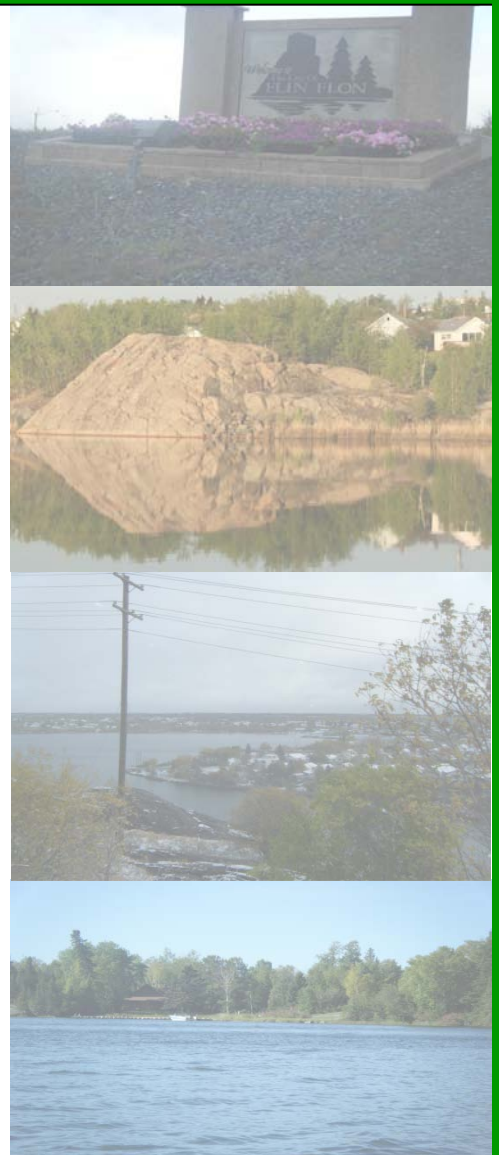


APPENDIX D

FLIN FLON DUST STUDIES AND RESULTS



APPENDIX D

FLIN FLON DUST STUDIES AND RESULTS

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D-1.0 FLIN FLON DUST STUDIES AND RESULTS

An important pathway to consider for the current HHRA is potential exposures to concentrations of the chemicals of concern (COC) present as part of indoor dust within Flin Flon area homes, schools and daycares. The Hudson Bay Mining and Smelting (HBMS) base metal smelting complex releases atmospheric emissions containing chemicals and particulate matter, including the COC. Gradually, wet and dry deposition causes the COC to settle onto local soils and other surfaces. Both the settled material and the airborne chemicals may be transferred into residential homes *via* human and local meteorological activity.

Outdoor yard soil can be transported indoors on clothing or shoes of humans or by animals, and combines with other sources to form household dust (U.S. EPA Region VIII, 2001). Studies have reported that between 20 and 30% of indoor contamination comes from outdoor soil sources (Rutz *et al.*, 1997). Exposure to concentrations of COC present in indoor environments is an important pathway of exposure for human health, especially for children.

During the problem formulation phase of the HHRA, it was recognized that there was no information on the concentrations of the COC in indoor dust in Flin Flon area. Therefore, an indoor dust survey was developed to fill this significant data gap. The primary objectives of this survey were as follows:

1. Measure concentrations of COC in indoor dust in the Flin Flon area;
2. Measure concentrations of the COC in outdoor soil samples to identify a relationship (if any) between indoor dust and outdoor soil concentrations;
3. Compare the data collected in Flin Flon area with other information and relationships reported in the literature;
4. If a relationship exists between COC concentrations in outdoor soil and indoor dust, use this relationship to predict indoor dust levels in indoor living spaces over the range of COCs levels reported in the soil survey; and,
5. Generate data that can be utilized to estimate human exposure to COC in indoor environments in the HHRA.

A total of 38 samples from homes, schools and daycares from the Flin Flon area were analyzed, including: Creighton (8); West Flin Flon (15); East Flin Flon (14) and Channing (1).

All dust samples were analysed for a total of 20 elements. However, the current study focused on the indoor dust and outdoor soil concentrations of the six COC being evaluated as part of the HHRA (*i.e.*, arsenic, cadmium, copper, lead, mercury and selenium).

Soil samples were collected from front lawns, back lawns, gardens and sandboxes for each of the Communities of Interest (COI) and were analyzed for 20 metals, however, the Final Metals Report only discusses the 12 metals associated with the CCME criteria (see Appendix B). The sample size was as follows, Creighton (68); West Flin Flon (107); East Flin Flon (141); Channing (18). For further details please refer to Appendix B (Final Metals Report).

A dust sampling event occurred in 2007, but due to insufficient volume of material collected the original study was unable to obtain the detection limits necessary for the risk assessment purposes (see Sub-Appendix J). The 2007 dust study did provide useful wipe sample results, limited information regarding lead in paint and a detailed tenant survey. These results are provided in Sub-Appendix J.

Due to the limited utility of the 2007 dust study, a supplemental dust sampling event took place in 2008 (see Sub-Appendix A). Dust samples were collected using a High Volume Small Surface Sampler vacuum (HVS3). A minimum of three sample areas (at least 1 square meter (sq. m) in size) were selected within each residence. Priority sample areas were selected based on the available carpeted floor area, high-traffic areas (family room) and children's play areas. Each sample area was measured and marked, and the HVS3 vacuum was pushed across the area at a consistent speed and pressure. On average, approximately 6 to 8 sq. m of carpeted floor space was sampled at each establishment in order to ensure adequate sample sizes for laboratory analysis. The composite dust sample was collected directly into a new laboratory-supplied nalgene bottle, which was capped and labeled prior to exiting the establishment. The bottles were collected in a plastic bin and stored at ambient room temperature until transported to Bodycote Testing Group, a CAEAL accredited laboratory. Between sampling events, the HVS3 vacuum was disassembled and sanitized with bottle brushes, methanol and alcohol wipes (see Sub-appendix A and D for further detail).

Results from the supplemental study indicated that lead dust vacuum results from two of 38 residential properties samples were exceptionally high. Results were compared to the 2007 dust wipe samples (Sub-Appendix J) which were low (less than available guidelines) and co-located yard soil was also low at one of the houses, with no yard soil sample having been taken at the other property. Based on the fingerprint of the metals within the samples, doubt was raised that the extreme levels found in these samples were connected to the facility. These two samples were considered statistical outliers. Both lead and arsenic vacuum dust results appeared elevated for about half of the homes tested although floor and windowsill wipe results (collected in 2007) for all homes were within acceptable ranges for lead. Therefore, concerns related to the accuracy of this data set were raised and all samples were sent to Testmark Laboratories in Sudbury for further analysis. Testmark was chosen since they have been used in previous studies for dust analysis, and they have a protocol developed for this type of analysis. In addition, splits of the larger vacuum dust samples were sent to Dr. John Drexler at the University of Colorado for speciation analysis to provide further insight into the source of the lead (and/or other metals) in the samples. Splits were also sent to the Queen's University Analytical Services Unit for confirmatory analysis. As detailed in Sub-Appendices E to G, dust sample analysis from Testmark was chosen as the dataset to complete the HHRA.

Soil samples were used to assist in evaluating the potential relationship between indoor dust and outdoor soils. Soils cores were collected by using a stainless steel soil probe with a 1.5 cm inner diameter core. A minimum of 10 cores were collected in an "X" pattern at each sample location. A composite sample was compiled for each sample depth and collected in a pre-labeled plastic-lined paper soil bag for laboratory analysis. The soil probe was cleaned with a plastic bottle brush and paper towel between sampling locations. A "dummy" core was collected and discarded at the start of each new sampling site in order to minimize the chances of cross contamination.

Tables D-1 and D-2 provide summaries of mean indoor dust and outdoor soil concentrations, respectively, from each community.

Table D-1 Summary of Mean Indoor Dust Concentrations by Community of Interest						
Community	Indoor Dust ($\mu\text{g/g}$)					
	Arsenic	Cadmium	Copper	Lead	Mercury	Selenium
Creighton	54.7	11.3	1,015	166	1.44	3.93
East Flin Flon	32.2	13.76	1,345	163	1.11	3.25
West Flin Flon	61.8	24.7	2,367	237	5.18	10.55
Channing	41.7	14.2	1,700	139	0.99	4.97
Total Residential Dataset (38)	48.9	17.6	1,688	192	2.78	6.3

() denotes number of samples

Table D-2 Summary of Mean Yard Soil (including front and backyards, gardens and sandboxes) Concentrations by Community of Interest						
Community	Yard Soil ($\mu\text{g/g}$)					
	Arsenic	Cadmium	Copper	Lead	Mercury	Selenium
Creighton	50.8	11.3	536.1	153.7	4.6	4.5
East Flin Flon	14.7	11.4	613.5	116.2	4.5	3.3
West Flin Flon	61.2	22	1845	290.9	91.9	27.9
Channing	13.9	7.6	274.6	90.3	1.9	1.5
Total Residential Dataset (334)	35.2	13.1	817.3	162.8	25.7	9.3

() denotes number of samples

Results of the residential survey indicated that the concentrations of the COC in dust and soil were consistently higher in West Flin Flon compared to the other communities. In general, dust levels of arsenic, cadmium, copper, and lead were higher than mean community soil concentration. These dust results were treated similarly to how the soil results were handled (*i.e.*, there were exceedances of the screening criteria; however there was no expected imminent health risk or need for immediate action, and this data was considered further in the HHRA). Although the lead concentrations in dust were higher than mean community soil concentrations, for the most part, the mean lead concentrations in each of the communities of interest did not follow similar trends as the other COC. This suggested that there is an alternate source of lead that may be present in some of the homes such as the presence of lead-based paints given the age of the dwellings.

To better understand the relationship between COC concentrations in indoor dust and outdoor soil, residential properties where both dust and soil samples were collected and analysed were considered. Analytical results were available for 25 co-located indoor dust and outdoor soil samples from residential properties across the Flin Flon area. The results are summarized in Table D-3.

Table D-3 Summary Statistics of Residential Indoor Dust and Outdoor Soil Data								
<i>Variable</i>	<i>Mean</i>	<i>Std Dev</i>	<i>Std. Error</i>	<i>N</i>	<i>Minimum</i>	<i>Maximum</i>	<i>Median</i>	<i>Skewness</i>
Indoor Dust								
Arsenic	51.52	26.72	5.34	25	13.8	138	44.2	1.45
Cadmium	17.18	8.53	1.71	25	4.88	41.3	14.9	1.3
Copper	1,691	751	150	25	541	3,250	1470	0.52
Lead	212	124	24.76	25	76	606	175	1.63
Mercury	2.56	2.5	0.5	25	0.27	11.9	2.08	2.4
Selenium	5.93	4.26	0.85	25	1.3	18.9	4.98	1.66
Yard Soil								
Arsenic	38.06	46.06	9.21	25	5	237	29.8	3.62
Cadmium	15.82	11.11	2.22	25	1.32	55.1	12.71	2.04
Copper	1,146	1152	230	25	72	5,260	720	2.2
Lead	186	160	32.08	25	23.2	804	148	2.6
Mercury	32.14	66.62	13.32	25	0.02	300	4.8	3.14
Selenium	13.1	21.38	4.28	25	0.7	97.3	4.25	3

To analyze the changes in dust concentrations relative to soil concentrations, concentration ratios were calculated. Concentration ratio (CR) values can be defined as the concentration of a specific metal observed in indoor dust ($\mu\text{g/g}$) divided by the concentration observed in co-located yard soil ($\mu\text{g/g}$). CR values were calculated for each site, the results of which have been summarized in Table D-4.

Table D-4 Summary Statistics of Residential Concentration Ratio Values^a								
<i>Variable</i>	<i>Mean</i>	<i>Std Dev</i>	<i>Std. Error</i>	<i>N</i>	<i>Minimum</i>	<i>Maximum</i>	<i>Median</i>	<i>Skewness</i>
Arsenic	2.29	1.75	0.35	25	0.38	8.5	1.62	1.98
Cadmium	1.75	2.18	0.44	25	0.35	11.29	1.11	3.78
Copper	3.19	4.99	1	25	0.52	25.69	1.67	4.16
Lead	2.04	2.69	0.54	25	0.38	12.65	1.27	3.19
Mercury	7.63	3.19	6.40	25	0.03	160.5	0.20	4.93
Selenium	1.37	1.89	0.38	25	0.19	9.23	0.82	3.36

^a CR value defined as [indoor dust $\mu\text{g/g}$]/[yard soil $\mu\text{g/g}$]

With the exception of mercury and selenium, all median CR values (n=25) were greater than 1.0. This indicates that indoor dust levels were on average 1.4 to 7.6 times higher than corresponding soil levels. However, the data also indicates that the CR values do not remain constant over a large range of yard soil concentrations, suggesting that indoor dust concentrations do not simply increase (in a linear fashion) with increasing soil concentration. A scatter plot of mercury CR values ($[\text{indoor dust} - \mu\text{g/g}] / [\text{yard soil} - \mu\text{g/g}]$) versus corresponding mercury concentrations in yard soil is provided in Figures D-1 and D-2. CR values which were greater than 1 were removed (n=5), therefore, the sample size was 20. The scatter plot indicates that CR values (describing the relationship between mercury in indoor dust and yard soil) vary with increasing yard soil concentrations. As yard soil concentrations increase, CR values tend to decrease. Figure D-2 represents a scatter plot of the transformed data set (ln transformed).

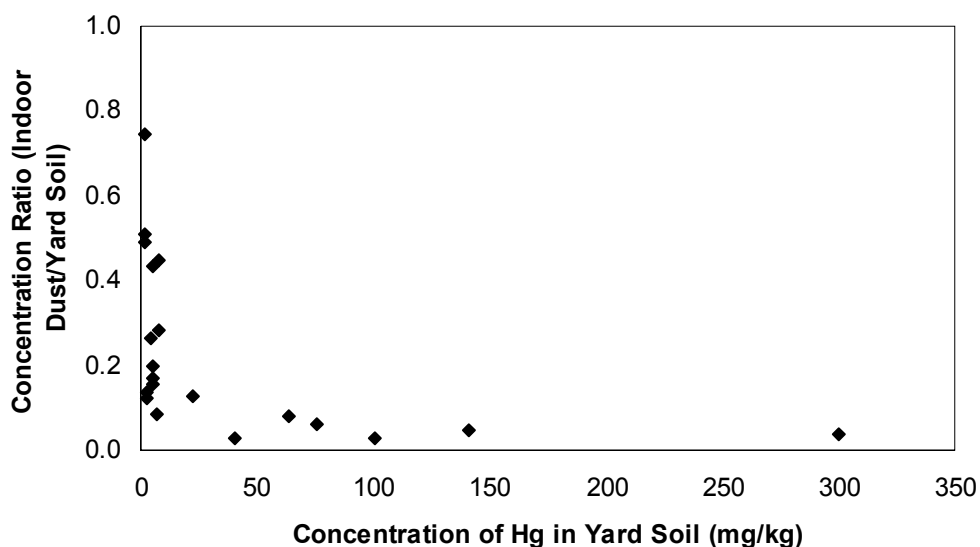


Figure D-1 Scatter Plot of Mercury CR Values ([indoor dust] / [yard soil]) Versus Mercury Concentrations in Outdoor Yard Soil ($\mu\text{g/g}$)

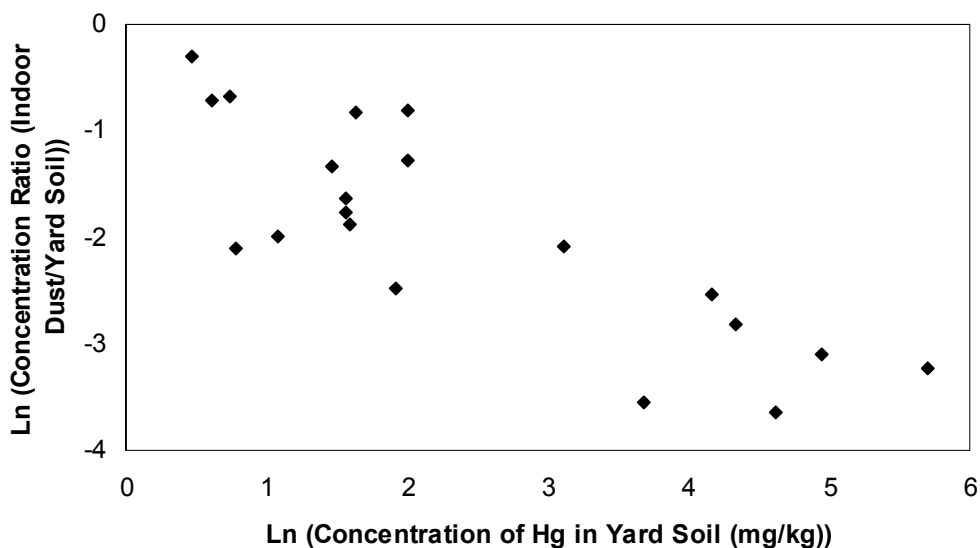


Figure D-2 Scatter Plot of Ln Transformed Data – Mercury CR Values Versus Mercury Concentrations in Outdoor Yard Soil ($\mu\text{g/g}$)

A scatter plot of selenium CR values ($[\text{indoor dust} - \mu\text{g/g}] / [\text{yard soil} - \mu\text{g/g}]$) versus corresponding selenium concentrations in yard soil is provided in Figure D-3. CR values which were greater than 1 were removed ($n=8$), therefore, the sample size was 17. The scatter plot indicates that CR values (describing the relationship between selenium in indoor dust and yard soil) vary with increasing yard soil concentrations. As yard soil concentrations increase, CR values tend to decrease. Figure D-4 represents a scatter plot of the transformed data set (ln transformed).

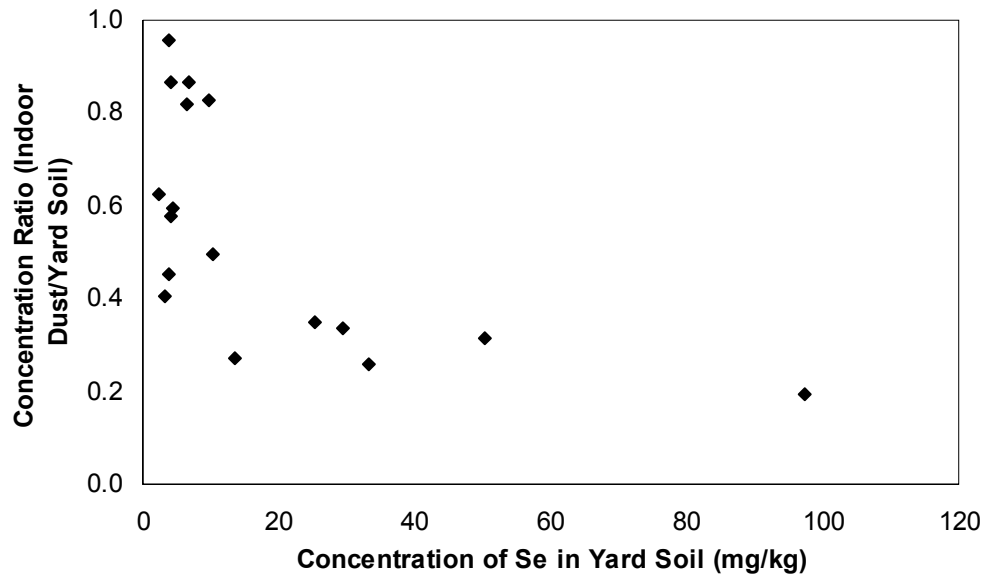


Figure D-3 Scatter Plot of Selenium CR Values ([indoor dust] / [yard soil]) Versus Mercury Concentrations in Outdoor Yard Soil ($\mu\text{g/g}$)

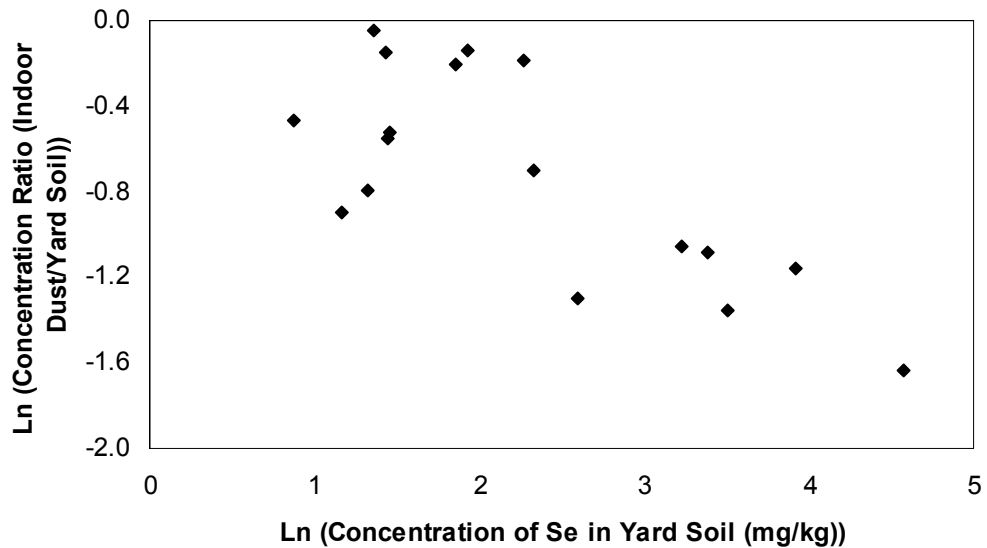


Figure D-4 Scatter Plot of Ln Transformed Data –Selenium CR Values Versus Selenium Concentrations in Outdoor Yard Soil ($\mu\text{g/g}$)

With the exception of mercury and selenium, all median CR values ($n=25$) were greater than 1.0. This indicates that indoor dust levels were on average 1.4 to 7.6 times higher than corresponding soil levels. However, the data also indicates that the CR values do not remain constant over a large range of yard soil concentrations, suggesting that indoor dust concentrations do not simply increase (in a linear fashion) with increasing soil concentration. Initial analyses suggest that at lower outdoor soil concentrations, other non-soil related sources become a more dominant source of indoor COC (e.g., lead paint, household sources, etc.).

This leads to the higher CR values observed in typical urban settings, and is consistent with results observed in other dust studies. As outdoor soil concentrations increase, soil appears to become a more significant contributor to ambient dust concentrations of the COC and indoor concentrations no longer match or exceed outdoor concentrations. This results in much lower CR values observed in metal-impacted areas, such as is present in some areas of Flin Flon. As such, CR values do not provide an accurate reflection of potential indoor dust concentrations given the full spectrum of COC soil concentrations observed in Flin Flon.

To establish a more statistically-robust comparison, linear regression equations were developed for each COC to predict indoor dust concentrations as a function of outdoor soil concentrations. Table D-5 provides the linear regression equations (*i.e.*, ln-transformed) which provided the best-fit based upon the paired outdoor soil and indoor dust concentration sets obtained from this study.

COC	Equation^a	R²	P Model Fit	N
Arsenic ^b	$\ln[\text{indoor dust}] = ((2.66416 \pm 0.7991) \times \log[\text{soil}])^2 + (3.21804 \pm 1.16241)^2$	0.3258	0.0029	25
Cadmium	$\ln[\text{indoor dust}] = 0.37204 \pm 0.1400 \times \ln[\text{soil}] + 11.29628 \pm 2.6904$	0.2347	0.0141	25
Copper	$\ln[\text{indoor dust}] = 0.38662 \pm 0.1095 \times \ln[\text{soil}] + 1248.30160 \pm 176.2077$	0.3513	0.0018	25
Lead	No statistically significant regression			
Mercury	$\ln[\text{indoor dust}] = 0.03324 \pm 0.00366 \times \ln[\text{soil}] + 1.49007 \pm 0.2663$	0.7802	<0.0001	25
Selenium	$\ln[\text{indoor dust}] = 0.17571 \pm 0.01970 \times \ln[\text{soil}] + 3.62776 \pm 0.4867$	0.7757	<0.0001	25

^a $\ln[\text{indoor dust}] = \beta_0 \pm \text{SE} \times \ln[\text{soil}] + C \pm \text{SE}$

^b Equation derived through regression $[\text{dust}] = (3.21804 + 2.66413 \log \text{soil})^2$

SE Standard error

It is important to note that the slope of the best fit linear regression line and the mean CR value for any COC are not equivalent. The slope of the regression line was determined by the method of least squares and represents the rate of change (over a specific concentration range) in the indoor dust level as a function of yard soil concentration, while CR values are defined as the concentration of metal in indoor dust ($\mu\text{g/g}$) divided by the concentration observed in co-located yard soil ($\mu\text{g/g}$).

In general, outdoor soil could not account for a large percentage of the variance observed in indoor dust concentrations. The regression models presented in Table D-5 were able to explain approximately 23 to 78% of the variation observed in indoor dust concentrations. With the exception of lead, regression equations for all COC were statistically significant and considered appropriate for the development of Flin Flon-specific dust-to-soil relationships. These relationships were used to generate dust exposure values for the Flin Flon HHRA. Figures D-5 through D-10 provide the regression plots for all COC with the exception of lead.

Since there was no statistically significant relationship observed between concentrations of lead in outdoor soil and indoor dust, a site-specific regression equation was not used to predict concentrations of lead in indoor dust. Instead, the IEUBK Multiple Source Analysis (MSA) module was used to predict concentrations of lead in indoor dust. This involves assigning a

value to represent the mass fraction (M_{SD}) of house dust that is derived from outdoor soil. The IEUBK default value of 0.70 g soil/g dust was used for the current assessment. In addition to the contribution of outdoor soil to indoor dust lead levels, the contribution of impacted outdoor air is also considered in the MSA. Again, using the IEUBK default values, an additive increment of 100 $\mu\text{g/g}$ of lead in indoor dust for every 1 $\mu\text{g}/\text{m}^3$ of lead in outdoor air was added to the contribution from outdoor soil. For example, for a given scenario in which the concentration of lead in outdoor soil is 200 $\mu\text{g/g}$ and the concentration in outdoor air is 0.1 $\mu\text{g}/\text{m}^3$, the predicted indoor dust concentration would be 150 $\mu\text{g/g}$ ($(200 \mu\text{g/g} \times 0.7) + (100 \mu\text{g/g} \times 0.1)$). Use of the MSA module was preferred over the use of the measured 95% UCLM indoor dust concentration because when deriving preliminary remediation goals (PRGs) or soil trigger concentrations (STCs) it allows the model to adjust the indoor dust concentration as the outdoor soil concentration is increased or decreased. It should be noted that the indoor dust concentration predicted using the MSA module and the EPCs for outdoor soil and outdoor air is similar to the measured EPC for indoor dust. For example, for the community of West Flin Flon, the MSA approach predicted an indoor dust concentration of 289 $\mu\text{g/g}$ based on an outdoor soil concentration of 370 $\mu\text{g/g}$ and an outdoor air concentration of 0.3 $\mu\text{g}/\text{m}^3$. The EPC for indoor dust in West Flin Flon based on measured data was 265 $\mu\text{g/g}$.

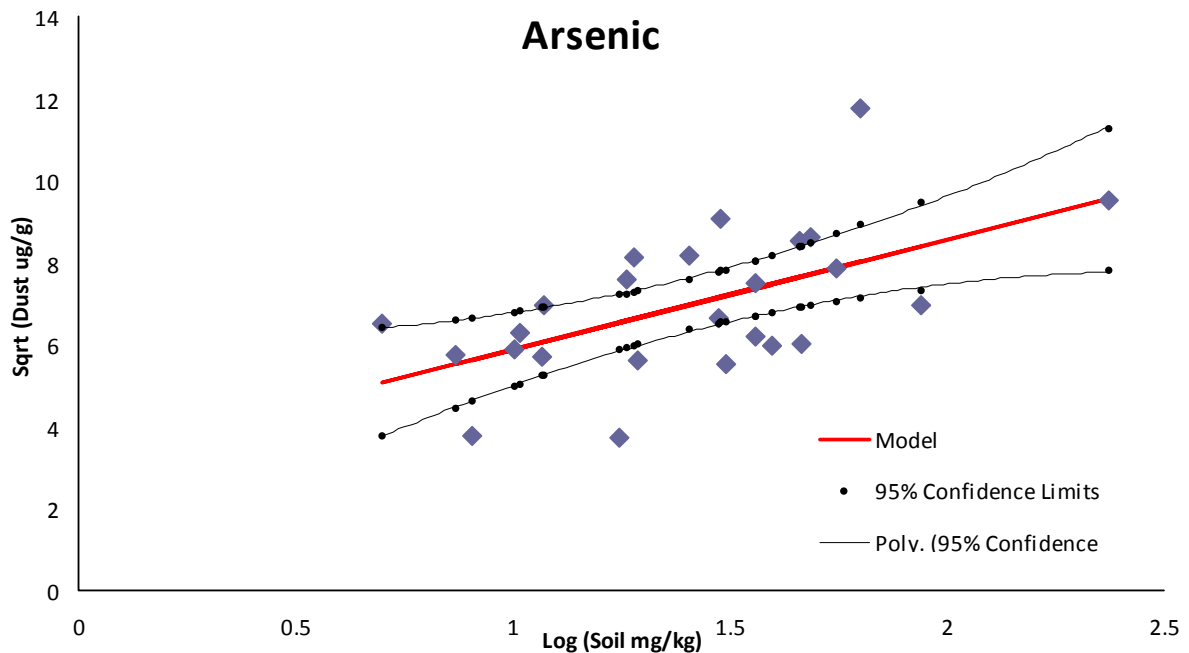


Figure D-5 Prediction Plot – Arsenic (Transformed)

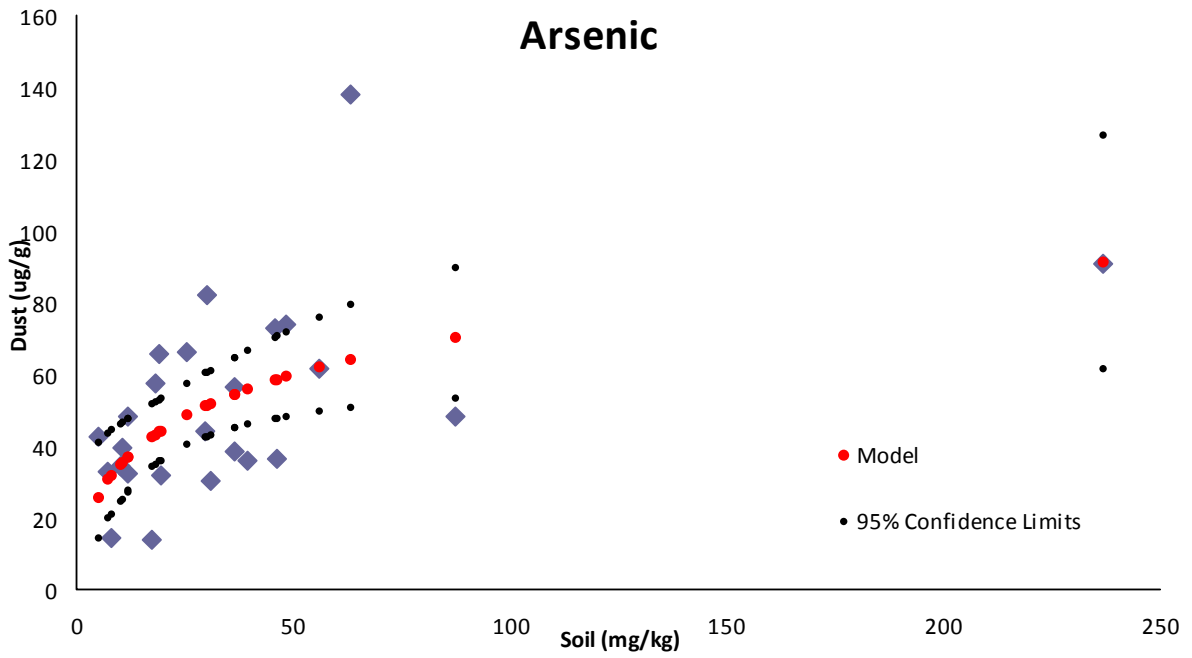


Figure D-6 Prediction Plot – Arsenic (Linear)

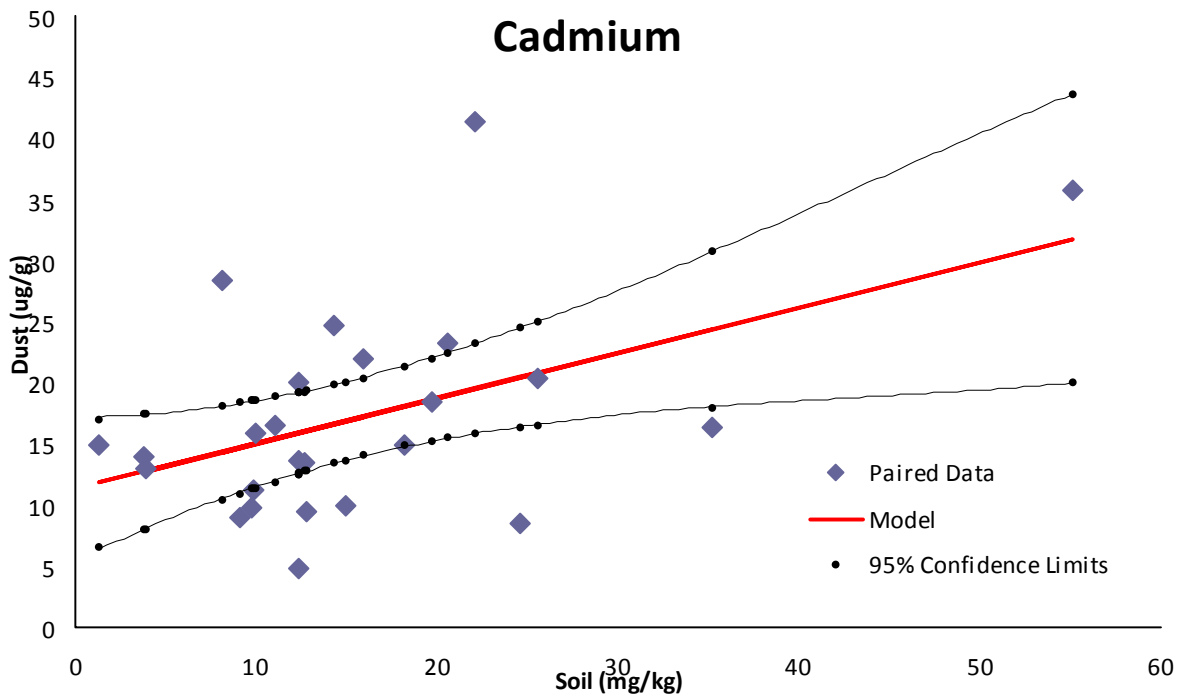


Figure D-7 Prediction Plot – Cadmium

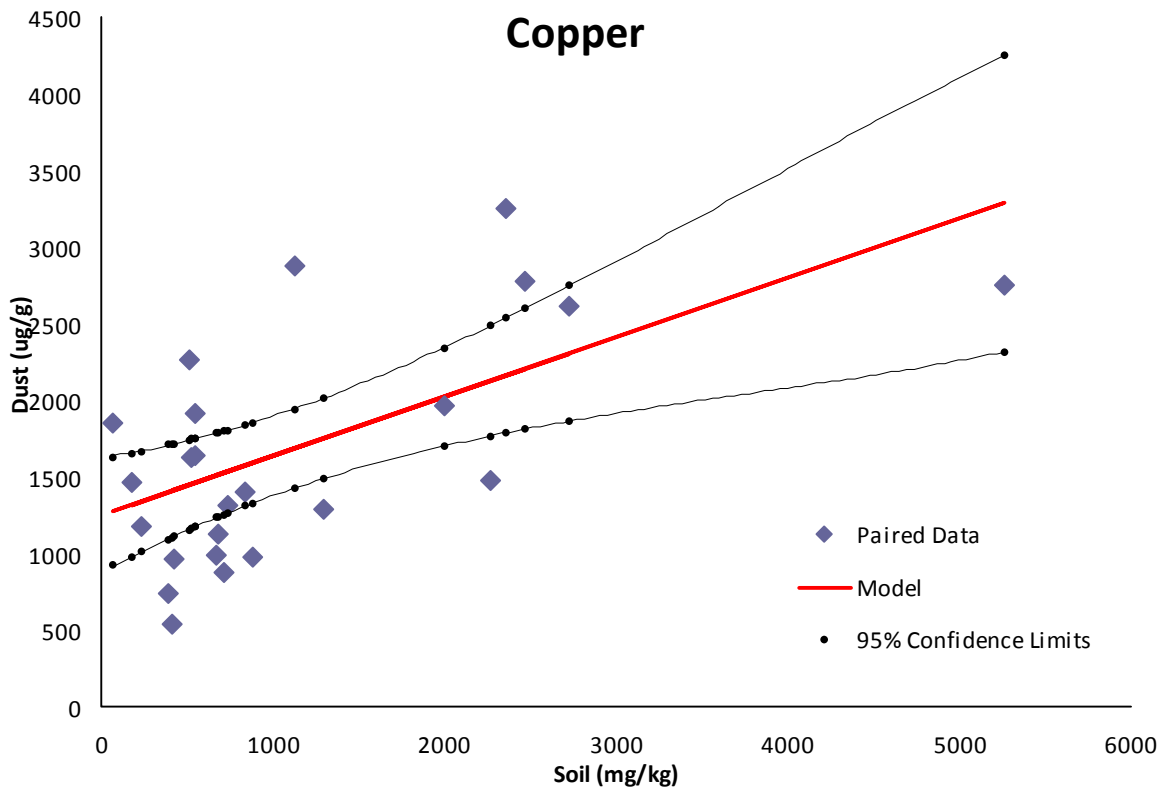


Figure D-8 Prediction Plot – Copper

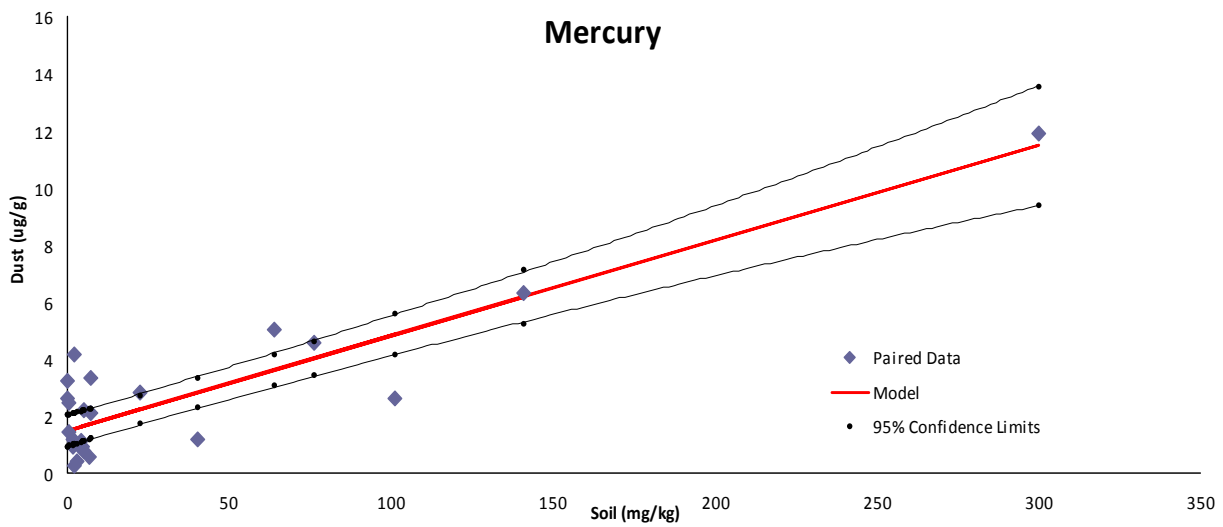


Figure D-9 Prediction Plot – Mercury

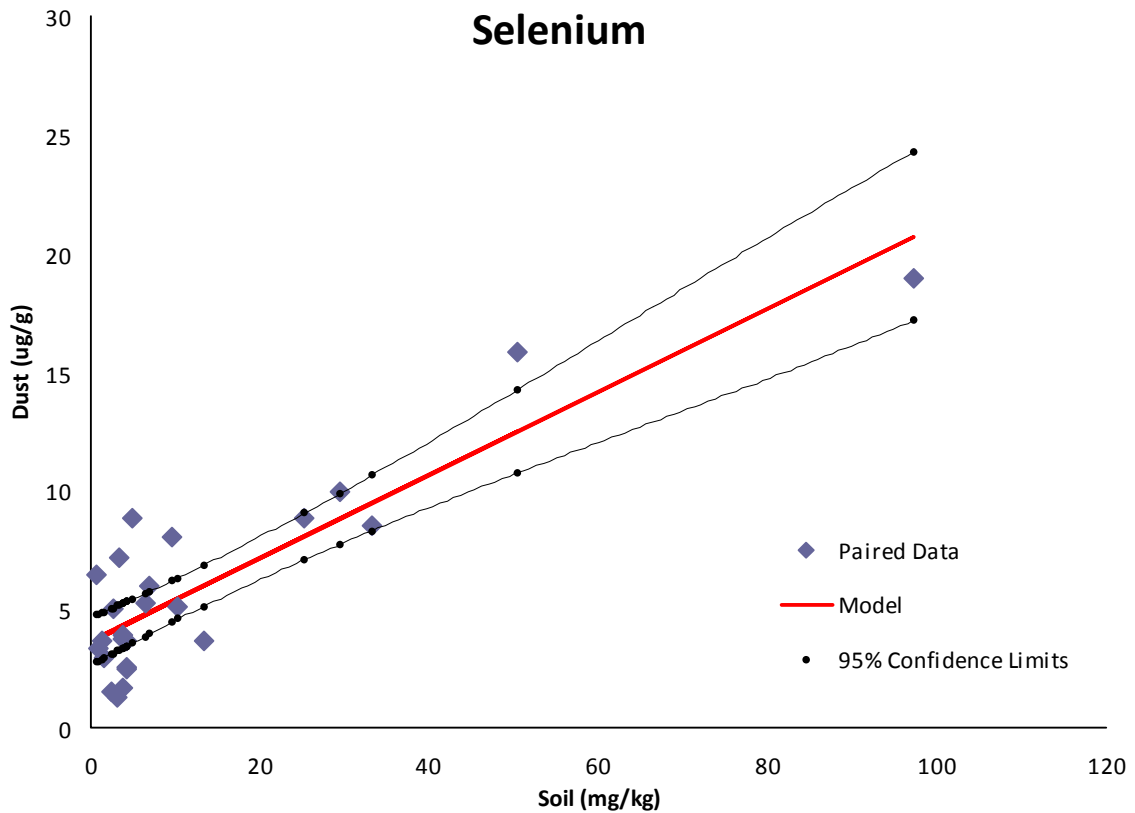


Figure D-10 Prediction Plot –Selenium

The slope of the best fit linear regression line (as reported above) and the CR value for any COC is not equivalent. The slope of the regression line was determined by the method of least squares and represents the rate of change (over a specific concentration range) in the indoor dust level as a function of yard soil concentration. CR values are defined as the concentration of metal in indoor dust ($\mu\text{g/g}$) divided by the concentration observed in co-located yard soil ($\mu\text{g/g}$). As previously demonstrated, the CR values do not remain constant over a large range of yard soil concentrations. As the concentration in outdoor yard soil increases, CR values decrease, suggesting that indoor dust concentrations do not simply increase (in a linear fashion) with increasing soil concentration. A comparison between predicted Cd indoor dust concentrations using the CR approach *versus* the linear regression equation provided in Table D-5 has been provided in Figure D-11.

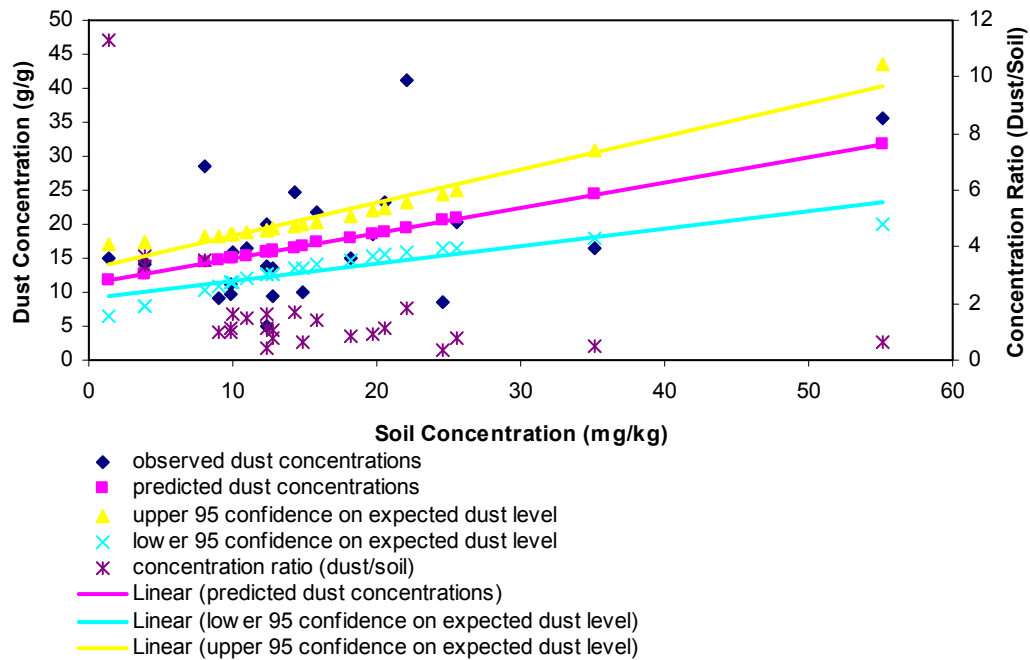


Figure D-11 Predicted Cd Concentrations in Indoor Dust using the 95% UCLM CR Value and the Linear Regression Equation

The 95% UCLM Cd predicted dust level derived from the linear regression equation for Cd was used to predict indoor dust levels over a range of yard soil concentrations observed in the Flin Flon area. Compared to the observed Cd concentrations, it appeared that the Upper Confidence Limit was an accurate predictor.

The 95% UCLM Cd CR value tends to decrease as the concentrations of Cd increase in the yard soil. It appears that the CR values can be used as a predictor of indoor dust concentrations when they are applied to relatively low concentrations of cadmium in soil. As concentrations in yard soil increase, however, CR values tend to decrease indicating high dust and soil concentrations similar to the linear regression methods. However, from Figure 4-1 that the indoor dust-to-soil ratio does not remain constant over a wide range of metal concentrations in soil and can be difficult if used to predict dust concentrations. Based on the discussion above, the use of regression equations to estimate indoor dust concentrations from outdoor soil concentrations is preferred to the use of CR values.

D-2.0 REVIEW OF OTHER INDOOR DUST STUDIES

Several studies have been conducted to define the relationship between levels of contaminants in household or indoor dust *versus* outdoor soil. Based on a review of the literature, there seems to be adequate evidence to suggest a relationship exists between levels of contaminants measured in indoor dust and the levels observed in nearby outdoor soil. However, there are varying degrees of certainty and strength to this relationship.

D-2.1 Literature Review of Relationships between Metal Concentrations in Outdoor Soil versus Indoor Dust

A recent study by Rasmussen (2004) collected indoor dust and outdoor soil and dust from 48 homes across the city of Ottawa, Ontario. Dust samples were collected into new vacuum cleaner bags by the homeowner. Outdoor soil samples consisted of a composite of the top 5 cm of soil from five locations in the yard surrounding the home being sampled. The results of the study indicated that the multi-element composition of indoor dust differed significantly from that of garden soil or street dust. For most elements, levels in household dust exceeded natural background concentrations for the region whereas most concentrations in garden soil and street dust were low in comparison with local background concentrations. The authors found no significant correlations for element concentrations in household dust *versus* street dust or household dust *versus* garden soil. Mean CRs for indoor dust *versus* garden soils were greater than 1 for most elements (see Table D-6). Rasmussen (2004) developed mean CR values for arsenic and selenium of 1.69 and 1.67 respectively. These were similar to the mean CR values developed using the current dataset of 2.29 and 1.67 for arsenic and selenium, respectively.

A study by Harrison (1979) found similar indoor/outdoor dust concentration ratios in a study of household and street dust in Lancaster area of the United Kingdom. The mean levels of total metals in household dust were found to be higher than in dust collected from rural roads, and to a lesser degree, urban roads. Mean concentration ratios for household dust *versus* urban road dust and for household dust *versus* rural road dust were greater than 1 for most elements (Table D-6). Lancaster area is not considered to be impacted by significant industrial emissions.

A population-based exposure survey conducted by Hwang *et al.* (1997) in Anaconda, Montana, examined the relationship between exterior soil and indoor dust in homes near a historic copper smelter operation. Geometric mean arsenic concentrations in five types of soil collected around the exterior of the homes ranged from 121 to 236 $\mu\text{g/g}$, with a total average soil concentration of 192 $\mu\text{g/g}$. An average arsenic level of 75.14 $\mu\text{g/g}$ was reported for indoor dust. Significant correlations reported between arsenic concentrations in indoor dust and each of the five soil types collected ranged from $r=0.42$ to $r=0.52$ ($p<0.0005$) (log-transformed data), with an average arsenic concentration ratio for indoor dust to outdoor soil of 0.391.

Calabrese (unpublished, as reported in Walker and Griffin, 1998) reported a similar indoor dust/outdoor soil concentration ratio of 0.387 for average arsenic concentrations in samples collected from a smaller sub-set of the same homes in Anaconda, Montana. Calabrese used different sampling methods and reported significantly different arsenic concentrations in soil (average=74.67 $\mu\text{g/g}$) and indoor dust (average=29.03 $\mu\text{g/g}$); however, the average indoor/outdoor concentration ratio was similar to the ratio observed by Hwang *et al.* (1997).

The Ontario Ministry of the Environment (MOE) recently conducted a review of available literature discussing the relationship between concentrations of metals in outdoor soil and indoor dust for the Rodney Street Community Human Health Risk Assessment in Port Colborne, Ontario (MOE, 2002). The review included an evaluation of a study conducted by PTI Environmental Services (PTI, 1994) at a contaminated site in Bartlesville, Oklahoma, in which levels of various metals were measured in outdoor soil and indoor dust. The study reported concentration ratios of indoor dust to outdoor soil for arsenic, cadmium, lead, and zinc, of 0.20, 0.35, 0.50, and 0.36, respectively (MOE, 2002). The MOE selected a CR value of 0.39, derived from the Hwang and Calabrese studies, to define the relationship between indoor dust and outdoor soil concentrations of nickel in Port Colborne.

Studies conducted in areas that have historically, or continue to be, impacted by neighbouring industrial activities, such as mining or smelting operations, provide evidence that a significant relationship exists between metal concentrations in outdoor soil and levels found in household dust. Regions or cities with less industrial activity have not shown evidence to support this relationship; however, in most studies, higher levels of contaminants are reported in household dust than in surrounding outdoor soil, which tends to be similar to natural background levels. Table D-6 provides a summary of the ratios between metal levels in indoor dust and outdoor soil reported in various studies.

<i>Relationship</i>	<i>Mean CRs</i>								<i>Reference.</i>
	<i>As</i>	<i>Pb</i>	<i>Ni</i>	<i>Se</i>	<i>Cu</i>	<i>Co</i>	<i>Cd</i>	<i>Zn</i>	
Indoor dust/ outdoor soil	1.69	5.50	3.37	1.67	13.90	1.03	16.37	6.18	Rasmussen, 2004
	0.391	-	-	-	-	-	-	-	Hwang <i>et al.</i> , 1997
	0.387	-	-	-	-	-	-	-	Calabrese (unpublished)
	0.20	0.50	-	-	-	-	0.35	0.36	PTI, 1994
Indoor dust/ street dust	4.9	6.95	3.62	2.5	4.48	1.05	13.39	6.20	Rasmussen, 2004
Indoor dust/ urban road dust	-	0.381	1.229	-	1.545	0.934	2.326	2.191	Harrison, 1979
Indoor dust/ rural road dust	-	1.302	2.324	-	2.797	1.197	5.095	3.939	Harrison, 1979

Regression analysis is an approach often used to examine the relationship between metal concentrations in indoor dust and outdoor soil. The slope of a regression equation (dust concentration/soil concentration) for a data set can be used to define the changes in dust concentration over a range of soil concentrations. However, it should be noted that the slope of a regression equation for the plot of indoor dust concentrations over outdoor soil concentrations and mean concentration ratios for the same data set are not analogous and should not be used for comparison of different data.

Regression analysis conducted by Murgueytio *et al.* (1998) on indoor dust and outdoor soil data collected during an exposure study in the Big River Mine Tailings site, south of St. Louis, Missouri, found a significant correlation between indoor dust lead concentrations and outdoor soil lead levels ($r^2=0.36$; $p<0.000$ [sic]).

The relationship between contaminants in indoor dust and outdoor soil in residential homes has also been examined by the U.S. Environmental Protection Agency (U.S. EPA) at several Superfund sites across the United States. Residential dust sampling at one Superfund site near Denver, Colorado, found only a weak correlation between the levels of arsenic and lead in soil and dust ($r^2=0.14$ and 0.18 , respectively; $p<0.01$). However, regression analysis of the paired soil and dust data revealed statistically significant regression line slopes, for both arsenic and lead (U.S. EPA Region VIII, 2001). A comparison of soil-dust relationships for arsenic and lead at other U.S. EPA Region VIII Superfund sites is presented in Table D-7.

Table D-7 Soil-Dust Relationships at Other U.S. EPA Superfund Sites^a		
Site	Slope of Regression Line (ppm dust/ppm soil)	
	Arsenic	Lead
Anaconda, MT	0.31	--
Bingham Creek, UT	--	0.43
Butte, MT	--	0.24
Deer Lodge, MT	0.001	-0.01
East Helena, MT	--	0.88
Flagstaff/Davenport, UT	--	0.06
Midvale OU1, UT	0.03	0.04
Leadville, CO	0.10	0.33
Murray Smelter, UT	0.17	0.19
Sandy City, UT	--	0.13
Sharon Steel, UT	--	0.76
Vasquez Blvd. & I-70, CO	0.06	0.34

^a U.S. EPA Region VIII, 2001

D-2.2 Indoor Dust Study for HHRA conducted in Port Colborne, Ontario

As part of a Community Based Risk Assessment conducted in 2002, indoor dust and co-located outdoor soil concentrations were collected from 31 residential homes and properties in Port Colborne, Ontario (JWEL, 2004). The study involved the collection of four types of indoor dust samples including indoor air samples, fabric surface samples, hard surface samples and attic samples. Various collection methods were used to collect each dust sample type, including air flow pumps and filters to collect indoor air samples of total suspended solids (TSP) and PM₁₀ particles, personal air sampling pumps with nozzles to collect dust from carpets and upholstery, moistened wipes to collect dust from hard surfaces and wipe and grab samples for the collection of attic dust. Soil cores were collected concurrently from the yards of the same residences. Analysis of the indoor dust and outdoor soil concentrations for cobalt, copper and nickel showed significant correlations between four dust sample types and concentrations in yard soil. Nickel in TSP ($r=0.36$, $p<0.049$), cobalt and nickel in fabric samples ($r=0.50$, $p<0.005$ and $r=0.61$, $p<0.0004$, respectively) and nickel in hard surface samples ($r=0.50$, $p<0.005$) were significantly correlated to soil concentrations. However, regression analysis indicated the strength the significant correlations was low; $r^2=0.13$ for nickel in TSP, $r^2=0.25$ and $r^2=0.37$ for cobalt and nickel in fabric samples, respectively, and $r^2=0.25$ for nickel in hard surface dust.

The results of the study were used to calculate reasonable maximum long-term average concentrations of metals in indoor dust (95% upper confidence limits on the mean). These values were then used to estimate human exposure as part of a human health risk assessment.

D-3.0 REFERENCES

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