CHAPTER 8

CONCLUSIONS AND RECOMMENDATIONS

CHAPTER 8:

CONCLUSIONS AND RECOMMENDATIONS

Table of Contents

Page

CHAPTER 8:

CONCLUSIONS AND RECOMMENDATIONS

List of Tables

Page

List of Figures

8.0 CONCLUSIONS AND RECOMMENDATIONS

This chapter provides a brief summary of the results of the HHRA. Separate assessments were completed for short-term (acute) and long-term (chronic) durations, with risk estimates provided for:

- Acute inhalation (24 hour durations);
- Acute oral (short-term soil and snow exposure events);
- Residential chronic multiple pathways (*i.e.*, inhalation, oral and dermal exposures); and,
- Commercial/industrial (outdoor worker) chronic multiple pathways (*i.e.*, inhalation, oral and dermal exposures).

Short-term or acute inhalation risk estimates are based on exposure periods that last from a few minutes to a few days and is characterized though consideration of 24-hour maximum air concentrations. For soil and snow pathways, acute exposure estimates are based on short-term transient exposure levels related to extreme activities and upper bound levels of COC in soil and/or snow. Short-term exposures are those defined as occurring over a less than lifetime duration or on an intermittent basis, and are typically characterized by a single pathway exposure only.

Long-term or chronic risks (>1 year to a lifetime) are characterized by comparing predicted exposures from all pathways with the exposure limits or toxicity reference values. Chronic health risks were estimated based on the assumption that an individual is continuously exposed to multimedia concentrations. The chronic risk estimates were based on an exposure duration of one year to an assumed lifespan of 80 years (Health Canada, 2006). For non-carcinogenic COC, this comparison is typically referred to as the Hazard Quotient (HQ) and is calculated by dividing the predicted exposure level by the exposure limit. If the total chemical exposure from all pathways is equal to or less than the exposure limit, then the HQ would be 1.0 or less, and no adverse health effects would be expected (refer to Chapter 4 for a more detailed discussion of this topic).

For chemicals with non-threshold-type dose responses (*i.e.*, carcinogens), the comparison is referred to as the Incremental Lifetime Cancer Risk Level (ILCR) or more simply a Cancer Risk Level (CRL). The ILCR represents an upper bound estimate of the additional incidence of cancer (*i.e.*, occurrence of cancer that would not be expected in the absence of the exposure) in a population of people exposed every day over their entire lifetime. The ILCR is calculated by multiplying the predicted exposure by the slope factor or unit risk value. The ILCR is expressed as the prediction that one person per n people would develop cancer, where the magnitude of n reflects the risks to that population. In the case of carcinogens, the acceptable risk level in Manitoba is considered to be an incremental increase in cancer risk of one-in-one hundred thousand (*i.e.*, one additional cancer per one hundred thousand people). Incremental lifetime cancer risks are calculated by multiplying a chemical- and route-specific cancer slope factor by facility related exposures.

8.1 Overview of Risk Assessment Results

8.1.1 Acute Endpoints

The results of the assessment of acute endpoints for each of the COC *via* the inhalation exposure pathways are summarized in Table 8-1.

Table 8-1 Acute Inhalation Risk Estimates		
COI and COC	***Exposure Duration***	***CR_{acute}***
West Flin Flon		
Arsenic	24 hrs	**2.5**
Cadmium	24 hrs	0.33
Copper	24 hrs	0.084
Lead	24 hrs	**1.2**
Mercury (inorganic)	24 hrs	0.028
Selenium	24 hrs	0.027
East Flin Flon and Channing		
Arsenic	24 hrs	0.73
Cadmium	24 hrs	0.060
Copper	24 hrs	0.016
Lead	24 hrs	0.20
Mercury (inorganic)	24 hrs	0.00016
Selenium	24 hrs	0.0089
Creighton		
Arsenic	24 hrs	0.10
Cadmium	24 hrs	0.0090
Copper	24 hrs	0.006
Lead	24 hrs	0.070
Mercury (inorganic)	24 hrs	0.0021
Selenium	24 hrs	0.0020

Bolded values highlighted in grey exceed a CR of 1.0.

Elevated risks associated with acute inhalation exposures are predicted for arsenic and lead in the West Flin Flon area only. By definition, acute exposure are short-term and transient in nature, typically occurring as a result of a unique or extreme weather condition or facility anomaly resulting in short-term deviations from average concentrations. The results of this evaluation indicated that some people may experience short-term and reversible health effects from the inhalation of arsenic and lead in air at intermittent times during facility operations, potentially occurring at times when air concentrations exceed the TRVs. These occurrences are somewhat rare (Table 8-2), the magnitude of exceedances are less than an order of magnitude, and the margins of safety inherent in the acute TRVs are large, indicating that the occurrence of acute health effects is unlikely. Table 8-2 indicates the number of samples collected for one year from September 2007 to August 2008 that are in excess of the acute TRV.

Table 8-2 Frequency of 24-hour CR_{acute} Exceedances						
COI	***Arsenic***	***Cadmium***	***Copper***	***Lead***	***Mercury***	***Selenium***
West Flin Flon	9 of 210	0 of 210	0 of 210	2 of 210	ND	ND
East Flin Flon and Channing	0 of 57	0 of 57	0 of 57	0 of 57	0 of 57	0 of 57
Creighton	0 of 59	0 of 59	0 of 59	0 of 59	0 of 59	0 of 59

ND – no data (short term air concentration predicted).

The results of the assessment of acute endpoints for each of the COC *via* the oral (soil and snow) exposure pathways are summarized in Table 8-3.

Table 8-3 Hazard Quotients for Acute Oral Exposures for Toddlers				
COI and COC	***Max Soil Concentration (µg/g)***	***HQ***	***Max Snow Concentration[a] (µg/L)***	***HQ_{acute}***
West Flin Flon				
Arsenic	237	**1.2**	147	0.03
Cadmium	71	0.42	183	0.05
Copper	7,810	NA	4940	0.50
Lead	820	NA	732	0.21

Table 8-3 Hazard Quotients for Acute Oral Exposures for Toddlers				
COI and COC	***Max Soil Concentration (µg/g)***	***HQ***	***Max Snow Concentration[a] (µg/L)***	***HQ_{acute}***
Mercury	971	**3.4**	1.8	0.0003
Selenium	286	NA	2.8	0.0005
East Flin Flon				
Arsenic	33	0.16	--	--
Cadmium	27	0.16	--	--
Copper	2,050	NA	--	--
Lead	333	NA	--	--
Mercury	18	0.062	--	--
Selenium	12	NA	--	--
Channing				
Arsenic	36	0.17	--	--
Cadmium	21	0.12	--	--
Copper	700	NA	--	--
Lead	266	NA	--	--
Mercury	7.0	0.024	--	--
Selenium	4.0	NA	--	--
Creighton				
Arsenic	300	**1.4**	--	--
Cadmium	32	0.19	--	--
Copper	1,800	NA	--	--
Lead	456	NA	--	--
Mercury	24	0.083	--	--
Selenium	18	NA	--	--

Bolded values highlighted in grey exceed an HQ of 1.0.
[a] Snow concentrations were not separated by COI.

For acute soil ingestion, marginal exceedances of the acceptable HQ (1.0) were noted for arsenic in West Flin Flon (HQ of 1.2) and Creighton (HQ of 1.4), and for mercury in West Flin Flon (HQ of 3.4). The results of this evaluation indicated that some people may experience short-term and reversible health effects such as irritation and gastrointestinal upset. However, the likelihood is rare, the magnitude of exceedances are small and the margins of safety inherent in the acute TRVs are large, indicating that the occurrence of acute health effects from acute soil ingestion is unlikely.

No acute HQ exceedances were noted for the snow pathway indicating that infrequent consumption of snow within the Flin Flon-Creighton area is not anticipated to result in adverse health effects.

8.1.2 Chronic Endpoints for the Residential Scenario

Long-term or chronic risks (>1 year to a lifetime) were characterized by comparing predicted exposures from all pathways with the TRVs. Chronic health risks were estimated based on the assumption that an individual is continuously exposed to COC in all media. The chronic risk estimates were based on an exposure duration of one year to an assumed lifespan of 80 years.

The results of the assessment of non-cancer endpoints for the toddler for each of the COC *via* the oral/dermal and inhalation exposure pathways are summarized in Table 8-4.

Table 8-4 Summary of Assessment Results for Non-Cancer Endpoints (Toddler)					
COC	***East Flin Flon***	***West Flin Flon***	***Creighton***	***Channing***	***Typical Background***
Arsenic	**1.5**	**1.9**	**1.7**	**1.5**	**1.2**
Cadmium	0.82	0.89	0.79	0.81	0.58
Copper	0.84	0.93	0.68	0.83	0.58
Lead	0.45	0.59	0.46	0.45	0.21
Inorganic mercury	0.33	**1.8**	0.35	0.30	0.20
Methyl mercury	**1.8**	**1.9**	**1.8**	**1.8**	0.42
Selenium	0.90	0.92	0.90	0.90	0.70

Bolded values highlighted in grey exceed the acceptable HQ of 1.0.

The results of the assessment of carcinogenic endpoints for arsenic and cadmium are presented in Table 8-5. No carcinogenic endpoints were evaluated for copper, lead, mercury and selenium for the current assessment.

Table 8-5 Summary of Assessment Results for Carcinogenic Endpoints (Lifetime Receptor)				
COC	***East Flin Flon***	***West Flin Flon***	***Creighton***	***Channing***
Arsenic	**3.0E-04**	**5.0E-04**	**1.7E-04**	**3.1E-04**
Cadmium	**2.5E-04**	**6.9E-04**	**4.5E-05**	**2.5E-04**

Bolded values highlighted in grey are in excess of the acceptable ILCR of 1.0×10^{-5}.

The chronic residential results are discussed further in Section 8.2.

8.1.3 Chronic Endpoints for the Outdoor Commercial/Industrial Workers

No unacceptable risks were predicted for any COC under an outdoor worker scenario for non-cancer effects (Table 8-6). Given that this scenario assumed that a worker would be exposed to the 95% UCLM concentration in soil every day spent at work during summer and winter months, it is anticipated that COC in soils found throughout the Flin Flon-Creighton area will not result in non-cancer health effects as a result of outdoor occupational activities.

Table 8-6 Hazard Quotients for an Outdoor Worker							
	Arsenic	***Cadmium***	***Copper***	***Lead***	***Mercury***	***Selenium***	***Methyl Mercury***
HQ	0.39	0.20	0.23	0.078	0.11	0.22	0.14

Exposure to arsenic and cadmium have been associated with elevated occurrences of various forms of cancer. The assessment of carcinogenic risks *via* the inhalation of arsenic and cadmium in ambient air for the outdoor worker would be similar to the assessment provided under the residential exposure scenario. In addition, the assessment of carcinogenic risks to arsenic *via* consumption of drinking water and market basket foods would be the same as those predicted under the residential scenario.

Under the outdoor worker scenario, chronic exposure to the 95% UCLM soil concentration is not anticipated to result in CRLs above the acceptable level of 1.0×10^{-5} (Table 8-7). This conservatively assumes that a receptor would spend their entire adult life working outdoors in the Flin Flon-Creighton area.

Table 8-7 Cancer Risk Estimates for an Outdoor Worker from Oral and Dermal Exposure to Arsenic in Soil	
Pathway	***ILCR***
Soil - Dermal Contact	2.7E-06
Soil - Ingestion	5.2E-06
Total Soil-Related ILCR	7.9E-06

8.1.4 Chronic Endpoints for the Recreational Scenario

Under a supplemental recreational assessment, it was assumed that receptors may spend a significant portion of the summer months swimming in local lakes. Exposure to COC was assumed to occur *via* incidental ingestion of surface water and sediment, as well as dermal contact of surface water with all skin. All HQs associated with incidental ingestion of surface water and sediment, and dermal exposure with surface water while swimming were very minor and are not considered to significantly contribute to overall risks for residential receptors or visitors to the Flin Flon-Creighton area (Tables 8-8 and 8-9).

Table 8-8 Hazard Quotients Associated with Exposure to COC in Surface Water and Sediment While Swimming

Pathway	***Arsenic***	***Cadmium***	***Copper***	***Lead***	***Mercury***	***Selenium***
Oral Exposure to Surface Water						
Toddler	0.015	0.0013	0.00012	0.00016	0.0092	0.00046
Child	0.0077	0.00063	0.000055	0.000080	0.0046	0.00023
Teen	0.0031	0.00026	0.000022	0.000033	0.0019	0.000094
Adult	0.0020	0.00017	0.000014	0.000021	0.0012	0.000066
Dermal Exposure to Surface Water						
Toddler	0.0019	0.00015	0.000015	0.000020	0.0011	0.000057
Child	0.0016	0.00013	0.000011	0.000016	0.00093	0.000046
Teen	0.0010	0.000080	0.0000069	0.000010	0.00058	0.000029
Adult	0.00071	0.000059	0.0000051	0.0000074	0.00043	0.000023
Oral Exposure to Sediment						
Toddler	0.050	0.060	0.012	0.040	0.0034	0.0064
Child	0.0062	0.0075	0.0013	0.0051	0.00043	0.00079
Teen	0.0034	0.0041	0.00072	0.0028	0.00024	0.00044
Adult	0.0029	0.0035	0.00060	0.0024	0.00020	0.00041
Total Exposure						
Toddler	0.067	0.061	0.012	0.041	0.0035	0.0070
Child	0.015	0.0082	0.0014	0.0052	0.00046	0.0011
Teen	0.0075	0.0045	0.00075	0.0028	0.00025	0.00057
Adult	0.0056	0.0037	0.00062	0.0024	0.00021	0.00050

Table 8-9 ILCR Estimates Associated with Exposure to Arsenic in Surface Water and Sediment While Swimming

Oral ILCR	**Dermal ILCR**	**Total ILCR from Swimming**
4.1×10^{-6}	3.9×10^{-7}	4.6×10^{-6}

Therefore, cancer and non-cancer risks associated with receptors swimming in lakes in the Flin Flon-Creighton area throughout the summer months are considered to be very minor and are not anticipated to result in the occurrence of adverse health effects.

8.2 Summary Discussion of Chronic Residential Results

Arsenic: Both non-cancer and cancer numerical risk estimates for arsenic exceeded standard acceptable benchmarks for both oral/dermal and inhalation exposures. The non-cancer TDI is protective of effects such as hyperpigmentation, keratosis, and possible vascular complications. Market basket foods were the main contributor to non-cancer arsenic-related risks under both the Typical Background scenario and the Flin Flon resident scenario. For carcinogenic risks, the inhalation of ambient air was the most significant source of risk. Consideration should be given to future reductions in smelter-related emissions, which would have a direct and immediate effect on reducing inhalation-related exposure and risks.

Although the consumption of drinking water and market basket foods are two of the largest sources of overall arsenic risk to residents of the COI, contributions from these sources are essentially equivalent to those experienced by receptors living in communities that are not impacted by a point source of arsenic emissions. The incremental increase in risks for Flin Flon area residents above background risks are primarily attributed to the incidental ingestion of soil and dust with elevated concentrations of arsenic and inhalation of ambient air.

The Community Health Status Assessment of Flin Flon and Creighton, completed by public health officials from Manitoba Health and Healthy Living and the Saskatchewan Ministry of Health found that the overall health status of the Flin Flon area population is as good if not better than the provincial averages for most of the indicators studied. It should be noted however, that non-melanoma skin cancers (NMSCs) had been excluded from the Community Health Status Assessment. This is generally consistent with international cancer statistics protocols and results from a lack of consistent reporting of these types of cancers. Given that one of the most common forms of cancer associated with arsenic exposure is skin cancer, the results of the Community Health Status Assessment may not have captured an increased incidence in all cancers that are possibly related to arsenic exposures in the Flin Flon area population relative to the Provincial averages.

The most powerful and persuasive piece of evidence in other weight-of-evidence evaluations was the urinary arsenic study results which compare urinary arsenic levels of an impacted community with those of a control community. It is recommended that a similar study be undertaken for the Flin Flon area, focusing on homes in West Flin Flon and Creighton in which a significant number of homes included within the residential soil sampling program contained concentrations of arsenic in excess of the PTC as presented below.

Cadmium: Oral/dermal and non-cancer inhalation exposures were within acceptable levels in all COI. It was not considered necessary to undertake more detailed assessment or actions related to cadmium in the study area.

Using the Health Canada cancer unit risk value indicates that concentrations of cadmium in ambient air may have the potential to result in an unacceptable increase in the risk of developing lung cancer for receptors spending a significant portion of their lifetime living in the Flin Flon area. Due to the conservatisms associated with the derivation of unit risk values and the acceptable cancer risk level of 1.0×10^{-5}, it is recommended that the results of the Community Health Assessment also be considered. It is noted that the Community Health Status Assessment would likely not be sensitive enough to detect an increase in cancer of the magnitude predicted in the HHRA given that the population of interest is only 8,000 to 9,000; however, it does provide some indication of current lung cancer incidence. That said, ILCRs for cadmium are elevated and consideration should be given to future reductions in smelter-related emissions, which would have a direct and immediate effect on reducing inhalation-related exposure and risks.

Copper: The estimated HQ values associated with copper exposures were less than 1.0 under all exposure and receptor scenarios. It was not considered necessary to undertake detailed assessment or actions related to copper in the study area. Overall, the health risks to Flin Flon-area residents associated with exposure to copper are within risk levels deemed to be acceptable by Health Canada and the CCME. Risk management measures or soil remediation are not considered to be necessary to prevent or reduce human health risks associated with exposure to copper in residential soils.

Lead: The assessment of lead exposure was completed using the excel-based HHRA exposure model used for all COC as well as the U.S. EPA IEUBK model. The Health Canada TRV of 3.6 µg/kg/day was used to assess risks based on the exposure predicted using the HHRA exposure model. The assessment completed using the IEUBK model predicted the probability of exceeding both a Blood Lead Level (BLL) of 10 µg/dL, as well as a BLL of 5 µg/dL, which has been suggested to be the level at which a decrease in cognitive ability may be affected in young children and the level currently in use in Ontario. Blood lead monitoring data is the most effective indication of recent exposure levels to lead from all sources. Given that blood lead data for the Flin Flon area is not available, the IEUBK model was used as an additional tool as it is widely acknowledged as a very effective method of assessing risks to young children from exposure to lead.

Using the HHRA model, estimated HQ values associated with lead exposures were less than 1.0 under all exposure and receptor scenarios. Using the U.S. EPA IEUBK model, BLLs were predicted for receptors in seven age categories for each of the four COI. The predicted BLLs represent the geometric mean for each age category calculated using standard assumptions that consider biological and behavioural differences in receptors, variability in repeat sampling, variability resulting from sampling locations, and analytical variability. Predicted BLLs for all age categories for all four COI and the Typical Background scenario, as well as the overall geometric mean concentrations, were well below 10 µg/dL. The communities of West Flin Flon and Creighton had higher predicted BLLs than East Flin Flon and Channing, but the highest predicted BLL of 5.6 µg/dL for the one to two years of age category was still well below 10 µg/dL. The probabilities of exceeding a BLL of 5 and 10 µg/dL are presented in Table 8-10.

Table 8-10 Blood Lead Levels in Children (0 to 7 Years) Predicted by the IEUBK Model (µg/dL)

Descriptive Statistic	***West Flin Flon***	***East Flin Flon***	***Creighton***	***Channing***	***Typical Background***
Geometric Mean	4.5	3.8	3.8	3.8	0.66
95^{th} Percentile BLL	9.7	8.2	8.2	8.2	1.4
Probability of exceeding a BLL of 5 µg/dL	41%	29%	28%	29%	0%
Probability of exceeding a BLL of 10 µg/dL	4.4%	2.1%	1.9%	2.1%	0%

Recognizing the limitations associated with using the IEUBK model for a community-based assessment, BLLs for each COI were predicted to represent the geometric mean value of a population of children exposed to homogenous lead levels under a similar exposure scenario. Results of this assessment indicate that geometric mean BLLs for each COI at the EPC soil and dust concentrations are well below 10 µg/dL. In addition, the 95^{th} percentile BLLs were also below 10 µg/dL for each COI. While these results may provide a general idea of the average community risk potential, the most appropriate method to assess risk would be to consider soil concentrations on a property by property basis (Section 8.3.4).

There are a significant number of residential properties in West Flin Flon that contain concentrations of lead in outdoor soil that are above the residential PTC protective of a BLL of 10 µg/dL as well as a few in East Flin Flon and Creighton. Assuming that the homes sampled as part of the residential soil sampling program are reflective of the distribution of lead throughout each of the COI, these results indicate that there is the potential that approximately 40% of properties in West Flin Flon contain levels of lead in excess of the PTC (refer to Table 8-15). A smaller percentage of properties in East Flin Flon (3%) and Creighton (13%) may also contain levels of lead in soil in excess of the PTC.

The assumptions and algorithms used within the IEUBK model are designed to provide a realistic assessment of the BLL in children exposed to lead through a number of exposure pathways. Since a significant percentage of homes in West Flin Flon and Creighton contain soil concentrations in excess of the PTC, the completion of a blood lead survey would be an appropriate method of reducing uncertainty in the exposure assessment and provide a more accurate measure of the levels occurring in young children in these communities. Based on the results of blood lead surveys completed in other communities in Canada such as Port Colborne, Ontario, measured BLLs are typically lower than those predicted using the IEUBK model. A blood lead survey should primarily focus on children up to the age of 7 years as they are the most sensitive to the impaired neurobehavioral development associated with elevated BLLs. Blood lead surveys are generally completed in late summer or early fall to assess children following a period of elevated exposure to outdoor soils. Given that lead has a half-life in blood of approximately 36 days, sampling during this time is likely to capture the highest BLL experienced by children throughout the year.

Mercury: The assessment of exposure and risk to mercury was completed for both organic (methyl mercury) and inorganic forms. While methyl mercury may be found in all forms of environmental media, it is generally recognized to be a small component of the total mercury measured in most. The fraction of methyl mercury in fish, drinking water, and ambient air is considered to be more significant. Therefore, the assessment of exposure and risks from methyl mercury was addressed *via* the consumption of local fish, market basket fish, ingestion of drinking water, and inhalation of ambient air.

Inorganic Mercury: HQ estimates for residents living in each of the COI ranged from 0.22 to 1.8, with the highest risk levels predicted for West Flin Flon. Health Canada has recommended an oral TDI for inorganic mercury of 0.3 µg/kg/day to be protective of adverse kidney effects. With the exception of the toddler in West Flin Flon, all HQs were equal to or below the acceptable value of 1.0 indicating that adverse effects associated with elevated exposure to inorganic mercury at the EPCs are not anticipated for most receptors. The highest risk levels were predicted to the toddler as a result of the elevated soil ingestion rate assumed for children of this age. Within West Flin Flon, exposure of the toddler to inorganic mercury, and subsequently risk levels, are dominated by contributions from soil. For all other receptors in each of the other COI, exposure and risk are dominated by the consumption of market basket foods and local fish, followed by exposure from dental amalgam for receptors other than the toddler and infant.

Concentrations of mercury in soil found on numerous properties sampled in West Flin Flon may result in the occurrence of adverse health effects in individuals that are highly exposed to impacted soils. Biomonitoring would be an appropriate option to more accurately assess mercury exposure to individuals in West Flin Flon. Human blood, hair, breast milk, and urine are generally used in measuring mercury exposure. In the blood, mercury has a short half-life (three days) thereby making it a useful medium for determining short-term exposures. However, long-term consumption of fish containing elevated methyl mercury levels can also be determined through blood analysis. Mercury analysis of hair is useful in that it reveals dietary exposure to methyl mercury, however, this has not been regarded as a suitable indicator for inorganic mercury exposure. For long-term, low level exposures to inorganic mercury, measurement through urine samples is the preferred medium.

Methyl Mercury: Exposure to methyl mercury was assumed to occur *via* the consumption of fish from market basket foods, consumption of local fish, consumption of drinking water, and inhalation of ambient air. HQ estimates for residents living in each of the COI ranged from 0.0026 to 1.9, with risk levels consistent throughout each of the COI. HQ estimates for the infant were significantly lower than those for other age groups because it was assumed that children under the age of 6 months would not consume fish. Although the primary route of exposure for all receptors other than the infant was the consumption of local fish, and each of these receptors consumed the same amount of fish on a per body weight basis, HQs were much higher for the toddler and child as a result of the use of lower RfDs for these age groups (*i.e.*, 0.2 µg/kg/day for the toddler and child relative to 0.47 µg/kg/day for the teen and adult) protective of neurodevelopmental effects.

Based on this assessment, it is recommended that fish consumption advisories be considered for this area, particularly for sensitive receptors.

Selenium: The estimated HQ values associated with selenium exposures were less than 1.0 under all exposure and receptor scenarios. It was not considered necessary to undertake detailed assessment or actions related to selenium in the study area.

Overall, the health risks to Flin Flon-area residents associated with exposure to selenium are expected to be similar to those observed in other parts of Canada and are within risk levels deemed to be acceptable by Health Canada and the CCME.

8.3 Provisional Trigger Concentrations

As discussed previously, risk assessments typically employ the 95% upper confidence limit of the mean (UCLM) to characterize the exposure point concentration (EPC) of a given exposure unit. The sample mean is based on a collection of samples from the exposure unit and therefore, uncertainty exists as to whether the sample mean is a true reflection of the population mean. As a result, the 95% UCLM can be thought of as an estimate of the true population mean for a given exposure unit. In this case, the exposure units were defined as the communities under assessment in the HHRA. The underlying assumption used when developing the chronic residential exposure scenarios was that individuals would move randomly within each community and, therefore, over time, come into contact with the average soil concentration within a given community (or exposure unit). In reality, individuals do not move in a random fashion within their community, but rather exhibit predictable spatial patterns in their movements. For example, many individuals will tend to spend the majority of their time between home and work or school. Therefore, the evaluation of risks on the basis of average EPCs (assuming random movement) in an area-wide risk assessment may underestimate risks for some receptors. As a result, in addition to predicting risks using the community-based EPCs, soil provisional trigger concentrations (PTCs) were derived for each COC to be protective of residential receptors. These PTCs can then be used to determine on a property-by-property basis, which properties contain concentrations that have the potential to cause unacceptable risks.

A PTC can be defined as the average COC soil concentration within an exposure unit (EU) that corresponds to an acceptable level of risk. In other words, the PTC is the EPC in soil within a given EU (*i.e.,* a residential property) which would yield an acceptable level of risk. Exceedances of the PTC do not necessarily indicate that conditions exist in which unacceptable health risks will occur, but rather that there is less certainty regarding the related risk level.

The PTCs are derived to determine if further consideration is required, and if warranted, to help focus the efforts of a biomonitoring program on those areas or properties that may be of the greatest concern. Should a biomonitoring program be completed, the results will be used to further evaluate risk levels.

Exposure to COC through the consumption of market basket foods is anticipated to be similar throughout each of the four COI. Exposure to COC through the inhalation of particulates in air and from the consumption of drinking water may vary from community to community. Since concentrations in air tend to be highest in West Flin Flon, environmental parameters associated with this community were used to back-calculate soil PTCs. The derivation of a single soil PTC using these parameters should be protective of residential properties in each of the four COI.

8.3.1 Arsenic

An arsenic PTC of 74 µg/g was derived for the Flin Flon-Creighton area. A comparison of the arsenic PTC of 74 µg/g with the results of the residential soil sampling program indicates that 40 of the 183 properties sampled contained concentrations of arsenic in excess of the PTC (Table 8-11). All of these properties were located in West Flin Flon and Creighton.

Table 8-11 Number of Properties with Concentrations of Arsenic in Outdoor Soil in Excess of the PTC of 74 µg/g

	West Flin Flon	*East Flin Flon*	*Creighton*	*Channing*	*Total*
# of Properties Sampled	77	66	30	10	183
# of Properties >74 µg/g	30 (39%)	0	10 (33%)	0	40 (22%)

8.3.2 Cadmium

A comparison of the cadmium soil PTC of 58 µg/g with the results of the residential soil sampling program indicates that 2 of the 183 properties sampled contained a maximum concentration of cadmium (58.1 and 70.8 µg/g) in excess of the residential soil PTC (Table 8-12). Both of these locations were located in the community of West Flin Flon. Although these concentrations are above the soil PTC, the HQs associated with these concentrations are still equal to an acceptable value of 1.0. As a result, levels of cadmium in Flin Flon-area residential soils are not anticipated to pose health risks to human health within any of the COI.

Table 8-12 Number of Properties with Concentrations of Cadmium in Outdoor Soil in Excess of the Residential Soil PTC of 58 µg/g

	West Flin Flon	*East Flin Flon*	*Creighton*	*Channing*	*Total*
# of Properties Sampled	77	66	30	10	183
# of Properties >58 µg/g	2 (2.6%)	0	0	0	2 (1.1%)

Given that concentrations of cadmium in soil in the Flin Flon-Creighton area are not anticipated to result in unacceptable risks, soil remediation objectives were not derived for cadmium. Current concentrations can safely remain in place without the need for measures to prevent or reduce exposure.

8.3.3 Copper

A copper soil PTC of 5,000 µg/g was derived for the Flin Flon-Creighton area. A comparison of the copper soil PTC of 5,000 µg/g with the results of the residential soil sampling program indicates that only 5 of the 183 properties sampled were in excess of the soil PTC (Table 8-13). All of these locations were in the community of West Flin Flon. Although these properties each contained a maximum concentration (5,260, 5,360, 5,470, 5,530 and 7,810 µg/g) in excess of the PTC, the HQs associated with these concentrations range from 1.0 to 1.1 assuming chronic exposure to these levels. Given this, and the minor contribution of soil and dust to the total daily copper exposure, levels of copper in Flin Flon-area soils are not anticipated to pose health risks to human health within any of the COI.

Table 8-13 Number of Properties with Concentrations of Copper in Outdoor Soil in Excess of the Residential Soil PTC of 5,000 µg/g

	West Flin Flon	***East Flin Flon***	***Creighton***	***Channing***	***Total***
# of Properties Sampled	77	66	30	10	183
# of Properties >5,000 µg/g	5 (6.5%)	0	0	0	5 (2.7%)

Given that concentrations of copper in soil in the Flin Flon-Creighton area are not anticipated to result in unacceptable risks, soil remediation objectives were not derived for copper. Current concentrations can safely remain in place without the need for measures to prevent or reduce exposure.

8.3.4 Lead

A simple evaluation of ambient soil and dust concentrations of lead in the COI may not be sufficient to provide an adequate and accurate basis on which to develop a reasonable PTC. As part of an overall weight-of-evidence approach, the following lines of evidence were reviewed and evaluated to aid in the development of an appropriate lead PTC:

- Exposure, risk, and PTC predictions from the HHRA Exposure Model;
- Predicted probabilities of exceeding BLL of concern using the IEUBK model; and,
- The empirical relationship between lead in soil and BLLs and how this information has formed the basis for PTC values derived at other sites.

An additional critical line of evidence would be site-specific blood lead data, if it were available for children within the study area.

Table 8-14 provides the potential PTCs mathematically derived for lead using the HHRA and IEUBK models. Also included in the table are site-specific soil lead criteria previously established by the U.S. EPA (2001).

Table 8-14 Provisional Trigger Concentrations (PTCs) and Soil Risk Management Levels (SRML) for Lead (µg/g)			
IEUBK Model Derived PTC	***HHRA Model Derived PTC***	***U.S. EPA SRML***	
		Play area	***Bare soil Remainder***
370 (protective of a 5% probability of exceeding a BLL of 10 µg/dlL)	**700**	400	1,200

The number of properties sampled as part of the residential soil sampling study that contained concentrations of lead in excess of a residential PTC of 370 µg/g within each community are presented in Table 8-15.

Table 8-15 Number of Properties with Concentrations of Lead in Outdoor Soil that Exceed a Residential PTC of 370 µg/g					
	West Flin Flon	***East Flin Flon***	***Creighton***	***Channing***	***Total***
# of Properties Sampled	77	66	30	10	183
# of Properties >370 µg/g	32 (42%)	2 (3%)	4 (13%)	0	38 (21%)

8.3.5 Mercury

An inorganic mercury soil PTC of 64 µg/g was derived for the Flin Flon-Creighton area. A comparison of the inorganic mercury PTC of 64 µg/g with the results of the residential soil sampling program indicates that all properties sampled in East Flin Flon, Creighton, and Channing were below the PTC, and that 40 of the 77 properties sampled in West Flin Flon were in excess of the PTC (Table 8-16).

Table 8-16 Number of Properties with Concentrations of Mercury in Outdoor Soil in Excess of the Residential PTC of 64 µg/g					
	West Flin Flon	***East Flin Flon***	***Creighton***	***Channing***	***Total***
# of Properties Sampled	77	66	30	10	183
# of Properties >64 µg/g	40 (52%)	0	0	0	40 (22%)

Since methyl mercury is not a significant component of residential soils, a soil PTC was not derived. Overall, unacceptable risks may occur to toddlers and children as a result of exposure to methyl mercury under the assumed exposure scenario. This assumes that these receptors will consume 1.5 local fish meals per week, 52 weeks per year at the 95% UCLM concentration (0.43 µg/g methyl mercury or 0.45 µg/g total mercury). Based on the assumed exposure scenario, and assuming that exposure from drinking water, air, and market basket fish remained constant, the back-calculated local fish concentration that would result in an HQ of 1.0 for the toddler is 0.19 µg/g ww total mercury. As a result of the conservative assumptions regarding local fish consumption rates, this value is significantly lower than the fish consumption guidelines recommended by regulatory agencies. Health Canada (2007) has established a guideline of 0.5 µg/g ww total mercury for commercially sold fish. Manitoba Water Stewardship indicates that sportfish with mercury concentrations ranging from 0.5 to 1.0 µg/g can be consumed up to 4 times per month by members of the general population, but women of child bearing age and children under 12 years old should only consume fish with concentrations below 0.5 µg/g (Manitoba, 2007). Similarly, Saskatchewan Environment and Resource Management (SERM, 1999) indicates that fish with an average mercury concentration of less than 0.5 µg/g may be eaten in unlimited amounts and that children and pregnant women should not consume fish containing mercury concentrations in excess of 0.5 µg/g (SERM, 1999). The

95% UCLM concentration for each fish species, independent of lake where caught, was below the 0.5 µg/g total mercury guideline, with the highest values for walleye (0.45 µg/g) and lake trout (0.49 µg/g) (refer to Chapter 4, Table 4-8). In addition, of the 11 lakes included in the Fish Study, the 95% UCLM concentration of all fish caught for three lakes (0.56 µg/g in Big Island, 0.60 µg/g in Denare Beach, and 0.54 µg/g in Hamell lake) was in excess of 0.5 µg/g guideline (refer to Table 4-9). It was also noted that there does not seem to be link between methyl mercury fish concentrations and proximity to the facility.

Of the 212 local fish collected and analyzed for total mercury, 17 (or 8%) contained concentrations above the 0.5 µg/g ww guideline, and 80 (or 38%) contained concentrations above the HHRA-derived concentration associated with an HQ of 1.0 (0.19 µg/g ww). Based on this assessment, it is recommended that fish consumption advisories be considered for this area, particularly for sensitive receptors.

8.3.6 Selenium

A selenium soil PTC of 170 µg/g was derived for the Flin Flon-Creighton area. A comparison of the selenium soil PTC of 170 µg/g with the results of the residential soil sampling program indicates that one sample (286 µg/g in West Flin Flon) was in excess of the PTC (Table 8-17). Although this property contained a concentration of selenium in excess of the PTC, the HQ for the toddler associated with this concentration is 1.1 assuming chronic exposure to this level. Given this, and the minor contribution of soil and dust to the total daily selenium exposure, levels of selenium in Flin Flon-area soils are not anticipated to pose health risks to human health within any of the COI.

Table 8-17 Number of Properties with Concentrations of Selenium in Outdoor Soil in Excess of the Residential PTC of 170 µg/g

	West Flin Flon	***East Flin Flon***	***Creighton***	***Channing***	***Total***
# of Properties Sampled	77	66	30	10	183
# of Properties >170 µg/g	1 (1.3%)	0	0	0	1 (0.55%)

Given that concentrations of selenium in soil in the Flin Flon-Creighton area are not anticipated to result in unacceptable risks, soil remediation objectives were not derived for selenium. Current concentrations can safely remain in place without the need for measures to prevent or reduce exposure.

8.3.7 Summary of Properties Exceeding PTCs

To provide additional information of the extent of contamination throughout the study area, Table 8-18 provides the number of properties with concentrations that exceed the residential PTC for at least one COC. West Flin Flon contained the greatest percentage of properties with exceedances of at least one PTC at 61%, followed by Creighton at 37%, and East Flin Flon at 3%.

Table 8-18	Number of Properties with Concentrations that Exceed the Residential PTC for at Least One COC				
	West Flin Flon	*East Flin Flon*	*Creighton*	*Channing*	*Total*
# of Properties Sampled	77	66	30	10	183
# of Properties >PTC	47 (61%)	2 (3%)	11 (37%)	0	60 (33%)

8.4 Overall Recommendations

The results of the HHRA indicate that concentrations of cadmium and arsenic measured in ambient outdoor air in the Flin Flon area may result in a very low to low increase in the risk of developing lung cancer, however because of the population size in the Flin Flon area, a detectable increase is not expected. Future reductions in smelter-related emissions containing arsenic and cadmium would have a direct and immediate effect on reducing inhalation-related exposure and risks. Given that the HBMS smelter ceased operation in mid-June 2010, concentrations of arsenic and cadmium in ambient air are anticipated to be reduced to levels below those that are associated with an increased risk of developing lung cancer. It is recommended that Provincial and HBMS air monitoring programs are continued in the future to ensure that concentrations meet regulated air standards.

Concentrations of arsenic, lead, and inorganic mercury in soils on a number of residential properties in the Flin Flon area are in excess of PTCs derived to be protective of health. It is recommended that to reduce uncertainty in the assessment of exposure and risk, a comprehensive biomonitoring study is completed in which blood and urine samples are collected from children living in the Flin Flon area. Analyses of arsenic and inorganic mercury in urine samples, and lead in blood, will provide measurements of actual exposure of children to these chemicals and allow for a more accurate assessment of potential risks.

Elevated exposure to methyl mercury was predicted for people frequently consuming fish from several of the local lakes within the Flin Flon area. It is unclear if elevated methyl mercury concentrations in fish are associated with the HMBS facility. Methyl mercury concentrations in fish generally appeared to increase with increasing distance from the HMBS facility, suggesting that methyl mercury levels in fish may not be directly linked to HMBS air emissions. Provincial fish advisories are in place to address these concerns for some lakes and it is recommended that the Provinces consider the need for advisories on additional lakes in the area.

The HHRA and biomonitoring study will inform the risk management decision making process by providing information on the actual levels of exposure and the pathways of potential concern. This allows for risk managers to focus on those environmental media that may be contributing most significantly to total exposure and those that may be most effectively mitigated.

Further details regarding biomonitoring are provided in Section 8.5.

8.5 HHRA and Biomonitoring

8.5.1 Introduction

As discussed previously, human health risk assessments (HHRAs) typically utilize a number of conservative (*i.e.*, protective) assumptions, which intentionally overestimate possible exposures and risks in relation to the COC. Because of this, it is often important to "ground truth" or validate the findings of an HHRA by collecting information on actual concentrations of the COC in human tissues or biological fluids. This evaluation of environmental contaminant exposures is commonly referred to as biomonitoring.

In a broad sense, biomonitoring refers to the various means that can be used to quantify or otherwise determine exposures and effects in human subjects through the use of biomarkers (Boogaard and Money, 2008). Biomonitoring includes the use of analytical methods to measure low levels of chemical substances or their breakdown products in biological media such as human body fluids or tissues (Albertini *et al*., 2006). It can also be used to measure changes in biochemical or physiological parameters which may be indicative of an effect. The parameters measured in biomonitoring are referred to as biomarkers.

The NAS/NRC (1989) identifies three main types of biomarkers: markers of exposure, markers of effect, and markers of susceptibility. A biomarker of exposure is defined as a xenobiotic substance or its metabolite(s), or the product of an interaction between a xenobiotic agent and some target molecule(s) or cell(s) that is measured within a compartment of an organism (NAS/NRC, 1989). Preferred biomarkers of exposure are typically analytical measurements of the substance of interest itself or substance-specific metabolites in readily obtainable body fluid(s) or excreta (*e.g*., blood, urine, hair).

Biomarkers of effect are any measurable biochemical, physiologic, or other alteration within an organism that, depending on magnitude, can be recognized as an established or potential health impairment or disease (NAS/NRC, 1989). A biomarker of susceptibility is an indicator of an inherent or acquired limitation of an organism's ability to respond to the challenge of exposure to a specific xenobiotic substance (NAS/NRC, 1989). It can be an intrinsic genetic or other characteristic, or a pre-existing disease condition that may result in changes related to absorbed dose, biologically effective dose, or target tissue responses.

More recently, biomarkers of exposure (#1 and #2) and effect (#3 and #4) have been further divided into the following (Boogaard and Money, 2008; ECETOC, 2005):

1. Biological monitoring or biomarkers of exposure (also: internal dose or body burden) (*e.g*., lead in blood, arsenic in urine);
2. Biochemical effect monitoring or biomarkers of effective dose (also: tissue dose) (*e.g*., determining adducts of a specific chemical to DNA or protein);
3. Biological effect monitoring or biomarkers of effect. Measured effects are usually reversible and are not indicative of an adverse health effect (*e.g*., cholinesterase activity in blood; albuminuria as biomarker of early renal function effects); and,
4. Clinical parameters or biomarkers of disease (*e.g*., albuminuria to assess renal function impairment in diabetics).

Most often, it is biomarkers of exposure that are considered in biomonitoring studies as they are linked to chemical exposures (they provide chemical-specific internal exposure data). Biomarkers of effect are often not substance-specific, and/or may not necessarily relate to potential adverse health effects (NAS/NRC, 1989; Boogaard and Money, 2008). Biomarkers of susceptibility are rarely considered in biomonitoring studies as they can be difficult to quantify in human subjects, and the same underlying limitations or characteristics may have different manifestations across individuals (NAS/NRC, 1989).

Biomonitoring can be advantageous over environmental monitoring (such as soil, air, water), as it provides information on the actual exposures received by human subjects and it accounts for exposures from all potential sources, pathways and routes (HERAG, 2007). This is an advantage when total exposures to a given substance are of interest, but can be a disadvantage if the intent is to attribute measured human exposure to a particular source, pathway or route.

Although biomonitoring studies have been conducted for many years, there is an increasing trend of such studies being applied under various situations where human exposure to chemicals is of concern. This is due in part to the increased availability and sensitivity of analytical methods, as well as public health and risk assessment practitioners' need, or desire, for more realistic exposure data (Doerrer and Holsapple, 2004; Boogaard and Money, 2008).

Presently, biomonitoring studies are frequently being conducted in conjunction with HHRAs of larger sites and communities. Some potential applications of biomonitoring include (from U.S. EPA, 2008):

1. Obtain more specific data related to exposures associated with site-related media (such as soils; drinking water; air; garden produce, *etc*.);
2. Identify the fate of the substance in the body;
3. Determine exposure trends;
4. Establish links between environmental exposures and adverse health effects;
5. Development of reference ranges; and,
6. Assist in decision making.

While biomonitoring studies can be used to identify whether or not exposure has occurred (or is occurring), and can establish temporal trends related to chemical exposure, such studies cannot (on their own) indicate whether measured exposure levels in human subjects suggest a potential human health risk (U.S. EPA, 2008). In addition, biomonitoring studies may not be able to determine when, how often, how much, and by what route exposure has occurred (U.S. EPA, 2008). By combining biomonitoring data with environmental data and HHRA findings, several lines of evidence can be available to assist in determining exposures and potential risks to COC (Figure 8-1).

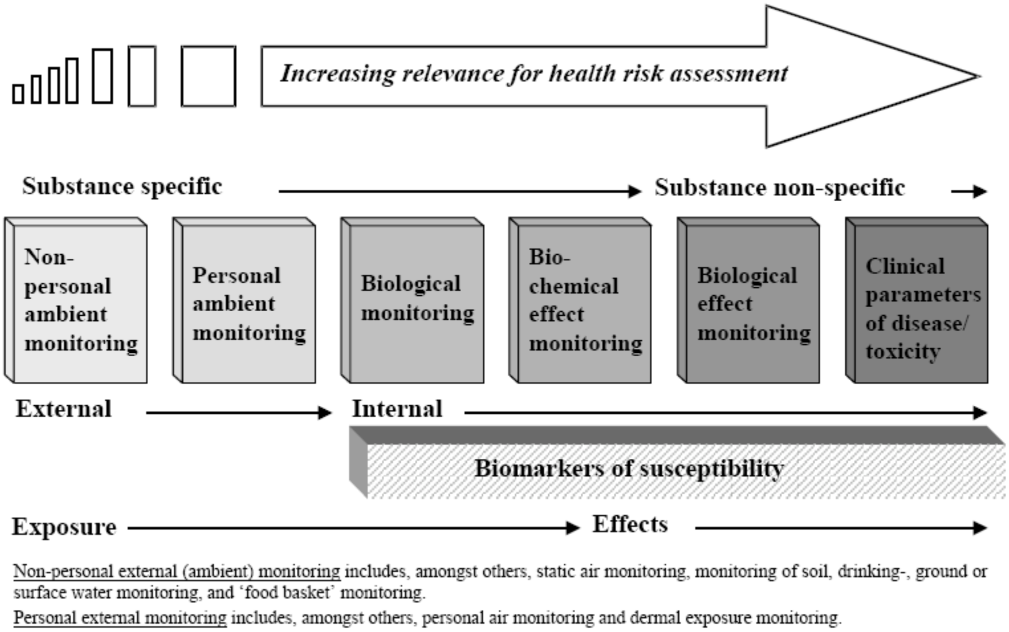

Figure 8-1 Monitoring Techniques as Part of the Exposure Dose Continuum (Taken from Figure 1 from Boogaard and Money, 2008).

8.5.2 Biomarkers of Exposure for Arsenic, Lead and Mercury

Based on the HHRA outcomes for the site, the COC being recommended for further study through biomonitoring are arsenic, lead and inorganic mercury. There are various types of biomonitoring that can be done for these substances. The ATSDR (2007a,b; 1999) toxicological profiles for these substances describe the biomarkers of exposure that are typically used for arsenic, lead and mercury. A brief summary of the information provided by ATSDR in these profiles is provided below. Further information will be developed as part of the biomonitoring study.

8.5.2.1 Non-steroid Anti-inflammatory Drugs

Arsenic in blood, urine, hair and nails can be used to indicate whether exposure has occurred. Blood arsenic levels measure exposures which have occurred within the very recent past, since arsenic is rapidly cleared from the blood. As such, blood arsenic concentrations are not considered reliable measurements of chronic low level exposure to arsenic.

Urinary arsenic is considered to be the most reliable indicator of recent arsenic exposure as arsenic absorbed in the lungs and gastrointestinal tract is excreted in the urine within a couple of days. It is important to remember when using urinary arsenic as a biomarker of exposure, that some organic species of arsenic that tend to occur in seafood, which are considered essentially non-toxic, are excreted in the urine in an unmetabolized form. If total urinary arsenic is measured, these essentially non-toxic forms of arsenic will be measured and can over estimate exposures to forms of arsenic that are of potential health concern. Thus, it is important in any urinary arsenic study to solicit information from the subjects on recent dietary intakes, lifestyle factors, recent activities *etc*., and to speciate urinary arsenic such that organoarsenicals originating from seafood are accounted for separately from the species that are of greatest toxicological concern.

Arsenic in hair and nails can be a useful indicator of past exposures to arsenic as arsenic tends to accumulate in these tissues. However, minimum exposure levels that produce measurable increases in arsenic levels in hair and nails have not been precisely defined.

8.5.2.2 Lead

Lead biomarkers of exposure that are in practical use today include measurements of total lead in blood, bone, urine and hair. Blood lead concentration is considered to be the most reliable biomarker for lead and is the most widely used with blood lead levels >10 µg/dL considered to be of concern. With a blood half-life of approximately 30 days, blood lead levels reflect relatively recent exposure but they cannot distinguish between low-level intermediate or chronic exposure and high-level acute exposure. As blood lead benchmarks for concern change to levels lower than 10 µg/dL, measured concentrations can be compared to levels of concern.

Lead in hair is another biomarker for lead, however, due to possible contamination of hair surface with environmental lead and contaminants in artificial hair treatments, this method is subject to error. Hair lead concentrations are also a relatively poor predictor of blood lead, particularly at low concentrations.

Lead in bone is considered a biomarker of cumulative exposure to lead because lead accumulates in bone over the lifetime and most of the lead body burden resides in bone tissue. Lead is not distributed uniformly in bone though, but accumulates in those regions of bone undergoing the most active calcification at the time of exposure. Furthermore, lead in bone analysis requires specialized x-ray equipment that may not be available in a particular community.

8.5.2.3 Mercury

Blood and urine mercury concentrations are commonly used as biomarkers of exposure to mercury and are reflective of recent exposures. Correlation of exposure at low levels to blood or urinary mercury levels is low. Hair has been used as a biomarker of exposure to methylmercury.

Blood levels of most forms of mercury peak sharply during and soon after short-term exposures, and as such monitoring of blood mercury levels is most useful soon after exposure. In workers chronically exposed to mercury, blood and urinary mercury concentrations may still be elevated for a long period of time after exposures cease as these individuals would likely have a high body burden of mercury.

Intra- and interindividual differences in total mercury levels in blood and urine (taken immediately or within a few days of exposure) have been reported to be substantial, possibly due to dental amalgams and ingestion of contaminated fish. Urine mercury measurement is reliable and simple but is a more appropriate marker for inorganic mercury, rather than organic forms of mercury because organic mercury represents only a small fraction of urinary mercury.

8.5.3 How Biomonitoring has been Used in Conjunction with Risk Assessment at Other Sites

The use of biomonitoring data in association with HHRA has increased in the last five to ten years, particularly on larger or wide-area sites involving larger populations. Coupling these two tools in large sites has been done previously with the following objectives:

- To assist in a weight of evidence evaluation;
- To obtain individual and/or community exposure data on COC of interest ;
- To refine or validate exposure estimates within the HHRA; and
- As a tool to measure the success of implementation of recommended risk management plans.

There are numerous studies which illustrate the uses of biomonitoring in large risk assessment projects in both Canada and the U.S. A selection of sites are briefly discussed in the following sections to provide context on the past and recent use of these tools on sites with similar issues to those present in the Flin Flon-Creighton area.

8.5.3.1 Teck Lead Smelter in Trail, BC

A specific example of the application of biomonitoring and HHRA is the assessment undertaken in Trail B.C. area, associated with emissions from 100 years of operations of a lead smelter, currently owned by Teck Ltd (Hilts *et al*, 1998; 2001). In 1975, a blood lead survey of children living within 1 mile of the smelter determined that only 4% of children tested had blood lead levels exceeding the standard of the day (40 µg/dL). No further action was recommended, in that no cases of lead poisoning had been reported by area physicians. With increased reports of lead exposure being associated with IQ deficits in the 1980s, a new blood lead study was undertaken in 1989 by smelter-owner Cominco and the BC Ministry of Health. The U.S. EPA level of no concern for lead in blood was 15 µg/dL at the time the study was conducted; 39% of children tested were above this level. As a result of this study, a Trail Community Lead Task Force was established to develop a strategy to reduce child lead exposures in the area. The Task force estimated the cost of soil replacement in Trail would exceed $55,000,000, which was not feasible financially, nor was it considered socially acceptable at the time. Therefore, the Task Force chose not to undertake any immediate soil replacement, but rather, focused on an extensive program of community education and case management, as well as environmental sampling and risk assessment to better understand the lead exposure pathways of importance in this area. By understanding the importance of specific exposure pathways (*i.e*., soil or dust ingestion, *versus* ingestion from drinking water), effective remedial action plans for reducing child exposures to lead could be clearly identified.

A detailed exposure assessment was conducted to identify predominant sources of lead in the community and from the smelter, and predominant exposure pathways (*e.g*., ingestion of soils; dusts; inhalation of air, *etc*.). In addition, a baseline risk assessment for lead was conducted, using the IEUBK model as a means to predict the probability of exceeding blood lead levels of 10 µg/dL in children, based on the environmental assessment data. The outputs of the IEUBK model were modified to also allow calculations of Hazard Indices, using the Health Canada TRV for lead. The geometric mean soil concentration for lead in the community was 720 µg/g. The biomonitoring data collected from the community was used as a means to validate the predicted blood lead results from the IEUBK (Hilts *et al*., 2001). In addition to the baseline risk assessment and initial blood monitoring, there has been an on-going blood lead testing program for young children to identify at risk families, and to track success of risk management efforts. The focus of the program has been on children in high risk age groups (6 to 60 months of age), living in specific areas where exposure levels are potentially higher, due to proximity to the smelter, and measured soil concentrations and risk estimates. Children living in low risk areas, or older children, are not subjected to unnecessary testing, unless requested by parents. Community education and intervention, case management, dust abatement and re-greening of bare soil areas all appear to have played a role in reduction of blood leads through the 1990s (Hilts *et al*., 1998).

In summary, the biomonitoring for blood lead was used as a direct indicator of exposure to lead in small children. These data were also used to validate the IEUBK model runs, and based on risk assessment of environmental data, areas within the community were prioritized for on-going bio-monitoring. Remedial goals and risk management decisions have been established based on the integration of risk assessment, using the IEUBK model in conjunction with modifications to predict Hazard Quotients, and blood lead data. Using an integrative approach recommended in the Baseline Risk Assessment report, different risk management approaches can be assessed. In 2001, the Trail community Lead Task Force recommended a long-term goal of having 90% of children aged 6 to 60 months with blood leads less than 10 µg/dL. It is unclear whether this goal has been revised more recently in light of increasing evidence of the potential for effects in children at lower blood lead levels. The remedial options recommended in 2001 are lengthy, and are presented in Hilts *et al*. (2001). The focus of the remedial planning was on reducing potential for dust and soil ingestion through community education, re-greening, minimizing potential for fugitive dust release from the site, *etc*. The blood lead monitoring program continues and is used as a tool to measure the success of the implementation of various remedial plans (Hilts *et al*., 2001; Hilts, 2003).

8.5.3.2 Sudbury Area Soils Study

The Falconbridge Arsenic Exposure Study (SARA, 2008) compared the urinary arsenic levels of individuals from an "impacted" community (Falconbridge) to those from a "control" community (Hanmer). While not directly part of the Sudbury Soils Study, the results of this study provided a unique dataset for use in the HHRA. Community residents were consulted to identify primary concerns and to provide feedback on the objectives of the study. The research team then developed the study methodology to address two specific questions that were deemed to be most important by residents:

1. Do Falconbridge residents have higher urinary arsenic levels than residents living in a comparison area with lower levels of arsenic in their soil; and,
2. What health risks relative to other communities are associated with the urinary arsenic levels of Falconbridge residents?

To address these questions, the research team developed a methodology that combined both the analysis of first morning void urine samples, and interviews that captured lifestyle information pertaining to potential arsenic exposure. The study was comparative in nature, meaning that the main questions above were addressed by comparing Falconbridge with a similar community with lower soil arsenic concentrations.

The results indicated that urinary arsenic measurements from the "impacted" community were similar to that of the control, despite the significantly higher soil concentrations present in the Falconbridge (95% UCLM soil concentration 78.7 *vs*. 4.27 µg/g for Falconbridge and Hanmer, respectively). Results of the survey also indicated that arsenic intakes for Falconbridge and Hanmer residents on average were within the typical daily intake of arsenic by Canadians; and therefore, residents are not at any increased risk from arsenic exposure compared to other Canadians in general.

Due to the unique nature of the arsenic sources within each community, the findings in the Falconbridge study could not be directly applied to other communities within the Greater Sudbury Area (GSA). However, arsenic speciation results provide some context for this comparison, and examination of the results of a speciation study (SARA, 2008) clearly showed that the forms of arsenic in the soil, dust and air were consistent between the various communities within the GSA. The comparison included the community of Falconbridge, thereby providing some confidence in the use of the Falconbridge Arsenic Exposure Study across the entire GSA. As such, the results of the Falconbridge Arsenic Exposure Study, which demonstrated no statistical difference in levels of arsenic in urine between Falconbridge and comparison (unexposed) community residents, indicated that arsenic exposure for all resident of the GSA were similar to those in other communities with significantly lower arsenic soil concentrations. Results of the survey were incorporated in the weight-of-evidence approach used to characterize overall health risks related to exposures of GSA residents to environmental concentrations of arsenic.

8.5.3.3 Eureka Mills (Eureka, Utah)

The following information on the Eureka Mills site has been obtained from the Baseline HHRA conducted by Syracuse Research Corporation (SRC, 2002) for the U.S. EPA.

The town of Eureka was a productive base- and precious-metal mining district in Utah. Between 1871 and 1902 a smelter and a number of mines were built in the area with Bullion Beck, Eureka Hill, Chief Consolidated, May Day, Godiva, and Uncle Sam being the most important. By the mid-1960's, mining activities were reduced with only sporadic mining occurring in the area since then. A risk assessment approach was used to characterize current and potential future risks which mining-related wastes pose to residential and recreational visitors who may be exposed in the vicinity of the site.

COCs at the site included arsenic, lead and mercury. Potential risks related to non-lead COC were evaluated using standard U.S. EPA risk assessment methods. For lead, potential health risks were investigated using direct measurement of blood lead in the community and by using IEUBK modeling approach. Speciation and bioaccessibility testing were conducted on soils sampled for lead and arsenic and were used in their assessment.

Risk assessment results for non-lead COC suggested that under the CTE exposure scenario to soil, that excess cancer or non-cancer risks to current or future residents and recreational visitors may occur. Overall these risks were primarily attributable to elevated arsenic concentrations.

Risks from exposure to lead were evaluated using a modeling approach (IEUBK model) and direct blood lead measurements. Using the modeling approach, 100% of the residential properties in Eureka and the outlying non-residential areas were estimated to exceed the U.S. EPA guideline of having a 5% probability of a blood lead value over 10 µg/dL. The predictions were supported by measured blood lead data. During 2000, over 250 blood lead samples were collected from 227 Eureka residents. Of these, 35 (approximately 15%) were found to have blood lead levels > 10 µg/dL and 34% of the blood lead samples collected from children up to 6 years old were found to exceed 10 µg/dL.

8.5.3.4 Anaconda Copper Mine, Montana, U.S.

This text is summarized from ATSDR (2007c). The Anaconda Copper Mine was discovered in 1881, and operated as a mine, which subsequently included smelting operations, through to the early 1980s, when it closed. This site was owned by the Atlantic Richfield Company (ARCO) and was listed on the National Priority List (NPL) in 1983. Historical mining in the area resulted in metals contamination in the Butte, Anaconda, and the Clark Fork River downstream to the Milltown dam in Missoula, which precipitated the inclusion of these areas in the NPL listing for environmental cleanup in the 1980s. A series of biomonitoring studies (focusing on young children) were conducted nationally across the U.S. on copper, zinc and lead smelter sites in 1975, and children in the Anaconda area were found to have elevated levels of arsenic in urine (8.6 µg/L geometric mean speciated arsenic) and hair relative to children in communities without excess arsenic exposure. A second biomonitoring study was conducted in the late 1970s, which was coupled with environmental site assessment media (soil; dust; drinking water, *etc*.). Attempts were made to correlate environmental media data with biomonitoring findings to examine the relative importance of various environmental media to total exposure levels measured. Additional biomonitoring studies have been conducted over the years, with several of these studies having looked for correlations between biomonitoring data and environmental media (urinary arsenic and surface soil concentrations of arsenic, *etc*.).

A baseline risk assessment study was completed in 1996 as part of the U.S. EPA Superfund process for lead and arsenic. Risks from lead were estimated to be within EPA's acceptable range, whereas risks from arsenic were considered unacceptable, and therefore arsenic was considered the risk driver at the site.

Some remedial actions were completed prior to the completion of the Baseline HHRA. Initial investigations indicated that the neighbourhood of Mill Creek (east of the Anaconda smelter) was severely impacted by contamination, and children in this neighbourhood had elevated levels of urinary arsenic. Temporary relocation of the children reduced their urinary arsenic levels to background, which confirmed sources related to their environment. Ultimately, a decision was made to permanently relocate these families. Additional investigations through the late 1980s, indicated that residential soils and dust in some other neighbourhoods were also impacted, and arsenic soil concentrations exceeding 250 µg/g in these neighbourhoods were removed and replaced with clean fill (completed in 1992). The level of 250 µg/g arsenic was selected as an action level, using screening level exposure calculations (*e.g.*, soil ingestion rate; soil arsenic concentration, bioavailability of arsenic in site soils, *etc*.), and an excess cancer risk of less than $1x10^{-4}$.

In summary, biomonitoring of children in the Anaconda area was useful in indicating elevated past exposures while the smelter operated and, later, in areas identified as having high soil concentrations of arsenic, it was used as a tool (in conjunction with the HHRA) to assist in identifying specific areas recommended for remedial action. It was also used to measure success of remedial actions.

8.5.4 Summary

A biomonitoring study can assist in the interpretation of the results of an HHRA by confirming or refuting the results that the HHRA forecast with respect to exposure commitments from all of the considered pathways of exposure in a particular community. As demonstrated in the studies above, biomonitoring has proven to be a valuable tool when used in conjunction with traditional risk assessment methods. Biomonitoring has been used in the validation of estimates of exposure; blood lead levels estimated using a modeling approach (IEUBK model) were validated against measured blood lead levels in children in both the Trail B.C. and Eureka Mills risk assessments. In the Trail B.C. HHRA, the biomonitoring of blood lead was additionally used as a direct indicator of exposure to lead in small children, and in this way, helped to refine the exposure assessment component of the HHRA. Biomonitoring has been used to assist in weight-of-evidence evaluations of potential risks as was done as part of the Sudbury Area Soils Study. Results of the Falconbridge Arsenic Exposure Study were incorporated in the weight-of-evidence approach used to characterize overall health risks related to exposures of Greater Sudbury Area residents to environmental concentrations of arsenic. The use of biomonitoring data in association with HHRA has helped to inform risk management planning, and can be used to measure success of remedial actions. The data collected in the biomonitoring study of children in the Anaconda area helped to determine arsenic exposure trends and in turn, to identify specific areas recommended for remedial action. Baseline arsenic exposure documented in the biomonitoring study was subsequently used to measure the success of risk-management actions.

Biomonitoring initiatives, including the measurement of urinary arsenic, blood lead, and urinary inorganic mercury levels in in children (under 15) in Flin Flon, Manitoba and Creighton, Saskatchewan, are recommended within HHRA for several reasons. As demonstrated, biomonitoring can help to refine and validate exposure estimates for COC which are associated with elevated levels of risk. More specifically, results from the detailed HHRA have indicated that the toddler receptor in West Flin Flon may have an elevated exposure to inorganic mercury and correspondingly, an elevated risk for potential health impacts. Biomonitoring is recommended to provide a more accurate assessment of mercury exposure to individuals in West Flin Flon. In the case of lead, a significant percentage of homes in West Flin Flon and Creighton were found to contain soil concentration in excess of those predicted to be protective of an acceptable blood lead level. Completion of a blood lead survey is recommended to help reduce uncertainty in the exposure assessment and provide a more accurate measure of the levels present in young children in these communities. Non-cancer and cancer health risks predicted from chronic exposure to inorganic arsenic were found in exceedance of acceptable risk levels. Despite elevated risk predictions, a weight-of-evidence evaluation conducted in the HHRA indicated that there are likely no unsafe exposures or increased health effects associated with arsenic levels within the Flin Flon area. Biomonitoring of urinary arsenic is recommended to validate the HHRA results and to contribute to the weight-of-evidence evaluation of potential risks.

8.6 Document Sign-Off

Intrinsik Environmental Sciences Inc. (Intrinsik) has provided this report to Hudson Bay Mining and Smelting Co., Limited (HBMS) and the Flin Flon Soils Study Technical Advisory Committee ("TAC"), solely for the purpose stated in the report. The information contained in this report was prepared and interpreted exclusively for HBMS and the TAC, and may not be used in any manner by any other party. Intrinsik does not accept any responsibility for the use of this report for any purpose other than as specifically intended by HBMS and the TAC. Intrinsik does not have, and does not accept, any responsibility or duty of care whether based in negligence or otherwise, in relation to the use of this report in whole or in part by any third party. Any alternate use, including that by a third party, or any reliance on or decision made based on this report, are the sole responsibility of the alternative user or third party. Intrinsik does not accept responsibility for damages, if any, suffered by any third party as a result of decisions made or actions based on this report.

The HHRA has been performed in accordance with accepted practice and usual standards of thoroughness and competence for the profession of toxicology and human health risk assessment. Any information or facts provided by others, and referred to or utilized in the preparation of this report, is believed to be accurate without any independent verification or confirmation by Intrinsik. The information, opinions and recommendations provided within the aforementioned report have been developed using reasonable and responsible practices, and the report was completed to the best of our knowledge and ability.

Intrinsik Environmental Sciences Inc.

Elliot Sigal, QP_{RA}	Adam Safruk, MES, B.Sc.
Executive Vice President	Scientist

8.7 References

Albertini, R., Bird, M., Doerrer, N., Needham, L., Robison, S. Sheldon, L. and Zenick, H. 2006. The use of biomonitoring data in exposure and human health risk assessments. Environmental Health Perspectives 114(11):1755-1762.

ATSDR. 1999. Toxicological profile for Mercury. U.S. Department of Health and Human Services. Public Health Service. Agency for Toxic Substances and Disease Registry. Atlanta, Georgia. http://www.atsdr.cdc.gov/toxprofiles/tp46.html

ATSDR. 2007a. Toxicological profile for Arsenic. U.S. Department of Health and Human Services. Public Health Service. Agency for Toxic Substances and Disease Registry. Atlanta, Georgia. http://www.atsdr.cdc.gov/toxprofiles/tp2.html

ATSDR. 2007b. Toxicological profile for Lead. U.S. Department of Health and Human Services. Public Health Service. Agency for Toxic Substances and Disease Registry. Atlanta, Georgia. http://www.atsdr.cdc.gov/toxprofiles/tp13.html

ATSDR. 2007c. Evaluation of Residential Soil Arsenic Action Level. Anaconda Co. Smelter NPL Site Anaconda, Deer Lodge County, Montana. EPA Facility ID: MTD093291656. U.S. Department of Health and Human Services. Public Health Service Agency for Toxic Substances and Disease Registry.

Boogaard, P.J. and Money, C.D. 2008. A proposed framework for the interpretation of biomonitoring data. Commentary. Environmental Health 7(Suppl I):S12

Doerrer, N.G. and Holsapple, M.P. 2004. Integration of biomonitoring exposure data into the risk assessment process. Inside OSHA Vol. 11(26). http://global.factiva.com.ezproxy.library.dal.ca.ga.default.aspx

ECETOC. 2005. Guidance for the Interpretation of biomonitoring data. The European Centre of Ecotoxicology and Toxicology of Chemicals. ECETOC Document No. 44. http://www.ecetoc.org/technical-documents.

Health Canada. 2006. Federal Contaminated Site Risk Assessment in Canada. Part 1. Guidance on Human Health Screening Level Risk Assessment (SLRA). Version 2.0. December, 2006.

Health Canada. 2007. Human Health Risk Assessment of Mercury in Fish and Health Benefits of Fish Consumption. Bureau of Chemical Safety, Food Directorate, Health Products and Food Branch. Health Canada, March 2007.

HERAG. 2007. Health Risk Assessment Guidance for Metals. HERAG. Fact Sheet 03. Indirect exposure *via* the environment / Consumer exposure. http://www.ebrc.de/ebrc/ebrc-projects.php

Hilts, S.R. 2003. Effects of smelter emissions reductions on children's blood lead levels. The Science of the Total Environment 303:51-58.

Hilts, S.R., White, E.R., and Yates, C.L. 2001. Identification, Evaluation and Selection of Remedial Options. Trail Lead Program. Trail, BC, Canada.

Hilts, S.R., Bock, S.E., Oke, T.L., Yates, C.L., and Copes, R.A. 1998. Effect of Interventions of Children's Blood Lead Levels. Environmental Health Perspectives. 106 (2): 79-83.

Manitoba, 2007. Mercury in Fish & Guidelines for the Consumption of Recreationally Angled Fish in Manitoba. Manitoba Water Stewardship, Water Quality Management Section.

NAS/NRC. 1989. Report of the oversight committee. In: Biologic markers in reproductive toxicology. Washington, DC: National Academy of Sciences, National Research Council, National Academy Press. Cited In: ATSDR, 2007a.

Saskatchewan Environment and Resource Management (SERM), 1999. Mercury in Fish: Annual Guidelines for Consumption, Updated with data through 1999.

SARA. 2008. Sudbury Area Risk Assessment. Volume II: Human Health Risk Assessment. May 2008. The SARA Group. http://www.sudburysoilsstudy.com/EN/indexE.htm

SRC. 2002. Baseline Human Health Risk Assessment Eureka Mills – Eureka, Utah. Prepared by Syracuse Research Corporation. September, 2002. Available at: http://www.epa.gov/region8/r8risk/pdf/hhra_eureka.pdf.

U.S. EPA. 2001. Lead: Identification of Dangerous Levels of Lead. Federal Register 40 CFR Part 745, Volume 66, Number 4. Friday January 5, 2001. U.S. Environmental Protection Agency.

U.S. EPA. 2008. Public health application of human biomonitoring. Workshop summary report. Final. United States Environmental Protection Agency. http://www.epa.gov/ncer/publications/workshop/09_25_07/