Concentrations and Latitudinal Variations of PBDEs in First Nation Peoples of the James Bay Region

By

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Author's declaration for electronic submission of a thesis

I hereby declare that I am the sole author of this thesis. This is a true copy of the thesis, including any required final revisions, as accepted by my examiners.

I understand that my thesis may be made electronically available to the public.

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Dedication

To my loving wife and my family - without your continuous support I could never have achieved as much as I have. Thank you from the bottom of my heart.

Abstract

Concerns regarding the persistence, bioaccumulation, long-range transport, and adverse health effects of polybrominated dipheyl ethers (PBDEs) have recently come to light. PBDEs are quickly becoming a ubiquitous pollutant and have been found globally in sediment, aquatic mammals, fish, bird eggs, human milk, serum, and adipose tissue. Due to their persistence and lipophilicity, PBDEs may biomagnify through the food chain and could pose a danger to those who consume these contaminated animals.

Many First Nation communities in the James Bay region rely in part on a subsistence diet; therefore, these communities have the potential to carry high levels of PBDEs. Additionally, PBDEs have been shown to be preferentially deposited in the sub-Arctic, making this contaminant of particular interest to the James Bay region of northern Ontario and Quebec, Canada.

By analyzing PBDE body burden (plasma), this contaminant was characterized with regards to its latitudinal variation and concentration. The study established a baseline PBDE level in the communities of the James Bay region and determined concentrations in the traditional foods of the First Nation communities located in the Mushkegowuk Territory of sub-Arctic Canada, to examine potential routes of exposure.

The PBDE body burden was found to be low in the Mushkegowuk communities when compared to more northern communities in Canada. Similarly, PBDE body burden in the US and southern Canada is higher than those of the James Bay region. The body burdens of the James Bay Cree are similar to Japanese and Swedish levels. Analysis of traditional foods shows relatively elevated levels of PBDEs in certain game species such as whitefish (*Coregonus clupeaformis)*, mallard duck (*Anas platyrhynchos*), and moose (*Alces alces)* compared to other traditional foods from the same area.

Data in this thesis can be used for human biomonitoring purposes and the animal data can be combined with consumption data in order to assess the contribution of PBDEs to body burden from a First Nation diet as well as provide guidance when developing safety guidelines for the consumption of traditional foods. While the sub-arctic has been identified as an environmental sink for PBDEs, the levels found in this study indicate that long-range transportation and dietary biomagnification of the contaminant may not be the primary exposure pathway. This may be due to low bioavailability of the contaminant, rapid metabolism, or alternate sources of exposure and should be investigated in future studies.

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Abbreviations and Symbols

μg	Micrograms (10^{-6})
BDE	Brominated diphenyl ether
BFR	Brominated flame retardant
DDE	Dichlorodiphenyltrichloroethene
DDT	Dichlorodiphenyltrichloroethane
decaBDE	Decabrominated diphenyl ether
HBCD	Hexabromocyclododecane
hexaBDE	Hexabrominated diphenyl ether
heptaBDE	Heptabrominated diphenyl ether
IUPAC	International Union of Pure and Applied Chemistry
ng	Nanograms (10^{-9})
octaBDE	Octabrominated diphenyl ether
PBDDs	Polybrominated dibenzo-p-dioxins
PBDEs	Polybrominated diphenyl ethers
PBDFs	Polybrominated dibenzofurans
PCBs	Polychlorinated biphenyls
PCDDs	Polychlorinated dibenzo-p-dioxins
PCDFs	Polychlorinated dibenzofurans
pentaBDE	Pentabrominated diphenyl ether
POP	Persistent organic pollutant
ppb	Parts per billion
ppm	Parts per million
T3	3,3',5-triiodothyronine
T4	Thyroxine or 3,3',5,5'-tetraiodo-L-thyronine
TBBPA	Tetrabromobisphenol A
tetraBDE	Tetrabrominated diphenyl ether

Chapter 1 – Introduction

1.1 Rationale

Polybrominated diphenyl ethers (PBDEs), a flame retardant used in a wide variety of household products, are of growing concern when released to the environment because they are stored in fat, resist breakdown, and may be endocrine disruptors, much like other persistent organic pollutants (POPs), such as polychlorinated biphenyls (PCBs). Recently, levels of other POPs in human plasma (a component of blood) have decreased dramatically; however, this is not the case for PBDEs (Ryan 2005). The concentration of PBDEs in plasma and other biological fluids has steadily increased since their inception in the 1970s. As of 2003, PBDE production was estimated at 35 million kilograms per year (Mazdai 2003). This extensive production arises because PBDEs are one of the least expensive flame retardant commercially available (Mazdai 2003). However, in 2004, two of the three PBDE commercial mixtures were voluntarily phased out of production. This has resulted in an increase in the production of other brominated flame retardants such as TBBPA (BSEF 2007).

Data are limited concerning current levels of PBDEs in sub-arctic First Nation populations (Liberda et al. 2005); however global temporal trends of PBDEs are alarming. It would have possible that PBDEs would become the most prevalent organohalogen contaminant by 2050 if regulations were not implemented (Ikonomou 2002a). The global ubiquity of PBDEs indicates that a monitoring program should be developed and implemented. Such a monitoring program should not only incorporate human tissue sampling but also diet. This is especially true for First Nations groups who subsist on wild fish and game.

1.2 Research Question

What are the concentrations and latitudinal variations of PBDEs in the First Nation peoples of the James Bay Region? Are these contaminants posing a threat to a traditional subsistence diet?

1.3 Research Objectives

In order to answer the research questions, the objectives below have been developed:

i) Determine concentrations and latitudinal variation of PBDEs in First Nations peoples of the James Bay region.

ii) Establish baseline PBDE levels in the traditional meats of the western James Bay region of northern Ontario.

iii) Determine what species of wild game are of concern from a consumption standpoint.

1.4 Significance of Research

Many studies have examined PBDE concentrations in farmed fish (e.g., Easton et al. 2002; Hayward et al. 2007). However, few studies have examined PBDE levels in wild fish and game traditionally eaten by sub-arctic First Nation populations. This is the first comprehensive study that will characterize PBDEs in fish and game that are traditionally eaten by First Nation people of sub-arctic Canada.

The results generated by this study will be made available to help First Nation groups monitor both latitudinal and temporal changes of PBDE concentrations in wild meats and in their own communities. Sub-arctic First Nations groups across Canada will benefit from the study because they eat similar wild game and fish and no other data of this type exists. Further, no study has been conducted where traditional foods were examined for PBDEs in an effort to identify the exposure levels in First Nation groups and their food. Thus, the present study provides First Nation groups across the country with important tissue dependent PBDE contaminant data not available prior to the present study. This project examines the extent to which traditional food sources expose First Nation peoples to PBDEs. Further, it develops a baseline for temporal and latitudinal trends.

1.5 Historical Context & Current Situation

First produced in the 1970s, PBDEs are widely used as flame retardants in goods such as plastics, electronics, textiles, and construction materials (Mazdai, 2003). In recent years, PBDEs have become an environmental contaminant of concern because they are lipophilic and may be endocrine disruptors like other persistent organic pollutants such as DDT (dichlorodiphenyltrichloroethane), PCBs (polychlorinated biphenyls), polychlorinated dibenzo-p-dioxins (PCDDs) and furans (PCDFs) (Hooper and McDonald 2000; de Wit 2002; Hale et al. 2003). Levels of legacy POPs in humans have, in general, decreased (see Hooper and Mcdonald 2000). However, this is not the case for PBDEs. The concentration of PBDEs in plasma and other biological materials has steadily increased since their inception (e.g., Meironyte et al 1999). This is particularly important to First Nations (aboriginal) peoples in Canada because they typically carry larger loads of many POPs compared to the general population (AMAP 1998). Recently, organochlorine contaminants in several of the James Bay communities were found at levels much higher than the southern Canadian group (Tsuji et al. 2006). This has created concern regarding PBDE levels in these communities because the disproportionate contaminant load may arise from a traditional diet which relies heavily on fish and other foods high in POPs and possibly from activities involving traditional activities in the environment such as hunting and trapping (Arquette et al. 2002; Ohta et al. 2002; Dewailly 2005; Tsuji et al. 2006).

PBDEs are slowly released over the life of the plastic, foam and other components, for which they were meant to be used as safeguards from fires. This has resulted in global concentrations of PBDEs in human tissues ranging from approximately 1 to 400 ng/g lipid (McDonald 2002; Ryan and Patry 2000). PBDEs were first detected in the environment in 1972 and have increased significantly in people and in environmental media throughout the 1980s and 1990s; this trend of increasing levels of PBDEs is expected to continue (Kalantzi 2004). Noren and Meironyte (2000) showed an exponential increase in the sum of all PBDEs in human milk from 1972 to 1996. More recently, Akutus et al. (2003) showed an increase in PBDEs from 0.03 ng/g lipid to 1.19 ng/g lipid for the years of 1973 and 2000 respectively. Eslami et al. (2006) have shown levels to range from 1.28 ng/g lipid to 1.34 ng/g lipid (geometric means) in 13 different regions of Japan. A voluntary phase out of the sole manufacturer of two lower brominated commercial PBDE mixtures (pentaBDE and octaBDE) in North America occurred in late 2004 (Rempes 2004).

Currently, only decaBDE is being manufactured. However, due to the lag time from current usage to the time the bioacummulative pollutant works itself through the food chain, the concentrations of PBDEs are expected to continue to increase the environment and biota for several years before reducing (McDonald 2002; Sjodin et al. 2003). Furthermore, it has been shown that decaBDE may be debrominated into BDEs of human health concern such as BDE-47 and BDE-99 (Birnbaum et al. 2004).

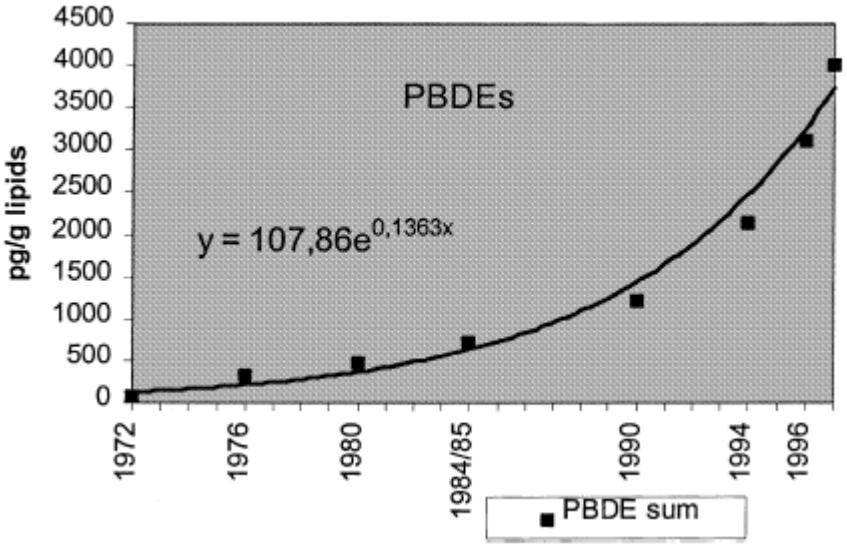

Figure 1. Concentrations an time related trends of PBDEs in human milk expressed as an exponential curve (Noren and Meironyte 2000)

Exposure to PBDEs is likely to occur through intake of foods prepared from animals with high fat content, especially fish, as well as through inhalation, ingestion, or dermal absorption of contaminated household and workplace dust (Meironyté et al. 1999; Darnerud et al. 2001; Jones-Otazo et al. 2005; Wada et al. 2005; Karlsson et al. 2007; Wu et al. 2007). In addition, PBDEs have been shown to be preferentially deposited in sub-Arctic latitudes (Hassanin et al. 2004) making this class of organohalogens a contaminant of particular interest to the sub-arctic Cree communities of the Mushkegowuk Territory (Fort Albany and Peawanuck/Weenusk First Nations) and Eastern James Bay (Cree Nation of Oujé-Bougoumou) who live in the latitudes of concern. Little exposure information is available for the PBDEs in sub-Arctic populations, therefore, in order to address concerns about PBDEs, the PBDE levels in plasma of three sub-

Arctic (south of 60^{o} and above 50^{o}) First Nation communities (n = 54) of the James Bay region of Quebec and Ontario, Canada were investigated.

1.6 Study Site & Justification

PBDEs have been shown to be deposited preferentially in sub-arctic latitudes (Hassanin et al. 2004). An investigation into possible sources of PBDEs in traditional foods of the sub-arctic region of Canada has never been done. To date, no study of any geographic region has been conducted concerning traditional foods of First Nation people. Due to the high annual consumption of fish and other wild game in Fort Albany, Oujé-Bougoumou, and Peawanuck/Weenusk First Nations (compared to non-aboriginal communities; Dewailly 2005), it is important to assess whether the traditional Cree diet is a significant route of exposure with respect to PBDEs.

According to Ryan (2005), exposure to PBDEs may occur as a result of dietary exposure, long-range deposition, and household dust. First Nations peoples in Canada carry an abnormally high environmental contaminant load compared to the general Canadian population, because many Aboriginal peoples have held onto their subsistence diet, which relies heavily on the consumption of wild game and fish. Many of these animals, especially high trophic level predators, have high contamination loads of POPs, and may include PBDEs (AMAP 1998). Arnout (2003) asserts that some PBDEs enter the food chain as a result of atmospheric deposition. This is of particular concern because PBDEs, like other POPs, are biomagnified through the food chain (Burreau et al. 1999; Gustafsson et al. 1999). The increase of POP levels in First Nation peoples is primarily due to a traditional diet, which relies heavily on fish and other wild game relatively high in contaminants. It is not known if traditional foods such as fish and wild game have elevated levels of PBDEs in this region.

1.7 Baseline Data

The results of a pilot study undertaken by Liberda et al. (2005) showed the mean lipid concentration of PBDEs for Oujé-Bougoumou Cree women (n=10) to be 28.99 ng/g lipid (SD=19.63, median = 23.83 ppb) when measured by high resolution gas chromatography-mass

spectrophotometry (HRGC-MS) and 26.96 ng/g lipid (SD=21.15, median=18.50 ppb) when measured by low resolution gas chromatography-mass spectrophotometry (LRGC-MS). Six PBDE congeners (28, 47, 66, 100, 153, and 154) constituted approximately 96% of the total sum of the 8 BDE congeners measured. The PBDE levels in the Oujé-Bougoumou Cree were substantially higher than these reported between 1996-2000 in Nunavut (Inuit from the Canadian Arctic) which had median values of 6.8 ppb for PBDE congeners 28, 47, 85, 99, 100, 153, 154, 183 (Ryan and Van Oostdam 2004). This is the only data available to date regarding sub-arctic First Nation peoples' PBDE contaminants loads. This type of data will take on greater importance if health effects now being reported in the scientific literature (e.g., Meerts et al. 2001; McDonald 2002; Mariussen and Fonnum 2003; Talsness et al. 2003) are substantiated by more comprehensive studies.

Chapter 2 - Methodology

2.1 Introduction

This section shall discuss the application of the selected methodology and concepts used in order to study PBDE exposure in First Nation communities. An ecosystem approach to human health was selected because it takes into account interactions and complexities that conventional approaches do not. In order to inform a portion of the ecosystem approach, it was necessary to employ the use of biomonitoring of PBDEs. By applying these methodologies and concepts, broad sets of parameters such as exposure levels, long range transport, threats to dietary intake, variation between communities, and community consultation were investigated.

2.2 Ecosystem Approach to Health

Lebel (2003) notes that "health cannot be considered in isolation." The environment, economy, and community all have effects that guide the health of the ecosystem (Lebel, 2003). A traditional sectoral approach is not adequate because of the complexities and interactions between humans and the environment. Therefore, focusing on a single factor such as exposure levels of a contaminant in biota will leave out other factors that affect human health. Lebel (2003) illustrates this importance by exemplifying a traditional biomedical approach to diagnosis and treatment. In Lebel's example a traditional approach to pathology would not take into account the connections between disease and socioeconomic factors or disease and environmental factors. This is because conventional approaches to health problems are typically conducted via empirical studies that look at a narrow set of risk factors (Forget and Lebel, 2001).

The ecosystem approach recognizes the links between human health and their biophysical, social, and economic environments (see Figure 2). This approach to human health promotes positive action on the environment which ultimately improves communal well-being and health and is the cornerstone theory driving this study (Lebel, 2003). Furthermore, Waltner-Towes (2004) notes that an ecosystem approach to human health will provide adaptive input for policy

development and analysis. Therefore, the ecosystem approach will inform future policy makers in such a way that adaptive measures and policies can be implemented.

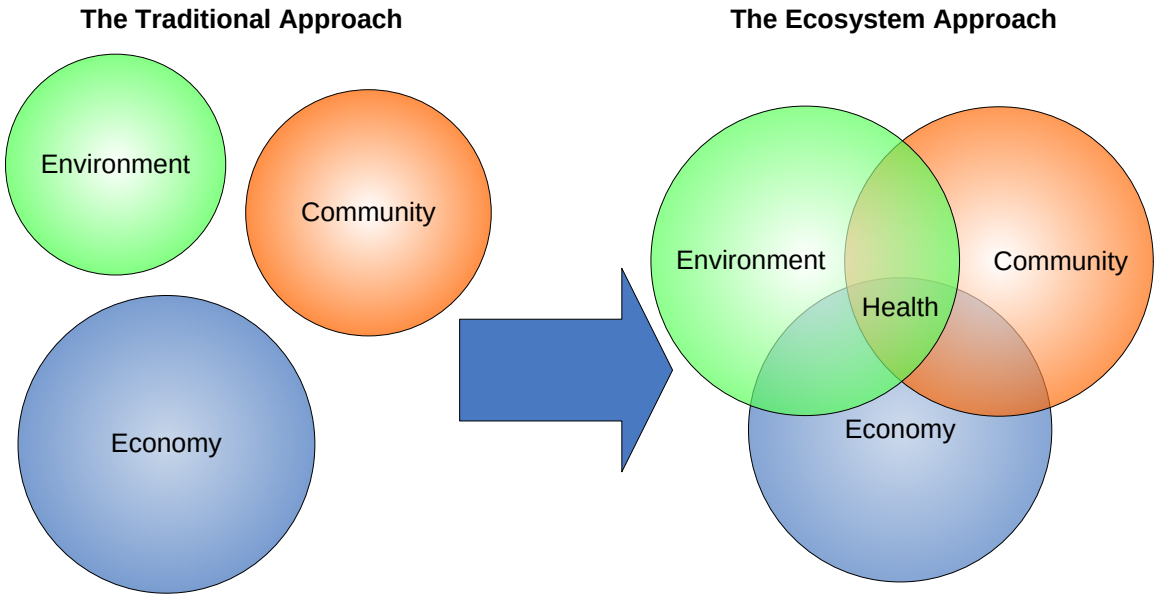

Figure 2. Conventional approach to health compared to the ecosystem approach (adapted from Forget and Lebel 2001; Lebel 2003;)

The ecosystem approach attempts to identify and assess both determinants of health of an ecosystem and health of the people who live in it (Forget and Lebel 2001). After evaluation of these determinants, effective measures can be taken to adopt social, economical, or biophysical changes, or a mixture of all three to mitigate adverse effects. The ecosystem methodology is grounded on three pillars: transdisciplinarity, participation, and equity (See figure 3, Yanggen, 2004).

- Transdisciplinarity: An inclusive vision of ecosystem-related areas of health concern.
- Participation: Involves all stakeholders such as the local community, scientific, and decision-making groups.
- Equity: Ensures fairness between gender and social groups.

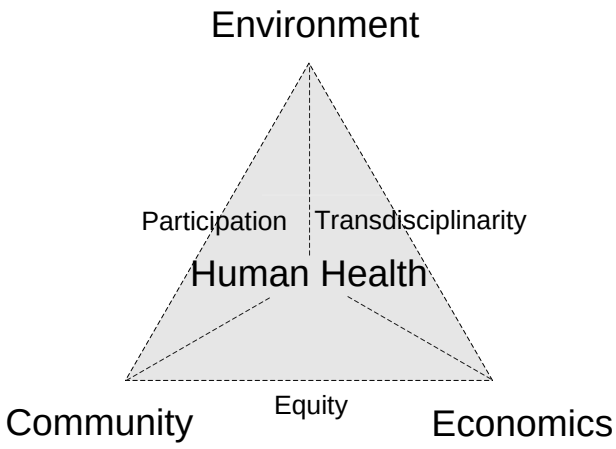

Figure 3. Ecosystem Approach to Health Conceptual Framework (recreated from Yanggen, 2004)

Ecosystem approaches to health require large interdisciplinary teams from a wide variety of backgrounds. This study did not conduct an ecosystem approach to health analysis of PBDEs and their effects on humans, but rather, used the ecosystem approach to health framework in order to guide the proposed research. For instance, socio-economic factors that contribute to health were not investigated in this study. However, the results from this study may inform future research that does look at community and economic factors. There are costs and benefits related to any modification in consumption patterns of traditional food for Aboriginal people that must be weighed (Van Oostdam et al. 1999, 2005).

While the theory of the ecosystem approach to human health guides the study, it is necessary to employ concepts which will be used to inform the link between the biophysical environment, contaminants, and human health. For example, if this study were informing the economics associated with environmental contaminants, concepts such as an economic risk assessment or a cost-benefit analysis would be performed. For this study, it is necessary to measure PBDEs in both humans and animals. In order to do this, a framework for this undertaking must be developed. In this study, a biomonitoring framework was used to quantify human and animal exposure.

2.3 PBDE Exposure

This study required the analysis of PBDE in biological material to quantify the contaminant load. Klaassen (2001) notes that exposure is "usually based on the concentration measurement of the toxicant itself or one of its metabolites in a readily available body fluid," or, more simply, that contaminant load is related to contaminant exposure.

For the PBDE analysis, the toxicant was measured in human plasma as well as in animal tissue, but could have just as easily been analyzed in milk or adipose tissue. Data obtained by direct analysis of the toxicant and not its metabolite reflects the actual received dose and hence are often employed (e.g., PCBs, DDT/DDE (dichlorodiphenyltrichloroethene); Klaassen 2001). Biological monitoring was used to measure PBDEs in humans and animal tissue.

2.4 Biological Monitoring

Biological monitoring (biomonitoring) is a procedure where exposure levels are measured in biota. Contaminant load can also be assessed by looking at biomarkers which result from the uptake of contaminants or their metabolites into body tissue or fluids (Klaassen 2001). Biomonitoring is important because when it "is appropriately used it can be more valuable in determining the level of risk from an environmental exposure than chemical analysis. This is because the tissue dose is directly measured, not merely environmental levels" (Klaassen 2001).

Hodgson (2004) and Klaassen (2001) note that when using biomarkers in environmental studies the following are necessary and were carefully considered when designing the study:

1) the objective of the biological monitoring must be defined;
2) a normal range of results appropriate for the study population should be defined;
3) decisions should be made for managing people with abnormal results;
4) a central collection point for results must be established to facilitate consistent analysis.

The concept of biomonitoring is used to inform the relationship between human and animal exposure to environmental contaminants. The framework was used to standardize sample collection, laboratory analysis, and statistical processing. This was particularly important due to the variety of animal species collected and the large number of human samples collected. Alternatively, it would have been possible to work within a different framework; such as one which analyzed the effects of environmental contaminants. However, these frameworks are often more invasive or laborious when sampling or analyzing humans and animal specimens (Klaassen 2001). Furthermore, the evaluation of environmental contaminants in biota is often undertaken as part of a biomonitoring study. Therefore, it was deemed more appropriate to implement a biomonitoring study.

2.5 Framework Justification

Figure 4, "Conceptual approach to proposed undertaking", illustrates how the theories and concepts outlined in this section inform the study. The ecosystem approach to human health informs the study by taking into consideration a multitude of factors that affect human health. While issues such as economics shall not be addressed in this report, it is important to note their role in human health. The results of this project can guide community leaders in how to plan for issues of environmental contamination with regards to human health and how they may overcome this problem as to not impact local selling of fish and game. For example, by identifying species that are highly contaminated, alternative species can be selected for consumption and sale.

This project informs the links between human health and the environment while acknowledging and attempting to provide guidance for future community and economic input into human health and environmental contaminants matters. Community participation and input was used when developing the project and continued throughout the sample collection process. For example, First Nation nurses were employed for blood collection and First Nation hunters provided input with respect to which traditional foods are hunted and also provided samples for analysis. This approach differs from conventional approaches to health problem frameworks. The latter are not

adequate for dealing with complex health issues because they are traditionally conducted via empirical studies that look at a narrow set of risk factors (Forget and Lebel 2001). Other researchers have used a similar approach. In the past, Yanggen et al. (2004) have used an ecosystem approach to health in order to evaluate the full causes and effects of pesticide use in commercial potato production. Further, Label (2003) illustrates the application of the ecosystem approach to human health with a transdisciplinary study involving mercury. The application of the ecosystem approach to toxins and human health has occurred in several studies.

While the ecosystem approach to health has many positive aspects, it also has many flaws and can be difficult to apply. For example, comprehensive ecosystem approaches require very large groups of professionals from a wide body of disciplines. Furthermore, these studies often take many years to undertake. It is important to note that these flaws are due to the scope of environment and health studies. Therefore, in order to overcome issues of temporal boundaries and spatial boundaries, conceptual and logistical challenges, only a small portion of the ecosystem approach to health was undertaken such that it could inform a portion of the problem. The focus, therefore, is on the biomonitoring of humans and traditional meats in order to quantify exposure. In turn, this information informs the link between human health and the environment in the ecosystem approach methodology.

2.6 Target Audience

As part of the ecosystem approach to human health, it is necessary to work with and inform the community in which the study is being conducted. This is particularly important because these contaminants are directly affecting the First Nation communities and the project was a result of community concerns of the emerging environmental contaminant PBDE. The potential of PBDEs to impact the health of First Nation people is a growing concern and therefore the study provides invaluable information of a potential route of PBDE exposure (the traditional diet) and baseline data on which to base future PBDE biomonitoring studies. Further, the First Nation hunters provided information regarding specific species that are eaten in these communities. Since animals may vary by region, it was important to identify traditional meats specific to the

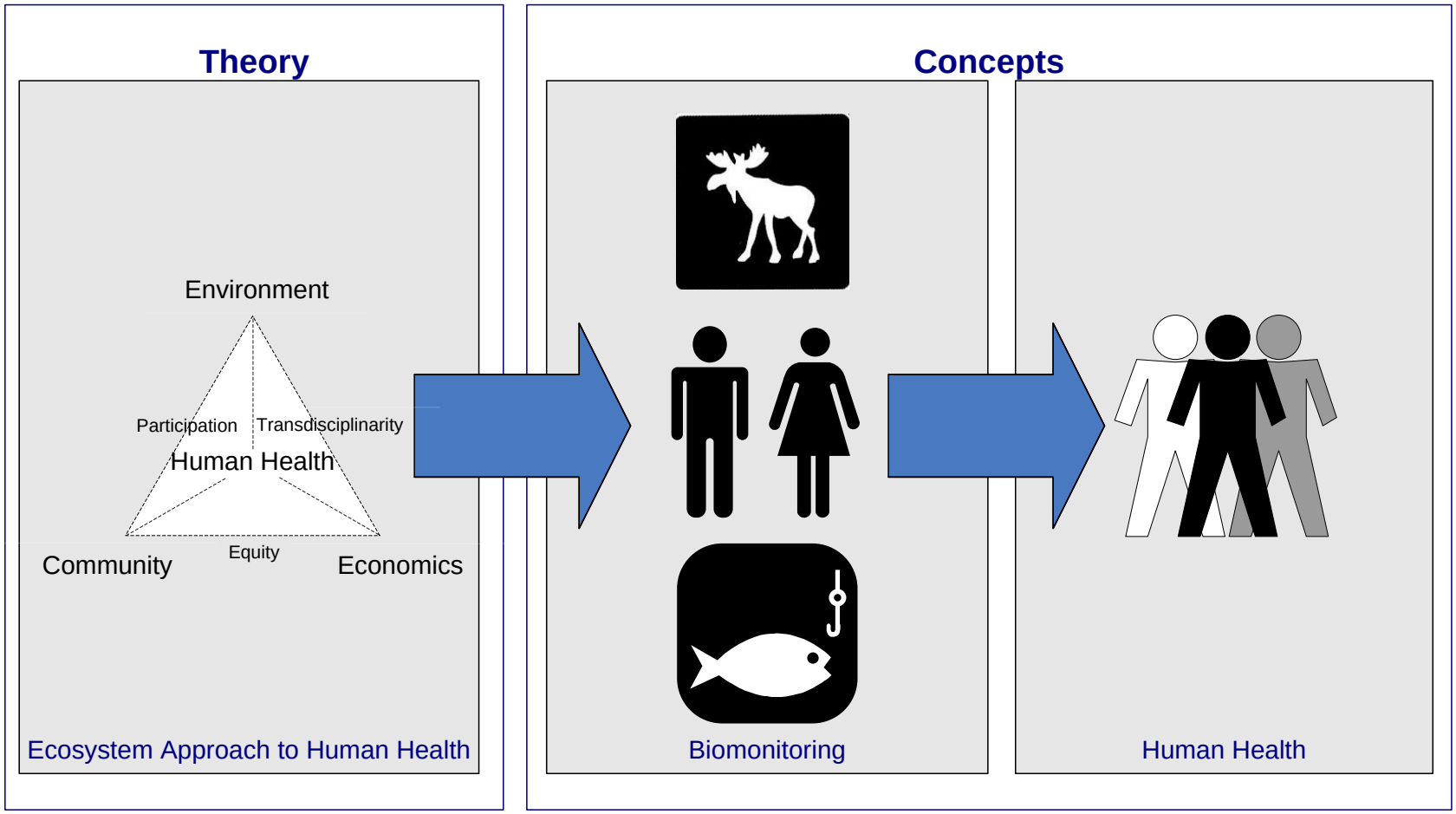

Figure 4. Conceptual approach to proposed undertaking

James Bay region. Additionally, community members also aided in blood sample collection (registered nurses) and identifying traditional meats which may pose a threat to human health. Ultimately, this information will be made available to First Nation communities, policy makers, and researchers as the findings will feed into future decision-making processes and studies alike.

2.7 Dissemination of findings

Dissemination of research findings from past projects has been extensively discussed in Tsuji et al. (1999). The study incorporates hands-on activities, modified open houses (these will be held over an extended period of time to accommodate people's schedules), educational booklets (e.g., Spent lead shot and the environment: the effects on wetlands and wildlife in the Mushkegowuk region. Fort Albany First Nation Special Environmental Health Report #1) and shall establish permanent displays in all the First Nation communities as part of an education and awareness program. The proposed study incorporates similar dissemination strategies, especially the use of modified open houses and information posters which have been found to be particularly effective in conveying information in the First Nation communities of the Mushkegowuk Territory. Decisions can then be made in concert with First Nation leaders and the results of this study on how to proceed in a fashion that will not affect human health or adversely impact the economy (e.g., selling of contaminated fish and game) and local diet. Further, information gathered during the tenure of the proposed project shall be disseminated through peer-reviewed journals and at academic conferences.

2.8 Geographic Boundaries

The study was limited to three First Nation populations located in northern Ontario and Quebec. The sampling locations were: Weenusk First Nation/Peawanuck, Fort Albany, and Oujé-Bougoumou (refer to figure 5). The three sites were chosen based on their latitudinal position and degree of isolation within the James Bay region.

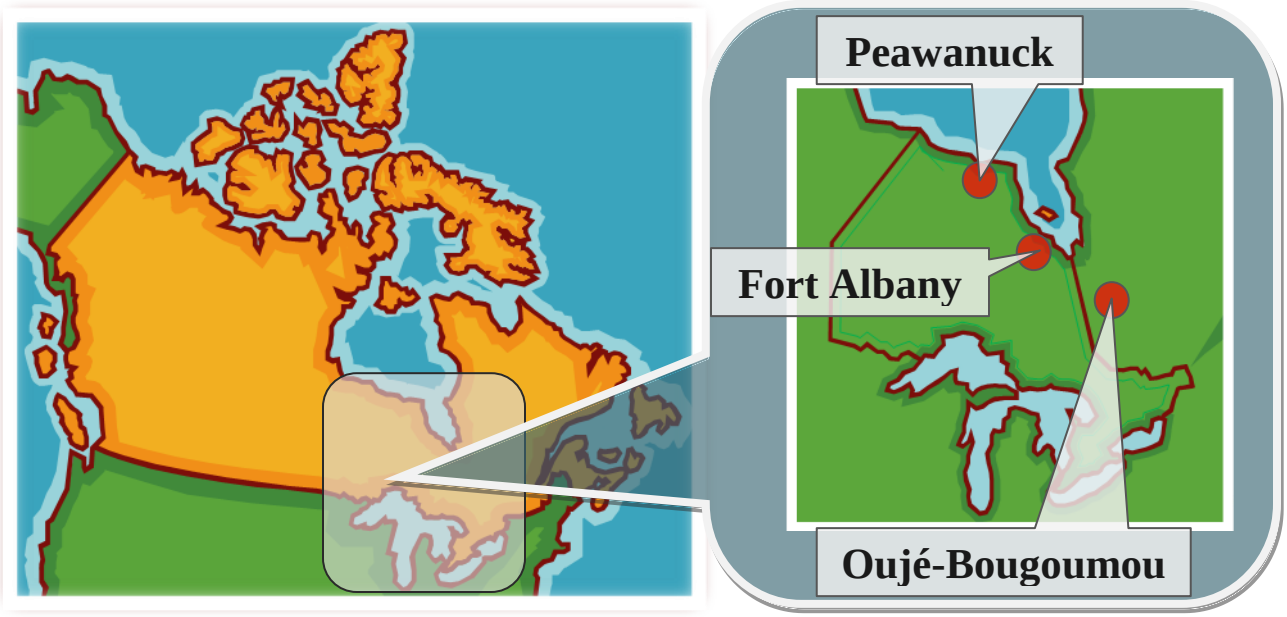

Figure 5. Sample Collection Locations of Proposed Study

2.9 Methodical Restraints of the Proposed Study

It is always a challenge to work in northern communities due to difficulty with poor weather, expensive long distance-travel, other logistical issues such as shortage of qualified personnel to assist in sample collection and cultural considerations such as depopulation of the communities due to hunting and other traditional activities. The success of other more complex projects undertaken prior to this study and the well-developed study team decreased the likelihood that these issues would inhibit study progress.

Recruitment into studies in First Nation communities can be challenging. Past studies have been able to overcome this with direct and frequent contact with the participants and their health care providers within the community while explaining the importance of the project on an individual basis and a community basis (e.g., Tsuji et al. 1999; Tsuji et al. 2006). First Nation health care providers as well as community members have been integral members of research teams. Past experience notes that First Nation peoples are highly motivated and cooperative when the importance of the work is explained and the reliability of the researchers is well established.

2.10 Conclusion

The methodology used for this project illustrates how the ecosystem approach and supporting concepts can be used in order to work towards solutions for human health concerns with regards to environmental contaminants. With regards to PBDEs, the use of biomonitoring was employed in order to quantify the exposure levels of PBDEs in humans and traditional meats. The concept of PBDE exposure, bioaccumulation, and biomagnification was described in order to explain how PBDEs can accumulate in human and animal tissues. By doing so, a variety of exposure routes were developed which would not have occurred if a conventional approach had been taken. Further, while only a portion of the ecosystem approach to human health was undertaken, the framework described how important data can be used by the community and the scientific community alike. Lastly, geographic boundaries were limited to the James Bay region due to their latitudinal position and varying degree of isolation within the sub-Arctic region of James Bay.

Chapter 3- Literature Review

3.1 Introduction

The following literature review provides background information regarding the characterization, health effects, exposure levels, transport, and bioaccumulation of PBDEs in biota and the biophysical environment. This review provided necessary information used to identify critical levels of health concern from PBDE exposure (both laboratory based and epidemiologically based) while noting global levels of PBDEs in biota. This is relevant to the study because results without context do not provide any decision-making information. PBDE levels in the First Nation communities can be compared to critical thresholds from laboratory analyese as well as other concentration levels around the world in order to determine if the James Bay region has levels which would require immediate intervention to prevent adverse health effects. The current status of PBDE production was evaluated and methods for risk assessment of other novel pollutants were presented. Reasons for the use of PBDEs in manufacturing were discussed and the difficulties in regulating this contaminant were addressed. Many of the papers presented in this review were biomonitoring studies which have been categorized into a variety of topics such as health effects, biomagnification, and long range transportation. These papers helped explain the full extent to which PBDEs may affect human health and the environment. As a result, it was possible to identify possible roadblocks in the collection and analysis of PBDEs in humans and animal tissues and develop a robust methodical procedure.

3.2 Background

POPs have plagued mankind since their inception, even though most of these chemicals have been manufactured to aid humanity to some degree. For instance, dichlorodiphenyltrichloroethane (DDT, an organochlorine pollutant) was used as an insecticide in order to control the spread of malaria (Kalantzi 2004). However, soon after coming into widespread usage, DDT was detected in the human milk. Many other new chemicals were created in order to fulfill various requirements. Chemicals such PCBs and DDT were in wide use

until they were banned in the 1970s. As expected, human tissue levels of PCBs and DDT peaked in these years but have decreased, largely due to the ban of these POPs (Kalantzi 2004). In contrast, PBDEs - first detected in the environment in 1972 - have increased throughout the 1980s and 1990s, and continue to increase up to our present day (Kalantzi 2004). When POPs such as PCBs and DDT entered the spotlight, largely though public and scientific outcry, attempts to reduce their use were finally made. For the most part, governmental action on PBDEs was fast compared to the legacy POPs and the majority of the citizenry did not known that these chemicals were even potential toxins.

3.3 Examples of the Ecosystem Approach Applied to Human Health and Environmental Contaminants

Lebel (2003) presented several examples of how the ecosystem approach to human health can be applied to environmental contaminants such as pesticides, heavy metals, and air pollution. For example, in Ecuador, the International Development Research Council (IDRC) worked with potato farmers who were experiencing health problems caused by high pesticide use (in Lebel 2003). By applying an ecosystem approach, researchers were able to work with community members to develop a solution that would reduce contaminant loads while not adversely impacting their livelihood. A conventional approach would have been to quantify the levels of contaminants in the population and compare them to acceptable levels while proposing a ban or reduction of current pesticides in use. However, this would have devastating effects on the local villages which rely on potato crops for their income. The ecosystem approach to human health allowed for a variety of solutions to be developed while working directly with the affected community. For example, financial considerations were of the utmost importance when looking at alternative pesticides for the potato crops. Not only were viable alternatives developed, but farmers were educated in how often and in what quantities these pesticides should be sprayed in order to maximize crop yield while reducing human exposure.

Lebel (2003) notes that it was due to the combined skill of staff and community members from a variety of backgrounds and cultures that it was possible to reduce pesticide dependence and its health effects. Different methods work depending on the type of culture involved. Lebel (2003) notes that these varying approaches "often proved to be complementary – offering insight into the pesticide problem from perspectives the project leaders defined as 'economic' (concerned with productivity and its financial results), 'instrumental'(focusing on hard science and statistical data), and 'interactive' (which stressed the role of the community members themselves in finding solutions."

In another example, Lebel (2003) cites how the ecosystem approach to human health was applied to the use of DDT, an insecticide often used to prevent malaria transmission by mosquitoes, in Mexico by seeking alternatives and working closely with the community to ensure that these solutions were feasible and long term. In this example, a variety of disciplines ranging from health professionals, to graphical information systems (GIS) specialists, to ecologists, and local community members from a wide variety of groups and interests all worked together in order to develop strategies to curb malaria. It is important to note that in all examples, community involvement was critical in order to ensure success. Further, these examples illustrate how the three pillars (transdisciplinarity, participation, and equity) of the ecosystem approach methodology are applied in issues of environment and human health. Lebel's research informed this study by illustrating the importance of community participation in matters of environment and health. A high degree of community participation was built into the study such as in the identification of traditional meats. This ensured that wild game and fish were analyzed from the diet specific to the James Bay region. Additionally, by working in a transdisciplinary approach, a variety of alternative sources of PBDE exposure (e.g., non-dietary intake) were brought to light (e.g., household dust and long range transport).

3.4 What are Brominated Fire Retardants?

Before one measures exposure levels of brominated flame retardants, it is important to understand what they are, why they are used, and the need for such products. This provides insight into why and possibly how this ubiquitous contaminant is found in a variety of environmental media and biota around the world and why they might present a serious enough health impact warranting investigation..

Fires have caused countless deaths and been the culprit of much property damage throughout history. Today, in the US alone, over three million fires are reported annually. Fires result in 29,000 injuries, 4500 deaths, and properly losses estimated in excess of $8 billion USD (Alaee and Wenning 2002). For these reasons, flame retardants have been added to many products.

Flame retardants are used in a wide variety of substances such as plastics, textiles, and electronic circuitry to prevent and slow down the spread of fire. Since many polymers are petroleum-based, and particularly combustible, flame retardants are applied to increase the fire resistance of these products and to help them satisfy fire safety regulations (Alaee et al. 2003). There are more than 175 different types of flame retardants, and they are typically divided into four groups: halogenated organic (typically brominated or chlorinated), phosphorus containing, nitrogen containing, and inorganic flame retardants (Birnbaum et al. 2004).

The cheapest commercially available flame retardants are brominated organic compounds such as polybrominated biphenyls (PBBs), polybrominated diphenyl ethers (PBDEs), tetrabromobisphenol A (TBBPA), and hexabromocyclododecane (HBCD), known collectively as brominated flame retardants (BFRs) (de Wit 2000). Globally, approximately five million metric tons of bromine is produced each year, and 38% of this total is used to produce BFRs (Birnbaum et al. 2004). BFRs, including PBDEs, are produced in very large quantities and used in a wide variety of materials in order to slow down the spread of fire. Looking at the volume production

alone, it is easy to see how inexpensive and ubiquitous flame retardants such as PBDEs are found throughout the built environment and ultimately become a source of exposure.

3.4.1 Reactive vs. Additive

The use of PBDEs as a flame retardant in an end product, be it textile or plastic, can occur in two different ways - additive and reactive. As an additive flame retardant, the commercial PBDE mixture is physically combined with the material being treated rather than chemically combined (Rempes 2004). The majority of flame retardant applications involve additive supplementation rather than reactive or chemical addition. Thus, additive flame retardants leech out of the chemical products. Once the flame retardant has been added as a component in a product, it becomes a diffuse source of contamination, and thus, environmental release is only a matter of time (Santillo 2003). This explains why PBDEs leech from commercial products and can ultimately wind up in the biophysical environment and biota.

3.4.2 Polybrominated Diphenyl Ethers (PBDEs)

Brominated diphenyl ethers are a group of aromatic brominated compounds in which one to ten hydrogen molecules in the diphenyl oxide structure are replaced by bromine. Commercial PBDEs typically contain three to ten bromine atoms depending on the mixture type and quality of the product. These PBDE compounds are designated tri, tetra, penta, hexa, hepta, octa, nona, and decabromodiphenyl ether (Rempes 2004). Commercially available products, typically known as technical or commercial mixes, are not pure substances; rather, they contain a mixture of brominated diphenyl ethers. Each PBDE group contains several PBDE congeners. Some of the more important congener groups are listed below:

- triBDE – BDE-28
- tetraBDE – BDE-47, BDE-49, BDE-66
- pentaBDE – BDE-85, BDE-99, BDE-100
- hexaBDE – BDE-153, BDE-154
- heptaBDE – BDE-183, BDE-190
- decaBDE – BDE-209

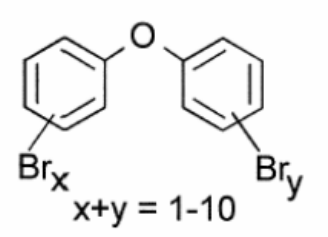

Figure 6. The chemical structure of a BDE molecule. X and Y represent 1 to 10 bromine atom(s) (de Witt 2000)

PBDEs, like PCBs, have the potential to involve 209 different congeners, this however is not seen in the environment due to the instability of some congeners (Birnbaum et al. 2004). PBDEs are numbered using the same International Union of Pure and Applied Chemistry (IUPAC) system as PCBs. The largest concentrations of PBDE congeners reported in samples are BDE-47 (2,4,2',4'- tetraBDE), BDE-99 (2,4,5,2',4'-pentaBDE), BDE-100 (2,4,6,2',4'-pentaBDE), BDE-153 (2,4,5,2',4',5'-hexaBDE), BDE-154 (2,4,5,2',4',6'-hexaBDE), and BDE-209 (2,3,4,5,6, 2',3',4',5',6'-decaBDE) (de Boer et al. 2001). It is important to understand how PBDEs are characterized with regards to their commercial mixtures as some of the congeners of concern (e.g., BDE-47 and BDE-99) are only present in certain mixes. The implications of this are discussed in the section on photolytic decomposition.

3.5 Temporal Trends

Many studies illustrate the temporal trend of increasing concentration of PBDEs (see Meironyte et al. 1999; Noren and Meironyte 2000; de Wit 2002; Ikonomou et al. 2002a; Ryan et al. 2002; Akutus et al. 2003). A study by Meironyte et al. (1999) measured the sum concentrations of PBDE congeners in Swedish human milk, and the results show an increase from 0.07 ng/g to 4.02 ng/g lipids during the 25-year study period (1972-1997). Meironyte et al. (1999) also note that the concentrations of several PBDE congeners analyzed in the study have doubled every 5 years over the past 25 years.

Akutus et al. (2003) measured pooled human milk samples between 1973 and 2000 and determined that the sum of PBDEs rose from <0.01 ng/g lipids in 1973 to a peak of 2.31 ng/g in 1998. Furthermore, Ikonomou et al. (2002a) notes that barring any changes in production or use, concentrations of PBDEs should continue on its current exponential growth, PBDEs would have overtaken PCBs as the most prevalent organohalogen contaminant by 2050 - even in the Canadian Arctic. It is important to determine a baseline value for PBDEs in the sub-Arctic region of the James Bay in order to monitor changes in levels, both increasing or decreasing, as exposure sources may still be significant.

3.6 Environmental Fate

It is necessary to understand how PBDEs enter the biophysical environment; the widespread use of PBDEs has resulted in numerous possible sources of emissions into the environment. These emission sources include, but are not limited to, municipal, hospital, and hazardous waste incinerators, facilities recycling plastics and metals from electronic devices, final disposal sites such as landfills, and accidental fires (Watanabe 2003). Furthermore, electronic equipment such as television sets and computers containing BFRs are also sources of PBDEs, particularly with regards to indoor air quality (Watanabe 2003).

The largest source of direct emissions of PBDEs into the environment are releases from factories producing PBDEs, and factories employing large amounts of PBDEs in their products such as polymers, plastics, and electronics industries (Watanabe et al. 2003). Watanabe et al. (2003) note significantly higher levels of BFRs downstream from plants which utilize flame retardants than those upstream which do not use such chemicals.

A recent environmental concern with regards to the formation of polybrominated dioxins (PBDDs) and furans (PBDFs) has come to light. Rempes (2004), Sakai et al. (2001), and Tohka and Zevenhoven (2002) note that PBDDs and PBDFs are formed during the manufacture of PBDEs, during the extrusion process of plastic recycling and during the incineration of waste

containing PBDEs. Therefore, special health and safety precautions should be taken when employees are working in areas where PBDDs and PBDFs are formed.

The EU risk assessment indicates that the majority of PBDE releases ultimately end up in soil (Sabine 2003a). This is especially true for decaBDE (BDE-209) due to its large molecular weight (McDonald 2002). From the soil, these pollutants travel via leeching with water or through wind erosion (Peltola 2003). During the warmer season, it is possible to volatilize PBDEs, which then travel long distances while adsorbed onto other particles. This information is used to explain why how PBDEs are released into the biophysical environment and identify possible sources of these emissions.

3.6.1 Long Range Transport

Understanding how PBDEs travel long distances is crucial to explaining the results observed in grazer animals of this study and illustrates the potential for increased human exposure. PBDEs are transported in the environment primarily by adsorption. Once these ubiquitous pollutants enter the environment, they will adsorb onto particles due to their low volatility, low solubility, and high affinity for carbon compounds (Peltola 2001). McDonald (2002) notes that PBDE transport in air is similar to that of chlorinated dioxins and furans, which have been found in the most remote regions of the world. PBDEs, much like other POPs, have been observed in air collected in the Arctic (Alaee et al. 2001) and in biota from similar northern regions (Ikonomou et al. 2002a).

A study by Ikonomou et al. (2002a) notes that levels of PBDEs in Arctic seals (*Pusa hispida)* were higher than levels found in human breast milk for the period between 1985 and 1996. Since materials containing BFRs are more or less non-existent in uninhabited areas of the northern Arctic, the appearance of PBDEs in the most remote regions of the globe is an indication of the efficiency of PBDEs in long-range transport (Ikonomou et al. 2002a). Furthermore, Ikonomou et al. (2002a) notes that air movement during the summer, when higher temperatures can volatilize PBDEs, carries air masses over industrial areas of North American and Japan prior to reaching

the northern Arctic. Thus, in order to reduce the environmental burden in the arctic, a reduction in production and use of all commercial mixes must occur in the aforementioned regions. The recent ban in the European Union of the commercial mixture pentaBDE is unlikely to reflect a change in the North American Arctic, but may reflect a change in the European continent. It is not certain how the voluntary phaseout of penta and octaBDE by the Great Lake corporation will change the contaminant levels in North America.

3.7 Human Exposure

PBDEs are released into the environment during their entire life cycle, that is, "from the cradle to the grave." Due to the ubiquitous nature of PBDEs, human exposure can occur in several ways; however, the main source of exposure may be via ingestion in foods (Ryan 2005; Sjodin 2003; Gill et al. 2004). Other exposure routes to PBDEs may include inhalation or dermal absorption of dust (Carson 2004). Dust which originates from aged electronic devices and deteriorating foam products can be mobilized by human activities and inhaled or absorbed (Sjodin et al. 2003). These findings are compounded by Sabine's research (2003b), which notes that flame retardants affect indoor air quality, ultimately becoming a route of PBDE exposure to humans and, in time, the natural environment.

Willford et al. (2005) and Wu et al. (2007) note that indoor release of PBDEs may be due to volatilization of lower brominated congeners or by the formation of dusts from the shearing off of treated products. While dietary intake once accounted for a large portion of legacy POP exposure, this does not appear to be the case for PBDEs. PBDEs can be added from five to thirty percent by weight in consumer products and therefore indoor air and household dust have a great potential to expose humans (Butt et al. 2004; Wilford et al. 2004). On average, North Americans spend approximately 90% of their time indoors and therefore have ample time to be subjected to contaminated dust and indoor air (Wilford et al. 2005). Of particular concern are children who often play on the floor and put their hands and other objects in their mouth which may be contaminated with PBDEs (Jones-Otazo et al. 2005; Wilford et al. 2005). Jones-Otazo et al. (2007) add that "ingestion of dust can lead to almost 100-fold higher exposure than 'average' for

a toddler with a high dust intake rate living in a home in which PBDE concentrations are elevated."

Wilford et al. (2005) have shown levels of dust to vary over three orders of magnitude from 170 ng/g (dry weight) to 170,000 ng/g (dry weight) with an arithmetic mean of 5500 ng/g (dry weight). With PBDEs at such high concentrations and close proximity to humans, it is apparent that this source of exposure is one which should be monitored. Jones-Otazo et al. (2005) and Wu et al. (2007) note that dust ingestion is a more significant route of PBDE exposure than inhalation. Ingestion of highly contaminated household dust may explain why some individuals have very high levels of tissue PBDE concentrations. However, Jones-Otazo et al. (2005) also stress that dietary factors such as fish consumption may also play a key role in solving this mystery.

Recently, Karlsson et al. (2007) found a positive relationship between PBDE concentrations in dust and plasma. However, this relationship was not dependent on household characteristics such as living area, floor material, or number of electronic devices. Wu et al. (2007) found statistically significant positive associations between human milk PBDE concentrations and consumption of dairy products and meat. Additionally, a strong association between PBDEs in human milk and household dust was observed. Household dust contaminated with PBDEs may also explain the difference in BDE body burden of European countries compared to those found in North America. Stapleton et al. (2004) note that North American demand for PBDEs greatly exceeds (e.g., North America represented 95% of the total pentaBDE commercial mixture) other continents and therefore, more products containing PBDEs are present in the homes of North Americans.

A high degree of variability between individuals has been observed in biota globally. The concentrations of PBDEs range from 17 to 462 ng/g lipid weight in the USA (She et al. 2001) and from 0.6 to 98 ng/g lipid weight in Sweden (McDonald 2002). This variability is likely due

to differing exposure routes such as diet, occupation, age, and use of products containing PBDEs (McDonald 2002). While PBDE concentration varies greatly between individuals, a human milk study from various regions in Canada found an average body burden of 0.64 μg/kg of PBDEs (Ryan et al. 2002). Using this data, it was determined that humans are exposed to PBDEs levels that are not considered harmful according to the most recent tests for lowest observed effect levels (Ryan et al. 2002; Ryan 2005).

Carson (2001) notes that in some aquatic species of the US and Europe, the concentration of total PBDEs equals or has surpassed the concentrations of total PCBs. The highest concentrations of PBDEs as single congeners found in the US occurred in bottom-feeding fish with a result of 57 mg/kg lipid weight of the pentaBDE congener BDE-47. A study by Sjodin et al. (2003) tested for variations in human exposure via seafood intake in Sweden. Swedes who did not consume fish had a median PBDE level of 0.4 ng/g lipid weight while Swedes who consumed 12-20 meals of fish per month had a median PBDE level of 2.2 ng/g lipid weight. Gill et al. (2004) estimates the daily intake of PBDEs on food consumption to range from 44 to 51 ng/day with fish contributing almost half the total amount.

While it is generally true that PBDEs have accumulated in the entire human population, those in high-risk occupations such as computer technicians, workers using PBDEs in the manufacturing process, and electronics dismantling plant workers are exposed to much higher levels of specific congeners such as BDE-153, BDE-153 and BDE-209 and typically have higher body burdens (Peltola 2001; Sjodin et al. 2003). McDonald (2002) notes that the exposure of pregnant women is of particular concern because the fetus may be especially sensitive to thyrotoxic PBDEs and these thyrotoxins can induce developmental effects. With only small changes in the thyroid hormone, developing fetuses and infants may develop developmental problems such as a decrease in intelligence and psychomotor skills (Klaassen 2001). Moreover, PBDEs, much like other POPs, are transferred via breast milk from mother to offspring and therefore postpartum exposure of the neonate is also a concern (Hooper and McDonald 2000).

This section provided input into how humans may be exposure to PBDEs. While one purpose of this study was to determine PBDE levels in the traditional meats of First Nation communities of the James Bay, other sources of PBDE exposure such as household dust and occupation were also identified.

3.8 Bioaccumulation, Bioavailability and Biomagnification

Bioaccumulation refers to the storage of a pollutant or other stable substance in living tissues, resulting in a much higher concentration of the pollutant in the tissue as compared to the surrounding environment. This is important to this study because it explains how PBDEs can accumulate in an individual. Furthermore, the pollutant may also be biomagnified through the food chain thus posing a threat to a traditional diet.

The phenomenon of high bioaccumulation in aquatic life occurs because aquatic life is more capable of absorbing large hydrophobic molecules than terrestrial mammals and hence may adversely impact people consuming these animals (Carson 2001). The bioconcentration factors from water to Baltic blue mussels were determined to be 1 300 000 for BDE-47, 1 400 000 for BDE-99 and 220 000 for BDE-153 (Gustafsson et al. 1999). Bioavailability refers to a substance's ability to be taken up by living tissues. The uptake efficiency of BDE-47, the most abundant PBDE congener in the environment, in pike (*Esox lucius*) as an amount remaining in the body after 9 days/single dose yielded a result >90%, which is higher than the uptake efficiency of CB-153 (<80%), the most abundant PCB congener in the environment (Burreau et al. 1999). The uptake efficiencies of BDE-99 and BDE-153 are similar to those of other PCB congeners studied: CB-31, CB-52, CB-77, and CB-118 (Burreau et al. 1999). Sediment concentrations compared to those in pike collected at the same site locations indicated high bioavailability for BDE-47, BDE-99, and BDE-100. CB-153 and other congeners were selected for comparison as they represent a stable and persistent environmental contaminant (PCBs), which has been present since the 1960s (Sjodin et al. 2003).

In the environment, a portion of the contaminant is available for biological uptake. This fraction is referred to as biologically available (or bioavailability; Klaassen 2001It is for this reason that the PBDE load found in sediment, for example, does not correlate to the total PBDE load found in humans living nearby. Bioaccumulation refers to the uptake of contaminants from the external environment and food (Klaassen, 2001). Johansen (2003) notes, POPs such as PCBs and DDT biomagnify "along the food chain, sometimes to thousands of times their original [concentration], posing special perils to animals, including human beings, who eat meat and fish." Colborn et al. (1996) exemplify biomagnification by following an example of a POP that has lodged itself in sediment. This POP sequestered by a microscopic organism which in turn is consumed by zooplankton (*Acartia tonsa*). The zooplankton is then eaten by mysids (*Neomysis americana)* which are consumed by lake trout (*Salvelinus namaycush*). Ultimately, by the time human eats the lake trout, its body may contain 25 million times the concentration of the pollutant found in a lake's sediment (Johansen, 2003).

Concentrations of the major BDE congeners typically increase in biota as trophic level increase, showing that BDEs are biomagnified (Burreau et al. 1999; Gustrafsson et al. 1999). Biomagnification refers to an increase in the concentration of a pollutant or other stable substance through the food chain. According to Peltola (2001), tetrabrominated and pentabrominated diphenyl ethers exhibit the highest biomagnification potential of all the PBDEs. Bioaccumulation of PBDEs is much higher in aquatic life than it is in terrestrial organisms (Carson 2001).

According to Burreau et al. (1999), BDE-47 appears to biomagnify to the largest extent. This is in agreement with studies where BDE-47 is the congener found in the greatest proportions, typically representing at least 50% of the sum of all PBDE congeners. Furthermore, Burreau et al. (1999) notes that while all congeners appear to biomagnify, tetraBDE and pentaBDE biomagnify more than triBDE, and considerably more than hexaBDE congeners. This difference in biomagnification is likely due to the size of the hydrophobic PBDE molecules (Carson, 2001).

Further, the tri to pentaBDE congeners all biomagnify more effectively than all the PCB congeners studied (Burreau et al. 1999).

McDonald (2002) argues that the BDE-209 molecule is too large to be absorbed, and thus, does not bioaccumulate. However, many studies have identified the BDE-209 congener in many biota samples, disproving McDonalds' hypothesis (Birnbaum et al. 2004; Eriksson et al. 2001; Tysklind et al. 2001). Part of this error may have been due to the technology used in identifying and quantifying samples. A HRGC-MS (high resolution gas chromatography – mass spectrophotometer) machine must be used to identify the BDE-209 congener, a bench top GC-MS does not have the resolution needed to accurately identify BDE-209 from the baseline (Ryan 2005).

The results of this section explain why PBDEs pose a threat to a traditional diet. PBDEs are bioavailable pollutants that can bioaccumulate in individuals. Furthermore, they have a great potential to biomagnify through the food chain thus possible posing as a danger to First Nation communities who may consume contaminated species.

3.9 Photolytic Decomposition

There is lack of consistency between congener patterns found in the environment and commercial mixtures available to industry. While this is partially due to the physical and chemical properties of PBDEs, the theory of photolytic decomposition has come to shed a light on the mysterious inconsistency. Although all of the PBDEs have low solubility (<1 μg/kg) and high K_{ow} values (octanol-air partition coefficient, used to describe the partitioning of PBDEs between water and environmental organic phases) (>5), the lower congeners have higher vapour pressures than the higher brominated congeners, and can be volatilized with less energy input.(Birnbaum et al., 2004). The concentrations of BDE-47, expected to be the most abundant congener found in the environment and in biota, are found in concentrations far greater than the commercial mixes provide, and can account for as much as 50% of the total BDE congeners

present (Ryan, 2005). While part of the mystery is solved by understanding the ability of humans to absorb lower brominated congeners, it is photolytic decomposition that completes the equation and may be the key to solving questions regarding PBDE uptake from household dust which is typically high in decaBDE.

Recent studies have investigated the stability of PBDEs. Particularly, decaBDE is reported to be extremely stable due to the full bromination of the diphenyl ring (Birnbaum et al. 2004). However, several studies have demonstrated the photolytic decomposition of several PBDEs including decaBDE to lower brominated congeners (Eriksson et al., 2001; Tysklind et al. 2001; Birnbaum et al., 2004). As well, Eriksson et al. (2001) notes that compounds with fewer than six bromine atoms degrade into polybrominated dibenzofurans (PBDFs). Photolytic decomposition may help in understanding how large quantities of lower brominated compounds occur in the environment and biota when the commercial mixtures produced globally do not contain such levels of lower brominated congeners.

Photolytic decomposition has important implications to future PBDE exposure because of the voluntary North American phase out of pentaBDE and octaBDE, two commercial mixtures of PBDEs. DecaBDE, a commercial mixture considered to be less bioaccumulative and toxic than pentaBDE and octaBDE, can be debrominated to the components of pentaBDE and octaBDE. Therefore, while octaBDE and pentaBDE mixes are no longer in production, it will still be possible to find an increase in these congener levels in both the environment and biota.

3.10 Health Effects

Based on the structural and mechanistic similarities it shares with PCBs, it is likely that PBDEs are thyroid hormone disruptors, neurobehavioral toxins and possibly have estrogenic activity (McDonald 2002; Meerts et al. 2001). Therefore, this report will focus on the aforementioned endocrine effects while mentioning other important findings relating to toxicology and physiology.

PBDEs may have the potential to cause other harmful effects. For instance, a study by Helleday et al. (1999) suggested that PBDEs have the same effect on human health as DDT and PCBs in terms of inducing genetic recombination, which can induce a number of diseases including cancer. Mariussen and Fonnum (2003) showed for the first time that several BFRs, including pentaBDE, were able to inhibit neurotransmitter uptake into synaptosomes and also inhibit dopamine uptake into synaptic vesicles. A study by Eriksson et al. (2002) shows that neonatal mice exposure to some BDEs caused aberrations in spontaneous motor behavior and reduced learning and memory in mice following exposure during a period of rapid brain development called "brain growth spurt." A review of neurobehavioural effects which affected changes in locomotion, rearing and total activity, combined with observed neonatal effects observed in mice, led to a safety threshold limit of 0.003 mg/kg/day for the PBDEs set by the USEPA based on the critical effect level of 0.8 mg/kg (VCCEP 2004; Health Canada 2004).

According to Fowles et al. (1994) and Brouwer et al (2001), the most sensitive end points of PBDE toxicity observed in animal bioassays are related to thyroid function, particularly induction of thyroid hyperplasia and alteration of thyroid hormone production. Furthermore, PBDEs have been shown to induce cytochrome P450 1A1 and 1A2, both *in vitro* and *in vivo*. Results from a new study by Brown et al. (2004) demonstrate the ability of BFRs, particularly PBDEs, to activate the AhR signal transduction pathway at moderate to high concentrations. This activation, although significant, may be in part due to dioxin and furan contaminants present in the commercial mixtures used for the analysis. Hamers et al. (2004) determined that di-ortho (2,2') substituted BDEs had the highest anti-AhR potency as well as the highest T4- transthyretin competing potency for OH-PBDEs.

It is important to identify critical effects in order to put First Nation exposure levels in perspective. Therefore using a variety of laboratory studies as well as epidemiological studies, the health effects associated with PBDE exposure were investigated. Below are the critical health

effects observed with the varying congener groups that were considered to be the most important.

3.10.1 Thyroid Effects

Thyroid hormones police brain development in both fetal and neonatal periods (McDonald 2002). They are responsible for controlling the proliferation of neuronal and glial cells, regulating neuronal migration and differentiation, and regulating neuronal connectivity and myelination. Furthermore, thyroid hormones regulate the development of cholinergic and dopaminergic systems in the cerebral cortex and hippocampus (McDonald 2002). There are indications that adults, and especially women, are experiencing hypothyroidism in endemic amounts (Muir 2003). Maternal hypothyroidism is one mechanism that can lead to neurodevelopmental deficits in the offspring.

The chemical structure of PBDEs is somewhat similar to the thyroid hormones thyroxine (T4) and 3,3',5-triiodothyronine (T3). It is this structural similarity that is believed to be one mechanism by which normal thyroid hormone effects are disrupted (McDonald 2002). It is thought that PBDEs alter the thyroid system via hormone mimicry, in which they bind to the transport protein transthyretin. Meerts et al. (2000) established that 14 congeners (out of the 17 tested) could be metabolized *in vitro* to produce metabolites that competed with the thyroid transport protein, transthyretin, thus creating a deficit in T4. Transthyretin is believed to be an important mechanism for the transport of T4 from the mother to the fetus and across the blood-brain barrier, thereby explaining concentrations found in unborn fetuses (McDonald 2002). Gill et al. (2004) notes that hydroxylated PBDE congeners (OH-PBDE) that resembled T4 and T3 were 1.22 and 1.42 fold more potent than T4, respectively. Both Legler and Brouwer (2003) and Zhou et al. (2001) observed reductions of total serum and free thyroid hormone (T4) when rodents were exposed, *in vivo* to various PBDE compounds.

A recent study on the temporal effects of the commercial PBDE mixture DE-71 on thyroid hormones consisting of 5, 20, and 31 day periods of dosing was undertaken by Stoker et al.

(2004). In the study, Wister rats were gavaged daily with 0, 3, 30, or 60 mg/kg DE-71 in corn oil from postnatal day (PND) 23-53 for males and 22-41 for females. The results indicate that serum T4 was significantly decreased at 30 and 60 mg/kg in the 5 and 20 day exposed female rats. This could be due to the competition between PBDEs and T4 for transthyretin (McDonald 2002). T3 decreased and TSH was elevated by 30 and 60 mg/kg in the 31 day exposed male rats. Both male and female groups (20 and 31 day exposure) dosed with 60 mg/kg produced decreased colloid area and increased follicular cell heights (indicative of hypothyroid state). Furthermore, increased liver to body weight ratios were observed at the two highest doses, while seminal vesicle and ventral prostate weights were reduced at 60 mg/kg in males. In females, a dose of 60 mg/kg caused a significant delay in the age of vaginal opening.

3.10.2 Neurobehavioral Effects

Maternal and fetal thyroid hormone levels are crucial to proper brain development in the fetus (McDonald 2002). The mother is the only source of thyroid hormones for the fetus during the first trimester, and is the major source in the second trimester. During these periods, abnormalities in maternal thyroid hormone levels can result in reduced intelligence (McDonald 2002). Various studies in newborn mice show that PBDEs cause learning and motor deficits that worsen with age (McDonald 2002; Eriksson et al. 2002). PBDE exposed mice typically exhibit permanent irregularities in motor behaviour and reduced learning and memory, which are signs of deficits in brain development. Branchi et al. (2001) were in agreement with other studies where their findings show that behavioural alterations worsen with increasing age. This becomes clearly evident around one month of age. A very interesting experiment by Bernes (1996) indicated that mice exposed to PBDEs expressed an inability to process new stimuli, as measured by hyperactivity. However, it should be noted that these experiments typically occurred at values which greatly exceed current human body burdens.

McDonald (2002) outlined three possible mechanisms by which PBDEs can adversely affect brain development:

- o thyroid hormone disruption
- o disruption of second messenger communications
- o alteration of neurotransmitter systems

While it is not yet known which of these mechanisms is responsible for neurodevelopmental deficits, it has long been established that thyroid hormone imbalance can produce such deficits in humans and rodents (McDonald 2002). McDonald (2002) believes that all PBDEs, which are co-planar due to the oxygen linkage between the diphenyl ring, induce similar effects as non-coplanar PCBs.

3.10.3 Reproductive Effects

While evidence suggests that PBDEs have a greater potential for thyroid hormone mimicry than for estrogenic mimicry, various studies have indicated that PBDEs interfere with estrogen pathways, which may result in sexual behavioural effects. Therefore, an investigation into this matter is necessary. In adults, sex hormones primarily influence reproductive functions such as sexual drive, sperm production, menstruation cycle and pregnancy (Bernes 1996). Sex hormones also play a critical role in the fetus where they guide the development of reproductive organs. Furthermore, these hormones influence the development of the thyroid gland, liver, immune system, and the brain (Bernes 1996).

Lichtensteiger et al. (2004) determined that female rats exposed to high doses of BDE-99 (10mg/kg/day) resulted in a “massive” impairment of sexual behaviour. This conclusion was reached by observing the effect on lordosis quotient (number of lordosis reactions/mating number) and on incentive behaviour to attract male rats. A study by Kuriyama and Chahoud (2003) was the first to report that rats exposed to single doses of BDE-99 (60 μg/kg and 300 μg/kg) affected male fertility. The dosage levels were well below the established NOAEL for pentaBDE (2 mg/kg/day). Treated rats had changes in relative testes, epididymis and spleen weight which is indicative of male fertility impairment and immunotoxicity. Furthermore, the

experiment resulted with significant decreases in daily sperm production as well as a decrease in spermatic count.

While it is not entirely known how PBDEs exert estrogenic effects, Legler and Brouwer (2003) suggest that PBDEs bind to and activate estrogen receptors in a similar fashion as they do for the Ah receptors. Both Legler and Brouwer (2003) and Meerts et al. (2001) show that BDE-47 was weak in estrogenic activity compared to estradiol. BDE-100, BDE-75, and BDE-51 possess estrogenic activities with potencies of about 10^{-6} in relation to estradiol. Brouwer et al. (2001) notes that BDE-100, BDE-75, BDE-51, BDE-30, and BDE-119 act as estrogen agonists that were 250,000 to 390,000 times less potent than the natural ligand, E_2. Typically, PBDEs with minimal bromination are the most estrogenic (Letcher and Bennett 2001). PBDEs and OH-PBDEs have estrogenic potencies in the same range as environmental bisphenol A (Meerts et al. 2001).

3.11 Risk Assessment

Assessing the risk of emerging environmental pollutants is a very difficult task. While many tools have been created to aid in risk assessment, Palm et al. (2002) have outlined a method for risk assessments for PBDEs, and other POPs, in order to provide background data for regulatory decision-making. This method has been broken down into six steps (modified from Palm et al., 2002):

- o chemical classification and determination of properties
- o acquisition of emission discharge data
- o assessment of the likely behaviour of the chemical in the environment
- o evaluation of fate in the environment
- o establish behaviours or concentrations in highly exposed localities
- o comparison of estimated and observed concentrations with effect or no observed effect levels (NOELs)

By using the above risk assessment strategy, a systematic method could be set in place to assess future environmental pollutants such as PBDEs in health and environmental risks. If such a study has been completed, a full understanding of fate and effects of that emerging contaminant will be demonstrated, setting the foundation for regulatory measures to be created (Palm et al. 2002). However, both Palm et al. (2002) and Gill et al. (2004) note that one stumbling block exists with regards to risk assessment methods for PBDEs: the quantity of data needed is very large. Therein lies the problem- data regarding new chemicals or pollutants of emerging concern are not available, and thus must be determined experimentally. Nevertheless, the method outlined by Palm et al. (2002) is a strategic action plan that can be used not only for PBDEs, but all POPs as well.

3.12 Global Action

Governmental action plays a difficult role with regards to regulating PBDEs. Reductions in the manufacturing of POPs and other toxic substances that are a hazard to the environment are in governmental support. However, governmental bodies are also required to protect its citizens from the dangers of domestic fires. How can governmental bodies regulate BFRs, particularly PBDEs, and not impact on the safety and well being of their citizens?

Governments are slowly taking action on the growing PBDE problem. In 1994, when a proposed ban of the production of PBDEs in the European Union could not be passed, Germany self regulated by setting maximum concentration levels for polychlorinated dibenzo-p-dioxins and furans, a byproduct of PBDE manufacturing (Sabine 2003a). In 1996 the OECD's Risk Reduction Programme identified PBDEs as a growing concern, and agreed on a voluntary industry commitment with international flame retardant producers to focus their production and application on the main technical PBDE formulations, penta, octa and deca-BDE (Sabine 2003a).

In 1996, Sweden banned various PBDE mixes. However, it was not until 2003 that the first major step in PBDE reduction occurred when the European Union banned the use of pentaBDE,

and heavily restricted the use of octaBDE by the end of 2004 under the EU Directive 2003/11/EC. Furthermore, the EU Directive also restricted the use of pentaBDE and octaBDE in electrical and electronic equipment which was phased out by July 2006 (Birnbaum 2004). California followed suit by issuing a ban on the manufacturing, processing and distribution of products containing more than a tenth of a percent of pentaBDE and octaBDE after January 1st 2008. Additionally, Great Lakes Chemical, the only producer of of PBDEs in North America, is voluntarily phasing out penta and octa commercial mixes by the end of 2004. Since there is no other North American based producer of penta and octa commercial mixes, the voluntary phase out will have a broad continental effect (Rempes 2004). While many skeptics argue that banning pentaBDE and octaBDE alone is not enough, a study by Ikonomou et al. (2002a) shows that PBDE level in human milk in Sweden have dropped since that country implemented measures to control environmental releases of PBDEs. Thus, action taken to control PBDE levels would be successful if other countries take similar approaches.

The UNEP POP Convention, signed in 2001, and the 1998 POP protocol of the UNECE Convention on Long-Range Transboundary Air Pollution are tools which may be used to restrict the use and release of POPs. BFRs, and more importantly, PBDEs, are not included within the substance lists for these conventions, but so far only the pentaBDE mix fulfils the screening criteria for the inclusion of new substances under both POP frameworks (Sabine 2003a).

In Canada, PBDEs were brought into light in 1997 when the Department of Fisheries and Oceans (DFO) funded a study on background levels of contaminants in aquatic life. This study resulted in an amendment to the Canadian Environmental Protection Act (CEPA) in 1999 to include PBDEs on the Domestic Substance List, requiring categorization and screening level risk assessments. This resulted in a regulatory review, which was completed in February of 2004 (Sabine 2003a; Health Canada 2004). Conversely, the federal US regulatory bodies are resisting control efforts, emphasizing that the flame-retardant properties of PBDEs outweigh the current, understood risks (Ikonomou et al. 2002a). It is important to understand current PBDE control

measures in order to inform future studies. For instance, the North American ban on pentaBDE and octaBDE could have various health implications (e.g., an increase in decaBDE use leading to a shift in sources of exposure).

3.13 Gap Analysis

Several data gaps remain with regards to PBDEs. This is primarily due to PBDEs being a newly emerging area of research. Ryan (2005) notes that many sources of human uptake are still unknown. He clarifies that while some of the total human uptake still comes from food, there is still a large percentage that is unaccounted for and may attributed to household dust. In the past, human uptake of POPs such as DDT and PCBs were almost entirely from food consumption; however, this does not appear to be the case with PBDEs. Ingestion of household dust containing PBDEs explains how a large variation between individuals is sometimes seen as well as why North American levels are much higher than other continents (e.g., Europe).

Wenning (2001) has identified several important data gaps such as the relationship between dietary levels and adverse health effects, relevant toxicity endpoints and likely effects-thresholds in humans and wildlife, and environmental fate and pathways of exposure. Gill et al. (2004) notes the need for data regarding PBDE body burden with particular attention to women of reproductive age, recreational and subsistence fishers, and aboriginal and individuals from Arctic countries that have a high intake of aquatic fish and mammals. Muir (2004) notes that several gaps in research are missing. First, risk assessment needs to move beyond the focus on average body burden, tissue or human milk concentrations, and account for the population distribution of the concentrations. Secondly, it is noted that not all interactions between congeners are the same. Exposures to varying PBDEs congeners and other harmful substances may be additive or synergistic, and this fact must be considered when assessing exposure in order to properly assess risk.

To date, there are no standardized analytical methods for the determination of PBDEs in the environment or biota. While there has been rapid development over the last 6 years, most

advances were for chlorinated pollutants such as PCBs (Covaci et al. 2003). Due to this lack of consistency between labs, de Boer et al. (2002) from Norway began an international laboratory study to try and compare results from unknown samples. The results from the first study demonstrated that analytical methods for the determination of PBDEs need further improvement, but provided global collaboration on analytical methods. In spite of this, Ryan (2005) stated that PBDEs have become the new focal point for many research groups and thus analytical test methods have improved greatly for the PBDEs within the last few years.

3.14 Discussion

PBDEs are a global issue that leads to cross-border contamination, and not merely localized environmental issues. Hale et al. (2003) and Ikonomou et al. (2002a) notes that PBDEs, particularly lower brominated PBDEs, are becoming greatly dispersed from urban areas to remote areas. This same pattern was observed with PCBs, DDTs, and other POPs which have since been banned.

The use and manufacture of PBDEs has resulted in widespread environmental contamination. Governments around the world have, to some degree, acknowledged this problem and initiated studies intended to provide information with regards to their significance and ultimately determine the need for regulatory action. So far, in North America, such action is just starting to take place.

Implementing a precautionary approach would include mandated health and safety tests before new chemicals such as PBDEs are allowed to come into the market place for global use. A precautionary approach would also require existing chemicals to undergo similar tests in order to evaluate their health and safety effects, and to determine if they should remain on the market. The European Union is taking the aforementioned step by implementing a program for the Registration, Evaluation, and Authorization of Chemicals, or REACH. This program transfers the burden of proof of health and safety from the government to the manufacturer (Northwest Watch 2004).

While welcomed, the chemical phase-outs and bans discussed in this report still leave problems to be addressed. First, many of the phase-outs and bans may still allow for decaBDE to remain on the market. While originally thought to be a stable, non-bioaccumulative compound, decaBDE has regularly been detected in breast milk and biota globally. Furthermore, new evidence points to the possibility of photolytic decomposition of decaBDE possibly accounting for the large occurrence of BDE-47 (a pentaBDE) in all environmental samples. Secondly, the phase-outs and bans do not remove or attempt to deal with any of the current products that contain large amounts of PBDEs (North West Watch 2004). The Northwest Watch study of 2004 admits that perhaps nothing can be done in this regards. Removal of products such as foam from furniture and carpet backings, which contain the most amount of PBDEs w/w, would be too costly, and may create additional sources of exposure to these chemicals by dust.

While knowledge of the toxicity and physiology of PBDEs is limited, what information that is available indicated that toxicological and physiological endpoints are similar to those seen for PCBs. While data are continually being produced regarding this subject matter, it appears that PBDEs disrupt thyroid hormones, produce deficits in neurodevelopment, affect reproduction, and may cause cancer. Due to the dominating ubiquity of BDE-47 in the environment and biota, it is essential to understand the basic toxicity and physiological end points before human health risk can be adequately assessed.

Information on the fate and impact of PBDEs in the environment must be moved to the forefront of research. Particular attention should be paid to the behavioural and cognitive impacts of PBDEs in both humans and wildlife. Arctic ecosystems and arctic communities should be a significant component of the aforementioned research, as they are positioned to become the ultimate sink for PBDEs.

3.15 Conclusion

Examples of how ecosystem approaches have been used in the past were presented in order to illustrate important parameters such as community involvement and equity. Specifically, examples regarding contaminants such as DDT and other pesticides were highlighted due to the similarities between these example cases and the current study. The studies demonstrated how biomonitoring results can be used in a broader context to inform not only health issues, but economic ones as well (e.g., reduction in pesticides to reduce health effects and save on insect control products).

Additionally, a variety of health effects associated with PBDE exposure have been identified. The critical health effects associated with PBDE exposure occur at very high exposure levels. These levels typically surpass current PBDE concentrations in humans around the world. Information regarding long range transportation, photolytic debromination of higher brominated compounds (e.g., BDE-209), and bioaccumulation factors were also presented. This informs the study by noting other possible sources of PBDE exposure in both humans and animals. This may prove to be important due to the North American phase out of two commercial PBDE mixtures in 2004. Further, it provides information that can be incorporated into an expansion of PBDE biomonitoring (future studies). For example, household dust was not considered an exposure pathway in the James Bay region and therefore not analyzed. Bans and voluntary phase outs were described for several regions around the world. It is not yet known, however, how soon these bans will affect levels in the biophysical environment and in biota.

Chapter 4– Methods

4.1 Introduction

This methods chapter describes how human and animal samples were collected and the analytical procedure used to analyze the specimens in the laboratory. Laboratory performance parameters are present in order to demonstrate both accuracy and precision of the PBDE analytical results. Lastly, the manipulations of all the data variables from the laboratory analysis are described in order to perform statistical analyses.

4.2 Analytical Background

Methods for PBDE analysis in the environment and biota are limited by analytical technologies. However, a new analytical technique takes advantage of carbon isotopes in order to conduct accurate congener analysis. While it is possible to analyze for PBDEs without using an isotope method, the identification of certain congeners becomes more difficult due to decreased instrumental resolution from the lack of a mass spectrophotometer. Therefore, a method which uses carbon isotopes, known as "isotope dilution," was used for the analysis of PBDEs in order to maintain high resolution between congeners and compatibility with available instrumental equipment.

Additionally, the choice between high resolution gas chromatography-mass spectroscopy (HRGC/MS) and low resolution gas chromatography-mass spectroscopy (LRGC/MS) existed. LRGC/MSs, also known as "bench-top GC/MSs", are far cheaper and smaller than their high resolution counterparts which require special training, large operation rooms, and upwards of $1 million to purchase. However, it is important to note that low resolution GC/MS limits the researchers' ability to successfully distinguish between the higher brominated compounds such as those found in the decaBDE commercial mixture; the HRGC/MS can successfully distinguish between all BDE congeners. Since only pentaBDE congeners were being analyzed, using a bench-top GC/MS would provide adequate results and hence was employed for the analysis.

4.3 Methods

In order to characterize the PBDE pollutant in humans and traditional foods, the pollutant must be measured in a variety of biota. Human samples as well as traditional foods and leeches (data unpublished) were analyzed in order to determine the extent of PBDE contamination. While there are minor variations to the analytical quantitative analysis of PBDEs in biota, they all involve the same principles: extraction, homogenization, lipid removal, column fractionation, and analysis via a GC/MS using stable ^{13}Carbon isotopes.

4.4 Field Sampling Methods

This section shall discuss the field sampling procedures. Human blood samples were used since PBDEs are lipophilic and accumulate in blood plasma. Confounding variables in studies that measure exposure may include age, cigarette smoking, sex, drug use, diet, or pre-existing health factors. Screening for this study included drug use restrictions and were limited to non-smokers. Therefore, sample size was limited by recruitment, however, statistical significance was obtained. Animal tissue was used for PBDE analysis because this represents what is typically eaten. Specimens were limited by the success of the First Nation hunters and therefore all meats obtained were sampled.

4.4.1 Human blood sampling

Blood samples for analyses were collected from the First Nation participants in 10-ml glass Vacutainers® (lavender-top, containing the anticoagulant, EDTA, Becton-Dickinson #7665) by a registered nurse who serves the First Nation communities. Immediately after blood collection, the blood was gently mixed with the anticoagulant in the Vacutainer, then centrifuged at room temperature (@1000-1200xg for 10 minutes). Once the plasma was separated, it was transferred from the Vacutainer® with polyethylene pipettes (Baxter #5214-10) to shatter-resistant, pre-cleaned (hexane) glass vials (Supelco #2-3178) with Teflon-coated lids. The plasma samples were then frozen and stored at -20°C. All plasma samples were then shipped frozen in polystyrene-insulated coolers over frozen gel packs to the Institut national de santé publique du Québec (INSPQ) laboratory for analysis. Twenty one samples from females between the ages of

19 and 38 were collected from Oujé-Bougoumou, thirteen samples from females between the ages of 26 and 42 were collected from Fort Albany, 10 samples from females between the ages of 23 and 69 and 10 samples from males between the ages 24 and 71 were collected from Peawanuck in 2005.

4.4.2 Bird sampling

Bird specimens were salvaged from the normal First Nation spring harvest of upland game birds during the summer of 2005. Birds were sexed and morphological measurements made. The birds were stored frozen until processing. Samples were thawed and a portion of the breast tissue sampled using stainless steel blades(blades were washed, then rinsed with distilled water and/or acetone [Optima, Fisher Scientific]); breast tissue samples were then individually placed in 60 mL Nalgene (# 2116-0060) straight-side, wide-mouth containers and shipped frozen to the INSPQ laboratorys for analyses.

4.4.3 Fish sampling

Random specimens were chosen from the First Nations catch during summer 2005. Fish were wrapped individually in tin foil and stored frozen until dissection. Samples were dissected using stainless steel blades (blades were washed, then rinsed with distilled water and/or acetone [Optima, Fisher Scientific]) in Fort Albany (where we have laboratory facilities) and a portion of fillets placed in Nalgene wide-mouth containers and shipped frozen to the INSPQ laboratory for analysis.

4.4.4 Large mammal sampling

Mammal samples were salvaged from the normal First Nation fall/winter harvest of moose (*Alces alces*) and caribou (*Rangifer tarandus*), of 2005. Samples were stored and frozen until processed in the laboratory. Samples were thawed and a portion of the tissue sampled using stainless steel blades (blades were washed, then rinsed with distilled water and/or acetone [Optima, Fisher Scientific]); tissue was then be placed in wide mouth Nalgene containers and

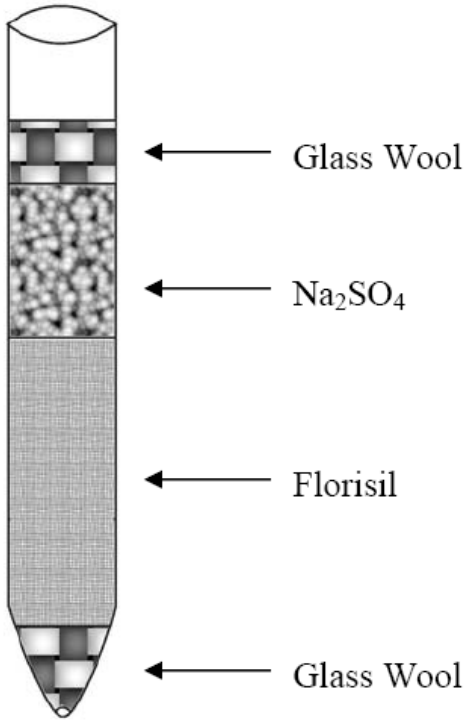

Figure 7. A Florisil Column

shipped frozen to the INSPQ laboratory for analyses. Analytical Procedure for Human Blood Plasma Samples

4.5 Analytical Procedure for Human Blood Plasma

This section shall discuss the analytical procedures in order to determine levels of PBDEs in blood plasma. The method used for analysis is referred to as "isotope dilution."

4.5.1 Extraction

Blood plasma samples were mixed with pesticide grade dichloromethane (Omnisolv), sodium sulphate (Fisher ACS, burned at 400 for 12 hours) and 2 ng 13C-PBDE 77 as internal standard (CIL). The brominated flame retardants were extract by ultrasonic treatment with sonicator MISONIX, (Fisher), using a 12.7 mm probe for 2 minutes at intensity 5. The extract was filtered on prewash glass whole (Corning) and sample was re-extracted twice with dichloromethane. The combined extracts were made to 100 ml with dichloromethane.

4.5.2 Fat determination

The sample fat content of was determined gravimetrically using a portion of the extract prepared above. Ten mls of the extract was transferred in pre-weighed aluminum dish and solvent was evaporated in chamber maintained under moderated vacuum until its complete evaporation. Residual material was weighed and consider as fat.

4.5.3 Purification GPC

The unused extract was evaporated near dryness with a rotary evaporator (Büchi RE-111) and reconstituted in 1.2 ml with dichloromethane. The concentrated extract was filtered on 0.45

membrane (Millex HV13) and purified with gel permeation chromatography (GPC) on liquid chromatography system (Water 510) mounted with two GPC columns (19*150mm and 19*300mm, Waters). The fraction containing BFRs was collected and evaporated to near dryness

4.5.4 Florisil Purification

The purified extract was reconstituted in hexane (1 ml) and further purified on a florisil dual columns system, containing each 1.5 g of deactivated 5 % Florisil (Fisher). After deposition on the first column, the fraction containing the BFRs was eluted with 10 ml of dichloromethane in hexane 25%. The collected fraction was evaporate with a Speed Vac Plus evaporator system (Savant).

4.5.5 Analysis

The purified extract was transferred to GC vial and made to a final volume of 100 µl with hexane. The extract was analysed for its BFRs content on an Agilent (Wilmington, DE, USA) 6890 Network chromatograph (GC) equipped with an Agilent 7683 series automatic injector and an Agilent 5973 Network mass spectrometer (MS). The GC was fitted with an Agilent 60 m XLB column (0.25 mm i.d., 0.25 mm film thickness). The carrier gas was helium, and all injections were 2 µL in splitless mode. The mass spectrometer was operated in selected ion monitoring (SIM), using electron capture negative ionisation (ECNI) with methane (99.97%) as the reagent gas. Masses 79 and 81 were monitored for all the brominated compounds with 13C-PBDE 77 as the internal standard. The target ion (81) was employed for quantification and the confirmation ion (79) from the same isotopic cluster was used to confirm the identity of the compound. The source temperature was kept at 150°C and the quadrupole was set to 103 °C. The injector and transfer line were kept at 275°C and 280°C, respectively.

4.6 Analytical Procedure for Animal Tissue Samples

The method for the analytical procedure for animal tissue used was the same of that of the blood plasma except that animal tissue samples were used in the ultrasonicator rather than plasma samples.

4.6.1 Quality Control Measures

Several quality control measures were employed in the PBDE analysis. Procedural blanks were used to determine background levels of PBDEs and remove these results from the samples. Additionally, a quality control (QC) sample is employed as part of the samples analyzed. The QC sample has been analyzed many times in the past; and thus, results could be compared for precision and accuracy. When detecting the target compounds, GC retention times were matched to the standard compounds. The signal to noise ratio was greater than three, and the isotopic ratio between the quantitative ion and the standard ion was within $\pm$ 20% of the theoretical value.

4.7 Performance Parameters for PBDE Analysis

General GC-MS performance parameters for the analyses are presented below. The limit of detection is the minimum concentration that can be detected. It represents a signal to noise ratio of three. The quantification limit is the minimum concentration that can be quantified with a certain degree of confidence. It represents a signal to noise ratio of 10. Linearity is defined as the range of concentration of the calibration standards that lie between the quantitation limit and the limit of linearity. Alternatively, linearity can be thought of as the closeness of a calibration curve to a specified straight line. Linearity is expressed as the maximum deviation of any calibration point on a specified straight line during any one calibration cycle. Repeatability is the precision for a given batch of analyses (i.e., same day and same calibration). Reproducibility is over a longer period of time (i.e., different days and different calibrations).

Table 1. BDE Parameters for GC-MS Analysis

PBDE Congener	**Limit of detection** (ng/g)	**Limit of quantification** (ng/g)	**Linearity** (ng/g)	**Repeatability** (% error)	**Recovery** (%)	**Reproducability** (% error)
BDE 47	0.03	0.49	0.49 to 40	12.0	82	27.1
BDE 99	0.02	0.56	0.56 to 40	14.8	82	-
BDE 100	0.02	1.6	1.6 to 40	14.5	72	-
BDE 153	0.01	0.74	0.74 to 40	7.1	80	-

4.8 Statistical Analysis

Statistical analyses were conducted using SPSS. A log-linear model was used to analyze frequency data (i.e., detectable concentrations and non-detectable concentrations) for BDE congeners. Log-linear models comprise a contingency-table approach to deal with the analysis of data which are not agreeable to multiple tests of independence by the chi-square test (Tsuji et al. 2005). Two-state log-linear modeling was conducted to examine the effect of location on detectability for BDE congeners with frequencies of detection <70% (BDE 47 = 61.1%; BDE 99 = 18.5%; BDE 100 = 25.9%). In the four-state case, the effect of location on quartile frequency of detection was analyzed for PBDE 153 (frequency of detection = 66.7%). An adjusted standardized residual (ASR) exceeding 1.96 is considered statistically significant ($p<0.05$). Detailed information regarding the application of log-linear modeling to organochlorine data can be found in Tsuji et al. (2005).

All congener data were lipid adjusted, but no imputations were made in cases where non-detections occurred, and then summed (ΣBDEs). The ΣBDEs was imputed (1/2 detection limits for non-detects) and the data were logged to adjust for non-normally distributed data. A one way ANOVA was performed on sex data for Peawanuck; no significant difference ($p>0.05$) was found, so gender data were combined for Peawanuck. ANOVA on ΣBDEs data for Oujé-Bougoumou, Fort Albany, and Peawanuck was performed with appropriate post-hoc tests. Only descriptive data are presented for wild game and fish as consumption data relating to the James Bay region is currently being accumulated. Once this data is available, dietary intakes can be estimated and statistical comparisons made.

4.9 Conclusion

Human blood plasma was chosen for analysis of PBDEs because of the contaminant's lipophilicy. Furthermore, drawing blood is less invasive than analyzing adipose tissue for PBDE content. While screening questions to reduce confounders were used to determine if individuals would be excluded from the study, statistical significance was still obtained. Additionally, tissue samples were analyzed on the basis of the animal parts that are eaten in a traditional diet.

Analytical methods for the analysis of PBDEs are limited by technology. This method employed the use of "isotope dilution," a novel method for the analysis of PBDEs which is quickly becoming a global standard technique due to the method's ability to accurately quantify individual congeners. A low resolution GC/MS had no bearing on the results as only low molecular weight BDEs were analyzed. Lastly, manipulation of the data for statistical analysis was described.

Chapter 5 – Results of PBDE Analyses

5.1 Introduction

The results of the PBDE analysis shall be described. First, the descriptive data for the human body burdens of the three communities are presented. The log-linear modeling data and ANOVA results to determine significant differences between communities are discussed. Lastly, the descriptive data for the animal tissue are also presented.

5.2 James Bay Community Body Burdens

This section details the results of the laboratory and statistical analysis for the human blood plasma samples.

5.2.1 Descriptives

Congener descriptive statistics by location are shown in Table 2, while, descriptives for ΣBDEs are shown in Table 3. The ΣBDEs ranged from not detectable (ND) to 29.93 ng/g lipid with geometric means being highest for Oujé-Bougoumou (3.5n g/g lipid) and lowest for Peawanuck (1.20 ng/g lipid), with the levels for Fort Albany being in between (2.28 ng/g lipid) the other two locations (Table 3, Figure 8).

Table 2. Descriptives for Sum of BDE Congeners 47, 99, 100, and 153 in ppb (ng/g lipid)

	Arithmetic Statistics					Geometric Statistics		
Location	n	Mean	SD	Minimum	Maximum	Mean	95% CI (upper)	95% CI (lower)
Oujé-Bougoumou	21	4.64	6.82	ND	29.93	3.50	5.22	2.22
Fort Albany	13	2.23	2.75	ND	9.29	2.28	3.56	1.32
Peawanuck	20	0.79	1.31	ND	5.42	1.20	1.63	0.84

ND, not detected

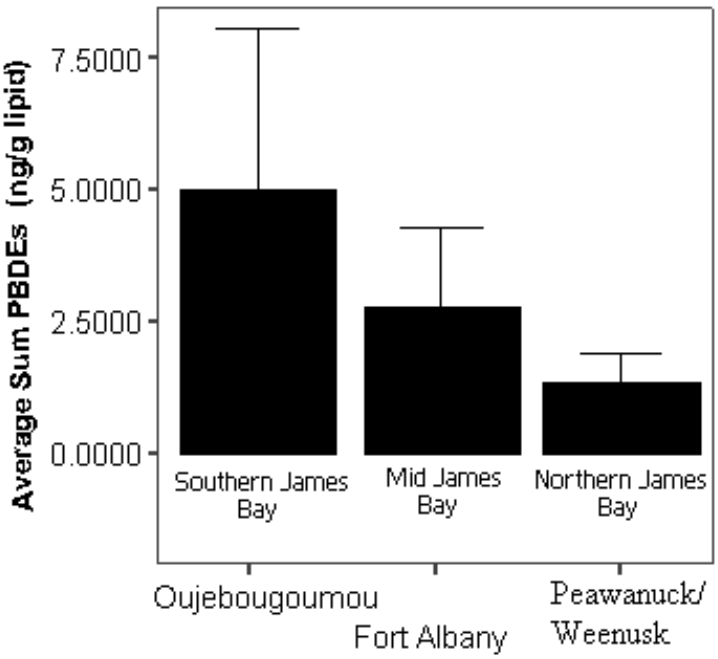

Figure 8. Sum of PBDEs of James Bay Communities (bars represent 95% CI of mean)

Table 3. Median, Minimum, and Maximum PBDE concentration in ppb (ng/g lipid)

Location	Median	Range	Frequency of Detection
Oujé-Bougoumou			
BDE-47	1.59	ND - 21.72	85.7%
BDE-99	ND	ND - 4.14	19.0%
BDE-100	ND	ND - 2.97	47.6%
BDE-153	0.44	ND - 2.30	95.2%
Fort Albany			
BDE-47	1.08	ND - 6.15	61.5%
BDE-99	ND	ND - 1.69	30.8%
BDE-100	ND	ND - 1.16	23.1%
BDE-153	ND	ND - 1.09	53.8%
Peawanuck			
BDE-47	ND	ND - 2.83	35.0%
BDE-99	ND	ND - 1.13	10.0%
BDE-100	ND	ND - 0.55	5.0%
BDE-153	ND	ND - 0.91	45.0%

ND, not detected

5.2.2 Log-linear modeling and ANOVA results

The two-state model tests for the effects of subject geographical location (Oujé-Bougoumou, Fort Albany, and Peawanuck) on the detectability (detection, non-detections) of PBDE congeners. The log-linear model contains 2 components, namely Location x Detectability (2 x 1 = 2df). The model of location on detectability provided ASRs greater than 1.96 in magnitude for Peawanuck and Oujé-Bougoumou. Specifically, significant detection was found for Oujé-Bougoumou for BDE-47 and BDE-100 (ASR = 2.96, 2.90 with df = 2 and p = 0.0039,0.0075 for PBDE-47 and BDE-100 respectively) and significant non-detection was found for Peawanuck for BDE-47 and BDE-100 (ASR = 3.02, 2.90 with df = 2 and p = 0.0039,0.0075 for PBDE-47 and BDE-100 respectively). There were no significant observations for the log linear model of BDE-99.

The four-state model tests for the effects of subject geographical location (Oujé-Bougoumou, Fort Albany, and Peawanuck) on the quartile of PBDE-153. The log-linear model contains 6 components, namely Location x Quartile (2 x 3 = 6 df). The effect of location on quartile provided a significant increase of detections in the 3rd quartile of the southern community, Oujé-Bougoumou (ASR = 2.26, p = 0.014), and 1st quartile of the northern community, Peawanuck (ASR = 2.59, p = 0.014). All values for the log linear models are presented in Table 4 (two state) and Table 5 (four state).

A one way ANOVA on the sum of BDE congeners and location with post-hoc analysis showed significant differences (p = 0.031) between Peawanuck, the most northern community of the study, and Oujé-Bougoumou, the most southern community. However, Fort Albany did not differ significantly (p > 0.463) with respect to both Oujé-Bougoumou and Peawanuck.

Table 4. Adjusted standardized residuals (ASR) from the two-state log linear model testing effect of location on frequency of detection

BDE Congener	Location	ASR Value
BDE-47	Oujé-Bougoumou	
	Not Detectable	-0.036
	Detectable	0.036
	Fort Albany	
	Not Detectable	-2.958
	Detectable	2.958
	Peawanuck	
	Not Detectable	3.019
	Detectable	-3.019
Pearson Chi-Square	11.72	
	DF = 2	
	p= 0.003	
BDE-99	Oujé-Bougoumou	
	Not Detectable	-1.305
	Detectable	1.305
	Fort Albany	
	Not Detectable	-0.080
	Detectable	0.080
	Peawanuck	
	Not Detectable	1.236
	Detectable	-1.236
Pearson Chi-Square	2.26	
	DF = 2	
	p= 0.3232	
BDE-100	Oujé-Bougoumou	
	Not Detectable	0.269
	Detectable	-0.269
	Fort Albany	
	Not Detectable	-2.902
	Detectable	2.902
	Peawanuck	
	Not Detectable	2.691
	Detectable	-2.691
Pearson Chi-Square	9.76	
	DF = 2	

Table 5. Adjusted standardized residuals (ASR) from the four-state log linear model testing effect of location on frequency of detection by quartile

BDE Congener	Location	ASR Value
BDE-153	Oujé-Bougoumou	
	1st Quartile	1.125
	2nd Quartile	-0.142
	3rd Quartile	-0.995
	4th Quartile	-0.097
	Fort Albany	
	1st Quartile	-3.553
	2nd Quartile	-0.374
	3rd Quartile	2.265
	4th Quartile	1.923
	Peawanuck	
	1st Quartile	2.591
	2nd Quartile	0.504
	3rd Quartile	-1.405
	4th Quartile	-1.855
___		
Pearson Chi-Square	15.98	
	DF = 6	
	p= 0.014	

5.3 James Bay Traditional Food Results

This section details the results of the laboratory and statistical analysis for the animal tissue samples.

5.3.1 Descriptives

Descriptives are given for wet weight and lipid-adjusted values of pintail duck (*Anas acuta*), mallard duck (*Anas platyrhynchos*), wavey goose (*Chen caerulescens*), Canada goose (*Branta canadensis*), godwit (*Limosa haemastica*), pickerel (*Stizostedion glaucum*), sturgeon (*Acipenser fulvescen*), whitefish (*Coregonus clupeaformis*), caribou (*Rangifer tarandus*), and moose (*Alces alces*) are presented in tables 4 and 5 and illustrated in figure 9 (lipid adjusted). The ΣBDEs ranged from not detectable (ND) to 6.13 ng/g wet weight with geometric means being highest for

whitefish (0.495 ng/g wet weight) and lowest for Wavey Goose (0.181 ng/g wet weight; Pickerel had no detection but only one sample was available for analysis).

Table 6. Descriptives for Sum of BDE Congeners 47, 99, 100, and 153 in ppb (ng/g wet weight)

		Arithmetic Statistics				Geometric Statistics		
Species	N	Mean	SD	Minimum	Maximum	Mean	95% CI (upper)	95% CI (lower)
Pintail Duck	20	0.40	53.7	ND	3.75	0.264	0.58	0.58
Mallard Duck	20	0.64	49.03	ND	3.02	0.459	0.87	0.87
Wavey Goose	20	0.27	23.11	ND	3.51	0.181	0.42	0.42
Canada Goose	21	0.26	25.85	ND	2.45	0.190	0.40	0.40
Godwit	19	0.29	22.43	ND	2.31	0.230	0.45	0.45
Pickerel	1	ND	-	-	-	ND	-	-
Sturgeon	12	0.31	9.49	ND	1.31	0.253	0.55	0.55
Whitefish	15	0.84	181.06	ND	6.13	0.495	1.16	1.16
Caribou	4	0.38	32.37	ND	1.52	0.287	1.62	1.62
Moose	14	0.24	89.48	ND	1.78	0.194	0.44	0.44

ND, not detected

Table 7. Descriptives for Sum of BDE Congeners 47, 99, 100, and 153 in ppb (ng/g lipid)

		Arithmetic Statistics				Geometric Statistics		
Species	N	Mean	SD	Minimum	Maximum	Mean	95% CI (upper)	95% CI (lower)
Pintail Duck	20	20.96	53.70	ND	219.92	1.88	5.95	0.03
Mallard Duck	20	29.60	49.03	ND	149.30	5.14	14.38	1.27
Wavey Goose	20	9.44	23.11	ND	88.66	1.51	4.18	0.09
Canada Goose	21	11.56	25.85	ND	89.82	1.76	4.88	0.17
Godwit	19	11.42	22.43	ND	85.65	2.66	6.88	0.56
Pickerel	1	ND	-	-	-	ND	-	-
Sturgeon	12	6.19	9.49	ND	22.80	2.03	6.43	0.07
Whitefish	15	97.33	181.06	ND	464.57	6.15	28.43	0.46
Caribou	4	16.18	32.37	ND	64.73	2.37	77.23	ND
Moose	14	37.63	89.48	ND	295.71	3.06	12.47	ND

ND, not detected

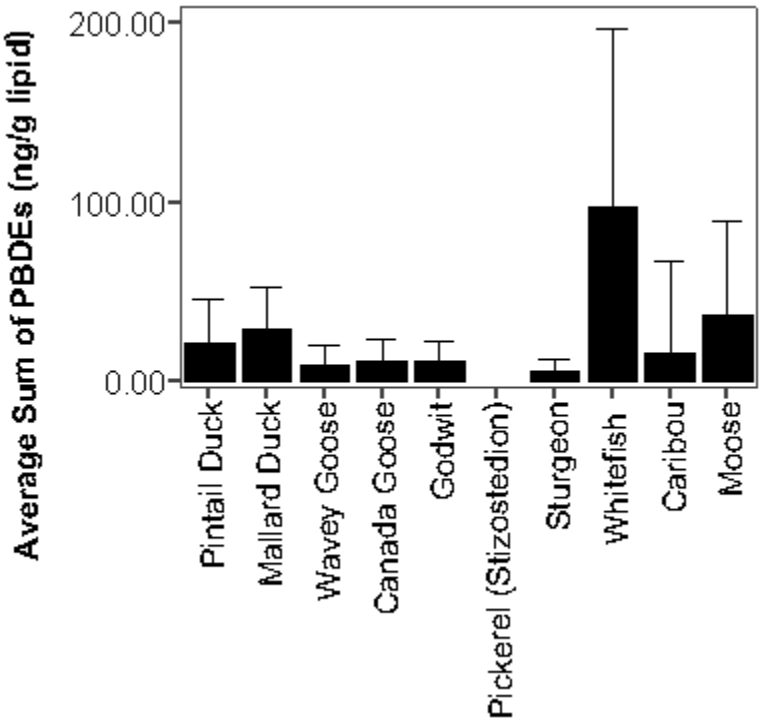

Figure 9. Average Sum of PBDEs (ng/g lipid) in traditional foods (Error bars show 95% CI of mean)

5.4 Conclusions

The ΣBDEs for humans in the three First Nation communities of the James Bay region ranged from not detectable to a maximum of 29.93 ng/g lipid. There was an unexpected significant difference between the geometric means of Peawanuck (1.20 ng/g lipid), the most northern community, and Oujé-Bougoumou (3.5 ng/g lipid), the most southern community. With regards to traditional meats, whitefish were found to contain the highest levels of PBDEs (0.84 ng/g wet weight). Of particular interest are the unexpected levels of PBDEs in the moose and caribou (grazers). The implications of these findings will be discussed in the next chapter.

Chapter 6– Discussion

6.1 Introduction

In this chapter, the concentration levels of PBDEs in the First Nation communities of the James Bay region are compared to North American levels as well as other parts of the world. A discussion on the congener profile in this study follows the global comparison. An explanation as to why the congener profile may be changing as well as how the variation between the three communities is also presented. Furthermore, the animal data from this study is compared to other wildlife throughout the world. Lastly, possible explanations for the effect of location on PBDE body burden are discussed and examples of how this study can inform future research are presented.

6.2 Comparison to Northern Canadian Levels

To date there has been little work published related to PBDEs and sub-arctic or Arctic aboriginal communities in Canada. Of the information available, in 2000 it was found that the mean sum of PBDE levels in human milk from Nunavik in Arctic Quebec, Canada, was 6.8 ng/g lipid (Pereg et al. 2003; Ryan and Van Oostdam 2004). Another northern Canadian study found levels at 23 ng/g and 11 ng/g lipid in maternal plasma and cord plasma respectively (Ryan and Van Osstdam 2004). However, since the samples were pooled it is difficult to determine individual exposure levels as this varies greatly between individuals. A pilot study by Liberda et al. (2005) found levels of PBDEs in a small sample size (n=10) to be 26.2 ng/g lipid (arithmetic mean). These levels are more than 5 times the concentration found in the same Oujé-Bougoumou cohort of this study. This variation is likely due to inter-laboratory analytical (instrumental and methodological) differences. Nevertheless, the data generated in the present study are internally consistent and comparable, since all samples were analyzed by the same laboratory at the same time. Caution is warranted with inter-laboratory comparisons especially as analytical methodology is constantly evolving. Nonetheless, it is important to put the results of the present study within the context of past studies.

Comparatively, Southern Canadian cities showed much higher levels of PBDE in milk (mean: 60 ng/g lipid, median 22 ng/g lipid) than body burden levels found in the James Bay region (Ryan and Van Oostdam 2004).

6.3 Comparison to Global Levels

Table 8 compares the results of this study to other results around the globe. In comparison to the Swedish (3.69 ng/g lipid) and Japanese (1.19 ng/g lipid) PBDE levels, the body burden of PBDEs found in the Mushkegowuk Territory are very similar (Meironyte et al. 1999; Akutus et al. 2003). This is likely because the use of PBDEs in manufacturing is relatively low in Europe and Japan and material goods containing PBDEs are not as readily available in the Canadian north compared to southern regions. The concentration values from the United States, for example, provide a wide range and typically higher level of contamination ranging from 2.08 ng/g – 199.2ng/g lipid (Schecter et al. 2003; Papke et al 2004; Mazdai et al 2003; Sojdin et al. 2001; Papke et al. 2001). This large variance is not abnormal as PBDE concentration levels can differ greatly between individuals due to varying pathways of exposure. Similarly, in southern Canada, levels of PBDEs in Ontario and Quebec (Ryan and Patry 2000) were found to be 8.25 ng/g lipid, while in British Columbia (Ryan et al. 2002) levels were a median of 49 ng/g lipid. The values of southern cities in North America are typically much higher than those of the James Bay Region – a phenomenon not typically seen with classic POPs like PCBs and DDT.

Table 8. Mean sum of PBDEs (BDE-47, BDE-99, BD-100, and BDE-153) compared globally

Year	Location	Tissue	Mean Sum PBDEs (ng/g lipid)	Reference
	Canada			
2005-2006	Oujé-Bougoumou	Plasma	4.64	Current study
2005-2006	Fort Albany	Plasma	2.23	Current study
2005-2006	Peawanuck	Plasma	0.79	Current study
2004	Oujé-Bougoumou	Plasma	26.00	Liberda et al 2005
2002	British Columbia	Milk	48.00	Ryan et al. 2002
1998-1999	Nunavut	Blood	28.30	Ryan and Van Oostdam 2004
1994-1995	Nunavut	Blood	18.60	Ryan and Van Oostdam 2004
1992	Ontario & Quebec	Milk	8.25	Ryan and Patry 2000
	USA			
2002	Texas	Milk	29.00	Schecter et al. 2003
2002	USA	Blood	197.60	Papke et al. 2004
2001	Indiana	Blood	40.80	Mazdai et al. 2003
2000	Austin & Denver	Milk	197.00	Sjodin et al., 2001
1988	Illinois	Serum	2.08	Papke et al. 2001
	Japan			
2000	Osaka	Milk	1.19	Akutus et al. 2003
1993	Osaka	Milk	0.66	Akutus et al. 2003
1983	Osaka	Milk	0.39	Akutus et al. 2003
1973	Osaka	Milk	0.03	Akutus et al. 2003
	Sweden			
1997	Sweden	Milk	3.64	Meironyte et al. 1999
1990	Sweden	Milk	1.12	Meironyte et al. 1999
1976	Sweden	Milk	0.29	Meironyte et al. 1999

6.4 A Changing Congener Profile?

Several recent studies have identified BDE-153 to be overtaking BDE-47 as the dominant congener in human samples (Thomas 2004; Weiss et al 2004; Fangstrom et al. 2004; She et al. 2007). For 16.67% of the human samples in the three James Bay communities (9 of 54), BDE-153 was found at greater levels than BDE-47. PBDEs as a whole were detected for 77.8% of all samples; however, BDE-153 was found to be the most detected congener (66.7%) when compared to BDE-47, BDE-99, and BDE-100 (individual congener detection is presented by location in table 3). The implications of this change in congener profile are unknown but it may be suggestive of a change in source, pathway, or the result of long term metabolism or half-lives of congeners in biota (She et al. 2007).

6.5 Latitudinal Variation of Human Blood Plasma Results

The results of Hassanin et al. (2004) indicate that BDEs as a whole, and more specifically, BDEs with a molecular weight greater than BDE-47 are preferentially deposited between latitudes of 60^{o}-50^{o}. This could help explain why BDE-153, the heaviest of the three congeners analyzed was detected more so than BDE-47 – especially in the lower latitude community of Oujé-Bougoumou ($p<0.05$). The results of this study demonstrate that the sum of BDEs was found highest in Oujé-Bougoumou, the most southern of the three First Nation communities ($p < 0.05$). Conversely, the most northern community, Peawanuck, had the lowest levels of BDEs body burden ($p < 0.05$).

The results of the log linear model demonstrate significant ($p < 0.05$) differences between BDE-47 and BDE-100 in that higher latitude communities had lower levels than the lower latitude communities. The four-state log linear model showed that BDE-153 had significantly ($p < 0.05$) more detections in the 3rd quartile in Oujé-Bougoumou and significantly more detections in the 1st quartile of Peawanuck. This means that the individuals in the Oujé-Bougoumou cohort had significantly more detections and higher concentrations of BDE-153 than the other First Nation communities. Conversely, Peawanuck had significantly less detections and lower concentrations of BDE-153 than the other First Nation communities.

6.6 Comparison of Animal Data

In order to properly compare PBDE concentrations between studies, it would have been prudent for other studies to report wet weight concentration as well as lipid adjusted data (only two studies reported wet weights). Due to this, it is difficult to compare concentrations between differing studies and lipid adjusted values must be compared regardless of the type of tissue analyzed.

Table 9 presents the mean sum of PBDEs (or their ranges if mean data is unavailable) for several wildlife studies across the globe. It is apparent that birds of prey such as the falcon (*Falco*

tinnunculus) and fish such as salmon (e.g., coho salmon (*Oncorhynchus kisutch*)) who prey on lower trophic levels contain higher levels of PBDEs than, for example, grazers (e.g., reindeer). The caribou and moose in this study (16.18 ng/g lipid and 37.63 ng/g lipid) contain levels much higher than those found in a study by Selstrom (0.5-1.7 ng/g lipid; in Law et al. 2002). The mean whitefish PBDE contamination level in this study (97.22 ng/g lipid) was lower than all other studies except for carp (*Cyprinus capio*) from the Detroit river (40.7 ng/g lipid). As a whole, the birds and fish of the studies in table 9 typically contain mean values or ranges that greatly exceed those found in this study.

Of particular concern are the high levels of PBDEs found in leeches (*Haemopsis spp.)* (data not published, n = 30). Mean levels of 378.8 ng/g wet weight were found in 30 leech samples and therefore may bioaccumulate through the food chain. Combining the results of this study with consumption data can give direct comparison to acceptable daily intakes (ADIs) and tolerable daily intakes (TDIs) developed by federal bodies such as Health Canada and international bodies such as the World Health Organization (WHO).

Table 9. Comparison of mean sum of PBDEs in wildlife

Species	Year	Location	Tissue	Mean Sum PBDEs (ng/g lipid)*	Reference
Birds					
Cormorants (*Phalocrocorax carbo*)	1996-1997, 1999-2000	England and Wales	Liver	1.8-140 ng/g†	Law et al. 2002
Starlings(*Poeoptera kenricki*)	1999	Sweden	Breast	5.7-13 ng/g	Sellstrom 1999 (in Law et al. 2003)
Gyrfalcons (*Falco rusticolus*)	1999	Baltic Sea	Egg	18 ng/g	Herzke et al. 2001
Sparrowhawk (*Accipiter nisus*)	1999	Baltic Sea	Egg	732 ng/g	Herzke et al. 2001
Falcon (*Falco tinnunculus*)	1999	Sweden	Egg	25-3800 ng/g	Sellstrom 2001(in Law et al. 2003)
Glaucous gulls (*Larus hyperboreus*)	1999	Bear Island (Arctic Ocean)	Liver	0.5-22 ng/g	Herzke et al. 2003
Herring gull (*Larus argentatus*)	1981 - 2000	Great Lakes	Egg	183 – 16,500 ng/g	Norstrom et al. 2002
Mammal					
Moose and reindeer	1999	Sweden	Breast	0.5-1.7 ng/g	Sellstrom 1999 (in Law et al. 2003)
Beluga whale (*Delphinapterus leucas)*	1989-2001	Canadian Arctic	Blubber	14.6 ng/g‡	Stern and Ikonomou (in Law et al. 2003)
Ringed seal (*Phoca Hispida*	2000	Holman Island (NWT)	Blubber	4.6 ng/g	Ikonomou et al. 2000

(Continued on next page)

* Range displayed if mean not provided
†Wet weight basis
‡ Wet weight basis

Fish (continued from previous page)					
Carp (*Cyprinus carpio*)	1999	Detroit River	Whole	40.7 ng/g	Rice et al. 2002
Large Mouth Bass (*Micropterus salmoides*)	1999	Detroit River	Whole	163 ng/g	Rice et al. 2002
Carp (*Cyprinus carpio*)	-	Great Lakes	-	2400 ng/g	Dodder et al. 2002
Chinook salmon (*Oncorhynchus tshawytscha*)	-	Lake Michigan	-	773 ng/g	Manchester-Neesvig et al. 2001
Coho salmon (*Oncorhynchus kisutch*)	-	Lake Michigan	-	8120 ng/g	Manchester-Neesvig et al. 2001

6.7 Accessibility and a Subsistence Diet

The effect of the sum of PBDE body burden on location may also be due to the accessibility of obtaining products which contain the PBDE flame retardant as well as a contribution from a subsistence diet. Oujé-Bougoumou is the only community that is accessible year round by road; Fort Albany and Peawanuck are fly-in communities. Hence, material goods containing PBDEs may be more readily available due to ease in transportation when compared to goods available in more isolated communities. It is therefore possible that accessibility of goods containing PBDE flame retardants may play a role in PBDE body burden from household and workplace dust - especially in the newly built community of Oujé-Bougoumou.

Further, traditional foods consumed by the Mushkegowuk Cree are contaminated with PBDEs at levels higher than those found in the human study participants themselves. Therefore, it appears that dietary exposure is not the main source of contamination since evidence of total PBDE body burden in the study group illustrates that humans contain lower levels than the foods they consume. However, the animal data collected should be combined with consumption data in order to properly assess dietary risk of PBDEs and compare the results to acceptable daily intakes (ADIs) and tolerable daily intakes (TDIs).

Accessibility of products containing the PBDE flame retardants may be the primary exposure pathway to human PBDE body burden. However, it is important not to completely rule out dietary intake from a subsistence lifestyle. For example, Moose and Caribou (herbivores) contain PBDE contaminants that could only have entered their body by long-range deposition and eating and foraging in contaminated areas. Therefore, these animals still have the potential to contribute to human body burden if consumed.

6.8 Future Recommendations

A dietary survey of traditional foods should be conducted for all seasons in order to assess the risk associated with the ingestion of animals contaminated with PBDEs. Currently, the levels of

PBDEs found in the James Bay region for First Nation people are lower than most of North America. However, a biomonitoring program should be set in place in order to observe any changes in contaminant load as the James Bay region is a dumping ground for PBDEs and the concentration in the environment may increase (Hassanin et al. 2004). An educational program should be created which educates the communities of the James Bay region of the danger of PBDE contaminants and informs residents which species should be eaten less frequently in order to avoid any adverse health effects. Lastly, household and workplace dust levels should be analyzed in the James Bay communities to determine if this is a major pathway of exposure.

Chapter 7 – Conclusions

This study measured the concentration and latitudinal variations of PBDEs in human plasma of the James Bay region. The biological monitoring of PBDEs via the methods used in this study is a measure of internal chemical dose which provides a more accurate assessment of health risk. However, careful interpretation of the results is needed and knowledge of toxicokinetic parameters (i.e., absorption, distribution, metabolism, and elimination) is essential, in order to completely characterize this emerging pollutant. The background literature was used to inform the study of critical health effects associated with PBDE exposure, determine levels and trends of PBDEs in humans around the world, as well as determine the current status of PBDE production in North America. This was important to the study because PBDE concentrations in humans and animals without context would only result in a meaningless exposure value. The literature provided background such that the PBDE concentrations could be compared to other studies.

Several examples of how the ecosystem approach to human health had been used for other studies were presented in order to inform the link between the biophysical environment, human health, and environmental contaminants. It is necessary to take such an approach because conventional approaches do not look beyond the exposure values themselves and can miss alternate sources of exposure. Additionally, by designing the study to inform the link between human health and the environment, a transparent process for which the results can be used and evaluated by a variety of other researchers as well as First Nation communities was created. Further, the opportunity to use this data in order to inform other aspects of the ecosystem approach is also available (e.g., the link between economics, human health, and the biophysical environment).

The results of this study (body burden and frequency of detection) illustrate a significant difference between PBDE body burden in Oujé-Bougoumou (lowest latitude) compared to Peawanuck (highest latitude). Findings suggests that heavier PBDE congeners are being deposited at lower latitudes than lighter congeners, and that there may be more sources of

PBDEs exposure, such as furniture or other plastic materials, in the lower latitude communities due to issues of accessibility. Body burden results also suggest that wild game and fish do not constitute a major source of PBDE exposure as Peawanuck eats the most wild meats and fish, followed by Fort Albany, and the Oujé-Bougoumou; while, Peawanuck's plasma levels are the lowest, followed by Fort Albnay with Oujé-Bougoumou having the highest PBDE plasma levels. Data are limited for PBDEs in traditional foods and foods in general. Since, there are no guidelines for safe levels of PBDE in meats targeted for human consumption, it is difficult to put wild meat and fish data in perspective.

The exposure pathway for PBDE body burden in the sub-arctic region is still largely unknown, but a subsistence diet may contribute, in small part, to the human body burden found in the First Nation communities. By using an interdisciplinary framework, other sources of potential exposure such as household dust were able to be identified. This is important because the biomonitoring framework used for this study can be expanded to include the new source of exposure. The presence of PBDEs in sub-arctic regions demonstrates the need for a monitoring program in order to track changes in exposure (i.e., dietary). The need for monitoring in sub-arctic regions is particularly important because these areas have been identified as sinks for PBDEs.

The PBDE body burden is lower in the Mushkegowuk territory communities when compared to more northern and southern communities in Canada. While the sub-arctic has been identified as an environmental sink for PBDEs, the levels found in this study indicate that long-range transportation of the contaminant may not be a significant exposure pathway yet. This may be due to low bioavailability of the contaminant, rapid metabolism, or alternate sources of exposure and may change in the future.

Similarly, PBDE body burden in the US is significantly higher than those of the James Bay region. The body burdens of the James Bay Cree are similar to Japanese and European levels. The traditional food data show relatively elevated levels of PBDEs in certain species such as

whitefish, mallard duck, and moose; thus, caution should be advised when consuming these foods as the full toxicological effects of PBDEs is still being characterized. However, current human PBDE levels of the James Bay region are below traditional food meat levels and well below other Canadian studies which suggests minimal input from the traditional diet.

The data generated by this study can be used to continue and further human biomonitoring programs in the James Bay area. Additionally, the animal data can be combined with First Nation peoples' consumption data in order to assess the contribution of PBDEs from a First Nation diet to a greater and more accurate extent. The results of this study illustrate the importance of monitoring human body burden as well as foods which are often consumed; especially in communities that rely on a subsistence diet. Biomonitoring studies such as these can identify possible causes of concern (e.g., species with high contamination load) and work towards reducing human body burden by implementing appropriate policies and educational campaigns thus feeding back into the ecosystem approach to human health.

In conclusion, the results of this study reveal that First Nation PBDE levels are lower than the national Canadian average and much lower than American levels. This is surprising given that classic POPs such as PCBs and DDT are typically found at higher levels in First Nation communities that practice a subsistence diet when compared to other non-subsistence communities. The levels of PBDEs in traditional foods does not appear to be a major source of PBDEs in that Peawanuck has the most traditional diet of the three communities studies but the lowest plasma concentrations; while, Oujé-Bougoumou has the highest PBDE levels but eats the least traditional foods. Nonetheless, given the long-range transportation and deposition patterns of PBDEs, the levels in the sub-arctic environment, animals, and humans is expected to rise in the future. Therefore, a long-term biomonitoring plan that incorporates traditional food meats and other important wildlife should be developed in order to safeguard human health and identify possible threats to a subsistence diet.

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