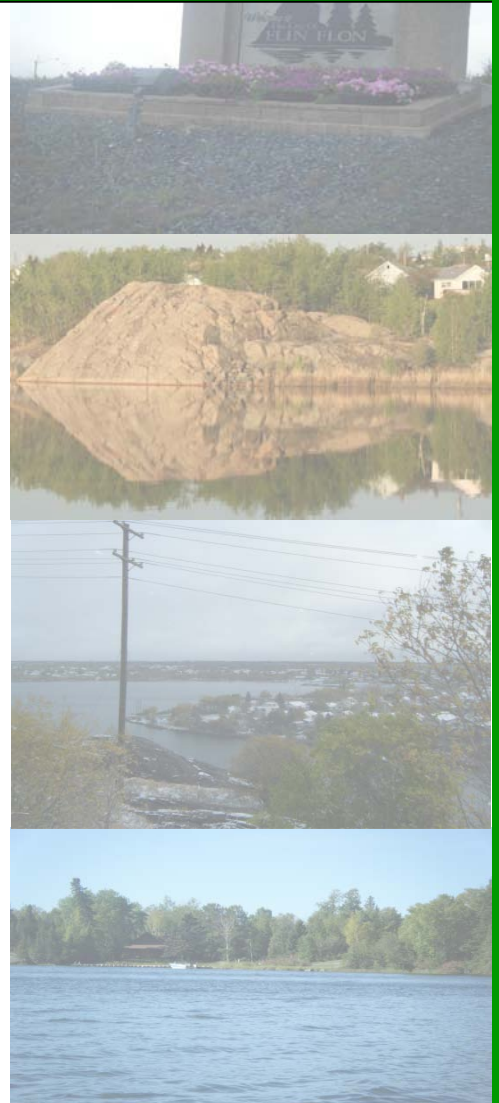# CHAPTER 2

LITERATURE REVIEW, DATA GAP ANALYSIS
AND SUPPLEMENTAL SAMPLING

CHAPTER 2:

LITERATURE REVIEW, DATA GAP ANALYSIS AND SUPPLEMENTAL SAMPLING

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2.0 LITERATURE REVIEW, DATA GAP ANALYSIS AND SUPPLEMENTAL SAMPLING

The purpose of this phase of the HHRA was to collect the necessary data to complete a review of all available primary scientific literature, reports prepared and data collected by government agencies, and information provided by HBMS. Information gathered as part of this exercise was used to determine if additional sampling and analysis was needed to adequately assess exposure and risk to people in the affected areas.

This section provides a summary of the literature reviewed as well as a data gap analysis for information relevant to the HHRA. Sections 2.1 through 2.8 describe information gathered describing metal content in soil, ambient air, home garden produce, drinking water, fish and sediment, surface water, and local blueberries, respectively. Section 2.9 discusses the bioaccessibility study conducted as part of the HHRA and Section 2.10 summarizes the food consumption survey conducted as part of the study. Appendix C provides individual summaries of the resources reviewed. This chapter also describes the supplemental data collected as part of the HHRA study.

2.1 Soil and Dust Data

2.1.1 Summary of Review

Numerous studies have been conducted by government agencies and independent researchers over the past 20 years or more in which concentrations of metals in soils in the Flin Flon area have been analyzed. Results of these studies have generally been in agreement that concentrations of several metals are elevated in soils surrounding the HBMS complex. In addition, a strong positive inter-correlation has been noted by several sources indicating that these metals share a common point of origin. The estimated distribution of contamination has varied among studies. Studies analyzing metal content in bogs and fens found elevated concentrations at distances as far as 100 km south-southeast from the smelter (Zoltai, 1988). Other studies involving the analysis of rain and snow found that the distribution of contamination was chemical specific, with distances ranging from a 33 to 217 km radius from the smelter such that the distribution of zinc exceeded that of lead, which in turn exceeded the distributions of arsenic and copper (Franzin *et al.*, 1979). McMartin *et al.* (1999) found that the distance from the smelter in which concentrations returned to regional background concentrations varied for each metal but averaged 70 km for cadmium, 76 km for lead, 84 km for zinc, 85 km for mercury, 90 km for copper, and 104 km for arsenic.

A large regional geochemical mapping project was conducted by the Geological Survey of Canada and Manitoba Energy and Mines studied soils in Flin Flon, central Manitoba, and Saskatchewan and has served as the basis for several additional soil studies (Henderson, 1995; McMartin *et al.*, 1996). Humus and till were sampled at over 1,600 sites within a 200 km radius of the HBMS smelter in Flin Flon. Samples were taken at an average spacing of 4 km, with a total of 1,817 samples of till and 1,639 samples of humus collected (McMartin *et al.*, 1999). Concentrations of metals were reported to vary significantly depending on the depth at which samples were collected. Several studies have found that concentrations are significantly higher in the upper soil layer composed of a large amount of organic material, commonly referred to as humus, relative to the underlying till layer (Henderson and McMartin, 1995; Henderson *et al.*, 1998; McMartin *et al.*, 1999; and, Manitoba Conservation, 2003).

Although the concentrations of metals in humus in the Flin Flon area are influenced by the smelter, concentrations in till can be heavily influenced by the composition and nature of the underlying bedrock and are modified by glacial erosion, transport, and deposition (McMartin *et al.*, 1996). Generally, concentrations of all smelter-related metals are higher in humus than in the underlying till. Concentrations in till are often not directly related to distance from the smelter and are generally less variable than concentrations in humus. Concentrations of mercury in till were found to be relatively constant regardless of distance (Henderson and McMartin, 1995). Overall, contamination of smelter-related metals is generally restricted to the organic surface horizon (humus) and concentrations are poorly correlated with concentrations in the underlying till. However, in close proximity to the smelter (<10 km), migration through the soil profile is significant and elevated concentrations are found to a minimum depth of 45 cm.

Manitoba Conservation has completed four significant soils studies in the Flin Flon-Creighton area. In 2000, as part of a study in which concentrations of several metals were measured in blueberries collected from a number of locations in the Flin Flon area, soil concentrations from the sampling locations were also measured. Three soil samples, at depths ranging from 1.5 to 7 cm bgs (below ground surface), were taken from each of 13 sites; 11 from the Flin Flon area, and two others likely representing regional background concentrations. Easting and northing coordinates were provided for each location (Manitoba Conservation, 2000). Since these sampling locations were not considered to be representative of areas in which people will spend prolonged durations, this data was not used to predict exposure to receptors in the HHRA.

During the summers of 1998, 1999, and 2003, Manitoba Conservation conducted a soils study for forest sites in the Flin Flon area (Manitoba Conservation, 2003). A total of 16 samples were taken at each of seven locations in Flin Flon and one reference location, at depths ranging from 0 to 15 cm bgs. The first round of sampling at these locations occurred during the summers of 1998 and 1999. These locations were re-sampled during the summer of 2003. As found in Henderson and McMartin (1995) and McMartin *et al.* (1999), the highest measured concentrations of smelter-related metals were generally found in the upper 0 to 6 cm within the organic layer of soil. As with the results of the blueberry study, sampling locations from this study were not considered to be representative of areas in which people will spend prolonged durations. However, the results of the 2003 study were used to represent concentrations of COC that wild game would be exposed to *via* incidental ingestion within calculations to predict concentrations in wild game meat consumed by residents (refer to Appendix O for a detailed explanation of the methodology used to predict concentrations of COC in wild game tissues).

In 2002, Manitoba Conservation designed and completed a home garden sampling program to address public concerns raised in response to discrepancies reported in the home garden surveys completed by Pip (1991) and HBMS (1994a). Nine home gardens from the Flin Flon area, located at varying distances and directions from the HBMS complex, were selected to characterize the potential influence of smelter-related emissions (Jones and Henderson, 2006). In addition, a garden located in the community of Cranberry Portage was selected to be representative of an area that is minimally impacted by smelter emissions. A garden in the town of The Pas was selected to represent a non-impacted control site. Three replicate soil samples were collected from each of eleven garden plots. Samples were collected from a depth of 10 cm.

Finally, Manitoba Conservation completed a surface soil sampling program in August, 2006 which involved the collection of soil from 93 sites in Flin Flon and 13 sites in Creighton (Manitoba Conservation, 2007). In addition, to characterize areas that are believed to be minimally impacted and non-impacted, samples were collected from Bakers Narrows Provincial Park and Cranberry Portage, respectively. Samples were collected from the top 2.5 cm of soil

within publicly accessible lands such as boulevards, parks, playgrounds, schoolyards, vacant lots and undeveloped areas but did not focus on residential properties.

The results of this study indicated that concentrations of the following twelve chemicals were found to be elevated relative to concentrations measured in the Cranberry Portage reference area:

- Antimony;
- Arsenic;
- Cadmium;
- Copper;
- Lead;
- Mercury;
- Molybdenum;
- Selenium;
- Silver;
- Sulphur;
- Thallium; and
- Zinc.

In addition to the finding that concentrations of these compounds were elevated across much of the study area, they were also strongly inter-correlated and are known to be past or present constituents of the HBMS smelter emissions. When compared to the CCME soil quality guidelines and interim remediation criteria for soils under a residential or parkland scenario, concentrations of arsenic, cadmium, copper, lead, mercury, selenium, and thallium all exceeded their respective guidelines. Although concentrations of zinc exceeded the CCME guideline, this guideline is protective of environmental health and does not contain a human health component. Concentrations of antimony, molybdenum, silver and sulphur did not exceed their corresponding interim remediation criterion.

For comparative purposes, the maximum concentrations of the eleven metals discussed above were identified from each of the Manitoba Conservation studies and compared to the CCME Human Health Guideline for residential/parkland properties (Table 2-1).

Table 2-1	Comparison of Maximum Soil Concentrations Measured During Four Separate Manitoba Conservation Studies (µg/g)				
Metal	*2006 Soils Study[b]*	*2000 Blueberry Study*	*2002 Home Garden Study*	*1998/1999/2003 Forest Soils Study*	*CCME HH Guideline*
Antimony	11.2	-	<10	8.6	20[a]
Arsenic	**407**	**210**	**50.4**	**210**	12
Cadmium	**70.9**	**125**	11	**85**	14
Copper	**5103**	**5,060**	747	**3,010**	1,100
Lead	**1,446.7**	**2,500**	**474**	**1,740**	140
Mercury	**653**	**44.3**	**25.9**	**30.6**	6.6
Molybdenum	8	<10	<4	9.8	10[a]
Selenium	**177.2**	-	6.3	25.2	28
Silver	7.5	-	1.8	3.2	20[a]
Thallium	**1.2**	**<10**	0.3	**1.3**	1.0
Zinc	11,150	24,700	3,120	15,300	NA

[a]   Value represents the CCME interim remediation criterion.
[b]   Value represents the average of three samples taken per location
Bold Values are in excess of the CCME Human Health Guideline for residential and parkland properties.
NA Not available

Although the maximum concentrations for some chemicals were higher in studies other than those reported in the 2006 soils study, inclusion of data from these studies would not result in the retention of additional COC for the HHRA. This comparison lends support to the use of the 2006 soils study as the primary source of Manitoba Conservation soil data for the preparation of the Problem Formulation and Terms of Reference for the HHRA.

2.1.2   Data Gap Analysis

The absence of soil samples taken from the front and backyards of residential properties was identified as a potential data gap. Although the Manitoba Conservation soils study had effectively characterized concentrations in public areas, an HHRA will commonly assume that chronic exposure events, for children in particular, will occur at the home. Exposure to metals in soil *via* incidental ingestion and dermal contact is most accurately characterized using values measured from children's play areas on residential properties. In addition, children and adults spend a significant amount of time indoors, particularly within their primary residence. This is particularly true for populations living within northern environments, which are subject to long, cold winters. As a result, exposure to COC within house dust is an important pathway to accurately assess. Although concentrations in indoor dust can be correlated to concentrations in outdoor soil and ambient air using generic assumptions, it is preferred to use actual measured concentrations collected from various hard and soft surfaces within homes.

2.1.3   Supplemental Soil and Indoor Dust Sampling Program

To aid in the completion of the HHRA, HBMS contracted Jacques Whitford to complete an additional soil sampling program in fall of 2007. This program was focused on characterizing metal content in outdoor soil and indoor dust on residential properties. This program is described in Chapter 3, Section 3.2.1.2 and detailed in Appendix B.

In addition, to address risks associated with exposure to chemicals in indoor dust, the Jacques Whitford study also involved the sampling and analysis of dust samples from homes and schools.

2.1.4   Indoor Dust Sampling Program

Ultimately, airborne particulates inside homes tend to settle out and become part of the dust that is found on furniture and floors/carpets leading to potential exposure pathways *via* dermal contact and ingestion. Sampling involved collections from both soft and hard surfaces as described below. Participants were requested not to vacuum two weeks prior to the sampling program.

Fabric Surfaces

Dust samples within the houses were collected (depending on presence in house) from carpeted floor areas at:

- The centre of the most frequently used play area for children under the age of six;
- The main entrance used for access and egress from the house;
- The secondary, less heavily used entrance to the house; and,
- The main hallway of the house and evident route of high traffic flow; and from two items of upholstered furniture such as:
    - A regularly used chesterfield; and,
    - An easy chair.

Up to five samples were recovered from fabric surfaces in each residence. Every attempt was made to recover the same number of samples from each residence. Where variations existed (*i.e.*, no carpet), additional samples were recovered from other surfaces deemed similar at the time of sampling. If there were no carpets, area rugs were sampled. If an insufficient number of surfaces existed for soft surface samples, additional hard surface samples were collected. Samples were collected where the most sensitive receptor spends the largest portion of their active time.

Hard Surfaces

Dust samples within the houses were collected from hard surfaced areas including a commonly contacted portion of:

- The kitchen tiled floor; and,
- The sill of a window commonly accessed and most likely to be contacted by a child (likely to be in the main living area).

A total of two samples (one from a window sill and one from a tiled floor) were recovered from hard surfaces in each residence. Every attempt was made to recover the same number of samples from each residence. Samples were collected in the main living area where the most sensitive receptor is likely to spend the largest portion of their active time.

Tenant Questionnaire

The occupants were asked to complete a questionnaire as part of the dust sampling program. The questionnaire administered to the occupants queried the number of people and their ages in the home; activities/hobbies undertaken in the home; characteristics of the home *etc*.

Lead-in-paint Sampling

Based on the historical use of lead in paints and the age of the community, samples of paint were taken from the interior of the homes selected for the indoor dust assessment. The data from the lead paint assessment was combined with the indoor dust sampling results for use in the HHRA.

2.1.4.1   Supplemental Dust Sampling Results

The results of the supplemental dust study are discussed in Chapter 4, Section 4.1.1.7 and detailed in Appendix D.

2.2   Air Data

2.2.1   Summary of Review

HBMS currently operates air sampling stations at Ruth Betts School in Flin Flon and at Creighton School in Creighton. HBMS historically operated an additional monitor at the Sewage Treatment Plant in Flin Flon but monitoring at this location was discontinued in 2002. HBMS reports data for TSP, PM_{10}, and $PM_{2.5}$ and metals associated with each of these fractions every six days. Manitoba Conservation operates a monitor on the Provincial building and analyzes for TSP and metals associated with TSP. Based on annual average concentrations of arsenic, cadmium, copper, lead, and zinc from 1998 to 2006, concentrations have generally declined or remained constant over this period (Table 2-2).

Table 2-2	Annual Average Metal Concentrations Associated with PM_{10} in Ambient Air (µg/m³)				
Location	*Lead*	*Copper*	*Cadmium*	*Zinc*	*Arsenic*
Ruth Betts School					
1998					
# samples	10	10	10	10	10
Maximum	3.1	1.0	0.32	2.8	0.32
Average	**0.81**	**0.42**	**0.11**	**0.98**	**0.091**
1999					
# samples	12	12	12	12	12
Maximum	3.5	0.94	0.33	3.4	0.099
Average	**0.60**	**0.23**	**0.064**	**0.60**	**0.042**
2000					
# samples	9	9	9	9	9
Maximum	1.9	0.53	0.28	2.8	0.17
Average	**0.41**	**0.17**	**0.046**	**0.63**	**0.044**
2001					
# samples	12	12	12	12	12
Maximum	0.60	1.5	0.19	3.3	0.13
Average	**0.24**	**0.50**	**0.060**	**0.94**	**0.048**
2002					
# samples	19	19	19	19	16
Maximum	0.38	1.0	0.16	2.2	0.071
Average	**0.15**	**0.35**	**0.024**	**0.56**	**0.023**
2003					
# samples	21	21	21	21	21
Maximum	0.47	1.0	0.083	2.8	0.20
Average	**0.16**	**0.33**	**0.020**	**0.78**	**0.043**
2004					
# samples	58	58	58	58	58
Maximum	0.15	1.2	0.024	4.2	0.077
Average	**0.033**	**0.13**	**0.0050**	**0.76**	**0.011**
2005					
# samples	37	37	38	37	37
Maximum	0.34	1.9	0.062	2.5	0.073
Average	**0.067**	**0.18**	**0.011**	**0.37**	**0.016**
2006					
# samples	46	49	49	49	49
Maximum	0.66	0.73	0.21	1.3	0.21
Average	**0.059**	**0.13**	**0.017**	**0.13**	**0.020**
Creighton School[a]					
2003					
# samples	210	210	210	210	210
Maximum	0.52	0.56	0.06	1.4	0.07
Average	**0.42**	**0.15**	**0.04**	**0.24**	**0.01**
2004					
# samples	340	340	340	340	340
Maximum	0.84	0.78	0.12	1.9	0.23
Average	**0.42**	**0.13**	**0.04**	**0.23**	**0.01**
2005					
# samples	354	354	354	354	354
Maximum	1.0	1.2	0.83	5.0	0.11
Average	**0.42**	**0.13**	**0.06**	**0.36**	**0.01**
2006					
# samples	346	346	346	346	346
Maximum	0.33	0.85	0.12	1.4	0.17
Average	**0.17**	**0.05**	**0.01**	**0.09**	**0.01**

a   Measurements are from the dichotomous air sampler.

Annual average concentrations of lead from the Creighton School are notably higher than those reported at Ruth Betts. While this may be an accurate reflection of differences in conditions resulting from wind patterns or the potential influence of the Tailings Management Area located to the north of Creighton, it may also be skewed by differences in air samplers used at these locations. In 2003, when a dichotomous air sampler replaced the previous air sampler located at the Creighton School, the concentrations of metals associated with the PM_{10} component increased significantly. According to HBMS, 2007 (pers. comm.), the dichotomous sampler uses an 8 cm filter that retains less particulates than the 10 cm filter used previously at the Creighton School and at Ruth Betts. The reported increase in concentrations of certain metals associated with the PM_{10} component may be the result of elevated detection limits that are used to analyze for metals in the smaller samples collected by the dichotomous sampler. As a result, it may not be appropriate to directly compare concentrations collected from the Ruth Betts and Creighton schools.

2.2.2 Data Gap Analysis

The air samples collected at Ruth Betts and the Creighton School have historically only been analyzed for five (arsenic, copper, cadmium, lead, and zinc) of the 11 metals reported to be elevated in soils in the Flin Flon area. As a result, based on discussions with Stephen West and Joel Nilsen of HBMS on August 27, 2007, HBMS agreed to add the remaining six metals (antimony, mercury, molybdenum, selenium, silver, and thallium) to the list of chemicals analyzed for at each location. Since the 8 cm filter used within the dichotomous monitor at the Creighton School would not likely retain enough particulates to allow for analysis of these additional metals (particularly mercury), HBMS indicated that they would operate an additional 10 cm monitor at this location to collect additional particulates. Sampling occured every three days.

Based on a comparison of metals associated with TSP at the Provincial Building, Ruth Betts School, and Creighton School, concentrations of arsenic, cadmium, copper, lead, and zinc are all highest at the Provincial Building (Table 2-3). Samples collected at the Provincial Building were considered to most accurately represent ambient air concentrations that receptors living within West Flin Flon would be exposed. Since the assessment of exposure *via* the inhalation pathway considered concentrations of metals associated with the PM_{10} component, correlating factors were used to estimate concentrations associated with PM_{10} based on measurements of metals in TSP. This is discussed in detail in Chapter 4 and Appendix I.

Table 2-3	Annual Average Metal Concentrations Associated with TSP in Ambient Air ($\mu g/m^3$)					
Location	*Lead*	*Copper*	*Cadmium*	*Zinc*	*Arsenic*	*Mercury*
Provincial Building						
1998	0.41	1.0	0.051	1.8	0.080	-
1999	0.48	0.68	0.051	1.5	0.066	-
2000	0.36	0.72	0.050	1.8	0.043	-
2001	0.26	0.91	0.048	2.7	0.051	-
2002	0.28	0.98	0.045	2.3	0.043	-
2003	0.22	1.0	0.032	1.8	0.054	-
2004	0.11	0.98	0.015	2.4	0.038	-
2005	0.18	0.89	0.022	1.6	0.043	-
2006	0.14	0.94	0.041	2.2	0.049	-
2007[a]	0.26	1.4	0.054	1.3	0.081	-
Ruth Betts School						
1998	0.65	0.98	0.077	1.36	0.067	-
1999	0.54	0.90	0.060	1.0	0.051	0.00049
2000	0.43	0.96	0.049	1.2	0.058	0.00043

Table 2-3	Annual Average Metal Concentrations Associated with TSP in Ambient Air (µg/m³)					
Location	*Lead*	*Copper*	*Cadmium*	*Zinc*	*Arsenic*	*Mercury*
2001	0.21	1.0	0.040	1.4	0.055	0.00080
2002	0.15	0.69	0.019	0.85	0.037	0.00056
2003	0.14	0.80	0.018	1.2	0.037	0.00079
2004	0.040	0.58	0.006	0.90	0.016	0.00076
2005	0.089	0.76	0.014	0.60	0.023	0.00056
2006	0.059	0.74	0.020	0.29	0.024	0.00050
2007[a]	0.086	0.87	0.025	0.59	0.044	0.00028
Creighton School						
1998	0.079	0.20	0.010	0.21	0.013	-
1999	0.10	0.37	0.010	0.27	0.014	-
2000	0.17	0.37	0.021	0.43	0.048	-
2001	0.091	0.65	0.013	0.52	0.032	-
2002	0.17	0.56	0.0055	0.77	0.15	-
2003	0.036	0.65	0.0041	0.43	0.0087	0.00032
2004	0.028	0.50	0.0034	0.47	0.0061	0.00035
2005	0.039	0.42	0.0067	0.33	0.011	0.00036
2006	0.024	0.45	0.0045	0.14	0.0078	0.00012
2007[a]	0.027	0.42	0.0048	0.076	0.0090	0.000083

[a]   Concentrations are the average for data sampled from January 1 to May 31 2007.
Note: From January, 1998 to mid-2005, samples from Ruth Betts, Sewage Plant, and Creighton School were only analyzed when wind direction was from the facility.

2.2.3   *Supplemental Air Sampling*

A summary of ambient air concentrations utilized in the HHRA, including the supplemental data collected by HBMS, is provide in Chapter 4, Section 4.1.1.2.

2.3   **Home Garden Data**

2.3.1   *Summary of Review*

Three primary studies have measured concentrations of metals in home garden soil and produce. The first was completed by Pip (1991) in which soil and vegetable samples were collected from home gardens at the end of the 1989 growing season at 12 locations that ranged in distance from 0.29 to 12.8 km northeast to southwest of the HBMS complex. Concentrations of cadmium, copper, and lead were measured in soil and produce. Overall, a strong intercorrelation was found for all metals in soil and although a significant level of variation was noted for concentrations of metals in produce, when concentrations in all produce were pooled (untransformed data), a significant inverse relationship was noted with distance from the smelter (log transformed data). Leafy tissues had significantly higher concentrations than fruits and below ground tissues. However, concentrations in leafy tissues were not significantly correlated with concentrations in soil. This was attributed largely to aerial deposition and the large surface area to volume ratio of leafy tissues (Pip, 1991).

In 1994, HBMS completed their own home garden survey by collecting soil and vegetable samples from seven gardens in Flin Flon and one in Creighton. In addition, vegetable samples were taken from a local grocery store and submitted for analysis to allow for a comparison to the results of the home garden survey. Consistent with Pip (1991), HBMS found that concentrations of metals in soil and vegetables decreased with increasing distance from the complex and that concentrations in vegetables were positively related to concentrations in soil. As found in the Pip (1991) study, concentrations in leafy vegetables, such as lettuce, cabbage,

chard, and beet tops, were significantly higher than those observed in other types of vegetables (HBMS, 1994a).

When compared to concentrations in vegetables collected from the local grocery store, HBMS (1994a) found that mean concentrations in carrots and tomatoes from home gardens were similar. However, lettuce from home gardens had concentrations of all metals except lead that were 50 to 75% higher than lettuce from the local grocery store. Concentrations of lead in lettuce from home gardens was approximately 70% lower than lettuce from the local grocery store (HBMS, 1994a).

To address public concerns raised in response to discrepancies reported in the home garden surveys completed by Pip (1991) and HBMS (1994a), Manitoba Conservation designed and completed a home garden sampling program in 2002. Nine home gardens from the Flin Flon area, located at varying distances and directions from the HBMS complex, were selected to characterize the potential influence of smelter-related emissions. In addition, a garden located in the community of Cranberry Portage was selected to be representative of an area that is minimally impacted by smelter emissions. A garden in the town of The Pas was selected to represent a non-impacted control site (Jones and Henderson, 2006).

Analysis of soil samples indicated that mean concentrations of several metals were significantly higher in Flin Flon gardens relative to the control site garden in The Pas (Jones and Henderson, 2006). Concentrations of arsenic, cadmium, copper, mercury, selenium, and zinc in soil were all highly correlated. Concentrations of arsenic, cadmium, copper, mercury, selenium, and silver in Cranberry Portage did not differ significantly from concentrations in The Pas, indicating that smelter emissions may not have a significant influence on conditions in Cranberry Portage (Jones and Henderson, 2006).

Concentrations of arsenic, cadmium, copper, lead, mercury, selenium, and zinc in vegetables from Flin Flon were generally higher than those from The Pas. Consistent with the Pip (1991) and HBMS (1994a) studies, concentrations in lettuce were typically higher than in other vegetables. Concentrations of certain metals (*e.g.*, cadmium and mercury) in vegetables were highly correlated with concentrations in soil, indicating that metals were absorbed from soil. A comparison between concentrations in washed and unwashed samples of lettuce indicated that atmospheric deposition was likely a contributing factor (Jones and Henderson, 2006). Overall, the Manitoba Conservation study concluded that although concentrations of certain metals were elevated in vegetables grown in Flin Flon gardens, the concentrations and anticipated consumption rates are not anticipated to result in human health concerns for individuals consuming home garden produce (Jones and Henderson, 2006).

Of these three primary studies, the Jones and Henderson report (2006) served as the primary source of home garden produce data for the HHRA. Although data from the Pip (1991) and HBMS (1994a) studies were considered to fill any data gaps, the data provided within the Manitoba Conservation study are more recent and robust. In addition, given that the study design and sampling program was developed to minimize the influence of several confounding factors (*e.g.*, differences in soil texture, amount of sunlight, proximity to fences or decks), the data from this study is considered to be a good representation of the effect of smelter-related emissions on concentrations of metals in home garden produce.

2.3.2 Data Gap Analysis

Although the Jones and Henderson (2006) report included a home garden sampling location (TQ0164) directly south of the HBMS complex, one was not selected within the town of Creighton. Given that the tailings impoundment is located directly north of Creighton, there is uncertainty as to whether or not the garden plots in this report are representative of the potential influence of wind-blown tailings on home gardens in Creighton. Home garden vegetable concentrations from the Jones and Henderson (2006) report are available for all metals that were retained for evaluation. No additional home garden sampling was completed as part of the HHRA.

2.4 Drinking Water Data

2.4.1 Summary of Review

Drinking water for residents of Flin Flon and Creighton is provided through separate municipal resources. Drinking water for Flin Flon is taken from Cliff Lake, which is supplied water from Trout Lake (also called Embury Lake) through active pumping. Drinking water for Creighton is taken from Douglas Lake.

Prior to the initiation of the HHRA, the most complete set of data for metal analysis in drinking water was the HBMS monthly sampling of drinking water taken from the Vocational Centre in Flin Flon. However, these data may not be appropriate for use in the HHRA to predict exposure to the general population because an additional filter is present at this sampling location and results may not be representative of the content of water within community-wide distribution. Three other sources of Flin Flon drinking water data were also available, likely collected as part of Manitoba Conservation or other Provincial monitoring programs. Concentrations of antimony, arsenic, cadmium, copper, lead, molybdenum, selenium, silver, thallium, and zinc were provided for treated Flin Flon drinking water from 2002 to 2006 (Table 2-4). Concentrations of these metals were relatively consistent across this sampling period with the exception of copper, which varied from 21 to 69 µg/L, and lead, which varied from 0.3 to 3.2 µg/L.

Table 2-4	Concentrations of Metals in Treated Drinking Water for Flin Flon from 2002 to 2006 (µg/L total metals)				
Chemical	*2002*	*2003*	*2004*	*2005*	*2006*
Antimony	0.7	0.6	0.6	0.6	0.5
Arsenic	1.9	2.0	1.8	2.5	2.3
Cadmium	0.68	0.61	0.65	0.61	0.92
Copper	21	21	48	45	69
Lead	0.7	0.3	0.6	1.5	3.2
Molybdenum	<0.1	<0.1	<0.1	<0.1	<0.1
Selenium	<0.2	<0.2	<0.2	<0.2	<0.2
Silver	<0.02	<0.02	<0.05	<0.05	<0.05
Thallium	<0.02	<0.02	<0.02	<0.02	<0.02
Zinc	140	95	110	86	85

Saskatchewan Environment (2006) also provided data for the Creighton Distribution System. The most recent data provided were for 2005 and 2006 in which concentrations (one per year) were provided for arsenic, cadmium, copper, lead, mercury, selenium, and zinc (Table 2-5). Concentrations of copper and lead in the Creighton drinking water were notably lower than those in Flin Flon drinking water during the same period.

Table 2-5	Concentrations of Metals Measured in the Creighton Distribution System in 2005 and 2006 (µg/L Total)	
Chemical	**February 2005**	**June 2006**
Arsenic	2.2	1.4
Cadmium	<0.5	0.4
Copper	13	3
Lead	<0.1	<0.1
Mercury	-	<0.05
Selenium	0.2	0.1
Zinc	110	46

2.4.2 Data Gap Analysis

For Flin Flon, concentrations of mercury in drinking water were not available for any sampling period. A single sample of drinking water for the years 2005 and 2006 was provided for the town of Creighton's Distribution System for seven of the priority metals (arsenic, cadmium, copper, lead, mercury, selenium, and zinc) (Saskatchewan Environment, 2006). No data were available for antimony, molybdenum, silver or thallium.

Based on discussions with Stephen West and Joel Nilsen of HBMS on August 28, 2007, HBMS arranged to sample water from several locations and provide a weekly analysis of the content of all potential COC in drinking water. HBMS provided measured concentrations of each chemical in drinking water collected on a bi-weekly basis from August 2007 to July 2008 for three locations in Flin Flon and one location in Creighton. These data were considered to be representative of any potential variability in drinking water quality that may result from seasonal fluctuations occurring in the source water body. Since there is a limited number of sample locations included within this program, members of the Technical Advisory Committee (TAC) decided that an additional short-term study in which multiple locations throughout Flin Flon and Creighton were included in a single sampling event should be completed to ensure that these data were representative of drinking water quality throughout the communities.

Jacques Whitford conducted a residential drinking water sampling program on a number of homes, schools, and daycares in Flin Flon (36) and Creighton (11) on March 6 and 7, 2008 to assess the current metal status of tap drinking water (Jacques Whiteford, 2008). For residential locations and daycares, drinking water samples were collected from the most frequently used tap, typically the kitchen. In schools, water samples were collected from a public water fountain (Jacques Whiteford, 2008).

2.4.3 Supplemental Drinking Water Sampling

Results of the supplemental drinking water sampling are provided in Chapter 4, Section 4.1.1.4 and the complete report is provided in Appendix Q.

2.5 Fish and Sediment Data

2.5.1 Summary of Review

Given that sport-fishing during both the summer and winter months is an important recreational activity in Flin Flon and Creighton, consumption of local fish may be a potentially significant route of exposure to metals released by the HBMS complex. When assessing exposure to chemicals as a result of the consumption of fish, there are several important factors to consider. Characterizing the concentrations of metals in fish should focus on those tissues, which are most likely to be consumed by humans. While many studies include the analysis of

concentrations in both liver and muscle tissue, it is common to assume that humans will generally only consume the muscle tissues and discard the organs and skin. Predicting concentrations of metals in muscle tissue based on measured concentrations in surface water and sediment can often be problematic due to several confounding factors associated with site-specific conditions and the limited rates of accumulation of many metals. Significant uncertainty can be reduced through the collection and analysis of fish tissues collected from lakes that are known to be frequented by local fishermen. Concentrations of metals within the tissues of fish from Flin Flon area lakes were documented in two separate studies. However, one of the studies focused on Ross and Schist Lakes which are known to be heavily influenced by liquid effluent from the HBMS complex as well as significant historic releases of sewage. As a result, use of fish data collected from either of these lakes may not be an accurate representation of levels that residents of Flin Flon or Creighton may be exposed to over a chronic duration due to public awareness of contamination in these lakes.

A 1982 study involved a survey of lakes that were considered to be within the deposition zone of the smelter (Harrison *et al.*, 1989; Harrison and Klaverkamp, 1990). Liver and skeletal muscle from white sucker (*Catostomus commersoni*) and northern pike (*Esox lucius*) were analyzed for arsenic, copper, cadmium, mercury, lead, selenium, and zinc. Thirteen lakes were sampled: five lakes from within an 8 km radius of the smelter (Flin Flon lakes) four lakes from 23 to 43 km east of the smelter (Manitoba lakes), and four lakes (not named) from 68 to 84 km northwest of the smelter (Saskatchewan lakes). Sediment cores were also collected and analyzed for copper, cadmium, mercury, selenium, and zinc in 1985 (three years after significant reductions in smelter emissions) (Table 2-6). Surface water samples were not taken because metal contamination within aquatic ecosystems is generally more accurately reflected in sediment composition. It was believed that measurements of metals in sediments would be a better predictor of contamination in biological tissues than surface water would be (Harrison and Klaverkamp, 1990).

Table 2-6	Summary of Mean Concentrations and Standard Deviations of Metals within the Top 2 cm of Sediments (µg/g dw)						
Location	*Lake*	*Distance (km)*	*Zinc*	*Copper*	*Cadmium*	*Mercury*	*Selenium*
Flin Flon/ Creighton	Hamell	4.5	8,408 (1,290)	2,775 (90)	56 (4.1)	3.27 (3.14)	11.1 (2.3)
	Douglas	5	12,625 (1,255)	1,900 (57)	60 (2.8)	3.77	12.6 (2.0)
	Phantom	5.4	6,988 (1,687)	1,385 (613)	36 (11.1)	9.22 (2.06)	4.5 (3.2)
	Cliff	5.6	5,950 (373)	1,278 (60)	21 (0.8)	6.39 (1.31)	4.0 (0.2)
	Meridian	7.6	2,858 (317)	568 (22)	15 (0)	2.69 (0.16)	3.4 (0.4)
Manitoba	Cleaver	23	1,238 (368)	198 (33)	7 (1.5)	1.30 (0.53)	2.5 (0.5)
	Naosap Mud	27	282 (190)	49 (22)	2 (1.3)	0.10 (0.11)	1.4 (0.2)
	Kotyk	30	507 (215)	76 (37)	3 (2)	0.35 (0.03)	2.4 (0.16)
	Nekik	43	170 (27)	34 (4)	<1	0.09 (0.05)	1.0 (0.2)

Table 2-6	Summary of Mean Concentrations and Standard Deviations of Metals within the Top 2 cm of Sediments (µg/g dw)						
Location	*Lake*	*Distance (km)*	*Zinc*	*Copper*	*Cadmium*	*Mercury*	*Selenium*
Saskatchewan	1	68	118 (5)	44 (1.3)	<1	0.11 (0.03)	0.9 (0.1)
	2	74	175 (13)	41 (1.3)	<1	0.14 (0.02)	1.3 (0.1)
	3	83	46 (21)	13 (5)	<1	0.03 (0.02)	0.2 (0.1)
	4	84	142 (5)	57 (0.8)	<1	0.22 (0.02)	0.9 (0.1)

The results of the sediment sampling indicated that the area receiving significant deposition of metals is likely much smaller than originally estimated by Franzin *et al*. (1979). Concentrations of metals in surface sediments of Saskatchewan lakes (68 to 84 km from the smelter) were equal to or less than lakes located in Ontario removed from significant anthropogenic sources. Based on the ratio of surface sediment to background concentrations of metals in sediments, it appears that the limit of deposition to the east was approximately 43 km as indicated by analysis of sediments in Nekik Lake (Harrison and Klaverkamp, 1990).

Analysis of metal content in fish indicated that generally, concentrations of most metals were found to be highest in the liver of fish from Flin Flon lakes (Harrison and Klaverkamp, 1990). While concentrations of metals such as cadmium, copper, selenium, and zinc were higher in the liver of fish from Flin Flon, relative to Saskatchewan, the opposite was true for mercury. Concentrations of mercury in the liver of both northern pike and white sucker were significantly higher in Saskatchewan lakes than in Flin Flon lakes. A similar pattern was observed in Sudbury, where concentrations of mercury in crayfish and perch were lower with increased proximity to the smelter. This was attributed to two factors. First, selenium has been reported to inhibit the accumulation of mercury by fish, therefore, increased concentrations of selenium in lakes closest to the smelter can have a significant impact on mercury accumulation. Secondly, methylmercury represents the largest component of total mercury within the tissues of fish. It has been reported that elevated concentrations of copper, cadmium, and zinc in sediments reduces the methylation of mercury (Jackson, 1984). This was supported by Harrison and Klaverkamp (1990) in which the highest concentrations of mercury in fish were associated with the lowest concentrations of copper, cadmium, selenium, and zinc in sediments. Although concentrations of some metals were elevated within liver tissues, as discussed previously, the HHRA focused on concentrations within the muscle tissues only.

Concentrations of metals in muscle tissue were not consistently higher in fish sampled from the Flin Flon lakes (Table 2-7). Concentrations of zinc in the muscle of white sucker were significantly higher in Flin Flon lakes relative to Saskatchewan lakes. Concentrations of cadmium in muscle tissue were below detection in all lakes with the exception of northern pike from Cliff Lake. Concentrations of lead and arsenic were below detection limits (<0.02 and <0.05 µg/g, respectively) in almost all lakes sampled. Detectable concentrations of arsenic were generally only found in lakes closest to the smelter. Concentrations of metals in the muscle of northern pike and the liver of white sucker were not significantly correlated with concentrations in sediments. However, concentrations of selenium in the liver of northern pike were positively correlated with selenium in sediment.

Table 2-7	Mean Concentrations and Standard Deviations of Metals in the Muscle Tissue of Northern Pike and White Sucker (µg/g ww)					
Group	*Species*	*Zinc*	*Copper*	*Cadmium*	*Mercury*	*Selenium*
Flin Flon	Northern Pike	5.6 (2.4)	0.16 (0.02)	<0.01	0.09 (0.03)	0.88 (0.38)
	White Sucker	5.5 (1.6)	0.22 (0.04)	<0.01	0.02 (0.01)	1.05 (0.49)
Manitoba	Northern Pike	5.1 (2.0)	0.12 (0.02)	<0.01	0.22 (0.09)	0.20 (0.06)
	White Sucker	3.0 (0.04)	0.25 (0.04)	<0.01	0.03 (0.01)	0.22 (0.12)
Saskatchewan	Northern Pike	5.7 (0.6)	0.22 (0.05)	<0.01	0.47 (0.25)	0.29 (0.10)
	White Sucker	3.5 (0.4)	0.34 (0.07)	<0.01	0.06 (0.03)	0.28 (0.12)

Overall, the Harrison and Klaverkamp (1990) study found that concentrations of metals in muscle tissue were not consistently higher in fish sampled from the Flin Flon lakes. Generally, the results of the study indicated that concentrations of metals found in fish from the area surrounding the smelter were similar to concentrations observed in fish from remote lakes from the Precambrian shield of Ontario, which have not been impacted by anthropogenic sources. Concentrations in fish tissues were generally not directly proportional to concentrations in sediments. Results were generally similar to those observed in Sudbury (Harrison and Klaverkamp, 1990).

A limnological survey of Ross Lake and Schist Lake was completed by HBMS in 1994. Sediment samples were collected from the top 5 cm using an Ekkman Box Corer (Table 2-8) (HBMS, 1994b).

Table 2-8	Concentrations of Metals in Sediment of Ross and Schist Lake (µg/g)	
Chemical	*Ross Lake*	*Schist Lake*
Arsenic		
#samples	9	31
Maximum	316	596
Average	201	85
Cadmium		
#samples	9	31
Maximum	769	481
Average	364	113
Copper		
#samples	9	31
Maximum	20,300	11,270
Average	11,230	1,870
Lead		
#samples	9	31
Maximum	385	235
Average	273	118
Mercury		
#samples	9	31
Maximum	179	12
Average	35	1.88
Zinc		
#samples	9	31
Maximum	64,700	91,100
Average	43,000	21,270

As a part of this survey, fish were collected from Schist Lake and analyzed for metal content in liver and muscle tissue. Of the eleven metals of potential concern for the HHRA, six were analyzed for in fish samples: arsenic, cadmium, copper, mercury, lead, and zinc. Fish samples were collected in July/August, 1994 (Table 2-9) and in October, 1994 (Table 2-10) using gill nets. Individual specimens were analyzed for metals in muscle tissue and liver tissue when sufficient tissue was available. Homogenizing of samples occurred when necessary. A total of 68 specimens were captured (HBMS, 1994b).

Table 2-9   Concentrations of Metals in the Tissues of Fish from Schist Lake Sampled in August, 1994 (µg/g)

Species	Tissue	Parameter	Copper	Zinc	Lead	Cadmium	Mercury	Arsenic
Northern Pike (n=5)								
Muscle		Mean	1.0	8.5	0.3	<0.1	0.047	0.046
		Max	2.7	12.1	0.5	<0.1	0.078	0.098
Liver		Mean	22.1	39.6	<0.3	<0.1	0.056	0.029
		Max	28.9	51.2	<0.3	<0.1	0.093	0.068
Walleye (n=1)								
Muscle		Mean	-	-	-	-	-	-
		Max	0.6	6.0	<0.3	<0.1	0.064	0.065
Liver		Mean	-	-	-	-	-	-
		Max	7.4	21.1	<0.3	0.5	0.04	0.11
Cisco (muscle n=12, liver n=6)								
Muscle		Mean	0.8	9.7	<0.3	<0.1	0.031	0.04
		Max	1.1	23.9	<0.3	<0.1	0.078	0.08
Liver		Mean	6.9	18	<0.3	0.2	0.148	0.140
		Max	12.6	34.2	<0.3	0.3	0.690	0.194
Lake Whitefish (muscle n=2, liver n=1)								
Muscle		Mean	0.5	6.1	0.4	<0.1	0.026	0.042
		Max	0.5	6.2	0.4	<0.1	0.026	0.063
Liver		Mean	-	-	-	-	-	-
		Max	7.1	22.1	<0.3	0.4	0.330	0.119

Table 2-10   Concentrations of Metals in the Tissues of Fish from Schist Lake Sampled in October, 1994 (µg/g)

Species	Tissue	Parameter	Copper	Zinc	Lead	Cadmium	Mercury	Arsenic
Northern Pike (n=19)								
Muscle		Mean	0.8	5.9	0.3	0.1	0.028	0.020
		Max	1.3	8.6	0.5	0.3	0.074	0.028
Liver		Mean	10	38.5	<0.3	0.1	0.018	0.019
		Max	20.2	59	<0.3	0.6	0.028	0.035
White Sucker (muscle n=4, liver n=2)								
Muscle		Mean	0.7	4.9	0.3	0.2	0.011	0.025
		Max	0.8	5.7	0.4	0.2	0.015	0.039
Liver		Mean	7.9	32.7	0.5	0.3	0.034	0.053
		Max	9.4	34.1	0.7	0.3	0.037	0.067
Cisco (muscle n=21, liver n=10)								
Muscle		Mean	0.5	5.4	0.3	0.2	0.012	0.031
		Max	0.7	7.2	0.4	0.4	0.02	0.045
Liver		Mean	4	21.7	0.4	0.3	0.018	0.068
		Max	15.7	44.5	1.0	0.7	0.020	0.094
Lake Whitefish (muscle n=4, liver n=2)								
Muscle		Mean	0.5	5.4	0.3	0.1	0.011	0.029
		Max	0.6	6.0	0.4	0.2	0.015	0.040
Liver		Mean	5.9	27.3	0.5	0.6	0.056	0.044
		Max	6.4	28	0.7	0.7	0.058	0.042

While this study offers important insight into the movement of metals within this water system and the impact on fish within this environment, the northern basin of Ross Lake receives treated

alkaline effluent overflow through Flin Flon Creek from HMBS tailings ponds, and Schist Lake receives water from Ross Lake *via* Ross Creek. In addition, water quality in Ross Lake is well known to be heavily compromised as a result of historic sewage discharge, therefore, it was not considered to be relevant to utilize fish data from these lakes within the HHRA.

2.5.2   Data Gap Analysis

Both studies described are relatively old and data may not be an accurate reflection of current levels in fish from the Flin Flon and Creighton area. Although concentrations in fish from the 1982 survey do not appear to be significantly elevated above background levels, concentrations of copper and zinc in muscle tissue from Schist Lake measured in 1994 were significantly higher than those reported in the 1982 survey. Lack of current data may create an area of uncertainty for the HHRA. In addition, concentrations of antimony, molybdenum, silver, and thallium in fish tissues have not been reported in any of the studies reviewed. Since there are several factors that influence the uptake of metals from the surrounding aquatic environment into the tissues of fish, it is recommended that tissue concentrations not be predicted based on more recent measurements of metals in sediments and surface water, rather measured data is preferred. This recommendation is supported by the findings of the 1982 survey which concluded that concentrations in fish were not proportional to concentrations in sediments.

2.5.3   Supplemental Fish and Sediment Data

Based on the results of the local food survey, a sampling program was developed for the collection and analysis of common sport fish from lakes at varying direction and distance from the smelter that are reported to be frequently used for fishing. This program was completed by Stantec in August, 2008. Results of this program were combined with data collected by Manitoba Water Stewardship during the same sampling period. Details of this study and its results are discussed in Chapter 4, Section 4.1.1.6. See Appendix E for the complete study.

2.6   Surface Water Data

2.6.1   Summary of Review

Concentrations of metals in surface water in the Flin Flon area were reported in two main studies, all of which focused primarily on Ross Lake and Flin Flon Creek as a result of the direct discharge of effluent overflow from the HBMS facility. Surface water chemistry is also available for areas that are considered to be representative of areas that may be impacted by smelter emissions but not by effluent discharge. These include Phantom Lake and Beaver Dam Creek (Stantec, 2005).

As part of the Environmental Effects Monitoring (EEM) report, surface water samples were collected from reference and exposure areas associated with the Flin Flon tailings impoundment system (FFTIS) discharge (the North Weir) and the Trout Lake mine discharge, both of which discharge into Flin Flon Creek (Stantec, 2005). Maximum surface water concentrations from exposure areas (E) (at or downstream of the discharge areas) and reference areas (R) (similar characteristics to the exposure areas but unaffected by the discharge) are presented in Table 2-11. Samples were collected in September, 2004.

Table 2-11   Surface Water Concentrations Measured in Areas Affected by HBMS Discharge and Selected Reference Areas (µg/L)

Chemical	Beaver Dam Creek (R)	North Weir (E)	Flin Flon Creek (R)	Flin Flon Creek (E)	Ross Creek (E)	Phantom Lake (R)	Schist Lake (E)
Arsenic	4.4	3.1	3.9	4	5.8	7.9	3.4
Cadmium	2.15	1.19	1.88	2.18	7.9	0.2	1.2
Copper	242	80	89	79	237	9	86
Lead	5.5	2.2	2.8	2.4	15.6	3.2	2.2
Mercury	<0.05	<0.05	<0.05	<0.05	0.1	0.06	0.05
Molybdenum	0.3	10.7	9.7	9	6.3	7	11
Zinc	610	330	390	480	920	140	280

E – indicates exposure area; R - indicates reference area.

Although the HHRA will consider potential exposure as a result of direct contact with surface water during recreational activities, it is not anticipated that many members of the community will be involved in recreational activities that would result in chronic exposure to surface water in Ross Lake or Flin Flon Creek. As a result, although concentrations of metals measured in the exposure areas of the EEM could be used to represent a worst-case scenario for dermal exposure to metals in surface water, concentrations measured in the reference areas are likely a more accurate representation of a chronic exposure scenario.

Concentrations of selected metals in the surface water of Douglas Lake were provided by Saskatchewan Environment (1993). Douglas Lake is an important resource to the town of Creighton because it serves as the raw source of municipal drinking water. However, the most recent data provided for Douglas Lake is for 1993 in which only arsenic and zinc were found above the laboratory detection limit (Table 2-12).

Table 2-12   Concentrations of Total Metals in Surface Water of Douglas Lake in August, 1993

Chemical	Concentration (µg/L total)
Arsenic	8
Cadmium	<1
Copper	<1
Lead	<5
Molybdenum	<5
Selenium	<1
Silver	<1
Zinc	82

2.6.2   Data Gap Analysis

Recent surface water concentrations are limited for lakes in the Flin Flon and Creighton areas that are likely to be used for ongoing recreational activities such as swimming. Although dermal uptake of metals from surface water is not anticipated to be a significant exposure pathway relative to soil-related pathways, use of appropriate surface water data within the HHRA is important to help reduce overall uncertainty.

2.6.3   Supplemental Surface Water Data

During the completion of the fish study conducted by Stantec in the summer of 2008, surface water samples were collected and analyzed for each potential COC. Manitoba Water Stewardship completed a follow-up surface water sampling program to supplement the data

collected by Stantec. Details of this study and its results are discussed in Section 4.1.1.10. The complete Stantec study is attached as Appendix E.

2.7   Blueberry Data

2.7.1   *Summary of Review*

Four studies were reviewed in which concentrations of metals in blueberries in Flin Flon were described (Wotton, 1979; Shaw, 1981; McEachern and Phillips, 1983; Manitoba Conservation, 2000). With the exception of Shaw (1981), these studies only measured concentrations of mercury and lead. It was generally observed that the greater concentrations were located in close proximity to the smelter, although no statistically significant trends were measured to support this.

Sampling conducted in 2000 by Manitoba Conservation reported lower metal concentrations than those measured between 1978 to 1981 (Wotton, 1979; Shaw, 1981; McEachern and Phillips, 1983). Mercury concentrations for washed berries were <0.01 µg/g (dry weight) for all samples taken from 13 locations in 2000 at distances ranging from 1.95 km to 155 km from the smelter (Table 2-13). In samples taken between 1978 and 1981, mercury concentrations for unwashed berries ranged from 0.01 to 0.22 µg/g (dry weight) and <0.02 to <0.10 µg/g (dry weight) for washed berries (Wotton, 1979; Shaw, 1981; McEachern and Phillips, 1983). These values were similar to background concentrations for unwashed (0.02 to 0.09 µg/g dry weight) and washed berries (<0.03 to <0.07 µg/g) (Wotton, 1979; Shaw, 1981; McEachern and Phillips, 1983). Generally, the washed berries had only slightly lower concentrations than the unwashed berries.

As part of the Manitoba Conservation study in 2000, concentrations of lead in washed blueberries were only found above the detection limit of 0.01 µg/g (dry weight) at two of the 13 sampling locations. Maximum concentrations of 0.3 µg/g were measured at locations 2.5 km south-southeast and 1.95 km north-northeast of the smelter. The highest concentrations of lead in unwashed blueberries (7.0 and 7.26 µg/g dry weight) were reported by Shaw (1981) from two samples taken in 1978 from a site 2.4 km southeast of the smelter. Concentrations of lead in blueberries in sites beyond 25 km of the smelter could not be determined analytically in the latter study.

Background concentrations of lead in berries sampled between 1978 and 1981 reported by Wotton (1979) and McEachern and Phillips (1983) exceeded the maximum concentration measured in 2000 (0.3 µg/g, dry weight, washed). Wotton (1979) reported background concentrations of <1.20 and <1.25 µg/g dry weight, washed and unwashed, respectively. The lead concentrations reported from non-reference sites in this study did not exceed the background concentrations to a great extent, with dry weight concentrations ranging from 1.18 to 2.42 µg/g (washed), and 1.16 to 2.43 µg/g (unwashed). McEachern and Phillips (1983) reported lead concentrations ranging from 0.61 to1.9 µg/g (dry weight, washed), and 0.94 to 1.6 µg/g (dry weight, unwashed) at a control site. Lead concentrations measured from sites not designated as background in this study ranged from 1.6 to 5.0 µg/g (washed) and 1.7 to 5.2 µg/g (unwashed). Many of the concentrations measured prior to 1990 exceeded the average lead concentrations in Canadian berries (0.05 µg/g) (McEachern and Phillips, 1983).

Table 2-13	Maximum Concentrations of Lead and Mercury in Washed Blueberries Sampled from 13 Locations in the Flin Flon Area (µg/g dw) (Manitoba Conservation, 2000)		
Location	*Date Sampled*	*Lead*	*Mercury*
TQ0151 (n=3)	08/15/2000	< 0.1	< 0.01
TQ0152 (n=3)	08/15/2000	< 0.1	< 0.01
TQ0153 (n=3)	08/16/2000	< 0.1	< 0.01
TQ0154 (n=3)	08/16/2000	< 0.1	< 0.01
TQ0155 (n=3)	08/15/2000	< 0.1	< 0.01
TQ0156 (n=3)	08/15/2000	< 0.1	< 0.01
TQ0157 (n=3)	08/15/2000	0.3	< 0.01
TQ0158 (n=3)	08/15/2000	< 0.1	< 0.01
TQ0159 (n=3)	08/15/2000	< 0.1	< 0.01
TQ0160 (n=3)	08/15/2000	< 0.1	< 0.01
TQ0161 (n=3)	08/15/2000	0.3	< 0.01
TQ0162 (n=3)	08/15/2000	< 0.1	< 0.01
TQ0163 (n=3)	08/17/2000	< 0.1	< 0.01

Only one study (Shaw, 1981) measured concentrations of metals other than lead and mercury in blueberries. A total of three sites were surveyed, with concentrations of arsenic, cadmium, copper, lead, mercury, molybdenum, selenium, silver, thallium and zinc measured. Concentrations of arsenic, copper, lead, mercury, selenium, and zinc were notably higher in blueberries sampled from the site in closest proximity to the smelter compared to two sites at beyond 25 km (Table 2-14).

Table 2-14	Concentrations (µg/g dw) of Various Elements in Blueberries in Flin Flon at Select Distances from the HBMS Complex (Shaw, 1981)		
	Sample Locations[a]		
Chemical	*Louis Lake (2.4 km SE)*	*Mystic Creek (26.5 km SE)*	*Cranberry Portage (36.2 km SE)*
Arsenic	1.74 to 2.0[b]	0.31	0.16
Cadmium	ND	ND	ND
Copper	21.1	3.7	3.7
Lead	7.0 to 7.26[b]	ND	ND
Mercury	0.22	0.01	0.02
Molybdenum	ND	ND	ND
Selenium	<0.2 to 0.22[b]	<0.3	0.07
Silver	0.5	ND	15
Thallium	ND	ND	ND
Zinc	24	ND	ND

[a] One sample was taken at each location
[b] Concentrations reported by two different labs (n=2)
ND No Data

2.7.2   Data Gap Analysis

Although concentrations of lead and mercury in blueberries were nearly all below detection in the Manitoba Conservation (2000) report, indicating that the consumption of local berries may not be a significant source of exposure, values reported by Shaw (1981) may indicate otherwise. Concentrations of lead were notably higher in this study, and concentrations of several other metals were found to be elevated at the sampling site located in close proximity to the smelter relative to the reference locations.

Based on personal communications with HBMS staff, it is understood that it is common for some residents of Flin Flon and Creighton to pick large amounts of blueberries to be consumed

over the course of the entire year. Although it is likely that residents will obtain a significant portion of their blueberry harvest from areas that are not immediately adjacent to the smelter and may not be significantly impacted by smelter-related emissions, a detailed evaluation of this exposure pathway in the HHRA was warranted.

2.7.3   *Supplemental Berry Collection*

During the completion of the fish study conducted by Stantec in the summer of 2008, wild blueberries were collected and analyzed for each potential COC. Details of this study and its results are discussed in Chapter 4, Section 4.1.1.9. The complete Stantec study is attached as Appendix E.

2.8   Snow Data

Jacques Whitford conducted snow column sampling at select locations on March 8 and 9, 2008 to provide a preliminary view of metal concentrations and metal deposition in snow around the City of Flin Flon (Jacques Whitford, 2008). A limited number of sites (n=12) from the Creighton and Flin Flon areas were selected for sampling based on the availability of non-disturbed snow profiles and relative accessibility. The sites were also selected to be downwind of prevailing northwest winds in the area and to be within the area of the Jacques Whitford residential soil sampling program. Snow column samples were collected in March to ensure total winter accumulation was obtained. The complete snow column sampling report is provided in Appendix R.

2.9   Bioavailability/Bioaccessibility

The ingestion of soils is often considered to be the major route of potential exposure to metals in humans (Sheppard *et al.*, 1995; Paustenbach, 2000). To effectively assess the dose of soil metals received by humans, the determination of bioavailability becomes an invaluable tool in risk assessment. Bioavailability is the fraction of a chemical which is ingested, inhaled, or applied on the skin surface that is absorbed and reaches the systemic circulation (Kelly *et al.* (2002). The approach for oral bioavailability assessment of contaminants can typically be divided into four fundamental processes: i) the oral intake of soil/dust including metals; ii) bioaccessibility; iii) intestinal absorption; and, iv) metabolism in the liver/intestines (Oomen *et al.*, 2006; Sips *et al.*, 2001). Out of these processes that construct the basis of bioavailability, bioaccessibility testing is a key component. The inclusion of bioaccessibility testing as part of the assessment process allows for a more realistic estimate of the systemic exposure to metals from soil and dust ingestion than using generic assumptions such as those employed to derive soil guideline values (EAUK, 2005a).

2.9.1   *Overview of Bioaccessibility*

Oral bioaccessibility can be defined as the fraction of a substance that is released from the soil or dust matrix during digestion, thus making it soluble and available for absorption through the gastrointestinal tract (Defra and Environment Agency, 2002). In effect, this fraction represents the upper limit of bioavailability. Oral bioaccessibility only takes into account the direct ingestion of soil and dust and does not incorporate other routes of exposure such as skin and lungs. The bioaccessible fraction is the fraction of the substance of interest that is dissolved from soil into chyme, and represents the maximum fraction available for intestinal absorption (Ruby *et al.*, 1999; Sips *et al.*, 2001). The dissolved substance may be absorbed and transported across the intestinal wall into the blood or the lymphatic system. Once dissolved, some of the substance may precipitate in the intestine, be bound to other substances or undergo chemical

transformation to an insoluble form. Any of the processes would lead to a portion of the substance remaining unavailable for absorption. Once distributed into the systemic circulation from the intestines or the liver, substances can ultimately start to exert systemic toxicity (Sips *et al.*, 2001). Thus, one can see the importance in assessing bioaccessibility as it will determine the amount of a soil- or dust-bound material that will actually become bioavailable to potentially exert effects in the body.

Bioavailability depends, in large measure, upon bioaccessibility. When bioaccessibility is low, oral bioavailability will also be low. Absolute oral bioavailability of soil-borne substances can be estimated on the basis of bioaccessibility in combination with the absorption and metabolism values from toxicological studies (Sips *et al.*, 2001). Hence, the effectiveness of methods in determining the bioaccessibility of soil contaminants may dictate the overall conclusions of risk assessments.

Toxicity data employed in most risk assessments (*e.g.,* reference doses [RfDs] and cancer slope factors [CSFs]) are typically developed, in part, from toxicological studies using animals. These studies generally use a highly bioavailable chemical form (*e.g.,* soluble inorganic salts, *etc.*) and delivery media (*e.g.,* food, water, *etc.*) to ensure a high dose reaches the target tissue. As such, RfDs and CSFs do not inherently address the availability of compounds in other environmental media, such as soils and dust. It is, therefore, important that the bioavailability of the compound present in soil or dust, relative to bioavailability of the chemical species and delivery media used by the critical toxicological study (*i.e.,* the study used to develop either the RfD or CSF), be quantitatively supported.

Absolute bioavailability refers to the fraction or percentage of a compound that is ingested, inhaled or applied to the skin that is absorbed and reaches systemic circulation (Hrudey *et al.*, 1996). Relative bioavailability, as it pertains to risk assessment, has been defined as *"the difference in absorption of a compound from the environmental medium of concern (e.g., food, soil and/or water) versus the absorption from the vehicle (or medium) used in the toxicological study from which the toxicity-based reference value is derived"* (Kelly *et al.,* 2002).

Traditionally, *in vivo* studies (*i.e.,* animal studies) have been used to determine the relative bioavailability of metals; however, *in vivo* studies can have significant associated time and cost constraints (Ruby *et al.,* 1999). Therefore, more rapid and inexpensive *in vitro* extraction studies (designed to simulate the human stomach and intestinal system) have been developed to provide a reasonable, yet conservative, approximation of true bioavailability by assuming relative bioavailability is equal to bioaccessibility. *In vitro* extraction studies have been designed to simulate the human gastrointestinal tract (*e.g.,* pH, temperature, and chemical composition of solutions in both the stomach and small intestine, *etc.*) in order to assess the mobilization of compounds from soil during the digestion process.

Given the importance of evaluating the potential toxicity of soil-bound COC to Flin Flon residents, *in vitro* bioaccessibility analyses were conducted. The objective of these analyses was to estimate the bioaccessible fractions of mercury, arsenic, cadmium, copper, lead, selenium, and zinc in Flin Flon soil samples. These results were then used to derive a relative absorption factor (RAF) for each COC.

An RAF based on a bioaccessibility evaluation is a simple quotient comparing the solubility of COC in soil and the exposure medium used to develop the RfD/CSF (*i.e.*, spiked food) in simulated digestive fluids. The RAF makes no assumptions about digestive differences between humans and other mammalian species, and is calculated as follows:

$$RAF = \frac{Bioaccessibility\ of\ Chemical\ in\ Soil}{Bioaccessibility\ of\ Chemical\ in\ Exposure\ Medium\ used\ to\ Develop\ the\ RfD}$$

Many different *in vitro* test methods are available to measure bioaccessibility of inorganic compounds in soil. Oomen *et al.* (2002) evaluated five different types of *in vitro* digestion models for three different soil types, producing a wide range of bioaccessibility results. Although data on bioaccessibility of lead and arsenic in soil are available, limited data are available for other metals such as nickel, copper, zinc, cadmium, and chromium (DEPA, 2003). At this time, no single *in vitro* method has been universally accepted (DEPA, 2003).

It is important to note that oral bioaccessibility testing is only applicable to the soil human exposure pathways, and not the food consumption pathways (EAUK, 2005a). While bioaccessibility testing may be a valuable addition to risk assessment practices, it is an evolving science and several uncertainties remain. Oral bioaccessibility results have been shown to vary considerably within and between contaminated sites. Therefore, it is prudent to only apply bioaccessibility data on a site-specific basis (EAUK, 2005a). Furthermore, bioaccessibility test results have been reported to be significantly affected by various factors such as physical-chemical properties of the contaminants (Dieter *et al.*, 1993; Freeman *et al.*, 1996; Gasser *et al.*, 1996; Ruby *et al.*, 1996; 1999), soil characteristics (Ruby *et al.*, 1993; 1996; 1999; Hamel *et al.*, 1998; 1999), the composition of digestive fluids (Guyton, 1991; Ruby *et al.*, 1992; Oomen *et al.*, 2000), and the presence of food constituents (Hack and Selenka, 1996). Hence, it is not realistic to propose a single value to represent the bioaccessibility of a given metal; site-specific values must be developed on a case-by-case basis. Table 2-15 provides an overview of bioaccessibility results for key COC, published in the primary literature, from a variety of different test methods and site-specific conditions.

Table 2-15			Collection of *In Vitro* Bioaccessibility Values From Primary Literature for Key COC			
COC	**Value (%)**		**Medium**	**Method**	**Source of Chemical**	**Reference**
	Mean ± SD	**Range**				
Arsenic	48.0 ± 3.0[a]	41.0 to 48.0[b]	Gastric phase	U.S. Pharmacopeia methodology for extraction. Total extractable metal digestion procedure followed modified EPA method 3051. Analysis performed using ICPMS	NIST Montana SRM 2710 (Control)	Hamel *et al.*, 1998
	13 ± 3.0[a]	4.5 to 25.0[b]			New Jersey, Jersey City composite soil	
	66.0 ± 8.0	-	Gastric + intestinal phase	Using mass-balance and soil recapture analytical methods	NIST Montana SRM 2710 (Control)	Hamel *et al.*, 1999
	34.0 ± 14.0	-			Slag material in Jersey City, U.S.	
	41.0 ± 2.0	-			Residential soil in Jersey City, U.S.	
	59.0 ± 2.0	-	Gastric phase	SBET method (BGS), United Kingdom	NIST Montana SRM 2710 (Control)	Oomen *et al.*, 2002
	50.0 ± 0.2	-			Flanders	
	11.0 ± 2.0	-			Oker 11	
	50.0 ± 1.0	-	Gastric + intestinal phase	DIN method (RUB), Germany	NIST Montana SRM 2710 (Control)	
	44.0 ± 3.0	-			Flanders	
	18.0 ± 3.0	-			Oker 11	
	41.2 ± 2.0	-		DIN method-without whole milk powder (RUB), Germany	NIST Montana SRM 2710 (Control)	
	30.0 ± 1.0	-			Flanders	
	11.0 ± 1.0	-			Oker 11	
	59.0 ± 1.0	-		In vitro digestion model (RIVM), Netherlands	NIST Montana SRM 2710 (Control)	
	95.0 ± 10.0	-			Flanders	
	19.0 ± 1.0	-			Oker 11	
	10.0 ± 0.4	-	Gastric + intestinal phase	SHIME method (Lab MET/Vito) Belgium	NIST Montana SRM 2710 (Control)	Oomen *et al.*, 2002
	6.0 ± 0.5	-			Flanders	
	1.0 ± 0.02	-			Oker 11	
	50.0 ± 1.0	-	Gastric + Intestinal phase (3 sections: duodenum, jejunum, and ileum)	TIM method (TNO) Nutrition, Netherlands	NIST Montana SRM 2710 (Control)	
	52.0 ± 1.0	-			Flanders	
	15.0 ± 3.0	-			Oker 11	
	24.8	-	Gastric phase	Invitro Gastrointestinal method (IVG)	Arsenic contaminated soil, excluding calcine [d]	Rodriguez and Basta, 1999
	21.9	-	Intestinal phase			
	23.0	-	Gastric+ intestinal phase	IVG with adsorption[c]		
	18.3	-	Gastric phase	Physiologically based Extraction test (PBET)		
	12.5	-	Intestinal phase			
	50.0[k]	50.0 to 50.0	Intestinal phase	PBET (PH 1.3)	Composite residential soil from Anaconda (ARS-I)	Ruby *et al.*, 1996
	32.0[k]	30.0 to 34.0		PBET (PH 2.5)		
	44.0[k]	44.0 to 44.0		PBET (PH 1.3)	Composite residential soil from Anaconda (ARS-II)	
	31.0[k]	20.0 to 32.0		PBET (PH 2.5)		

Table 2-15   Collection of *In Vitro* Bioaccessibility Values From Primary Literature for Key COC

COC	Value (%)		Medium	Method	Source of Chemical	Reference
	Mean ± SD	*Range*				
	34.0[k]	32.0 to 36.0		PBET (PH 2.5)	Composite house dust sample from Anaconda (AHD-I)	
Lead (Pb)	64.8 ± 9.4	52.4 to 77.2	Gastric phase	Using modified method (Mass-balance and soil recapture) of Hamel *et al.*, 1999; Ellickson *et al.*, 2001; 2002; Ruby *et al.*, 1992; 1993;1996	House dusts from carpets of urban homes.	Yu *et al.*, 2006
	12.1 ± 8.2	4.9 to 32.1	Intestinal phase			
	34.0 ± 7.5[a]	29.0 to 46.0[b]	Gastric phase	U.S. Pharmacopeia methodology for extraction. Total extractable metal digestion procedure followed modified EPA method 3051. Analysis performed using ICPMS	NIST Montana SRM 2710 (Control)	Hamel *et al.*, 1998
	46 ± 16[a]	22 to 58[b]			New Jersey, Jersey City composite soil	
	62.0 ± 1.0	-	Gastric + intestinal phase	Using mass-balance and soil recapture analytical methods	NIST Montana SRM 2710 (Control)	Hamel *et al.*, 1999
	70.0 ± 11.0	-			Contaminated soil from Bunker- Hill, ID, U.S.	
	39.0 ± 14.0	-			Slag material in Jersey City, U.S.	
	69.0	-			Residential soil in Jersey City, U.S.	
	90.0 ± 2.0	-	Gastric Phase	SBET method (BGS), United Kingdom	NIST Montana SRM 2710 (Control)	Oomen *et al.*, 2002
	91.0 ± 2.0	-			Flanders	
	56.0 ± 4.0	-			Oker 11	
	68.0 ± 2.0	-	Gastric + intestinal	DIN method (RUB), Germany	NIST Montana SRM 2710 (Control)	
Lead	40.0 ± 2.0	-	Gastric + intestinal phase	DIN method (RUB), Germany	Flanders	Oomen *et al.*, 2002
	23.0 ± 1.0	-			Oker 11	
	46.0 ± 2.0	-		DIN method-without whole milk powder (RUB), Germany	NIST Montana SRM 2710 (Control)	
	31.0 ± 3.0	-			Flanders	
	16.0 ± 2.0	-			Oker 11	
	11.0 ± 2.0	-		In vitro digestion model (RIVM), Netherlands	NIST Montana SRM 2710 (Control)	
	66.0 ± 9.0	-			Flanders	
	29.0 ± 2.0	-			Oker 11	
	3.0 ± 0.3	-		SHIME method (Lab MET/Vito) Belgium	NIST Montana SRM 2710 (Control)	
	4.0 ± 1.0	-			Flanders	
	1.0 ± 0.1	-			Oker 11	
	17.0 ± 3.0	-	Gastric + Intestinal phase (3 sections:	TIM method (TNO) Nutrition, Netherlands	NIST Montana SRM 2710 (Control)	
	13.0 ± 3.0	-			Flanders	

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Table 2-15   Collection of *In Vitro* Bioaccessibility Values From Primary Literature for Key COC

COC	Value (%)		Medium	Method	Source of Chemical	Reference
	Mean ± SD	Range				
	4.0 ± 1.0	-	duodenum, jejunum, and ileum)		Oker 11	
	13.0[e]	-			Top Soil, particle size <53µm	
	15.0[e]	-			Top Soil, particle size 53 to 100 µm	
	29.0[e]	-	Gastric phase	Simulated stomach acid digestion procedure[f]	Top Soil, particle size 100 to 500 µm	Rasmussen, 2004
	74.0[e]	-			Urban house dust<53µm	
	55.0[e]	-			Urban house dust 53 to 100µm	
	60.0[e]	-			Suburban house dust <100µm	
	23.0	0.70 to 36.3	Gastric phase	*In vitro* Gastrointestinal method (IVG) with dough[g]		Schroder *et al.*, 2004[h]
	0.56	0.02 to 1.16	Intestinal phase		Contaminated soil	
	32.2	1.40 to 64.4	Gastric phase	*In vitro* Gastrointestinal method (IVG) without dough[g]		
	1.06	0.03 to 3.23	Intestinal phase			
	9.5[i]	5.0 to 9.9	Gastric phase	PBET (PH 1.3)		
	4.6[j]	1.0 to 3.6	Intestinal phase			
	3.8[i]	2.7 to 3.8	Gastric phase	PBET (PH 2.5)	Composite mine waste materials from Butte, MT (BMW-I)	
	2.7[j]	0.94 to 1.2	Intestinal phase			
	1.3[i]	1.1 to 1.3	Gastric phase	PBET (PH 4.0)		
	-	0.48 to 1.84	Intestinal phase			Ruby *et al.*, 1996
	35.0[i]	22.0 to 35.0	Gastric phase	PBET (PH 1.3)		
	8.3[j]	4.0 to 4.0	Intestinal phase		Composite mine waste materials from Butte, MT (BMW-II)	
	13.0[i]	6.0 to 13.0	Gastric phase	PBET (PH 2.5)		
	9.80[j]	3.0 to 13.0	Intestinal phase			
	70.0[i]	58.0 to 70.0	Gastric phase	PBET (PH 1.3)	Composite residential soil from Bartlesville (BVS)	
Lead	29.0	12.0 to 70.0	Intestinal phase	PBET (PH 1.3)	Composite residential soil from Bartlesville (BVS)	
	26.0[i]	22.0 to 26.0	Gastric phase	PBET (PH 3.0)		
	29.0[j]	11.0 to 26.0	Intestinal phase			
	83.0[i]	72.0 to 83.0	Gastric phase	PBET (PH 1.3)		Ruby *et al.*, 1996
	54.0[j]	25.0 to 83.0	Intestinal phase		Composite residential soil from Salt Lake City (SCS)	
	22.0[i]	11.0 to 22.0	Gastric phase	PBET (PH 2.5)		
	18.0[j]	7.0 to 8.0	Intestinal phase			
	16.0	12.0 to 16.0	Gastric phase	PBET (PH 1.3)	Composite tailings sample from Copperton (CT-1)	
	3.0	1.2 to 1.7	Intestinal phase			
	8.0	6.8 to 8.0	Gastric phase	PBET (PH 2.5)		

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Table 2-15   Collection of *In Vitro* Bioaccessibility Values From Primary Literature for Key COC

COC	Value (%)		Medium	Method	Source of Chemical	Reference
	Mean ± SD	*Range*				
	0.6	0.2 to 0.3	Intestinal phase			
	10.0	8.0 to 10.0	Gastric phase	PBET (PH 1.3)	Composite tailings sample from Copperton (CT-2)	
	1.1	0.4 to 0.7	Intestinal phase			
	6.0	4.0 to 6.0	Gastric phase	PBET (PH 2.5)		
	2.1	0.7 to 1.0	Intestinal phase			
	49.0	39.0 to 49.0	Gastric phase	PBET (PH 1.3)	Composite stream channel sample from Bingham (CT-3)	
	14.0	5.0 to 8.0	Intestinal phase			
	24.0	22.0 to 24.0	Gastric phase	PBET (PH 2.5)		
	17.0	7.0 to 7.0	Intestinal phase			

[a]   Values recorded at the 1000:1 liquid(gastric fluid) to solid ratio, because authors stated that 1000:1 ratio provided the most representative extractability for most metals.

[b]   Range represent values recorded at various liquid to solid ratios (100:1 to 5000:1)

[c]   Incorporated iron hydroxide gel in an in vitro procedure to stimulate intestinal absorption, results showed statistical resemblance to *in vivo* bioaccessibility test results

[d]   Reported values that were similar to the results of the *in vivo* method of assessing bioaccessibility

[e]   Results presented as relative bioavailability factors calculated by expressing the migratable metal content as a percentage of the total metal content for each sample medium

[f]   Originally developed for toy safety.

[g]   A dosing vehicle (wet feed) equivalent to the amount of gastric extraction solution.

[h]   Authors claim that The PBET method of extraction which do not used food in the extraction to mimic fasting conditions, has been correlated with relative bioavailavle Pb as estimated by two animal models (weanling rats and swine). The authors also claim that the IVG method is an accurate predictor of relative bioavailable As in contaminated soils and waste materials as estimated by a juvenile swine model while utilizing food in the extraction procedure.

[i]   Data presented as relative bioaccessibility (*bioaccessibility of Pb from the test substrate was corrected for the recovery of a soluble Pb spike)* based on gastric phase bioaccessibility data from PBET, the range represent the range of bioaccessibility data from the gastric phase based on time.

[j]   Data presented as relative bioaccessibility based on intestinal phase bioaccessibility data from PBET, the range represent the range of bioaccessibility data from the intestinal phase based on time.

[k]   Data presented as relative bioaccessibility *(calculated as average soluble As mass in small intestinal simulation divided by total As mass in the reaction vessel, corrected for recovery of soluble As spike in small intestinal simulation)* based on intestinal phase bioaccessibility data from PBET, the range represent the range of bioaccessibility data from the intestinal phase based on time.

The biggest uncertainly in any *in vitro* bioaccessibility test is knowing how closely the values relate to human bioavailability. No *in vitro* method can recreate the physiological process of the human gastrointestinal tract, the presence of food, and the effect of microbial communities perfectly. Standardized soil reference material and bioaccessibility methods may improve human bioaccessibility data. In the absence of human bioaccessibility data, the best validation of the *in vitro* results has been done using *in vivo* studies in rats and pigs (Sips *et al.*, 2001).

Bioaccessibility does not only vary between substances, it also varies from site to site and between the compositions of different contaminants (EAUK, 2005b). As observed in Table 2-15, the bioaccessibility of lead and arsenic varied greatly in different geographical locations. Geological differences between land uses will affect the properties of metals within the soil as well (EAUK, 2005b). For example, the addition of phosphates or organic matter in soil will desorb non-bioaccessibile metals bound to iron oxides and convert them into bioaccessible forms (EAUK, 2005b).

Another uncertainty regarding bioaccessibility is the comparability of the *in vitro* results within and between laboratories (EAUK, 2005b), which are based on such factors as differences in procedures (*e.g.*, source of glassware, solvents, reagents, *etc.*, preparation prior to bioaccessibility testing, instrumentation calibration, *etc.*). Human error will also contribute to inter- and intra-laboratory differences in results. There is no single method suitable for all metals, since such a method would need to satisfy the key requirements such as simulation of the human gastrointestinal conditions, simplicity and cost effectiveness (Danish EPA, 2003).

The use of bioaccessibility in risk assessment is considered to be valid and applicable at specific sites, provided that some basic data requirements are fulfilled to permit regulatory groups (*e.g.*, Health Canada) to ascertain the validity and defensibility of such measurements (Health Canada, 2007). The following parameters have been suggested as relevant for investigation at specific sites:

1. Soil particle size – Researchers have suggested that the <250 um fraction, typically used in risk assessment, may not be representative of the soil size fraction of greatest relevance to human health risk assessment. Therefore studying a range of soil particle sizes, to ascertain if the particle size affects the bioaccessibility, has been proposed (Health Canada, 2007).

2. Ratio of simulated bioaccessibility fluid volume to soil mass – Researchers have observed that, for some elements and soils, the ratio of the fluid to the soil mass during the bioaccessibility extraction can affect the bioaccessibility (Health Canada, 2007). To determine if this variable affects the bioaccessibility results, measurements with varying liquid volume to soil mass ratios are recommended (Health Canada, 2007).

2.9.2 Approach for Bioaccessibility Testing in this Study

The bioaccessibility study was designed in two parts: the first was to investigate the two parameters mentioned above (soil particle size and the ratio of simulated bioaccessibility fluid to soil mass) for three soils and determine the most appropriate method for measuring bioaccessibility, and the second was to measure the bioaccessibility in the remaining soils using the selected method (Method 2).

Soil samples collected as part of the Residential Lawn and Garden Soil Sampling for Metals Study (Jacques Whiteford, 2008) were utilized for the bioaccessibility study. The bioaccessible arsenic, cadmium, copper, lead, mercury, selenium, and zinc content in Flin Flon area soils was determined using an *in vitro* physiologically based extraction test (PBET) methodology coupled to elemental detection (ASU and ESU, 2008). The *in vitro* extraction consisted of a two-phase protocol (*i.e.*, simulating both gastric and intestinal phases of absorption) designed to simulate a human receptor. It is a modification of two previously published methods, that of Ruby (1993; 1996) and Rodriguez and Basta (1999) incorporating the 100:1 liquid to solid ratio, low molecular weight organic acids, pepsin, bile and pancreatin concentrations used by Ruby and the NaCl and pH conditions of Rodriguez. It should be noted that while this study employed an *in vitro* procedure to determine bioaccessibility of all of the COC in Flin Flon soils, published *in vivo* validation of these methods has only been conducted for lead and arsenic and as such only results for lead and arsenic have been utilized in the HHRA.

Various stages of the bioaccessibility study were conducted at different research laboratories. The ESG laboratory (Analytical Sciences Division) at the Royal Military College of Canada conducted the in vitro bioaccessibility extractions to measure the bioaccessible arsenic, cadmium, copper, lead, mercury, selenium, and zinc content of the soil samples. The Analytical Services Unit (ASU) at Queens University in Kingston, Ontario carried out the analysis of the resulting extracts, as well as the analysis of total metals and metalloids in the <45 µm fraction of three soils in Part 1 of the study. The Analytical Sciences Group (ASG) at the Royal Military College, Kingston, Ontario digested several of the bioaccessibility extracts prior to analysis at ASU and analyzed mercury in the <45 µm fraction of two of the three soils used in Part 1. The total concentration of the elements of interest in the <250 µm fraction of the soils was determined as part of the Residential Lawn and Garden Soil Sampling for Metals Study (Jacques Whiteford, 2008). A detailed description of the specific extraction procedures and quality control methods are provided in Appendix G (ASU and ESU, 2008).

2.9.1.1   Part 1 Analyses: Method Validation

For Part 1 of the study, three methods were tested on three soils (FF208B, CS102B, FF276F), in triplicate (Table 2-16). The first method (Method 1) used an extraction fluid volume:soil mass of 100:1 and the <45 µm fraction of the soils; the second method (Method 2) used an extraction fluid volume:soil mass of 100:1 and the <250 µm fraction of the soils; and the third method (Method 3) used an extraction fluid volume:soil mass of 2,000:1 and the <250 µm fraction of the soils. The full summary of results is provided in Appendix G (ASU and ESU, 2008).

Table 2-16	Summary of Part 1 Bioaccessibility Testing					
Average[a] Bioaccessibility (%)						
		Phase 1 (Gastric)			*Phase 2 (Gastric + Intestinal)*		
COC		*Method 1*	*Method 2*	*Method 3*	*Method 1*	*Method 2*	*Method 3*
		<45 100:1	<250 100:1	<250 2,000:1	<45 100:1	<250 100:1	<250 2,000:1
Mercury	FF208B	0.14	0.20	<1.7	1.1	1.6	4.6
	CS102B	<1.7	<2.6	<50	4.6	4.7	<51
	FF276F	0.060	0.11	<0.4	1.0	1.2	2.9

Table 2-16	Summary of Part 1 Bioaccessibility Testing					
Average[a] Bioaccessibility (%)						
COC		*Phase 1 (Gastric)*			*Phase 2 (Gastric + Intestinal)*		
		Method 1	*Method 2*	*Method 3*	*Method 1*	*Method 2*	*Method 3*
		<45 100:1	<250 100:1	<250 2,000:1	<45 100:1	<250 100:1	<250 2,000:1
Arsenic	FF208B	39	56	85	30.4	42	92
	CS102B	12.1	17.9	BND	11.8	16	BND
	FF276F	35	39	<63	32	41	67
Cadmium	FF208B	78	86	BND	42.9	43	BND
	CS102B	85	65	BND	46	36	BND
	FF276F	79	78	81	30	36	71
Copper	FF208B	36	58	83	29.8	43	66
	CS102B	31	24.7	34	28	23.6	45
	FF276F	35	37	51	26	25	59
Lead	FF208B	85	85	BND	17	15	43
	CS102B	70	55	74	7.5	7.3	36
	FF276F	59	49	74	6.3	6	33
Selenium	FF208B	<8.1	<19	BND	<8.1	17	BND
	CS102B	SND	BND	BND	SND	BND	BND
	FF276F	<1.6	<3.5	BND	1.6	4.1	BND
Zinc	FF208B	69	82	BND	31	36.8	33
	CS102B	61	54	BND	27	20	<93
	FF276F	65	60	77	23	21	37

BND = % bioaccessibility cannot be calculated because the total concentration is less than the bioaccessible concentration

SND = % bioaccessibility cannot be calculated because the total concentration is less than the detection limit.
[a]   No substitutions for undetectable values were made when calculating averages.

The bioaccessibility results were generally higher using Method 3. However, the higher dilution used in Method 3 (2,000:1) resulted in a large number of undetectable sample results (56% of all measurements), in spite of the attempt to use three samples that had relatively high concentrations of metals and metalloids in them. Therefore, to obtain as many detectable results as possible, it was recommended that Method 3 not be used for the remaining samples, since the detection was compromised with this method and the remainder of the samples have lower concentrations of contaminants than the three samples analyzed in Part 1 (ASU and ESU, 2008).

Statistical analysis revealed that although the smaller particle size (<45 µm) may have higher contaminant concentrations, the chemical or mineralogical form of these contaminants is less bioaccessible when bioaccessibility is expressed as a percent of the total concentration. Thus, since the statistics do not support the use of the <45 µm fraction (Method 1), bioaccessibility analyses proceeded using the <250 µm fraction (Method 2) (ASU and ESU, 2008).

2.9.1.2   Part 2 Analyses

Soil collected as part of the surface soil concentration study was again utilized for part two of the bioaccessibility analyses. Bioaccessibility testing was conducted on a total of 50 soil samples. Table 2-17 outlines the number of samples tested from each community. Sample specific results are included as part of Appendix G (ASU and ESU, 2008).

Table 2-17	Summary of Soil Locations for Bioaccessibility Testing
COI	*Soil Sample ID Number*
Creighton (n=10)	CS102B, CS104F, CS104B, CS108F, CS111F, CS112B, CS112G, CS114F, CS125F, CS129F
West Flin Flon (n=20)	FF201F, FF202F, FF206F, FF206B, FF208B, FF209B, FF210B, FF221F, FF225B, FF229F, FF231F, FF232F, FF235B, FF239F, FF254F, FF258F, FF271F, FF276F, FF277F, FF278F
East Flin Flon (n=10)	FF302F, FF305B, FF307F, FF310F, FF312B, FF323F, FF324B, FF341F, FF343F, FF344F
Channing (n=10)	FF401B, FF402F, FF402B, FF402G, FF404F, FF406F, FF408F, FF408B, FF408G, FF410

Table 2-18 provides the results of the bioaccessibility analyses on soil samples conducted for both gastric and gastric+intestinal phases, where available. In the case of most of the COC evaluated in the current assessment, bioaccessibility provides a reasonably conservative approximation of the overall bioavailability.

Table 2-18		Summary of Part 2 Bioaccessibility Results (%)							
	Total		*Creighton*		*West Flin Flon*		*East Flin Flon*		*Channing*	
COC	*mean*	*95 UCLM*	*mean*	*95 UCLM*	*mean*	*95 UCLM*	*mean*	*95 UCLM*	*mean*	*95 UCLM*
Phase 1 (Gastric)										
Mercury	1.2	1.5	1.8	3.0	0.4	0.8	1.4	1.9	1.9	3.5
Arsenic	28	33	15	18	15	42	29	32	35	45
Cadmium	83	87	79	92	89	96	83	91	76	85
Copper	40	43	37	43	46	50	39	44	32	38
Lead	65	69	50	57	69	74	75	86	64	74
Selenium	42	55	64	NC[2]	28	38	91	NC	NC	NC
Zinc	64	68	60	72	67	72	72	77	56	62
Phase 2 (Gastric + Intestinal)										
Mercury	3.1	3.5	4.7	6.9	2.6	3.1	2.1	3.1	3.3	4.3
Arsenic	33	37	21	25	37	42	30	33	41	55
Cadmium	42	46	35	41	46	50	36	39	49	60
Copper	33	35	30	35	34	37	29	32	37	45
Lead	11	17	8	10	10	12	8	9	17	35
Selenium	43	57	64	NC	28	40	91	NC	NC	NC
Zinc	23	25	18	23	27	30	21	25	21	25

%   Bioaccessibility = Bioaccessible Concentration (mg/kg)/Total Concentration (mg/kg) x 100%
NC Not calculated. Due to concentrations below detection limit, bioaccessibility values could not be calculated for selenium.

A comparison of the Phase 1 (gastric) and Phase 2 (gastric + intestinal) total bioaccessibility (%) results indicated that cadmium, copper, lead and zinc bioaccessibilities were generally higher in Phase 1, while bioaccessibilities for arsenic, selenium and mercury were slightly higher in Phase 2 (see Figure 2-1). A paired t-test comparing percent bioaccessibility values between phases showed significant differences for all elements (ASU and ESU, 2008), while particularly notable differences in the bioaccessibilities of cadmium, lead and zinc between the gastric and gastric+intestinal phases was evident. Phase 1 (gastric) absorption was selected to represent COC bioaccessibility in the gastrointestinal tract for purposes of the risk assessment as it provides a comparable (for arsenic, mercury and selenium) or more conservative (for cadmium, copper, lead and zinc) bioaccessibility estimate than the use of Phase 2 (gastric + intestinal) bioaccessibility.
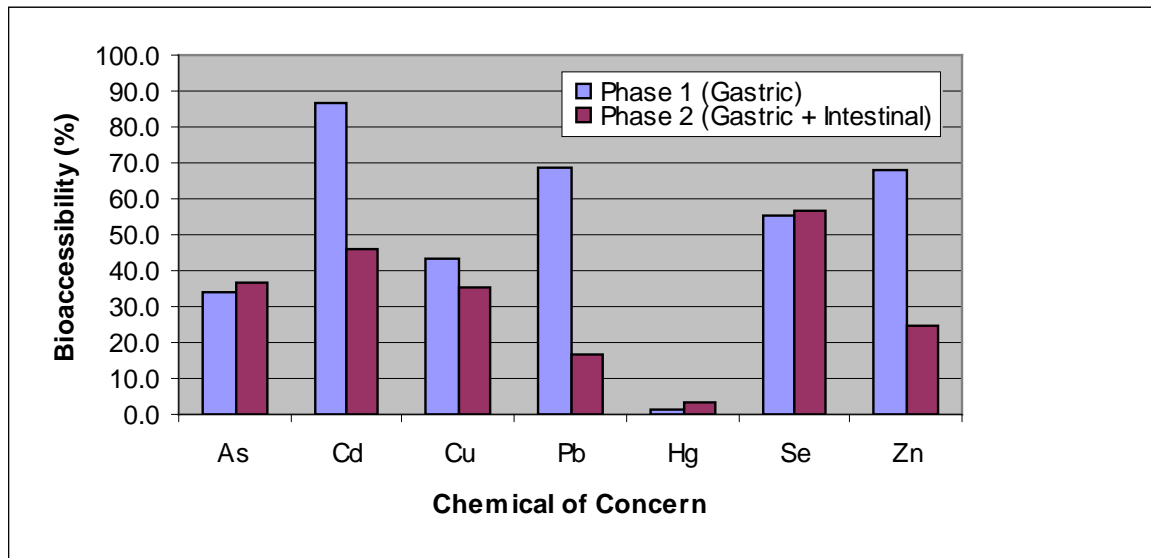
Figure 2-1   Phase 1 (Gastric) *versus* Phase 2 (Gastric + Intestinal) Bioaccessibility (%) of COC

While the gastrointestinal absorption of lead (and all chemicals for that matter) in humans occurs in two organs: the stomach (gastric phase), and the small intestine (intestinal phase), only the gastric phase has been validated for *in vitro* bioaccessibility testing for lead and only the gastric phase bioaccessibility results for lead are considered acceptable by many regulatory authorities (including Health Canada, U.S. EPA, and OMOE). This is somewhat counter-intuitive as research has shown that for metals the actual absorption occurs in the small intestine, with very small to negligible amounts being absorbed in the stomach (Mushak, 1991). While lead and probably most other metals will certainly be solubilized at stomach pH, it is important to remember that solubilization does not necessarily equal uptake.

While bioaccessibility provides one aspect of the potential uptake of a particular COC, a variety of other factors can affect the overall bioavailability of lead, including:

- GI absorption is dependent on soluble forms of lead; insoluble forms are very poorly absorbed;
- GI absorption is higher in children than in adults (*i.e.*, 40 to 50% of ingested lead absorbed in children *versus* 3 to 10% of ingested lead absorbed in adults); data are limited for older children and adolescents;
- GI absorption increases markedly if subjects are fasted;Nutritional status also influences GI absorption of lead (*e.g.*, most well characterized influences are iron and calcium; Iron and calcium deficient subjects absorb greater amounts of lead than non-deficient subjects);
- Lead absorption may increase during pregnancy;
- Gastrointestinal absorption of inorganic lead occurs primarily in the duodenum (Mushak, 1991); exact mechanisms of absorption are unknown and may involve active transport and/or diffusion through intestinal epithelial cells or between cells, and may involve ionized lead (Pb+2) and/or inorganic or organic complexes of lead;
- WHO documents on lead report that 50% of soluble lead is absorbed in the small intestine;

- Saturable mechanisms of absorption have been inferred from measurements of net flux kinetics of lead in the *in situ* perfused mouse intestine, the *in situ* ligated chicken intestine, and *in vitro* isolated segments of rat intestine; by analogy to other divalent cations, saturable transport mechanisms for lead may exist within the mucosal and serosal membranes and within the intestinal epithelial cell;
- Numerous observations of nonlinear relationships between blood lead concentration and lead intake in humans suggest the existence of a saturable absorption mechanism or some other capacity-limited process in the distribution of lead in humans; where the percentage of ingested lead that is absorbed may decrease with increasing rate of lead intake; however, still no firm basis for discerning if the gastrointestinal absorption of lead is limited by dose - the dose at which absorption becomes appreciably limited in humans is not known;
- This saturation observation can be seen in the curvilinear dose-response relationships between lead dose and blood lead, where the increment in blood lead per unit of intake decreases with increasing blood lead;
- Lead intake-blood lead relationships also vary with age as a result of age-dependency of gastrointestinal absorption of lead, and vary with diet and nutritional status (Mushak 1991);
- In immature swine that received oral doses of lead in soil, lead dose-blood lead relationships were nonlinear (curvilinear); however, dose-tissue lead relationships for bone, kidney, and liver were linear. The same pattern (nonlinearity for blood lead and linearity for tissues) was observed in swine administered lead acetate intravenously (Casteel *et al.*, 1997). This raises the question of whether there is an effect of dose on absorption or on some other aspect of the biokinetics of lead;
- Rat studies with lead acetate also suggest a capacity-limited process at the level of the intestinal epithelium;
- The U.S. EPA IEUBK model assumes that GI absorption of lead is sum of a capacity limited process and unlimited process; fractional absorption is decreased at high intake levels (>5 mg/kg/day); and,
- Blood lead levels are linked to both external intakes and interactions with lead that deposits in bone tissue (*i.e.*, resorption, formation processes can increase or decrease blood lead independently of external exposures, depending on age and physiological status, such as pregnancy, menopause, osteoporosis, prolonged immobility).

As such, while the bioaccessibility of lead is certainly an important factor, there are a variety of other factors which impact upon the bioavailability of lead and the ultimate potential of the exposure to lead to adverse effects. The gastrointestinal absorption of lead varies depending on a number of factors including speciation, solubility, particle size (if lead is in a matrix like food or soil), the exposure medium (*e.g.,* food, water, soil), and the age and physiological state of the exposed animal (*e.g.,* fasting or fed, nutritional status, pregnancy status, *etc.*).

The results of the bioaccessibility study indicates that in total as much as 69% (representing the 95% UCLM) of the lead present in Flin Flon soils becomes solubilized (*i.e.*, is available for absorption) in the gastric phase of the study. Drexler and Brattin (2007) have related relative *in vivo* bioavailability (RBA) and *in vitro* bioaccessibility (IVBA) estimates from a large dataset of lead-contaminated soils and wastes. A highly significant correlation coefficient between the two sets of data was found and the following linear regression equation relating the two derived:

$$RBA = 0.878 * IVBA - 0.028$$

This equation allows an estimate of RBA when only IVBA is known. In the current study, the IVBA estimates for lead (69% for soil) results in estimates for soil RBA of 58%. These values

were utilized in the current assessment.

2.9.3 Recommended Relative Absorption Factors

As noted previously, a RAF based on a bioaccessibility evaluation is a simple quotient comparing the solubility of contaminants in soil and the exposure medium used to develop the RfD/CSF (*i.e.*, spiked food) in simulated digestive fluids. Table 2-19 provides the recommended RAFs for each of the COC used in the current human health risk assessment.

Table 2-19	Summary of Recommended Relative Absorption Factors (RAF) for Soil
Chemical	**RAF**
Arsenic	33
Cadmium	100
Copper	100
Lead[a]	58
Mercury	100
Selenium	100

[a] The RAF for lead in soil has been adjusted based on the Drexler and Brattin (2007) regression equation.

The complete bioaccessibility report, including each analytical report, can be viewed in Appendix G, and a discussion of the application of these bioaccessibility values is provided in Chapter 4. It is important to keep the purpose of the bioaccessibility study in context. The purpose of the study was to estimate the relative difference in bioaccessibility between metals in soil from the Flin Flon area, and those used in the toxicological studies used to derive the TRVs utilized in the HHRA. The study was not intended to measure the absolute bioavailability of metals in soil and dust from the Flin Flon area. Since the results of the study are used in a relative manner, these uncertainties are not expected to significantly affect the results or conclusions of the HHRA.

2.10 Flin Flon and Creighton Food Survey Summary

Flin Flon and Creighton area residents participated in a food consumption survey in the winter and spring of 2008 (see Appendix P for a copy of the survey and associated map). The response rate was approximately 17%, which included a return number of approximately 500 surveys out of a total of 3,000 which were distributed for completion. Overall, the purpose of the survey was to gather information on the local foods that may commonly be collected and consumed from the Flin Flon/Creighton area. This included local fish, wild game, wild berries and wild mushrooms. Specifically, information on the location where the food item was collected (fished, hunted or gathered from), the type of food, the frequency that food items were consumed, and the season when these foods were generally consumed was determined. As well, information was gathered regarding which family members consume local food items.

Results from the survey were used as a basis for the development of a sampling plan for local food sources to ensure that the proposed food collection that was completed in the summer and fall of 2008 was specific to the local area residents and relevant for the purposes of the HHRA.

2.10.1 Fish

Results from the fish consumption data analysis indicated that approximately 99% of the respondents consume locally caught fish. Specifically, 97% of adults, 90% of teens, 90% of children and 87% of toddlers represented by the survey eat locally caught fish from Flin Flon

and surrounding areas. Seasonal trends for local fish consumption were also determined and results showed that almost half (45%) of households consume locally caught fish all year round, while 27% consume local fish in the summer and 11, 9 and 7% eat local fish in the spring, winter and fall respectively. Also, the majority of survey respondents indicated that they most frequently consume fish between 1 to 1.5 times per week but that this frequency was highest during the summer months. The questionnaire did not ask residents to indicate the serving sizes of local foods as it is often very difficult to estimate the mass of food items such as fish fillets, wild berries, or wild game servings. There is significant room for error in this estimation and data can often be highly skewed. As a result, a generic meat serving of 8 oz. is recommended by a number of sources for predicting exposure from the consumption of impacted meats. This is discussed further in Chapter 4.

Overall, these results strongly indicate that locally caught fish is a significant food source which needed to be sampled for further analysis. As well, since summer is the most common time of year to catch and consume local fish it was recommended to be the season in which fish sampling should occur.

In terms of fish type, the top four fish species that residents reported catching and consuming from local area lakes comprised 97% of all the fish caught and consumed, were Pickerel (66%), Northern Pike (16%), Trout (10%), and Perch (5%) (Figure 2-2). Also, the survey indicated that of those residents that reported consuming local fish, only 16% (73 of 457) reported that they consume parts other than the filet.

Other 2 %   Not Identified 1 %

Northern Pike 16 %

Trout 10 %

Perch 5 %

Pickerel 66 %
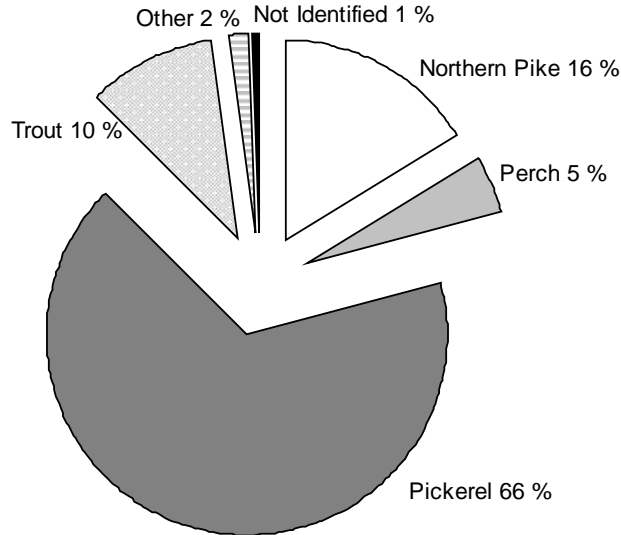
Figure 2-2   Percentage of Survey Respondents Reporting Catching and Consuming Various Species of Local Fish (Survey Entries for Fish = 1,065)

The local food survey distributed to residents included a local map (see Appendix P) over laid by concentric rings moving out from Flin Flon in 5 km increments. Moving outwards from Flin Flon, Areas 1 to 7 were labeled and used for the purpose of identifying where local food items were obtained. The results of the survey indicated that the most common Areas where fish are caught for consumption purposes were Area 5, which is approximately 25 km from Flin Flon (39% of survey respondents reported catching fish in this area); and Area 6, which is 30 km from Flin Flon (23% survey respondents reported catching fish in this area). Approximately 9% of survey respondents indicated that they caught fish from lakes outside of the map area (> 40 km from Flin Flon), and 6% indicated catching fish from lakes in Area 3 (approximately 15 km from Flin Flon). The survey indicated that less than 5% of respondents reported catching fish in each of the remaining Areas (Figure 2-3).

Area 1 (1 %)
Area 2 (4 %)
Not Identified (11 %)
Area 3 (6 %)
Area 4 (5 %)
Outside of area map (9 %)
Area 7 (2 %)
Area 5 (39 %)
Area 6 (23 %)
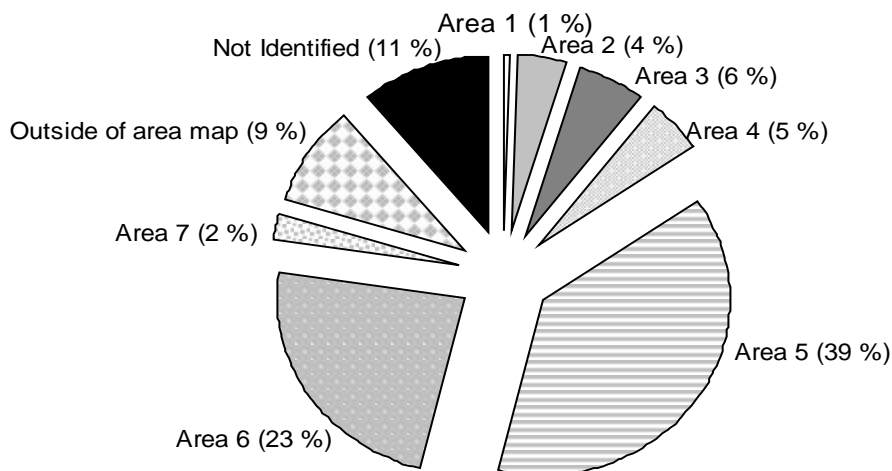
Figure 2-3   Percentage of Respondents that Reported Catching Fish in Each of the Designated Areas Surrounding Flin Flon (Survey Entries for Fish = 1,065)

For each Area, respondents identified the lakes from which they caught fish from regularly.

In Area 1 (5 km from Flin Flon), the main lake where people reported fishing was Cliff Lake (n=6). The remaining lakes that were used for fishing in this area were not identified. In Area 2 (10 km from Flin Flon), Big Island Lake (30%), Phantom Lake (20%) and Trout Lake (20%) were the principle lakes that were identified as primary fishing locations (Figure 2-4).
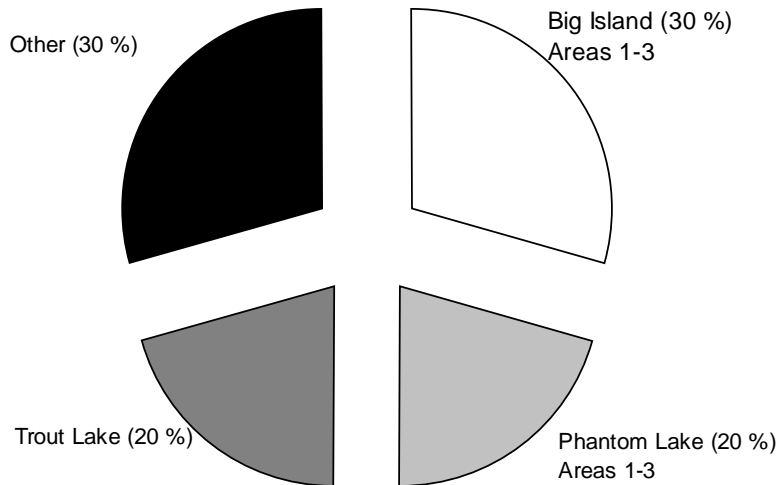
Figure 2-4   Percentage of Respondents that Reported Catching Fish From Lakes Identified within Area 2 (n=44)

In Area 3, the three lakes where the greatest amount of fishing was indicated were Mystic Lake (25%), White Lake (20%) and Birch Lake (15%) (Figure 2-5).
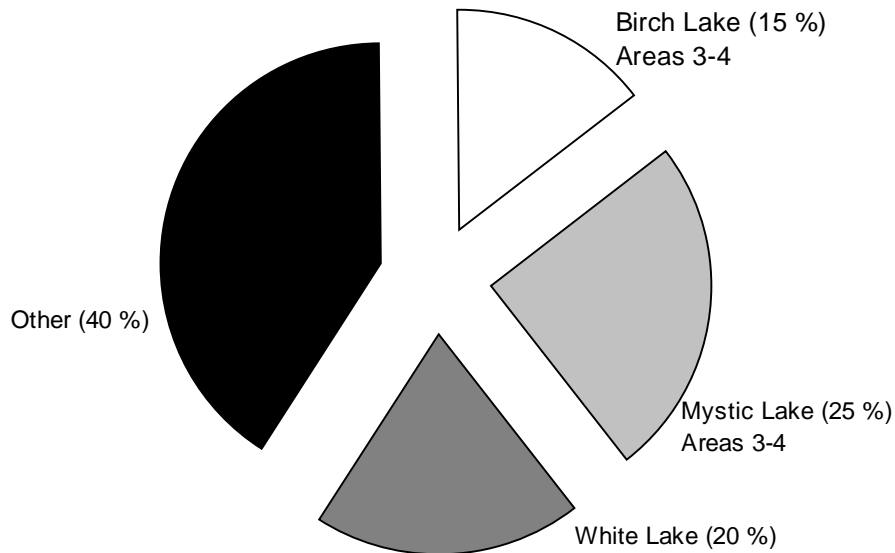
Figure 2-5   Percentage of Respondents that Reported Catching Fish From Lakes Identified within Area 3

In Area, 4 the most commonly identified fishing areas were Thompson Lake (29%), Denare Beach (20%), and Bakers Narrows (11%) (Figure 2-6).
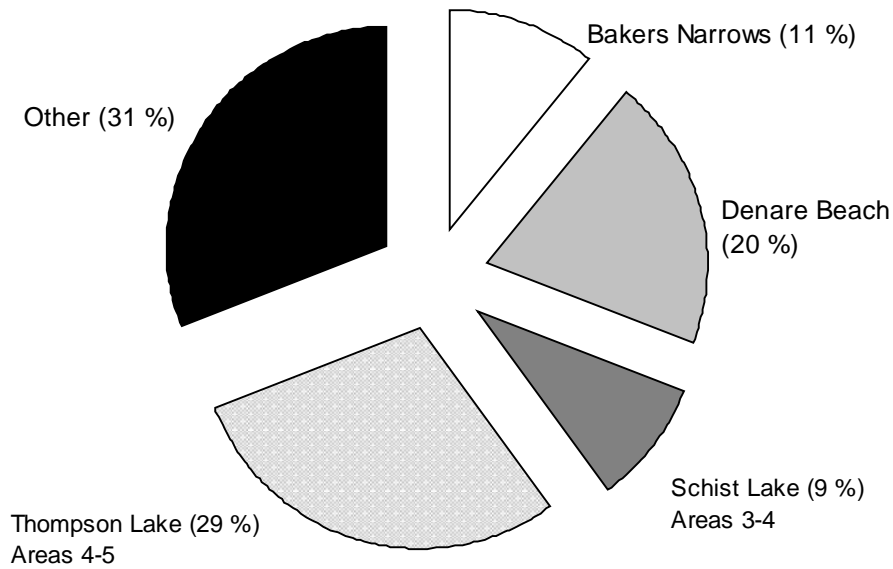
Figure 2-6   Percentage of Respondents that Reported Catching Fish From Lakes Identified within Area 4 (n=55)

In Area 5, which is approximately 25 km from Flin Flon, the two main lakes/areas where fishing was reported were Athapapuskow (38%) and Kisseynew (31%) (Figure 2-7).
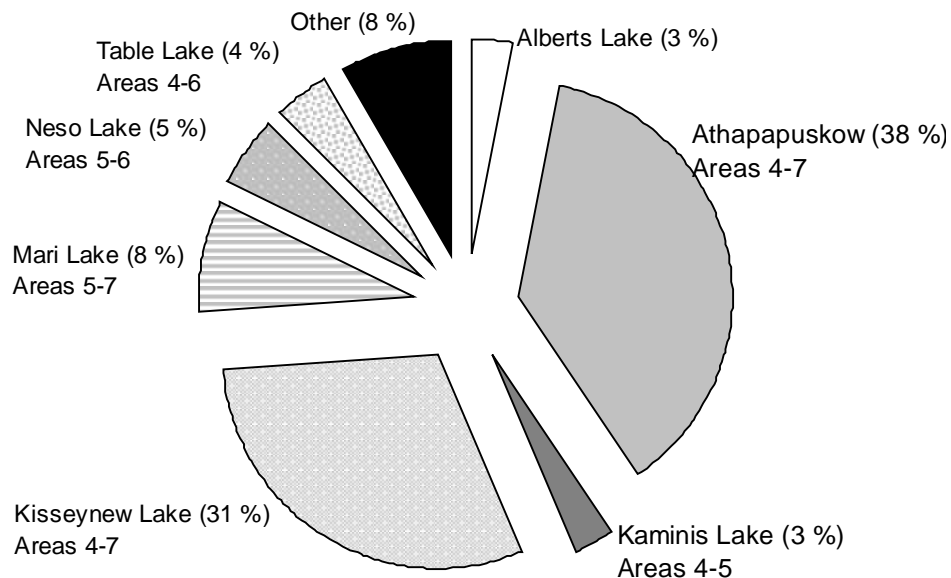
Figure 2-7   Percentage of Respondents that Reported Catching Fish From Lakes Identified within Area 5

Area 6, which is approximately 30 km from Flin Flon, the three main lakes where fishing was reported were Amisk Lake (46%), Maraiche Lake (24%) and Naosap Lake (13%) (Figure 2-8).
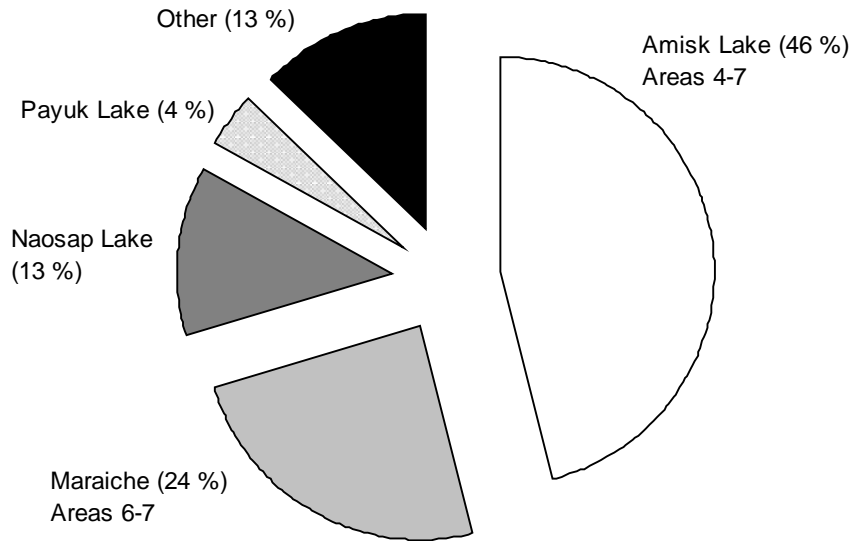
Figure 2-8   Percentage of Respondents that Reported Catching Fish From Lakes Identified within Area 6

Area 7 was not commonly reported as an area where survey respondents indicated fishing. Lakes reported are indicted in Figure 2-9.
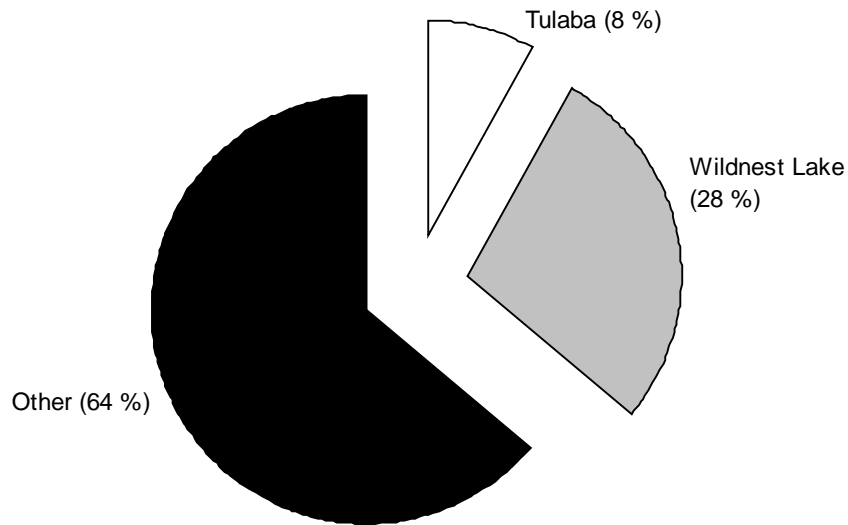
Figure 2-9   Percentage of Respondents that Reported Catching Fish From Lakes Identified within Area 7 (n=25)

Lastly, Area 8 (outside of the map area given to survey respondents), which is greater than 40 km from Flin Flon, was indicated to have several lakes/areas where local residents commonly fish (Figure 2-10).
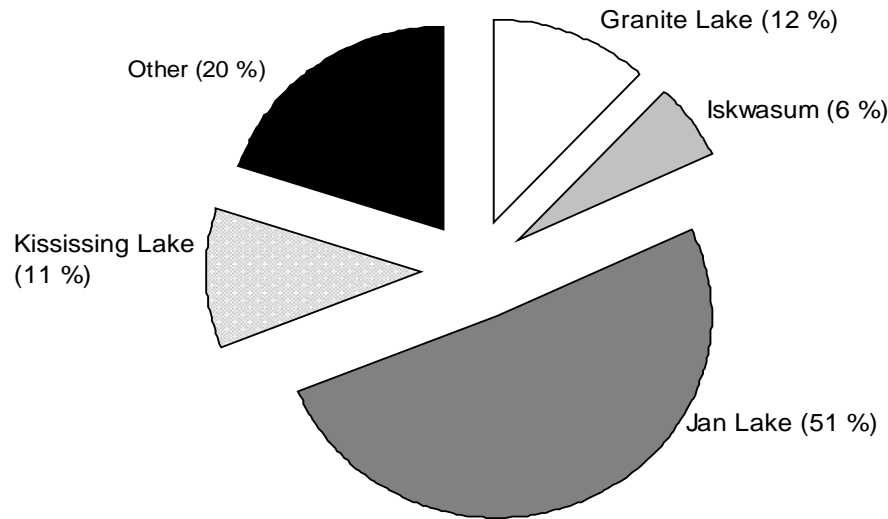
Figure 2-10   Percentage of Respondents that Reported Catching Fish From Lakes Identified within Area 8 (n=100)

2.10.2   Wild Game

Results from the survey showed that there were fewer individuals in Flin Flon and area that consume wild game in comparison to locally caught fish. It was revealed that 53.9% of adults consume wild game, whereas the percentages decreased for teens, children and toddlers and the results were 40.8, 34.7 and 38.6%. Forty percent consume wild game all year round, followed by 27% who consume it in the fall, and 23% consuming wild game in the winter. Again, similar to fish consumption, individuals consuming wild game do so with a frequency of approximately 1 to 1.5 times per week most commonly. In particular, fall is when the greatest percentage of individuals consumes wild game 1 to 1.5 times per week.

Flin Flon and Creighton residents generally consume the muscle portion of the moose, game birds, chicken, and deer from the local area (Figure 2-11).

Trends from the survey results indicate that wild game is generally hunted from further away from the center of Flin Flon (Figure 2-12).
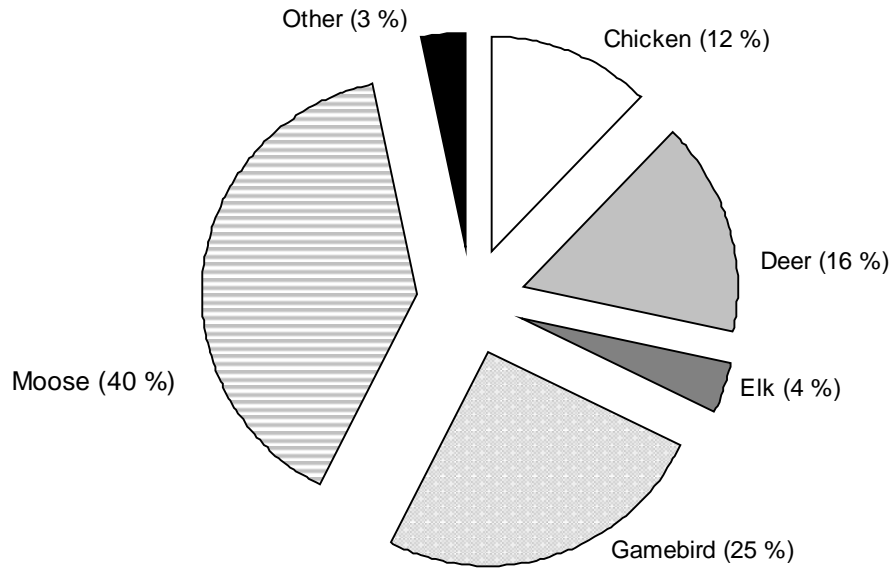
Figure 2-11   Type of Wild Game Caught by Local Area Residents of (Survey Entries for Wild Game = 480)
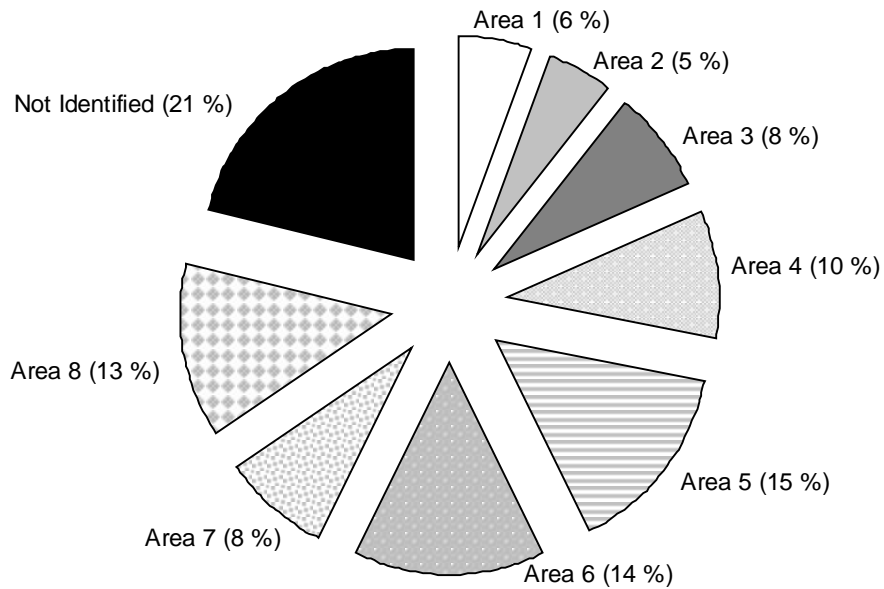

Figure 2-12   Percentage of Wild Game Gathered by Area Relative to Flin Flon (Survey Entries for Wild Game = 480)

2.10.3   Wild Berries

Wild berries, particularly blueberries, were reportedly consumed by approximately 85% of survey respondents. Specifically, approximately 90% of adults consume local berries followed by 80, 75 and 65% for teenagers, children and toddlers. Berries are consumed 38% across all seasons, however, 31% overwhelmingly indicated that they consumed berries in the summer season, followed by 21% in the fall. The least amount of berries were consumed in the winter (7 %) and spring (3%). Residents consuming berries did so most frequently 1-1.5 times per week.

In terms of quantities consumed, there was a high variability of responses, therefore, standardization of these responses needed to occur before the analysis could take place. Using the following estimates all data entries were converted to grams per season.

Estimates:

1.   Two cups of frozen blueberries = 275 grams (used for all types of berries);
2.   Ice cream pail/tub = 2 L; and,
3.   In response to the question *How much local wild berries would your family eat per season*?, a "small amount" was assumed to be 500 mL.

Figure 2-13 demonstrates the wide range of responses from the Flin Flon Survey respondents concerning berry consumption. In order to compare the average Flin Flon berry consumption rates to the average Canadian berry consumption rate, the 95th percentile of berry consumption quantities for each season was calculated, averaged and converted to grams per day. The resulting value for the Flin Flon respondents' berry consumption is 103 g/household/day.
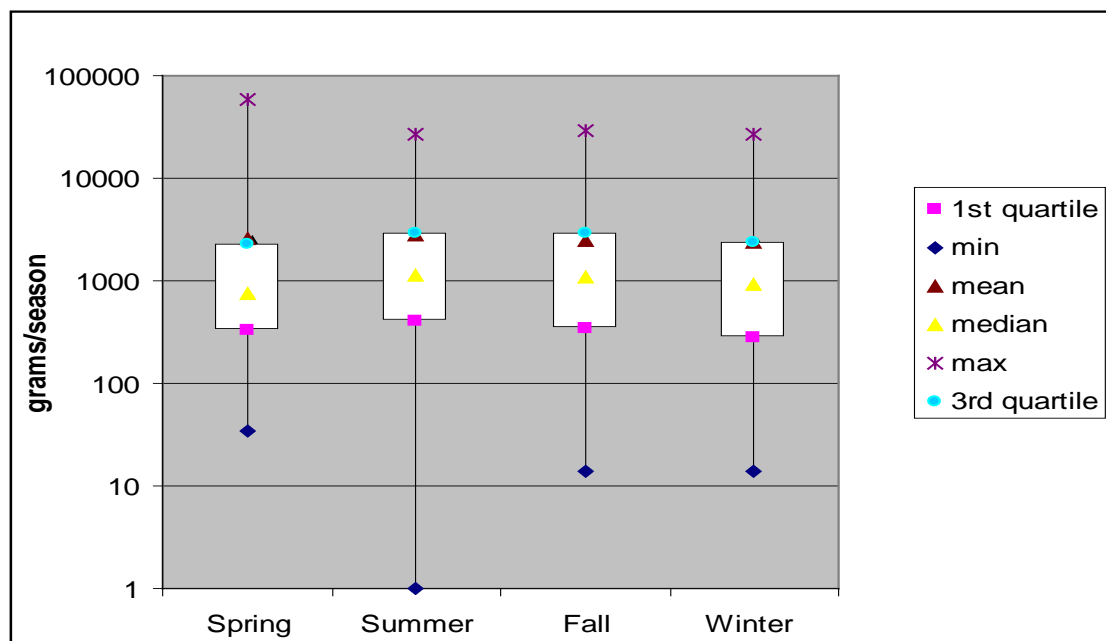

Figure 2-13   Survey Respondent's Berry Consumption Quantities for the Spring, Summer, Fall and Winter

Based on the survey respondents, there was an average of 2.5 individuals per household. Therefore, the 95th percentile berry consumption is 41.2 g/person/day.

Appendix A of the Health Canada document entitled Human Health Risk Assessment for Priority Substances, provides general Canadian population reference values of 1.64, 7.75 and 1.99 g/person/day for the mean consumption of cherries, strawberries, and blueberries respectively. Adding these values together provides a berry intake estimate of 11.38 g/person/day.

Given that the 95th percentile berry consumption rate based on the local food survey is 41.2 g/person/day, it is clear that the Flin Flon residents eat considerably more berries than the average Canadian per day. This is an indication that site specific food intake parameters are necessary for the Flin Flon HHRA.

Relative to Flin Flon, the most common areas reported for berry picking were Area 1 (33%) and Area 2 (18%) (Figure 2-14). Other berries that were reportedly picked and consumed include cranberries, pin cherries, and strawberries.
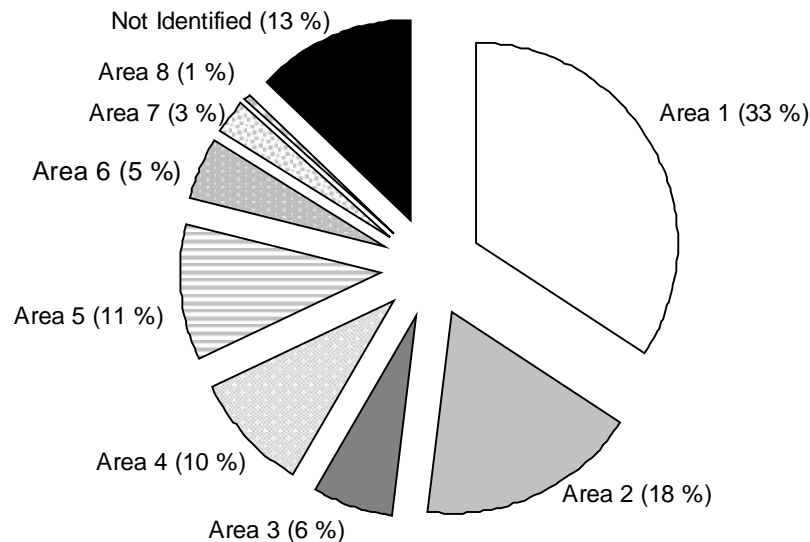

Figure 2-14   Percentage of Wild Berries Gathered by Area Relative to Flin Flon (Survey Entries for Wild Berries = 687)

2.10.4   *Wild Mushrooms*

Consumption of local wild mushrooms is not widespread in the Flin Flon-Creighton area. In total, approximately 15% of survey respondents indicated that they consume local wild mushrooms. Quantities of the mushrooms were quite variable. Figure 2-15 indicates that Area 1 appeared to be the most common for mushroom picking followed by Area 4 and Area 2.
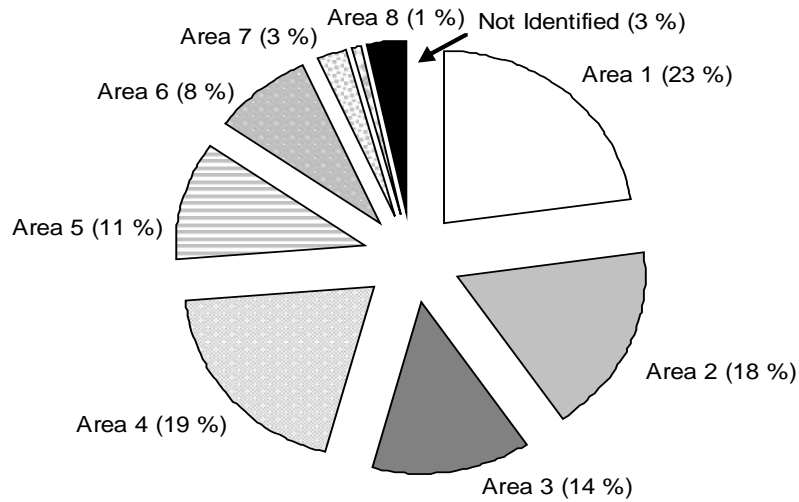
Figure 2-15   Percentage of Wild Mushrooms Gathered by Area Relative to Flin Flon (Survey Entries for Wild Berries = 118)

Figure 2-16 demonstrates the wide range of responses from the Flin Flon survey respondents concerning wild mushroom consumption. However, it is important to note that the information presented in this figure represents the majority of, but not the entire population of mushroom eating survey respondents due to lack of necessary information, such as type of wild mushrooms being consumed.
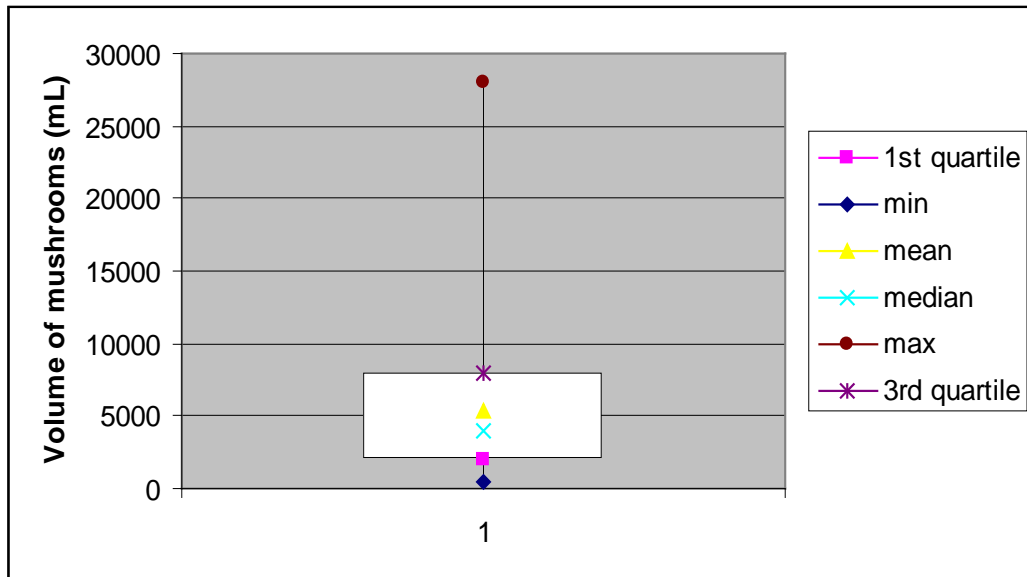

Figure 2-16   Volume of Wild Mushrooms Consumed by Local Residents (Six-number Data Summary)

Given the limited number of respondents that reported consuming wild mushrooms frequently, the assessment of exposure to COC through the consumption of wild mushrooms was not assessed in the HHRA.

2.11   References

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