bethke, 1995; Larter and Nagy, 2000), although regional and species differences exist. Concentrations in caribou kidneys are among the highest recorded in wildlife in North America (Muir et al., 1997).

Lead uptake and intoxication has been demonstrated in several free-ranging avian species in Canada including northern pintails, snow geese, Canada geese, and common loons (Daoust et al., 1998; Tsuji et al., 2002). The source of lead in these birds was believed to be ingestion of spent lead shots or of lead fishing sinkers rather than exposure from long-range atmospheric deposition.

Although potentially toxic on its own at high levels, selenium is also considered in many contaminant monitoring and research programs because it is essential in many enzymes and proteins and can reduce the toxic effects of

some metals such as mercury (Muir et al., 1997; Dietz et al., 1998). High levels of selenium are often found in wildlife species with elevated levels of mercury, particularly marine mammals such as polar bears and seals, and have also been demonstrated in common loons and common mergansers (Dietz et al., 1998; Scheuhammer et al., 1998). This element is thought to form toxicologically inert complexes with inorganic mercury (Scheuhammer et al., 1998).

1.5.2.2. Parameters of health in wildlife

Free-living wildlife species are exposed to a variety of naturally occurring stressors throughout their lifetime, and healthy animals have the ability to respond to these stresses and maintain health or homeostasis. Assessment of the overall health of wild mammals and birds is of interest to many investigators and a number of reviews of the various measurements and indices used in wildlife health evaluation have been published (eg. Kirkpatrick, 1980; Huot, 1988; Franzmann et al., 1995). Some of the measurements used for assessment of overall health can also act as biomarkers of exposure to environmental contaminants.

Biomarkers of exposure to environmental contaminants are measures of normal biological processes, or health parameters, that are altered with exposure to the contaminant of interest. Such biomarkers have been identified at all levels of biological organization, from the molecular to the population level (Hugget et al., 1992; deMarch et al., 1998). Biomarkers can be very specific for exposure to the contaminant of concern (eg. eggshell thinning from DDE exposure) or can be more general measures of individual or population health

which may also be influenced by factors other than contaminants (eg. reduced nutritional stores, reduced reproductive success, altered immunological response). Some biomarkers, such as mixed function oxidase response, have been well-studied in some free-ranging species (deMarch et al., 1998), however, in many cases, there are data gaps with regard to normal biological processes in wildlife.

Some examples of biomarkers used in studies of environmental contaminants include: alterations in immune response (biochemical or functional, ie. decreased resistance to infectious disease), developmental alterations, alterations in thyroid function, reproductive impairment, gross and histopathological alterations in various organs or tissues, impaired growth, loss of nutritional stores, genetic alterations and molecular and biochemical changes such as enzyme induction, hepatic porphyria and alterations in vitamin A (retinol) homeostasis (Hugget et al., 1992; Fox, 1993; deMarch et al., 1998).

Many of these potential biomarkers are also influenced by other factors. For example, a very commonly used measure of the overall health of an animal is an assessment of nutritional reserves, or body condition, through measurement of protein or fat reserves or both. In addition to exposure to contaminants, factors that can greatly influence an animal's body condition include, but are not limited to, infectious (viral, bacterial, parasitic) and non-infectious diseases, food availability, age, reproductive status and season (Franzmann et al., 1995).

A review of the literature has identified a number of specific biological effects of exposure to organochlorine and metal contaminants in free-ranging wildlife. The review is not exhaustive but focuses primarily on the more commonly reported effects at the individual or population level and will be addressed by contaminant type. Specific threshold tissue levels for some of these biological effects of exposure to environmental contaminants are presented in Chapter 6 General Discussion, Section 6.2.2.

1.5.2.3 Toxicological effects of exposure to organochlorines

The specific mechanisms by which the varied biological effects of environmental exposure to organochlorines (OCs) occur are incompletely understood. Several experimental studies have demonstrated that 2,3,7,8-TCDD (the most potent dioxin congener) can elicit a number of acute and chronic toxic responses including carcinogenic, teratogenic, reproductive and immune effects, as well as a number of biochemical alterations (Safe, 1990). Based on their ability to produce similar toxic effects, many OC compounds are recognized as being 'dioxin-like' (Webster and Commoner, 1994). Many of these toxic effects are thought to be produced through the same cytoplasmic aryl hydrocarbon (Ah) receptor-based mechanisms (Safe, 1990; Webster and Commoner, 1994). Some related compounds have only a weak affinity for the Ah receptor but are still toxicologically important because they are found at relatively high levels in the environment (Webster and Commoner, 1994).

It is apparent that TCDD and related compounds are capable of altering a variety of biochemical processes resulting in varied acute and chronic

toxicological effects (Webster and Commoner, 1994). Dose-response studies have shown that for many of the toxic effects, the only difference between these compounds is their relative toxic potency (Safe, 1990). One approach for assessing the potential hazard that a mixture of dioxin and dioxin-like compounds may pose is the use of 2,3,7,8-TCDD toxic equivalents (TCDD-TEQs) (eg. van den Berg et al., 1998).

The most commonly reported effects will be discussed in various categories relating to their biological systems. These categories are not mutually exclusive and considerable overlap exists between the types and likely mechanisms of action that produce these effects. Several biochemical alterations have also been associated with exposure to OCs, such as enzyme induction, vitamin A and thyroid hormone alterations and morphological changes in the thyroid gland, but will not be included in this review. Summaries of some of these biochemical effects in free-ranging wildlife have been recently published (eg. Rolland, 2000; Fox, 2001).

1.5.2.3.1 Endocrine disruption

The endocrine system has a tremendous influence on a variety of biological processes including growth, reproduction, metabolic processes, sexual attributes, and behaviour (Blood and Studdert, 1999). Biological organs and systems influenced by hormones, which are thus potential targets for endocrine disrupting contaminants, include: external and internal genitalia, brain, skeleton, thyroid, liver, kidney, and immune system (Colburn et al., 1993).

In a review of Great Lakes species, a number of biological effects were identified including: thyroid dysfunction in birds and fish; decreased fertility in birds, fish and mammals; gross birth deformities and decreased hatching success in birds, fish, and turtles; metabolic and behavioural abnormalities in birds; demasculinization and feminization of male fish and birds and defeminization and masculinization of female fish and birds; and compromised immune systems in birds and mammals (Colburn and Thayer, 2000). The list of species affected included nine avian species (bald eagle, black-crowned night heron, Caspian tern, common tern, double-crested cormorant, Forster's tern, herring gull, osprey and ring-billed gull), three mammalian species (mink, otter, beluga whale), two native species of fishes (lake trout and sauger) and one species of reptile (snapping turtle). It was determined that all of the abnormalities found could be mediated through the endocrine system (Colburn and Thayer, 2000).

Environmental endocrine disruptors have been defined as "any exogenous agent that causes adverse health effects in an intact organism, or its progeny, consequent to changes in endocrine function" (Gillesby and Zacharewski, 1998). Many chemicals that are found in the environment have been placed under the broad category of endocrine disruptors, including organochlorine contaminants, currently used pesticides and metals (Colburn et al., 1993; Guillette et al., 1995; Gray et al., 1998; Gillesby and Zacharewski, 1998; Colburn and Thayer, 2000).

While high doses can lead to overt toxic endpoints such as death, cancer and teratogenic effects (Colburn and Thayer, 2000), low doses can lead to

functional alterations of endocrine organ systems including the thyroid and urogenital systems (Guillette et al., 1995; Brinbaum, 1995; Gillesby and Zacharewski, 1998).

The potential effects of exposure to these compounds, particularly in developmentally sensitive embryos, are thought to be diverse, multisystemic, latent, and permanent (Colburn et al., 1993; Fry, 1995).

1.5.2.3.2 Developmental effects

Developmental effects are those that occur due to alterations in growth or differentiation of an organism (Blood and Studdert, 1999). There are a number of examples of developmental effects attributed to environmental exposure to OCs. Many of these have been documented in the Great Lakes region, particularly in colonial fish-eating birds where developmental abnormalities have been seen in nine species (Fox, 2001).

A classic example of developmental effects was discovered in the 1970s, when a series of anomalies seen in chicks from a variety of fish-eating avian species was identified and subsequently named the Great Lakes embryo mortality and edema syndrome (GLEMEDS) (Gilbertson et al., 1991). The suite of lesions was characterized by an elevated incidence of embryonic and chick mortality, growth retardation and various developmental abnormalities. Specifically, lesions included: bill deformities, club feet, missing eyes, crossed bills, defective feathering, as well as subcutaneous, pericardial and peritoneal edema and liver enlargement, necrosis and porphyria. The occurrence of these effects was strongly associated with exposure of the colonies to dioxin and

dioxin-like compounds expressed as TCDD-TEQs (Gilbertson et al., 1991; Fox, 2001). Their incidence, along with the degree of environmental contamination, has decreased dramatically since the 1970s (Gilbertson et al., 1991, Giesy et al., 1994). These types of effects have also been reproduced in experimental studies with exposure to coplanar PCBs in other avian species such as chicken, American kestrel, and common tern (Hoffman et al., 1998).

1.5.2.3.3 Reproductive effects

Although other environmental contaminants have also been implicated as causing reproductive effects, those due to exposure to OCs are among the best documented and understood. Reproductive effects in wildlife in general can be mediated at many different physiological levels both in adults and in embryos (Fry, 1995).

Specific effects on avian embryos resulting in reduced reproductive success include mortality or reduced hatchability and failure of chicks to thrive. Effects on adult birds include acute mortality, sub-lethal stress, reduced fertility, eggshell thinning, suppression of egg formation, and impaired incubation and chick-rearing behaviours (Fry, 1995).

A well-known example of OC contaminants causing reproductive impairment is the eggshell thinning effect of DDE in a variety of raptor and seabird species which led to widespread population declines in the 1950s and 1960s in North America (Lowe and Stendell, 1991; Bowerman et al., 1995; Fry, 1995; Blus et al., 1997; Fox, 2001). With the decline in use of DDT in the 1970s,

levels of DDT and its metabolites in the environment declined and many, but not all, affected populations have recovered dramatically (Donaldson et al., 1999).

More recent studies have also uncovered populations suffering from impaired reproduction associated with exposure to a variety of organochlorine contaminants. Various bird populations including bald eagles in the Great Lakes region (Giesy et al., 1994; Bowerman et al., 1995; Donaldson et al., 1999), lesser black-backed gulls in Finland (Hario et al., 2000), wood ducks in Arkansas wetlands (White and Seginak, 1994), double-crested cormorants in the Netherlands (Dirksen et al., 1995), and great blue herons in the Strait of Georgia, British Columbia (Fox, 2001) have been found to have negative reproductive effects of varying types and degrees associated with exposure to OCs, primarily PCBs and/or PCDDs and PCDFs in the environment. Over the last two decades, field and laboratory studies have shown the mechanisms of toxicity of OC contaminants on reproduction in birds to be quite diverse (Fry, 1995).

In mammals, many European otter populations in western Europe have shown dramatic declines and even extirpation in some areas. OCs, especially PCBs, have been implicated in both the original declines and poor success in repopulating these habitats. Recently, PCB levels have been monitored in the remaining populations by analysing fecal samples in populated areas and are still found to be relatively elevated in some of these areas (Mason and MacDonald, 1993a, 1993b, 1994; Leonards et al., 1997).

Negative reproductive effects have been seen in beluga whales that are exposed to a variety of environmental contaminants in the St. Lawrence River

system. These effects vary from a high incidence of mammary tumors in sexually mature females to reduced productivity in older females compared to similarly aged whales in less-contaminated areas (Béland et al., 1993).

1.5.2.3.4 Immunological effects

The immune system seems to be a particularly sensitive target to exposure to organochlorine contaminants such as PCBs, PCDDs and PCDFs (Muir et al., 1997). Evidence from both experimental and field studies have shown immunotoxic effects of these compounds in a variety of avian and mammalian species.

Reviews of the literature have shown that generalization of the immunotoxic effects of PCBs in thymus, spleen and lymph nodes cannot be made across species and therefore the following paragraphs describe studies conducted in various species.

In experimental animals, PCBs have been shown to affect both humoral and cell-mediated immune systems as well as the function of the mononuclear phagocytic lineage of cells (Kirkvliet, 1994; Tryphonas, 1995). Although these effects appear biologically relevant, because of redundancy and immunological reserve, measurable changes in immune function will not necessarily result in an increased susceptibility to disease in an organism (Kirkvliet, 1994).

Various studies in birds have shown both *in vivo* and *in vitro* immune effects. Suppression of T-cell-mediated immunity and thymic atrophy in herring gull and Caspian tern chicks in the Great Lakes surveyed from 1992-2000 was found to be associated with the level of *in ovo* exposure to PCBs (USDHHS,

2000; Fox, 2001). A recent study on black guillemot chicks in a PCB contaminated fjord in Labrador demonstrated no alterations in thyroid function, but severe suppression of T-cell mediated immune response and an increase in gonadal abnormalities related to PCB exposure (Kelly et al., 2000; Grasman et al., 2000). Experimental studies in various avian species have demonstrated splenic atrophy, atrophy of lymphoid tissues, thymic involution and edema and increased susceptibility of ducklings to hepatitis virus with exposure to varying mixtures of PCBs (USDHHS, 2000). Another recent experimental study in American kestrels showed somewhat conflicting results between the sexes. Adult females exposed to a commercial mixture of PCBs in the diet had a significantly higher antibody response than control birds while adult males had significantly suppressed antibody production (Smits and Bortolotti, 2001).

Sagerup et al. (2000) measured OC contaminants and intestinal helminth parasitic infection in glaucous gulls in Norway. They found a positive correlation between ten of 14 organochlorine contaminants measured, including PCBs, and intestinal nematode intensity, the most prevalent and virulent group of intestinal parasites in these birds. The authors conceded that correlative studies such as this one cannot confirm causal relationships, however, they speculated the association may be related to immunosuppression leading to an increased susceptibility to parasitic infections.

Experimentally, several studies have shown that harbour seals fed fish highly contaminated with OCs had impaired immune functions compared to those fed less contaminated fish (Bernhoft et al., 2000). Differences in immune

functions included: impaired natural killer cell activity, *in vitro* T-cell function, antigen-specific *in vitro* lymphocyte proliferative responses, and *in vivo* delayed-type hypersensitivity and antibody responses to ovalbumin (Bernhoft et al., 2000). Ross et al. (1996) found significant seasonal variation in natural killer cell activity in both control and exposed groups of seals. Exposed seals had significantly higher levels of Ah-receptor binding contaminant burdens, as measured by TCDD-TEQ levels, than did control animals fed less contaminated fish. The authors cautioned that, although there was a significant correlation between TCDD-TEQ levels and immunosuppression, there may also have been an immunotoxic contribution from non Ah-receptor binding classes of chemicals that were not measured in their study (Ross et al., 1995, 1996).

In a study of harbour porpoises found dead in United Kingdom waters, Jepson et al. (1999) found that animals that died due to infectious diseases (n=33) (primarily pneumonias due to parasitic or parasitic and bacterial infections) had significantly higher levels of PCBs in blubber (25 congeners) than healthy animals whose deaths were attributed to physical trauma (n=34). These results were not confounded by other measured variables including age, sex and nutritional status. A similar earlier study conducted in the same area did not find a significant association between infectious disease mortality and PCB levels (Kuiken et al., 1994). Jepson et al. (1999) attribute the different findings in the previous study to a smaller sample size (n=48), and therefore, less statistical power to detect a difference between the two groups. Furthermore, the earlier study also included a large percentage of decomposed porpoise carcasses

which may have reduced the reliability of the PCB results, whereas only freshly dead or slightly decomposed carcasses were included in the later study (Jepson et al., 1999).

IgG levels measured in polar bears in Svalbard, Norway, were negatively correlated with PCB and hexachlorobenzene levels in plasma (Bernhoft et al., 2000). Concentrations of OCs, in particular PCBs, were found in the Svalbard polar bears in the same order of magnitude as those found to produce immunotoxic effects or reduced immune response and resistance to infection in studies with experimental animals. It is thought that arctic mammals such as polar bears may be more susceptible to low levels of OC contaminants, particularly during periods of fasting or starvation when lipid deposits containing the contaminants are mobilized (Muir et al., 1997; Polischuk et al., 2002).

Land mammals too have shown immunotoxic effects from exposure to OCs. An experimental study in ranched mink fed a diet based on fish captured downstream from a bleached kraft pulp mill showed no detrimental effects on a broad range of pathological, biochemical, hematological, behavioural or reproductive variables (Smits et al., 1995). There was, however, a significantly suppressed delayed type hypersensitivity immune response in the exposed animals (Smits et al., 1996).

Immunotoxic effects of PCBs and other contaminants, in some cases, are thought to be associated with disturbances at the population level. The impairment of immune function associated with exposure to environmental contaminants may have been a contributing factor in the 1987-1991 morbillivirus

epizootics in populations of harbour seals, grey seals and striped dolphins in Europe (Muir et al., 1997; Fox, 2001). During the 1988-9 seal epizootic, nearly 18,000 harbour seals died in the North, Irish and Baltic Seas. Particularly high levels of mortality occurred in more polluted areas.

In summary, numerous associations have been found between immunological measures and environmentally relevant OC concentrations in various free-living avian and mammalian species. However, it is unclear at present whether these associations may be contributing to population level effects in wildlife species.

1.5.2.3.5 Neurological effects

Experimental studies with avian and mammalian wildlife have demonstrated a variety of effects with exposure to PCBs that may be related to neurological impairment. These include: alterations in central nervous system neurotransmitter levels, retarded learning, increased activity, and behavioural changes (USDHHS, 2000).

Exposure to dioxin and dioxin-like compounds has been demonstrated to have an effect on neurological development in various avian species. The brains of great blue heron (Henshel et al., 1995), double-crested cormorants, and eagles have been shown to have gross cerebral asymmetry which correlated with levels of TCDD and TCDD-TEQs in sibling eggs (Henshel, 1998). Similar changes were not seen in uncontaminated reference heron and cormorant colonies. These changes were also seen experimentally in chickens exposed to varying degrees of TCDD in ovo. All four species reviewed by Henshel (1998)

have shown similar measurable neurological effects in spite of the differences in contaminant mixtures to which they were exposed.

1.5.2.4 Toxicological effects of exposure to metals

The toxicological effects of exposure to metals vary with the metal, species, metabolism or biotransformation, and rates of elimination (Dietz et al., 1997).

For many metals, toxicity is reduced by the formation of inert complexes. For example, the cytoplasmic protein metallothionein (MT), is induced by, and binds with, cadmium, forming biologically inert Cd-MT complexes within tissues (Goyer, 1995). Similarly, selenium is thought to reduce the toxicity of mercury by forming inert complexes (mercuric selenide) that are stored in the liver (Dietz et al., 1997). Marine mammals appear to be tolerant to high hepatic concentrations of mercury, where this metal is bound with selenium (Law, 1996; Wagemann et al., 2000). Demethylation is also an important component of detoxification for mercury (Thompson, 1996; Dietz et al., 1997). The organic form of mercury (MeHg) is lipophilic and therefore more biologically active, making it more toxic than inorganic forms.

Specific toxic effects associated with individual metals as well as the beneficial effect of selenium on mercury toxicity are discussed in the following sub-sections.

1.5.2.4.1 Cadmium

Systems reported to be affected by high levels of cadmium in animals and humans include renal, reproductive, immune, nervous, hematopoetic, cardiovascular, hepatic, and skeletal systems (Muir et al., 1997; USDHHS,

1999). The primary target of chronic cadmium toxicity in mammals and birds is the kidney (Scheuhammer, 1987; Alden and Frith, 1991; USDHHS, 1999), and the earliest light microscopic change in this organ is tubular necrosis (Alden and Frith, 1991). However, studies documenting biological effects associated with high cadmium concentrations in tissues in wildlife populations are rare.

Cadmium concentrations in healthy wild birds vary widely among species and among populations within species. In supposedly unpolluted areas, cadmium concentrations are consistently several orders of magnitude higher in pelagic seabirds than in terrestrial birds (Furness, 1996). Larison et al. (2000) found microscopic renal lesions and reduced concentrations of skeletal calcium in white-tailed ptarmigan with renal cadmium levels > 100 µg/g wet weight (ww) sampled in the Colorado ore-belt region, an area with high concentrations of trace metals including cadmium. In a study of common eiders and king eiders in the Canadian Arctic, concentrations of cadmium in king eiders from one location were among the highest recorded in eider ducks (up to 232.5 µg/g dry weight in kidney, approximately 58 µg/g ww) (Wayland et al., 2001). Sections of kidneys, liver, ovaries, testes and spleens from a subsample of birds were examined histologically and all tissues were considered to be normal or had lesions that were felt to be very mild or mild and were not associated with cadmium concentrations.

In a review of cadmium toxicity in small mammals, Cooke and Johnson (1996) cited only one field study that demonstrated effects due to cadmium exposure. That study (Hunter et al., 1984) found ultrastructural renal and hepatic

lesions that were associated with high cadmium levels in common shrews. In spite of these lesions, the shrews appeared to be in good condition and no evidence of clinical renal dysfunction was found on urine analysis.

Although large mammals such as ungulates and seals have been shown to have some of the highest levels of cadmium recorded in wildlife (Muir et al., 1997), field-based evidence for biological effects related to high cadmium levels is lacking (CACARII, 2003). However, few studies have investigated potential biological effects in these species (eg. Sonne-Hansen et al., 2000).

1.5.2.4.2 Mercury

Mercury is found in a variety of inorganic and organic forms with varying degrees of biological toxicity (Thompson, 1996). Methylmercury (MeHg) is the most stable, bioavailable and toxic form of mercury.

Methylmercury has been associated with neurological impairment; reproductive effects such as increased early embryonic mortality and increased number of unfertilized eggs; and weight loss in birds at relatively low dietary concentrations (Scheuhammer, 1987; Thompson, 1996). Marked species differences in mercury levels have been found, with pelagic seabirds accumulating particularly high levels compared to other avian species. The toxicological significance of mercury in seabirds is difficult to assess (Thompson, 1996). In mammals, methylmercury is primarily a neurotoxin but has also been shown to interfere with spermatogenesis. Developing fetuses and nursing offspring are particularly susceptible to the neurological effects of MeHg (Dietz et al., 1997).

Field studies of free-ranging birds have shown associations between high levels of mercury and various health parameters, including body condition, immune function and intestinal parasitic infection. Higher total mercury levels were found in some tissues of common loons that died in poor body condition in eastern Canada compared to apparently healthy birds (Daoust et al., 1998; Scheuhammer et al., 1998). However, it was unclear whether the higher levels of mercury in specific tissues of these emaciated birds were a cause of the emaciation or had instead resulted from a redistribution of mercury secondary to muscle wasting from other causes.

In a series of field studies on king eiders and common eiders in the Canadian Arctic, Wayland et al. (2001, 2002, 2003) examined relationships between tissue concentrations of trace metals and selected health parameters, including body condition, immune function (cell-mediated and antibody response) and gastrointestinal infection by helminth parasites. Elevated mercury and cadmium levels were found to be consistently associated with relatively poor body condition (Wayland et al., 2003). However, other associations found between metal levels and these health parameters were generally not consistent across species or studies. In common eiders, numbers of gastrointestinal nematodes increased as total and organic mercury concentrations increased (Wayland et al., 2001), and cell-mediated immunity was positively related to hepatic selenium concentrations (Wayland et al., 2002). Overall, the authors concluded that most of the results from the studies did not consistently show adverse health effects that could be attributed to elevated tissue metal levels.

1.5.2.4.2.1 Selenium concentrations in relation to mercury toxicity

Selenium levels have been found to have a beneficial effect on mercury toxicity (Thompson, 1996; Goyer, 1995) that is believed to result from the formation of toxicologically inert inorganic Hg-Se complexes (Scheuhammer et al., 1998). In free-ranging mammalian species with elevated total mercury concentrations in the liver, hepatic selenium levels were strongly positively correlated with mercury levels, often at a 1:1 molar ratio (Scheuhammer et al., 1998; Eisler, 1985). A similar relationship has been found between inorganic mercury and selenium in liver and kidney in piscivorous birds (Scheuhammer et al., 1998), however, other studies on avian species have shown more variable results. In migratory birds, the relative rates of accumulation of the two elements may influence the ratio found in the liver at a given time (Ohlendorf, 1996). Determination of selenium concentrations, in addition to total and organic mercury levels in kidney and liver tissues, is recommended when making toxicological assessments of these contaminants in piscivorous birds (Scheuhammer et al., 1998).

Selenium is a beneficial or essential element in small amounts, however, exposure to high concentrations of selenium in plants and water can lead to toxicity in various species of wildlife (Eisler, 1985; O'Toole and Raisbeck, 1997). High levels of selenium have been documented in areas contaminated by fly-ash, sewage sludge and mining and smelting emissions (Eisler 1985; Heinz, 1996). In North America, significant adverse effects from selenium exposure have been documented in waterfowl populations in selenium-contaminated

wetlands in the western United States (Ohlendorf, 1996; O'Toole and Raisbeck, 1997). In eastern North America, in general, soil selenium content tends to be low.

1.5.2.4.3 Lead

Biological effects of lead reported in birds based on experimental studies and clinical cases of chronic lead poisoning in free-ranging waterfowl include: lethargy, weakness, anorexia, anemia, reduced weight gain or weight loss, diarrhea, coma and death (Locke and Thomas, 1996; Dietz et al., 1998). In addition, Rocke and Samuel (1991) found that the number of immunologic cells decreased with increased lead concentrations in tissues of experimentally exposed male mallards. The toxicity of lead in birds seems to be more pronounced in young, altricial birds compared to adults (Scheuhammer, 1987).

The most common route of exposure of lead in wildlife is ingestion of spent lead shots or lead sinkers used for fishing. Affected groups include waterfowl, loons and raptors. Lead from ammunition embedded in tissues does not appear to be sufficiently absorbed to be a toxicological risk (Locke and Tomas, 1996).

Mining and smelting operations have been implicated as the source of lead-contaminated sediments in the Coeur d'Alene River Basin in Idaho. Biological effects in a number of free-ranging waterfowl species, including death, have been attributed to ingestion of the lead-contaminated sediments in this area (Hoffman et al., 2000). In a study of Canada goose goslings fed a diet incorporating sediments from this area, toxic effects including hematological effects (such as increased free erythrocyte protoporphyrin concentrations in

blood and depression of red blood cell δ-aminolevulinic acid dehydratase [ALAD], both sensitive indicators of lead exposure and toxicity in birds), oxidative stress and reduced growth and survival were seen in treated birds compared to untreated controls and controls fed clean sediment (Hoffman et al., 2000).

In humans, several neurobehavioural effects have been attributed to low-level lead exposure in early childhood. Due to increased intestinal absorption and increased sensitivity of developing tissues, children are more sensitive to lead exposure than adults (Hansen et al., 1998). In mammals, the hematopoetic, cardiovascular, nervous, gastrointestinal and renal systems are reportedly affected by lead toxicity, and chronic, low-level exposure during pre-natal and post-natal stages may result in reduced growth and permanent neurobehavioural deficits (Ma, 1996).

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2. HARVEST SURVEYS

2.1 Introduction

Harvest statistics are "counts, or estimates, of the quantity of a particular species of fish and wildlife taken in a specific area or by a specific group of people over a period of time" (Usher and Wenzel, 1987). The data collected from harvest surveys can reflect current Aboriginal land use and the domestic or subsistence use of wildlife (Armitage, 1990) and can be used for a variety of reasons such as basic social or biological research (eg. nutritional studies or wildlife management); land rights negotiations; and compensation or mitigation purposes (Usher and Wenzel, 1987). Native harvest surveys have been used for these purposes in many communities across northern Canada (eg. JBNQ, 1982; Prevett et al., 1983; Gamble, 1987; Tobias and Kay, 1993).

One of the first coordinated attempts at collecting information about Aboriginal hunting efforts came from a comprehensive series of harvest surveys which was initiated after the James Bay and Northern Québec Agreement was signed in 1975 (JBNQ, 1982). Surveys were conducted over five consecutive years, and the information collected was used to help negotiate guaranteed levels of harvesting for Cree and Inuit in the region. These surveys have subsequently been used as a model for those conducted in other areas (Usher et al., 1985) (eg. Armitage, 1990; Tobias and Kay, 1993).

Armitage (1990) documented the harvest by Innu hunters from Sheshatshit and Utshimassit (Davis Inlet), Labrador (see map, Figure 1.1) in 1987 for its

contribution to the "domestic production" of the communities. For Innu, domestic production "encompasses all hunting, trapping, fishing, and gathering activities as well as cutting firewood, craft production, child care and food production in the household" (Armitage, 1990). For the Innu people in Labrador, the transition from full-time subsistence in the country to village-based living was initiated in the 1960s when the government designated villages for them at Sheshatshit and Utshimassit. In December, 2002, residents of Utshimassit moved to the new community of Natuashish. This move was initiated and supported by the *Mushuau* Innu people[2].

At the time of Armitage's study, many Innu families lived two lives, one in the community and the other, for varying amounts of time on a seasonal basis, in temporary camps in the country (Armitage, 1990). For a number of Innu families, this is still the case today. The time spent in the country provides families with an opportunity to have access to country foods as well as an opportunity for healing and a chance to renew their relationships with each other, away from the communities, using the skills required for life in the country (Armitage, 1990). However, with the changing demography of the Innu population, along with fluctuations in wildlife populations, it is unclear whether the results obtained by Armitage (1990) reflect contemporary harvesting practices.

The primary objective of these harvest surveys was to determine, quantify

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2 Mushau Innu (literally translated as "People of the Barrens") traditionally inhabited the barrens of northern *Nitassinan* (Québec-Labrador peninsula). In the 1960s, the Mushuau Innu settled in the government-designated community of Utshimassit.

and rank (according to the number harvested and the edible weight provided by each species) the species of wild animals that were caught and killed by the residents of the two Labrador Innu communities, Sheshatshit and Utshimassit, between 01 September 1999 and 31 May 2000. The survey results facilitated the scientific determination of species that formed a substantial part of the Innu diet, and subsequently, the selection (in conjunction with consultations with members of the two communities) of four species to include in the detailed animal heath and contaminant assessment phase of the overall study.

2.2 Materials and methods

2.2.1 Selection of sample population

A stratified random sampling process was used to sample the population of eligible hunters (male and female). An overall list of potential hunters was derived from 1995 community health lists of inhabitants of the two Labrador Innu communities. Based on typical Innu hunting practices, males and females, 13 years of age and older (born before 01 January 1986), were considered to be eligible hunters to be surveyed. Non-Innu men and women who were known to be integrated into Innu families were also included as eligible hunters. Those who were either known to have moved, unable to hunt due to a disability, or deceased were removed from the list. The remaining list formed the sampling frame.

This list of eligible hunters was stratified into two groups, "outpost" and "non-outpost", according to their participation in recent Outpost Programs

sponsored by the community Band Councils. The Outpost Programs were initiated in the early 1970s to assist Innu families in accessing hunting areas. These programs have played an important role in facilitating present-day harvesting activities for many Innu families (Armitage, 1990). The assumption behind this method of stratification was that those who participated in the Outpost Program in the past tended to hunt regularly while in the country and may have been active hunters around the community as well. This stratification was used to ensure that those who were more likely to be frequent hunters would be included in the study. In Sheshatshit, the only available list of Outpost Program participants was from spring 1999. In Utshimassit, the lists of participants from 1997, 1998 and 1999 were accessed.

Sample size calculations to determine the number of Innu people to be surveyed from this sampling frame utilized the standard deviation from the harvest surveys conducted in Sheshatshit and Utshimassit in 1986 – 87 (Armitage, 1990), and the goal was to be within 25% of the true mean with 95% confidence. These calculations resulted in the necessity to survey 26 people per village, or approximately 4% of the sampling frame for Sheshatshit and 7% for Utshimassit. However, because there was a much smaller number of participants than non-participants in the Outpost Programs, the desired survey proportion for the outpost participant population was set at a minimum of 20% (27 people for Sheshatshit and 5 people for Utshimassit) in order to obtain reasonably accurate estimates for this stratum of the population. For consistency, the desired sample size was increased to 30 individuals from each

sub-population in Sheshatshit and 30 non-participants in the Outpost Program and 12 participants in the Outpost Program in Utshimassit for each hunting period surveyed.

The eligible hunters in the sampling frame were assigned computer-generated random numbers, and "outpost" and "non-outpost" residents were randomly selected until the approximate number of required hunters was surveyed.

2.2.2 Survey design

The survey was designed to answer questions regarding which species were hunted during the time period being considered, and how many of each species were harvested (struck and retrieved).

A preliminary list of species for the harvest survey was derived from a comprehensive list of potentially hunted species compiled by the Innu Nation for use in harvest diaries. The list included individual species (eg. willow ptarmigan) as well as related groups of species (eg. ducks) for use if the respondent did not identify the particular species.

The survey was developed in close consultation with the Innu co-researchers and Innu hunters in each community, and those animals that were not identified as being normally hunted or eaten were removed from the list. The remaining species on the list were identified by both the English and Innu-aimun names. Commonly recognized names were used in place of formal names where it was deemed appropriate (eg. partridge instead of grouse). The species

were categorized into groups familiar to Innu hunters: "land animals", "flying animals" and "fish" (which also included marine mammals). A category of "other" was added to each group to account for any species not included in the survey list. For ease of administration, the order of the species within each category was arranged beginning with those most commonly hunted. Initial drafts of the survey were piloted on Innu Nation staff members and revisions were made, based on comments received (a copy of the survey used in both communities is presented in Appendix D).

2.2.3 Survey administration

The surveys were administered by the Innu co-researcher for each community in the language chosen by the interviewee. In a single interview, each respondent was asked to recall what animals were hunted and the number of each species struck and retrieved for the specified hunting period. The vast majority of surveys were completed in person; however, some surveys were completed over the telephone when repeated requests for a personal interview were found to be impossible. A few surveys were left with the person being surveyed, to be completed on their own. This was done only when personal or telephone interviews could not be accomplished, and only when the Innu co-researcher felt that the respondent would be able to answer the survey questions without the need of verbal assistance.

Respondents were given the option of either filling in the actual number of each species of animal hunted, or circling the applicable range corresponding to that animal. The ranges listed for each animal were based on the number

typically killed during the hunting period. For example, for mammals such as beaver, the ranges listed tended to be small (1-4, 5-9, 10-14, etc.) whereas for birds and fish, the ranges given were larger (1-14, 15-29, etc.), reflecting the typical harvest for more easily accessible and plentiful animals.

To reflect seasonal differences in hunting patterns, surveys were administered for two separate hunting periods for residents of Sheshatshit: "fall" (September 1, 1999 to January 31, 2000) and "winter/spring" (February 1, 2000 to May 31, 2000). All respondents were surveyed as soon as logistically possible after the end of each hunting period. In general, surveys were completed within a few months of the end of each hunting period, while a few of the late respondents took as long as nine months to respond. Beyond nine months, it was felt that recall bias would be too great, especially with similarity of seasons (eg. surveying for fall 1999 was limited to the period prior to fall 2000).

Due to administrative difficulties, only surveys for the period including September 1, 1999 to January 31, 2000 were completed in Utshimassit, and only 16 surveys were completed from the randomly selected list of eligible hunters. An additional 14 surveys were also completed by known hunters not included on the random sample list. Due to the small number of randomly selected surveys completed, harvest estimates for Utshimassit were not calculated. The qualitative data from the surveys completed (randomly selected and known hunters) are reported in the Results.

2.2.4 Statistical analyses

For Sheshatshit, harvest data for each species were summarized by hunting season. Where ranges of animals were selected instead of numbers, the midpoint of the range was substituted as an estimate of the number of animals harvested. If that number was not a whole number, the next lowest whole number was chosen (eg. 14 for the range: 10-19).

The distributions of the harvest estimates were right skewed, with most animals harvested by a few hunters and the majority of respondents reporting no hunting. Therefore, the normal practice of using the median to represent central tendency of right-skewed data would produce an estimate of zero, which would be a clear under-estimation of the true parameter. Furthermore, a zero estimate would be inappropriate for the future uses of the survey (eg. for selection of species for contaminant testing and health assessment, or for land-rights negotiations and/or compensation). Therefore, an arithmetic mean (number of animals per respondent) and standard deviation were calculated to determine the central tendency and variation of the harvesting data, by species, harvest season, and sub-population sampled (outpost vs. non-outpost participant).

The harvest data, by species, harvest season and sub-population sampled were then extrapolated from the sampled population to the total population of eligible hunters to produce community harvest estimates for each hunting period. This was done by determining the overall population means and standard error of the estimate of the mean, then multiplying by the total population of eligible hunters, stratified by outpost and non-outpost participants.

Total edible weight for each type of species was then calculated by multiplying the estimated total number harvested (and standard error) for each species with edible weight values for each species, as determined from Armitage (1990) and JBNQ (1982), which reflect the large proportion of each animal that is typically eaten. For certain species, edible weight values could not be obtained from these sources. Therefore, an approximate value was selected from another species within the same family.

Finally, a grand total of edible weight for all species harvested was calculated, and the proportion that each species represented of this grand total was determined in order to formulate the list of animal species of country food that could be considered to form a substantial part of the Innu diet (i.e. species forming $\geq$ 1% of the grand total edible weight).

2.3 Results

Table 2.1 lists the number and proportion of eligible hunters surveyed in Sheshatshit for each hunting period (fall 1999 and winter/spring 2000), by outpost participation strata. The inverse of these proportions were used as weighting factors within each stratum to determine the total community harvest estimates.

In the fall and winter/spring surveys, 47% and 55% of respondents, respectively, were women. For the fall survey, 50% of "outpost" and 45.7% of "non-outpost" respondents in Sheshatshit recalled hunting during that period,

Table 2.1. Number and proportion of eligible harvesters surveyed in Sheshatshit for the fall (01 September 1999 to 31 January 2000) and winter/spring (01 February 2000 to 31 May 2000) hunting periods.

	Total eligible hunters	Fall		Winter/Spring	
		Number Surveyed	Proportion Surveyed	Number Surveyed	Proportion Surveyed
Non-outpost	556	35	6.3%	24	4.3%
Outpost	135	18	13.3%	31	23.0%
Total	691	53	7.7%	55	8.0%

while 19.4% of "outpost" and 11.5% of "non-outpost" respondents recalled hunting during the winter/spring hunting period.

In Table 2.2, the harvest estimates for Sheshatshit, in total and by season, are listed by animal type (mammals, birds and fish), along with the standard errors of these estimates. The species within each animal type are ordered with respect to the total harvest estimate. A total of 10 mammalian species, 11 avian species and 8 fish species were reported to be harvested in the fall, while 5 mammalian species, 11 avian species and 7 fish species were reported to be harvested during the winter/spring hunting period.

The following species were reported as harvested between 01 September, 1999 and 31 January, 2000 by 30 hunters in Utshimassit: caribou, black bear, porcupine, seal (generic), red fox, snowshoe hare, Arctic hare, Arctic fox, muskrat, spruce grouse, grouse (generic), willow ptarmigan, rock ptarmigan, Canada goose, eider duck (generic), scoter (generic), duck (generic), pintail, gull (generic), tern (generic), Arctic char (searun and landlocked), brook trout (searun), lake trout, salmon, cod, rockcod, lake whitefish and sculpin. Qualitative differences of the survey results from Utshimassit compared to Sheshatshit included the harvesting of marine species such as seal, Arctic char, and eider ducks, as well as more northerly ranging terrestrial species such as Arctic hare and Arctic fox.

Table 2.3 lists the estimated total edible weight by species, in kilograms, of the total community harvest estimates for Sheshatshit. The species are ordered

Table 2.2. Community harvest estimates of number of animals killed (standard error) by season and in total for Sheshatshit for the period 01 September 1999 – 31 May 2000, listed by animal type.

		Fall [a]	Winter/Spring [b]	Total harvest estimate
Mammals	Caribou	2349 (1037)	145 (69)	2494 (1039)
	Snowshoe hare	615 (242)	394 (346)	1009 (422)
	Porcupine	498 (166)	159 (76)	657 (182)
	Muskrat	76 (62)	0	76 (62)
	Red fox	76 (62)	0	76 (62)
	Beaver	55 (41)	18 (14)	76 (44)
	Mink	55 (55)	0	55 (55)
	Marten	55 (55)	0	55 (55)
	Otter	14 (14)	7 (7)	21 (15)
	Black bear	7 (7)	0	7 (7)
Birds	Spruce grouse	4906 (1769)	1106 (608)	6011 (1871)
	Grouse (generic)	3801 (1444)	394 (311)	4194 (1477)
	Willow ptarmigan	2211 (829)	207 (166)	2419 (846)
	Rock ptarmigan	1658 (1327)	332 (228)	1990 (1346)
	Ruffed grouse	1327 (1064)	283 (235)	1610 (1090)
	Canada goose	905 (346)	435 (249)	1341 (426)
	Duck (generic)	401 (214)	35 (35)	435 (217)
	Black duck	138 (76)	159 (104)	297 (129)
	Black scoter	0	131 (104)	131 (104)
	Surf scoter	124 (111)	0	124 (111)
	Loon (generic)	62 (55)	0	62 (55)
	Oldsquaw duck	7 (7)	35 (35)	41 (35)
	King eider	0	35 (35)	35 (35)
	White-winged scoter	0	35 (35)	35 (35)
	Blue winged teal	0	35 (35)	35 (35)
	Gull (generic)	28 (28)	0	28 (28)
	Owl (generic)	14 (14)	0	14 (14)
Fish	Rainbow smelt	4187 (3289)	974 (719)	5162 (3367)
	Brook trout	1762 (719)	332 (1660)	2094 (738)
	Lake trout	1147 (511)	104 (69)	1251 (516)
	Atlantic salmon	442 (269)	0	442 (269)
	Lake whitefish	193 (193)	0	193 (193)
	Northern pike	90 (69)	0	90 (69)
	Brook trout (searun)	55 (55)	35 (35)	90 (65)
	Rockcod	62 (62)	0	62 (62)
	Arctic char (landlocked)	0	35 (35)	35 (35)
	Arctic char	0	35 (35)	35 (35)
	Ouananiche[c]	0	21 (21)	21 (21)

[a] Fall includes the period from September 1, 1999 to January 31, 2000.
[b] Winter/Spring includes the period from February 1, 2000 to May 31, 2000.
[c] Landlocked Atlantic salmon

Table 2.3. Estimates of community harvest edible weight (and standard error) (kg) and percentage of total community harvest edible weight by species for Sheshatshit for September 1, 1999 – May 31, 2000.

Species	Edible weight	Total harvest [a] estimate (SE)	Total edible weight (SE)	Percent of Total edible weight
Caribou	61.7	2494 (1039)	153880 (64106)	**88.72**
Porcupine	4.76	656 (182)	3123 (866)	**1.80**
Canada goose	2.1	1341 (426)	2816 (895)	**1.62**
Spruce grouse	0.35	6011 (1871)	2104 (655)	**1.21**
Atlantic salmon	3.77	442 (269)	1666 (1014)	0.96
Lake trout	1.2	1251 (516)	1501 (619)	0.87
Grouse (generic)	0.35	4194 (1477)	1468 (517)	0.85
Brook trout	0.5	2094 (738)	1047 (369)	0.60
Snowshoe hare	0.84	1009 (422)	848 (354)	0.49
Willow ptarmigan	0.35	2419 (846)	847 (296)	0.49
Rock ptarmigan	0.35	1990 (1346)	697 (471)	0.40
Black bear	95.3	7 (7)	667 (667)	0.38
Beaver	7.9	76 (44)	600 (348)	0.35
Ruffed grouse	0.35	1610 (1090)	564 (382)	0.32
Duck (generic)	0.77	435 (217)	335 (167)	0.19
Black duck	0.77	297 (129)	229 (99)	0.13
Lake whitefish	0.6	193 (193)	116 (116)	0.07
Rainbow smelt	0.02	5162 (3367)	103 (67)	0.06
Black scoter	0.77	131 (104)	101 (80)	0.06
Otter	4.75	21 (15)	100 (71)	0.06
Rockcod	1.59	62 (62)	100 (99)	0.06
Surf scoter	0.77	124 (111)	96 (85)	0.06
Northern pike	1	90 (69)	90 (69)	0.05
Loon (generic)	1.08	62 (55)	67 (59)	0.04
Muskrat	0.64	76 (62)	49 (40)	0.03
Brook trout (searun)	0.5	90 (65)	45 (33)	0.03
Arctic char (landlocked)	1.2	35 (35)	42 (42)	0.02
Oldsquaw duck	0.77	41 (35)	32 (27)	0.02
King eider	0.77	35 (35)	27 (27)	0.02
White-winged scoter	0.77	35 (35)	27 (27)	0.02
Blue-winged teal	0.77	35 (35)	27 (27)	0.02
Ouananiche[b]	1	21 (21)	21 (21)	0.01
Arctic char	0.5	35 (35)	18 (18)	0.01

[a] Calculations based on total harvest estimates presented in Table 2.2
[b] Landlocked Atlantic salmon

with respect to the total edible weights. The total edible weight of caribou harvested was 153,880 kg or 88.7 % of the total edible weight of all species combined.

The species that comprised at least 1% of the estimated total edible weight harvested between 01 September 1999 and 31 May 2000 for Sheshatshit were: caribou, porcupine, Canada goose and spruce grouse. These four species as well as Atlantic salmon, lake trout and brook trout were included in the short-list from which residents of Sheshatshit selected two species for subsequent health and contaminant analyses. The fish species were included because their total edible weights were close to 1% of the total community harvest (salmon: 0.97%, lake trout: 0.86%, brook trout: 0.60%). Furthermore, in contrast to harvesting of birds and mammals, a large proportion of fishing activity takes place during the summer months (a period not included in the harvest surveys). Thus, the annual fish harvest was likely underestimated.

The following species were considered to provide a large proportion of the country food for Utshimassit, based on edible weight: caribou, seals (generic), Arctic char, Canada goose, brook trout, snowshoe hare, black bear. This list was derived in part from our survey results, but primarily from the harvest estimates reported by Armitage (1990) for the community in 1987.

2.4 Discussion

The primary use of these harvest surveys was to create ranked lists of species, based on hunter recall, that represent a substantial part of the diet (i.e. comprising at least 1 % of the estimated total edible weight) for the residents of

each Labrador Innu community in 1999 - 2000. These lists would then be used to assist researchers and community residents in choosing species for detailed animal health and contaminant study to be carried out in collaboration with hunters from each community over the following two years.

Gatherings of representative elders and hunters from each community were held to discuss the lists produced and to choose two species from each community list for the subsequent animal health and contaminant study. The species were chosen based on concerns over the health of the animals in relation to contaminants, from both local and long-range sources, and current and historical industrial developments in the hunting area (eg. hydroelectric developments, mining). The cultural significance of the animals to the community residents was also an important factor in the choice of species. Sheshatshit residents met on October 5, 2000 and chose caribou and porcupines. Utshimassit elders met on March 7, 2001 and chose Canada geese and black bear for the detailed study.

Although the relative proportions of the various species harvested are likely reflective of the community harvest during the hunting periods surveyed, in general, for most species, the harvest estimates appear to be inflated. Based on a community population of 1092 (from the 1999 census), the *per capita* country food harvest in Sheshatshit for the nine month period we surveyed was 160 kg/person. This estimate is considerably higher than that determined by Armitage (1990) in 1987 for both Sheshatshit (34 kg/person/year) and Utshimassit (101 kg/person/year). The average annual harvest estimate for the Cree of northern

Québec for 1972 – 79 was 121 kg/person (JBNQ, 1982), and that of the Inuit community of Makkovik, Labrador for a one year period in 1980 – 1981 was 85 kg/person (Alton Mackey and Orr, 1987).

The number of caribou that was reported killed during the 1999/2000 hunting season was over 10 times the number killed in 1987 as reported by Armitage (1990). This increase likely reflects, in part, a change in hunting practices. Caribou migration patterns changed around 1999, resulting in much easier access to the animals and a correspondingly larger harvest than in 1987. However, it is also likely that there was some degree of overestimation of the caribou harvest.

Some of the selection, misclassification, confounding and extrapolation factors that may have resulted in an overestimation of the harvest are discussed below. Therefore, the results are presented with the caution that the estimates should not be assumed to accurately reflect the actual community harvest for that period.

First, overestimation could have resulted from non-response bias. Non-response bias can occur if the sample population differs, for some reason, from those not surveyed. When samples are stratified, such as was done for these surveys, it is important to ensure that individuals are appropriately categorized and that a representative sample of each group is obtained. No statistically significant difference was found in age or gender distribution between "outpost" and "non-outpost" groups or between those surveyed and those not surveyed.

Several types of misclassification bias should also be considered in the possible overestimation. Survey responses may be inaccurate or biased if the

survey is somehow misleading or difficult to understand or administer. Misclassification was minimized by standardizing the surveys and their administration between seasons and communities. In Sheshatshit, one interviewer, who was a trusted member of the community, conducted all surveys. Surveys were administered in the respondent's language of choice. Overall, it was felt that there was a general acceptance of the study and the surveys. Before the survey was administered, respondents were instructed to recall only their own harvest. However, it is possible that some of the respondents may have accounted for the total harvest of the hunting party or household rather than their individual harvest only. Independent confirmation of the harvest estimates was not possible.

Recall failure is another source of potential misclassification bias. The surveys were designed to reflect seasonal hunting periods to assist the respondent's recall. As well, all surveys were administered within nine months of the end of the hunting period. However, it is often difficult to accurately recall harvests of smaller animals such as waterfowl, gamebirds and fish (Usher et al., 1985; Berkes, 1990). In general, the interviewer felt that the responses were reasonable, given each individual circumstance. However, it is possible in some cases that the harvests were overestimated to some degree.

With regard to the extrapolation factor, the total population of eligible hunters may have been overestimated leading to overestimates of the total community harvest. To determine the total population of eligible hunters, community lists of residents were used that were derived in 1995, four years prior to the harvest

survey. Although the co-researchers in each community updated the lists to account for those who were no longer resident or were otherwise not considered to be eligible hunters, there were some difficulties in keeping the list up-to-date. Inaccuracies in the community lists likely introduced some degree of non-response bias.

These sources of bias could influence the data and must be considered when interpreting the harvest statistics. Although most of the annual hunting by community residents takes place during the hunting periods covered by the surveys, hunting during the summertime, while limited, could alter these totals somewhat. Furthermore, there is yearly variation in what is harvested due to the opportunistic nature of the hunt, and thus, data collected over a number of years would give a more accurate representation of the community harvest effort over the long term.

Qualitatively, the range of species harvested was similar to the survey results reported by Armitage (1990). One notable difference was that no moose were reported to be harvested in these surveys, whereas in 1986-87, moose accounted for the third largest amount of country food (by edible weight) (Armitage, 1990). When the species harvested were considered by group for Sheshatshit, Armitage (1990) found that fish were second to caribou in terms of edible weight, followed by small game and migratory waterfowl. Whereas small game (including hare, porcupines, grouse and ptarmigan) were second to caribou, followed by fish and migratory waterfowl in these surveys. As discussed

previously, the importance of fish was likely underestimated in the present surveys.

In Utshimassit, Armitage (1990) found that the most important species for country foods were caribou, fish and seals. For Québec Cree, fish, moose and waterfowl were the leading country foods based on edible weight (JBNQ, 1982), while for Inuit in Makkovik, Labrador, leading country foods were caribou, fish and seal (Alton Mackey and Orr, 1987).

Due to possible overestimation of the numbers of harvested animals, a second set of community harvest estimates was calculated that provided much more conservative values. These were determined by reducing the overall population of eligible hunters based on the assumption that those residents randomly selected for interview that were "non-respondents" were all non-hunters. The calculations for the second set of estimates are based on the response rates for the winter/spring 2000 surveys, where 50% of the non-outpost group and 38% of the outpost group who were contacted responded to the survey. An accurate response rate could not be determined for the fall surveys because of some initial errors in classifying the sample populations that were subsequently rectified before the winter/spring surveys were started. Table 2.4 contains the results of these much more conservative estimates of harvested animals.

2.4.1 Conclusions

Hunting remains an important activity for many residents of both Sheshatshit and Utshimassit, and a wide variety of species are hunted by residents of both communities.

Table 2.4. Alternate conservative community harvest estimates of number of animals killed (and standard error) by season and in total for Sheshatshit for the period 01 September 1999 – 31 May 2000 listed by animal type.

		Fall [a]	Winter/Spring [b]	Total [c] harvest estimate
Mammals	Caribou	1231 (543)	76 (36)	1307 (544)
	Snowshoe hare	322 (127)	206 (181)	529 (221)
	Porcupine	261 (87)	83 (40)	344 (96)
	Muskrat	40 (33)	0	40 (33)
	Red fox	40 (33)	0	40 (33)
	Beaver	29 (22)	11 (7)	40 (23)
	Mink	29 (29)	0	29 (29)
	Marten	29 (29)	0	29 (29)
	Otter	7 (7)	4 (4)	11 (8)
	Black bear	4 (4)	0	4 (4)
Birds	Spruce grouse	2570 (927)	579 (319)	3149 (980)
	Grouse (generic)	1991 (757)	206 (163)	2197 (774)
	Willow ptarmigan	1158 (434)	109 (87)	1267 (443)
	Rock ptarmigan	869 (695)	174 (119)	1043 (705)
	Ruffed grouse	695 (557)	148 (123)	843 (571)
	Canada goose	474 (181)	228 (130)	702 (223)
	Duck (generic)	210 (112)	18 (18)	228 (114)
	Black duck	72 (40)	83 (54)	156 (67)
	Black scoter	0	69 (54)	69 (54)
	Surf scoter	65 (58)	0	65 (58)
	Loon (generic)	33 (29)	0	33 (29)
	Oldsquaw duck	4 (4)	18 (18)	22 (18)
	King eider	0	18 (18)	18 (18)
	White-winged scoter	0	18 (18)	18 (18)
	Blue winged teal	0	18 (18)	18 (18)
	Gull (generic)	14 (14)	0	14 (14)
	Owl (generic)	14 (14)	0	14 (14)
Fish	Rainbow smelt	2194 (1723)	510 (376)	2704 (1764)
	Brook trout	923 (376)	174 (87)	1097 (386)
	Lake trout	601 (268)	54 (36)	655 (270)
	Atlantic salmon	232 (141)	0	232 (141)
	Lake whitefish	101 (101)	0	101 (101)
	Northern pike	47 (36)	0	47 (36)
	Brook trout (searun)	29 (29)	18 (18)	47 (34)
	Rockcod	33 (33)	0	33 (33)
	Arctic char (landlocked)	0	18 (18)	18 (18)
	Arctic char	0	18 (18)	18 (18)
	Ouananiche[d]	0	11 (11)	11 (11)

[a] Fall includes the period from September 1, 1999 to January 31, 2000.
[b] Winter/Spring includes the period from February 1, 2000 to May 31, 2000.
[c] Estimates are based on the assumption that non-respondents to the surveys were likely to be non-hunters.
[d] Landlocked Atlantic salmon

Based on the period including September 1, 1999 to May 31, 2000, the most important species in terms of edible weight of country food for residents of Sheshatshit were caribou, porcupine, Canada geese and spruce grouse in that order. Several fish species also provide a large proportion of country food for the community, however, as a considerable amount of fishing activity takes place during the summer months, a period not covered in our surveys, it is unclear how large a proportion of the annual country food harvest they comprise. The fall season appears to be a more important hunting period compared to the winter/spring, both in terms of the number of respondents who reported hunting and the total number of animals harvested. A representative sample of residents of Utshimassit was not obtained and, therefore, harvest estimates for that community could not be calculated.

Although the relative proportion of species harvested in Sheshatshit is likely accurately reflected by these surveys, the total harvest estimates based on the original calculations (Table 2.2 and Table 2.3) appear to be inflated. The true harvest by community residents for the hunting period surveyed likely falls between the original harvest estimates and the alternative, more conservative, harvest estimates that were calculated (Table 2.4).

2.5 References

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3. HEALTH PARAMETERS OF CARIBOU IN RELATION TO TISSUE CONCENTRATIONS OF ENVIRONMENTAL CONTAMINANTS

3.1 Introduction

The caribou is the dominant large terrestrial herbivore in many northern ecosystems. In eastern Canada, the largest population of caribou is the George River herd (GRH). The range of the GRH includes much of Labrador and extends across the Ungava peninsula well into the province of Québec. The GRH underwent rapid growth up to the mid-1990s when it was estimated at approximately 700,000 animals and has recently declined to approximately 400,000 animals (R. Otto, pers. comm. 2003).

Several studies investigating the biology and ecology of the GRH have been published (eg. Parker, 1981; Huot, 1989; Crête et al., 1990; Hearn et al., 1990; Crête and Huot, 1993). Some have focused on health parameters such as parasite burdens (Huot and Beaulieu, 1985; Lankester and Luttich, 1988), while others have concentrated on tissue contaminant levels (Crête et al., 1989; Robillard et al., 2002).

Contaminants such as metals, organochlorines (OCs) and radionuclides in northern ecosystems have been the focus of recent research programs in Canada and worldwide (AMAP, 2002; CACAR II, 2003). A consistent finding of many studies of environmental contaminants throughout northern Canada is elevated levels of cadmium in the kidneys of ungulates, including caribou (Muir et al., 1996; Dietz et al., 1998).

An important route of exposure to environmental contaminants for terrestrial animals is through the diet. The diet of GRH caribou consists of a variety of plant types including grasses and sedges, woody stems and twigs, leaves, lichens and mosses (Parker 1981; Crête et al., 1990). Lichens and mosses, which lack root systems, take up metals and minerals directly through the air and are often used as monitors of atmospheric deposition of metals (Dietz et al., 1998). High cadmium levels in GRH caribou were first documented in the 1980s (Crête et al., 1989), and subsequent studies have also demonstrated elevated levels (Robillard et al., 2002). Beginning in the 1980s, both the Québec and Newfoundland and Labrador provincial governments released public advisories regarding the consumption of liver and kidneys of caribou and other ungulates based in part on these findings (Ministère d'Agriculture, Pecheries et Alimentation, 2002; Department of Tourism, Culture and Recreation, 2003). Separate assessments of the risk to public health in Québec from consumption of these organs have been published, some with contradictory conclusions (eg. Archibald and Kostatsky, 1991; Robillard and Bélanger, 1997).

The risk of high renal cadmium levels to the health of free-ranging ungulates is not well understood. There have been no published reports of adverse health effects due to cadmium toxicity in wild ungulates despite markedly elevated concentrations in some animals (Dietz et al., 1998). However, the potential biological effects of cadmium, or other contaminants, in ungulates, including caribou, have not been investigated.

Innu in Labrador hunt caribou primarily from the GRH on a seasonal basis. Caribou have important nutritional, cultural and spiritual significance for the Innu people (Armitage, 1990; Loring, 1997). In addition to the concerns raised over contaminant levels in caribou, there is a general concern by the Innu people for the health of the caribou populations in Labrador in relation to industrial developments and other potential impacts on the ecosystems (Innes, 1998).

This study was carried out in collaboration with the Innu Nation in order to evaluate the health of important species hunted by Labrador Innu, including caribou, in relation to tissue environmental contaminant concentrations. Specifically, the objectives were:

1 - to assess the health of caribou killed during the regular subsistence hunt by determining body condition and hepatic parasite burdens and examining selected tissues for gross and microscopic pathological abnormalities;

2 - to determine the tissue concentrations of a wide range of OC and metal contaminants in these caribou; and

3 - to examine the relationship between important health parameters and tissue contaminant levels in the caribou through multiple regression analyses.

3.2 Materials and methods

3.2.1 Caribou locations and sample collection

In order to make sure that the results were clearly representative of the type of animals that make up the Innu harvest, the sample population consisted of twenty-seven caribou from the GRH that were shot during the regular subsistence hunt by residents of Labrador Innu communities of Sheshatshit and

Utshimassit (Davis Inlet). The caribou were killed in south-central and coastal-northern Labrador (approximately 61°W - 64°W and 53°30'N - 56°N) (Figure 1.1, Page 3). Samples were collected from caribou killed in February 2001 (winter, n=13) and in October and December 2001 (fall, n=14).

All methods for collection, handling, processing and disposal of samples were agreed upon by the Innu Nation co-researchers, community elders, hunters and their family members, and the researchers from the AVC. In order to retain strong community acceptance of the research and its findings, a high degree of importance was placed on maintaining the traditions surrounding the hunt, and ensuring that the caribou were still considered to be suitable for community distribution and consumption after sampling.

Once killed, each caribou was cleaned by removing the gastro-intestinal tract, leaving the thoracic cavity intact. During examination of the carcasses in the field, as described below, any visible abnormalities were noted, and comments from the hunter regarding the health of the animal were recorded.

All caribou, with the exception of one adult, were aged either by size (calf, immature) (n=4) or by cementum annular count of the primary (middle) incisor (n=22) (Matson's Laboratory, Milltown, Montana, U.S.A.). Each cementum age estimate was accompanied by a code indicating the level of certainty (A= certain of age estimate, B= some error of estimate possible, C= error of estimate likely). Fourteen (63.6%) had a certainty code of A, and the remaining eight (36.4%) had a certainty code of B.

On site, gross examination of the carcass and organs was limited to the exterior of the animal and to the abdominal organs (stomachs, intestines, liver, kidneys, uterus, fetus [if present] and bladder), testes and thyroid glands. The kidneys, thyroid glands, testes, and liver were collected and put on snow or ice for up to 12 hours until they could be frozen. Kidneys with attached fat were wrapped in aluminum foil (not rinsed in hexane) and placed in a plastic bag for subsequent contaminant analyses.

After examination and sample collection were complete, all caribou were taken back to the communities for distribution, while study samples were frozen at -20°C and transported to the Atlantic Veterinary College (AVC) further examination and sampling.

At the AVC, the tissues and organs were thawed and examined in more detail for gross abnormalities. Peri-renal fat was collected for OC analysis and was manipulated using stainless steel instruments rinsed three times in hexane, re-wrapped in aluminum foil triple-rinsed in hexane and kept frozen at -20°C. One half of one kidney was submitted for metal analysis. Medial sections of all kidneys, transverse sections of skin covering the tarso-metatarsal joint from a subsample of animals, and sections of grossly abnormal hepatic tissue were taken from thawed samples, and then fixed and processed routinely for light microscopic examination.

Body condition was evaluated by determining the kidney fat index (KFI), a recognized index of body fat in caribou throughout a wide range of body

conditions (Huot, 1988; Chan-McLeod et al., 1995). If both kidneys from a caribou were collected, the average of the two values was calculated.

For parasite recovery, the livers were examined at the AVC for the presence of the large American liver fluke, *Fascioloides magna* as per Lankester and Luttich (1988). Hepatic cysts that were consistent with cestode larvae were also noted.

3.2.2 Analysis of tissue contaminants

All kidney samples were analysed at the AVC Diagnostic Toxicology Laboratory for cadmium, lead and selenium. Sample digestion for analysis followed EPA (Environmental Protection Agency) Method 200.2 Revision 2.8. Analytical methods for selenium followed Fleuerstein and Schlemmer (2000). An atomic absorption spectrophotometer (AAS) (Perkin Elmer Analyst 800) equipped with a transversely heated graphite atomizer was used for selenium and lead analysis and an AAS equipped with a flame apparatus fuelled by acetylene and air with a manual sampling system was used for cadmium (Pesce and Kaplan, 1987). Sample batches were run with blanks, spikes and duplicates. Minimum detection limits were 0.4 µg/g wet weight (ww) for cadmium, 0.05 µg/g ww for lead and 0.05 µg/g ww for selenium. Standard reference materials (National Institute of Standards and Technology [NIST] pine needles for lead; Environment Canada, National Water Research Institute TMDA54.2 and National Research Council lobster hepatopancreas [NRC TORT-2] for cadmium; NRC TORT-2 for selenium) were analyzed at a minimum of every 10 samples and recoveries were within +/- 15%.

Analysis of kidney samples for total mercury was carried out at Philip Analytical Services (Halifax, Nova Scotia, Canada) following EPA method 245.6. Approximately 0.3 g of homogenized tissue sample was digested and analyzed for total mercury using cold vapour atomic absorption spectrophotometry (Leeman PS200 Mercury Analyzer). Reagent blanks, duplicates, reference materials (eg. dogfish liver DOLT-2 or dogfish muscle DORM-2) and method spikes were prepared and analyzed in the same manner as the samples. One reagent blank, one duplicate, one spike and one reference material was analyzed for every 20 samples with a minimum of one per batch. A total quality control effort of 10% was maintained. The minimum detection limit for mercury was 0.01 µg/g ww. The kidney was chosen for metal analysis because of results from numerous previous studies showing moderate to high levels of cadmium in caribou kidneys (eg. Crete et al., 1989; Elkin and Bethke, 1995; Larter and Nagy, 2000).

Peri-renal fat samples were analysed for OC pesticides and PCBs at the Environmental Quality Laboratory (Environment Canada, Moncton, New Brunswick, Canada). Analyses included 20 OC pesticides (hexachlorobenzene [HCB], alpha- hexachlorocyclohexane [α-HCH], gamma-HCH [lindane], aldrin, dieldrin, methoxychlor, mirex, sum of endosulfans [alpha- and beta-endosulfan], sum of chlordanes [heptachlor, heptachlor epoxide, gamma-chlordane (-CHL), *cis*-chlordane, *trans*-nonachlor] and sum of dichlorodiphenyltrichloroethanes [DDT] and metabolites [*o,p'* and *p,p'* - DDD, DDE, DDT]) and 24 PCB congeners (8, 18, 28, 29, 44, 50, 52, 66, 77, 87, 101, 104, 105, 118, 126, 128, 138, 153,

170, 180, 187, 195, 206, 209). Analyses were performed using capillary column gas chromatography (GC) with an electron capture detector (ECD) (Agilent 5890 GC with dual ECD detectors). Tissue samples containing approximately 2 g of fat with surrogate solution added were blended with sodium sulfate and methylene chloride using an ultrasonic extractor. The extract was subsampled for gravimetric tissue analysis. The remaining extract was cleaned first through a gel permeation chromatograph followed by a small silica-gel micro column. Fractionation was completed on a silica gel column using hexane and 50/50 hexane/methylene chloride prior to GC/ECD analysis. Results were reported in ng/g dry weight with detection limits ranging from 3 ng/g (PCBs) to 20 ng/g (methoxychlor). Lipid weight concentrations were calculated by dividing the concentration of the contaminant in dry weight by the % lipid of each sample. Wet weight concentrations were determined by multiplying the dry weight concentration by the % dry matter (100 - % moisture) of the sample. All sample batches were run with procedural blanks, spikes and duplicates along with certified reference materials (NIST 1974a mussel tissue). Recoveries for certified reference materials were all within the certified ranges. Analytical duplicates were within 9%.

Due to cost restrictions, only samples of peri-renal fat tissue from the two oldest caribou were analysed for seven polychlorinated dibenzo-*p*-dioxins (PCDDs), ten polychlorinated dibenzofurans (PCDFs) and three non-ortho substituted, or coplanar, PCBs (coPCBs) at Axys Analytical Services Ltd. (Sidney, British Columbia, Canada). Analytes included 2,3,7,8-

tetrachlorodibenzo-*p*-dioxin (TCDD), 1,2,3,7,8-PeCDD, 1,2,3,4,7,8-HxCDD, 1,2,3,6,7,8-HxCDD, 1,2,3,7,8,9-HxCDD, 1,2,3,4,6,7,8-HpCDD, OCDD, 2,3,7,8-TCDF, 1,2,3,7,8-PeCDF, 2,3,4,7,8-PeCDF, 1,2,3,4,7,8-HxCDF, 1,2,3,6,7,8-HxCDF, 1,2,3,7,8,9-HxCDF, 2,3,4,6,7,8-HxCDF, 1,2,3,4,6,7,8-HpCDF, 1,2,3,4,7,8,9-HpCDF, OCDF and coPCB congeners 77, 126 and 169. Homogenized tissue samples were dried with anhydrous sodium sulphate and ground to a free-flowing powder with a glass mortar and pestle. Surrogate standard solutions were added to the ground sample and the mixture was soxhlet extracted with 1 : 1 dichloromethane:hexane, subsampled for gravimetric lipid analysis and concentrated by rotary evaporation. The extract was cleaned up sequentially on a Biobeads SX-3 column, layered silica gel column, carbon/Celite column and Florisil column where it was fractionated using 5% dichloromethane:hexane (F1 fraction) followed by dichloromethane (F2 fraction). The F1 fraction was loaded onto an alumina column cutpointed for PCBs and eluted with hexane (discarded) and 1:1 dichloromethane:hexane. The eluate was evaporated and an aliquot of carbon-labeled PCB recovery standards was added. The F2 fraction was loaded onto an alumina column cutpointed for dioxins. Once eluted and evaporated, an aliquot of carbon-labeled dioxin recovery standards was added.

PCDD/F analyses were completed using high-resolution gas chromatography with high-resolution mass spectrometric detection (HRGC/HRMS) (Ultima Autospec MS equipped with a Hewlett Packard 5890 GC). Detection limits were <0.07 pg/g ww for PCDD/F congeners. Trace

amounts of individual PCDD/F congeners were found in the procedural blanks (0.009 - 0.706 pg/g, *n*=4). Analytical duplicates were within 16%.

PCB congener analyses was carried out using HRGC/HRMS (VG high resolution MS equipped with a Hewlett Packard 5890 GC). Detection limits were <2.6 pg/g ww for individual PCB congeners. Trace amounts of individual PCB congeners were found in the procedural blanks (0.260 - 0.507 pg/g, *n*=3). Analytical duplicates were within 5%.

For quantification of both PCDD/F and PCB congeners, all samples were spiked with ^{13}C-labeled surrogate standards prior to analysis. Concentrations of target analytes were calculated using the isotope dilution method of quantification and were corrected based on the percent recovery of surrogate standards. Results were reported in pg/g ww. Sample detection limits were reported for each target analyte for each sample, based on a minimum detectable area for that compound in the chromatogram.

The 2,3,7,8-TCDD Toxic Equivalencies (TCDD-TEQs) for PCDD, PCDF and coPCB data were reported using World Health Organization toxic equivalency factors (WHO TEFs) for mammals (van den Berg et al., 1998).

3.2.3 Statistical analysis

The following parasitological parameters were determined for hepatic *F. magna* and cestode infections: prevalence (number of caribou infected with a particular parasite/number of caribou examined), abundance (number of a particular parasite counted in each caribou), mean abundance (total number of a particular parasite counted/number of caribou examined) and mean intensity

(total number of a particular parasite counted/number of infected caribou) (hereafter referred to as intensity) of infection, following Bush et al. (1997).

Contaminant values less than minimum detection limits (mdl) were replaced with ½ of the mdl. For sumPCB, because it was the summation of individual congeners, each having its own minimum detection limit, the mean mdl of the components of the sum was used as the mdl for the sum. Because the mdl for PCDD, PCDF and coPCBs were very low, values that were less than the mdl were considered to be insignificant and were not included in the TCDD-TEQ calculations.

Geometric (for non-normally distributed variables) or arithmetic (for normally distributed variables) means and 95% confidence intervals were used for descriptive statistics.

Pearson's correlation coefficients were determined among the contaminants measured in order to identify significant correlations between individual contaminants. Contaminants detected in at least 20% of the samples tested (mercury, selenium, cadmium and lead in kidney and HCB and alpha-HCH in fat) were included in univariate statistical analyses described below.

Health parameter and contaminant variables that were not normally distributed (KFI, hepatic parasite abundance, renal cadmium and lead) were transformed (natural log [ln]) to meet the assumptions of the parametric statistical tests used.

Simple associations among and between individual health parameters, contaminants and confounding variables (season, age, gender) were examined

using χ^2 statistics for categorical variables (gender, season, hepatic parasite prevalence), pair-wise Pearson's correlation coefficients and *t*-tests for normally distributed continuous variables and Spearman's correlation and Kruskal-Wallis test for non-normally distributed continuous variables (alpha-HCH only). Results of $p<0.05$ were reported as statistically significant. For analyses including health parameters, we also reported trends ($0.05<p<0.2$) in order to account for any near-significant relationships. Those contaminants and potential confounders found to have unconditional associations with health parameters at $p<0.2$ were included as potential predictor variables in multiple variable regression analyses (Dohoo et al., 2003).

Multiple variable regression analyses were used to simultaneously determine associations between health parameters and contaminant levels, controlling for possible confounding factors. Only KFI, cestode and *F. magna* prevalence and abundance were used as health parameters for regression analyses because parasite intensity levels only utilize data from infected caribou, and gross and microscopic lesions were rare, severely limiting the power to find significant risk factors. Multiple logistic (for *F. magna* and cestode prevalence) and linear (for KFI and *F. magna* and cestode abundance) regression models were developed to determine the associations between health parameters and contaminant levels. Those variables found to be significant at $p<0.2$ on univariate analysis with health parameters were offered to the regression models in a manual, forward step-wise manner. Two-way interaction variables between significant main effects were assessed, where applicable. The final models

included those variables that were significant at $p<0.1$ in order to insure that all potential associations were considered and to account in part for the relatively small sample sizes.

Goodness of fit of the final models was determined using standard regression diagnostic procedures (including analysis of residuals, leverage values, Cook's distance [linear regression] and Pearson and deviance residuals [logistic regression]). Statistical analyses were completed using STATA (Statistical Package, v.8.0; Stata Press, College Station, Texas, U.S.A.).

3.3 Results

3.3.1 Descriptive results

Of the 27 caribou examined, 16 (59%) were female and 14 (52%) were killed in the fall. No seasonal differences were seen in the age distribution (χ^2 = 12.3, df = 8, p=0.14) or sex distribution (χ^2 = 0.3, df = 1, p=0.58). Adults (≥2 years) represented 74% of caribou sampled. Ages ranged from 0 (calf) to 12 years old and the mean age was 3 years (95%CI: 2.0 - 4.4).

3.3.1.1 Health parameters

Although KFI was only measured in 25 (93%) caribou because of gunshot wounds or damage to the kidneys during cleaning, all caribou examined had measurable fat stores and appeared to be in adequate body condition for the time of year (geom. mean KFI: 47.9; 95% CI: 38.2 - 60.1). All caribou were considered to be healthy by the hunters.

On laboratory examination, *F. magna* was found in 78% (95% CI: 62 - 94) of the livers based on the presence of encapsulated and migrating flukes, with a

geometric mean abundance of 4.2 flukes (95% CI: 2.0 - 8.2) and geometric mean intensity of 8.3 flukes (95% CI: 4.9 - 14.0). Most capsules contained two flukes and a single capsule contained four flukes. However, the number of flukes per capsule was not consistently recorded. The most heavily infected caribou, a 7-year-old female, had a total of 30 capsules in the liver and 67 encapsulated and migrating *F. magna*.

Cestode larvae were found in livers with a prevalence of 50% (95% CI: 48 - 52), a geometric mean abundance of 0.6 larvae (95% CI: 0.3 - 1.1) and a geometric mean intensity of 1.9 larvae (95% CI: 1.3 - 3.0).

No significant gross or microscopic lesions, other than in the liver, were noted in most animals. The exceptions were a marked unilateral interstitial nephritis with multiple cyst formation in a 12-year-old female, possibly the result of a previous bacterial infection, and a mild multifocal interstitial nephritis in another animal. Although not considered to be clinically significant, all other kidneys examined contained very small numbers of lymphocyte aggregates. Gross and microscopic lesions associated with hepatic *F. magna* parasitism varied among animals depending on the intensity of the infection. Infected livers had between 1 and 40 % of their parenchyma affected. Even in the most severely infected liver (30 capsules), there were numerous areas that appeared normal microscopically. Therefore, the lesions were not considered to have been clinically significant. No gross or microscopic evidence of *Besnoitia sp.* cysts, as described by Ayroud et al. (1995), was observed on examination of skin covering the tarso-metatarsal joint of eight of the animals.

3.3.1.2 Tissue prevalence and concentrations of contaminants

Table 3.1 provides a summary of the prevalence and concentrations of metals in the kidneys, and OC pesticide and PCB contaminants in the peri-renal fat of the caribou. Prevalence and concentrations of TCDD-TEQs in fat tissue from the two oldest caribou (11 and 12 years) are also reported.

All caribou had detectable levels of mercury, cadmium and selenium in kidney samples and 82% of kidney samples had detectable levels of lead (Table 3.1). Only three of the 20 individual organochlorine pesticides measured (HCB, α-HCH and γ-CHL) and four out of 24 PCB congeners (87, 101, 118, 209) were detected in peri-renal fat samples (reported as sumPCB) (Table 3.1).

The majority of the TCDD-TEQ in the two animals tested was comprised of coPCBs (81.1% and 77.5%, respectively) with PCB126 being the primary contributing congener (79.4% and 76.7% of the total TEQ, respectively).

3.3.2 Univariate analytical results

Table 3.2 shows results of simple associations between health parameters (KFI and hepatic parasite infection) and season, gender and age. Although a linear association was not found between KFI and age, a quadratic form of age $(\text{age} - \chi\ \text{age})^2$ was significantly negatively associated with KFI ($r=-0.50$, $p \leq 0.05$). Caribou infected with *F. magna* were significantly older than those free of infection, and abundance increased significantly with increasing age. A greater abundance of cestodes was seen in winter compared to fall. A seasonal difference was also seen with body condition, with higher KFI in winter caribou

Table 3.1. Concentrations of selenium, metals and organochlorine contaminants in caribou from the George River herd, Labrador, in 2001.

Contaminant		*n*	units[a]	number above mdl[b]	concentration[c]	mdl
	selenium	27	µg/g ww	27(100)	1.2 (1.2 - 1.3)[e] (0.9 - 1.5)	0.05
metals [d]	mercury	27	µg/g ww	27(100)	0.66 (0.58 - 0.75)[e] (0.36 - 1.10)	0.01
	cadmium	27	µg/g ww	27(100)	6.5 (4.7 - 8.9) (1.5 - 44.0)	0.4
	lead	27	µg/g ww	22(82)	0.09 (0.06 - 0.13) (nd - 0.75)	0.05
organochlorine [f] pesticides	HCB [g]	27	ng/g ww	27(100)	24.2 (21.5 - 26.9)[e] (11.8 - 36.2)	2.3 (2.2 - 2.4)[e]
	alpha-HCH [h]	27	ng/g ww	6(22)	1.4 (1.2 - 1.5) nd - 4.8	2.3 (2.2 - 2.4)[e]
	gamma-chlordane	27	ng/g ww	3(11)	1.7 (1.5 - 1.9) nd - 4.7	3.0 (2.9 - 3.2)[e]
	sumPCB [i]	27	ng/g ww	2(7)	1.3 (1.0 - 1.5) nd - 10.3	2.3 (2.2 - 2.4)[e]
TCDD-TEQ [j]		2	pg TEQ/g ww	2(100)	0.530, 0.797	<1.0

[a] ww=wet weight
[b] Total number of samples with values above the mdl and (percent)

Table 3.1 continued

[c] Geometric means, unless otherwise indicated, with (95% confidence intervals) and (ranges) for sample sizes greater than two; for sample sizes of two, both values are presented; nd = below minimum detection limit (mdl); concentrations below the mdl were assigned a value of ½ mdl for statistical calculations.
[d] Concentration in kidney; arithmetic mean % moisture (95% confidence interval) of kidney samples = 77.8% (77.2 - 78.4)
[e] Arithmetic mean (95% confidence interval)
[f] Concentration in mesenteric fat; arithmetic mean % lipid (95% confidence interval) of fat samples (based on ww) = 65.2% (60.3 - 70.2)
[g] Hexachlorobenzene
[h] alpha-hexachlorocyclohexane
[i] Sum 24 polychlorinated biphenyl (PCB) congeners (8, 18, 28, 29, 44, 50, 52, 66, 77, 87, 101, 104, 105, 118, 126, 128, 138, 153, 170, 180, 187, 195, 206, 209); minimum detection limit used is the mean mdl of all components used in the sum
[j] sum of World Health Organization 2,3,7,8-TCDD Toxic Equivalence Factors for PCDD/Fs and coPCBs for mammals (van den Berg et al., 1998); mean % lipid of fat samples = 63.5%.

Table 3.2. Univariate analyses of health parameters as a function of age, gender and season in caribou from the George River herd, Labrador, in 2001.

	Health parameters[a]				
	kidney fat index (KFI)	*Fascioloides magna* [b]		cestode	
		prevalence	abundance	prevalence	abundance
Age (years)	ns	INF>nINF [c] <0.01	0.53 <0.01	nINF>INF [d] <0.2	ns
Gender	ns	ns	ns	ns	ns
Season	winter>fall[e] <0.05	ns	ns	ns	winter>fall[f] <0.05

[a] Statistical tests used were Pearson's correlation and *t*-test. Results are presented as correlation coefficient and *p*-value in the Table or as geometric (KFI and cestode abundance) or arithmetic (age) means and (95% confidence interval) for each group below; ns = not statistically significant, $p>0.2$.

[b] Prevalence and abundance calculations for *F. magna* are based on the presence and number of migrating and encapsulated flukes

[c] Age of *F. magna* infected caribou: 3.9 years (2.4 - 5.3); non-infected caribou: 1.2 years (0 - 2.4)

[d] Age of cestode infected caribou: 2.4 years (0.6 - 4.1); non-infected caribou: 3.4 years (2.4 - 4.4)

[e] KFI of winter caribou: 59.9 (42.1 - 85.1); fall caribou: 40.2 (29.8 - 54.4)

[f] Cestode mean abundance in winter caribou: 1.1 (0.3 - 2.4); fall caribou 0.3 (0.0 - 0.7)

INF=infected

nINF=not infected

compared to fall caribou. No associations were found between health parameters and gender. Trends toward simple associations ($0.05<p<0.2$) are listed in Table 3.2.

Results of simple associations between contaminant concentrations and age, season and gender showed that renal cadmium concentrations increased with increasing age ($r=0.73$, $p<0.01$), illustrated graphically in Figure 3.1. Higher levels of renal mercury were found in animals sampled in the fall (0.80 µg/g ww; 95% CI: 0.68 - 0.91) compared to the winter (0.52 µg/g ww; 95% CI: 0.43 - 0.60) ($p<0.001$). However, HCB concentrations in fat were higher in winter caribou (27.4 ng/g ww; 95% CI: 24.1 - 30.6) compared to fall caribou (21.3 ng/g ww; 95% CI: 17.4 - 25.3) ($p<0.01$).

Table 3.3 shows results of simple associations between health parameters (KFI and hepatic parasite infection) and contaminants. Caribou infected with *F. magna* had significantly higher renal cadmium levels than uninfected animals and as the abundance of flukes increased, renal cadmium concentrations increased as well. Renal cadmium levels were also significantly higher in caribou infected with cestode larvae. Other trends toward simple associations ($0.05<p<0.2$) between these health parameters and various contaminants are listed in Table 3.3.

Table 3.4 shows results of univariate analyses among health parameters. The abundance of *F. magna* was significantly higher in caribou infected with cestode larvae compared to those free of cysts ($p<0.05$). Other trends towards simple associations ($0.05<p<0.2$) among health parameters are listed in Table

Figure 3.1 The relationship between renal cadmium concentration and age in caribou collected from the George River herd in 2001 in Labrador, Canada.

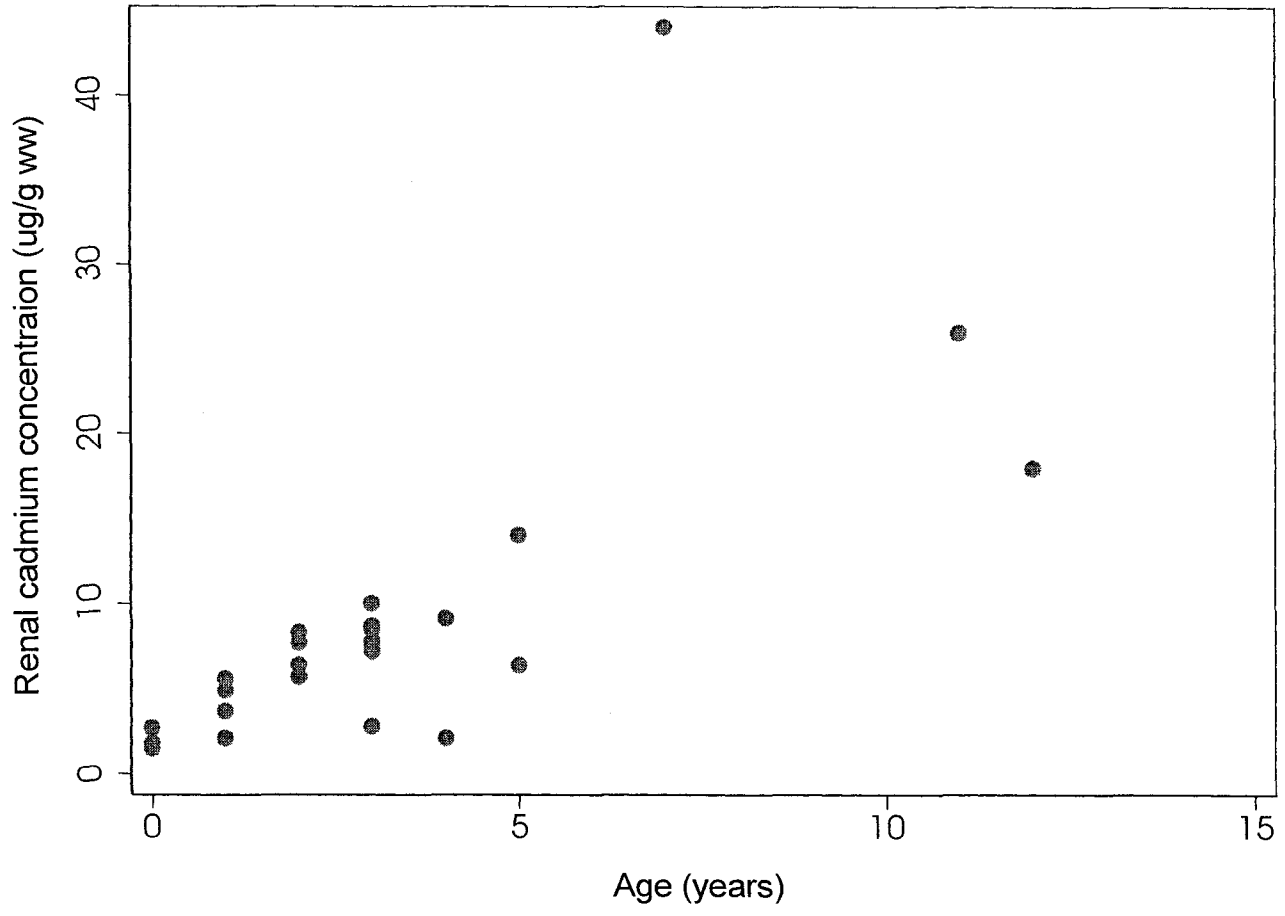


Table 3.3. Univariate analyses of health parameters as a function of tissue contaminant concentrations in caribou from the George River herd, Labrador, in 2001.

Contaminant [b]	Health parameters [a]				
	kidney fat index (KFI)	*Fasioloides magna* [c]		cestode	
		prevalence	abundance	prevalence	abundance
cadmium	ns	INF>nINF [d] <0.01	0.68 <0.01	INF>nINF[e] <0.05	ns
lead	ns	INF>nINF [f] <0.2	0.37 <0.1	ns	-0.26 <0.2
mercury	ns	nINF>INF [g] <0.2	ns	ns	ns
selenium	0.36 <0.1	ns	ns	INF>nINF[h] <0.1	0.39 <0.2
HCB	0.31 <0.2	ns	ns	ns	0.27 <0.2
alpha-HCH	ns	ns	ns	ns	ns

[a] Statistical tests used were Pearson's correlation and *t*-test (cadmium, lead, mercury, selenium, HCB) and Spearman correlation and Kruskal-Wallis test (alpha-HCH). Results are presented as correlation coefficient and *p*-value in the Table or as geometric (cadmium and lead) or arithmetic (mercury and selenium) means and (95% confidence interval) for each group below; ww=wet weight; ns = not statistically significant, $p>0.2$.

[b] HCB=hexachlorobenzene (ng/g ww); alpha-HCH=alpha-hexachlorocyclohexane (ng/g ww)

[c] Prevalence and abundance calculations for *F. magna* are based on the presence and number of migrating and encapsulated flukes.

[d] Renal cadmium concentration in *F. magna* infected caribou: 8.0 µg/g ww (5.7 - 11.1); non-infected caribou: 4.7 µg/g ww (1.7 - 6.0)

[e] Renal cadmium concentrations in cestode infected caribou: 8.4 µg/g ww (5.7 - 12.5); non-infected caribou: 4.6 µg/g ww (2.8 - 7.5)

[f] Renal lead concentrations in *F. magna* infected caribou: 0.10 µg/g ww (0.06 - 0.15); non-infected caribou: 0.07 µg/g ww (0.03 - 0.16)

[g] Renal mercury concentrations in *F. magna* infected caribou: 0.64 µg/g ww (0.54 - 0.75); non-infected caribou: 0.72 µg/g ww (0.57 - 0.87)

[h] Renal selenium concentrations in cestode infected caribou: 1.24 µg/g ww (1.17 - 1.31); non-infected caribou: 1.16 µg/g ww (1.08 - 1.24)

INF=infected; nINF=not infected

Table 3.4. Univariate analyses among health parameters (kidney fat index [KFI] and *Fascioloides magna* and cestode infection) in caribou from the George River herd, Labrador, in 2001.

Health parameters [a]		KFI	*F. magna* [b]	
			prevalence	abundance
KFI		-	ns	0.29
cestode	prevalence	INF>nINF [c] <0.1	ns	<0.2 INF>nINF [d] <0.05
	abundance	ns	ns	ns

[a] Statistical tests used were Pearson's correlation and *t*-test. Results are presented as correlation coefficient and *p*-value in the Table or as geometric means and (95% confidence interval) for each group below; ns = not statistically significant, $p>0.2$.
[b] Prevalence and abundance calculations for *F. magna* are based on the presence and number of migrating and encapsulated flukes.
[c] KFI in cestode infected caribou: 57.4 (43.4 - 75.9); non-infected caribou: 43.8 (31.6 - 60.6)
[d] *F. magna* abundance in cestode infected caribou: 7.8 (2.6 - 19.4); non-infected caribou: 2.3 (0.7 - 6.0)
INF=infected; nINF=not infected

3.4. Among individual contaminants, renal selenium and cadmium concentrations were positively correlated (r=0.47, $p<0.05$).

3.3.3 Multiple variable regression analyses

Multiple regression analyses were performed for KFI and *F. magna* and cestode infection (prevalence and abundance) based on the results of univariate analyses with contaminants and potential confounders. Significant final models are presented in Table 3.5.

Season, kidney selenium concentration, HCB concentration in fat, *F. magna* abundance, cestode prevalence and a quadratic form of age were found to be associated with KFI in univariate analyses (Tables 3.2, 3.3 and 3.4), and therefore were offered as independent variables for the regression analysis. Multiple linear regression analysis results showed that the only significant predictor of KFI was age (Table 3.5). Figure 3.2 shows the relationship between KFI and age including the predicted values for KFI based on the regression model presented in Table 3.5. Based on this small sample size, it appears as though young adults (3 to 7 year-old) attain the highest levels of KFI relative to younger and older animals.

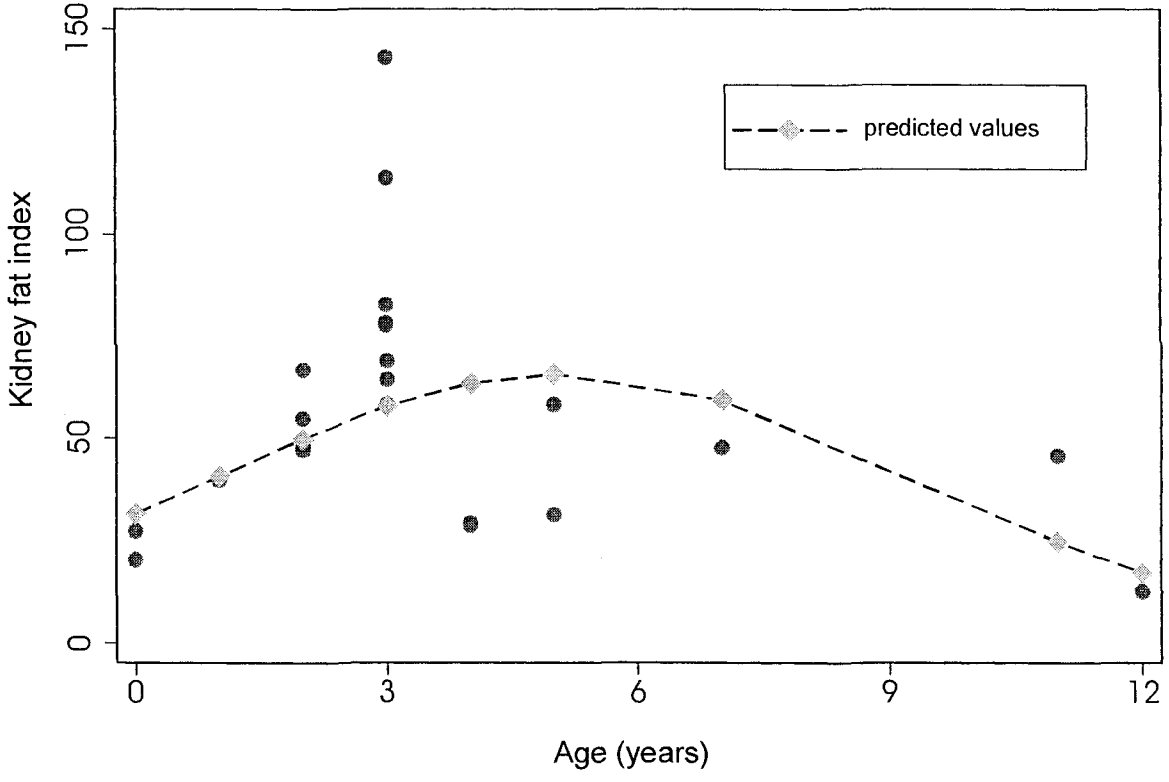
In univariate analyses, variables found to be associated with *F. magna* abundance were age, KFI, renal lead and cadmium concentrations and cestode prevalence (Tables 3.2, 3.3 and 3.4). These variables were offered as independent variables for the regression analysis, along with kidney selenium concentration which was significantly associated with renal cadmium concentration (based on a *t*-test; $p<0.05$) and was felt to be a potentially

Table 3.5. Results of multiple regression analyses for kidney fat index (KFI) and *Fascioloides magna* infection in caribou from the George River herd, Labrador, in 2001.

Health parameter	Regression Analysis	Coefficient (SE)	Regression equation [a]	Intercept	F-value	Model *p*-value	*n*	Adjusted R^2
ln(KFI)	linear	0.12 (0.05) -0.03 (0.01)	age (age - χ age)^2	3.69 (0.16)	6.81	0.005	24	0.34
ln(*F. magna* abundance)	linear	1.69 (0.32) -4.02 (1.83)	ln(cadmium) (µg/g ww) selenium (µg/g ww)	3.28 (1.99)	14.2	<0.0001	26	0.51
F. magna prevalence	logistic	2.05 (0.88)	ln(cadmium) (µg/g ww)	-2.03 (1.38)	7.24 [b]	<0.01	27	0.26 [c]
	logistic	1.00 (0.47)	age	-0.89 (0.95)	7.90 [b]	<0.01	26	0.28 [c]

[a] p-values for coefficients were all significant at $p \leq 0.05$
[b] LR test (chi-square) value
[c] Pseudo R^2

Figure 3.2 The predicted relationship between kidney fat index (KFI) and age in caribou collected from the George River herd in 2001 in Labrador, Canada.



confounding variable. Multiple linear regression analysis for *F. magna* abundance showed renal cadmium and selenium levels were both significant independent variables (Table 3.5). For this sample of caribou, as cadmium levels increased and selenium levels decreased, *F. magna* abundance increased. Standard regression diagnostic procedures confirmed a reasonable fit of all final models.

Age and renal cadmium, lead and mercury concentrations were associated with prevalence of *F. magna* in univariate analyses (Tables 3.2 and 3.3) and, therefore, were offered as independent variables for the regression analysis along with renal selenium concentration (for similar potential confounding reasons). Depending on which variable was entered into the model first, two separate significant final models were produced, each with a single significant predictor. Renal cadmium concentration and age were each found to be significant independent variables in separate models for *F. magna* prevalence on multiple logistic regression analysis, and both predictors had a positive relationship with *F. magna* prevalence (Table 3.5). For this sample of caribou, animals with high levels of renal cadmium and older animals had increased probability of *F. magna* infection. Standard regression diagnostic procedures confirmed a reasonable fit of all final models.

Multiple logistic (prevalence) and linear (abundance) regression analyses for cestode infection did not result in significant final models that fit the data and, therefore, are not presented.

3.4 Discussion

Renal cadmium concentrations were similar to those found in earlier studies on the GRH (Crête et al., 1989; Robillard et al., 2002) and elsewhere (Elkin and Bethke, 1995; Larter and Nagy, 2000). Increasing levels of cadmium seen with increasing age have been widely reported in caribou and other ungulate species (Crête et al., 1989; Dietz et al., 1998). Kidney concentrations of mercury, lead and selenium in our study were low and fell within the ranges previously reported in the GRH (Robillard et al., 2002) and other herds (Elkin and Bethke, 1995).

Levels of OCs found in the sampled caribou were low and appeared to be within the range of those reported in other Canadian herds (Elkin and Bethke, 1995; Hebert et al., 1996). Hexachlorobenzene was the most abundant OC contaminant detected in our study and has been reported in Arctic terrestrial herbivores (Braune et al., 1999). The levels of OCs found are well below those reported to be associated with adverse health effects in mammals (de March et al., 1998) and are unlikely to cause health problems in the caribou tested.

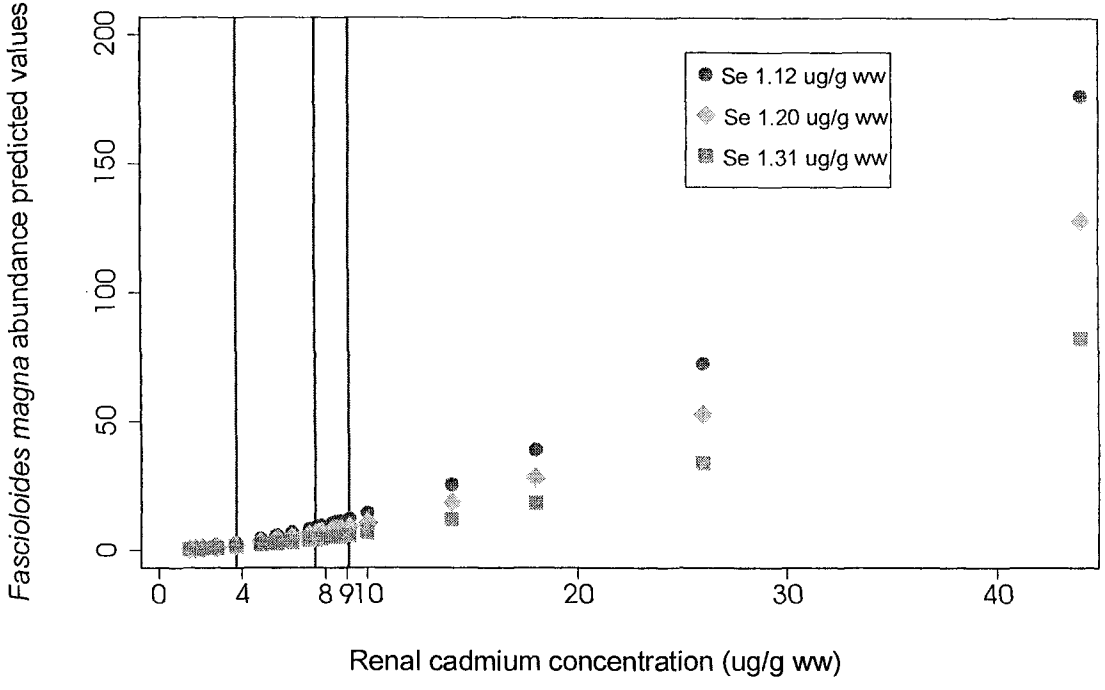
The GRH is the only caribou herd in North America endemically infected with *F. magna* (Wobeser et al., 1985). Other definitive hosts include wapiti and white-tailed deer (Pybus, 2001). The prevalence of *F. magna* infection in the caribou sampled (78%) was significantly higher ($p<0.05$, based on χ^2 statistic) than the prevalence found in other studies on the GRH conducted in the 1980s. Lankester and Luttich (1988) found a prevalence of 58%, Huot and Beaulieu (1985) 49%, and, based on the presence of capsules only, Parker (1981) found

a prevalence of 15%. The mean intensity of infection in this study was similar to that reported by Lankester and Luttich (1988). Increased prevalence of *F. magna* may occur with increased definitive host density, however, other factors such as the local conditions of wetland habitats, ungulate use and movements, and seasonal variations in temperature and moisture, may also play a role (Pybus, 2001).

Based on multiple linear regression analysis, *F. magna* abundance was best predicted by renal cadmium and selenium concentrations, as illustrated in Figure 3.3. The relationship was apparent at relatively low levels of cadmium (<10 µg/g ww). Selenium levels had an apparent protective effect on *F. magna* infection when cadmium levels were considered. Increasing cadmium concentrations from 3.7 µg/g ww to 9.1 µg/g ww (25^{th} to 75^{th} percentile) increased *F. magna* abundance from 3 flukes to 12 flukes when selenium concentrations were low (1.12 µg/g ww , 25^{th} percentile) and from 1 fluke to 6 fluke when selenium concentrations were high (1.31 µg/g ww, 75^{th} percentile).

Although age was associated with both renal cadmium and *F. magna* infection (older caribou had higher levels of cadmium and a greater abundance of *F. magna*), controlling for age did not significantly alter the final model. Age is clearly related to both cadmium levels and *F. magna* infection, however, it is unclear, based on the number of animals examined, whether cadmium is associated with *F. magna* infection independent of age (within age cohorts). For most age cohorts, only a small number of animals were examined (range 1 to 4)

Figure 3.3 The relationship between renal cadmium concentration and *Fascioloides magna* abundance at low (1.12 µg/g ww, 25th percentile), median (1.20 µg/g ww) and high (1.31 µg/g ww, 75th percentile) concentrations of renal selenium in caribou collected in 2001 in Labrador, Canada.



note: Solid lines indicate 25th (3.7 µg/g ww), median (7.5 µg/g ww) and 75th (9.1 µg/g ww) percentile concentrations of cadmium.

which precluded separate analyses within age groups. However, within the 3 - year-old cohort, cadmium and *F. magna* abundance were significantly positively correlated (r=0.68, $p<0.01$, n=8).

As found on univariate analyses, both age and cadmium concentration were found to be significant predictors in separate final models for *F. magna* prevalence on multiple regression analysis (Table 3.5). Again, it is unclear, based on the animals examined, whether cadmium levels are predictive of *F. magna* prevalence independent of age.

There has been inconsistent evidence in the literature of immunotoxic effects of cadmium in experimental animals with chronic cadmium exposure (Scheuhammer, 1987; Taylor et al., 1999) and the relationship between high cadmium levels and trematode parasite infection in caribou or other ungulate species has not previously been examined. The possibility of an increased susceptibility to *F. magna* infection in caribou with elevated levels of cadmium accumulation seems unlikely but cannot be entirely dismissed.

The positive role of selenium in host immunity is well documented in experimental and domestic animals (Dhur et al., 1990; Arthur et al., 2003), however, the protective role of selenium in relation to parasite infection has not been convincingly demonstrated. Findings indicate that selenium deficiency results in immunosuppression, and supplementation with low doses of selenium increases immune function. However, variations in experimental results indicate that the relationship between selenium levels, immune response and protective immunity is complex (Radostitis et al., 2000). It is unclear whether these

findings can be extrapolated to free-ranging wildlife such as caribou. Our results showed that selenium levels were not by themselves associated with *F. magna* infection. However, when cadmium and selenium levels were considered together, as selenium levels decreased abundance of *F. magna* increased. It may be that any immunotoxic effects of Cd, as demonstrated by increasing *F. magna* abundance may be more pronounced when Se levels are lower.

Host-parasite interactions are complex and can be affected by a variety of host, parasite and environmental factors (Wakelin, 1996; Hoberg et al., 2001). It has been established that, in general, the host immune response is an important regulator of parasitic infections in mammals (Wakelin, 1996). The specific immunoparasitology of *F. magna* in caribou, however, has not been examined. Pybus (2001) ventures that "there is little evidence to suggest an effective immune response to *F. magna* infection in most hosts".

On univariate analysis, renal cadmium and selenium levels were positively correlated ($r=0.47$, $p<0.05$). In free-ranging mammalian and avian species with elevated total mercury concentrations in the liver, selenium levels were generally strongly positively correlated, often at a 1:1 molar ratio (Eisler, 1985; Scheuhammer et al., 1998). The beneficial effect of selenium relative to mercury toxicity has been well documented (Goyer, 1995; Thompson, 1996) and is believed to result from the formation of toxicologically inert Hg-Se complexes (Scheuhammer et al., 1998). However, a similar protective relationship has not been consistently reported with cadmium. Investigations into the association

between selenium and cadmium in horses (Junnila et al., 1987) and rats (Jamall and Smith, 1985) have been equivocal.

An alternative explanation for the association seen between renal cadmium concentrations and *F. magna* infection could relate to concomitant exposure of caribou to both cadmium and *F. magna* larvae while foraging. Pybus (2001) suggested that the variation in prevalence and intensity of *F. magna* infection within host populations may reflect individual differences in the use of emergent vegetation (and thus exposure to infective stages of the parasite). However, there is no evidence that this foraging behaviour would also increase an animal's exposure to cadmium.

Although evidence for immunotoxicity with cadmium exposure in animals is scant, the potential for renal toxicity has been well established. The first site of injury from cadmium toxicity in mammals is the renal proximal convoluted tubule (Alden and Frith, 1991). The threshold for significant renal tubular damage in mammals and birds is generally reported as 100-200 µg/g ww (Cooke and Johnson, 1996). However, a specific threshold level for cadmium toxicity in most wildlife species, including caribou, has not been evaluated.

In experimental animals, the threshold for renal tubular injury has been shown to vary considerably with species and route of exposure (Alden and Frith, 1991). Microscopic and biochemical evidence of damage has been reported in rats with chronic cadmium exposure at renal concentrations as low as 2 - 4 µg/g ww (Brzóska et al., 2003). There is also evidence that exposure to cadmium

can result in disturbances in calcium balance and decreases in bone density (Taylor et al., 1999).

Relatively few field studies have demonstrated tissue injury due to cadmium toxicity in free-ranging wildlife (Beyer, 2000), and there have been no published reports of adverse health effects due to cadmium toxicity in wild ungulates, despite markedly elevated tissue concentrations in some animals (Dietz et al., 1998). However, such studies rarely involve microscopic or ultrastructural examination of affected organs.

Recent studies examining the relationship between cadmium levels and biological effects in free-ranging species have shown varied results. Larison et al. (2000) examined white-tailed ptarmigan from the Colorado ore belt and found that those with renal cadmium concentrations >100 µg/g ww had histopathological evidence of renal injury. Ptarmigan with renal cadmium concentrations above this threshold level also had reduced concentrations of skeletal calcium compared to controls. In a study conducted on ringed seals from Greenland, Sonne-Hansen et al. (2000) found no evidence of cadmium-induced renal toxicity or skeletal demineralisation. Mean renal cadmium concentration in the seals examined was 44.5 (±40.8) µg/g ww.

The highest concentration of renal cadmium in the sampled caribou was 44 µg/g ww, less than half of the commonly reported toxic renal threshold (100-200 µg/g ww). On microscopic examination of kidneys from all caribou sampled, no evidence of proximal tubular injury was found, although some degree of autolysis and freezing artifacts may have masked subtle changes.

Other pathological findings in the caribou were limited to hepatic lesions due to *F. magna* infection which were grossly evident. The most severe infection found was 67 flukes (30 capsules). While hepatic lesions were extensive, they were felt to be clinically insignificant, as sufficient hepatic reserves were apparent. Although the presence of flukes in the liver causes noticeable damage, significant pathological findings have not been reported in definitive hosts except with severe infections. Mortality due to infection was reported in wapiti with >500 flukes (Pybus, 2001). Mulvey and Aho (1993) reported subtle negative health effects such as decreased weight gains in young male white-tailed deer with moderate to heavy infections of *F. magna* (>10 flukes) compared to uninfected or lightly infected deer (0-10 flukes). No reports of morbidity or mortality due to *F. magna* infection in caribou have been published (Pybus, 2001). No association was found between *F. magna* infection and body condition as measured by the KFI. Similarly, Lankester and Luttich (1988) found no association between *F. magna* infection and back fat depth in the GRH. Due to the cross-sectional nature of the present study, differences in weight gains could not be examined.

Neither tissue contaminant levels nor hepatic parasite infection appeared to have a negative impact on the body condition of the caribou in this study. However, due to the relatively small sample size (n=27), there was a limited power to detect more than a few significant associations with KFI. On regression analysis, the only significant predictor of KFI in caribou was age, with young adults (3 to 7 year-old) attaining the highest levels of KFI relative to

younger and older animals, demonstrating that seasonal differences found were mainly due to different ages of caribou killed in the two seasons. A similar parabolic relationship between KFI and age was seen in female caribou from the GRH by Parker (1981), and Thomas et al. (1989) noted that KFI decreased with increasing age in adult female barren-ground caribou.

Fifty percent of the caribou that were examined were infected with cestode cysts in the liver that were consistent with the larval form of *Taenia hydatigena*. *T. hydatigena* has a wide range of intermediate hosts, including domestic livestock and wild cervids, and prevalence and intensity differ among different hosts species (Jones and Pybus, 2001). Accounts of clinically significant lesions due to *T. hydatigena* in wild intermediate hosts are rare. Addison et al. (1979) found no association between physical condition of moose in Ontario and intensity of *T. hydatigena* infection. Similarly, no significant lesions were found in relation to *T. hydatigena* cysts in the liver in these caribou, and there was no association between indices of infection and KFI in the final model.

3.4.1 Conclusions

The sampled caribou appeared to have sufficient fat reserves for the time of year, with the greatest fat reserves found in young adults (3 to 7 years old) compared to younger and older caribou. Lesions identified in carcasses included two occurrences of chronic interstitial nephritis, as well as chronic hepatitis in those caribou infected with *F. magna*. None of these lesions was felt to have been of clinical significance and statistical analysis of pathological findings was not performed due to the small number of lesions found.

The levels of detected contaminants fell within the reported ranges for caribou in Canada and elsewhere. All contaminants were well below the reported thresholds for toxic effects in other species, although renal cadmium levels approached the reported toxic threshold in one case, which was expected. There was no evidence of renal tubular toxicity due to chronic cadmium exposure, however, most caribou examined were young (≤5 years), with relatively low levels of renal cadmium.

Age was associated with renal cadmium levels and *F. magna* infection (older caribou had increased levels of both), and, on multiple regression analysis, *F. magna* abundance was best predicted by renal cadmium and selenium levels. It is unclear, based on the small number of animals examined, whether a relationship between *F. magna* infection and cadmium levels exists independent of age, however, the possibility of some degree of immunotoxicity related to cadmium levels (possibly tempered by selenium concentrations) cannot be ruled out entirely, and further investigations in larger populations of animals may be warranted.

The caribou we examined were representative of those typically harvested by Innu hunters. Caribou are an important source of country food for Innu people in the area. While this study does not directly address the suitability of caribou for human consumption relative to tissue contaminant levels, the results may be useful in assisting Labrador Innu communities in making this type of risk determination.

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4. HEALTH PARAMETERS OF PORCUPINES IN RELATION TO TISSUE CONCENTRATIONS OF ENVIRONMENTAL CONTAMINANTS

4.1 Introduction

The porcupine is a large rodent native to North America whose range includes nearly all of Canada, with the exception of the high Arctic regions and some more southern islands such as Newfoundland (Dodge, 1982). Porcupines are terrestrial herbivores present throughout most of Labrador, Canada. In winter, their diet consists of cambium layer and inner bark of trees as well as new twigs and buds while in spring and summer, a variety of ground vegetation is also consumed (Banfield, 1974; Dodge, 1982). In Labrador, porcupines are hunted regularly for food by Innu people. The harvest, sharing, processing and consumption of country food (such as porcupines) contribute to the spiritual and cultural identity of Aboriginal peoples (Van Oostdam et al., 1999) including the Innu (Armitage, 1990).

In recent years, several northern terrestrial and marine animal species have been the focus of research on contaminants. A wide range of environmental contaminants such as organochlorine pesticides, polychlorinated biphenyls (PCBs) and metals (eg. mercury and cadmium) have been found in northern ecosystems, often far from their sources of discharge into the environment (AMAP, 2002; CACARII, 2003). The presence of many of these contaminants in the North is a concern because of their persistence and the potential for accumulation in animals, particularly top predators such as polar bears (Braune et al., 1999; Muir et al., 1999). Although environmental

contaminants are generally less of a concern in terrestrial herbivores, some species, such as caribou, have been found to accumulate high levels of cadmium in their kidneys and liver (Crête et al., 1989; Gamberg and Scheuhammer, 1994). Smaller herbivores, such as snowshoe hare and ptarmigan, have also been found to carry high cadmium concentrations in areas with mineral-rich soils (Dietz et al., 1998; Larison et al., 2000). There is little known about environmental contaminants in porcupines. One small study in the Yukon tested four porcupines and found moderate levels of cadmium in the liver (Muir et al., 1997). In general, there has been relatively little attention focused on environmental contaminants in terrestrial wildlife species in Labrador. Also, although the accumulation of these environmental contaminants in some species has been well-documented, relatively little research has been directed at assessing the relationship between contaminant levels and various biomarkers and health parameters in northern ecosystems. However, there are some recent exceptions (eg. Sonne-Hansen et al., 1999; Wayland et al., 2001, 2002, 2003; Kuzyk et al., 2003). In general, a knowledge gap remains concerning the biological effects of these contaminants in northern wildlife species (CACAR II, 2003).

This study was carried out in Labrador in collaboration with the Innu Nation in order to evaluate the health of important wildlife species hunted by Labrador Innu, including porcupines, in relation to tissue environmental contaminant concentrations. Specifically, the objectives were:

1 - to assess the health of porcupines killed during the regular seasonal hunt by determining body condition and intestinal helminth parasite burdens and examining selected tissues for gross and microscopic pathological abnormalities;
2 - to determine the tissue concentrations of a wide range of organochlorine and metal contaminants in these porcupines; and
3 - to examine the relationship between important health parameters and tissue contaminant levels in the porcupines through multiple regression analyses.

4.2 Materials and methods

4.2.1 Porcupine locations and sample collection

In order to make sure that the results were representative of the types of animals that make up the Innu harvest, the sample population consisted of twenty-nine porcupines that were killed by a blow to the head with a blunt instrument, usually a large stick, during the regular subsistence hunt by residents of the Labrador Innu community of Sheshatshit. The porcupines were collected in south-central Labrador (approximately 62°W to 67°W and 53°N to 54°N) (Figure 1.1, Page 3) during the periods 7 - 12 October 2000 and 22 September to 16 October 2001 (fall, n= 23), and 16 - 19 May 2001 (spring, n=6).

All methods for collection, handling, processing and disposal of samples were agreed upon by the Innu Nation co-researchers, community elders, hunters and their family members, and the researchers from the AVC. In order to retain strong community acceptance of the research and its findings, a high degree of importance was placed on maintaining the traditions surrounding the hunt, and

ensuring that the porcupines were still considered to be suitable for consumption and community distribution after sampling.

Once killed, the porcupines were brought to permanent or temporary camps in the hunting area to be cleaned and were either eaten on site or brought back to the community for distribution. Each porcupine was cleaned within 48 hours of being killed by removing the intestines through an abdominal incision, and samples were taken as described below.

During cleaning of the carcass, any visible abnormalities were noted, and comments regarding the condition of the porcupine from the hunter or person cleaning the carcass were recorded. In most cases, samples were taken by researchers as the porcupines were being cleaned, however, due to lack of proximity, for three of the porcupines, the animals were cleaned by the participating hunter or family member, and samples were later obtained by researchers from the discarded gastro-intestinal tract.

On site, gross examination of each porcupine was limited to the exterior of the animal and, in most cases, intraabdominal organs (intestinal tract and usually, but not always, stomach, liver, uterus). Intestines and livers were collected and either processed on site or stored in a plastic bag with or without aluminum foil wrapping (not rinsed in hexane) and kept on snow or ice for up to 24 hours until they could be frozen at -20˚C. At the AVC, within three months of the sample collection, the collected tissues and organs were thawed and examined in more detail for gross abnormalities. Mesenteric fat and livers were sampled for organochlorine analysis and were manipulated using stainless steel

instruments rinsed three times in hexane, re-wrapped in aluminum foil triple-rinsed in hexane and re-frozen at -20°C. A portion of each liver collected (at least 10 g) was submitted for metal analysis. Grossly abnormal tissues were sampled for standard microscopic examination.

Two porcupines that were considered to be unfit (thin for the time of year) for consumption by the hunter and/or the person cleaning the animal (generally an older female family member) were taken to the AVC for necropsy. A full post mortem examination was performed on the remains of each of these animals, and tissues were sampled for standard microscopic examination in order to determine the cause of the perceived abnormalities.

For each porcupine, body condition was assessed subjectively on the basis of the amount of adipose tissue present in the mesentery or subcutaneous and/or perirenal fat stores. As the carcasses were examined and mesenteric fat was dissected from the intestines, a body condition score (BCS) was assigned to 28 porcupines on a scale of 0 to 4: 0 - no fat present; 1 - slight or questionable fat present; 2 - fat present but not conspicuously so; 3 - conspicuous fat; 4 - abundant fat. For the purposes of statistical analyses, the assigned body condition scores were dichotomized into body condition categories of "poor to fair body condition" (BCS = 1 or 2) and "moderate to good body condition" (BCS = 3 or 4). Due to an oversight during field collection, one porcupine did not receive a specific body condition score at the time of sample collection.

For intestinal helminth parasite recovery, the descending colon was discarded except for a segment containing the 5 - 10 most proximal pellets. The

intestines were opened longitudinally with scissors within a 10 L bucket, and the mucosa was bluntly scraped between thumb and forefinger and rinsed into the bucket. Once thoroughly rinsed, the intestine was discarded. Water was added to the bucket to bring the volume to 10 L and the contents were stirred vigorously with a spatula for at least 1 minute. Immediately after stirring, a 10 % aliquot was taken using a 1 L bucket. The sample was run through a 150 or 250 µm sieve and was placed in a water-tight plastic container and preserved for later examination by adding a measured volume of 37% formaldehyde for a final concentration of 10% formalin.

For quantification and identification of intestinal helminths, an additional aliquot of either 4% or 10% was taken from each sample and examined under a stereoscopic microscope at a magnification of six power. Individual nematodes were counted and approximately 50 males and 50 females recovered for identification. Cestode numbers were considered unreliable due to varying levels of post mortem autolysis to which these parasites are particularly sensitive. All scolices found and a representative sample of cestode segments were recovered for identification. Results obtained from the aliquots were used to estimate the abundance of intestinal nematode parasites of each porcupine.

4.2.2 Analysis of tissue contaminants

See section 3.2.2 for specific analytical methodologies.

Samples of liver were analysed at the AVC Diagnostic Toxicology Laboratory for cadmium, lead and selenium and at Philip Analytical Services (Halifax, Nova Scotia, Canada) for total mercury. Mesenteric fat and liver

samples were analysed for organochlorine (OC) pesticides and PCBs at the Environmental Quality Laboratory (Environment Canada, Moncton, New Brunswick, Canada). Liver was tested rather than kidney because results from a previous study showed moderate levels of cadmium in four porcupine livers (Muir et al., 1997).

Due to cost restrictions, only samples of mesenteric fat tissue from two porcupines (one in poor body condition and the other in moderate body condition) were analysed for seven polychlorinated dibenzo-*p*-dioxins (PCDDs), ten polychlorinated dibenzofurans (PCDFs) and three non-ortho substituted, or coplanar, PCBs (coPCBs) at Axys Analytical Services Ltd. (Sidney, British Columbia, Canada). The 2,3,7,8-TCDD Toxic Equivalencies (TCDD-TEQs) for PCDD, PCDF and coPCB data were reported using World Health Organization toxic equivalency factors (WHO TEFs) for mammals (van den Berg et al., 1998).

4.2.3 Statistical analysis

The following parasitological parameters were determined for intestinal parasites: prevalence (number of porcupines infected with a particular parasite/number of porcupines examined) for nematodes and cestodes and abundance (number of a particular parasite counted in each porcupine) and mean abundance (total number of a particular parasite counted/number of porcupines examined) for nematodes only, following Bush et al. (1997). Cestode numbers were considered unreliable due to varying levels of post mortem autolysis to which these parasites are particularly sensitive, therefore, only the prevalence (based on the presence of segments) was determined.

Contaminant values less than minimum detection limits (mdl) were replaced with ½ of the mdl. For sumPCB and sumCHL, because they were the summation of individual components, each having its own minimum detection limit, the mean minimum detection limit of the components of the sum were used as the mdl for the sum. Because the mdl for PCDD, PCDF and coPCBs were very low, values that were less than the mdl were considered to be insignificant and were not included in the TCDD-TEQ calculations.

Geometric (for non-normally distributed variables) or arithmetic (for normally distributed variables) means and 95% confidence intervals were used for descriptive statistics.

Pearson's correlation coefficients were determined among the contaminants measured in order to identify significant correlations between individual contaminants. Contaminants detected in at least 20% of the samples tested (by tissue) (mercury, selenium, cadmium, and alpha-HCH in liver and HCB, alpha-HCH, sumCHL and sumPCB in fat) were included in univariate statistical analyses described below.

Some of the health parameter (intestinal nematode abundance) and contaminant (liver selenium and liver and fat alpha-HCH concentrations) variables were transformed (natural log [ln]) to meet assumptions of normality for the parametric statistical tests used. Non-parametric statistical tests were used for those variables for which ln transformation did not result in a normal distribution (cadmium and sumCHL and sumPCB concentrations).

Simple associations among health parameters, contaminants and confounding variables (season, gender) were examined using χ^2 statistics for categorical variables (body condition, gender, season), pair-wise Pearson's correlation coefficients and *t*-tests for normally distributed variables and Spearman's correlation and Kruskal-Wallis test for non-normally distributed variables. We reported results of $p<0.05$ as statistically significant. For analyses including health parameters, we also reported trends ($0.05<p<0.2$) in order to account for any near-significant relationships. Those contaminants and potential confounders found to have unconditional associations with health parameters at $p<0.2$ were included as potential predictor variables in multiple variable regression analyses (Dohoo et al., 2003).

Multiple variable regression analyses were used to simultaneously determine associations between health parameters and contaminant levels, controlling for possible confounding factors. Only body condition and intestinal nematode parasite abundance were used as health parameters for regression analyses. The prevalence of both total nematode and total cestode intestinal parasites was 100%. Gross and microscopic lesions were rare thus the power to detect significant risk factors for these health parameters would be limited.

Multiple logistic (for body condition) and linear (for intestinal nematode abundance) regression models were developed to determine the associations between health parameters and contaminant levels. Those variables found to be significant at $p\leq 0.2$ on univariate analysis with the health parameters were offered to the regression models in a forward, manual stepwise process. Two-

way interaction variables between significant main effects were assessed, where applicable. The final models included those variables that were significant at $p \leq 0.1$ in order to insure that all potential associations were considered and to account in part for the relatively small samples sizes.

Goodness of fit of the final models was determined using standard regression diagnostic procedures (including analysis of residuals, leverage values, Cook's distance [linear regression], Pearson and deviance residuals [logistic regression]). Statistical analyses were completed using STATA (Statistical Package, v.8.0; Stata Press, College Station, Texas, U.S.A.).

4.3 Results

4.3.1 Descriptive results

Of the 29 porcupines sampled, 13 (45%) were female, 11 (38%) were male and 5 (17%) were of unknown gender. Twenty (69%) of the porcupines sampled were killed in fall 2001, 3 (10%) were killed in fall 2000 and the remainder in spring 2001. For subsequent analyses, fall porcupines included those collected in fall 2000 and 2001 as no difference was found in gender ($\chi^2 = 2.25$, d.f=1, $p>0.1$) or body condition score ($\chi^2 = 4.49$, d.f.=3, $p>0.2$) between the two years. No difference was seen in the gender distribution between seasons ($\chi^2 = 0.504$, d.f.=1, $p>0.2$).

4.3.1.1 Health parameters

The two thin porcupines were both females; one was killed in the fall, the other in the spring. On post-mortem examination, no significant abnormalities were identified that could account for the poor condition of these animals. The

remaining 27 porcupines were considered to be healthy and suitable for eating by the participating Innu families, and no significant gross or microscopic lesions were noted in these animals. Based on mesenteric fat reserves, body condition scores ranged from 1 to 4 with 17 (61%) porcupines considered to be in poor to fair body condition (BCS=1 or 2) and 11 (39%) considered to be in moderate to good body condition (BCS=3 or 4). Body condition score was not determined for one porcupine because of an error during field collection.

Intestinal tracts from two of the porcupines were not recovered intact, thus, only 27 of the 29 porcupines were examined for intestinal helminth parasites. All intestines examined contained large numbers of grossly visible nematodes and cestodes (Figure 4.1). All nematodes were identified as the oxyurid *Evaginuris evaginata*. Nematode abundance ranged from 14,900 to 288,800 (geometric mean = 89,100; 95% CI 67,350 - 117,890). The cestodes were identified as members of the anoplocephalid parasite family commonly found in porcupines, *Monoecocestus americana* and *M. variabilis*.

4.3.1.2 Tissue prevalence and concentrations of contaminants

Tables 4.1 and 4.2 provide summaries of the prevalence and concentrations of metal, OC pesticide and PCB contaminants in fat and/or liver of the porcupines. TCDD-TEQs in fat tissue from two porcupines are also reported. Liver tissue was not collected from 12 porcupines following the wishes of the individual hunters and there was insufficient liver tissue from one other porcupine for organochlorine analyses. Two of the porcupines had insufficient fat reserves for contaminants analysis. Therefore, there were 17 samples of

Figure 4.1 Two portions of porcupine intestine showing intraluminal contents, including large numbers of nematode (white arrows) and cestode (black arrowheads) parasites.

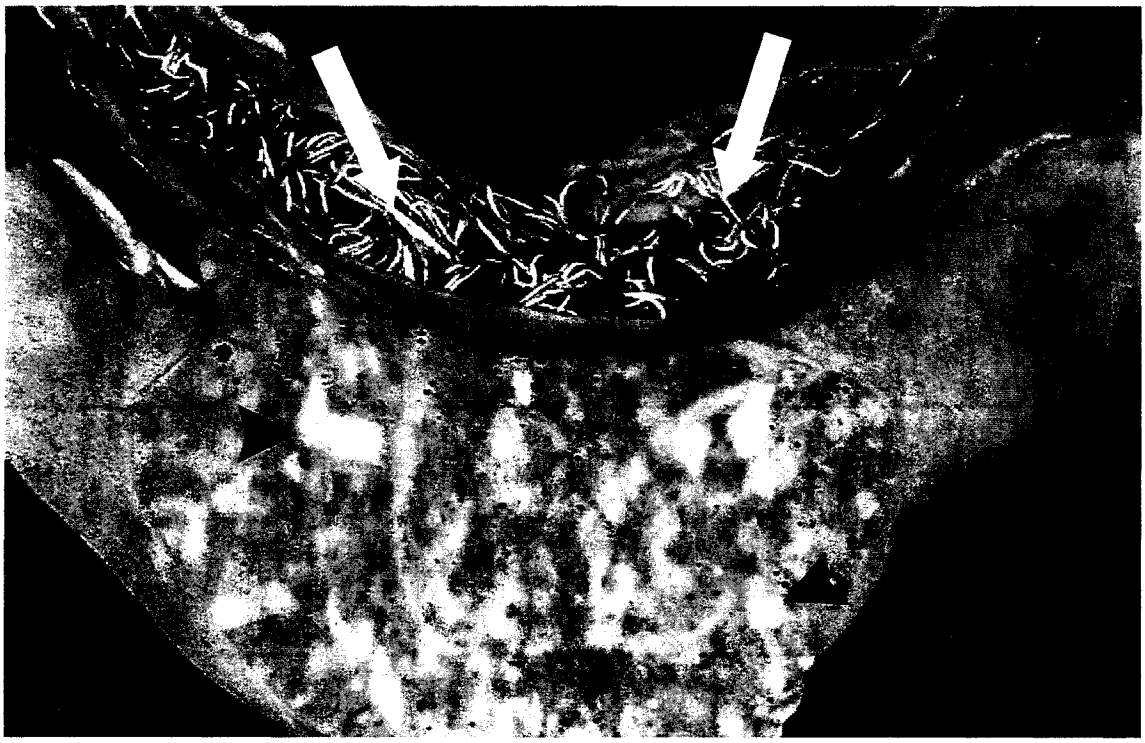

Table 4.1. Concentrations of metals and selenium in liver of porcupines from south-central Labrador in 2000 - 2001.

Contaminant	*n*	number above mdl[a]	concentration[b] µg/g ww	mdl µg/g ww
mercury	17	15(88)	0.02 (0.01 - 0.02) (nd - 0.40)	0.01
cadmium	17	6(35)	0.32 (0.22 - 0.46) (nd - 2.2)	0.4
lead	17	2(12)	0.05, 0.31	0.05
selenium	17	17(100)	0.21 (0.14 - 0.33) (0.06 - 0.91)	0.05

[a] Total number of samples with values above the mdl (percent)
[b] Arithmetic (mercury) or geometric (cadmium, lead, selenium) means with (95% confidence intervals) and (ranges) for sample sizes greater than two; for sample sizes of two, both values are presented; nd = below minimum detection limit (mdl); concentrations below the mdl were assigned a value of ½ mdl for statistical calculations; ww = wet weight; arithmetic mean % moisture (95% confidence interval) of 16 liver samples = 73.9% (72.4 - 75.3)

Table 4.2. Concentrations of organochlorine contaminants in liver and mesenteric fat tissue of porcupines from south-central Labrador in 2000 - 2001.

Contaminant		Porcupines tested in total		Liver samples			
		n	number above mdl[b]	*n*	number above mdl[c]	concentration[a] ng/g ww[d]	mdl ng/g ww[e]
organochlorine pesticides[g]	HCB	29	21(72)	16	0	nd	0.8 (0.7 - 0.9)
	HE	29	1(3)	16	1(6)	2.5	1.6 (1.4 - 1.7)
	trans nonachlor	29	1(3)	16	0	nd	0.8 (0.7 - 0.9)
	gamma-CHL	29	2(7)	16	0	nd	1.1 (0.9 - 1.2)
	cis-CHL	29	4(14)	16	0	nd	0.8 (0.7 - 0.9)
	sumCHL[h]	29	7(24)	16	1(6)	2.5	1.1 (0.9 - 1.2)
	p,p'-DDE	29	2(7)	16	0	nd	1.3 (1.2 - 1.5)
	o,p'-DDT	29	1(3)	16	1(6)	4.7	2.4 (2.1 - 2.6)
	sumDDT[i]	29	3(10)	16	1(6)	4.7	2.1 (1.9 - 2.3)
	alpha-HCH	29	15(52)	16	6(38)	0.56 (0.43 - 0.73) (nd - 1.9)	0.8 (0.7 - 0.9)
PCBs	sumPCB[j]	29	6(21)	16	0	nd	0.8 (0.7 - 0.9)
TCDD-TEQ[k]		2	100	0	na	na	na

Table 4.2 continued

Contaminant		Porcupines tested in total		Fat samples			
		n	number above mdl[b]	*n*	number above mdl[c]	concentration ng/g ww[f]	mdl ng/g ww[e]
organochlorine pesticides[g]	HCB	29	21(72)	27	21(78)	2.6 (2.0 - 3.3)[e] (nd - 6.9)	2.0 (1.7 - 2.2)
	HE	29	1(3)	27	0	nd	3.9 (3.5 - 4.3)
	trans nonachlor	29	1(3)	27	1(4)	2.4	2.0 (1.7 - 2.2)
	gamma-CHL	29	2(7)	27	2(7)	2.7, 6.5	2.6 (2.3 - 2.9)
	cis-CHL	29	4(14)	27	4(15)	1.1 (0.9 - 1.3) (nd - 3.5)	2.0 (1.7 - 2.2)
	sumCHL[h]	29	7(24)	27	6(22)	1.5 (1.2 - 1.9) (nd - 6.5)	2.6 (2.3 - 2.9)
	p,p'-DDE	29	2(7)	27	2(7)	2.8, 3.1	3.3 (2.9 - 3.6)
	o,p'-DDT	29	1(3)	27	0	nd	5.9 (5.2 - 6.5)
	sumDDT[i]	29	3(10)	27	2(7)	2.8, 3.1	5.2 (4.7 - 5.8)
	alpha-HCH	29	15(52)	27	12(44)	1.4 (1.1 - 1.9) (nd - 4.1)	2.0 (1.7 - 2.2)
PCBs	sumPCB[j]	29	6(21)	27	6(22)	1.2 (1.0 - 1.5) (nd - 5.0)	2.0 (1.7 - 2.2)
TCDD-TEQ[k]		2	100	2	2(100)	0.19, 0.35 [l]	<1.0

[a] Geometric means, unless otherwise indicated, with (95% confidence intervals) and (ranges - for concentration column only) for sample sizes greater than two; for sample sizes of one or two, values are presented; nd=below minimum detection limit (mdl); concentrations below the mdl were assigned a value of ½ mdl for statistical calculations; na=not applicable.

[b] Number of porcupines with contaminant concentrations above the mdl in either liver or fat tissue and (percent).

Table 4.2 continued

[c] Number of samples with contaminant concentrations above the mdl for that tissue and (percent).
[d] ww = wet weight; arithmetic mean % lipid (95% confidence interval) of liver samples (based on wet weight) = 2.7% (2.3 - 3.0)
[e] Arithmetic mean (95% confidence interval)
[f] Arithmetic mean % lipid (95% confidence interval) of fat samples (based on wet weight) = 51.6% (44.2 - 59.1)
[g] HCB=Hexachlorobenzene; HE=heptachlor epoxide; gamma-CHL= gamma-chlordane; *cis*-CHL=*cis*-chlordane; alpha-HCH=alpha-hexachlorocyclohexane;
[h]Sum chlrodanes = sum of (HE, *trans* nonachlor, gamma- & *cis*-CHL); minimum detection limit used is the mean mdl of all components used in the sum
[i] Sum DDT and metabolites (*p,p'*-DDE and *o,p'*-DDT) ; minimum detection limit used is the mean mdl of all components used in the sum
[j] Sum 24 polychlorinated biphenyls (PCB) congeners (8, 18, 28, 29, 44, 50, 52, 66, 77, 87, 101, 104, 105, 118, 126, 128, 138, 153, 170, 180, 187, 195, 206, 209); minimum detection limit used is the mean mdl of all components used in the sum
[k] sum of World Health Organization 2,3,7,8-TCDD Toxic Equivalence Factors for PCDD/Fs and coPCBs for mammals (van den Berg et al., 1998), nd=0; mean % lipid of fat samples = 44%.
[l] pg TEQ/g ww

liver tested for metals, and 16 samples of liver and 27 samples of fat tested for OC pesticides and PCBs. All 29 porcupines had at least one tissue (liver or fat) tested for OC pesticides and PCBs, producing overall denominations of 29 for Table 4.2.

A majority of the liver samples tested had detectable levels of mercury, all with concentrations near the detection limit. Cadmium and lead were detected in a smaller proportion of the porcupines tested, and detectable levels of selenium were found in all of the liver samples (Table 4.1).

Of the 20 individual OC pesticides measured in fat and liver samples, eleven (aldrin, dieldrin, mirex, methoxychlor, gamma-HCH, alpha-& beta-endosulfan and *o,p*' & *p,p*'-DDD, *o,p*'-DDE and *p,p*'-DDT) were not detected in any of the samples tested. The most prevalent organochlorine was HCB which was detected in 78% of fat samples, but none of the liver samples. Alpha-HCH was detected in 52% of porcupines overall in either fat or liver samples, or both (Table 4.2).

PCDD/Fs and coPCBs were found in both of the porcupines in which they were tested with minimum detection limits of <0.07 pg/g wet weight (ww) for PCDD/Fs and <2.6 pg/g ww for coPCBs (Table 4.2). The majority of the TCDD-TEQ was comprised of coPCBs (79% and 92%) with PCB 126 being the primary contributing congener (77% and 87% of the total TCDD-TEQ, respectively).

4.3.2 Univariate analytical results

Seasonal differences were found with the two main health parameters (body condition and intestinal nematode abundance). A greater proportion of porcupines killed in the spring were in poor to fair body condition compared to those killed in the fall (χ^2 =3.94, d.f=1, p<0.05). In addition, spring porcupines had significantly greater intestinal nematode abundance (geom. mean=160,090; 95%CI: 95,810 - 267,520) compared to fall porcupines (geom. mean=78,000; 95% CI: 57,230 - 106,280) (t-test, p<0.01). No associations were found between these health parameters and gender.

Table 4.3 shows simple associations (p<0.05) between contaminant concentrations and season and gender. Hepatic selenium concentrations were higher in fall porcupines compared to spring porcupines, however, spring porcupines had higher levels of hepatic alpha-HCH than fall porcupines. Gender differences were also seen with higher hepatic alpha-HCH concentrations in females compared to males and higher levels of mesenteric fat sumCHL in males compared to females.

Table 4.4 lists the results of simple associations between health parameters and contaminant levels. Liver cadmium concentrations were significantly higher in porcupines in poor to fair body condition compared to those in moderate to good body condition, and there was a significant positive correlation between liver alpha-HCH concentrations and intestinal nematode abundance. No significant simple associations (p<0.05) between health parameters and concentrations of

other contaminants were found. Trends towards simple associations ($0.05<p<0.2$) are listed in Table 4.4.

Among contaminants, significant positive associations were found between mesenteric fat alpha-HCH and HCB (r=0.49, $p<0.05$), sumCHL (r=0.61, $p<0.001$) and hepatic cadmium levels (r=0.75, $p<0.01$). A significant positive association was also found between mesenteric fat sumCHL and liver cadmium concentration (r=0.52, $p<0.05$).

In examining the relationship between body condition and intestinal nematode abundance, a trend was found with a greater abundance of intestinal nematodes in animals in poor to fair body condition (geom. mean=101,292 ; 95% CI 72,662 - 141,202) compared to those in moderate to good body condition (geom. mean=68,505; 95% CI 38,378 - 122,282) (*t*-test, $p=0.11$).

4.3.3 Multiple variable regression analyses

Multivariate regression analyses were performed for the two main health parameters: body condition and intestinal nematode abundance.

The variables found to be associated ($p<0.2$) with body condition on univariate analysis (season, cadmium concentration in liver, sumPCB and HCB concentration in fat and intestinal nematode abundance) (Table 4.4 and Section 4.3.2) were included in the full multiple logistic regression model. The regression analysis did not result in any significant final models ($p<0.05$) that fit the data. However, both season and liver cadmium concentration predicted the outcome perfectly (all fall porcupines were in poor to fair body condition and all porcupines in moderate to good body condition had hepatic cadmium concentrations below

Table 4.3. Univariate analyses of contaminants as a function of season and gender of porcupines from south-central Labrador in 2000 - 2001.

Contaminant[a]	season	gender
mercury (liver)	ns	ns
cadmium (liver)	ns	ns
selenium (liver)	fall>spring[b]	ns
HCB (fat)	ns	ns
sum CHL (fat)	ns	male>female[c]
alpha-HCH (fat)	ns	ns
alpha-HCH (liver)	spring>fall[d]	female>male[e]
sumPCB (fat)	ns	ns

[a] Statistical tests used: t-test (selenium, mercury, HCB, alpha-HCH) and Kruskal-Wallis test (cadmium, sumCHL, sumPCB); results are presented as arithmetic (mercury and HCB) or geometric (cadmium, selenium, sumCHL, alpha-HCH and sumPCB) means and (95% confidence interval) for each group below.
[b] Hepatic selenium concentration in fall porcupines: 0.28 µg/g wet weight (0.17 - 0.45); spring porcupines: 0.09 µg/g wet weight (0.05 - 0.16), $p>0.05$.
[c] Mesenteric fat sumCHL concentration in males: 2.0 ng/g wet weight (1.3 - 3.2); females: 1.2 ng/g wet weight (0.9 - 1.6), $p>0.05$.
[d] Hepatic alpha-HCH concentration in spring porcupines: 0.9 ng/g wet weight (0.8 - 1.0); fall porcupines: 0.5 ng/g wet weight (0.4 - 0.7), $p>0.05$.
[e] Hepatic alpha-HCH concentration in females: 0.7 ng/g wet weight (0.4 - 1.1); males: 0.4 ng/g wet weight (0.3 - 0.6), $p>0.05$.
ns= not significant, $p>0.05$.

Table 4.4. Univariate analyses of health parameters (body condition and intestinal nematode abundance) of porcupines from south-central Labrador in 2000 - 2001 as a function of tissue contaminant concentrations.

contaminant [a]	body condition	intestinal nematode abundance
mercury (liver)	ns	ns
cadmium (liver)	poor > good [b]	ns
selenium (liver)	ns	ns
HCB (fat)	poor > good [c]	ns
sum CHL (fat)	ns	ns
alpha-HCH (fat)	ns	ns
alpha-HCH (liver)	ns	0.64 [e]
sumPCB (fat)	poor > good[d]	ns

[a] Statistical tests used: t-test, Pearson's correlation (mercury, selenium, HCB, alpha-HCH) and Kruskal-Wallis test, Spearman's correlation (cadmium, sumCHL, sumPCB); results are presented as *p*-value and correlation coefficient in the Table or arithmetic (mercury and HCB) or geometric (selenium, cadmium, alpha-HCH, sumCHL and sumPCB) means and (95% confidence interval) for each group below.
[b] Hepatic cadmium concentration in porcupines in poor to fair body condition: 0.39 µg/g wet weight (0.21 - 0.70); moderate to good condition: 0.2 µg/g wet weight (0.2 - 0.2), $p<0.05$.
[c] Mesenteric fat HCB concentration in porcupines in poor to fair body condition: 3.4 ng/g wet weight (2.4 - 4.4); moderate to good condition: 2.6 ng/g wet weight (1.6 - 3.6), $p<0.2$.
[d] Mesenteric fat sumPCB concentration in porcupines in poor to fair body condition: 1.5 ng/g wet weight (1.1 -2.1); moderate to good condition: 1.1 ng/g wet weight (0.9 - 1.3), $p<0.2$.
[e] $p<0.05$
ns= not significant, $p>0.2$.

the minimum detection limit). Therefore these variables should be considered significant correlates of body condition, although interaction and confounding between the two variables could not be assessed due to the nature of the data.

The variables found to be associated ($p<0.2$) with intestinal nematode abundance on univariate analysis (season, body condition and liver alpha-HCH concentration), as well as the interaction term season*body condition were offered to the full multiple linear regression model (Table 4.4 and Section 4.3.2). The regression analysis resulted in a final model with only one significant predictor of intestinal nematode abundance: liver alpha-HCH concentration. The final regression equation with coefficient (SE) and intercept (SE) was:

ln(nematode abundance) = 0.90 (0.31) ln[alpha-HCH] + 11.95 (0.24)

(overall F-value= 8.78 and overall p-value<0.05; n=15; adjusted R^2=0.36)

Standard regression diagnostic procedures were used to confirm a reasonable fit of the final model.

4.4 Discussion

Little comparative data for contaminants exists for porcupines. Porcupines are terrestrial herbivores, and many contaminant-related studies have focused primarily on predator species such as marine mammals and piscivorous birds which tend to accumulate relatively high levels of contaminants through the food web (Braune et al., 1999; Muir et al., 1999). In the Yukon, high cadmium levels have been found in other terrestrial herbivores, such as caribou, moose, ptarmigan and snowshoe hare (Gamberg, 1996 reported in AMAP 1998). These high levels are thought to be linked to mineral-rich soils in the area and

accumulation of cadmium in plants eaten by these species (Dietz et al., 1998). One country food contaminant study in the Yukon examined four porcupines which were found to have moderate levels of cadmium in liver: 38.4 ± 24.3 µg/g dry weight (Muir et al., 1997), approximately five times higher than the porcupines sampled in this study. In eastern Canada, high cadmium levels have been found in large terrestrial herbivores, such as caribou and moose (Brazil et al., 1989; Crête et al., 1989). However, this accumulation has not been shown to occur in smaller terrestrial herbivores. Porcupines in the present study had cadmium, mercury and lead levels similar to or slightly lower than small herbivores (willow ptarmigan and snowshoe hare) in Québec (Langlois and Langis, 1995).

The low levels of OC contaminants found in the present study, including the most prevalent, HCB and alpha-HCH, are consistent with what has been found in many other northern terrestrial herbivore species in Canada (Elkin and Bethke, 1995; Braune et al., 1999). The levels of organochlorines in four Yukon porcupines were generally low (<10 ng/g ww), and only one porcupine had levels of PCDDs and PCDFs >1 pg/g, although the tissue tested and level found was not specified (Muir et al., 1997).

The concentrations of all metal and organochlorine contaminants in these porcupines, including alpha-HCH and cadmium, were near or below the detection limits. Although there are no data available on thresholds for biological effects of these contaminants in porcupines, the concentrations found were much lower than threshold levels reported for other mammalian or avian species, often by a

factor of 1000 or more, particularly for OC compounds (deMarch et al.,1998; CACARII, 2003).

Overall, it was found that spring porcupines were thinner and had higher intestinal nematode abundance than those collected in the fall. In more southern ranges, porcupines of all ages and both sexes have been shown to undergo severe nutritional stress in late winter (Roze, 1989; Sweitzer and Berger, 1992, 1993), and a similar seasonal depletion of nutritional reserves was seen in our study. It is commonly known that intestinal nematodes are ubiquitous and abundant in porcupines (Dodge, 1982; Roze, 1989). In a study of Innu zoology, Innu informants from Québec also described the two types of intestinal parasites that we found in porcupines (Clément, 1995). However, the association between season and intestinal nematode abundance has not been previously reported in porcupines.

It was found that thinner porcupines had higher hepatic cadmium concentrations. Liver cadmium levels were reported per gram of tissue. Thus, any condition that could result in a decrease in size of the liver (such as loss of nutritional condition) with no change in the absolute concentration of cadmium would result in an increased relative cadmium concentration per gram of tissue. It is possible that this was a factor in the porcupines examined.

On multiple regression analysis, concentration of hepatic alpha-HCH was found to be the only significant predictor of intestinal nematode abundance. For the porcupines sampled, increasing levels of alpha-HCH were associated with increasing abundance of intestinal nematodes. However, data for both liver

contaminant concentrations and health parameters were collected for only a subset of porcupines (n=16 for body condition and n=15 for intestinal parasites), and only six porcupines had detectable levels of these contaminants. Although statistically significant, the associations found between liver contaminant concentrations and health parameters are based on a small number of porcupines that may not be representative of the population as a whole.

Large numbers of nematodes and cestodes were found in the intestines of every porcupine examined. In general, references to these parasites in porcupines in the literature suggest that they are likely innocuous (Dodge, 1982; Roze, 1989), perhaps because of their ubiquitous nature. However, Roze (1989) points out that the health impacts of these parasites on their hosts have not been evaluated. Innu people participating in the study who were experienced with cleaning porcupines were very familiar with these intestinal parasites, and those who commented on them, attached little importance to them regarding the health of the animals.

In domestic production animals such as sheep and cattle, the impact of subclinical infections by intestinal helminth parasites on growth and production have been well established (Sykes, 1978; Radostits et al., 1994), and intestinal helminth infections can cause chronic and debilitating disease in humans and experimental animals (Gause et al., 2003). Parasites, including intestinal helminths, obtain all the energy needed for growth, reproduction and survival from their host (Wakelin, 1996). Factors that may influence the number of parasites in an individual animal include the degree of exposure to infective

stages of the parasite and the immunological response of the host (Wakelin, 1996; Gause et al., 2003). In turn, host immune response can vary with individual genetic variation (Wakelin and Apanius, 1997), as well as with age, health or diet differences (Wakelin, 1996). Due to the scope and collaborative nature of this study, some potentially influencing factors, such as age, could not be assessed.

The large numbers of intestinal helminth parasites found in all porcupines examined indicate that these animals are able to function with remarkable parasite burdens. It seems unlikely that these parasites are entirely innocuous given the energy requirements of such a large parasitic biomass. In the winter and spring, when food availability is diminished and energy demands are high, these parasites could have greater impact on the nutritional condition of the porcupines. It is quite conceivable that a high number of intestinal nematodes could negatively affect the nutritional condition of the porcupine hosts. However, based on the small number of animals examined in this study, a significant association between intestinal nematode numbers and fat reserves was not found, although, there was a trend towards greater numbers of intestinal nematode parasites in thinner porcupines. Season was a highly significant predictor of body condition in the porcupines studied and season was also found to be a significant predictor of nematode abundance. The small sample size and the subjective measure of body condition, could have reduced the ability to detect significant associations between nematode abundance and body condition.

Although some statistically significant associations were found between contaminant levels and health parameters, it appears unlikely that they are biologically relevant as, overall, the tissue concentrations of contaminants were very low. In a study of apparently healthy glaucous gulls, Sagerup et al. (2000) found a positive association between intestinal nematode intensity and a variety of OC pesticides and PCBs. However, the concentrations of organochlorines in the gulls were up to 10,000 times greater than the levels found in the porcupines.

4.4.1 Conclusions

As expected, spring porcupines were thinner than those killed in the fall, and all porcupines had large numbers of intestinal nematode parasites. In addition to being thinner, spring porcupines also had a greater abundance of intestinal nematodes. The biological significance of the high intestinal parasite loads and of the seasonal variation in nematode abundance is unclear. Although some associations were found between contaminant levels and health parameters (body condition and intestinal nematode parasites), it is unlikely that these associations are biologically significant as metal and OC contaminants were generally near or below minimum detection limits in liver and fat tissues for a large part of the porcupine population sampled.

Porcupines are an important source of country food for Innu people in Labrador. Although this study does not directly address the question of the suitability of porcupine for human consumption relative to tissue contaminant levels, the results may be useful in assisting the participating Innu communities in making this type of risk determination.

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5. HEALTH PARAMETERS OF CANADA GEESE IN RELATION TO TISSUE CONCENTRATIONS OF ENVIRONMENTAL CONTAMINANTS

5.1 Introduction

The Canada goose is a migratory waterfowl species found throughout North America (Mowbray et al., 2002). The breeding area for the North Atlantic Population (NAP) includes Labrador (CWS, 2000; Mowbray et al., 2002) where it is an important species for subsistence hunting by Innu in both spring and fall seasons. The NAP of Canada geese overwinters on the eastern seaboard of the United States (Mowbray et al., 2002) where, like other species that use the Atlantic flyway for migration, these birds are more likely to be exposed to several persistent environmental contaminants. Agricultural grains are important in the winter diet of Canada geese while leaves, stems and berries from a variety of aquatic and terrestrial plants are consumed during the breeding season (Bellrose, 1978; Mowbray et al., 2002)

In northern ecosystems, persistent environmental contaminants, such as metals, organochlorine pesticides, PCBs, PCDD/PCDFs and radionuclides, have been the focus of recent research programs in Canada and worldwide (CACAR, 1997; AMAP, 1998). Although many persistent organochlorines, such as PCBs and DDT, were severely restricted or banned in North America in the 1970s, elevated tissue concentrations of some of these compounds have been found, even in recent years. Observed levels in northern Canada are a concern to the health of animals and people who consume them (CACAR, 1997).

In North America, waterfowl hunted in eastern regions have been shown to have higher tissue concentrations of metal and organochlorine contaminants than in more western regions (Foley, 1992; Braune et al., 1999). Although studies have documented tissue organochlorine and metal residue levels in Canada geese and other avian species in Arctic and subarctic Canada, there have been relatively few studies that have examined possible links of these contaminants to biological effects (Muir et al., 1999). Little attention has been focused on environmental contaminants in waterfowl in Labrador.

This study was carried out in collaboration with the Innu Nation in order to evaluate the health of important species hunted by Labrador Innu, including Canada geese, in relation to tissue contaminant concentrations. Specifically, the objectives were:

1- to assess the health of Canada geese killed during the regular seasonal hunt by determining body condition and intestinal helminth parasite burdens and examining selected tissues for gross and microscopic pathological abnormalities;

2 – to determine the tissue concentrations of a wide range of organochlorine and metal contaminants in the geese; and

3 - to examine the relationship between important health parameters and tissue contaminant levels in the geese through multiple regression analyses.

5.2 Materials and methods

5.2.1 Geese locations and sample collection

In order to make sure that the results were representative of the types of animals that make up the Innu harvest, the sample population consisted of forty-

three Canada geese that were killed with shotguns during the regular seasonal hunt by residents of the Labrador Innu communities of Sheshatshit and Utshimassit (Davis Inlet). The geese were killed in south-central and coastal-northern Labrador (approximately 60° W to 67° W and 53° N to 56° N) (Figure 1.1, page 3). Samples were collected from geese killed between May 12 - 20, 2001 (spring, n=29) and September 6 - 17, 2001 (fall, n=14).

All methods for collection, handling, processing and disposal of samples were agreed upon by the Innu Nation co-researchers, community elders, hunters and their family members, and the researchers from the AVC. In order to retain strong community acceptance of the research and its findings, a high degree of importance was placed on maintaining the traditions surrounding the hunt and ensuring that the geese were still considered to be suitable for consumption and community distribution after sampling.

Once killed, the geese were brought to permanent or temporary camps in the hunting area or directly back to the community to be cleaned. Each goose was cleaned within 72 hours of being killed by removing the intestines through an incision made cranial to the vent, and samples were taken as described below.

During cleaning of the carcass, any visible abnormalities were noted and comments from the hunter or person cleaning the goose were recorded. In most cases, researchers were present when the geese were being cleaned. In some instances (n=15) where logistical problems meant that researchers could not accompany hunters, particularly with the fall geese, the intact geese were

examined and samples were taken by the hunters alone. Once cleaned, geese were either eaten on site or brought back to the communities for distribution.

On site, gross examination of the geese was limited to the exterior of the carcass and the abdominal contents (intestines, and usually, but not always, liver, gizzard, and testes or ovary and oviduct). The abdominal contents were collected from each goose, wrapped in aluminum foil (not rinsed in hexane), placed in a plastic bag and frozen immediately or placed on snow or ice for up to 72 hours until they could be frozen at –20°C. All samples were transported to the AVC for laboratory examination.

At the AVC, the tissues and organs were thawed and examined in more detail for gross abnormalities. Mesenteric fat and livers were collected for organochlorine analysis and were manipulated using stainless steel instruments rinsed three times in hexane, re-wrapped in aluminum foil rinsed three times in hexane, and re-frozen at –20°C. A portion of each liver (at least 10 g) was submitted for metal analysis at AVC. Grossly abnormal tissues were sampled for standard microscopic examination, including several sections of cloaca and two sections of liver.

Body condition was evaluated subjectively by the amount of adipose tissue present in the mesentery. As the mesenteric fat was dissected from the intestines, a body condition score (BCS) on a scale of 0 to 5 was assigned: 0 - no fat present; 1 - slight or questionable fat present; 2 - fat present but not conspicuously so; 3 - conspicuous fat; 4 - abundant fat; 5 - extreme fat.

For parasite recovery, the ceca were cut at their attachment to the ileum and each portion (ceca and small intestine) was placed in separate plastic buckets. Each portion was opened longitudinally with scissors and the mucosa was bluntly scraped between thumb and forefinger and rinsed into the bucket. Once thoroughly rinsed, the portion of cecum or small intestine was discarded. Cecal contents were sifted through a 106 µm sieve. Small intestinal contents were transferred to a 1 L glass beaker, water was added to a final volume of 500 ml, and the beaker was placed on a stir plate to thoroughly mix the contents for at least 2 minutes. Immediately after stirring, a 10% aliquot was taken, using a 50 ml beaker, sifted through a 106 µm sieve and placed in a water-tight plastic container for storage. After rinsing the sieve with copious amounts of water, a second 10% aliquot was taken in the same manner as the first. Cecal contents and the two aliquots of intestinal contents were preserved for later examination by adding a measured volume of 37% formaldehyde for a final concentration of 10% formalin.

One 10% aliquot of intestinal contents and the entire volume of cecal contents were examined under a dissecting microscope at 12 power for nematodes, trematodes and cestodes. All helminth parasites found were counted and recovered for identification, by site. Results obtained were used to estimate the abundance of intestinal helminth parasites of each goose.

5.2.2 Analysis of tissue contaminants

See section 3.2.2 for specific analytical methodologies.

Three of the forty-three geese were excluded from contaminant analyses for budgetary reasons. The excluded geese were felt to be typical of those sampled (killed in the spring and BCS of 3) and were from hunters who submitted multiple geese. For the remaining 40 geese, 38 had liver tissue collected. Samples of liver were analysed at the AVC Diagnostic Toxicology Laboratory for cadmium, lead and selenium, and at Philip Analytical Services (Halifax, Nova Scotia, Canada) for total mercury. Mesenteric fat samples and liver samples from the 40 geese were analysed for organochlorine pesticides and PCBs at the Environmental Quality Laboratory (Environment Canada, Moncton, New Brunswick, Canada). Liver tissue was chosen for analysis because metals tend to accumulate in both liver and kidney (Scheuhammer, 1987) and of the two organs, liver tissue was more readily sampled from geese that were killed and cleaned in a traditional manner. Moderate levels of mercury, selenium and cadmium in liver have been reported in some species of free-ranging waterfowl (Braune et al., 1999; Wayland et al., 2001).

Due to cost restrictions, a representative subsample of mesenteric fat samples (n=21) was analysed for PCDD, PCDF and coPCBs (coplanar or non-ortho substituted PCBs) (hereafter referred to as dioxin-tested geese) at Axys Analytical Services Ltd. (Sidney, British Columbia, Canada). The selection of the subsample of dioxin-tested geese was based in part on season as well as the availability of fat tissue. Of the 40 geese selected for OC and metal contaminant analysis, 12 had insufficient fat tissue collected for the additional dioxin analysis. Of the 28 remaining geese, in order to achieve a representative sample of both

seasons, all nine remaining fall geese were selected and 12 of the 19 remaining spring geese were randomly chosen. The 2,3,7,8-TCDD Toxic Equivalencies (TCDD-TEQs in fat) for PCDD, PCDF and coPCB data were reported using World Health Organization toxic equivalency factors (WHO TEFs) for birds (van den Berg et al., 1998).

5.2.3 Statistical analysis

5.2.3.1 Data management and descriptive statistics of health parameters and contaminants

The following parasitological parameters were determined: prevalence (number of geese infected with a particular parasite/number of geese examined), abundance (number of a particular parasite counted in each goose), mean abundance (total number of a particular parasite counted/number of geese examined) and mean intensity (total number of a particular parasite counted/number of infected geese) (hereafter referred to as intensity) of infection, following Bush et al. (1997). These measures of parasite infection were determined for nematodes, total trematodes (sum of all trematode species), trematode families or individual species with at least 20% prevalence and total helminths (sum of nematodes and total trematodes). Cestode numbers were considered unreliable due to varying levels of post mortem autolysis to which these parasites are particularly sensitive, therefore, only the prevalence (based on the presence of segments) was determined.

For the group of dioxin-tested geese, BCS was dichotomized. All geese in that group had a BCS of 3 or 4, with exception of two geese with a BCS of 5.

Thus, dioxin-tested geese with BCS of 4 and 5 were grouped together for statistical analyses.

Contaminant values less than minimum detection limits (mdl) were replaced with ½ of the mdl. For sumDDT, sumPCB and sumCHL, because they were the summation of their individual components, each having its own minimum detection limit, the mean minimum detection limit of the components of each sum was used as the mdl for each sum. Because the mdl for PCDD, PCDF and coPCBs were very low, values that were less than the mdl were considered to be insignificant and were not included in the TCDD-TEQ calculations.

For descriptive statistics, sumDDT included all *o,p'*- and *p,p'*- metabolites of DDT, including *p,p'*-DDE (the most prevalent metabolite). For univariate analyses, *p,p'*-DDE was analysed separately from the other metabolites of DDT in order to assess which metabolites (*p,p'*-DDE or the sum of the other DDT metabolites) were associated with the health parameters. Similarly, for descriptive statistics, sumCHL included all chlordanes and metabolites, including HE (the most prevalent component). However, for univariate analyses (for fat tissue only), HE was assessed separately and was excluded from the sum CHL.

Geometric (for non-normally distributed variables) or arithmetic (for normally distributed variables) means and 95% confidence intervals were used for descriptive statistics.

Pearson's (for normally distributed variables) and Spearman's (for non-normally distributed variables) correlation coefficients were determined among

the contaminants measured in order to identify significant correlations between individual contaminants.

5.2.3.2 Analytical statistics for associations between health parameters and contaminants/confounders

Only BCS and measures of intestinal helminth parasite infection (abundance and prevalence) were used as health parameters for regression analyses because parasite intensity levels only utilize data from infected geese and gross and microscopic lesions were rare limiting the power to find significant risk factors.

For two geese, there was only a liver specimen, and for two other geese, there was only a fat specimen, thus, an imputation strategy for contaminant results among liver and fat samples was employed in order to minimize the effect of these missing data on the power to detect associations for subsequent regression analyses. Estimates for the missing data for *p,p'*-DDE and dieldrin in liver and fat tissues were imputed using simple linear regression equations based on the contaminant concentrations in both tissues (ln transformed) for the entire sample. Pearson's correlation coefficients between fat and liver levels were 0.91 for dieldrin and 0.82 for *p,p'*-DDE. For the remaining contaminants, the proportion of liver samples with concentrations above the mdl was <50%, therefore, imputation between liver and fat samples was not attempted.

Contaminants detected in at least 20% of the samples tested (by tissue) (mercury, cadmium, lead, selenium, sumCHL, *p,p'*-DDE, sumDDT, and dieldrin in

liver, and HCB, HE, sumCHL, *p,p'*-DDE, sumDDT, dieldrin, sumPCB, and TCDD-TEQ in fat) were included in univariate statistical analyses.

Univariate analyses were conducted between health parameters, contaminants and potential confounders. For the subset of dioxin-tested geese, TCDD-TEQ concentrations were also included in the univariate analyses. Health parameter and contaminant variables were transformed (ln) to meet assumptions of normality for parametrical tests used. Simple associations among and between individual health parameters (BCS and intestinal helminth prevalence and abundance), contaminants and confounding variables (gender, season) were examined using χ^2 statistics for categorical variables, pair-wise Pearson's correlation coefficients and *t*-tests for normally-distributed continuous variables, and Spearman's correlation and Kruskal-Wallis test for non-normally distributed continuous variables (eg. hepatic cadmium, lead, sumDDT and sumCHL concentrations). Results of $p<0.05$ were reported as statistically significant. For analyses including health parameters, trends were also reported ($0.05<p<0.2$) in order to account for any near-significant relationships. Those contaminants and potential confounders found to have unconditional associations with health parameters at $p<0.2$ were offered as potential predictor variables to multiple variable regression analyses (Dohoo et al., 2003).

Multiple variable regression analyses were used to simultaneously determine associations between health parameters and contaminant levels, controlling for possible confounding factors. Multiple logistic (for intestinal helminth prevalence and BCS for dioxin-tested geese) and linear (for BCS and

intestinal helminth abundance) regression models were developed to determine the associations between health parameters and contaminant levels. Those variables found to be significant at $p<0.2$ on univariate analysis with the health parameters were offered to the full regression models in a manual, forward step-wise manner. Two-way interaction variables between significant main effects were assessed, where applicable. The final models included those variables that were significant at $p<0.1$ in order to insure that all potential associations were considered and to account in part for the relatively small sample sizes. For the subsample of dioxin-tested geese, only models for which TCDD-TEQ was found to be a significant predictor are reported because models that did not include TCDD-TEQ as a significant predictor did not differ substantially from the models developed with data from all geese.

Goodness of fit of the final models was determined using standard regression diagnostic procedures (including analysis of residuals, leverage values, Cook's distance [linear regression] and Pearson and deviance residuals [logistic regression]). All statistical analyses were completed using STATA (Statistical Package, v.8; Stata Press, College Station, Texas, U.S.A.).

5.3 Results

5.3.1 Descriptive results

Of the 43 Canada geese sampled, 29 (67%) were killed in spring and the remaining 14 (33%) were killed in fall. Of the spring geese, 13 (45%) were female, 14 (48%) were male, and 2 (7%) were of unknown gender. In the fall, it was often not possible to examine the entire carcass prior to sample collection

due to logistical difficulties with obtaining the entire carcass, and therefore gender data were missing for most (n=13). For similar reasons, age data were not collected for any of the geese.

5.3.1.1 Health parameters

The geese appeared to be in adequate body condition for the time of year, although a small number (three) were thinner than expected. Body condition scores were normally distributed with a range of 1 to 5 (mean 3.0; 95%CI: 2.7 – 3.3). BCS was strongly positively correlated with % lipid in fat (r=0.77) and liver (r=0.56) tissues ($p<0.01$). No significant lesions were noted in the animals and organs examined. All examined geese were considered to be healthy and suitable for eating by the hunters and their families.

Intestinal nematode, trematode and cestode data are presented in Table 5.1. Samples of intestinal parasites from eight geese were not recoverable due to marked post-mortem autolysis. The cecal sample from one goose was also lost, leading to complete intestinal samples from only 34 of the geese examined.

Helminth parasites were found in the intestinal tracts of all geese examined. Nematodes were the most prevalent group of helminths, (found in 100% of examined geese and therefore intensity was not applicable), and were also the most abundant (Table 5.1). Nematodes were found primarily in the ceca and were identified as *Trichostrongylus tenuis*.

Trematodes were found in 76% of geese. The most prevalent trematodes were the notocotylids (which include the species *Catatropis harwoodi* and *Notocotylus attenuatus)*, followed by the species *Zygocotyle lunata* and

Table 5.1. Descriptive statistics for intestinal helminth parasite prevalence, abundance and intensity of infection by parasite type for Canada geese killed in May and September 2001 in Labrador.

Helminth [a]	*n* infected	prevalence	abundance	intensity
nematodes	34	100	28.7 (19.0 - 43.3) (2 - 262)	na
total trematodes [b]	26	76 (61 - 91)	7.2 (3.1 - 16.2) (0 - 2972)	43.7 (16.2 - 117.8)
notocotylids [c]	17	50 (32 - 68)	1.6 (0.6 - 3.6) (0 - 460)	8.1 (3.4 - 19.1)
Zygocotyle lunata	10	29 (13 - 46)	0.3 (0.1 - 0.5) (0 - 8)	1.8 (1.1 - 2.9)
Paramonostomum alveatum	9	26 (11 - 42)	1.3 (0.3 - 3.7) (0 - 2512)	66.3 (16.3 - 270.2)
total helminths [d]	34	100	57.1 (36.7 - 88.7) (3 - 3021)	na
cestodes [e]	12	36 (20 - 52)	na	na

[a] Prevalence (%) or geometric mean and (95% confidence interval) and (range - for abundance only)
[b] sum of all trematode species
[c] notocotylids include the species *Catatropis harwoodi* and *Notocotylus attenuatus*
[d] sum of total trematodes and total nematodes
[e] cestode prevalence based on the presence of segments
na = not applicable

Parmonostomum alveatum. *Parmonostomum alveatum* was present in the greatest intensity, followed by the notocotylids and *Z. lunata* (Table 5.1). The distribution of trematodes differed between sites with *P. alveatum* dominating in the small intestine and the notocotylids dominating in the cecae. Nematodes and trematodes had low numbers present in most geese examined. Cestode segments were present in less than half of the geese. Other trematodes found in small numbers include cotylurids and echinostomes (identified as *Echinostoma trivolvis* based on adult worm morphology).

No lesions attributable to parasitism were noted on gross examination of intestinal muscosa, however, some degree of post-mortem autolysis was present in all intestines examined which may have masked subtle changes.

5.3.1.2 Tissue prevalence and concentrations of contaminants

Tables 5.2 and 5.3 provide a summary of prevalence and concentrations of metals, organochlorine (OC) pesticides, and PCBs in the liver and/or fat of the 40 selected geese. Prevalence and concentrations of PCDD/PCDFs and coplanar PCBs are also reported for the fat samples of the 21 selected geese receiving additional analyses. For two geese, there was insufficient fat tissue collected for analysis due to poor body condition of the geese, however, liver tissue from these geese was included for testing. For two other geese, the livers were not collected in order to follow the wishes of the individual hunters, but fat samples were analysed. Therefore, there were 38 samples of liver and fat tested for most contaminants and 21 samples of fat tested for specific dioxins, furans and

Table 5.2. Concentrations of selenium and metals in livers of Canada geese from Labrador in 2001.

Contaminant [a]	*n*	number above mdl[b]	concentration µg/g wet weight[c]	mdl µg/g wet weight
selenium	38	38 (100)	0.51[d] (0.46 - 0.57) (0.17 - 0.84)	0.1
mercury	37	33 (89)	0.01 (0.010 - 0.014) (nd - 0.03)	0.01
cadmium	38	10 (26)	0.3 (0.22 - 0.32) (nd - 1.3)	0.4
lead	38	12 (32)	0.04 (0.03 - 0.05) (nd - 0.31)	0.05

[a] Geometric means, unless otherwise indicated, with (95% confidence intervals) and (ranges - for concentration column only); nd = below minimum detection limit (mdl); concentrations below the mdl were assigned a value of ½ mdl for statistical calculations.
[b] Total number of samples with values above the mdl and (percent).
[c] Arithmetic mean % moisture (95% confidence interval) of liver samples = 69.5% (68.1 - 71.0)
[d] Arithmetic mean

Table 5.3. Concentrations of organochlorine contaminants in liver and fat tissue of Canada geese from Labrador in 2001.

Contaminant[a]		geese tested in total (liver or fat)		Liver				
		n	number above mdl [b]	*n*	number above mdl [c]	concentration ng/g ww [d]	range ng/g ww	mdl ng/g ww [e]
organochlorine pesticides [g]	HCB	40	27(68)	38	0	nd		0.9 (0.9 - 1.0)
	HE	40	20(50)	38	7(18)	1.2 (1.0 - 1.7)	(nd - 24.1)	1.8 (1.7 - 1.9)
	trans nonachlor	40	19(48)	38	2(5)	1.0 , 1.2		0.9 (0.9 - 1.0)
	gamma-CHL	40	4(10)	38	1(3)	1.3		1.2 (1.2 - 1.3)
	cis-CHL	40	2(5)	38	1(3)	0.8		0.9 (0.9 - 1.0)
	sumCHL [h]	40	24(60)	38	9(24)	1.0 (0.7 - 1.3)	(nd - 25.1)	1.2 (1.2 - 1.3)
	sum rest CHL [i]	nc	nc	nc	nc	nc		nc
	p,p'–DDE	40	38(95)	38	31(82)	7.5 (4.4 - 12.7)		1.5 (1.5 - 1.6)
	sumDDT [j]	40	38(95)	38	31(82)	9.0 (5.3 - 15.1)		2.4 (2.3 - 2.6)
	sum rest DDT [k]	40	26(65)	38	10(26)	2.3 (1.5 - 3.6)	(nd - 106.5)	2.4 (2.3 - 2.6)
	dieldrin	40	26(65)	38	20(53)	5.3 (3.3 - 8.6)	(nd - 84.1)	2.7 (2.6 - 2.9)
	alpha-HCH	40	2(5)	38	1(3)	1.00		0.9 (0.9 - 1.0)
	gamma-HCH	40	1(3)	38	0	nd		0.9 (0.9 - 1.0)
PCBs	sumPCB [l]	40	34(85)	38	5(13)	0.6 (0.5 - 0.8)	(nd - 12.0)	0.9 (0.9 - 1.0)
TCDD-TEQ [m]		21	21(100)	0	na	na		na

Table 5.3 continued

Contaminant [a]		geese tested in total (liver or fat)		Fat				
		n	number above mdl [b]	*n*	number above mdl [c]	concentration ng/g ww [f]	range ng/g ww	mdl ng/g ww [e]
organochlorine pesticides [g]	HCB	40	27(68)	38	27(71)	3.3 (2.4 - 4.5)	(nd - 14.2)	2.2 (2.0 - 2.3)
	HE	40	20(50)	37	19(51)	5.2 (3.6 - 7.5)	(nd - 51.1)	4.4 (4.1 - 4.7)
	trans nonachlor	40	19(48)	38	18(47)	2.8 (1.9 - 4.3)	(nd - 47.3)	2.2 (2.0 - 2.3)
	gamma-CHL	40	4(10)	37	3(8)	1.5 (1.4 - 1.7)	(nd - 2.9)	2.9 (2.7 - 3.1)
	cis-CHL	40	2(5)	38	1(3)	2.7		2.2 (2.0 - 2.3)
	sumCHL [h]	40	24(60)	38	20(53)	5.7 (3.5 - 9.3)	(nd - 70.0)	2.9 (2.7 - 3.1)
	sum rest CHL [i]	nc	nc	36	19 (53)	3.6 (2.5 - 5.3)	(nd - 47.2)	2.9 (2.7 - 3.1)
	p,p'--DDE	40	38(95)	38	36(95)	214.9 (114.0 - 405.2)	(nd - 2366.4)	3.7 (3.4 - 3.9)
	sumDDT [j]	40	38(95)	38	36(95)	276.8 (142.1 - 539.3)	(nd - 4107.3)	5.8 (5.4 - 6.2)
	sum rest DDT [k]	40	26(65)	38	26(68)	44.6 (20.3 - 98.1)	(nd - 1997.1)	5.8 (5.4 - 6.2)
	dieldrin	40	26(65)	36	24(67)	21.4 (12.1 - 37.8)	(nd - 293.3)	6.6 (6.1 - 7.0)
	alpha-HCH	40	2(5)	38	1(3)	2.6		2.2 (2.0 - 2.3)
	gamma-HCH	40	1(3)	38	1(3)	6.9		2.2 (2.0 - 2.3)
PCBs	sumPCB [l]	40	34(85)	38	34(89)	17.6 (10.6 - 29.2)	(nd - 946.7)	2.2 (2.0 - 2.3)
TCDD-TEQ [m]		21	21(100)	21	100	5.7 (3.0 - 10.9)[n]	(0.4 - 36.0)[n]	<1.0 [n]

[a] Geometric means, unless otherwise indicated, with (95% confidence intervals) and (ranges - for concentration column only) for sample sizes greater than two; for sample sizes of one or two, values are presented; nd=below minimum detection limit (mdl); concentrations below the mdl were assigned a value of ½ mdl for statistical calculations.

[b] Number of geese with contaminant concentrations above the mdl in either liver or fat tissue and (percent).

[c] Number of samples with contaminant concentrations above the mdl for that tissue and (percent).

[d] ww=wet weight; geometric mean % lipid and (95% confidence interval) of liver samples (based on wet weight) = 3.3% (2.7 - 4.0)

[e] Arithmetic mean minimum detection limit and (95% confidence interval)

[f] Arithmetic mean % lipid and (95% confidence interval) of fat samples (based on wet weight) = 61.4% (54.7 - 68.2)

Table 5.3 continued

[g] HCB=Hexachlorobenzene; HE=heptachlor epoxide; CHL= chlordane; sumDDT=dichlorodiphenyltrichloroethanes and metabolites (DDE and DDD); HCH=hexachlorocyclohexane; PCB=polychlorinated biphenyls.
[h] Sum chlordane = sum of (HE, *trans* nonachlor, gamma- & *cis*-CHL); minimum detection limit used is the mean mdl of all components used in the sum
[i] Sum rest CHL = sum CHL not including HE
[j] Sum DDT and metabolites (*p,p'*-DDE, *p,p'*-DDE, *p,p'*-DDT and *o,p'*-DDT) ; minimum detection limit used is the mean mdl of all components used in the sum
[k] Sum rest DDT = sumDDT not including *p,p'*-DDE
[l] Sum 24 PCB congeners (8, 18, 28, 29, 44, 50, 52, 66, 77, 87, 101, 104, 105, 118, 126, 128, 138, 153, 170, 180, 187, 195, 206, 209); minimum detection limit used is the mean mdl of all components used in the sum
[m] sum of World Health Organization 2,3,7,8-TCDD Toxic Equivalence Factors for PCDD/Fs and coPCBs for birds (van den Berg *et al.*, 1998), nd=0; arithmetic mean % lipid (95% confidence interval) of fat samples = 72.2 % (64.6 - 79.9)
[n] pg TEQ/g
nc=not calculated
na=not analyzed

coPCBs. There was insufficient liver tissue from one goose for mercury analysis, therefore, 37 liver samples were analysed for mercury.

In a small number of samples, interference from the tissue precluded accurate analysis of some contaminants (PCBs 128 [n=1], 195 [n=6], 206 [n=3] and 209 [n=7], dieldrin [n=2], HE [n=1], and gamma-chlordane [n=1]), leading to missing values.

A majority of geese had detectable levels of mercury and selenium in liver tissue. Cadmium and lead were detected in less than half of the geese tested (Table 5.2). Of the 20 individual OC pesticides measured in fat and liver samples, eight (heptachlor, aldrin, mirex, methoxychlor, alpha- & beta-endosulfan and *o,p'*-DDD, -DDE) were not detected in any of the samples tested (Table 5.3).

In liver tissue, *p,p'*-DDE and dieldrin were the most prevalent individual contaminants. In fat tissue, detectable levels of *p,p'*-DDE were found in all but two of the samples. HCB, HE, sumCHL (excluding HE), sumDDT (excluding *p,p'*-DDE) dieldrin and sumPCB were also detected in a majority of the samples tested (Table 5.3). The highest concentration of OCs found was sumDDT in fat (geom. mean 276.8 ng/g ww) (comprised primarily of *p,p'*-DDE). All fat samples tested had detectable levels of PCDDs, PCDFs and coPCBs (Table 5.3).

Table 5.4 shows Pearson's (for normally distributed contaminant variables) and Spearman's (for non-normally distributed contaminant variables) correlation coefficients between the various contaminants measured. For all examined

Table 5.4. Univariate analyses among tissue contaminants in Canada geese from Labrador in 2001.

Contaminant [ab]		Hg	Cd	Pb	Se	HCB	HE	t-nonachlor	sum CHL[c]	sum CHL	*p,p'*-DDE	*p,p'*-DDE	sum DDT	sum DDT	dieldrin	dieldrin	sumPCB
	tissue	liver	liver	liver	liver	fat	fat	fat	fat	liver	fat	liver	fat	liver	fat	liver	fat
Cd	liver	0.34[d]															
Pb	liver	ns	ns														
Se	liver	0.33[d]	0.59[e]	ns													
HCB	fat	0.40[d]	0.35[d]	ns	0.53[e]												
HE	fat	ns	0.47[e]	ns	0.53[e]	0.72[e]											
t-nonachlor	fat	0.34[d]	0.50[e]	ns	0.58[e]	0.77[e]	0.83[e]										
sum CHL[c]	fat	ns	0.47[e]	ns	0.58[e]	0.77[e]	0.84[e]	0.99[e]									
sumCHL	liver	ns	ns	ns	ns	ns	0.44[e]	0.40[d]	0.34[d]								
p,p'-DDE	fat	0.41[d]	0.36[d]	ns	0.51[e]	0.75[e]	0.70[e]	0.61[e]	0.63[e]	ns							
p,p'-DDE	liver	ns	ns	ns	0.35[d]	0.60[e]	0.54[e]	0.47[e]	0.45[e]	0.34[d]	0.83[e]						
sumDDT	fat	ns	0.39[d]	ns	0.61[e]	0.82[e]	0.80[e]	0.77[e]	0.78[e]	ns	0.81[e]	0.76[e]					
sumDDT	liver	ns	ns	ns	ns	ns	0.40[d]	ns	ns	0.41[d]	0.36[d]	0.51[e]	0.48[e]				
dieldrin	fat	ns	0.41[d]	ns	0.61[e]	0.81[e]	0.76[e]	0.82[e]	0.81[e]	ns	0.63[e]	0.62[e]	0.89[e]	0.37[d]			
dieldrin	liver	ns	0.44[e]	ns	0.60[e]	0.71[e]	0.63[e]	0.79[e]	0.78[e]	0.62[e]	0.65[e]	0.68[e]	0.76[e]	0.42[e]	0.79[e]		
sumPCB	fat	0.55	0.41[d]	ns	ns	0.56[e]	0.68[e]	0.63[e]	0.65[e]	ns	0.60[e]	0.35[d]	0.52[e]	ns	0.55[e]	0.40[d]	
total TEQs	fat	ns	0.50[d]	ns	0.51[d]	0.78[e]	0.81	0.81[e]	0.80[e]	ns	0.62[e]	ns	0.79[e]	ns	0.79[e]	0.63[e]	0.75[e]

[a] Statistical tests used: Pearson's and Spearman's correlation (cadmium, lead, sumCHL, sumDDT in liver); results are presented as Pearson's correlation coefficient or Spearman's rho in the table.
[b] Hg=mercury; Cd=cadmium; Pb=lead; Se=selenium; HCB=Hexachlorobenzene; HE=heptachlor epoxide; sumCHL= sum of chlordanes (HE, gamma-chlordane, *trans* nonachlor, gamma- & *cis*-chlordane); sum DDT= dichlorodiphenyltrichloroethanes and metabolites except *p,p'*-DDE (*o,p'*-DDE, *p,p'*-DDT and *o,p'*-DDT); sumPCB=sum of 26 polychlorinated biphenyl congeners; total-TEQ= sum of World Health Organization 2,3,7,8-TCDD Toxic Equivalence Factors for PCDD/Fs and coPCBs for birds (van den Berg *et al.*, 1998).
[c] sumCHL not including HE
[d] $p<0.05$
[e] $p<0.01$
ns=not significant, $p>0.05$.

geese, most correlations among and between metals and OCs in fat and liver were significant and in a positive direction ($p<0.01$ or $p<0.05$).

5.3.2 Univariate analytical results

Tables 5.5 - 5.7 show correlations and associations (at various significance levels) between the various health parameters and the contaminants measured and possible confounding variables (such as season and gender) for all geese sampled. Due to considerable multi-collinearity among the contaminants measured (Table 5.4), it would be erroneous to make conclusions about specific contaminants based on these results. The Tables are intended to show the reader what was found among individual variables, and to present the variables that were utilized in the final multiple variable regression analyses. Noteworthy observations among these univariate analyses are presented below.

5.3.2.1 Health parameters by season and gender

Table 5.5 shows associations between health parameters and season for all geese sampled. Body condition scores were significantly higher in spring geese compared to fall geese. Spring females had greater BCS (arithmetic mean: 3.6; 95% CI: 3.1 – 4.1) than spring males (arithmetic mean: 2.9; 95% CI: 2.2 – 3.5) ($p<0.05$).

5.3.2.2 Tissue contaminants by season and gender

Table 5.6 reports associations ($p<0.05$) between the various contaminants and season, and between contaminants and gender among spring geese. All contaminants measured in the geese, except for hepatic lead, sum CHL and sum DDT, were significantly higher in spring than in fall ($p<0.05$), including

Table 5.5. Univariate analyses of health parameters as a function of season hunted for Canada geese from Labrador in 2001.

Health parameter [a]		Season	
		spring	fall
BCS[b]		3.2 (2.7 - 3.6)	2.6 (2.2 - 2.9)[g]
nematodes	abundance	ns	
total trematodes	prevalence	ns	
	abundance	ns	
notocotylids	prevalence	ns	
	abundance[c]	1.3 (0.6 - 3.1)	3.9 (0.3 - 24.4)[e]
Zygocotyle lunata	prevalence	ns	
	abundance	ns	
Paramonostomum alveatum	prevalence[d]	33.3 (14.3 - 52.3)	0.0 (0 - 0)[f]
	abundance[c]	2.1 (0.4 - 6.6)	0.0 (0 - 0)[f]
total helminths	abundance	ns	
total cestodes	prevalence	ns	

[a] Statistical tests used: t-test (BCS [body condition score], nematode, total trematode and total helminth abundance), Kruskal-Wallis test (notocotylids, *Z. lunata,* and *P. alveatum* abundance), chi-squared test (parasite prevalences).
[b] Arithmetic mean (95% confidence interval) for each group
[c] Geometric mean (95% confidence interval) for each group
[d] Percentage (95% confidence interval) for each group
[e] $p<0.2$
[f] $p<0.1$
[g] $p<0.01$
ns=not significant at $p>0.2$.

Table 5.6. Univariate analyses of contaminants as a function of season and gender (in spring) of Canada geese from Labrador in 2001.

Contaminant [a]		tissue	Season		Gender (spring)	
			spring	fall	male	female
	selenium	liver	0.58 (0.53 - 0.63)	0.39 (0.31 - 0.47)[d]	0.65 (0.59 - 0.71)	0.50 (0.43 - 0.58)[c]
metals [e]	mercury	liver	0.014 (0.012 - 0.017)	0.009 (0.007 - 0.013)[c]	ns	
	cadmium	liver	0.31 (0.24 - 0.40)	0.2 (0.2 - 0.2)[c]	ns	
	lead	liver	ns		ns	
organochlorine	HCB	fat	6.2 (5.2 - 7.3)	1.2 (0.9 - 1.5)[d]	ns	
pesticides [f]	HE	fat	8.4 (5.5 - 12.8)	2.1 (1.5 - 3.0)[d]	ns	
	t-nonachlor	fat	5.3 (3.2 - 8.7)	1.0 (0.8 - 1.1)[d]	ns	
	sumCHL	fat	6.1 (4.0 - 9.4)	1.3 (1.0 - 1.5)[d]	ns	
	sumCHL	liver	ns		ns	
	p,p'-DDE	fat	631.4 (425.8 - 935.4)	39.4 (12.1 - 128.3)[d]	383.2 (152.5 - 962.7)	120.7 (48.0 - 304.0)[b]
	p,p'-DDE	liver	15.6 (9.3 - 26.4)	2.2 (1.1 - 4.5)[d]	ns	
	sumDDT	fat	211.5 (113.7 - 393.5)	3.1 (2.1 - 4.5)[d]	1387.1 (753.0 - 2558.0)	557.2 (289.7 - 1070.6)[b]
	sumDDT	liver	ns		ns	
	dieldrin	fat	44.0 (25.2 - 76.7)	3.6 (2.5 - 5.0)[d]	ns	
	dieldrin	liver	11.0 (6.5 - 18.7)	1.4 (1.3 - 1.5)[d]	18.9 (9.0 - 39.7)	6.0 (2.7 - 13.2)[b]
PCBs [g]	sumPCB [h]	fat	28.5 (14.8 - 54.8)	7.7 (3.9 - 15.2)[c]	ns	
TCDD-TEQ [i]		fat	19.2 (12.3 - 26.1)	1.8 (0.8 - 2.9)[d]	na	

[a] Statistical tests used: t-test or Kruskal-Wallis test (cadmium, lead, sum CHL and sumDDT in liver); arithmetic (selenium) or geometric means and (95% confidence interval) are presented for each group.
[b] $p<0.05$
[c] $p<0.01$

Table 5.6 continued

[d] $p<0.0001$
[e] reported as µg/g ww (wet weight)
[f] reported as ng/g ww; HCB=Hexachlorobenzene; HE=heptachlor epoxide; sumCHL= sum of chlordanes except HE (gamma-chlordane, *trans* nonachlor, gamma- & *cis*-chlordane); sum DDT= dichlorodiphenyltrichloroethanes and metabolites (*p,p'*-DDE, *p,p'*-DDE, *p,p'*-DDT and *o,p'*-DDT)
[g] polychlorinated biphenyls (ng/g ww)
[h] sum of 24 PCB congeners (8, 18, 28, 29, 44, 50, 52, 66, 77, 87, 101, 104, 105, 118, 126, 128, 138, 153, 170, 180, 187, 195, 206, 209)
[i] sum of World Health Organization 2,3,7,8-TCDD Toxic Equivalence Factors for PCDD/Fs and coPCBs for birds (van den Berg *et al.*, 1998); pg TEQ/g ww
ns=not significant at $p>0.05$
na=not analysed

Table 5.7. Univariate analyses of health parameters of Canada geese from Labrador in 2001 as a function of tissue contaminant concentrations.

				total								total	
			nematode	trematodes		notocotylid		*P. alveatum*		*Z. lunata*		helminths	cestode
Contaminant [ab]	tissue	BCS	abundance	prevalence	abundance	prevalence	abundance	prevalence	abundance	prevalence	abundance	abundance	prevalence
mercury	liver	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns
cadmium	liver	ns	-0.2 [ll]	ns	0.25 [ll]	ns	ns	ns	ns	ns	ns	ns	ns
lead	liver	ns	-0.25 [jj]	ns	ns	nd	0.35[kk]	ns	ns	ns	ns	ns	ns
selenium	liver	ns	-0.25 [ll]	ninf>inf [e]	-0.3 [jj]	ninf>inf [l]	-0.35[kk]	ns	ns	ns	ns	-0.35[kk]	ns
HCB	fat	0.40 [jj]	ns	ninf>inf [f]	ns	ns	ns	inf>ninf [t]	ns	ns	ns	ns	inf>ninf [ff]
HE	fat	0.26 [ff]	ns	ns	ns	ns	ns	inf>ninf [u]	0.31[jj]	ns	ns	ns	ns
t-nonachlor	fat	0.29 [kk]	ns	ninf>inf [g]	ns	ninf>inf [m]	ns	inf>ninf [v]	ns	ns	ns	ns	inf>ninf [gg]
sum CHL [c]	fat	ns	ns	ninf>inf [h]	ns	ninf>inf [n]	ns	inf>ninf [w]	ns	inf>ninf [dd]	ns	ns	ns
sum CHL	liver	ns	ns	ns	ns	inf>ninf [o]	0.26[ll]	ns	ns	ns	ns	ns	ns
p,p'-DDE	fat	0.22 [ll]	ns	ns	ns	ns	ns	inf>ninf [x]	ns	ns	ns	ns	inf>ninf [hh]
p,p'-DDE	liver	0.27 [kk]	ns	ns	ns	ns	ns	inf>ninf [y]	0.24[ll]	inf>ninf [ee]	ns	ns	ns
sumDDT	fat	0.40 [jj]	ns	ninf>inf [i]	ns	ninf>inf [p]	ns	inf>ninf [z]	ns	ns	ns	ns	ns
sumDDT	liver	0.46 [ii]	ns	ns	ns	ns	ns	inf>ninf [aa]	0.44[jj]	ns	ns	ns	ns
dieldrin	fat	0.39 [jj]	ns	ninf>inf [j]	ns	ninf>inf [q]	-0.40 [jj]	inf>ninf [bb]	ns	ns	ns	ns	ns
dieldrin	liver	ns	ns	ninf>inf [k]	ns	ninf>inf [r]	ns	ns	ns	ns	ns	ns	ns
sumPCB	fat	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns
TCDD-TEQ	fat	good>poor [d]	ns	ns	ns	ninf>inf [s]	ns	inf>ninf [cc]	ns	ns	ns	ns	ns

[a] Statistical tests used: t-test and Pearson's correlation or Kruskal-Wallis test and Spearman's correlation (cadmium, lead, sumCHL and sumDDT in liver); results are presented as correlation coefficient and *p*-value in the Table (BCS and parasite abundances) or below as arithmetic (selenium) or geometric means and (95% confidence interval) for each group (for *parasite prevalences*).

[b] HCB=Hexachlorobenzene; HE=heptachlor epoxide; sumCHL= sum of chlordanes (HE, gamma-chlordane, *trans* nonachlor, gamma- & *cis*-chlordane); sumDDT=dichlorodiphenyltrichloroethanes and metabolites except *p,p'*-DDE *(o,p'*-DDE, *p,p'*-DDT and *o,p'*-DDT); sumPCB=sum of 26 polychlorinated biphenyl congeners; TCDD-TEQ= sum of World Health Organization 2,3,7,8-TCDD Toxic Equivalence Factors for PCDD/Fs and coPCBs for birds (van den Berg *et al.*, 1998).

[c] sumCHL not including HE

Table 5.7 continued

[d] TCDD-TEQ in geese in moderate to good body condition: 13.8 pg TEQ/g (8.5 - 22.5); in poor to fair body condition: 3.3 pg TEQ/g (1.3 - 8.4), $p<0.01$
[e] hepatic selenium concentration in geese not infected with intestinal trematodes: 0.64 µg/g ww (0.57 - 0.71); in infected geese: 0.53 µg/g ww (0.46 - 0.60), $p<0.01$
[f] HCB concentration in fat in geese not infected with intestinal trematodes: 5.6 ng/g ww (2.5 - 12.6); in infected geese: 3.9 ng/g ww (2.8 - 5.6), $p<0.2$
[g] t-nonachlor concentration in fat in geese not infected with intestinal trematodes: 5.4 ng/g ww (1.6 - 18.5); in infected geese: 3.1 ng/g ww (1.8 - 5.5), $p<0.2$
[h] sumCHL concentration in fat in geese not infected with intestinal trematodes: 6.7 ng/g ww (2.2 - 20.7); in infected geese: 4.1 ng/g ww (2.4 - 6.9), $p<0.2$
[i] sumDDT concentration in fat in geese not infected with intestinal trematodes: 150.2 ng/g ww (16.9 - 1331.8); in infected geese: 58.0 ng/g wwt (21.6 - 156.3), $p<0.2$
[j] dieldrin concentration in fat in geese not infected with intestinal trematodes: 57.8 ng/g ww (12.7 - 263.2); in infected geese: 18.6 ng/g ww (9.2 - 37.6), $p<0.1$
[k] dieldrin concentration in liver in geese not infected with intestinal trematodes: 12.2 ng/g ww (3.2 - 46.9); in infected geese: 6.5 ng/g ww (3.4 - 12.3), $p<0.2$
[l] hepatic selenium concentration in geese not infected with notocotylids : 0.61 µg/g ww(0.54 - 0.68); in notocotylid-infected geese: 0.51µg/g ww (0.43 - 0.60), $p<0.05$
[m] t-nonachlor concentration in fat in geese not infected with notocotylids : 4.4 ng/g ww (2.3 - 8.9); in notocotylid-infected geese: 2.9 ng/g ww (1.3 - 6.3), $p<0.2$
[n] sumCHL in fat in geese not infected with notocotylids : 5.8 ng/g ww(3.1 - 10.6); in notocotylid- infected geese: 3.8 ng/g ww (1.8 - 7.9), $p<0.2$
[o] sum CHL in liver in geese infected with notocotylids : 1.2 ng/g ww (0.7 - 2.3); in geese not infected with notocotylids : 1.0 ng/g ww(0.5 - 1.9), $p<0.1$
[p] sumDDT in fat in geese not infected with notocotylids : 143.7 ng/g ww (37.9 - 544.1); in notocotylid- infected geese: 38.8 ng/g ww (12.2 - 123.9), $p<0.1$
[q] dieldrin concentration in fat in geese not infected with notocotylids : 53.2 ng/g ww(20.6 - 137.4); in notocotylid- infected geese: 12.5 ng/g ww (5.8 - 26.9), $p<0.01$
[r] dieldrin concentration in liver in geese not infected with notocotylids : 11.4 ng/g ww (4.8 - 27.3); in notocotylid- infected geese: 5.2 ng/g ww (2.5 - 11.0), $p<0.1$
[s] TCDD-TEQ in fat in geese not infected with notocotylids : 11.2 ng/g ww (2.6 - 48.1); in notocotylid- infected geese: 6.0 ng/g ww (2.1 - 16.7), $p<0.2$
[t] HCB concentration in fat in geese infected with *P. alveatum:* 6.0 ng/g ww (4.5 - 8.1); in geese not infected with *P. alveatum*: 3.8 ng/g ww (2.5 - 5.7), $p<0.05$
[u] HE concentration in fat in geese infected with *P. alveatum:* 9.2 ng/g ww (4.5 - 18.6); in geese not infected with *P. alveatum*: 5.0 ng/g ww (2.9 - 8.6), $p<0.1$
[v] t-nonachlor concentration in fat in geese infected with *P. alveatum:* 5.4 ng/g w w (1.7 - 17.2); in geese not infected with *P. alveatum*: 3.0 ng/g ww (1.7 - 5.3), $p<0.2$
[w] sumCHL concentration in fat in geese infected with *P. alveatum:* 6.4 ng/g ww (2.4 - 16.7); in geese not infected with *P. alveatum*: 4.1 ng/g ww (2.3 - 7.1), $p<0.2$
[x] *p,p'*-DDE concentration in fat in geese infected with *P. alveatum:* 627.5 ng/g ww (313.7 - 1255.0); in geese not infected with *P. alveatum*: 320.1 ng/g ww (162.1 - 629.0), $p<0.1$
[y] *p,p'*-DDE concentration in liver in geese infected with *P. alveatum:* 21.2 ng/g ww (7.0 - 64.6); in geese not infected with *P. alveatum*: 8.2 ng/g ww (4.5 - 14.9), $p<0.1$
[z] sumDDT concentration in fat in geese infected with *P. alveatum:* 246.8ng/g ww (75.1 - 811.6); in geese not infected with *P. alveatum*: 45.9 ng/g ww (15.6 - 135.1), $p<0.05$
[aa] sumDDT concentration in liver in geese infected with *P. alveatum:* 5.4 ng/g ww (1.4 - 21.4); in geese not infected with *P. alveatum*: 1.9 ng/g ww (1.1 - 3.2), $p<0.05$
[bb] dieldrin concentration in fat in geese infected with *P. alveatum:* 47.2 ng/g ww (15.2 - 146.9); in geese not infected with *P. alveatum*: 19.0 ng/g ww (8.9 - 40.9), $p<0.1$
[cc] TCDD-TEQ in geese infected with *P. alveatum:* 15.2 pg TEQ/g (8.2 - 28.0); in geese not infected with *P. alveatum*: 4.7pg TEQ/g (1.5 - 14.8), $p<0.05$
[dd] sumCHL concentration in liver in geese infected with *Z. lunata:* 6.5 ng/g ww (2.5 - 16.6); in geese not infected with *Z. lunata*: 4.2 ng/g ww (2.4 - 7.3), $p<0.2$
[ee] *p,p'*-DDE concentration in liver in geese infected with *Z. lunata:* 16.1 ng/g ww (3.8 - 67.9); in geese not infected with *Z. lunata*: 8.8 ng/g ww (5.2 - 14.9), $p<0.2$
[ff] HCB concentration in fat in geese infected with intestinal cestodes: 5.4 ng/g ww (3.2 - 9.1); in geese not infected with intestinal cestodes: 3.8 ng/g ww (2.5 - 5.7), $p<0.2$
[gg] t-nonachlor concentration in fat in geese infected with intestinal cestodes: 5.2 ng/g ww (1.9 - 14.2); in geese not infected with intestinal cestodes: 2.9 ng/g ww (1.6 - 5.2), $p<0.2$
[hh] *p,p'*-DDE concentration in fat in geese infected with intestinal cestodes: 540.8 ng/g ww (257.1 - 1137.6); in geese not infected with intestinal cestodes: 314.0 ng/g ww (150.5 - 654.7), $p<0.2$
[ii] $p<0.01$
[jj] $p<0.05$
[kk] $p<0.1$
[ll] $p<0.2$
ns= not significant, $p>0.2$
ww=wet weight
inf=infected; ninf=not infected

TCDD-TEQ in the subsample of dioxin-tested geese. Among spring geese, males had higher levels of selenium and dieldrin in liver, and *p,p'*-DDE and sumDDT in fat tissue compared to females ($p<0.05$). Gender differences in TCDD-TEQ among spring geese were not evaluated due to the small sample sizes for this subset of geese.

5.3.2.3 Health parameters by tissue contaminants

Results of univariate analyses between health parameters and tissue contaminant levels in all geese examined ($p<0.2$) are listed in Table 5.7. All simple associations found between BCS and OCs in fat and liver were positive. For dioxin-tested geese, TCDD-TEQ concentrations were higher in geese in good body condition.

The direction of associations between intestinal helminths and OC contaminants varied by parasite species. Associations found between OC contaminants and *P. alveatum*, *Z. lunata* and cestode infection were all positive (infected geese had greater OC concentrations), while most associations between contaminants and notocotylids and total trematodes were negative (infected geese had lower OC concentrations).

5.3.2.4 Univariate analysis results in spring geese

Because spring geese were found to have significantly higher levels of most contaminants compared to fall geese, biological effects due to contaminant exposure may be more apparent in spring geese. To determine whether associations between health parameters, contaminants and potential confounders were present in spring geese only, univariate analyses were also

carried out on the subsample of 29 spring geese. TCDD-TEQ concentrations were not included in the analyses of spring geese because of the small number of spring geese tested for dioxins and related compounds (n=12).

Similar results were found in the spring subsample as were found for all geese with the exception of some additional significant associations. Selenium was found to be negatively associated with nematode (r=-0.43), total trematode (r=-0.43) and total helminth (r=-0.53) abundance and HCB concentrations in fat were positively associated with nematode abundance (r=0.51) ($p<0.05$). In addition, *p,p'*-DDE in liver was positively associated with *Z. lunata* prevalence (geometric mean *p,p'*-DDE = 31.3 ng/g ww [95%CI: 7.9 – 123.9] in infected geese; *p,p'*-DDE = 11.4 ng/g ww [95%CI: 6.5 – 20.0] in those free of infection) ($p<0.05$).

5.3.3 Multiple variable regression analyses

Multiple regression analyses were performed for health parameters for all geese as well as for spring geese only. For the subset of dioxin-tested geese, analyses were performed for those health parameters found to be associated with TCDD-TEQ on univariate analyses (BCS and *P. alveatum* prevalence). Significant final models for each group of geese are presented in Table 5.8. Because season was frequently related to various contaminants (Table 5.6), for analyses including all examined geese and dioxin-tested geese, season was only added to the model after other contaminant variables were allowed to enter the final model. This enabled the determination of what correlates of season may be related to the health parameter of interest.

Table 5.8. Results of multiple regression analyses of health parameters for samples from all geese, dioxin-tested geese and spring geese only.

	Health parameter	Regression Analysis	Coefficient (SE)	Independent variable	Intercept (SE)	F-value[a]	Model *p*-value	*n*	adjusted[b] R^2
All geese	BCS	linear	0.23 (0.09)	ln(dieldrin in fat)	2.37 (0.31)	6.30	0.02	38	0.13
	notocotylid prevalence	logistic	-1.47 (0.60) 1.34 (0.79)	ln(dieldrin in fat) ln(sumCHL in fat)	3.06 (1.31)	9.43	0.01	27	0.25
Dioxin-tested geese	BCS	logistic	0.81(0.36)	ln(dieldrin in fat)	-3.24 (1.44)	6.74	0.01	21	0.24
			1.44 (0.70)	ln(HCB in fat)	-2.43 (1.16)	5.65	0.02	21	0.20
			0.60 (0.26)	ln(sumDDT in fat)	-3.11 (1.35)	7.69	0.01	21	0.28
			1.00 (0.49)	ln(TCDD-TEQ in fat)	-2.48 (1.22)	6.05	0.02	21	0.22
	Paramonostomum alveatum prevalence	logistic	1.64 (0.89) -10.53 (6.34)	ln(TCDD-TEQ in fat) selenium (in liver)	1.94 (2.98)	7.65	0.02	14	0.40
Spring geese	BCS	linear	0.38 (0.14) -1.04 (0.39)	ln(dieldrin in fat) male	2.37 (0.56)	5.79	0.01	25	0.29
	ln(nematode abundance)	linear	1.46 (0.44) -4.01 (1.42)	ln(HCB in fat) selenium(in liver)	3.18 (1.17)	9.32	0.002	21	0.45
	notocotylids prevalence	logistic	-0.81 (0.37)	ln(dieldrin in fat)	3.11 (1.53)	5.96	0.02	24	0.18
	ln(total helminth abundance)	linear	-6.39 (2.21)	selenium (in liver)	7.96 (1.33)	8.40	0.01	23	0.25

[a] For logistic regressions, LR test (chi-squ) value.
[b] For logistic regressions, Pseudo R^2 value.

No interaction variables between main effect variables remained in any of the final models. Standard regression diagnostic procedures confirmed a reasonable fit of all final models.

5.3.3.1 All geese

Significant final models were produced for BCS, and notocotylid prevalence. No variables remained significant in the models for the other health parameters.

Season and concentration of several OC contaminants (HCB, HE, *p,p'*-DDE, sumDDT and dieldrin in fat, *p,p'*-DDE and sumDDT in liver) were found to be associated with BCS in univariate analyses (Tables 5.5 and 5.7) and were offered as independent variables for multiple regression analysis. The only significant predictor of BCS was dieldrin in fat, having a positive association with BCS (Table 5.8).

Selenium and sumCHL concentrations in liver and sumCHL, sumDDT and dieldrin fat were found to be associated with notocotylid prevalence in univariate analyses (Table 5.7). However, dieldrin and sumCHL in fat were the only remaining significant predictors of notocotylid prevalence, with dieldrin having a negative association and sum CHL having a positive association with notocotylid prevalence (Table 5.8).

5.3.3.2 Dioxin-tested geese

Significant final models were produced for BCS and *P. alveatum* prevalence. No variables remained significant in the models for the other health parameters.

Season, concentrations of HCB, sumCHL, *p,p'*-DDE, sumDDT, dieldrin and TCDD-TEQ in fat, concentrations of *p,p'*-DDE, sumDDT and dieldrin in liver, total trematode abundance, *P. alveatum* prevalence and abundance, and total helminth abundance were associated with BCS on univariate analysis and so were offered as independent variables. Depending on which variable was entered into the model first, four separate significant final models were produced, each with one OC contaminant as the only significant predictor (HCB, sumDDT, dieldrin, and TCDD-TEQ in fat) (Table 5.8). All significant predictors had a positive association with BCS.

Season, BCS, concentrations of sumCHL, sumDDT, dieldrin and TCDD-TEQ in fat and concentrations of selenium, mercury, sumCHL, *p,p'*-DDE and sumDDT in liver were associated on univariate analysis with *P. alveatum* prevalence. Selenium in liver and TCDD-TEQ in fat were found to be significant predictors of *P. alveatum* prevalence in dioxin-tested geese (Table 5.8), with selenium being negatively associated and TCDD-TEQ being positively associated.

5.3.3.3 Spring geese

Significant final models were produced for BCS, nematode abundance, notocotylid prevalence, and total helminth abundance. No variables remained significant in the models for the other health parameters.

Gender, concentrations of dieldrin and sumPCB in fat, concentrations of mercury and sum DDT in liver, and total trematode prevalence were associated on univariate analysis with BCS and thus were offered as independent variables.

Gender and dieldrin in fat were found to be significant predictors of BCS in spring geese (Table 5.8). Dieldrin in fat was positively associated with BCS, and males were negatively associated with BCS.

Gender, concentrations of selenium, cadmium and lead in liver, and concentration of HCB in fat were associated on univariate analysis with nematode abundance. HCB and selenium were found to be significant predictors of nematode abundance in spring geese (Table 5.8). HCB in fat was positively associated, and selenium in liver was negatively associated, with nematode abundance.

Concentrations of sumDDT and dieldrin in fat, and selenium, cadmium, *p,p'*-DDE and dieldrin in liver were associated on univariate analysis with notocotylid prevalence. Dieldrin in fat was found to be the only significant predictor of notocotylid prevalence in spring geese, and was negatively associated.

Gender, concentrations of HCB, *p,p'*-DDE and sumDDT in fat, and concentrations of selenium and dieldrin in liver were associated on univariate analysis with total helminth abundance. Selenium was found to be the only significant predictor of total helminth abundance in spring geese (Table 5.8) and produced a negative association.

5.4 Discussion

5.4.1 Health parameters

The seasonal and gender differences in fat reserves (BCS) seen in the sampled geese are consistent with previous reports on condition dynamics in

Canada geese (Hanson, 1962; Mowbray et al., 2002). Fat reserves in Canada geese have been shown to vary dramatically on a seasonal basis as well as with forage availability, age, gender and physiological (eg. reproductive state) and disease processes (Hanson, 1962). Not all potentially influencing factors were accounted for, however, most geese examined appeared to be in adequate body condition. Associations between BCS and most OC contaminant concentrations were all positive (higher concentrations in fatter geese), even when season was considered. These associations are most likely related to the high lipid solubility of OC contaminants and are discussed in more detail in Section 5.4.3.

The helminth species found appeared to be part of the normal helminth fauna for the NAP of Canada geese, and intestinal helminth infection did not appear to be impacting the health of the geese. Similarly, in a study of Canada geese, white-fronted geese and snow geese wintering in Texas, U.S.A., intestinal helminths did not appear to cause tissue damage, and no correlations were found between intestinal helminth intensities and host weight (Purvis et al., 1997).

Some other gastrointestinal helminths, such as gizzard nematodes of the genera *Amidostomum* and *Epomidiostomum,* have been found at high prevalences in geese, including Canada geese (Purvis et al., 1997), and these parasites have been reported to cause significant lesions in host species (Herman and Wehr, 1954; Tuggle and Crites, 1984). Gizzard nematodes were not evaluated in this study, because only seven gizzards were collected, based on the requests of individual hunters.

5.4.2 Tissue contaminant levels

Organochlorine contaminants were found at detectable levels in the tissues of a majority of the geese examined. As OCs are found primarily in the lipid fraction of a tissue, results from different tissues and studies can be compared by lipid-normalizing the OC concentrations (Braune et al., 1999). However, individual contaminants may not be at an equilibrium in individual birds (not constant across lipid stores), and therefore, comparisons should be interpreted with caution (Braune et al., 1999). Based on lipid weight estimations, OC levels found in pooled pectoral muscles of Canada geese killed by hunters in all regions of Canada from 1988 - 95 (Braune et al., 1999) and in pectoral muscles of resident Canada geese in Chicago, Illinois (Levengood et al., 1999) were similar to, or higher than the geometric mean concentrations found in this study. Similar ranges of OC concentrations were found in fat tissue of Canada geese killed in New York State in 1983 - 84 (Foley, 1992).

Braune et al. (1999) found that Canada geese and other herbivorous waterfowl have relatively low levels of OCs in pectoral muscle tissue. In general, levels of OCs were lowest in herbivorous species such as geese, swans and terrestrial gamebirds (ptarmigan and grouse), and highest in fish-eating birds such as loons and mergansers. As an example, ranges for sumPCB in pools of pectoral muscle for Canada geese and common mergansers were: nd -108 and 92 - 9744, respectively (ng/g lipid weight) (Braune et al., 1999). For comparison, lipid weight geometric mean concentration (95%CI) of sumPCB in fat tissue from geese in our study was 28.7 (17.3 - 47.6) ng/g.

Similarly, Foley (1992) found that wood ducks and Canada geese always contained low concentrations of OC contaminants (HCB, CHL, sumDDT, sumPCBs, dieldrin) in their tissues, significantly lower compared to other waterfowl species (American black duck, mallard, scaup and bufflehead) killed in New York state in 1983 - 84. For example, Foley (1992) found arithmetic mean (±SD) concentrations of sumDDT in fat tissue of 260(440) for Canada geese, 1330(2520) for black ducks and 1480(3220) for scaup (ng/g wet weight). For comparison, wet weight geometric mean concentration (95%CI) of sumDDT in fat tissue from geese in this study was 277 (142 - 539) ng/g.

The levels of OC contaminants measured in the sampled Canada geese were lower than those found in Arctic seabirds, particularly for PCBs. Ivory gulls and northern fulmars had levels of PCBs 300-600 times higher than those found in geese in this study (CACARII, 2003). Smaller Arctic seabirds such as dovekies, thick-billed murres and black-legged kittiwakes had levels of PCBs 30 - 200 times higher than these geese. Other OCs such as HCB, HCH and sumCHL were also considerably higher in the Arctic seabirds. Differences between the geese in this study and Arctic seabirds were not as great for sumDDT and dieldrin concentrations (1.5-20 times higher in Arctic seabirds for sumDDT and 4-15 times higher for dieldrin) (CACAR II, 2003).

There is little information on PCDD/F and coPCB levels in geese for comparison. Trace levels of these compounds were found in one pool of pectoral muscles from Canada geese in British Columbia analysed by Braune et al. (1999). Residues found in liver tissue in some Arctic seabirds were 50 - 1000

times higher (based on lipid weight estimations) than those found in fat tissue in geese in this study (CACARII, 2003).

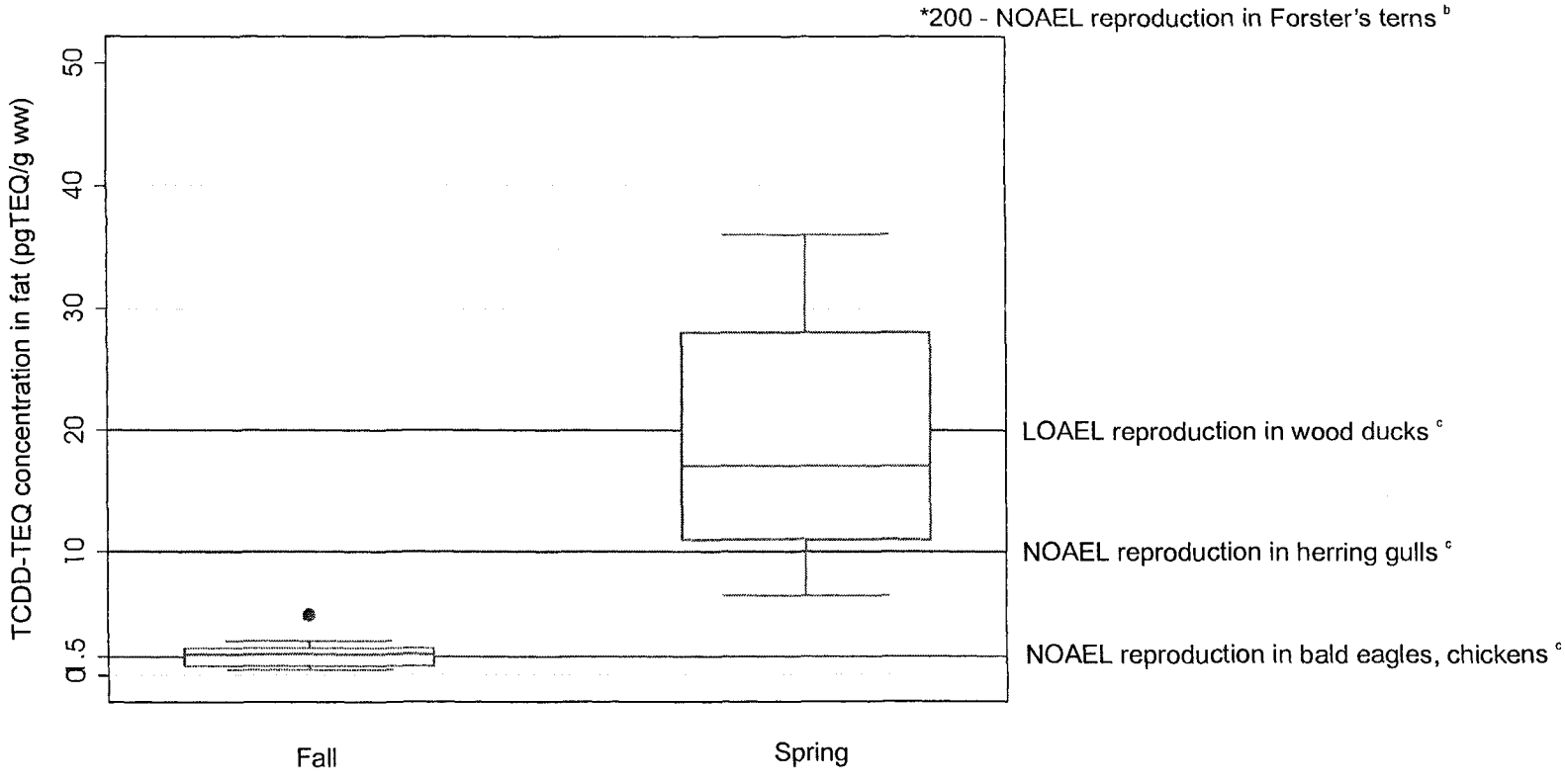
All liver metal levels found in the geese in this study were at or near the minimum detection limits. Liver mercury levels were considerably lower than levels in fish-eating birds such as common mergansers (mean 10.0 - 17.5 µg/g wet weight in liver) killed in Québec (Langlois and Langis, 1995) and in dabbling ducks and sea ducks from various regions of Canada (range 0.016 - 2.3 µg/g wet weight in liver) (Braune et al., 1999).

Significant correlation among contaminants was found. In general, OCs were positively correlated indicating that coaccumulation of these compounds occurred, although the toxicological significance of coaccumulation in the geese is unclear. Correlations between levels of PCBs and *pp*'-DDE have been found in bald eagles in the Great Lakes area of Canada and between current-use pesticides such as organophosphates and residual organochlorines (primarily *p,p*'-DDE) in tree swallows in Ontario, Canada (Bowerman et al., 1995; Bishop et al., 2000).

For most of the contaminants measured in the Canada geese in our study, concentrations were well below reported threshold levels for biological effects in avian species even when considered by season (deMarch et al., 1998; Braune et al., 1999; CACARII, 2003), with the exception of TCDD-TEQ and *p,p'*-DDE and sumDDT concentrations.

Figure 5.1 illustrates the TCDD-TEQ concentrations in fat tissue by season in relation to reported threshold levels for biological effects in avian species. In

Figure 5.1 TCDD-TEQ concentrations[a] in fat by season in Canada geese collected in 2001 in Labrador, Canada.



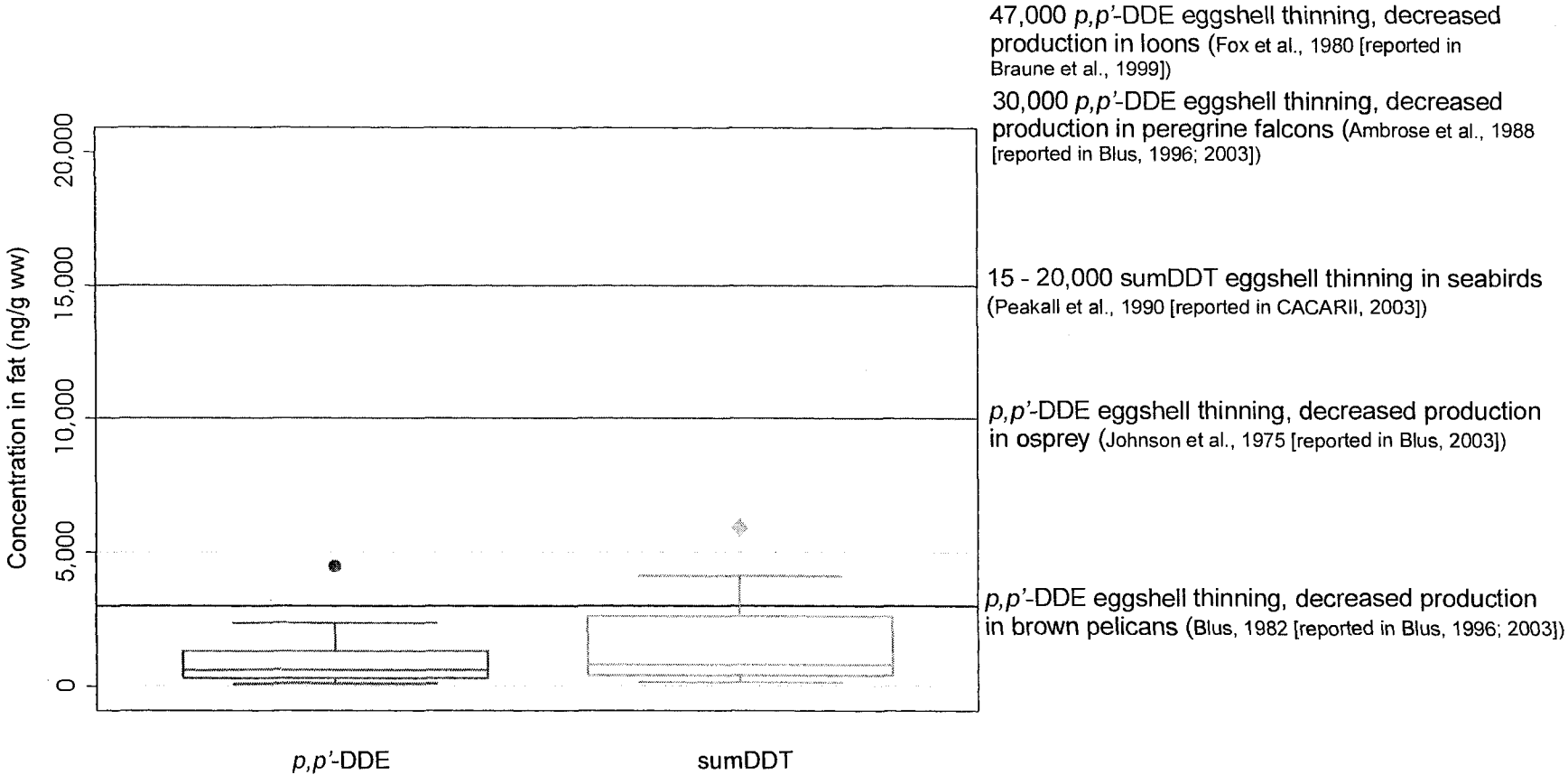
[a] TCDD-TEQ calculations were based on PCDD, PCDF and coPCB concentrations and World Health Organization avian toxic equivalence factors (TEFs) as reported in van den Berg et al. (1998).
[b] Kubiak et al., 1989 (reported in deMarch et al., 1998)
[c] Giesy 1994b (reported in deMarch et al., 1998)
Solid lines represent reported threshold levels for reproductive effects in avian species.

fall geese studied, concentrations (1.8 pgTEQ/g) slightly exceeded the NOAEL (no adverse effect level) for reproduction in chickens and bald eagles (1.5 pg/g ww). In the spring geese studied, the geometric mean TCDD-TEQ level (19.2 pgTEQ/g) also exceeded the NOAEL for herring gulls (10 pgTEQ/g ww) and approached the LOAEL (lowest adverse effect level) for reproduction for wood ducks (20 pgTEQ/g ww) (Figure 5.1). However, tremendous differences in sensitivity to reproductive effects of dioxin-like contaminants have been demonstrated in wild birds. For example, for Forster's terns, threshold levels are considerably higher (NOAEL for hatching success=200 pgTEQ/g ww, LOAEL=2200 pgTEQ/g ww) (deMarch et al., 1998). Therefore, comparisons between species must be made cautiously.

Figure 5.2 illustrates *p,p'*-DDE and sumDDT concentrations in fat tissue of spring geese from our study in relation to reported threshold levels for eggshell thinning in free-ranging avian species. A small number of the spring geese had concentrations of *p,p'*-DDE and sumDDT in fat that approached or exceeded the threshold levels for eggshell thinning reported for eggs in brown pelicans.

In addition to the effects that different contaminants can induce independently, some studies have demonstrated interactive biological effects between and among environmental contaminants. Laboratory studies have shown some evidence for additive and non-additive (synergistic or antagonistic) effects among PCBs, dioxins and furans (Tryphonas, 1995; van den Berg et al., 1998; USDHHS, 2000). Interactions have also been found between PCBs and methyl mercury and PCBs and *p,p'*-DDE (USDHHS 2000).

Figure 5.2 *P,p'*-DDE and sumDDT concentrations in fat in Canada geese collected in spring 2001 in Labrador, Canada.



note: Solid lines indicate reported threshold levels for biological effects in bird eggs.

5.4.3 Seasonal and gender differences in contaminants

Organochlorines were found at levels 4 (sumPCB) to 23 (sumDDT) times higher in spring geese compared to those collected in fall. One possible explanation for this seasonal difference relates to the accumulation of these contaminants in wintering areas and/or during spring migration. Canada geese are primarily a migratory species and those killed in Labrador are most likely part of the NAP which breeds from western Greenland to Labrador, eastern Québec and insular Newfoundland in Canada (CWS, 2000; Mowbray et al., 2002). The NAP migrates south through the Canadian Maritime provinces and along the New England coast of U.S.A., and winters along the Atlantic coast south of Long Island Sound, New York, to Pea Island, North Carolina, U.S.A. (Mowbray et al., 2002). Alternatively, it is possible that some of the geese killed in the spring near the Québec-Labrador border may be part of the Atlantic population (AP) which breeds in northern Québec along Ungava Bay and eastern Hudson's Bay and the interior Ungava Peninsula (CWS, 2000). The AP migrates south along the eastern shores of Hudson Bay and James Bay in Canada, across New York State and winters from New England south to South Carolina (Mowbray et al., 2002). Regardless of the grouping, geese killed in the spring in Labrador have recently migrated from the industrialized northeastern U.S.A. and are likely a mixture of breeding adults and yearlings because family bonds are strong within the first year and yearlings typically return to their natal areas with their parents in the spring (Bellrose, 1978). Conversely, geese killed in the fall are migrating south for the winter and are a mixture of adults, yearlings and juveniles (young

of the year) from Labrador or Greenland. The latter group would have not yet been exposed to the more industrialized areas encountered during the migratory and wintering periods and thus, they would be expected to have tissue contaminant concentrations close to background levels. Age ratios for the NAP Canada geese killed during the Atlantic Canada fall hunting season from 1991 - 1999 have ranged from approximately 0.5 to 1.5 juveniles/adult (estimated from Figure 24, CWS, 2000).

Another potential contributing factor to the seasonal differences in OC concentrations in geese relates to lipid stores. In other species, the distribution and dynamics of body lipid stores have been shown to affect the levels and distribution of OCs (eg. Anderson et al., 1984; Henrickson et al., 1996; Norstrum et al., 1998; Olafsdottir et al., 1998; Polischuk et al., 2002). For Canada geese, peak condition is reached shortly after the spring migration (Hanson, 1962; Mowbray et al., 2002). Thus, as OC residues are acquired, they are stored in the expanding lipid pool. Fat stores subsequently diminish throughout the nesting and brood-rearing periods and are at an annual low during the summer moult (Hanson, 1962; Mowbray et al., 2002). During this period of dynamic fat store mobilization, plasma lipid levels increase and OC compounds that have been accumulated and stored in fat are expected to be more readily metabolized and excreted (Clark et al., 1987). Although OC contaminants in general are relatively persistent, they are metabolized and excreted by birds and mammals at rates which vary with the pharmacokinetics of the compound and the lipid pool dynamics of the species (Norstrom et al., 1986; Clark et al., 1987). Half-lives for

various OCs in different species have been reported and are similar to those based on an experimental model developed for juvenile herring gulls by Clark et al. (1987). Half-lives in the herring gulls were in the order of 50 (for DDD) to 700 days (for DDE) (Clark et al., 1987). However, in cackling geese, the half-life of DDE was shown to be much shorter (63 days) (Anderson et al., 1984), perhaps because it was measured during a period of decreasing fat stores (Clark et al., 1987). Based on the results for herring gulls and other species, it is possible that the OC levels could have decreased considerably during the four months between spring and fall migrations due to the mobilization of lipids and subsequent increased availability for metabolism and excretion of OCs during that time, however, it seems unlikely that the 16 fold decrease in *p,p'*-DDE levels seen in this study can be explained entirely by increased clearance rates alone.

In general, the clearance rates of highly lipophilic OC compounds are inversely related to lipid pool size; OCs are cleared more quickly when fat stores in an animal are small (Clark et al., 1987) (ie. after the summer moult for Canada geese). In late summer and fall, as lipid stores expand again in preparation for fall migration, exposure to OC contaminants in Labrador is most likely much lower than during winter and early spring in industrialized areas. Thus, any remaining OC residues would be diluted, and samples taken in the fall would have lower OC concentrations than in the spring.

Regardless of the mechanisms behind the seasonal differences in tissue OC levels, it is apparent that exposure to OCs is greater in spring/summer than in fall. Biological effects due to these contaminants would be more likely to

occur after the spring migration, during nesting and brood-rearing periods when lipid stores containing the OCs are mobilized, compared to fall when contaminant burdens are lower and lipid deposits are likely more stable.

Explanations are less obvious for the higher levels of hepatic mercury, cadmium and selenium in spring geese compared to fall. It is possible that the slightly higher levels of mercury and cadmium seen in spring could be attributed to increased exposure in the wintering area. Geese undergo a moult of all their feathers in summer, and mercury residues can be deposited in the replacement feathers during their formation (Scheuhammer, 1987), which could result in decreased fall tissue residues. Selenium levels can vary with forage type (Ohlendorf, 1996), thus, seasonal differences in hepatic selenium levels may reflect the seasonal differences in diet. As with the seasonal differences in OC residues, the lower levels of mercury, cadmium and selenium seen in the fall geese may also be related to the demographic makeup of the birds sampled.

Gender differences in some contaminant levels were also found on univariate analysis. Concentrations of *p,p'*-DDE and sumDDT in fat and selenium and dieldrin in liver were 1.3 to 3.2 times higher in males than females. Similar gender differences in OC and trace element concentrations have been demonstrated in some other studies of wild birds (eg. Donaldson and Braune, 1999). It is generally thought that females are able to deposit a proportion of their contaminant burden into developing eggs during the breeding season (Donaldson and Braune, 1999; Levengood, 1999). Not all contaminants measured in the sampled geese were lower in females; perhaps because the

geese were sampled early in the breeding season and some of the females had not completed laying their eggs. Unfortunately, gender data were not available for the fall geese and similar comparisons could not be made for that season.

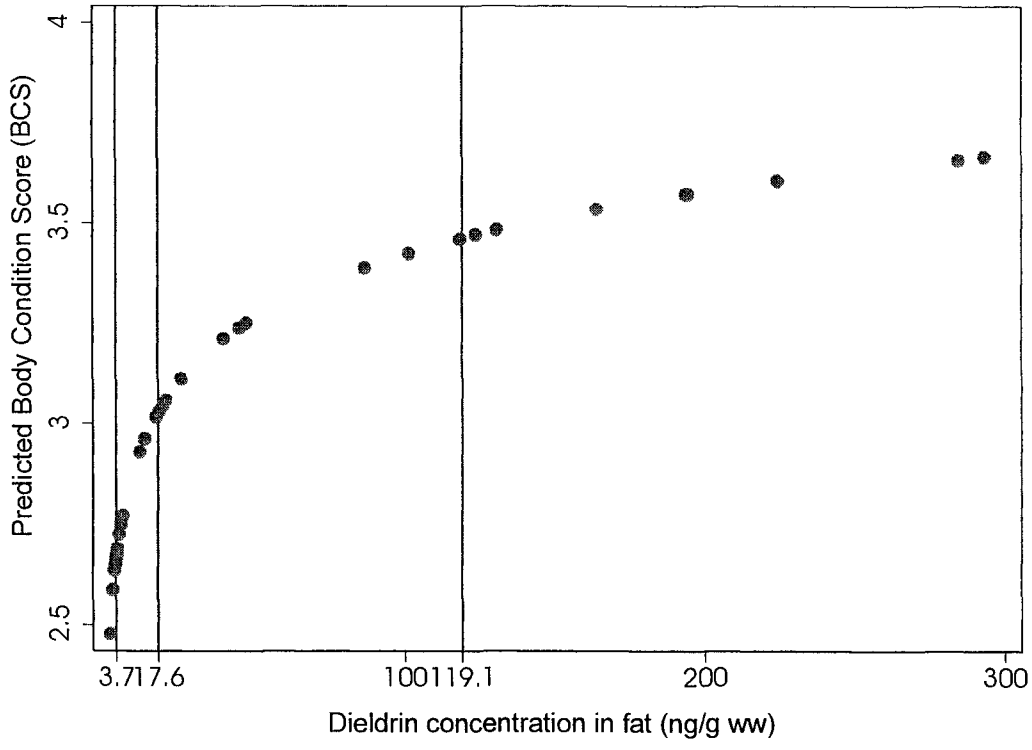
5.4.4 Associations between contaminants and health parameters based on multiple variable regression analyses

5.4.4.1 Contaminants and BCS

Dieldrin concentration in fat was found to be a significant predictor of BCS. In all groups (including dioxin-tested geese and spring geese only), geese with higher levels of dieldrin tended to be in better body condition (greater fat stores). For all geese, increasing dieldrin concentration in fat from 3.7 to 119.1 ng/g ww (from the 25th percentile to the 75th percentile) resulted in an increase in predicted BCS from 2.7 to 3.5 (Figure 5.3). For the dioxin-tested geese, several separate regression models were developed each with a single OC contaminant, including dieldrin, as the only significant predictor (Table 5.8).

As OCs are stored in the lipid fraction of a tissue, on a wet weight (fresh weight) basis, the OC concentration reflects, in part, the proportion of lipid in the tissue. BCS was strongly positively correlated with % lipid in fat tissue ($r=0.77$, $p<0.01$). Thus, the association seen between BCS and dieldrin reflects, in part, the increased OC storage capacity per gram of fat tissue of geese with higher BCS (larger fat reserves). On a lipid weight basis, OCs are often negatively correlated with BCS reflecting the increased concentration of OCs in smaller lipid compartments (Henrikson et al., 1996). Because dieldrin and other OC contaminants have a high affinity for lipids, their distribution in an organism is

Figure 5.3 The predicted relationship between dieldrin concentration in fat and body condition score (BCS) in Canada geese collected in 2001 in Labrador, Canada.



note: Solid lines indicate 25th (3.7 ng/g wet weight [ww]), median (17.6 ng/g ww) and 75th (119.1 ng/g ww) percentile concentrations of dieldrin.

dependent on the distribution and dynamics of the lipid stores, which can vary dramatically on a seasonal basis in many free-ranging species, as previously discussed.

For the spring geese (the only group for which gender was consistently recorded), gender (when considered with dieldrin concentration) was also a significant predictor of BCS. In spring geese, increasing the dieldrin concentration from 16.9 to 130.3 ng/g ww (from the 25th percentile to the 75th percentile) resulted in an increase in predicted BCS from 3.4 to 4.2 in females and 2.4 to 3.2 in males. Other studies have also shown that female geese in spring tend to have greater fat reserves than males (Hanson, 1962).

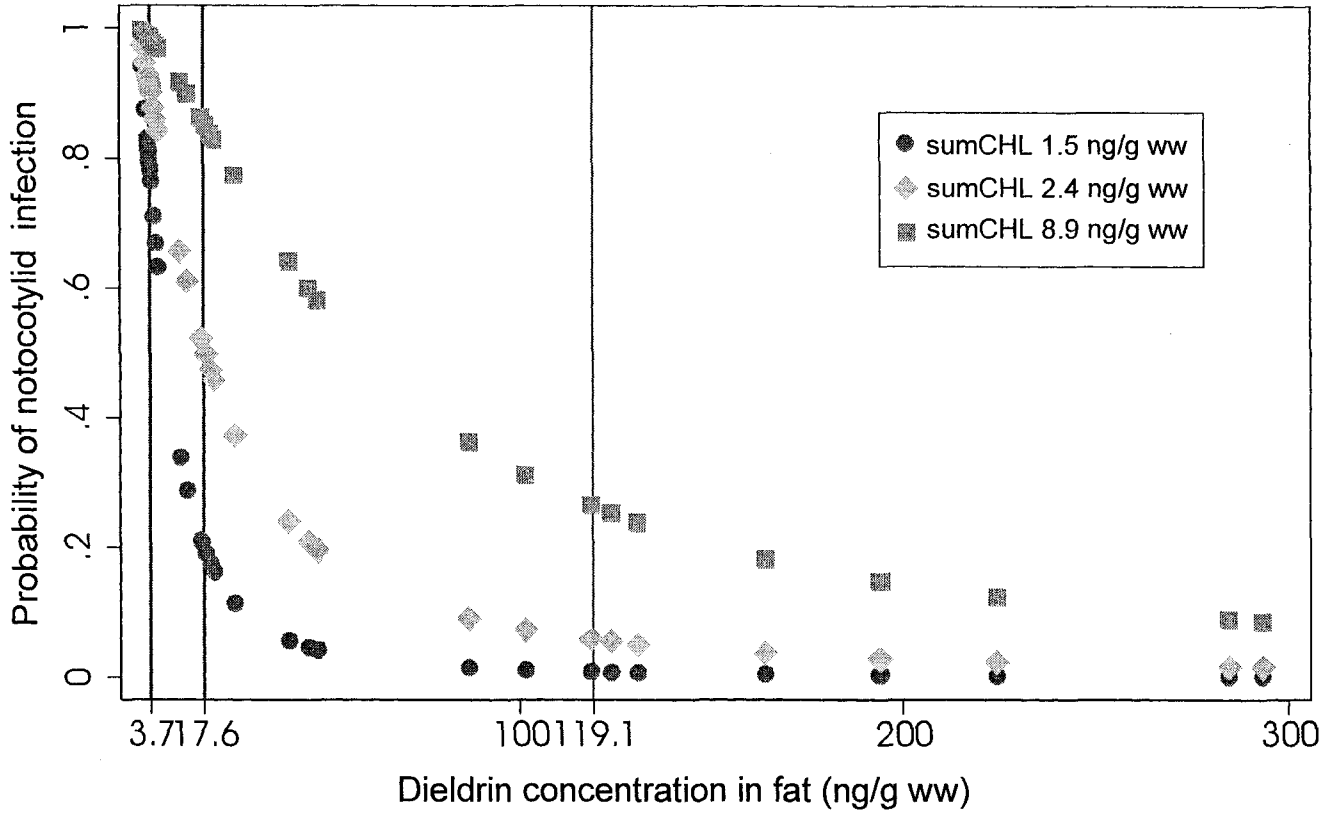
5.4.4.2 Contaminants and helminth parasites

Both positive and negative associations were found between OC contaminant levels and intestinal helminth parasite infection in the geese, based on multiple regression analyses. The most consistent finding was a negative association between certain intestinal parasitic parameters and hepatic selenium levels (Table 5.8).

5.4.4.2.1 OC contaminants and helminth parasites

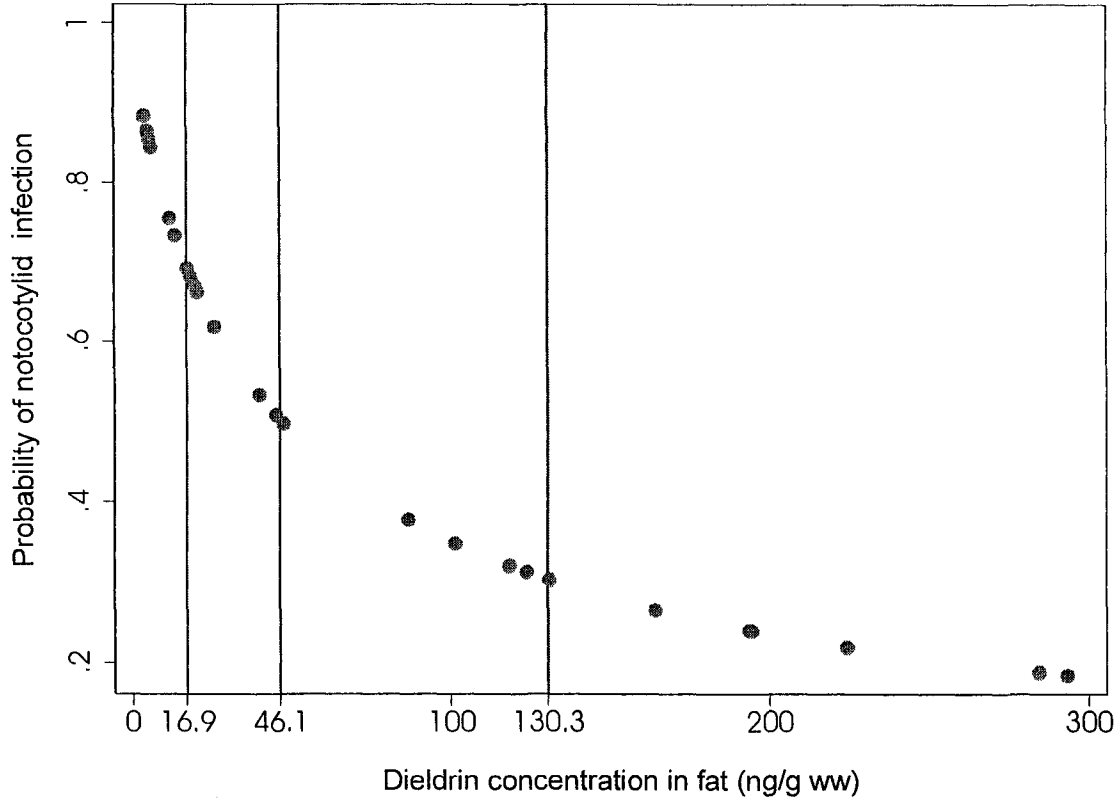
For all geese and spring geese only, dieldrin concentrations in fat were negatively associated with notocotylid infection. Figures 5.4 and 5.5 illustrate the relationship between probability of notocotylid infection and dieldrin concentrations in all geese and spring geese, respectively. Figure 5.4 showed how this relationship varies with low, medium and high levels of sumCHL (see model in Table 5.8). Because the final models contained transformed variables,

Figure 5.4 The predicted relationship between dieldrin concentrations in fat and probability of notocotylid infection at low (1.5 ng/g wet weight [ww], 25th percentile), median (2.4 ng/g ww) and high (8.9 ng/g ww, 75th percentile) concentrations of sumCHL in fat in Canada geese collected in 2001 in Labrador, Canada.



note: Solid lines indicate 25th (3.7 ng/g ww), median (17.6 ng/g ww) and 75th (119.1 ng/g ww) percentile concentrations of dieldrin.

Figure 5.5 The predicted relationship between dieldrin concentrations in fat and probability of notocotylid infection in a subsample of Canada geese collected in spring 2001 in Labrador, Canada.



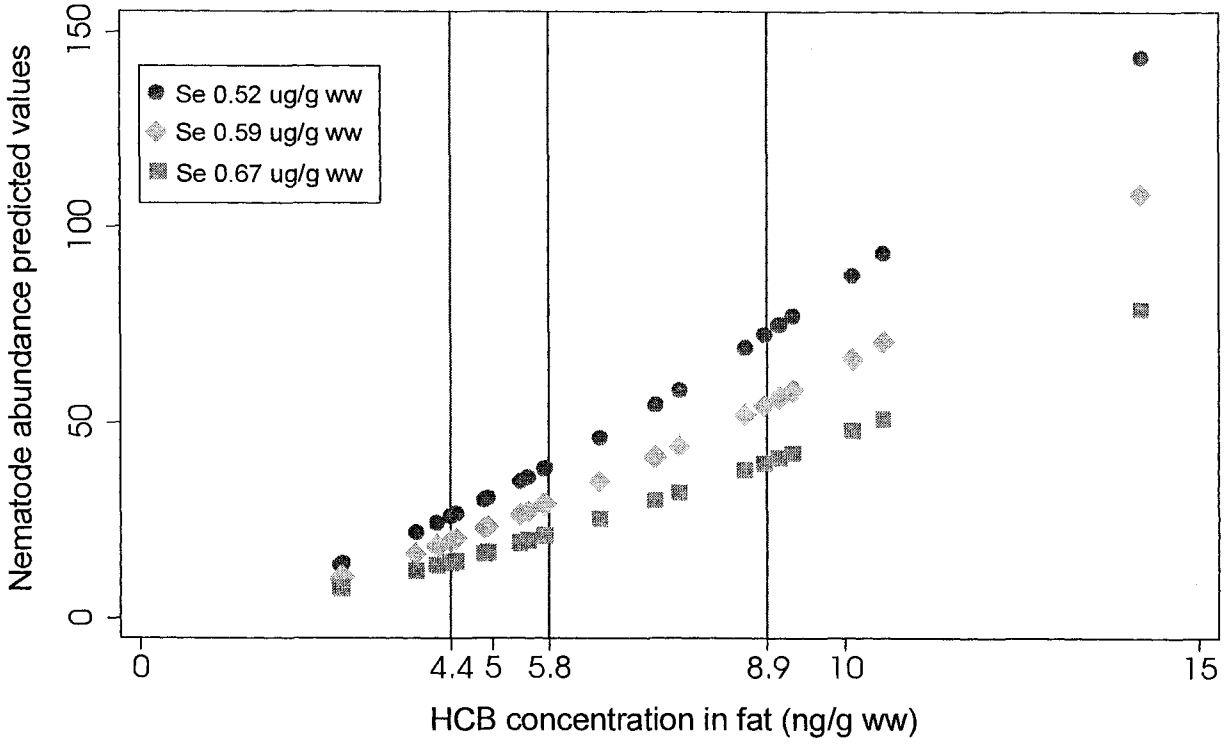
note: Solid lines indicate 25th (16.9 ng/g wet weight [ww]), median (46.1 ng/g ww) and 75th (130.3 ng/g ww) percentile concentrations of dieldrin.

interpretation of the coefficients is made easier by graphing the variables within the range of values found in the dataset and identifying specific values at important cut points, such as 25th, 50th and 75th percentiles.

In all geese, increasing dieldrin concentration in fat from 3.7 to 119.1 ng/g ww (25th to 75th percentile) decreased the probability of notocotylid infection from 0.79 to 0.01 when sumCHL concentrations were low (1.5 ng/g ww, 25thpercentile) and from 0.98 to 0.26 when sumCHL concentrations were high (8.9 ng/g ww, 75th percentile). For spring geese, increasing dieldrin concentrations in fat from 16.9 to 130.3 ng/g ww (25th to 75th percentile) decreased the probability of notocotylid infection from 0.69 to 0.30. When selenium levels were taken into account, OCs in fat were positively associated with nematode abundance in spring geese and *P. alveatum* prevalence in dioxin-tested geese. Figure 5.6 shows the relationship between HCB concentrations and predicted nematode abundance in spring geese at low (25th percentile), median and high (75th percentile) concentrations of selenium. Increasing HCB concentrations from the 25th to the 75th percentile (4.4 to 8.9 ng/g ww) increased predicted nematode abundance from approximately 26 to 72 nematodes if selenium concentration was low (0.52 µg/g ww or 25th percentile) and from approximately 14 to 40 nematodes if selenium concentration was high (0.67 µg/g ww or 75th percentile).

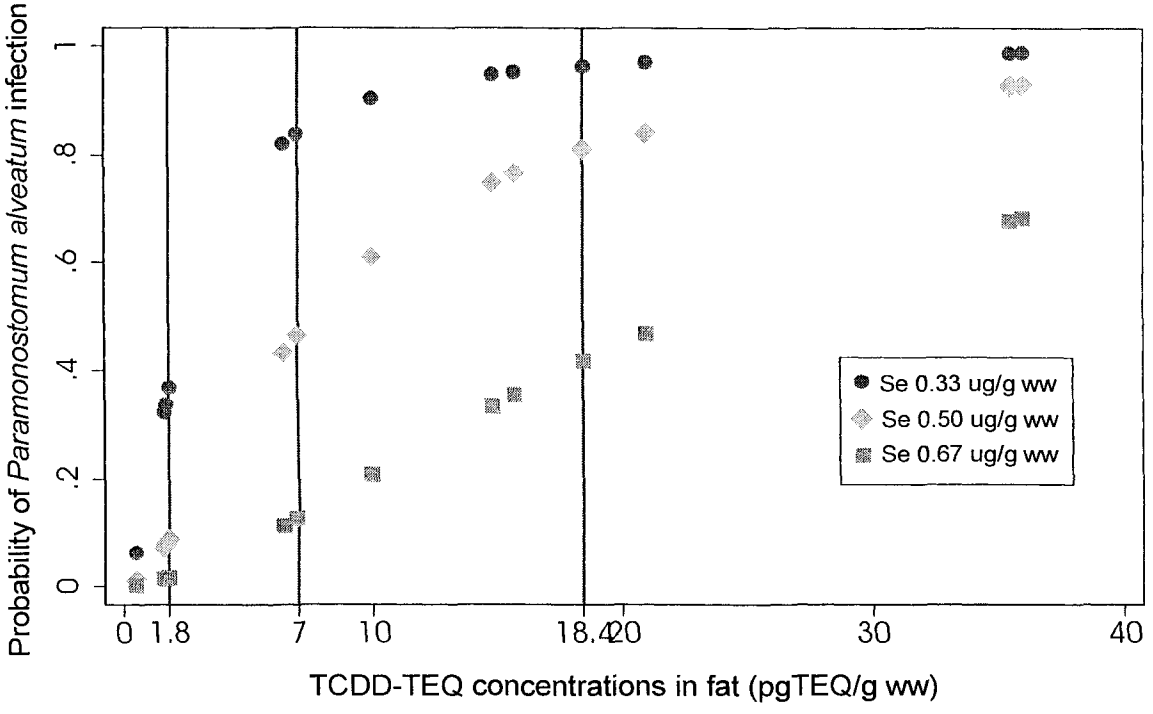
Figure 5.7 shows the relationship between TCDD-TEQ concentrations and *P. alveatum* prevalence in dioxin-tested geese at low (25th percentile), median and high (75th percentile) concentrations of hepatic selenium. Increasing TCDD-

Figure 5.6. The relationship between HCB concentrations in fat and nematode abundance predicted values at low (0.52 µg/g wet weight [ww], 25th percentile), median (0.59 µg/g ww) and high (0.67 µg/g ww, 75th percentile) concentrations of hepatic selenium in a subsample of Canada geese collected in spring 2001 in Labrador, Canada.



note: Solid lines indicate 25th (4.4 ng/g ww), median (5.8 ng/g ww) and 75th (8.9 ng/g ww) percentile concentrations of HCB.

Figure 5.7 The relationship between TCDD-TEQ concentrations in fat and probability of *Paramonostomum alveatum* infection at low (0.33 µg/g wet weight [ww], 25th percentile), median (0.50 µg/g ww) and high (0.67 µg/g ww, 75th percentile) concentrations of hepatic selenium in dioxin-tested Canada geese collected in 2001 in Labrador, Canada.



note: Solid lines indicate 25th (1.8 pgTEQ/g ww), median (7.0 pgTEQ/g ww) and 75th (18.4 pgTEQ/g ww) percentile concentrations of TCDD-TEQ.

TEQ concentrations from the 25th to the 75th percentile (1.8 to 18.4 pgTEQ/g ww) increased the probability of *P. alveatum* infection from 0.37 to 0.96 if hepatic selenium concentrations were low (0.33 µg/g ww or 25th percentile) and from 0.02 to 0.42 if hepatic selenium concentrations were high (0.67 µg/g ww or 75th percentile). Although statistically significant, this association was based on only a small subsample of geese (n=14) due, in part, to the high cost of PCDD/F analysis.

OC contaminants have been shown to be associated with immunotoxic effects in marine mammals such as polar bears and seals (eg. Ross et al., 1995 Bernhoft et al., 2000) and some studies have shown evidence of immunotoxic effects in free-ranging avian species (eg. Grasman et al., 2000; USDHHS 2000; Fox 2001). Few studies, however, have specifically examined the relationship between OC concentrations and parasitic infection. In glaucous gulls, Sagerup et al. (2000) found that increasing levels of a variety of OC pesticides and PCBs in the livers of apparently healthy gulls were positively associated with total intestinal nematode intensity. The authors speculated that the association may have been related to immunosuppression due to OC exposure resulting in an increased susceptibility to parasitic infection. TCDD-TEQ levels were not measured in the gulls, however, the levels of *p,p'*-DDE were approximately 20 times and PCBs were approximately 1000 times the levels found in geese in this study.

Although positive associations were found between HCB concentrations and nematode abundance in spring geese and sumCHL concentrations and

notocotylid prevalence in all geese, the levels of these OC contaminants were much lower than would be expected to lead to biological effects in the geese, based on published studies (Weimeyer, 1996).

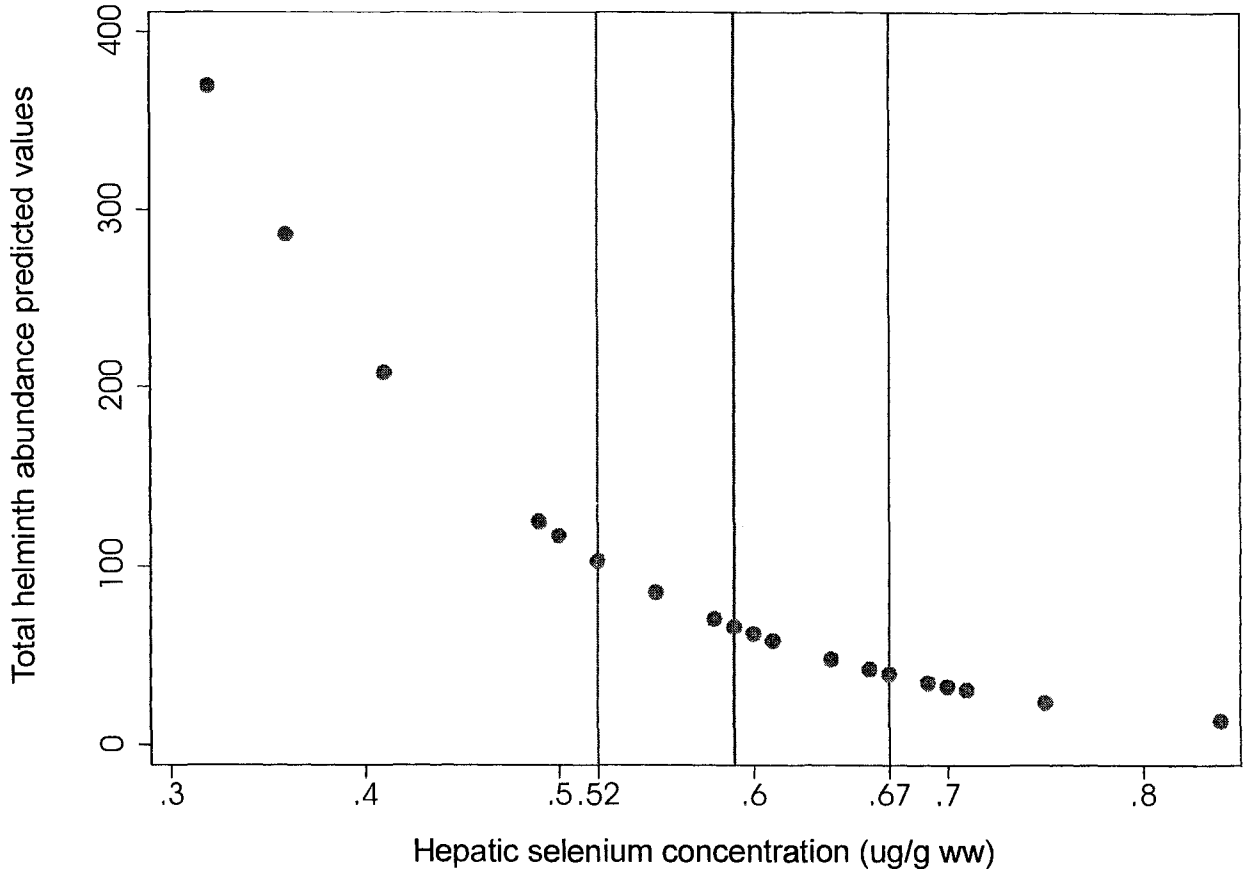
5.4.4.2.2 Selenium and helminth parasites

In the subsamples of spring geese and dioxin-tested geese, selenium concentrations were generally lower in geese with higher parasite numbers. This relationship was apparent with trematodes as well as nematodes either alone or in conjunction with elevated levels of OCs (Table 5.8).

As illustrated in Figure 5.6, hepatic selenium levels appeared to have a protective effect on nematode abundance in spring geese when HCB concentrations were considered. Similarly, in dioxin-tested geese, selenium levels appeared to have a protective effect on *P. alveatum* prevalence when TCDD-TEQ concentrations were considered (Figure 5.7). In addition, selenium levels alone were found to be negatively associated with total helminth abundance in spring geese (Figure 5.8). Increasing hepatic selenium concentrations from the 25th to the 75th percentile (0.52 to 0.67 µg/g ww) decreased predicted total helminth abundance by approximately 64 parasites.

Selenium plays an important role in many biological processes including the immune system (Dhur et al., 1990; Arthur et al., 2003). At high levels, selenium has toxic effects in birds and mammals (Heinz, 1996), however, adverse effects from selenium deficiency may also occur in domestic and wild animals (eg. white muscle disease in domestic lambs and calves) (Ohlendorf,

Figure 5.8. The relationship between hepatic selenium concentrations and total helminth abundance predicted values in a subsample of Canada geese collected in spring 2001 in Labrador, Canada.



note: Solid lines indicate 25th (0.52 µg/g wet weight [ww]), median (0.59 µg/g ww) and 75th (0.67 µg/g ww) percentile concentrations of hepatic selenium.

1996). Like other essential elements, it is usually homeostatically regulated, and uptake and loss of dietary selenium is rapid in avian tissues, especially liver (Ohlendorf, 1996). Normal dry-weight concentrations of selenium in livers and kidneys of several species of birds in freshwater habitats are between 4 and 10 µg/g (Ohlendorf, 1996) (approximately 1.0 - 2.5 µg/g ww). Selenium concentrations in the geese in the present study were slightly below this level.

Selenium is found in the anti-oxidant enzyme glutathione peroxidase, and the requirement for selenium is related to the amount of oxidant activity in an animal. The positive role of selenium in host immunity has been well documented in experimental and domestic animals. Several experimental studies have demonstrated positive effects of selenium and vitamin E supplementation on immunity in healthy animals (Dhur et al., 1990; Radostits et al., 2000), including chickens challenged with *Eimeria tenella* oocysts (Colnago et al., 1984). The interaction between selenium and vitamin E in relation to immunity, however, is unclear, and studies do not always clarify their separate roles (Dhur et al., 1990).

Overall, the relationship between selenium levels, immune response and protective immunity is complex (Dhur et al., 1990; Arthur et al., 2003), and the protective role of selenium in relation to parasite infection has not been convincingly demonstrated in experimental and domestic animals.

In free-ranging wildlife species, the relationship between selenium and parasitic infection has rarely been examined, and appears inconclusive. In a study of metal and selenium levels in free-ranging king and common eider ducks

in the Canadian Arctic in relation to various health parameters, Wayland et al. (2001) found no association between liver selenium levels and intestinal parasite abundance.

Selenium levels in this study were found to be positively associated with most metal and OC contaminants in all groups of geese examined. Elsewhere, selenium levels have been found to vary with the concentrations of some contaminants, most notably mercury. In free-ranging mammalian species with elevated total mercury concentrations in the liver, selenium levels were strongly positively correlated, often at a 1:1 molar ratio (Eisler, 1985; Scheuhammer et al., 1998). The beneficial effect of selenium relative to mercury toxicity has been well documented (Goyer, 1995; Thompson, 1996) and is believed to result from the formation of toxicologically inert Hg-Se complexes (Scheuhammer et al., 1998). In birds, the relationship between hepatic mercury and selenium appears to be more variable. In migratory birds, the relative rates of accumulation of the two elements may influence the ratio found in the liver at a given time (Ohlendorf, 1996). Similar relationships between selenium levels and other contaminants have not been well documented in free-ranging wildlife.

5.4.5 Analytical considerations

Correlation among contaminants can be problematic for statistical analyses in observational studies such as this. In assessing the relationship between contaminant levels and health parameters using multivariate regression analyses, the potential for multicollinearity (correlation among independent variables, or contaminants) must be considered. Several methods have been

proposed to help address the issue of multicollinearity in epidemiological studies, including principal components analysis (PCA) (Dohoo et al., 1996).

In an attempt to address this issue, PCA was used to condense the list of OC contaminants into a smaller number of uncorrelated variables. The fundamental assumption behind PCA is that there are some underlying factors that are responsible for the correlation among observed variables. Applying the assumptions to this study, it would appear that certain OC contaminants are likely to be present together due to common means of exposure (eg. through long-range atmospheric transport) and/or common biochemical properties. The PCA induces a partition of the correlation between contaminants in the original data into groups of uncorrelated variables called "factors". Factors are formed as the weighted averages of the groups of original correlated variables within each factor (Lafi and Kaneene, 1992). Therefore, a new dataset of factors is created that is considerably smaller than the original dataset.

PCA was conducted on the geese dataset. Those OC contaminants detected in fat samples of >20% of geese were included in the PCA (HCB, HE, sumCHL excluding HE, *p,p'*-DDE, sumDDT excluding *p,p'*-DDE, dieldrin and sumPCB). Two separate PCAs were conducted, one on all geese examined, and another on the subsample of dioxin-tested geese which included TCDD-TEQ in the original variables (8 altogether). The first principal component (or factor) (PC1) explained 75% of the total original variance for all geese and 79% for dioxin-tested geese. For both groups of geese, PC1 was positively associated with all of the original variables, indicating that geese with high PC1

scores had overall high levels of all OCs in fat tissue. Univariate and multivariate analyses were performed between PC1 and the health parameters for all geese and dioxin-tested geese in place of the original OC variables. PC1 was not found to be a significant predictor of any of the health parameters on multivariate analyses for all geese or the subsample of dioxin-tested geese, thus, the results of these analyses are not presented here.

One major limitation to the use of PCA is that the principal components produced from the original variables have no intrinsic meaning; they are merely mathematical constructs (Dohoo et al., 1996). Thus, the principal components produced from a PCA of a series of OC contaminants do not necessarily have toxicological relevance. Therefore, the lack of statistically significant results between health parameters and PC1 does not necessarily rule out a biological relationship between the health parameters measured and individual OCs or groups of OCs.

An example of a method used to combine a number of related compounds into a single measure with toxicological significance is the use of TCDD-TEQ (van den Berg et al., 1998). This measure was developed to determine the overall toxicological equivalency of a mixture of dioxin-like compounds (dioxins, furans and PCBs), in relation to 2,3,7,8-TCDD, that is based on the toxicological potential of individual compounds as they relate to specific biological endpoints (van den Berg et al. 1998). At the present time, it is not possible to assess the toxicological potential of a mixture including other OCs or metals in a similar manner.

The purpose of the univariate and multivariate analyses used in our study was to identify associations between contaminants and health parameters which may be biologically relevant. Aside from the issue of multicollinearity of independent variables, one other limitation of the approach used is related to the large number of variables examined and analyses performed. The probability of making a type I error (identifying an association where none exists) increased with each analysis. One method used to decrease the potential for Type I error in such instances is to apply a Bonferroni correction, however, it is generally accepted that this method of correction is too conservative and reduces the ability to identify true associations (Dohoo et al., 2003). Although some of the associations found in this study may have been spurious, results were interpreted conservatively to reflect this possibility and placed more emphasis on the general trends and associations found to be consistent across related contaminants and health parameters. Results were also interpreted in the context of the relevant literature and biological plausibility.

5.4.6 Conclusions

Overall, the levels of contaminants found were similar to those reported in other studies on Canada geese and were generally lower than those found in other waterfowl species. Spring geese had significantly higher concentrations of all contaminants measured compared to fall geese. In general, the levels of contaminants were well below the reported thresholds for biological effects in free-ranging animals with the exception of dioxins, furans and coPCBs (reported

as TCDD-TEQ), particularly in spring geese, which approached the reported thresholds for reproductive effects in some avian species.

In general, metal and organochlorine contaminants did not appear to be having an obvious negative impact on the health of the Canada geese examined in this study. Various contaminants were associated with BCS and certain parasitological parameters. However, the associations were not always consistent across subsets of geese or parasite parameters.

The associations seen between BCS and various OC contaminants were likely related to the high fat solubility of OC contaminants in general. The associations between parasitological parameters and various OC contaminants are difficult to interpret. Limited sample size may have been partly responsible for the inconsistent and limited detection of significant associations between contaminants and parasitological parameters measured. Alternatively, contaminant levels may have been below levels for the biological effects examined. Further investigations into potential relationships between health parameters and contaminant levels in Canada geese would benefit from an increased sample size of spring geese only.

One consistent finding was a negative association between selenium levels and intestinal trematode, nematode and total helminth abundance. However, a protective effect in relation to parasite infection has not been convincingly demonstrated yet. Selenium was also consistently positively correlated with metal and OC contaminant levels on univariate analyses. The biological

significance of these associations is unclear in free-ranging Canada geese and may warrant further investigation.

Canada geese are an important source of country food for Innu people in the area. Although this study does not directly address the question of the suitability of geese for human consumption relative to tissue contaminant levels, the results may be useful in assisting Labrador Innu communities in making this type of risk determination.

5.5 References

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6. GENERAL DISCUSSION

6.1 The rationale and benefits of the collaborative research model used in this thesis

This study was initiated by the Innu Nation in response to concerns expressed by Innu elders and hunters regarding perceived declines in overall environmental health and integrity of country food over the last number of decades. Observable physiological changes noted in wildlife include decreased reproductive success, altered behavioural patterns, declining fat reserves, increased parasite loads and changes in the quality of meat and organs. Concurrent with these observed changes, several regional industrial activities, including mining, military training exercises and hydroelectric development, had been initiated in the hunting areas. Many Innu felt that there were connections between these regional industrial activities and the decline in health and changes in behaviour observed in the animals. Furthermore, in recent years, the issue of environmental contaminants in northern ecosystems has come to the forefront. Contamination of the air, land and water is often cited by Innu as being potentially detrimental to the health and well-being of a variety of birds, mammals and fish (Armitage, 1990; Innes, 1998).

Several studies were undertaken in other northern regions to document pollution levels and determine the potential impacts of these contaminants in the environment, the animals and the people of the North (CACAR, 1997; AMAP, 1998). However, little attention was given to these questions in relation to the Innu of Labrador. The concerns of Innu people regarding contaminants in the

environment are broad. Opinions conveyed by Innu people during individual meetings and community gatherings throughout the course of the study were not formally documented. However, several people expressed concerns over the potential impact of contaminants on the health of animals and Innu people as well as on Innu culture, particularly with regards to Innu youth. Concerns were raised regarding the potential negative impact of contaminants in wildlife on the perception of country foods as being culturally, physically and socially beneficial.

As previously discussed, the issue of environmental contaminants in northern ecosystems is complex and the potential impacts of these compounds are varied. In particular, for the Innu and other First Nations and Inuit people with intimate ties to the land and animals, country food contamination is a concern at many levels. The scope of this collaborative study was limited to assessing the health of particular animals important to Innu people in Labrador as country foods, in relation to environmental contaminants. Many of the other concerns related to the broad issue of environmental contaminants, such as the potential exposure to people through eating country foods, were not addressed because the ability to adequately address this particular concern was outside of the expertise of the research group.

With their long history of hunting country foods in Labrador, Innu people have gathered a large amount of important knowledge relevant to the objectives of this research project. This traditional ecological knowledge (TEK) represents collective and individual experiences that are intergenerational and intertwined with cultural beliefs and value systems (Kuhn and Duerden, 1996). Although

formal collection and documentation of TEK was beyond the scope of this study, because of the collaborative nature of the research, Innu TEK was incorporated at all levels of the study.

From the initial stages, one Innu Nation co-researcher worked full-time on all aspects of the study and a number of other Innu Nation staff in Sheshatshit and Utshimassit participated as required. A great deal of importance was placed on the development of trusting relationships with participating families and other community members. Whenever possible, the Innu Nation and AVC co-researchers spent extended periods of time with participating families in the country during hunting trips prior to and during sample collection. Attempts were made to minimize interference with regular hunting activities and animal handling practices and to consult regularly with the individual hunters throughout the study period. The animals examined were hunted opportunistically and reflected the seasonal nature of the hunts as well as the demographic range of the animals normally killed. The types of samples and the manner in which they were taken, handled and ultimately disposed of were directly influenced by the wishes of elders and hunters collectively, as well as on an individual basis. All of the animals examined were eaten immediately or brought to the communities for distribution except for two porcupines, which were considered to be too thin and were not eaten.

Emphasis was placed on informing the participating hunters verbally and in writing of the interim findings as test results became available, and incorporating feedback where offered. In November 2002, gatherings were held in both

communities for all community members during which presentations of the results were made. Summary reports were also prepared and distributed individually to all participating hunters.

The collaborative nature of the study insured that the examination of the animals and the collection of tissue samples were meaningful and acceptable scientifically and culturally to the AVC and Innu Nation co-researchers, as well as participating families and community members. Throughout the study, compromises were made by all parties, which enabled important cultural traditions to be maintained, in the context of scientific rigour.

For example, the elders of Utshimassit chose the black bear as one of the two species to be tested for that community. However, during the time available for sample collection, no bears were hunted around Utshimassit, and one bear was killed near Sheshatshit. Therefore a representative sample of black bears, the fourth species chosen for study, was not examined. The choice of black bear for this study by Utshimassit elders may have reflected a more traditional hunt and emphasis on the cultural importance of the species. During the gathering at which the animals were chosen for study, one elder expressed concerns about the bears near Utshimassit being tranquilized and moved by provincial wildlife officials. The potential impacts of this practice on the bears' health and their suitability for eating were questioned. Unfortunately, these concerns could not be addressed in this study. The one bear killed near Sheshatshit during the study period was necropsied on site, and a report of the necropsy findings, including contaminants analysis results, was produced and

presented to the submitting party. A copy of the report and accompanying cover letter are shown in Appendix E. This bear was shot because it was exhibiting aggressive behaviour. Nothing was found on examination of the bear to account for this behaviour. Tissue contaminant levels (metals and OCs) were all low except hepatic cadmium concentration, which was moderate (13.3 µg/g ww).

The parameters chosen for health assessment of the animals reflected relevant biomarkers in relation to contaminant exposure, some of which were also meaningful to Innu hunters and those familiar with handling and cleaning the animals. An evaluation of some of the health parameters, such as body condition and gross abnormalities, can be applied to harvested animals during regular hunting activities. Innu hunters and those cleaning the animals commonly used an assessment of fat stores to evaluate the health of an animal, as was apparent during the study. Although few gross lesions were evident in the animals and tissues examined, comments made during community gatherings and during individual conversations indicated that evidence of gross abnormalities is also used as an assessment of health in a variety of animal species. In general, intestinal helminth parasites, although sometimes grossly apparent, are difficult to assess quantitatively in the field. The advantages of assessing this health parameter in hunter-killed animals are that intestines are usually easily accessible for sampling and laboratory quantification of parasite infection can serve as a general index of immune function. In caribou, *F. magna* (large American liver fluke) infection can be easily recognized grossly and with

some extra effort during cleaning of the carcass, a rough estimate of the fluke abundance can also be made.

In addition to the animals examined for the study, a small number of animals that were found to be unhealthy by Innu hunters were submitted for necropsy to the AVC (three ducks and one Canada goose from one hunter, and one porcupine from another hunter which was also included in the study). Necropsy reports of these submissions were produced and presented to the submitting parties. Copies of the reports and cover letters are shown in Appendix E. The Canada goose and porcupine were considered to be thin for the time of year. The Canada goose had a chronic fracture of the pubic bone which explains its poor condition. No explanation for the poor condition of the porcupine could be found. Contaminant concentrations from the animals were all low.

6.2 Interpretation of results in the context of contaminant studies in free-ranging wildlife

All of the analysed contaminants in our study have the potential to cause negative health effects in bird and mammal populations, including humans, based on laboratory studies and accidental human exposures (Gilman et al., 1997). However, establishing a causal association between environmental contaminants and biological effects in free-ranging wildlife is inherently difficult, and the following cautions should be considered before drawing conclusions regarding the results found among the three species tested.

Often a weight-of-evidence approach is used, based on associations between identified effects and the presence and relative concentrations of contaminants measured in the environment and/or in animals. In addition to many logistical difficulties of obtaining samples from free-ranging wildlife, a variety of confounding factors add to the difficulty in evaluating the effects of these contaminants in wild populations on specific health parameters and interpreting the results in the context of other studies. Some of these confounding factors include those related to methodology of wildlife studies in general. For many wildlife populations, there are limited baseline data for many health parameters, which leads to difficulties in determining what is normal and what are deviations from normal. Furthermore, wild populations are exposed to many environmental pressures which can be known or potential sources of variation of health parameters, such as, weather patterns (Bishop et al., 2000), predator-prey relationships and disease exposure (Larson et al., 1996), natural population cycles, habitat alterations and human disturbance. Depending on the species and population, it can be difficult to access individuals or populations, particularly if species are threatened or endangered.

There are also several biological factors related to environmental contaminants that lead to differences in the composition of contaminants to which free-ranging populations are exposed. For example, DDT is degraded or metabolized over time in the environment and in biota to DDE and DDD. Thus, the composition of a mixture of DDT that a free-ranging animal is exposed to through the food chain can be considerably different from that available for use

in a laboratory setting. For some organochlorine pesticides and PCB congeners, the metabolites are as toxic as, or more toxic than, the parent compounds and can also be very persistent in certain tissues (Fry, 1995; Leonards et al., 1997).

Free-ranging populations are often exposed to mixtures of contaminants that may coaccumulate and may not necessarily be accounted for in contaminant studies. For example, correlations have been found in bald eagles in the Great Lakes between levels of PCBs and *p,p'*-DDE (Bowerman et al., 1995) and in tree swallows in Ontario between current-use pesticides such as organophosphates and residual organochlorines (primarily *pp'*-DDE) (Bishop et al., 2000). Interactive biological effects have been demonstrated between and among environmental contaminants. Laboratory studies have shown some evidence for additive and non-additive (synergistic or antagonistic) effects among PCBs, dioxins and furans (Tryphonas, 1995; USDHHS, 2000). Interactions have also been found between PCBs and methyl mercury and PCBs and *p,p'*-DDE (USDHHS, 2000).

Variations in species susceptibility to various environmental contaminants have also been reported. For example, mink are among the most sensitive species to the toxic effects of dioxin exposure (Hochstein et al., 1998). In contrast, the closely related ferret appears to be less sensitive to PCBs than mink (Leonards et al., 1997). Thus, caution must be used in extrapolating results from studies on one species to another, even if they are closely related.

6.2.1 Comparison of results across the three species examined

A few generalities across species in relation to contaminant levels and health parameters can be made based on the results of this study. While the three species studied are all herbivores, porcupines have relatively small home ranges and can be considered non-migratory, caribou migrate seasonally within the Québec-Labrador peninsula and Canada geese migrate through and/or breed throughout Labrador and overwinter in the Northeastern United States, leading to challenges in making generalities across the three species, as noted below.

For all three species, most of the animals examined appeared to be healthy based on an assessment of fat reserves and gross and microscopic examination of carcasses. In general, the helminth parasites examined did not appear to be negatively impacting the health of the animals examined, although there was some evidence of an association between increased intestinal nematode abundance and poor body condition in porcupines.

Caribou and porcupines had very low levels of most contaminants in their tissues. However, as expected, the cadmium level in caribou kidneys was elevated in older animals. No evidence of renal histopathology or other negative health effects that could be attributed to elevated cadmium levels was found in the caribou. In the caribou, elevated cadmium levels and decreased levels of selenium were associated with an increased abundance of *F. magna* (the large American liver fluke). This association may, in part, be related to age, however, and warrants further investigation. Although no evidence was found to indicate

negative health effects relative to cadmium levels or *F. magna* infection in the caribou examined, the average age of the animals was low (3 years), and health effects, if present, would be more likely to be apparent in older animals.

In the geese, significantly higher levels of contaminants, particularly organochlorines, were found in those hunted in the spring compared to those hunted in the fall. However, overall, the geese had low levels of organochlorine contaminants relative to other avian species in North America. The seasonal differences in contaminant levels were likely due to a number of related factors including: contaminant levels in the overwintering areas, differences in age distribution between seasons, and distribution and dynamics of lipid stores in the geese.

The identification of these recognizable characteristics in relation to contaminant burdens (age in caribou and season in Canada geese) does not imply that current harvesting practices should be altered based on the findings of this study. Risk management decisions regarding country food contaminants should be made at the community and individual levels, based on sound research, (ideally undertaken locally), as well as a variety of social (such as age and lifestyle considerations) and cultural factors that may be protective to Innu families and communities.

In relation to reported threshold levels for biological effects, the tissue concentrations of organochlorines and metals appear to be well below levels of concern with the possible exception of TCDD-TEQ in geese. However, these comparisons must be made cautiously, as little comparative data exists for

dioxins and related compounds in Canada geese, and large differences in species sensitivity have been reported.

Overall, we found little evidence that tissue contaminant levels were associated with negative health effects in the Canada geese. However, in a small (n=14) subsample of geese, elevated TCDD-TEQ levels were associated with increased odds of intestinal trematode infection (*P. alveatum*) when selenium levels were also taken into account. Laboratory and field-based studies have demonstrated decreased immune responses in free-ranging wildlife species with elevated levels of organochlorines including dioxin-like compounds. However, it is unclear whether exposure to environmentally relevant concentrations of these compounds can lead to an increased susceptibility to intestinal parasites.

Although elevated selenium levels can lead to toxicity in mammals and birds, selenium levels were considered in this study because of the reported association between elevated mercury and selenium levels. Selenium and inorganic mercury are often found in a 1:1 molar ratio, particularly in marine mammals. A similar relationship has also been reported in birds, where selenium has been reported to reduce the toxic potential of mercury through the formation of insoluble, non-toxic complexes.

In all three species examined, mercury levels were very low. However, in caribou and Canada geese, selenium levels were found to be positively associated with cadmium. Positive associations were also found between organochlorine levels and selenium in the geese. Furthermore, in both caribou

and Canada geese, selenium levels were consistently negatively associated with helminth parasite infection (hepatic *F. magna* in caribou and intestinal nematodes and trematodes in Canada geese), either alone or when metal or organochlorine contaminants were considered. A similar relationship was not found in the porcupines.

Selenium, an essential trace element, plays an important role in many biological processes (Dhur et al., 1990; Arthur et al., 2003). It is generally accepted that selenium has an important immunological function, however, it has not been convincingly related to parasite infection. The role of selenium in the health of free-ranging wildlife has not been thoroughly examined. Few studies on biological effects of environmental contaminants in free-ranging wildlife have considered selenium levels. In a series of studies on eiders in the Canadian Arctic, Wayland et al. (2001, 2002, 2003) examined various health parameters, including intestinal helminth parasite infection, body condition, stress response and immunological function, in relation to tissue metal levels, including selenium. The authors found some evidence of a negative relationship between stress response and liver selenium levels. However, no other health parameters were consistently associated with selenium levels.

Apart from the documented association with increased mercury levels, it is unclear what the role of selenium may be in free-ranging wildlife species in relation to environmental contaminants and health effects.

6.2.2 Study results compared to reported thresholds for biological effects

One other method used to assess the potential health risks due to contaminant exposure is to compare our study contaminant levels to reported threshold levels for biological effects derived from laboratory, or more preferably, field studies in similar species (deMarch et al., 1998). Although this approach is commonly used, extrapolation of threshold levels from the laboratory to the field is fraught with much uncertainty. In addition to the factors discussed above, wild animals are generally exposed to mixtures of contaminants modified by the environment throughout their lifetimes, while laboratory experiments often involve exposure to one contaminant over relatively short periods of time (CACARII, 2003).

Although there are no data available specifically for the species studied in this thesis (caribou, porcupines and Canada geese), we have endeavoured to compare threshold levels for biological effects with those obtained in our study in order to put our study results in context. The threshold data used for comparison are presented in Tables 6.1 and 6.2 and are adapted in part from summary tables presented in deMarch et al. (1998), Braune et al. (1999) and CACARII (2003). The reported threshold levels are often tissue specific and may be for different tissues (eg. brain) than those that we sampled, and therefore, interpretations should be made with caution.

For the geese in the current study, only DDT and *p,p'*-DDE concentrations in fat in some spring geese and TCDD-TEQ concentrations in fat in spring and

Table 6.1. Threshold tissue residue levels for selected contaminants associated with biological effects in avian species (adapted from reviews presented in Braune et al., 1999; deMarch et al., 1998; CACARII, 2003).

Contaminant [a]	Species	Tissue	Effects	Concentration [b]	Source [c]
Mercury	non-marine birds	liver & kidney	likely toxic effects	20-30	1
Cadmium	non-marine birds	liver	likely toxic effects	40	2
		kidney	likely toxic effects	100	
Selenium	waterfowl	liver	to prevent reproductive problems	3.0 - 6.0	3
		liver	to prevent health problems	10.0	3
		liver	possible lethal effects	>20.0	3
Lead	waterfowl	liver	background levels	<2	4
			subclinical poisoning	2 - 6	4
			clinical poisoning	6 - 15	4
			severe clinical poisoning	>15 - 20	4
Dieldrin	mallard duckling	liver	NOAEL[d] - survival, growth and behavioural parameters	1.0	5
	mallard duckling	liver	LOAEL[e] - survival, growth and behavioural parameters	7	5
	sparrowhawks & kestrels	liver	NOEL[f] - population decline	1.0	5
HE	Canada geese	egg	decreased nest success	>10	6
	kestrels	egg	decreased nest success	<3	7
	passerines	brain	lethality	>8.0	6
HCB	chicken	egg	NOEL - egg hatching	100	6
	Canada geese	egg	no effect on reproduction	2.97	6

Table 6.1 continued

Contaminant [a]	Species	Tissue	Effects	Concentration [b]	Source [c]
sPCB	chickens	egg	decreased hatching success	1 - 5	8
	terns, cormorants, doves, eagles	egg	decreased hatching success	8 - 25	8
	great cormorants, gulls, passerines, pheasants	brain	lethality	75 - 300	8
	chickens	egg	NOAEL - hatching success	0.36	9
	chickens	egg	LOAEL - hatching success	1.5	9
	Forster's tern	egg	NOAEL - hatching success	2.3	9
	double-crested cormorants	egg	LOAEL - egg mortality	3.5	9
	common tern	egg	LOAEL - hatching success	7.6	9
	black-crowned night heron	egg	NOAEL - reproduction	10.9	9
sumDDT	seabirds	egg	eggshell thinning	15 - 20	10
p,p'-DDE	brown pelicans	egg	eggshell thinning, decreased productivity	3.0	7, 11
	peregrine falcons	egg	eggshell thinning, decreased productivity	30	7, 11
	loon	egg	eggshell thinning, decreased productivity	47	12
	osprey	egg	eggshell thinning, decreased productivity	10	7
Mirex	kestrels	liver	reduced sperm count	1.6	6
TCDD-TEQ [j]	bald eagle, chicken	egg?	NOAEL - reproduction	1.5 [g]	9
	herring gull	egg?	NOAEL - reproduction	10 [g]	9
	wood duck	egg?	LOAEL - reprduction	20 [g]	9
	Forster's tern	egg?	NOAEL - reproduction	200 [g]	9

[a] HE= heptachlor epoxide HCB=hexachlorobenzene; sumPCB= sum polychlorinated biphenyl congeners; sumDDT= sum dichlorodiphenyltrichloroethanes and metabolites (including *p,p'*-DDE); TCDD-TEQ= sum of World Health Organization 2,3,7,8-TCDD Toxic Equivalence Factors for PCDD/Fs and coPCBs

Table 6.1 continued

[b] µg/g wet weight (ww)
[c] 1- Thompson, 1996; 2- Furness, 1996; 3- Heinz, 1996; 4- Pain, 1996; 5- Peakall, 1996; 6- Weimeyer, 1996; 7- Blus, 2003; 8- Hoffman, 1996; 9- deMarch et al., 1998; 10- CACARII, 2003; 11- Blus, 1996; 12- Braune et al., 1999.
[d] NOAEL = No adverse effect level
[e] LOAEL = Lowest adverse effect level
[f] NOEL = No effect level
[g] pgTEQ/g ww

Table 6.2. Threshold tissue residue levels for selected contaminants associated with biological effects in mammalian species (adapted in part from reviews presented in deMarch et al., 1998 and CACARII, 2003). Thresholds are given for wild mammals where available, otherwise those for laboratory mammals are used.

Contaminant	Species	Tissue	Effects	Concentration	Source [b]
Mercury	mammals	liver & kidney	likely toxic effects	30 [a]	1
Cadmium	small mammals	kidney	likely toxic effects	100 [a]	2
	marine mammals	liver	potential renal dysfunction	>20 - 200 [a]	3
	marine mammals	kidney	potential renal dysfunction	>50 - 400 [a]	3
Selenium	terrestrial mammals	liver	hepatic lesions	>7 [a]	4
Lead	mammals	liver	clinical signs	>30 [c]	5
	mammals	kidney	clinical signs	>90 [c]	5
sumPCB [d]	mink	fat	reproductive effects	13 - 25 [a]	6
		muscle	reproduction	9.0 [e]	7
	seals	? [f]	NOEL [g] - vitamin A reduction	4.0 [e]	3
		?	poor reproductive success	77 [e]	7
	otter	muscle	reproduction	7.5 [e]	7
	marine mammals	?	LOEL [h] - vitamin A reduction	11.0 [e]	3
TCDD - TEQ [i]	otter	?	NOAEL [j] - vitamin A reduction	84 [l]	3
	mink	?	LOAEL [k] - kit survival	490 [l]	3
	seals	fat?	immunosuppression	69 [m]	8

Table 6.2 continued

[a] µg/g wet weight (ww)
[b] 1- Thompson, 1996; 2- Cooke and Johnson, 1996; 3- CACARII, 2003; 4- Dietz et al., 1998; 5- Ma, 1996; 6- O'Hara and Rice, 1996; 7- deMarch et al., 1998; 8- Ross et al., 1995.
[c] µg/g dry weight
[d] Sum polychlorinated biphenyl congeners
[e] µg/g lipid weight
[f] ? = tissue was not specified
[g] NOEL = No effect level
[h] LOEL = Lowest effect level
[i] sum of 2,3,7,8-TCDD Toxic Equivalence Factors for PCDD/Fs (polychlorinated dibenzo-p-dioxins/furans) and coPCBs (coplanar PCBs)
[j] NOAEL = No adverse effect level
[k] LOAEL = Lowest adverse effect level
[l] pgTEQ/g ww
[m] pgTEQ/g lipid weight

fall geese approached or exceeded the threshold for reproduction in some avian species. However, the threshold levels are based on concentrations in eggs (Table 6.1).

For caribou and porcupines, all tissue contaminants were well below the threshold levels except for one caribou with renal cadmium concentrations that approached the threshold levels for toxic effects (based on small mammals) (Table 6.2).

6.3 Recommendations for further research

Although, in general, environmental contaminants do not appear to be impacting the health of caribou, porcupines and Canada geese in Labrador, several aspects of the study may warrant further investigation. A representative sample of caribou typically hunted by Innu was examined, which included a number of younger animals and calves. Some evidence was found to indicate an association between renal cadmium and selenium levels and *F. magna* infection. Further studies would benefit from focusing on older caribou, which tend to have higher levels of cadmium and greater prevalence and abundance of *F. magna.* A more detailed assessment of the potential renal histopathology related to cadmium levels, particularly in older animals, would also be of interest as these types of studies have rarely been undertaken in free-ranging ungulates.

For Canada geese, further investigations should be targeted at spring geese which we found to have significantly higher levels of most contaminants measured as compared to geese hunted in the fall. It appears as though health effects from contaminant exposure would be more likely to occur in the spring or

summer, when organochlorine contaminants are at higher levels and fat reserves containing OCs are being mobilized. Specifically, further investigation into the levels of TCDD-TEQ in geese in relation to immunological or reproductive parameters would be of interest.

No evidence of contaminant-related health effects in porcupines was found, however, the health impacts of the high intestinal parasite burdens found in these animals may warrant further investigation.

Most of the animals examined appeared to be healthy, which is often the case with hunter-killed animals. Over the long term, the collection and examination of animals with observed abnormalities during the regular seasonal hunt would lead to a greater understanding of the health of these and other species hunted by Innu people in Labrador. This type of long-term collaboration could help to further address the concerns expressed by Innu people regarding the overall health of the land and animals in a manner that would be relevant and meaningful to all participants.

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Appendix A. Reference list of species names mentioned in the Thesis.

American black duck	*Anas rubripes*
American eel	*Anguilla rostrata*
American kestrel	*Falco sparverius*
Arctic char (gen)	*Salvelinus alpinus*
Arctic char (landlock)	*Salvelinus alpinus*
Arctic Fox	*Alopex lagopus*
Arctic Hare	*Lepus arcticus*
Atlantic herring	*Clupea harengus*
Atlantic Puffin	*Fratercula arctica*
Atlantic salmon	*Salmo salar*
bald eagle	*Haliacetus leucocephalus*
Barrow's goldeneye	*Bucephala islandica*
beaver	*Castor canadensis*
beluga whale	*Delphinapterus leucas*
black bear	*Ursus americanus*
black guillemot	*Cepphus grylle*
black scoter	*Melanitta nigra*
black-crowned night heron	*Nycticorax nycticorax*
black-legged kittiwake	*Ryssa tridactyla*
blue-winged teal	*Anas discors*
brook trout	*Salvelinus fontinalis*
brook trout (sea run)	*Salvelinus fontinalis*
brown pelican	*Pelecanus occidentalis*
bufflehead	*Bucephala albeola*
burbot	*Lota lota*
cackling goose	*Branta canadensis*
Canada goose	*Branta canadensis*
Canada goose (AP)	*Branta canadensis interior*
Canada goose (NAP)	*Branta canadensis canadensis*
capelin	*Mallotus villosus*
caribou	*Rangifer tarandus*
Caspian tern	*Sterna caspia*
cattle (domestic)	*Bos taurus*
chicken	*Gallus domesticus*
common eider	*Somateria mollissima*
common goldeneye	*Bucephala clangula*
common loon	*Gavia immer*
common merganser	*Mergus merganser*
common murre	*Uria aalge*
common tern	*Sterna hirundo*
cormorants (gen)	*Phalacrocorax spp.*
dolphins (gen.)	*Delphinidae spp.*
double-crested cormorant	*Phalacrocorax auritus*
dovekie	*Alle alle*
eider duck (gen)	*Somateria spp.*
ferret	*Mustela spp.*
Forster's tern	*Sterna forsteri*
glaucous gull	*Larus hyperboreus*
goat (domestic)	*Capra hircus*
great blue heron	*Ardea herodias*
great cormorant	*Phalacrocorax carbo*
great skua	*Stercorarius skua*
greater black-backed gull	*Larus marinus*
greater scaup	*Aythya marilla*
green-winged teal	*Anas crecca*
grey seal	*Halichoerus grypus*
gulls (gen)	*Larus spp.*
harbour porpoise	*Phocoena phocoena*
harbour seal	*Phoca vitulina*
harlequin duck	*Histrionicus histrionicus*
herring gull	*Larus argentatus*
horse (domestic)	*Equus caballus*
ivory gull	*Pagophila eburnea*

king eider	*Somateria spectabilis*
kittiwake	*Rissa sp.*
lake trout	*Salvelinus namaycush*
lake whitefish	*Coregonus clupeaformis*
lesser black-backed	*Larus fuscus*
lesser scaup	*Aythya affinis*
lobster	*Homarus americanus*
longnose sucker	*Catostomus catostomus*
loon (gen.)	*Gavia sp.*
lynx	*Lynx canadensis*
mallard	*Anas platyrhynchos*
marten	*Martes americana*
merganser (gen.)	*Mergus spp.*
mink	*Mustela vison*
moose	*Alces alces*
muskrat	*Ondatra zibethicus*
northern cod	*Gadus callerias*
northern fulmar	*Fulmarus glacialis*
northern pike	*Esox lucius*
northern pintail	*Anas acuta*
oldsquaw duck	*Clangula hyemalis*
osprey	*Pandion haliaetus*
European otter	*Lutra lutra*
otter	*Lutra canadensis*
ouananiche (landlocked Atlantic	*Salmo salar*
peregrine falcon	*Falco peregrinus*
pheasant	*Phasianus sp.*
polar bear	*Ursus maritimus*
porcupine	*Erethizon dorsatum*
rainbow smelt	*Osmerus mordax*
rat	*Rattus sp.*
red fox	*Vulpes vulpes*
red-breasted	*Mergus serrator*
red-throated Loon	*Gavia stellata*
ring-billed gull	*Larus delawarensis*
ringed seal	*Phoca hispida*
rock ptarmigan	*Lagopus mutus*
rockcod	*Microgadus tomcod*
ruffed grouse	*Bonasa umbrellus*
sauger	*Stizostedion canadense*
scaup	*Aythya sp.*
scoter (gen.)	*Melanitta spp.*
sculpin (gen.)	*Cattidae spp.*
seals (gen)	*Phoca spp.*
sheep (domestic)	*Ovis sp.*
snapping turtle	*Chelydra serpentina*
snow goose	*Chen caerulescens*
snowshoe hare	*Lepus americanus*
snowy owl	*Nyceta scandiaca*
sparrowhawk	*Accipiter gularis*
spruce grouse	*Falcipennis canadensis*
stickleback (gen.)	*Gasteroseidae spp.*
striped dolphin	*Stenella coeruleoalba*
surf scoter	*Melanitta perspicillata*
terns (gen)	*Sterna spp.*
thick-billed murre	*Uria lomvia*
tree swallow	*Tachycineta bicolor*
wapiti	*Cervus elaphus*
weasel (gen.)	*Mustela spp.*
white-fronted goose	*Anser sp.*
white sucker	*Catostomus commersoni*
white-winged scoter	*Melanitta fusca*
white-tailed deer	*Odocoileus virginianus*
white-tailed ptarmigan	*Lagopus leucurus*
willow ptarmigan	*Lagopus lagopus*
wolf	*Canis lupus*
wood duck	*Aix sponsa*

Appendix B. List of acronyms and abbreviations used in the Thesis with selected definitions

2,3,7,8-TCDD	2,3,7,8-Tetrachlorodibenzo-p-dioxin (also refered to as dioxin or TCDD)
AAS	Atomic absorption spectrophotometer (spectrophotometry)
Ah receptor	Aryl hydrocarbon (dioxin) receptor
ALAD	delta-aminolevulinic dehydratase
AMAP	Arctic Monitoring and Assessment Programme
AP	Atlantic population (Canada geese)
AVC	Atlantic Veterinary College
BCS	Body condition score
CACAR	Canadian Arctic Contaminants Assessment Report
CHL	Chlordane
CI	Confidence interval
coPCB	Coplanar PCBs. PCB congeners that take on a planar configuration and are dioxin-like. These include both mono- and non-ortho PCBs.
DDD	1,1-dichloro-2,2-bis(4-chlorophenyl)ethane
DDE	1,1-dichloro-2,2-bis(4-chlorophenyl)ethylene
DDT	1,1,1-trichloro-2,2-bis(4-chlorophenyl)ethane
ECD	electron capture detector
EPA	Environmental Protection Agency (US)
GC	gas chromatograph (chromatography)
HCB	Hexachlorobenzene
HCH	Hexachlorocyclohexane
HpCDD	Heptachlorodibenzo-*p*-dioxin
HpCDF	Heptachlorodibenzofuran
HRGC	High resolution gas chromatography
HRMS	High resolution mass spectrometer
HxCDD	Hexachlorodibenzo-*p*-dioxin
HxCDF	Hexachlorodibenzofuran
KFI	Kidney fat index
ln	natural log

LOAEL	lowest-observed-adverse-effect-level. Lowest concentration or amount of a substance, found by experiment or observation, which causes an adverse alteration of morphology, functional capacity, growth, development, or life span of the target organism distinguishable from normal (control) organisms of the same species and strain under defined conditions of exposure. (IUPAC Compendium of Chemical Terminology, 2nd ed 1997)
LOEL	lowest-observed-effect-level. Lowest concentration or amount of a substance, found by experiment or observation, that causes any alteration in morphology, functional capacity, growth, development, or life span of the target organism distinguishable from normal (control) organisms of the same species and strain under the same defined conditions of exposure. (IUPAC Compendium of Chemical Terminology, 2nd ed 1997)
NAP	North Atlantic population (Canada geese)
NOAEL	no-observed-adverse-effect-level. Greatest concentration or amount of a substance, found by experiment or observation, which causes no detectable adverse alteration of morphology, functional capacity, growth, development, or life span of the target organism under defined conditions of exposure. (IUPAC Compendium of Chemical Terminology, 2nd ed 1997)
NOEL	no-observed-effect-level. Greatest concentration or amount of a substance, found by experiment or observation, that causes no alterations of morphology, functional capacity, growth, development, or life span of the target organisms distinguishable from those observed in normal (control) organisms of the same species and strain under the same defined conditions of exposure. (IUPAC Compendium of Chemical Terminology, 2nd ed 1997)
OCDD	Octachlorodibenzo-*p*-dioxin
OCDF	Octachlorodibenzofuran
OCs	Organochlorines
PCA	Principal Components Analysis
PCB	Polychlorinated biphenyls
PCDDs	Polychlorinated dibenzo-*p*-dioxins
PCDFs	Polychlorinated dibenzofurans
PeCDD	Pentachlorodibenzo-*p*-dioxin
PeCDF	Pentachlorodibenzofuran
SD	Standard deviation
SE	Standard error
sumCHL	sum of chlordanes
sumDDT	sum of DDT, DDE and DDD concentrations
sumPCB	sum of a number of individual PCB congeners
TCDD-TEQ	TCDD toxic equivalents (calculated by summing applicable TEFs)
TCDF	Tetrachlorodibenzofuran
TEF	Toxic equivalency factor

Appendix C. Pilot study summary report

Innu Nation Country Food Contaminant Pilot Study - 1997

Summary Report, January 2001

Prepared by: Beth Pollock, Atlantic Veterinary College (in consultation with: Drs. Andy Tasker, John VanLeeuwen, Scott McBurney, Pierre-Yves Daoust, Atlantic Veterinary College)

This summary report is based on the Innu Nation report "Country Food Contaminant Study 1997 Final Report" written by Larry Innes, April 1998.

In Spring 1997, researchers Scott McBurney and Pierre-Yves Daoust visited two outpost camps - Minipi and Anukuashash-nipi and sampled a number of different animals. In October and November 1997, caribou were sampled from Western Labrador along the Esker and Wabush roads. The animals sampled included:

Minipi	4 brook trout
	3 pike
	2 spruce partridge
	2 surf scoters
	1 black scoter
	1 Canada goose
Anukuashash-nipi	1 spruce partridge
	1 muskrat
	1 porcupine (found dead)
Western Labrador	9 caribou

Samples of **muscle, liver** (**kidney** in the caribou) and **fat** were taken from the animals. The samples were sent to laboratories in Prince Edward Island (PEI) and British Columbia to test for contaminants.

What are contaminants?

Contaminants are substances that are not normally found in animals or the environment or they are found in larger than normal amounts. Some contaminants are created by human activity like burning garbage or making transformer oil while others are formed by natural processes like forest fires. Contaminants are not

necessarily harmful to the environment or to human or animal health unless they are present in large amounts.

Which contaminants were looked for?

Two main groups of contaminants were tested for in the samples, heavy metals and "Persistent Organic Pollutants".

Heavy Metals

These metals are natural and are found in rocks and soil. Some human activities like mining and flooding land for reservoirs can cause some of these metals to be found in unnatural amounts in some areas. Cadmium, mercury and lead are metals that can become contaminants.

"Persistent Organic Pollutants (POPs)"

This group includes human-made chemicals like pesticides which are used in agriculture for controlling insects and other pests (toxaphene, chlordane, DDT) and chemicals used in industry like PCBs (found in oils and transformers). It also includes dioxins which are industrial waste products and PAHs which are found in smoke.

When are contaminants a concern?

Normally, the amount of contaminants in the ground, air and water is very, very small. In some areas, if there has been a chemical spill or other source of local pollution, the amount of contaminants can be higher. Some of these chemicals, especially "Persistent Organic Pollutants" can move with the air currents (wind and clouds) from southern areas to the North and then get into the ground and water when it rains or snows. Another way they can travel north is in the bodies of ducks and geese when they migrate from the South in the spring.

If animals eat or drink these contaminants, they do not get rid of them easily, so they can build up in their bodies over time. Contaminants tend to build up in animal organs such as the kidney and liver, as well as in the fat. When larger animals eat these animals, then they also eat the contaminants that have built up in their bodies. This is the reason that older animals and larger animals that eat other animals (like large fish) often have more contaminants in their bodies.

Modern technologies allow most of these chemicals to be measured in very, very small amounts. If some contaminants are found in animals in small amounts, it does not necessarily mean that they are causing harm to the animals or that the animals are not safe to eat. These contaminants may cause harm to animals or people only if they are present in very large amounts. Just like breathing in a small amount of smoke from a cigarette every once in a while will not hurt you but if you

smoke many cigarettes every day for many, many years, it can be harmful to your health.

The federal government (Health Canada) has recommended guidelines for the amount of certain contaminants that is safe in foods. These guidelines are made to help protect the health of people and the environment. The guidelines reflect what MAY be harmful if that amount is eaten EVERYDAY for a LIFETIME. The guidelines come from studies on animals like rats or monkeys in research laboratories or from cases of accidental exposure to high levels of contaminants in people. These guidelines are very cautious and include a large safety factor in their number (usually of 100 times or more).

In general, if the contaminant levels in food are below the guidelines, it means that government scientists feel that it is safe to eat the food as you normally would. If the levels are at or slightly above the guidelines, there is a lot of uncertainty over whether or not eating them may be harmful.

Unfortunately, there is still a lot of work that needs to be done to understand how much of these contaminants is safe and how much may not be safe. These guidelines are the scientists' best guess about safe levels of contaminants in foods so that is what we have used here to compare to the animals that were sampled in this study.

What was found in the animals in this study?

In many of the samples, either very, very small amounts of contaminants or no contaminants were found.

Some of the samples had levels of certain contaminants that were above the recommended guidelines.

- **Caribou** kidneys were above the guideline levels for mercury and cadmium.

- In the **goose**, some of the pesticide levels in its fat were above the guidelines but in the muscle and liver the levels were well below the guidelines.

- In the **scoters**, the levels of some pesticides (toxaphene and chlordane), PCBs and dioxins were well above the guidelines in the fat but were very low in the muscle and liver samples except for dioxins which were close to guideline in the muscle. The level of mercury in the liver of the black scoter was slightly above the guideline level.

- In the **fish**, all the contaminants were below the guidelines except for dioxins in the fat of one **pike**, which was slightly above the recommended guideline.

- The levels of all contaminants measured in the **muskrat, porcupine** and **spruce partridge** were well below the guidelines.

- The **porcupine** was found dead and seemed to have died from starvation during the winter.

What do these findings mean?

The levels of contaminants found were not high enough to be harmful to the animals. Because only a few animals were looked at, it is impossible to tell if the contaminants that were found in some of the samples would also be found in other animals in Nitassinan.

If the level of a contaminant in a sample is near or over the guideline - it **does not mean that you shouldn't eat those kinds of animals**. It means that there needs to be more information collected about those country foods. In order to be confident about the information collected, many animals have to be looked at. When only a few samples are looked at, they may not show what all of the animals are like.

Eating country foods is healthy for many different reasons. In most places across the North, there is no reason to stop eating them or to change how much they are eaten because of contaminants.

To find out more about these contaminants in the animals of Nitassinan, it is important to collect more information. After more animals are looked at, the information about these chemicals in the animals has to be considered along with what is known about the many

benefits of eating country foods.

Once all of the information gathered is put together, it can help people to make their own choices about eating country foods.

TECHNICAL SUMMARY OF RESULTS

Innu Nation Country Food Contaminant Pilot Study - 1997

The levels of contaminants in the animals tested were compared to the guideline levels as set by Health Canada (CACAR 1997). Where guideline levels are not reported, but Health Canada Acceptable Daily Intake (ADI) levels are reported, the values were compared to the ADI (assuming consumption of 100 g of that tissue and 60 kg person). In general, the levels for ADI are much more conservative than the more general guidelines as they assume daily intake of that level over a lifetime.

For this summary, the samples tested are reported as the percentage of the ADI (assuming consumption of 100 g of that tissue and 60 kg person and no other source of exposure to the contaminant). "Low" levels refer to <40% of the ADI, "medium" 40-80% the ADI and "high" refer to >80% of the ADI. This method is

used only to standardize the use of the terminology "low, medium, and high" in reference to contaminant levels in wildlife tissues. It is not in any way intended as a consumption recommendation. The value of ADIs, how they are calculated, and their use by health professionals as guidelines for consumption of traditional foods is a debated topic (Furgal et. al.1999).

Definitions for contaminants and technical terms are given at the end of the report.

FISH:

metals - No lead or arsenic was detected in any of the samples. In all tissues tested, levels of mercury were below the guideline levels for total mercury as set by Health Canada for edible portions of fish (0.5 ppm).

O/C pesticides - *data not available*

PCBs - In all tissues tested, the levels of PCBs were well below the guideline levels for fish set by Health Canada (2 ppm).

Dioxins/furans, co-planar PCBs - These compounds were measured only in the largest pike. The total TEQ in the fat is slightly higher than the recommended Health Canada guideline (17.9 ppt - guideline 15 ppt).

PAHs - In all tissues tested, PAHs were not detected or were in very low concentrations, well below the guideline levels.

SPRUCE GROUSE: 3 birds that were collected were combined into one sample

metals - No lead, mercury or arsenic was detected in any of the tissues. The level of cadmium measured was in the normal range for birds (Furness 1996)

O/C pesticides - All of the O/C pesticides measured were below 40% of the ADI and classified as "low".

PCBs - The levels of PCBs were well below the guideline levels for poultry consumption set by Health Canada (0.5 ppm on a fat basis).

Dioxins/furans, co-planar PCBs - These compounds were not measured.

PAHs - PAHs were not detected in the tissues tested.

SCOTERS: Two surf scoters and one black scoter were analysed.

metals - No lead or arsenic was detected in any of the samples. The level of cadmium measured was in the normal range for birds of this type (Furness 1996). Mercury levels in the surf scoter muscle and liver were

"low" (16.4% ADI) or not detected. In the black scoter, the liver mercury levels exceeded the ADI (167%)

O/C pesticides - Toxaphene in the fat of the surf scoters surpassed the ADI (204 and 205 %) and in the black scoter, the levels were well below the ADI or "low". Levels in muscle and liver in all 3 ducks were "low".

Levels of chlordane in fat in all 3 birds were well above the ADI (116 - 320 %). The levels of chlordane in the liver and muscle were in the "low" range.

HCHs in all tissues were "low" (<7.1% ADI).

DDT in all tissues were also "low" (<6.4% ADI).

PCBs - The levels of PCBs in fat were well above the Health Canada guideline levels for poultry consumption (0.5 ppm on a fat basis) at 1.193 ppm wet weight in the black scoter and 2.204 and 2.368 ppm wet weight in each of the two surf scoters. The levels of PCBs measured in the muscle in all 3 ducks were well below the Health Canada guidelines.

Dioxins/furans, co-planar PCBs -These compounds were measured only in the fat of one surf scoter and the muscle of the other. The total TEQ is greatly in excess of the ADI in the fat (3217%) and in the muscle is "high" (90% ADI). Most of the TEQ is contributed by the dioxin-like PCBs (140 ppt, vs 53 ppt for dioxins/furans).

PAHs - PAHs were not detected in the tissues tested.

CANADA GOOSE:

metals - No lead, arsenic or mercury was detected in any of the samples. The level of cadmium measured was in the normal range for birds of this type (Furness 1996).

O/C pesticides - Toxaphene in the fat tissue was 87.5% of the ADI or "high". The levels in the liver and muscle were "low" (≤33% ADI).

Levels of chlordane in fat were well above the ADI (332%). The levels of chlordane in the liver and muscle were "low" (≤ 36%ADI).

HCHs were not detected in any tissue tested.

Levels of DDT in fat (1.25 µg/g) exceeded the Health Canada guideline of 1 µg/g on a fat basis.

PCBs - The levels of PCBs in all tissues were well below the Health Canada guideline levels for poultry consumption (0.5 ppm on a fat basis).

Dioxins/furans, co-planar PCBs -These compounds were measured only in the fat tissue. The total TEQ was in excess of the ADI (123%). Most of the TEQ was contributed by the dioxin-like PCBs (4.8 ppt vs 2.6 ppt for dioxins/furans).

PAHs - PAHs were not detected in the tissues tested.

MUSKRAT:

metals - No lead, cadmium, arsenic or mercury was detected in any of the samples.

O/C pesticides - Toxaphene, chlordane, HCHs and DDT levels in all tissues measured were "low" (≤15.7% ADI).

PCBs - The levels of PCBs in all tissues were well below the Health Canada guideline levels for beef consumption (0.2 ppm on a fat basis).

Dioxins/furans, co-planar PCBs - These compounds were measured only in the fat tissue. The total TEQ was "low" (38.7% ADI). Most of the TEQ was contributed by the PCDD/Fs (1.4 ppt vs 0.82 ppt for dioxins/furans).

PAHs - PAHs were not detected in the tissues sampled.

PORCUPINE:

metals - No lead, cadmium, arsenic or mercury was detected in any of the samples.

O/C pesticides - Toxaphene was not detected in the fat or muscle.

The levels of chlordane were "low" in the fat (10.0% ADI) and not detected in the muscle.

HCHs and DDT levels in fat and muscle were well below the ADI or "low" (≤0.1%).

PCBs - The levels of PCBs in all tissues were well below the Health Canada guideline levels for beef consumption (0.2 ppm on a fat basis).

Dioxins/furans, co-planar PCBs -These compounds were not measured.

PAHs - PAHs were not detected in the tissues tested.

CARIBOU: 9 caribou were sampled ranging in age from 1 to 13 years.

metals - No arsenic was detected in any of the tissues tested.

Mercury levels in kidneys were above the ADI in all samples (110-422%). In muscle, the levels were not detected or were very low.

Cadmium was not detected in any of the muscle samples. The levels in the kidneys were well in excess of the ADI (280-4766%).

Lead was detected in only 2 of the kidney samples and was found in very low levels.

O/C pesticides - Toxaphene was not detected in the samples tested.

Chlordane levels were "low" (≤1.3%ADI).

HCH levels were "low" (≤2.5%ADI).

DDT levels in all tissues measured were "low" (0% ADI).

PCBs - The levels of PCBs in all tissues were well below the Health Canada guideline levels for beef consumption (0.2 ppm on a fat basis).

Dioxins/furans, co-planar PCBs -These compounds were not measured.

PAHs - PAHs were not detected in the tissues sampled.

BIOLOGICAL EFFECTS:

For all contaminants measured, the levels were below the reported thresholds for biological effects in animals. Based on the information available, there does not appear to be documented health risks to the animals due to contaminants at the levels measured. (Braune et al. 1999)

DEFINITIONS

Acceptable Daily Intake (ADI) - The daily intake of a substance from all sources during a person's entire lifetime that should not cause an appreciable risk to health on the basis of all known facts. It is usually calculated from toxicity experiments with laboratory animals.

Bioaccumulation - Some contaminants are excreted more slowly than they are absorbed and are stored in the body for long periods of time. The total amount of contaminant in the body may increase over time if an animal is continually exposed to a bioaccumulating contaminant.

Biomagnification - Contaminants "biomagnify" when their concentration increases at each level of the food chain. As smaller animals are eaten by larger animals, they also consume their contaminant burdens. This results in higher concentrations of contaminants in top predators.

Cadmium - Cadmium is a heavy metal and enters the environment both through natural processes and human activity (mining and smelting). Tobacco smoke is a major route of exposure to cadmium in people. Cadmium can be found in the terrestrial, freshwater and marine environments. Cadmium tends to bioaccumulate in organisms but does not biomagnify up the food chain.

Chlordane - Chlordanes are o/c pesticides that are not currently used in Canada and enter the Northern ecosystem through long-range airborne transport.

Contaminant - Substances that are foreign to a natural system or present in unnatural concentrations. Some contaminants are created by human activity, while others are the result of natural processes. Contaminants are not necessarily harmful to the environment or human or animal health.

Coplanar PCBs - Thirteen of the 209 PCBs (eg. PCB-77, 126, 169) have been shown to share certain toxicological properties with dioxins due to their chemical configuration and have been assigned TEQs. These congeners are often included in the consideration of total TEQ of a sample along with dioxins and furans.

DDT (dichlorociphenyltrichloroethane) - The use of DDT was stopped in Canada in 1989. It is an o/c pesticide and is still used in some countries for malaria control. It is very persistent in the environment and can be found throughout northen Canada. Most of the DDT found in the North is from long range transport through the atmosphere. However, there are suggestions that it was regularly applied at several locations in Labrador by the US military, including Goose Bay and Minipi Lake.

Dioxins and furans (PCDD/PCDF) - These two chemical families are closely related. There are 75 dioxins and 135 furans. The most toxic is 2,3,7,8-tetrachlorodibenzodioxin (2,3,7,8-TCDD). Although produced by natural sources (forest fires, volcanoes), the majority of dioxins are by-products of certain chemicals or chemical processes (chlorine bleaching used in pulp and paper mills) or combustion (incineration). Small sources of dioxins include wood burning, cigarette smoke, barbecuing. Furans are a trace contaminant of PCBs.

HCH (hexachlorocyclohexanes) - HCHs are a group of pesticides that are still in large scale use throughout the world today. The most commonly known and the most toxic form of HCHs is Lindane which is still registered for use in Canada.

Heavy metals - The metals of most concern as contaminants in the North are cadmium, lead and mercury. They have both natural and human sources. In large amounts, they can be harmful to animals and people.

Lead - Most of the lead found in the environment is due to human activity. Some sources include the improper disposal of batteries, ammunition, mining, smelting, ceramic glazing. Plants and animals can bioaccumulate lead but it does not tend to biomagnify up the chain.

Mercury - Mercury is a metal and exists in 3 forms: organic, inorganic and elemental. It is present in the air, water and soil both from natural sources and as a result of human activity. Elevated levels of mercury are found in flooded areas like reservoirs built for hydroelectric power generation. The organic form of mercury (methylmercury) is the most toxic. Methylmercury can bioaccumulate and biomagnify up the food chain.

Organochlorine (O/C) pesticides - These compounds do not occur naturally and are persistent in the environment. They include chlordane, toxaphene, HCH, DDT. They are fat soluble and bioaccumulate in living organisms and can biomagnifiy up the food chain. Many can be transported long distances through the atmosphere.

"Persistent Organic Pollutants" (POPs) - A group of contaminants that stay in the environment for a long time. Organic compounds contain carbon, usually combined with hydrogen and other elements such as chlorine. POPs include organochlorine pesticides such as DDT and chlordane as well as PCBs, PAHs and dioxins and furans. The majority of POPs in the North originate from distant sources. In large amounts, they are potentially harmful to living organisms.

Pollutant - A pollutant is a contaminant that is present at concentrations that are harmful to the environment or to human or animal health.

Polychlorinated Biphenyls (PCBs) - PCBs are a family of 209 different compounds and they do not occur naturally. Their many uses include: in capacitors, transformers, hydraulic fluids, adhesives, lubricants, flame retarders. PCBs have been banned from most uses in North America and are restricted for others. They are stable and persistent in the environment and are fat soluble and can bioaccumulate and biomagnify up the food chains.

Polycyclic Aromatic Hydrocarbons (PAHs) - Natural sources of PAHs include forest fires, volcanoes and fossil fuels. Sources from human activity include the incomplete combustion of fossil fuels, woods and garbage; cigarette smoke and vehicle exhaust. PAHs do not tend to accumulate in animals as most animals can rapidly eliminate it from their systems. Common PAHs include benzo[a]pyrene.

Toxaphene - Toxaphene is an o/c insecticide that was extensively used in southern countries and is still used in some areas. It is found in northern areas due to long-range transport and has been found in high levels in fish in the Yukon and Northwest Territories. It is a complex mixture of 670 chemicals.

Toxic equivalency factors (TEQs) - These numbers are assigned to individual dioxins and furans on the basis of how toxic they are in comparison to 2,3,7,8-TCDD which has been assigned a TEQ of 1. These numbers are used to express the toxicity of these compounds on a common basis.

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Appendix D. Copy of harvest survey

Innu Nation Harvest Survey

Introduction

A small study of several birds, animals and fish was conducted by Innu Nation staff and scientists at three Innu hunting camps during the spring and fall of 1997. Small pieces of muscle, fat, liver and kidney were taken and tested for certain harmful chemicals. While some chemicals were found in a few of the samples, the small number of animals examined does not allow us to make any conclusions or recommendations. However, the results did show that additional studies are needed.

Innu Nation and its research partners are now conducting a detailed three year study to better understand the levels of chemicals in Innu country food. Once we have enough information, we hope to be able to decide what should be done about the levels of chemicals in the animals.

Part of the study is the hunting diary that every Innu person over the age of 13 should have. Everyone should be recording everything that they hunt for a whole year, beginning from September 1999. This Harvesting Survey is also a part of the study and will help the research team find out more about the animals that are killed through the year. Innu Nation researchers in each community will be doing interviews with hunters every few months. Once we have the results of the interviews, and the research team has consulted with Innu elders, hunters and other experts, we will choose four groups of animals for detailed study. From this study, we hope to find out about any risks that the chemicals might pose to human health, and to inform Innu about where these chemicals might be in the animals that they kill and eat.

Your participation in this study is very important and any personal information collected during the study will be treated as strictly confidential by the researchers. Our objective is to keep people informed about the work we are doing, and to give people a chance to have their questions answered.

Date _____ Name of Interviewer _____ Name of Responder _____

Community _____ Interview # _____

1. Did you go hunting between September 1, 1999 and January 31, 2000? *(circle one)*

YES *(if YES go to question2)*

NO *(if NO, finish interview)*

2. How many times did you go hunting? *(circle one answer)*

once

twice

once a month

once every two weeks

once a week

twice a week

more than twice a week

other _____

3. Did you kill any land animals (for example: caribou, hare, beaver, porcupine) during this time? *(circle one)*

YES *(if YES, go to question 4)*

NO *(if NO, go to question 5)*

4. How many of the following animals did you kill yourself? *(write the number if it is known, or circle the closest range of numbers if they are not sure exactly how many)*

Caribou	atiku	**number**_____	1-9	10-19	20-29	30-39	40-49	50-59	60+
Arctic Hare	mishtapush/mishtapushu	**number**_____	1-9	10-19	20-29	30-39	40-49	50-59	60+
Snowshoe Hare	uapush	**number**_____	1-9	10-19	20-29	30-39	40-49	50-59	60+
Porcupine	kaku	**number**_____	1-9	10-19	20-29	30-39	40-49	50-59	60+
Muskrat	utshashku/uatshashku	**number**_____	1-9	10-19	20-29	30-39	40-49	50-59	60+
Red Fox	uishuau-atsheshu	**number**_____	1-9	10-19	20-29	30-39	40-49	50-59	60+
Arctic Fox	uapatsheshu	**number**_____	1-4	5-9	10-14	15-19	20-24	25-29	30+
Black Bear	mashku	**number**_____	1-4	5-9	10-14	15-19	20-24	25-29	30+
Moose	mush	**number**_____	1-4	5-9	10-14	15-19	20-24	25-29	30+
Otter	nitshuku/nitshiku	**number**_____	1-4	5-9	10-14	15-19	20-24	25-29	30+
Beaver	amishku	**number**_____	1-4	5-9	10-14	15-19	20-24	25-29	30+
Martin	uapishtan	**number**_____	1-4	5-9	10-14	15-19	20-24	25-29	30+
Mink	atshikash	**number**_____	1-4	5-9	10-14	15-19	20-24	25-29	30+
Weasel (gen.)	shikush	**number**_____	1-4	5-9	10-14	15-19	20-24	25-29	30+
Polar Bear	uapashku	**number**_____	1-4	5-9	10-14	15-19	20-24	25-29	30+
Wolf	maikan/menike	**number**_____	1-4	5-9	10-14	15-19	20-24	25-29	30+
Lynx	pishu	**number**_____	1-4	5-9	10-14	15-19	20-24	25-29	30+
Other (Kutakuts)		**number**_____	1-4	5-9	10-14	15-19	20-24	25-29	30+
_____	_____								

5. Did you kill any animals that fly (for example: geese, ducks, grouse, ptarmigan), during this time? *(circle one)*

YES *(if YES, go to question 6)*

NO *(if NO, go to question 7)*

6. How many of the following flying animals did you kill yourself? *(write the number if it is known, or circle the closest range of numbers if they are not sure how many)*

Ruffed Partridge	pashpashtshu	**number**_____	1-14	15-29	30-44	45-59	60-74	75-89	90+
Spruce Partridge	innineu	**number**_____	1-14	15-29	30-44	45-59	60-74	75-89	90+
Partridge (gen)	pineu	**number**_____	1-14	15-29	30-44	45-59	60-74	75-89	90+
Rock Ptarmigan	kashkanatshish/kashketshish	**number**_____	1-14	15-29	30-44	45-59	60-74	75-89	90+
Willow Ptarmigan	uapineu	**number**_____	1-14	15-29	30-44	45-59	60-74	75-89	90+
American Black Duck	innishipl/pashkuaship	**number**_____	1-14	15-29	30-44	45-59	60-74	75-89	90+
Blue-Winged Teal	amishkuniship	**number**_____	1-14	15-29	30-44	45-59	60-74	75-89	90+
Green-winged Teal	amishkunniship/nishipiss	**number**_____	1-14	15-29	30-44	45-59	60-74	75-89	90+
Red-throated Loon	ashumuaku/kashakat	**number**_____	1-14	15-29	30-44	45-59	60-74	75-89	90+
Loon (gen.)	muaku	**number**_____	1-14	15-29	30-44	45-59	60-74	75-89	90+
Canada Goose	nishk	**number**_____	1-14	15-29	30-44	45-59	60-74	75-89	90+
Oldsquaw duck	ahaueu/aueu	**number**_____	1-14	15-29	30-44	45-59	60-74	75-89	90+
Black Scoter	shashteship	**number**_____	1-14	15-29	30-44	45-59	60-74	75-89	90+
White Winged Scoter	mamuku	**number**_____	1-14	15-29	30-44	45-59	60-74	75-89	90+
Surf Scoter	mitshikuatan, papukutshat	**number**_____	1-14	15-29	30-44	45-59	60-74	75-89	90+
Scoter (gen.)	umamuku/matshakutakai	**number**_____	1-14	15-29	30-44	45-59	60-74	75-89	90+
King Eider	utshimau-shiship	**number**_____	1-14	15-29	30-44	45-59	60-74	75-89	90+
Common Eider	mishta-missip	**number**_____	1-14	15-29	30-44	45-59	60-74	75-89	90+
Eider Duck (gen)	missip/passipats	**number**_____	1-14	15-29	30-44	45-59	60-74	75-89	90+
Harlequin Duck	nutshipaushtukueshish	**number**_____	1-14	15-29	30-44	45-59	60-74	75-89	90+
Ducks (gen)	shiship	**number**_____	1-14	15-29	30-44	45-59	60-74	75-89	90+
Greater Scaup	peptshukuteu	**number**_____	1-9	10-19	20-29	30-39	40-49	50-59	60+
Lesser Scaup	napeship	**number**_____	1-9	10-19	20-29	30-39	40-49	50-59	60+
Scaups (gen)	kaishinikanukutesht	**number**_____	1-9	10-19	20-29	30-39	40-49	50-59	60+

Red-breasted Merganser	mishtishuku	**number**_____	1-9	10-19	20-29	30-39	40-49	50-59	60+
Merganser (gen.)	ushuku/katshinukutesht	**number**_____	1-9	10-19	20-29	30-39	40-49	50-59	60+
Snowy Owl	uapakanu	**number**_____	1-9	10-19	20-29	30-39	40-49	50-59	60+
Owl (gen)	uhu	**number**_____	1-9	10-19	20-29	30-39	40-49	50-59	60+
Atlantic Puffin	tshetshukuteshu	**number**_____	1-9	10-19	20-29	30-39	40-49	50-59	60+
Black Guillemot	katashishipat/shikauniss	**number**_____	1-9	10-19	20-29	30-39	40-49	50-59	60+
Common Murre	ketshinukuteu	**number**_____	1-9	10-19	20-29	30-39	40-49	50-59	60+
Thick-billed Murre	innukut	**number**_____	1-9	10-19	20-29	30-39	40-49	50-59	60+
Dovekie	tshumushumash	**number**_____	1-9	10-19	20-29	30-39	40-49	50-59	60+
Cormorants (gen)	kakatshiship	**number**_____	1-9	10-19	20-29	30-39	40-49	50-59	60+
Gulls (gen)	tshiashku/tshinashku	**number**_____	1-9	10-19	20-29	30-39	40-49	50-59	60+
Terns (gen)	tshinash/apishtshinashkuish	**number**_____	1-9	10-19	20-29	30-39	40-49	50-59	60+
Northern Pintail	uapinniship	**number**_____	1-9	10-19	20-29	30-39	40-49	50-59	60+
Barrow's Goldeneye	mishikushuku	**number**_____	1-9	10-19	20-29	30-39	40-49	50-59	60+
Common Goldeneye	mishikushku	**number**_____	1-9	10-19	20-29	30-39	40-49	50-59	60+
Other	(Kutakuts)	**number**_____	1-9	10-19	20-29	30-39	40-49	50-59	60+
Other	(Kutakuts)	**number**_____	1-9	10-19	20-29	30-39	40-49	50-59	60+

7. Did you kill fish, (for example: char, trout, salmon , seal, cod) during this time?*(circle one)*

YES *(if YES, go to question 8)*

NO *(if NO, finish interview)*

8. How many of the following kinds of fish did you kill yourself? (write the number if it is known, or circle the closest range of numbers if they are not sure how many)

Arctic char (landlock)	memishkushkiteu	**number**_____	1-14	15-29	30-44	45-59	60-74	75-89	90+
Arctic char (gen)	shushashu	**number**_____	1-14	15-29	30-44	45-59	60-74	75-89	90+
Lake trout	kukamess	**number**_____	1-9	10-19	20-29	30-39	40-49	50-59	60+
Brook trout (sea run)	uinipeku-matameku	**number**_____	1-14	15-29	30-44	45-59	60-74	75-89	90+
Brook trout	matameku	**number**_____	1-14	15-29	30-44	45-59	60-74	75-89	90+
Atlantic salmon	utshashumeku	**number**_____	1-9	10-19	20-29	30-39	40-49	50-59	60+
Lake whitefish	atikameku	**number**_____	1-9	10-19	20-29	30-39	40-49	50-59	60+
Northern pike	tshinusheu	**number**_____	1-9	10-19	20-29	30-39	40-49	50-59	60+
Northern cod	unushui/ueushu	**number**_____	1-9	10-19	20-29	30-39	40-49	50-59	60+
Rockcod	tamakat	**number**_____	1-9	10-19	20-29	30-39	40-49	50-59	60+
Rainbow Smelt	kauapishiss	**number**_____	1-9	10-19	20-29	30-39	40-49	50-59	60+
Longnose sucker	mikuashai	**number**_____	1-9	10-19	20-29	30-39	40-49	50-59	60+
White sucker	makatsheu	**number**_____	1-9	10-19	20-29	30-39	40-49	50-59	60+
Lobster (gen.)	ashatsheu	**number**_____	1-9	10-19	20-29	30-39	40-49	50-59	60+
Crabs (gen.)	pemituteu	**number**_____	1-9	10-19	20-29	30-39	40-49	50-59	60+
Shellfish (gen.)	esh	**number**_____	1-9	10-19	20-29	30-39	40-49	50-59	60+
American eel	upimishui/pemituteu	**number**_____	1-9	10-19	20-29	30-39	40-49	50-59	60+
Ouananiche	uanan	**number**_____	1-9	10-19	20-29	30-39	40-49	50-59	60+
Stickleback (gen.)	kaushkentitshekenshit	**number**_____	1-9	10-19	20-29	30-39	40-49	50-59	60+
Burbot	minai	**number**_____	1-9	10-19	20-29	30-39	40-49	50-59	60+
Capelin	kashkanamekush	**number**_____	1-9	10-19	20-29	30-39	40-49	50-59	60+
Atlantic herring	kaushkanusht	**number**_____	1-9	10-19	20-29	30-39	40-49	50-59	60+
Sculpin (gen.)	mishtakuakai	**number**_____	1-9	10-19	20-29	30-39	40-49	50-59	60+
Seals (gen)	atshuku	**number**_____	1-4	5-9	10-14	15-19	20-24	25-29	30+
Whale (gen.)	mishtameku	**number**_____	1-4	5-9	10-14	15-19	20-24	25-29	30+

Dolphins (gen.)	uapimeku	**number**____	1-4	5-9	10-14	15-19	20-24	25-29	30+
Other	(Kutakuts)	**number**____	1-4	5-9	10-14	15-19	20-24	25-29	30+

Thank you for your time and participation in the study. The results of all of the surveys together will be presented to the communities for review and discussion in the fall of 2000

Comments: _____

Appendix E. Necropsy reports of animals submitted to the AVC during the study period.

Necropsy Report — Harvest and Country Food Contaminant Study -Innu Nation, Environment Canada and the Atlantic Veterinary College

Black Bear case#X-23911-00 January 29, 2001

Copies to Louie and Mary Adele Penashue, Sheshatshit and Basile Penashue, Innu Nation

History

The bear had been hanging around the cabin of Louie and Mary Adele Penashue on the Churchill Road. it had eaten some food near the cabin and had returned over a period of days. It was aggressive and threatening the people at the cabin. In their experience, this behavior was unusual and they had not had a similar problem with a bear in the country before. They were wondering if something was wrong with it to make it act this way or if it was thin and hungry. Because of aggressiveness, it was shot on July 17, 2000. It was gutted (stomach, intestines and liver removed) at the cabin and the carcass brought to Sheshatshiu on the same day. it was then frozen at the North West River Provincial Wildlife Office (Kirk Greening 497-8479) to be shipped to AVC or necropsied in Sheshatshiu at a later date. It was decided to necropsy the bear on site in Labrador because of the difficulties associated with shipping it to AVC. It was taken out of the freezer on October 3 and transported on October 6 to the cabin where it had been killed. Gross examination was performed as Louie and Mary Adele Penashue skinned and cut up the bear.

Mary Adele Penashue felt that the bear was not as fat as should normally be expected at this time of the year. There were worms found in the abdomen and the intestines, Mary Adele feels this is normal as most healthy animals have them.

COMMENTS

This bear has fair fat reserves, which is less than expected at this time of year. There does not appear to be any disease affecting this bear that may have caused the agressive behaviour. The results of the toxicology tests are normal (cadmium, lead, selenium and mercury).

Thank you for allowing us to examine this bear. If there are any questions about this report, please do not hesitate to contact Basile or Beth Pollock at the Atlantic Veterinary College (902) 566-0639.

Beth Pollock, DVM/ Pierre-Yves Daoust, Wildlife Pathologist

Details of necropsy examination and test results

Gross examination

Necropsy, October 7, 2000. This is a young adult female black bear which is cleaned (liver, stomach and intestines removed). It is in fair body condition (fat stores: coronary groove 2+; subcutis 2+; kidney 3+). It had been shot in the head and there was some associated hemorrhage at the base of the calvarium. A small number of nematodes (approximately 10 cm in length) were found free in the abdomen or embedded in the peri-renal fat. They were collected and fixed in 10% formalin.

Histology

Samples of kidney, lung, heart and brain were examined. No lesion can be seen in any of these organs.

Toxicology (all values are reported on a wet weight basis)

kidney - lead	**0.13 ppm (normal range 0.2 - 2.0 ppm)**
- selenium	**1.80 ppm (normal range 1.00-1.70 ppm)**
- cadmium	**13.3 ppm (normal range 0.4 - 9.3; toxic threshold is 100 ppm)**
- total mercury	**0.36 ppm (normal range 0.02 - 0.83 ppm)**

(Toxic threshold levels from Environmental Contaminants in Wildlife - interpreting tissue concentrations. 1996. SETAC special publication)

Diagnosis

No abnormal finding

COMMENT

Nothing could be found on microscopic examination of the brain, heart, lungs and kidneys of this bear which could explain its apparent aggressive behavior. The fact that this animal was only in fair body condition, at a time of year when it should have gotten ready for winter, may have had a lot to do with this behavior.

Addendum (3 October, 2003): Subsequent cementum analysis of a premolar tooth estimated the age at 11 years.

Results of organochlorine and PCB analysis of fat tissue conducted on May 21, 2002:

All OC pesticides were below detection limits except for HCB (2.4 ng/g ww) which was near the detection limit.

All PCB congeners were below detection limits except for PCB 153 and 180.

sumPCB concentration 8.3 ng/g ww.

The organochlorine concentrations are very low and can be considered to be background levels.

Necropsy Report - Harvest and Country Food Contaminant Study - Innu Nation, Environment Canada and Atlantic Veterinary College

Canada goose, American black duck, 2 unknown ducks case #A-16976-00 September 29, 2000. Copies to Simeon Rich, Sheshatshiu and Basile Penashue, Innu Nation.

History

Received on July 19, 2000 from Simeon Rich, Sheshatshiu Labrador. Shot in June 2000 near the Pinus River along the Churchill road. The goose is plucked and cleaned. Simeon is concerned that the goose appears thin and does not feel that it is safe to eat. The black duck still has its feathers and is not cleaned. It was shot near the same location as the goose. Two unidentified, plucked and cleaned ducks and a goose gizzard were packaged with the goose.

COMMENTS

The Canada Goose is very thin. It has an old injury to its back with a small fracture (broken bone) and a local infection. We do not know what caused this injury. The goose is probably thin because of the injury to its back combined with the spring migration. The results from the toxicology tests are very low.

The black duck has good fat reserves. This duck appears healthy. The results from the toxicology tests are low. Some of the contaminants (DDT, pesticides and PCBs) that were tested for were not found in the samples.

The 2 unknown ducks appear healthy.

Thank you for giving us these birds to examine. If there are any questions about this report, please do not hesitate to contact Basile Penashue at the Innu Nation (497-8398) or Beth Pollock at the Atlantic Veterinary College (902)566-0639.

Beth Pollock, DVM

Details of necropsy examination and test results

Gross examination

01 Canada goose - This is an immature female Canada goose which is plucked and cleaned (wings, legs, head, intestines, liver removed) weighing 1489 g. It is in poor body condition (no fat around heart, gizzard, minimal amount in the subcutis, moderate atrophy of the pectoral muscles). There are multiple gunshot wounds throughout the body. There is a chronic fracture of the left pubic bone in two places with minimal associated bony callus. Adjacent to the most proximal pubic fracture, between the ischium and the pubic bone, encapsulated in the soft tissue there is a 3 cm x 1 cm sequestrum of brown, crumbly material. The adrenal glands appear moderately enlarged.

04 American black duck - This is an adult male (enlarged testicles) American black duck which is intact and feathered weighing 1238 g. It is in good body condition (extensive fat stores in the coronary groove, surrounding the gizzard and in the subcutis). There are multiple gunshot wounds in the cranial half of the body and the neck is severed midway between the head and body. The syrinx is preserved in formalin.

02 unknown duck - This is an adult male (enlarged testicles) unidentified duck which is plucked and cleaned (wings, legs, head, intestines, liver, gizzard removed) weighing 513 g. It is in poor body condition (minimal fat stores in coronary groove and in subcutis, mild atrophy of the pectoral muscles). The lungs are severely hemorrhagic.

03 unknown duck - This is an adult male (enlarged testicles) unidentified duck which is plucked and cleaned (wings, legs, head, intestines, gizzard removed) weighing 839 g. It is in moderate body condition (moderate fat stores in coronary groove and in subcutis, no atrophy of the pectoral muscles). There are multiple gunshot wounds throughout the body. The syrinx is preserved in formalin. The syrinx appears similar to that of the American black duck.

Bacteriology

01 Canada goose - Samples of kidney and lung were submitted for aerobic culture.

Lung - no microbial growth

Kidney - small numbers of alpha hemolytic *Streptococcus* and *Staphylococcus* sp. and few *Bacillus* sp. were cultured.

Histology

01 Canada goose - Samples of lung, heart, kidney, adrenal gland, pectoral and leg muscle, ovary and sequestrum were examined. The sequestrum is encapsulated by fibrous connective tissue. Many giant cells are also present surrounding the foreign material. Much of this material is birefringent under polarized light, but its nature cannot be determined.

04 American black duck - Samples of brain, lung, heart, adrenal glands, liver, kidney, duodenum, pancreas, cecum, testicle were examined. Sections of kidney show several small aggregates of mononuclear leukocytes (mainly lymphocytes?) in the interstitium. The spleen contains unusually prominent lymphoid follicles.

02 unknown duck - Samples of lung, heart, kidney, adrenal gland, pectoral and leg muscle were examined and no abnormal findings were noted. A large proportion of the cells of the adrenal cortical tissue showed moderate fatty change.

03 unknown duck - Samples of lung, heart, kidney, pectoral and leg muscle, testicle, liver were examined and no abnormal findings were noted.

Toxicology(all values are reported on a wet weight basis)

01 Canada goose -

kidney - lead 0.14 ppm (<2 ppm is considered background level in healthy birds)

- selenium 1.65 ppm (toxic threshold is >10 ppm)

- cadmium <0.5 ppm (toxic threshold is 100 ppm)

- total mercury 0.02 ppm (toxic threshold is 20-30 ppm)

muscle - DDT <.0001 ppm

- Toxaphene <0.1 ppm

- Total PCBs <0.01 ppm

04 American black duck-

kidney - lead 0.42 ppm (<2 ppm is considered background level in healthy birds)

- selenium 3.09 ppm (toxic threshold is >10 ppm)

- cadmium 1.29 ppm (toxic threshold is 100 ppm)

- total mercury 1.3 ppm (toxic threshold is 20-30 ppm)

muscle - DDT <.0001 ppm

- Toxaphene <0.1 ppm

- Total PCBs <0.01 ppm

(toxic threshold levels from Environmental Contaminants in Wildlife - interpreting tissue concentrations. 1996. SETAC special publication)

Diagnoses

01 Canada Goose -

Focal trauma, chronic (cellulitis and pubic fractures)

04 Black Duck -

None

02 Unknown duck -

None

03 Unknown duck -

None

COMMENTS

The Canada Goose was very thin and had an old injury to its back which included a small fracture and a local infection. The cause of this injury could not be determined. This injury combined with the recent spring migration likely contributed to the bird's poor body condition. The toxicology results show that the contaminant burden is minimal and did not contribute to the poor health of this goose.

The American black duck was on good body condition, as indicated by its extensive fat reserves. No significant abnormalities were found in this duck. The toxicology results show that the contaminant burden is minimal and insufficient to impair the health of this duck.

No significant abnormalities were found in the two unknown ducks.

Necropsy Report — Harvest and Country Food Contaminant Study

Innu Nation, Environment Canada and the Atlantic Veterinary College

Porcupine case#X- 19947-01-10 January 29, 2001

Copies to Kathleen Nuna, Sheshatshit and Basile Penashue, Innu Nation

History

The porcupine was killed by Terry Andrew on September 14, 2001 and was given to Kathleen Nuna. While she was cleaning it she noticed some white spots and moles in the inner wall of inside of it and it was skinny. Kathleen Nuna said that this time of the month now porcupine would be fat and healthy. This porcupine didn't seem alright and she couldn't eat it. The cleaned carcass and intestines were given to Basile Penashue who froze them and submitted them to the Atlantic Veterinary College for testing.

COMMENTS

This porcupine was very thin for the time of year it was killed. Although the person cleaning the porcupine (Kathleen Nuna) noticed some white spots and moles on the inner walls, I failed to look closely for them so samples were not taken of this area for microscopic examination. By examining the other tissues (brain, liver, muscle and kidneys), no cause for the thin condition of this animal or the spots described could be found.

The results of the tests for contaminants showed very low amounts of only a few contaminants in the liver, fat and kidney. These amounts were much lower than what can cause health problems in animals.

There were greater than average numbers of pinworms (parasites) found in the intestines. The high numbers of these parasites are likely related to the poor condition of the porcupine, but are unlikely to have caused it.

I deeply regret not examining this porcupine more closely, I agree completely with Kathleen Nuna that there was something wrong (seemed unhealthy) with this porcupine, but I could not find an explanation for the thin condition of the porcupine or the white spots and moles described.

Thank you for allowing us to examine this porcupine. If there are any questions about this report, please do not hesitate to contact Basile Penashue at the Innu Nation or Beth Pollock (709) 458-2061.

Beth Pollock, DVM in consultation with Pierre-Yves Daoust, Wildlife Pathologist

Details of necropsy examination and test results

Gross examination

Necropsy, October 11, 2001. This is a female porcupine of unknown age which is partially cleaned (stomach and intestines removed). It is in poor body condition (fat stores: subcutis 1+). Its fur and quills had been singed and scraped and the carcass had been partially cut into pieces.

The liver, kidneys, and uterus remained with the carcass. The intestines were submitted separately. The contents were harvested for parasite identification and quantification.

Histology

Samples of kidney, liver, muscle and brain were examined. No lesion can be seen in any of these organs. Samples of the body wall were not taken for histological examination.

Toxicology (all values are reported on a wet weight basis)

kidney - lead	0.31 ppm (no effect <1.3 ppm)
- selenium	0.18 ppm (normal range 1.00-1.70 ppm)
- cadmium	<0.4 ppm (toxic threshold is 100 ppm)
- total mercury	0.01 ppm (30 ppm lethal limit)

(Toxic threshold levels from Environmental Contaminants in Wildlife - interpreting tissue concentrations. 1996. SETAC special publication)

Fat and liver - tested for organochlorine pesticides, PCBs

fat was tested for dioxins and furans

Only the organochlorines alpha-BHC, heptachlor epoxide, and one PCB congener were detected in the liver sample. The organochlorine HCB and some dioxins were detected in the fat sample.

All of the contaminants found at detectable levels were at very low concentrations (<10 ng/g ww for OC pesticides and PCBs and <1 pg/g ww for dioxins and furans).

Parasitology

Large numbers of two types of parasites were found in the intestine: the nematode *Evaginuris evaginata* (a pinworm) and a cestode (tapeworm) that was not identified or counted.

There were 188800 pinworms counted (range for 27 porcupine collected from Labrador 14900 - 288800, geometric mean 89100).

Diagnosis

Open

COMMENT

This porcupine was very thin for the time of year it was killed. Although the person cleaning the porcupine (Kathleen Nuna) noticed some white spots and moles on the inner walls, I failed to look closely for them so samples were not taken of this area for microscopic examination. By examining the other tissues (brain, liver, muscle and kidneys), no cause for the thin condition of this animal or the spots described could be found.

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