APPENDIX M

PUBLIC REVIEW DRAFT

BASELINE HUMAN HEALTH RISK ASSESSMENT
FOR THE
OMAHA LEAD SITE
OMAHA, NEBRASKA

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LIST OF ACRONYMS AND ABBREVIATIONS

ABA	Absolute Bioavailability
ABS	Dermal Absorption Fraction
AF	Adherence Factor
AFo	Oral Absorption Fraction
AQS	Air Quality System
ASARCO	American Smelting and Refining Company
AT	Averaging Time
ATSDR	Agency for Toxic Substances and Disease Registry
bgs	Below Ground Surface
BHHRA	Baseline Human Health Risk Assessment
BW	Body Weight
CBW	Council Bluffs Waterworks
CDC	Centers for Disease Control
CF	Conversion Factor
COPC	Chemical of Potential Concern
CSM	Conceptual Site Model
CTE	Central Tendency Exposure
CVAA	Cold Vapor Atomic Absorption
DA	Absorbed Dose per Event
DAD	Dermal Absorbed Dose
DCHD	Douglas County Health Department
DI	Daily Intake
DIL	Daily Intake Lifetime
DQA	Data Quality Assessment
ED	Exposure Duration
EF	Exposure Frequency
EPC	Exposure Point Concentration
EV	Event Frequency
FDA	Food and Drug Administration
GM	Geometric Mean
GSD	Geometric Standard Deviation
HEAST	Heath Effects Assessment Summary Tables
HI	Hazard Index
HIF	Human Intake Factor
HVS3	High Volume Small Surface Sampler
HQ	Hazard Quotient
HUD	Housing and Urban Development
IARC	International Agency for Research on Cancer
ICP-AES	Inductively Coupled Plasma Atomic Emission Spectroscopy

IEUBK	Integrated Exposure Uptake Biokinetic Model
IR	Intake Rate
IRIS	Integrated Risk Information System
IVBA	*In Vitro* Bioaccessibility
LOAEL	Lowest-observed-adverse-effect-level
MRLs	Minimal Risk Levels
MUD	Metropolitan Utilities District
NOAEL	No-observed-adverse-effect-level
NPL	National Priorities List
NTP	National Toxicology Program
OLS	Omaha Lead Site
P10	Probability of having a blood lead level that exceeds 10 µg/dL
PA/SI	Preliminary Assessment/Site Investigation
PPRTVs	Provisional Peer Reviewed Toxicity Values
RAGS	Risk Assessment Guidance for Superfund
RBA	Relative Bioavailability
RfD	Reference Dose
RI	Remedial Investigation
ROD	Record of Decision
RME	Reasonable Maximum Exposure
SA	Surface Area
SAP	Sampling and Analysis Plan
SF	Slope Factor
STSC	Superfund Health Risk Technical Support Center
TAL	Target Analyte List
TWA	Time-Weighted Average
UCL	Upper Confidence Limit
USEPA	United States Environmental Protection Agency
WOE	Weight of Evidence
XRF	X-Ray Fluorescence

EXECUTIVE SUMMARY

1.0 INTRODUCTION

This document is the final baseline human heath risk assessment (BHHRA) for the Omaha Lead Site (OLS) located in Omaha, Nebraska. The purpose of this document is to characterize the risks to area residents, both now and in the future, from site-related contaminants present in environmental media, assuming that no steps are taken to remediate the environment or to reduce human contact with contaminated environmental media. The results of this final assessment are intended to help inform risk managers and the public about potential human risks attributable to site-related contaminants and to help determine where there is a need for action at the site.

2.0 SITE CHARACTERIZATION

Site Location and Description

The OLS is located in Omaha, Nebraska. The site encompasses the eastern portion of the greater metropolitan area, excluding Omaha's Central Business District (Figure ES-1). It includes residential properties, child-care facilities, vacant lots, and other residential-type properties (schools, churches, parks) that have been contaminated as a result of historic air emissions from lead smelting/refining operations.

Basis for Concern

In the past, two lead smelters operated in the eastern part of the OLS. Historical smelting, refining, and recycling activities that occurred at these smelters resulted in the release of lead and other smelter-related metals into air in the form of particulate matter. These particulates in the air were transported by wind and deposited onto soil in the vicinity of the site. Humans may be exposed to lead and other smelter-related metals in soil by a number of pathways, and adverse health effects may occur if concentrations and exposure levels are high enough.

One measure of human exposure to lead is the concentration of lead in the blood, generally expressed in units of micrograms of lead per deciliter of blood (µg/dL). Concentration values above 10 µg/dL are generally considered to be elevated and of potential health concern. The Douglas County Health Department (DCHD) has been monitoring lead levels in the blood of children living in Douglas County for a number of years. Figure ES-2 shows the percentage of children with elevated blood leads (> 10 µg/dL) in the area of the site compared with State and national data. As shown, the frequency of elevated blood lead values in the area of the OLS have been consistently higher than elsewhere. This indicates that children living in the area of the site are being exposed to lead in the environment at levels that exceed other locations, and which may be of potential health concern to some children.

3.0 SITE CONCEPTUAL MODEL

Exposure Media and Pathways of Potential Concern

Figure ES-3 presents a general conceptual model of how smelter-related contaminants that have been released to the environment at the OLS might result in exposure of humans. The environmental medium of chief concern is surface soil that has been impacted by the wet or dry deposition of metal-containing airborne particulates released from the smelters. The human population of chief concern is residents in the area of the site, now or in the future, including both children and adults. Residents might be exposed to smelter-related contaminants in site soils by a number of different pathways, including ingestion, inhalation, and dermal contact with contaminated soil or dust, and ingestion of home-grown produce that may have taken up contaminants from the soil. However, not all of these are likely to be of equal concern for all contaminants. Section 4 describes the exposure pathways that are of chief concern for lead, and Section 5 describes the exposure pathways that are of chief concern for other (non-lead) contaminants.

Selection of Chemicals of Potential Concern

Chemicals of Potential Concern (COPCs) are chemicals which exist in the environment at concentration levels that might be of potential health concern to humans and which are or might be derived, at least in part, from site-related sources. The chief COPC at this site is lead. However, several other chemicals were identified that might also be of potential concern to humans, including the following:

CHEMICALS OF POTENTIAL CONCERN	
Aluminum	Iron
Antimony	Lead
Arsenic	Manganese
Cadmium	Mercury
Chromium	Thallium
Cobalt	Vanadium
Copper	Zinc

4.0 EVALUATION OF RISKS FROM LEAD

4.1 Exposure Assessment

Population of Chief Concern

The population of chief concern for lead exposure is young children (age 0-84 months). This is because young children tend to have higher intakes of lead than adults, tend to absorb more lead than adults, and are inherently more sensitive to lead than adults. If environmental exposures to lead in a residential area are acceptable for young children, exposures are usually also acceptable for older children and adults, including pregnant women.

Exposure Pathways of Chief Concern

Young children can be exposed to lead from a variety of sources. At this site, the environmental medium of chief concern is outdoor soil that became contaminated with lead released from historic smelter operations. Exposure to smelter-contaminated soil may occur through ingestion of soil or dust, or through inhalation. Dermal absorption of lead from soil or dust is considered to be very minor, as is uptake of lead from soil into home-grown produce.

In addition to these exposures to smelter-related releases of lead, children may also be exposed to lead from other sources as well. This includes lead from leaded paint, as well as lead in drinking water and food from grocery stores. Because risk from lead depends on exposure from all of these sources, these exposure pathways are also included in the risk evaluation for lead.

4.2 Toxicity Assessment

Non-Cancer Effects

Excess exposure to lead can result in a wide variety of adverse effects in children. However, the most important effect is on the nervous system. The effects of chronic low-level exposure on the nervous system include decreased performance in various types of tests of intelligence, attention span, hand-eye coordination, etc. The overall weight of the available evidence provides clear evidence of nervous system effects in young children occurring at blood lead levels in the range of 5-10 µg/dL, and possibly lower. Such effects on the nervous system are long-lasting and may be permanent.

Cancer Effects

Studies in animals indicate that chronic oral exposure to very high doses of lead salts may cause an increased frequency of tumors of the kidney. However, there is only limited evidence

suggesting that lead may be carcinogenic in humans, and the non-carcinogenic effects on the nervous system are usually considered to be the most important and sensitive endpoints of lead toxicity.

Current Guidelines for Protecting Children from Lead

The USEPA currently identifies 10 µg/dL as the concentration level at which effects begin to occur that warrant avoidance. For convenience, the probability that an observed blood lead value will exceed 10 µg/dL is referred to as P10. The EPA has established a health-based goal there should be no more than a 5% chance that a child will have a blood lead value above 10 µg/dL (USEPA 1994b,1998b). That is, if P10 is less than 5%, risks from lead are considered acceptable.

4.3 Overview of the IEUBK Model

The USEPA has developed a mathematical model for evaluating lead risks to residential children. This model is referred to as the Integrated Exposure Uptake Biokinetic (IEUBK) model. This model requires as input data on the levels of lead in all potentially contaminated environmental media (soil, dust, water, air, diet) at a specific location, and on the amount of these media taken in (by ingestion or inhalation) by a child living at that location. Given these inputs, the model calculates an estimate of the distribution of blood lead values that might occur in a population of children exposed to the specified conditions, including the value of P10.

4.4 Inputs to the IEUBK Model

For most IEUBK model exposure parameters, the default values recommended by current EPA guidance were used. Data from the site were used to estimate:

- The average concentration of lead in outdoor surface soil at each property
- The average concentration of lead in indoor dust at each property
- The average concentration of lead in air (site-wide)
- The average concentration of lead in drinking water (site-wide)
- The relative bioavailability (absorbability) of lead in soil and dust

These data were then used to calculate the value of P10 at each property inside the final focus area that has not already been remediated by USEPA.

4.5 Results

The results of the lead risk evaluation are presented in Table ES-1 and in Figure ES-4. Key points to note include the following:

- Of the 28,478 properties evaluated, a total of 19,445 homes (68%) are predicted to have P10 values at or below the health-based goal of 5%, and 9,033 properties (32%) have values that exceed the goal.
- Of these 9,033 properties, 3,177 have P10 values between 5% and 10%, 3,051 properties have P10 values between 10% and 20%, and 2,805 properties have P10 values greater than 20%.
- The location of properties with P10 values greater than the health-based goal of 5% were widespread across the OLS final focus area and were found within all zip codes with the exception of 68117 (which only had 2 properties).

These results indicate that a number of homes or parcels within the final focus area have soil lead levels that are of potential health concern to children who may reside there, now or in the future.

4.6 Uncertainties in the Evaluation of Risks from Lead

It is important to recognize that the exposure and risk calculations for lead presented in this section are subject to uncertainty that arises from a number of different sources. This includes:

- uncertainty in measured lead concentrations in soil
- uncertainty in estimated lead concentrations in indoor dust
- uncertainty in lead absorption from soil and dust
- uncertainty in the IEUBK model calculations and predictions

Because of these uncertainties, risk predictions for lead may not be highly accurate for every property, and risk managers and the public should take these uncertainties into account when interpreting the risk conclusions for lead at this site.

5.0 EVALUATION OF RISKS FROM NON-LEAD CONTAMINANTS

Although lead was the primary contaminant released to the environment from the historic operation of the smelters in the OLS, other metal and metalloid contaminants may also have been released. The purpose of this section is to provide an evaluation of the risks to area residents from these chemicals.

5.1 Exposure Assessment

The risk assessment focused on potential health risks to residents (both children and adults) from incidental ingestion of surface soil and indoor dust, as well as dermal contact with soil and indoor dust (when data permitted). Other pathways were judged to be sufficiently minor that quantitative evaluation was not warranted.

5.2 Quantification of Human Exposure

Exposure of residents (adults and children) to non-lead chemicals of potential concern in site soils and dusts was evaluated on a property-by-property basis. Exposure was calculated in accord with standard equations recommended by USEPA. In brief, the amount of chemical ingested or absorbed per day from each medium was calculated from information on the concentration of the chemical in the medium and the amount of medium that is ingested or contacted. Because there are usually differences between individuals in the level of exposure due to differences in intake rates, body weights, exposure frequencies, and exposure durations, calculations were performed for individuals that are "average" or are otherwise near the central portion of the range, and on intakes that are near the upper end of the range (e.g., the 95th percentile). These two exposure estimates are referred to as Central Tendency Exposure (CTE) and Reasonable Maximum Exposure (RME), respectively. Values of CTE and RME parameters for soil and dust were in accord with standard default values recommended by USEPA for evaluation of residents. Because only limited data were available on the concentration of non-lead COPCs in residential yards, each yard was evaluated using the highest detected concentration of each chemical. This approach is expected to tend to overestimate actual exposure levels.

5.3 Toxicity Assessment

A toxicity assessment is usually divided into two parts: the first characterizes and quantifies the non-cancer effects of the chemical, while the second addresses the cancer effects of the chemical. This two-part approach is employed because there are typically major differences in the time-course of action and the shape of the dose-response curve for cancer and non-cancer effects.

Non-cancer effects of an ingested chemical are usually described in terms of a Reference Dose (RfD). An RfD is an estimate (with uncertainty spanning perhaps an order of magnitude) of a daily exposure to the human population (including sensitive subgroups) that is likely to be without an appreciable risk of deleterious effects during a lifetime.

For cancer effects of ingested chemicals, toxicity is usually expressed in terms of the Slope Factor (SF), which has dimensions of risk of cancer per unit dose. At this site, arsenic is the only chemical of potential concern that is known to cause cancer effects when ingested.

Toxicity values (RfD and SF values) for each non-lead COPC evaluated in this assessment were selected from EPA's database or other sources in accordance with USEPA guidance. In the absence of site-specific data, the relative bioavailability (RBA) for all non-lead chemicals in all media was assumed to be 1.0. This is expected to be a conservative assumption, tending to overestimate the true degree of absorption and risk.

5.4 Risk Characterization

5.4.1 Basic Methods

Non-Cancer

The potential for non-cancer effects is evaluated by comparing the estimated daily intake of the chemical over a specific time period with the RfD for that chemical derived for a similar exposure period. This comparison results in a non-cancer Hazard Quotient (HQ), as follows:

$$HQ = DI / RfD$$

where:

HQ = Hazard Quotient
DI = Daily Intake (mg/kg-day)
RfD = Reference Dose (mg/kg-day)

If the HQ for a chemical is equal to or less than one (1E+00), it is believed that there is no appreciable risk that non-cancer health effects will occur. If an HQ exceeds 1E+00, there is some possibility that non-cancer effects may occur, although an HQ above 1E+00 does not indicate an effect will definitely occur. This is because of the margin of safety inherent in the derivation of all RfD values. However, the larger the HQ value, the more likely it is that an adverse effect may occur.

If an individual is exposed to more than one chemical, a screening-level estimate of the total non-cancer risk is derived simply by summing the HQ values for that individual. This total is referred to as the Hazard Index (HI). If the HI value is less than 1E+00, non-cancer risks are not expected from any chemical, alone or in combination with others. If the screening level HI exceeds 1E+00, it may be appropriate to perform a follow-on evaluation in which HQ values are added only if they affect the same target tissue or organ system (e.g., the liver). This is because chemicals which do not cause toxicity in the same tissues are not likely to cause additive effects.

Cancer

The excess risk of cancer from exposure to a chemical is described in terms of the probability that an exposed individual will develop cancer because of that exposure by age 70 which is considered a lifetime. For each chemical of concern, this value is calculated from the daily intake of the chemical from the site, averaged over a lifetime (DI_L), and the slope factor (SF) for the chemical, as follows:

$$\text{Excess Cancer Risk} = 1 - \exp(-DI_L * SF)$$

Excess cancer risks are summed across all chemicals of concern and all exposure pathways that contribute to exposure of an individual in a given population.

The level of total cancer risk that is of concern is a matter of personal, community, and regulatory judgment. In general, the USEPA considers excess cancer risks that are below about 1E-06 to be so small as to be negligible, and risks above 1E-04 to be sufficiently large that some sort of remediation is desirable. Excess cancer risks that range between 1E-04 and 1E-06 are generally considered to be acceptable, although this is evaluated on a case by case basis, and USEPA may determine that risks lower than 1E-04 are not sufficiently protective and warrant remedial action.

5.4.2 Results

Non-Cancer Risks

Table ES-2 shows the estimated non-cancer Hazard Quotients (HQs) for residential CTE (Panel A) and RME (Panel B) scenarios, including both children (age 0-6 years) and adults (age 7-30 years). As shown, risks from most COPCs in surface soil are below a level of potential concern (HQ ≤ 1) for both child and adult residents with CTE and RME scenarios. An exception is arsenic, which results in an HQ > 1 at about 11% of the properties. In addition, there are a small number of properties (< 1% of the total) where antimony, mercury and/or thallium yield HQ values above 1. Summation of non-cancer HQ values for chemicals that act on the same target tissue does not result in a substantial increase in non-cancer risk at most properties.

Cancer Risks

The only COPC at this site that is carcinogenic by the oral or dermal route is arsenic. Table ES-3 presents a summary of the estimated cancer risks to CTE residents and RME residents from arsenic in soil, including both the ingestion and dermal pathways. As seen, estimated cancer risks to CTE residents are within USEPA's target risk range (1E-06 to 1E-04) at all properties. Estimated risks to RME residents are also within USEPA's target risk range at most properties, although risks exceed 1E-04 at 141 locations (5% of the properties with data). The excess individual lifetime cancer risks at these 141 properties range from 1E-04 to 1E-03.

Summary of Risks from Non-Lead COPCs

Based on the evaluations described above, the frequency of properties with chemical-specific HQ values or organ-specific HI values above 1E+00 is very low except for arsenic. At some properties, the concentrations of arsenic are sufficient that, if the soil is ingested by child and/or adult residents over a long period of time, the risk of both cancer and adverse non-cancer effects would be higher than normally considered acceptable by USEPA.

5.5 Uncertainties

Quantitative evaluation of the risks to humans from environmental contamination is frequently limited by uncertainty regarding a number of key data items, including concentration levels in the environment, the true level of human contact with contaminated media, and the true dose-response curves for non-cancer and cancer effects in humans. This uncertainty is usually addressed by making assumptions or estimates for uncertain parameters based on whatever limited data are available. In general, the assumptions are intentionally conservative, such that risk calculations based on the assumptions are more likely to overestimate than underestimate actual human exposure and risk. It is important for risk managers and the public to keep this in mind when interpreting the results of a risk assessment.

Figure ES-1 Site Location
Omaha, Nebraska

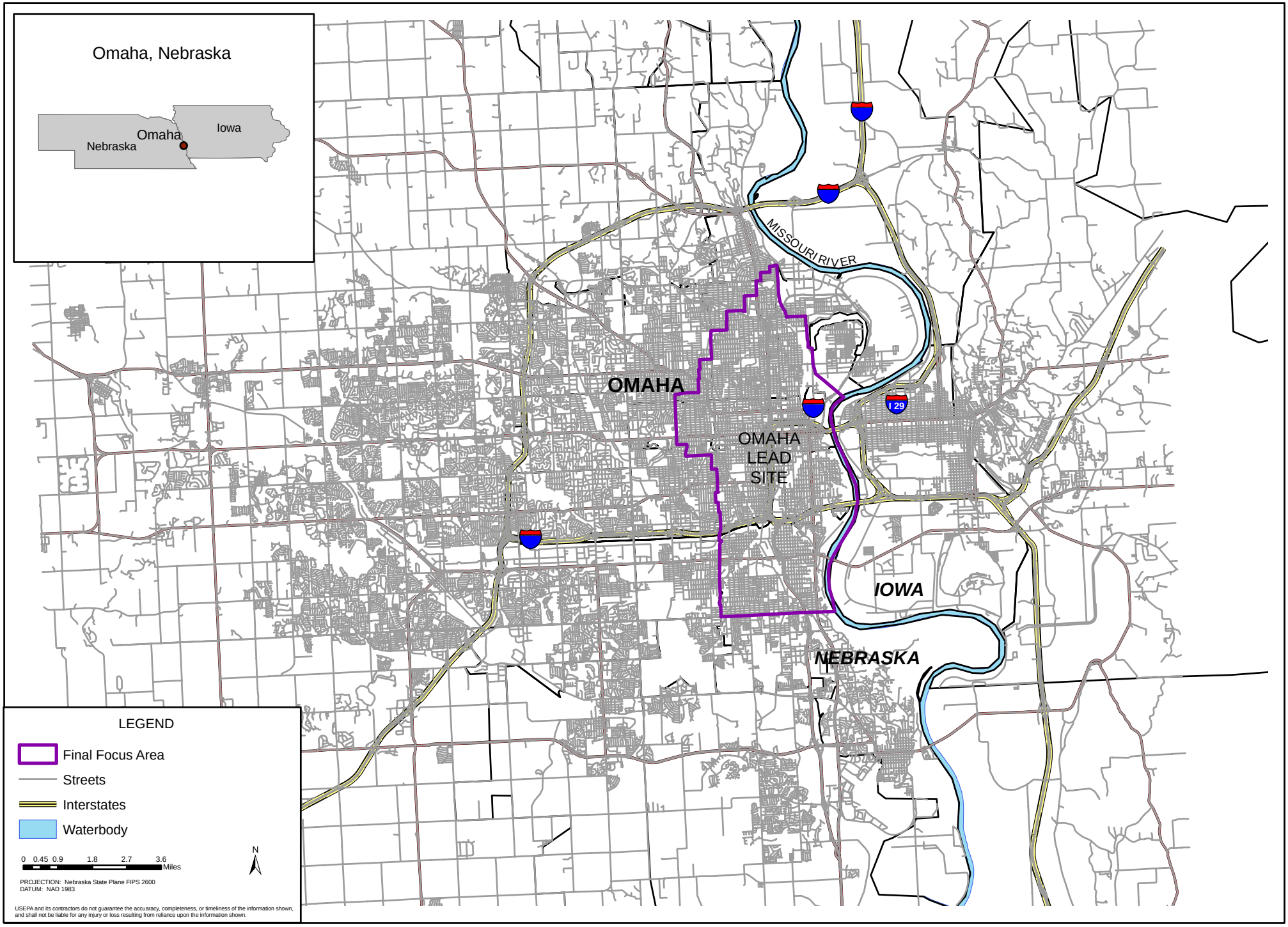


FIGURE ES-2 ELEVATED BLOOD LEAD LEVELS BY YEAR

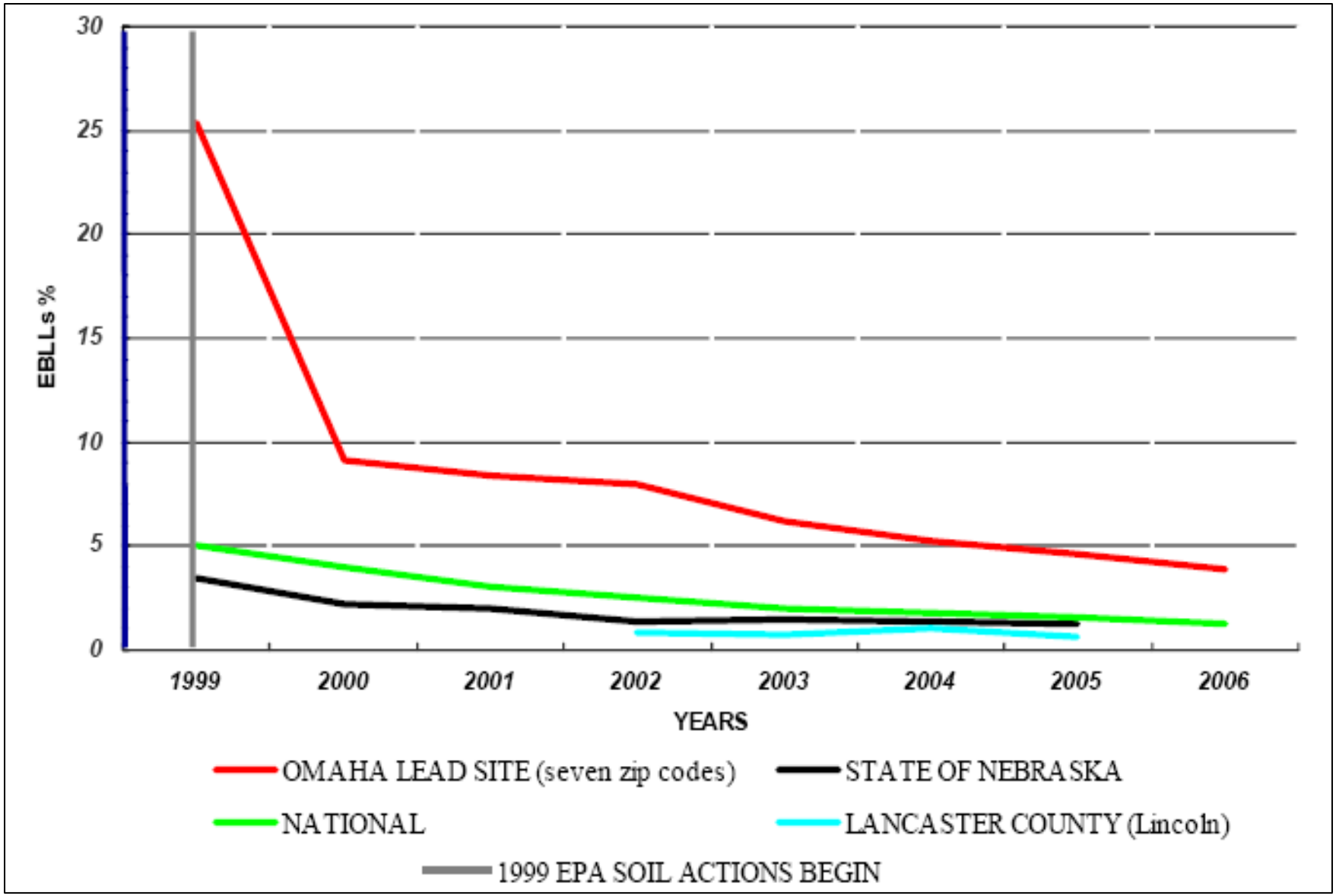


Figure ES-3. General Site Conceptual Model for Human Exposure at the Omaha Lead Site

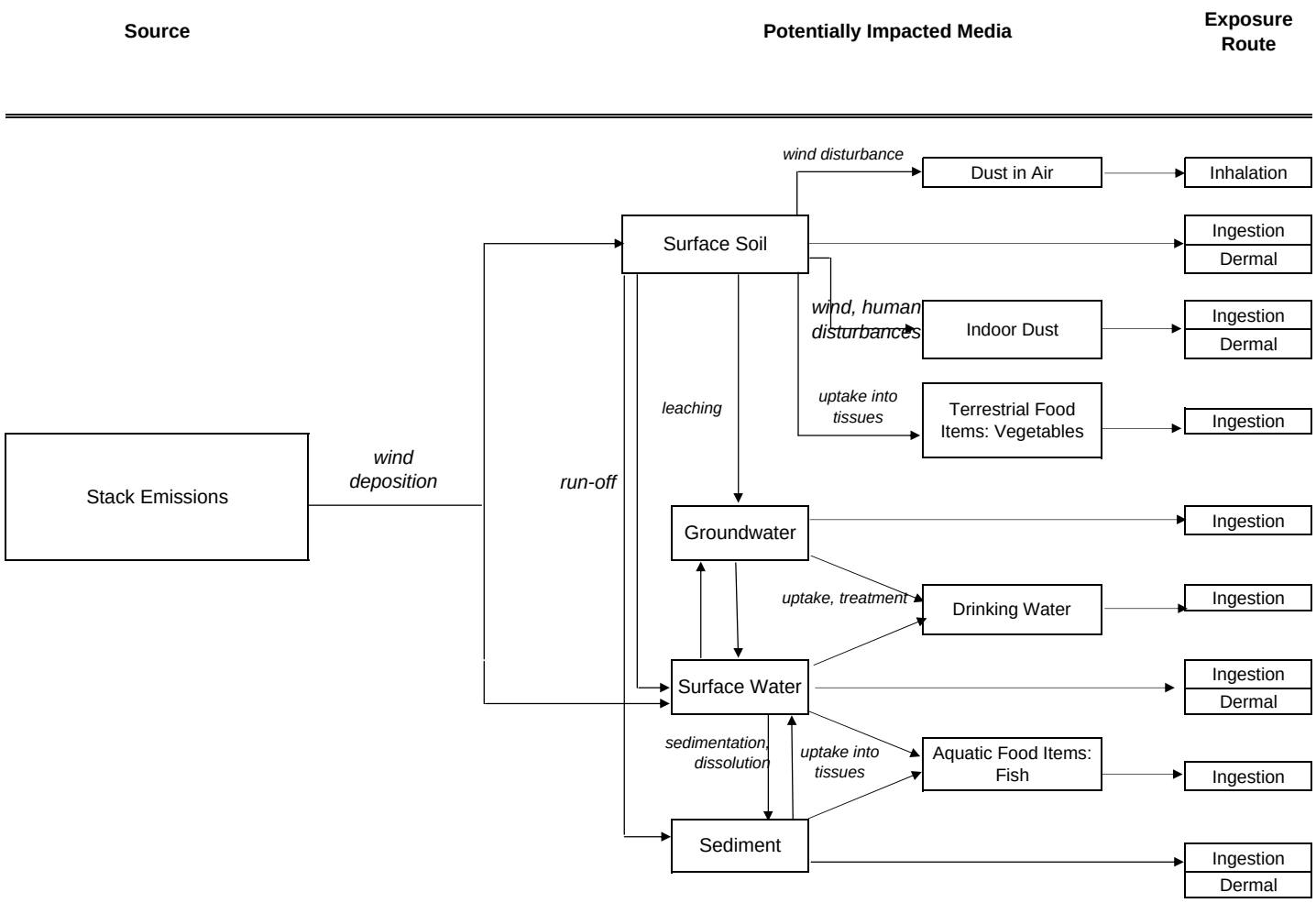
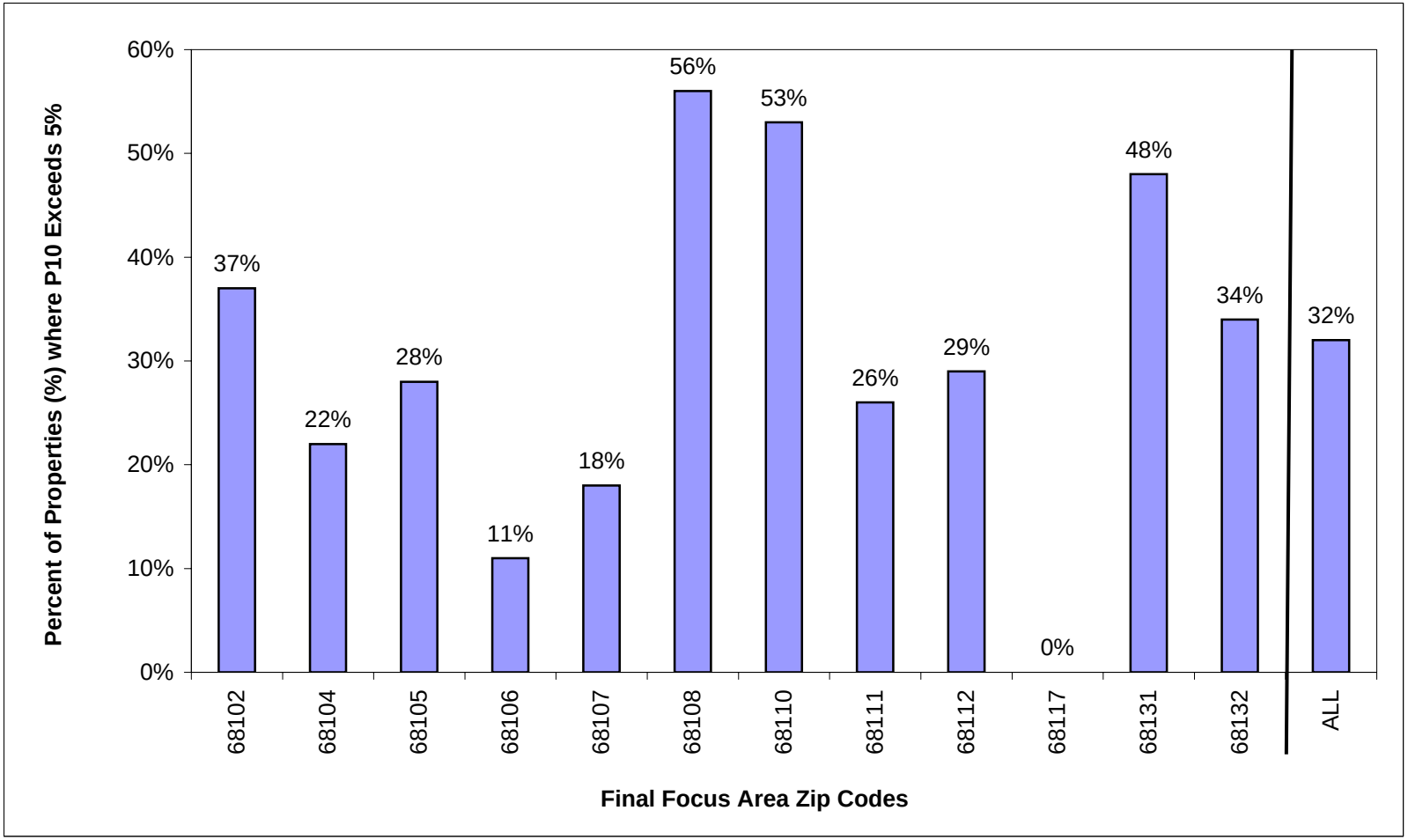


Figure ES-4. Percent of Properties in the Final Focus Area that Exceed EPA's Health Based Goal for Lead (P10>5%)



P10 = probability of having a blood lead level that exceeds 10 µg/dL

Table ES-1. Summary of Risks to Residents from Exposure to Lead in Surface Soil

ZIP CODE	NUMBER OF PROPERTIES EVALUATED	ESTIMATED NUMBER AND PERCENT OF PROPERTIES WITHIN THE SPECIFIED P10 RANGE					TOTAL NUMBER AND PERCENT OF PROPERTIES WHERE P10 EXCEEDS 5%
		≤ 5%	>5% to ≤10%	>10% to ≤20%	>20% to ≤50%	>50%	
68102	71	45	6	9	11	0	26
		63%	8%	13%	15%	0%	37%
68104	27	21	3	3	0	0	6
		78%	11%	11%	0%	0%	22%
68105	4,953	3,585	518	463	343	44	1,368
		72%	10%	9%	7%	1%	28%
68106	165	147	7	7	3	1	18
		89%	4%	4%	2%	1%	11%
68107	7,069	5,762	587	424	258	38	1,307
		82%	8%	6%	4%	1%	18%
68108	3,324	1,468	499	653	625	79	1,856
		44%	15%	20%	19%	2%	56%
68110	2,170	1,011	332	384	381	62	1,159
		47%	15%	18%	18%	3%	53%
68111	7,295	5,422	766	634	421	52	1,873
		74%	11%	9%	6%	1%	26%
68112	162	115	17	20	8	2	47
		71%	10%	12%	5%	1%	29%
68117	2	2	0	0	0	0	0
		100%	0%	0%	0%	0%	0%
68131	1,955	1,019	286	312	269	69	936
		52%	15%	16%	14%	4%	48%
68132	1,285	848	156	142	110	29	437
		66%	12%	11%	9%	2%	34%
ALL	28,478	19,445	3,177	3,051	2,429	376	9,033
		68%	11%	11%	9%	1%	32%

P10 = Probability of exceeding a blood lead value of 10 µg/dL (%).

Table ES-2. Estimated Total Non-Cancer Risks to Residents from Ingestion and Dermal Exposure from Surface Soil

Panel A. CTE Receptor

ANALYTE	TOTAL NUMBER OF PROPERTIES	NUMBER OF PROPERTIES WITHIN THE SPECIFIED RISK RANGE							
		Total Non-Cancer Hazard Index (HI)							
		Child				Adult			
		≤1	2 - 5	>5	Max	≤1	2 - 5	>5	Max
Aluminum	219	214	0	0	0.08	214	0	0	0.01
Antimony	2,843	2,843	0	0	0.9	2,843	0	0	0.09
Arsenic	3,046	2,970	71	5	8	3,046	0	0	0.8
Cadmium	3,046	3,046	0	0	0.06	3,046	0	0	0.01
Cobalt	214	214	0	0	0.2	214	0	0	0.02
Chromium III	3,046	3,046	0	0	0.001	3,046	0	0	0.0001
Chromium VI	3,046	3,046	0	0	0.08	3,046	0	0	0.01
Copper	214	214	0	0	0.1	214	0	0	0.02
Iron	214	214	0	0	0.3	214	0	0	0.03
Manganese	219	214	0	0	0.1	214	0	0	0.01
Mercury	2,832	2,830	1	1	6	2,832	0	0	0.6
Thallium	3,046	3,040	6	0	4	3,046	0	0	0.4
Vanadium	214	214	0	0	0.04	214	0	0	0.00
Zinc	27,737	27,737	0	0	0.1	27,737	0	0	0.01

Panel B. RME Receptor

ANALYTE	TOTAL NUMBER OF PROPERTIES	NUMBER OF PROPERTIES WITHIN THE SPECIFIED RISK RANGE							
		Total Non-Cancer Hazard Index (HI)							
		Child				Adult			
		≤1	2 - 5	>5	Max	≤1	2 - 5	>5	Max
Aluminum	219	214	0	0	0.2	214	0	0	0.03
Antimony	2,843	2,841	2	0	2.5	2,843	0	0	0.3
Arsenic	3,046	2,736	253	57	24	3,038	8	0	3
Cadmium	3,046	3,046	0	0	0.2	3,046	0	0	0.02
Cobalt	214	214	0	0	0.5	214	0	0	0.1
Chromium III	3,046	3,046	0	0	0.003	3,046	0	0	0.0003
Chromium VI	3,046	3,046	0	0	0.2	3,046	0	0	0.03
Copper	214	214	0	0	0.4	214	0	0	0.04
Iron	214	214	0	0	0.8	214	0	0	0.1
Manganese	219	214	0	0	0.3	214	0	0	0.04
Mercury	2,832	2,829	1	2	17	2,831	1	0	1.8
Thallium	3,046	3,029	16	1	11	3,046	0	0	1.1
Vanadium	214	214	0	0	0.1	214	0	0	0.01
Zinc	27,737	27,737	0	0	0.4	27,737	0	0	0.04

-- = Not applicable

Table ES-3. Summary of Total Cancer Risks to Residents (Children and Adults) from Ingestion and Dermal Exposure to Arsenic in Surface Soil

ZIP CODE	NUMBER OF PROPERTIES EVALUATED	ESTIMATED NUMBER AND PERCENT OF PROPERTIES WITHIN THE SPECIFIED RISK RANGE							
		Estimated Total Cancer Risk							
		CTE				RME			
		≤1E-06	>1E-06 to ≤1E-05	>1E-05 to ≤1E-04	>1E-04	≤1E-06	>1E-06 to ≤1E-05	>1E-05 to ≤1E-04	>1E-04
68102	6	1	5	0	0	0	1	5	0
		17%	83%	0%	0%	0%	17%	83%	0%
68104	7	1	5	1	0	0	1	5	1
		14%	71%	14%	0%	0%	14%	71%	14%
68105	466	49	373	44	0	1	51	372	42
		11%	80%	9%	0%	0%	11%	80%	9%
68106	33	6	22	5	0	0	6	23	4
		18%	67%	15%	0%	0%	18%	70%	12%
68107	800	94	674	32	0	0	105	664	31
		12%	84%	4%	0%	0%	13%	83%	4%
68108	276	15	257	4	0	0	15	257	4
		5%	93%	1%	0%	0%	5%	93%	1%
68110	259	15	241	3	0	0	16	240	3
		6%	93%	1%	0%	0%	6%	93%	1%
68111	800	69	697	34	0	0	79	690	31
		9%	87%	4%	0%	0%	10%	86%	4%
68112	6	0	5	1	0	0	0	5	1
		0%	83%	17%	0%	0%	0%	83%	17%
68131	172	19	145	8	0	0	24	140	8
		11%	84%	5%	0%	0%	14%	81%	5%
68132	221	15	188	18	0	1	20	184	16
		7%	85%	8%	0%	0%	9%	83%	7%
ALL	3046	284	2612	150	0	2	318	2585	141
		9%	86%	5%	0%	0%	10%	85%	5%

1.0 INTRODUCTION

1.1 Purpose

This document is the final baseline human heath risk assessment (BHHRA) for the Omaha Lead Site (OLS) located in Omaha, Nebraska (CERCLIS ID NESFN0703481). The purpose of this document is to assess the potential risks to humans, both now and in the future, from site-related contaminants present in environmental media, assuming that no steps are taken to remediate the environment or to reduce human contact with contaminated environmental media.

This final risk assessment is an extension and update of an interim risk assessment that was previously performed at the site (NHHS 2004). This final risk assessment incorporates additional data that were not available at the time of the interim assessment.

The results of this final assessment are intended to help inform risk managers and the public about potential human risks attributable to site-related contaminants and to help determine where there is a need for action at the site. The overall management goal is to ensure protection of humans from deleterious effects of exposures to site-related contaminants for both current and future land uses.

The methods used to evaluate risks in this assessment are consistent with current United States Environmental Protection Agency (USEPA) guidelines for human health risk assessment (USEPA 1989a, 1991a, 1991b, 1992a, 1997a, 2001a, 2002a, 2002b, 2004a) provided by the USEPA for use at Superfund sites.

1.2 Organization

In addition to this introduction, this report is organized into the following sections:

Section 2 This section provides a description of the site and a review of data that are available to characterize the nature and extent of environmental contamination at the site.

Section 3 This section provides the Site Conceptual Model that summarizes how people may be exposed to site-related contaminants in the environment, and identifies chemicals of potential concern (COPCs) in soil that warrant quantitative evaluation of exposure and risk.

Section 4 This section evaluates exposure and risk to residential children from lead in soil and dust.

Section 5 This section evaluates exposure and risk to area residents (adults and children) from non-lead chemicals of potential concern.

Section 6 This section provides full citations for USEPA guidance documents, site-related documents, and scientific publications referenced in the baseline risk assessment.

2.0 SITE CHARACTERIZATION

A detailed description of the site has been provided in the Remedial Investigation (RI) report for the site (Black and Veatch 2008c). A summary of information that is needed for the evaluation of human exposure and risk from lead and other site-related contaminants is provided below.

2.1 Site Location and Description

The OLS is located in Omaha, Nebraska. As shown in Figure 2-1, it encompasses the eastern portion of the greater metropolitan area, excluding Omaha's Central Business District (USEPA 2005a). It includes residential properties, child-care facilities, vacant lots, and other residential-type properties (schools, churches, parks) that have been contaminated as a result of historic air emissions from lead smelting/refining operations (see Figure 2-2).

2.2 Focus Area Boundary

Under Superfund, one important objective of the RI is to characterize the nature and extent of site-related contamination. In performing an RI, the most common approach is to establish an initial focus area to guide initial sampling efforts, and then to expand this focus area as needed based on the results that are obtained.

For the OLS, the original boundary of the focus area was established at the time the site was added to the National Priorities List (NPL) by establishing a perimeter surrounding properties that had been determined at the time to have a soil lead concentration that exceeded 1,200 milligrams per kilogram (mg/kg) (USEPA 2005a). This area encompassed approximately 13.8 square miles.

As RI sampling continued, the focus area was expanded to include a total of about 20 square miles, bounded by Ames Avenue to the north, L Street to the south, 45th Street to the west, and the Missouri River to the east (USEPA 2005a). In 2004, the focus area was further expanded to include a portion of the area bounded by Redick Street to the north, Harrison Street to the south and 52nd Street to the west. This expanded focus area is shown by the grey area in Figure 2-3.

Most recently, the focus area has been expanded again to include additional areas to the west and north. The final focus area is shown in Figure 2-3 by the purple line. The final focus area is based on a geospatial analysis of lead in surface soil that is described in Appendix A. In brief, the approach is based on evaluating the density (frequency) of properties within a neighborhood (a circle with a radius of 500 meters) that have a mean soil lead concentration above 400 mg/kg (the default level of concern and soil screening level for residential properties). The density is expected to be highest near former smelters, and to decrease as a function of distance away from the smelters. When the density decreases to 5% or less, the occurrence of properties above 400

mg/kg is expected to be infrequent, and the relative contribution of the smelter to the soil lead level is expected to be low. On this basis, the final focus area boundary is defined as a line that encloses properties located in neighborhoods where the density of properties with mean soil concentrations of 400 mg/kg or higher exceeds 5%.

Based on this approach, the final focus area encompasses all or part of the following 13 zip codes: 68102, 68104, 68105, 68106, 68107, 68108, 68110, 68111, 68112, 68117, 68131, 68132, 68198. It includes a total of 39,305 residential residential properties and vacant lots (see Table 2-1 for details).

2.3 Topography

The topography of the Omaha metropolitan area is characterized by a hilly upland in the western portion of the city and the Missouri River floodplain area to the east. The altitude at the OLS ranges from approximately 1,030 to 1,200 feet above mean sea level (Black and Veatch 2001).

2.4 Meteorology

Omaha has a climate that is continental in character, with contrasts in both temperature and precipitation. Seasons typically consist of severe winters, wet springs, and warm summers with moderate thunderstorm activity. The normal annual total precipitation in the area is approximately 28 to 30 inches, and the mean annual lake evaporation is 28 inches, resulting in a net precipitation of 0-2 inches (Black and Veatch 2008c). The prevailing winds are from the south/southeast and from the north/northwest, along the river valley (see Figure 2-4).

2.5 Surface Water

The Missouri River is the predominant surface water feature near the OLS, located immediately east of the site. It supports recreational fishing and boating. The OLS is located outside of the 500 year floodplain of the Missouri River (Black and Veatch 2008c).

Based on air transport modeling, the furthest north that site-related impacts to the Missouri River might be expected is about river mile 629. Surface water runoff from the OLS flows easterly towards the Missouri River. Direct surface water runoff from the Final Focus Area is expected to occur from Read Street (approximately river mile 624) to the Douglas County Line (approximately river mile 609).

2.6 Land Use

Current land use at the OLS consists of a mixture of residential, industrial, and commercial uses (Black and Veatch 2001). Future land use at the OLS is reasonably expected to remain approximately similar to current land use patterns, with a mixture of residential, industrial and

commercial properties. However, land use at individual properties may change over time. That is, some residential properties could be converted to non-residential use in the future, and current non-residential properties located within the OLS could be converted into residential properties in the future, particularly into multi-unit dwellings (USEPA 2005a).

2.7 Drinking Water

Residences located within the OLS focus area receive their drinking water from the municipal water supply (USEPA 2005a).

Sources of municipal water include wells and surface water intakes from the Missouri River. The Metropolitan Utilities District (MUD) provides drinking water to Omaha and Papillion, Nebraska, and Carter Lake, Iowa. MUD operates 52 municipal wells and one surface water intake. The 52 municipal wells are all located outside of a 4-mile radius of the OLS. The surface water intake is located north of the OLS at river mile 623. Council Bluffs Waterworks (CBW) provides drinking water to Council Bluffs, Crescent and Underwood, Iowa, and operates 2 municipal wells and one surface water intake. The municipal wells and surface water inlet are located approximately 2-3 miles east-northeast of the site, approximately 2,000 feet downstream of river mile 619 (Black and Veatch 2001).

2.8 Basis for Concern

Two lead processing facilities conducted operations within the boundary of the OLS. The locations of these two facilities are shown in Figure 2-5.

The American Smelting and Refining Company, Inc. (ASARCO) operated a lead refinery for a period of more than 120 years (early 1870s to 1997) at 500 Douglas Street. This facility processed lead bullion containing gold, silver, antimony, and bismuth. The ASARCO facility was closed in 1997. The property is currently owned by the City of Omaha and has been reutilized for both commercial and public purposes (Black and Veatch 2008c).

A secondary lead smelter and a lead battery recycling plant operated from the early 1950s until 1982 at 555 Farnam Street. Two different owners operated this facility: Aaron Ferer and Sons Co., and Gould Electronics, Inc. (Gould). Douglas County purchased the property in the early 1980s. The property has been remediated and developed into a county park (Black and Veatch 2008c).

Historical smelting, refining, and recycling activities at these two former smelters resulted in the release of lead and other smelter-related metals into air in the form of particulate matter. These particulates in the air were transported by wind and deposited onto soil in the vicinity of the site. Humans may be exposed to lead and other smelter-related metals in soil by a number of

pathways, and adverse health effects may occur if concentrations and exposure levels are high enough.

One measure of human exposure to lead is the concentration of lead in the blood, generally expressed in units of micrograms of lead per deciliter of blood (µg/dL). As discussed in greater detail in Section 3.1, concentration values above 10 µg/dL are generally considered to be elevated and of potential health concern. The Douglas County Health Department (DCHD) has been monitoring lead levels in the blood of children living in Douglas County for a number of years. Figure 2-6 shows the percentage of children with elevated blood leads (> 10 µg/dL) in the area of the site compared with State and national data. As shown, the frequency of elevated blood lead values in the area of the site has been consistently higher than elsewhere (USEPA 2005a). This indicates that children living in the area of the site are being exposed to lead in the environment at levels that exceed other locations, and which may be of potential health concern to some children.

2.9 Potential Sources

The USEPA conducted an apportionment investigation to determine the sources of lead contamination found in the soil of residential properties at the site. Soil samples were collected from the former ASARCO refinery property, the former Gould property and residential properties. These soil samples were examined using an electron microprobe to characterize the chemical form of the lead that was present and to compare the types of lead found in soil samples at the smelter areas to that found in residential properties. More than 90 percent of the community soils studied contained lead in a form that was characterized as pyrometallurgical (i.e., smelter-related) (Black and Veatch 2008c). Based on this, the study concluded that the ASARCO and Gould facilities are significant sources of the lead at the OLS (USEPA 2005a).

2.10 Response Actions to Date

Time-Critical Removal Action

USEPA initiated two time-critical removal actions at the OLS site (USEPA 1999, 2001b, 2002c, 2003a) to eliminate or reduce ingestion exposure to lead in soil. In 2004, the two removal actions were combined into a single response action that encompassed residential properties with soil lead concentrations equal to or greater than 1,200 mg/kg and properties (childcare facilities or residential properties) where children with elevated blood lead levels reside with soil lead concentrations greater than 400 mg/kg (Black and Veatch 2008c and USEPA 2005a).

Remedial Action

An Interim Record of Decision (ROD) was signed in 2004 (USEPA 2005a) that included the following remedial actions to address site risks:

- Excavation and removal of lead-contaminated soils at properties with the greatest human health risk. These generally include:
  - A residential property where at least one non-foundation sample exceeds 800 mg/kg for lead.
  - Residences where a child with an elevated blood lead level (>10 µg/dL) resides and with any non-foundation sample exceeding 400 mg/kg lead.
  - Child-care facilities and other high-child impact areas with any non-foundation sample exceeding 400 mg/kg lead.
- Stabilization of loose and flaking exterior lead-based paint, in cases where it threatens the protectiveness of the soil remedy implemented at a property.
- High-efficiency interior cleaning at properties where soil cleanup actions are conducted and lead in interior dust exceeds Housing and Urban Development (HUD) standards.
- Participation in a comprehensive program with other public and private entities to characterize and address all potential lead sources.
- Health education for the Omaha community and medical professionals to support public awareness, exposure prevention programs, in-home assessments, blood-lead screening programs and diagnosis, treatment, and surveillance programs.

As of September 2008, a total of 36,000 properties (residential, commercial, other, etc.) have been investigated, and USEPA has remediated 4,100 properties.

2.11 Site Investigations

Several investigations have collected data at the OLS. The available data are summarized in Tables 2-2 through 2-3 and are briefly described below.

2.11.1 Surface Soil

Table 2-2 presents a summary of the three soil investigations conducted at or in the vicinity of the OLS. Each investigation is summarized in the following sections. Electronic data are provided in Appendix B and summary statistics are provided in Tables 2-4 and 2-5.

Preliminary Assessment/Site Inspection

A Preliminary Assessment/Site Investigation (PA/SI) was conducted from 1998 to April 2001 to support removal action and remedial action at the Omaha Lead Site. Soil samples were collected from childcare facilities and from properties where children with elevated blood lead levels resided in order to evaluate the properties for potential removal action. Soil samples were also collected from other residential properties to identify the extent of lead contamination from the industrial emission sources in downtown Omaha. These samples were collected along four

sampling corridors that extended from downtown Omaha to the north, south, east and west. Along a sampling corridor, surface soil samples were collected from approximately eight properties, every tenth of a mile until the soil lead concentrations were consistently found to be below 400 mg/kg. Additionally, grab samples of subsurface soil were collected at every other property within the sampling corridor. Sample depths were 0-8 inches below ground surface (bgs), 8-16 inches bgs, and 16-24 inches bgs (Black and Veatch 2001).

At all properties, a minimum of five surface soil samples were collected: four from the yard and one from the drip zone (within 3 feet of the house foundation). Surface soil samples consisted of five-point composite samples of the top one inch of soil (0-1" bgs). Yard sample locations were generally determined by dividing the property into four quadrants of approximately equal areas and collecting one composite sample from each quadrant (Black and Veatch 2001). Surface and subsurface soil samples were homogenized and sieved with a number 10 (2 mm) mesh sieve prior to analysis. All samples were analyzed using X-Ray Fluorescence (XRF) and approximately 10% of all field samples were analyzed for the 23 target analyte list (TAL) Metals using Inductively Coupled Plasma Atomic Emission Spectroscopy (ICP-AES) and Cold Vapor Atomic Absorption (CVAA) for mercury (Black and Veatch 2001).

Background Summary Report

In November of 1999, a total of thirty soil samples were collected approximately 8 miles north of the former ASARCO facility to determine background concentrations of metals. The samples consisted of 23 surface soil (0-1" bgs) samples and 7 subsurface soil samples (0-8", 8-16" and 16-24" bgs). Surface soil samples consisted of 5-point composite samples, whereas subsurface samples were collected as grab samples. Soil samples were homogenized and sieved through a number 10 mesh sieve (2 mm). The samples were analyzed by XRF for 15 of the 23 TAL Metals (arsenic, lead, zinc, silver, barium, cadmium, cobalt, chromium, copper, iron, mercury, nickel, manganese, antimony, and selenium) as well as 7 additional analytes (molybdenum, rubidium, tin, strontium, titanium, uranium, and zirconium). Twenty-nine of the soil samples were also analyzed for 23 metals by ICP-AES (the 23 TAL Metals (except for mercury) and bismuth) (Jacob 2000).

Remedial Investigation/Feasibility Study

From May 2001 to May 2008, USEPA collected surface soil samples to support site characterization, remedial investigation and risk assessment activities. At residential properties, the yard was divided into 4 quadrants of approximately equal areas located around the home (at most properties this consisted of 2 quadrants in the front yard and 2 quadrants in the backyard). A 5-point composite surface soil (0-1 inches in depth) sample was collected from each quadrant. Surface soil samples were also collected from play areas, gardens, and the drip zone around each residence (the area within 6-30 inches of the foundation of a residence) in the same manner. Surface soil samples were homogenized and sieved prior to analysis. Surface soil samples

collected from March 1999 to May 2000 were sieved through a number 60 mesh sieve (250 µm), whereas surface soil samples collected thereafter were sieved through a number 10 mesh sieve (2 mm) (Black and Veatch 2008c).

All surface soil samples were analyzed by XRF and approximately 5% of all field samples were analyzed for the 23 TAL Metals using ICP-AES and CVAA for mercury (Black and Veatch 2008c).

2.11.2 Paired Indoor Dust-Outdoor Soil Samples

Two investigations have collected paired outdoor soil-indoor dust samples at the site. Each data set is briefly described below and summarized in Table 2-3. Electronic data are provided in Appendix B and summary statistics are presented in Table 2-6.

2003-2004 Remedial Investigation

During 2003-2004, indoor dust samples were collected from 159 homes, which had not previously been investigated for soil, to collect data to support the draft risk assessment.

Soil

At each property, surface soil (0-1” bgs) samples were collected as described above, with a five-point sample collected from each yard quadrant. All soil samples were sieved using a number 10 mesh screen (2 mm) and analyzed by XRF. A total of 159 soil samples were submitted for analysis by ICP-AES.

Indoor Dust

A total of 636 indoor dust samples were also collected at the 159 properties where soil samples were collected, including 3 dust samples and a single wipe sample. Dust samples were collected at each home from the following locations: (1) entryway, (2) windowsill and (3) floor. Samples were collected in accordance with the Field Sampling Plan (Black and Veatch 2008c) using a High Volume Small Surface Sampler (HVS3) vacuum. Dust samples were sieved using a number 100 sieve (150 µm) and analyzed for lead (only) using ICP.

2007 Supplemental Data Collection Activities

In the fall of 2007, paired outdoor soil-indoor dust samples were collected from 98 properties in accordance with the Field Sampling Plan (Black and Veatch 2007) to address some of the limitations identified in the existing outdoor soil-indoor dust dataset (see Black and Veatch 2007 for details).

The properties for sampling were selected based on the concentration of lead in yard soil measured in previous investigations. An effort was made to sample properties at which children currently reside. Samples from an approximately equal number of properties were collected from the following soil lead concentration bins:

CONCENTRATION BIN NUMBER	SOIL-LEAD CONCENTRATION (mg/kg)	NUMBER OF PROPERTIES SAMPLED
1	< 250	20
2	>250 - 500	20
3	>500 - 750	21
4	>750 – 1,000	15
5	>1,000 – 1,250	9
6	>1,250	13

Collection of an approximately equal number of samples from each concentration bin increases the confidence of the best fit line through the data that defines the outdoor soil to indoor dust relationship.

The details of the 2007 surface soil and indoor dust sample collection procedures are discussed in the following sections.

Surface Soil

One 20-point composite surface soil (0-1 inches bgs) sample was collected from each property. The composite sample was collected by dividing up the property into 4 quadrants and collecting five grab samples from each quadrant (e.g., from the four corners away from drip zones and the center of each quadrant). The composite soil sample was sieved in the field, using a number 60 mesh screen to isolate the soil particles in the fine fraction (< 250 µm). The fine fraction of the soil samples were submitted for analysis by the USEPA Region 7 laboratory for the 23 TAL Metals using SW846 6010 and SW 846 7471 (mercury). Three additional analytes were added to the analytical suite: silicon, titanium and zirconium. These chemicals are thought to be good indicator chemicals for deriving an outdoor soil to indoor dust relationship because they are present in both media and are not thought to have significant alternative indoor sources.

Indoor Dust

One 3-point composite indoor dust sample was also collected from each property consisting of indoor dust collected from 3 locations in the home: (1) entryway; (2) living area; and (3) bedroom. Samples were collected in accordance with the Field Sampling Plan (Black and Veatch 2007) using a HVS3 vacuum. The composite dust samples were sieved in the field using a number 60 mesh screen to isolate the dust particles in the fine fraction (250 µm). The fine fraction dust samples were submitted to the EPA Region 7 laboratory for analysis of the 23 TAL

Metals and silicon, titanium, and zirconium by Method SW846 6010 and SW846 7471 (mercury).

2.11.3 Tap Water

In December 2007, a total of 154 tap water samples were collected from 77 residences to support the final human health risk assessment. A total of two tap water samples were collected at each home: a "first-flush" sample and a "post-flush" sample. The first-flush sample represents a water sample that is collected first thing in the morning, before any household use of water (e.g., running the shower, flushing the toilet, running the sink). The post-flush sample represents a water sample that is collected after allowing the tap to run for approximately 5 minutes. Both samples were collected from the kitchen sink in accordance with the Field Sampling Plan (Black and Veatch 2007) using 1 liter, high density polyethylene sample bottles. Both the first-flush and post-flush samples were collected by residents. Field personnel added 1 milliliter of nitric acid to the sample containers to preserve the samples. Water samples were analyzed for lead by the USEPA Region 7 laboratory in Kansas City, KS, using EPA Method 200.8. These data are provided electronically in Appendix B and are summarized in Table 2-7.

2.11.4 Ambient Air

Concentrations of lead in ambient air were obtained from USEPA's Air Quality System (AQS) database (USEPA 2008b). This database contains ambient air monitoring results for contaminants that are considered hazardous air pollutants or criteria air pollutants by the Clean Air Act, including lead. The database contains annual summaries of the air pollution measurements for the current year and ten previous years. As shown in Figure 2-7, 11 of the 14 monitoring stations in Douglas County are located within the Omaha Lead Site. Table 2-8 presents details on each station, including the time period(s) when lead measurements were collected at each station and Table 2-9 presents the mean concentration of lead measured from 1997 (the year ASARCO ceased operations) to 2002 (the most current data available).

2.12 Data Useability Assessment

In accord with guidelines for assessing data quality (USEPA 1989a, 1992b), the available data were reviewed to determine if they were appropriate for use in the risk assessment. The following sections describe the data useability findings.

2.12.1 Review Against Data Useability Criteria

The soil, dust, and tap water data were reviewed in accord with USEPA's (1992b) Guidance for Data Useability in Risk Assessment. Detailed data useability worksheets are presented in Appendix H. The results of the review are briefly presented below:

- Data Sources: the sampling and analytical methods used in the PA/SI and RI are comparable and appropriate to combine across data sets, with the exception of the indoor dust. For indoor dust, different sieve sizes were used to prepare samples collected in 2003-2004 (Number 100 mesh screen, 150 µm) and 2007 (Number 50 mesh screen (250 µm). Thus, these datasets should not be combined together for use in the risk assessment.
- Documentation: field documentation was appropriate, no deviations from the sampling and analysis plan (SAP) were reported.
- Analytical Methods and Detection Limits: the analytical methods used to quantify metals in soil, dust and tap water are appropriate for use in risk assessment. The detection limits were found to be adequate for all chemicals, in all media, with the exception of thallium in soil.
- Data Quality Indicators: Data quality indicators (completeness, comparability, representativeness, precision and accuracy) were met for soil, dust, and tap water.
- Data Review: Laboratory data were reviewed by the USEPA Region 7 Laboratory. XRF data were reviewed by the field contractor. Data were qualified accordingly.
- Reports to Risk Assessor: All USEPA (1992b) recommended data and documentation were available for use in the risk assessment.

Based on this evaluation, USEPA's data useability criteria have been satisfied for surface soil, indoor dust and tap water data collected at the site.

2.12.2 XRF vs ICP

As described above, the concentration of lead and other metals in soil were analyzed by two different methods: XRF and ICP. In some cases, XRF data may be less accurate than ICP data. Thus, whenever ICP data were available at a sampling location, these data were preferred over XRF data from the same station, and only the ICP data were included in the risk assessment. If only XRF data were available for a sampling location, then the XRF results were included if they were determined to be adequate for use in risk assessment.

The adequacy of XRF data were determined by conducting a Data Quality Assessment (DQA) of XRF data sets. This assessment is presented in Appendix C. In brief, the first step was to evaluate the XRF detection limit of the analyte in soil. If the XRF detection frequency was high (>80%), the detection limit was considered adequate. If the detection frequency was < 80%, the XRF detection limit was compared to the level needed for risk assessment purposes. If the detection limit was below the level needed for risk assessment purposes, the detection limit was considered adequate. If the detection limit frequency was adequate, then the strength of the correlation between paired XRF and ICP results was evaluated. If the strength of the correlation was good ($R^2 > 0.7$), the data were considered to be adequate for use in risk assessment. Otherwise, the XRF data were not considered to be adequate. Based on the DQA in Appendix C, only the XRF data for lead and zinc were determined to be adequate for use in the risk assessment. XRF data for other metals were not used.

For lead and zinc, all XRF data were adjusted to estimate the ICP-equivalent concentrations, using the chemical-specific parameters from the ICP/XRF regressions (see Appendix C for details):

$$[\text{ICP-equivalent concentration}] = a_i + b_i \bullet [\text{XRF concentration}]$$

where:

a = intercept from the ICP/XRF regression line for chemical "i"
b = slope from the ICP/XRF regression line for chemical "i"

2.12.3 Current *vs* Historic Site Conditions

As mentioned above, USEPA is continuing to perform remediation of yard soil at residences at the OLS that exceed the interim trigger for action. Soil samples collected from properties that have already been remediated by USEPA were excluded from this risk assessment, since they do not reflect current conditions. A total of 3,293 of the 4,100 properties that have been remediated were located within the final focus area. These 3,293 properties were excluded from evaluation in this risk assessment.

While excluding remediated properties from this evaluation may underestimate the original, overall risks to residents from site-related contaminants at the site, it is necessary to exclude these properties in order to provide decision-makers with information on potential risks to residents under current (remediated) site conditions. This information will be used to help determine if there is a need for additional action at properties that have not yet been remediated in order to protect public health.

2.12.4 Data Qualifiers

Data qualifiers assigned by the laboratory were reviewed in order to determine if individual measurements were reliable and appropriate for use. All data points were considered appropriate for use in the risk assessment except for soil samples that were "R" qualified (rejected for not meeting quality control requirements), "0" (no results reported) and "N/A-I" (no results reported) by the laboratory.

3.0 SITE CONCEPTUAL MODEL

3.1 Scope of the Baseline Risk Assessment

This risk assessment evaluates the potential risks to residents at current residential properties (including child-care facilities) and at properties that could be converted to residential land use in the future (vacant lots). Commercial properties, school, parks, and churches were not evaluated by this risk assessment.

3.2 Exposure Media and Pathways of Potential Concern

Figure 3-1 presents a conceptual model of how smelter-related contaminants that have been released to the environment at the OLS might result in exposure of residents, including both children and adults. The environmental medium of chief concern is surface soil that has been impacted by the wet or dry deposition of metal-containing airborne particulates released from the smelters. Residents might be exposed to smelter-related contaminants in site soils by a number of different pathways, including ingestion, inhalation, and dermal contact with contaminated soil or dust, and ingestion of home-grown produce that may have taken up contaminants from the soil. However, not all of these are likely to be of equal concern for all contaminants. Section 4.1 describes the exposure pathways that are of chief concern for lead, and Section 5.1 describes the exposure pathways that are of chief concern for other (non-lead) contaminants.

3.3 Selection of Chemicals of Potential Concern

Chemicals of Potential Concern (COPCs) are chemicals which exist in the environment at concentration levels that might be of potential health concern to humans and which are or might be derived, at least in part, from site-related sources.

The chief COPC at this site is lead. The procedure used to identify other (non-lead) COPCs for the evaluation of risks to residents is shown in Figure 3-2. Chemicals that are not likely to contribute significant risks to humans are eliminated, while chemicals that might be of potential concern are retained for either quantitative or qualitative evaluation, depending on the availability of data. It is important to note that this COPC selection procedure is intended to be conservative; that is, it is expected that some chemicals will be identified as COPCs that are actually of little or no concern, but that no chemicals of authentic concern will be overlooked.

The steps in the COPC selection process are described below.

Step 1: Eliminate chemicals for which no toxicity values are available

If a chemical does not have a toxicity value, this is identified as a source of uncertainty unless the chemical is a beneficial nutrient (e.g., calcium, magnesium, potassium and sodium).

Step 2: Eliminate chemicals never detected

In accord with USEPA (1989a), a chemical may be eliminated from the quantitative risk assessment if it is detected only infrequently in a site medium. In this risk assessment, chemicals were excluded only if they were never detected in any site sample. However, if the analytical detection limit for a chemical that was never detected was sufficiently high that the chemical would not have been detected even if it were present at a level of concern, that chemical was identified as a source of uncertainty.

Step 3: Eliminate chemicals detected, but whose maximum value is below a level of concern

If a chemical is detected at least once, but the maximum detected concentration is below a level of health concern, that chemical may be eliminated from further consideration. The level of health concern used in this evaluation is based on Regional Screening Level values for residential exposure to soil that have been calculated by USEPA (USEPA 2008c). The target risk levels were set to a non-cancer hazard quotient (HQ) value of 0.1 and a cancer risk level of 1E-06.

Appendix E presents detailed results of the COPC selection process. Table 3-1 lists the COPCs identified for quantitative evaluation.

4.0 EVALUATION OF RISKS FROM LEAD

Lead is the contaminant of primary human health concern at the OLS. This section describes the toxicity of lead and the approach that the USEPA uses to evaluate risks to humans from exposure to lead, and applies that method to characterize the level of risk that would exist if no further action is taken at the site to reduce human exposure to lead at the site.

4.1 Exposure Assessment

4.1.1 Population of Chief Concern

For residential land use, young children (age 0-84 months) are the sub-population of chief concern for exposure to lead. This is because young children (1) tend to have higher intakes (expressed in terms of mg/kg-day) of lead-contaminated environmental media such as soil, dust, and paint, (2) tend to have a higher absorption fraction for ingested lead, and (3) are more sensitive to the toxic effects of lead than are older children or adults. If environmental exposures to lead in a residential area are acceptable for young children, exposures are usually also acceptable for older children and adults, including pregnant women.

4.1.2 Exposure Pathways of Chief Concern

The generalized conceptual site model (CSM) for exposure of residents to smelter-related contaminants was presented above in Figure 3-1. The detailed CSM for exposure of residents to lead is presented in Figure 4-1. The following sections describe which of these pathways are likely to be potentially significant for residents (children) and that have been included in the quantitative evaluation of the total exposure to lead in the risk assessment.

Young children can be exposed to lead from a variety of sources. As shown Figure 4-1, at this site the environmental medium of chief concern is outdoor soil that became contaminated with lead released from historic smelter operations. Although few children intentionally ingest large amounts of soil, many children ingest small quantities as a consequence of normal hand-to-mouth activities during play outside. In addition, soil may be tracked into indoor living spaces where it becomes mixed into indoor dust, which may also be ingested by indoor hand-to-mouth activities.

Several other pathways also contribute to the total exposure to smelter-related lead. Release of soil particles into air can occur, and children can be exposed by breathing the air. Although this pathway is usually small compared to the ingestion pathways, it is evaluated quantitatively. Dermal contact with soil or dust may also occur, but dermal absorption of lead is generally considered to be a minor pathway compared to ingestion, and is not usually evaluated quantitatively. Similarly, ingestion of lead in home-grown garden vegetables may occur, but

uptake of lead from soil into plants is usually not extensive, and ingestion from this source is not evaluated quantitatively in this risk assessment.

In addition to these exposures to smelter-related releases of lead, children may also be exposed to lead from other sources as well. One of these is lead in indoor or outdoor paint. The main concern is for direct ingestion of leaded paint. This occurs mainly when the paint is in poor condition (i.e., peeling and chipping), and a child can directly ingest flakes or chips of paint. This exposure pathway is not evaluated in this assessment because data on the amount of paint ingested from paint chip ingestion is not available. Leaded paint may also contribute to total lead levels in soil and dust. To the extent that this is occurring at this site, this pathway is evaluated by using measured or calculated levels of lead in soil and dust. Lead is also present in water used for drinking, and in food purchased from stores. Because risk from lead depends on exposure from all of these sources, these exposure pathways are also included in the risk evaluation for lead.

4.2 Toxicity Assessment

Excess exposure to lead can result in a wide variety of adverse effects in children. The following sections summarize the most characteristic and significant of the adverse effects of lead on children, and current guidelines for classifying exposures as acceptable or unacceptable.

Neurological Effects

The effect of lead that is usually considered to be of greatest concern in children is impairment of the nervous system. Many studies have shown that animals and humans are most sensitive to the effects of lead during the time of nervous system development, and because of this, the fetus, infants and young children (0-6 years of age) are particularly vulnerable. The effects of chronic low-level exposure on the nervous system are subtle, and normally cannot be detected in individuals, but only in studies of groups of children. Common measurement endpoints include various types of tests of intelligence, attention span, hand-eye coordination, etc. The overall weight of the available evidence provides clear evidence of neurocognitive effects in young children occurring at blood lead levels in the range of 5-10 µg/dL, and possibly lower (USEPA 2006). Such effects on the nervous system are long-lasting and may be permanent.

Effects on Heme Synthesis

A characteristic effect of chronic high lead exposure is anemia stemming from lead-induced inhibition of heme synthesis and a decrease in red blood cell life span. Lead interferes with heme synthesis by inhibiting the enzymes ALA-D and ferrochelatase (USEPA 2007a). Decreases in ALA-D activity can be detected at blood lead levels below 10 µg/dL in children and adults (ACGIH 1995, USEPA 2006). At concentrations of 20-30 µg/dL, erythrocyte ALA-D activity is halved and ferrochelatase is inhibited (USEPA 2006). Heme synthesis is inhibited not

only in red blood cells but in other tissues. Several key enzymes that contain heme, including those needed to form vitamin D, also show decreased activity following lead exposure (USEPA 1986). The Centers for Disease Control (CDC) (1991) reviewed studies on the synthesis of an active metabolite of vitamin D and found that impairment was detectable at blood lead levels of 10 - 15 µg/dL.

Cardiovascular Effects

Both human and animal studies support the conclusion that lead exposure increases the risk of cardiovascular effects, including increased blood pressure, increased incidence of hypertension, cardiovascular morbidity and mortality (USEPA 2006).

Renal Effects

In adults, reduced renal function (measured by decreased renal clearance of creatinine) has been observed in adults at blood lead levels below 10 µg/dL. Low levels of lead can also act as a co-factor with other risk factors to increase the risk for renal dysfunction, increasing the susceptibility of sensitive populations such as those with diabetes, hypertension and chronic renal insufficiency for adverse renal effects (USPEA 2006). Individuals with chronic, high level exposure to lead (40-100 µg/dL), such as occupationally exposed adults, have an increased risk for lead nephropathy which can result in small, fibrotic kidneys and ultimately kidney failure (USEPA 1986, 2006).

The renal effects of lead exposure in children are difficult to access, as most studies only measured early biological effect markers which have unknown clinical significance (USEPA 2006).

Immune System Effects

The main immune system targets of lead are macrophages and T lymphocytes, leading to a potential for increased tissue inflammation, reduced cell-mediated immunity, and increased risk of autoimmunity (USEPA 2006 and 2007a). Additionally, studies of lead exposure in both animals and humans have found an association between increased production of IgE (an immunoglobulin involved in allergic responses and asthma) and blood lead concentrations. In children, significant associations between serum IgE levels and increasing blood lead levels were observed at blood lead levels of less than 10 µg/dL (USEPA 2006 and 2007a). In animals, neonatal exposure to lead resulting in blood lead levels less than 10 µg/dL was found to produce persistent later-life immunotoxicity (USEPA 2006).

Cancer Effects

Studies in animals indicate that chronic oral exposure to very high doses of lead salts may cause an increased frequency of tumors of the kidney (USEPA 1989b, ACGIH 1995). In humans, evidence of lead carcinogenicity is limited (USEPA 2006). Based on the available data, lead has been classified as a probable human carcinogen by the International Agency for Research on Cancer (IARC) (IARC 2005), and is reasonably anticipated to be a human carcinogen by the National Toxicology Program (NTP) (NTP 2003, 2004). USEPA classifies lead as a probable human carcinogen (USEPA 2008d). Nevertheless, the non-carcinogenic effects on the nervous system are usually considered to be the most important and sensitive endpoints of lead toxicity (USEPA 1986, 2006, 2007a).

Current Guidelines for Protecting Children from Lead

It is currently difficult to identify what degree of lead exposure, if any, can be considered safe for infants and children. As discussed above, some studies report lead-induced effects in children and adults beginning at around 10 µg/dL or even lower. Of special concern are the claims by some researchers that effects of lead on neurobehavioral performance, heme synthesis, the immune system, and fetal development may not have a threshold value, and that the effects are long-lasting (USEPA 1986, 2006). On the other hand, some researchers and clinicians believe the effects that occur in children at low blood lead levels are so minor that they need not be cause for concern.

After a thorough review of all the data, the USEPA currently identifies 10 µg/dL as the concentration level at which effects begin to occur that warrant avoidance. Likewise, the CDC has established a guideline of 10 µg/dL in preschool children which is believed to prevent or minimize lead-associated cognitive deficits (CDC 1991, 2005). For convenience, the probability that an observed blood lead value will exceed 10 µg/dL is referred to as P10. The USEPA has established a health-based goal there should be no more than a 5% chance that a child will have a blood lead value above 10 µg/dL (USEPA 1994b,1998a). That is, if P10 is ≤ 5%, risks from lead are considered acceptable.

4.3 Overview of the IEUBK Model

The USEPA has developed a standard approach for evaluating the risks of lead exposure in residential children (USEPA 1994a, 2001c). This approach employs a set of mathematical equations and calculations to predict the likely distribution of blood lead values in a population of children exposed to a specified set of environmental lead levels. This model is referred to as the Integrated Exposure Uptake Biokinetic (IEUBK) model. This model requires as input data on the levels of lead in all potentially contaminated environmental media (soil, dust, water, air, diet) at a specific location, and on the amount of these media taken in (by ingestion or inhalation)

by a child living at that location. Dermal exposure to lead is considered to be a minor source of exposure and is not included in the model.

The inputs to the IEUBK model are selected to reflect estimates of central tendency values (i.e., arithmetic means or medians). These estimated inputs are used to calculate an estimate of the central tendency (the geometric mean (GM)) of the distribution of blood lead values that might occur in a population of children exposed to the specified conditions. Assuming the distribution is lognormal, and given (as input) an estimate of the variability between different children (this is specified by the geometric standard deviation or GSD), the model calculates the expected distribution of blood lead values, and estimates the value of P10. This value may be compared to the EPA health-based goal (P10 ≤ 5%) in order to determine if environmental contamination levels at that location are acceptable or not.

4.4 Inputs to the IEUBK Model

Table 4-1 presents the IEUBK model inputs used in the risk assessment. For most parameters, the default values recommended by the guidance for the model were used (USEPA 2004a, USEPA 2008a). Details on the site-specific model inputs used in the risk assessment are described below.

4.4.1 Environmental Concentration Data

USEPA guidance (USEPA 1994a) specifies that environmental concentration values provided to the IEUBK model should generally be the best estimates of the arithmetic mean concentration in a medium, averaged over the exposure unit of concern (a property). The following sections describe the derivation of these input values for each environmental medium included in the IEUBK model calculations.

Outdoor Yard Soil

For outdoor yard soil, the exposure point concentration (EPC) for each property was calculated as the mean of all surface soil samples collected at that property from locations where children may be regularly exposed to soil (USEPA 2007b). At most residences, this was usually the mean of four composite surface soil samples collected from 4 quadrants of the yard (2 samples from the front yard and 2 samples from the back yard). At some properties, additional samples were collected from garden areas, play areas, and drip zones. In these cases, samples from garden areas and play areas were included in calculating the yard-wide average lead concentration for the property. However, soil samples from the drip zone were excluded from the yard-wide average lead concentration, because the drip zone is not thought to be an area where children are likely to play on a regular basis. In addition, the drip zone may contain lead-based paint which could significantly overestimate exposure if included in estimating EPCs.

As discussed previously (see Section 2.11.1), most measures of lead in yard soil are obtained using XRF analysis of "coarse" soils (i.e., < 2 mm). However, the ideal measure would be lead measured in the fine fraction (< 250 µm) using ICP. This is because it is considered likely that children are exposed mainly to the finer faction, and because ICP measurements are considered to be more accurate than XRF measurements. Therefore, all XRF measurements based on the coarse fraction were converted to estimate the concentration that would be expected had the fine fraction been analyzed by ICP. Appendix F presents the details of how this conversion is achieved. In brief, the calculations is as follows:

$$\text{Fine ICP} = \text{Coarse XRF} \cdot 1.16 \cdot 1.04$$

where:

1.16 = Conversion factor from coarse XRF to coarse ICP
1.04 = Conversion factor from coarse ICP to fine ICP

Appendix G presents the EPCs calculated in this way for each property evaluated by the risk assessment.

Indoor Dust

Indoor dust is a medium that is composed in part of outdoor soil and in part of material derived from other sources. Soil can be a dominant source of lead in indoor dust at residences. The IEUBK model incorporates a soil-to-dust transfer factor to describe the potential for lead in soil to be transported indoors and contribute to the concentration of lead in dust. This transfer factor is called M_{sd} and it is defined as the mass fraction of soil-derived particles in indoor dust (gram soil/gram dust) (USEPA 1998b). The M_{sd} is used to estimate the concentration in dust as follows:

$$Pb_{dust} = D0 + M_{sd} \bullet Pb_{soil}$$

where:

D0 = The concentration of lead in dust (µg Pb/g dust) that is not attributable to soil
Pb_{dust} = concentration of lead in indoor dust (µg Pb/g dust)
Pb_{soil} = outdoor soil lead concentration (µg Pb/g soil)
M_{sd} = mass fraction of soil in dust (g soil/g dust)

M_{sd} values can range from 0 to 1, with low values indicating that the contribution of outdoor soil to indoor dust is minimal, and high values indicating that soil is a dominant source of lead in indoor dust. The USEPA default assumption for M_{sd} is 0.70. That is, the default assumption is that 70% (by mass) of indoor dust is derived from outdoor soil (USEPA 1994a). However, the average fraction of dust derived from soil is likely to vary between sites, so USEPA Region 7 has

obtained data to derive a site-specific equation that relates the concentration of lead in dust to that in soil for the OLS. The data and the analysis are presented in Appendix F. The resulting equation is:

$$C_{dust} = 42 + 0.74 \cdot C_{soil}$$

This equation was used to calculate the concentration of lead in dust at each property, based on the mean concentration of lead in outdoor soil at the property. Note that this equation is used to estimate the dust concentration even in cases where a measured value for dust was available (N = 98). This is because the measured value in dust is conditional on the current conditions at the property (percent grass cover in the yard, number of people and pets residing at the property, frequency of house cleaning, etc.), while the goal is to perform a generic assessment that includes expected values for all future as well as current residents.

Drinking Water

Data on the concentration of lead in tap water at 98 properties in the Final Focus Area were collected in 2007 and 2008 (Black and Veatch 2008a and 2008b). Samples of two types were collected at each residence: "first-flush" samples (i.e., the water that emerges when the water is first turned on after standing overnight) and "post-flush" samples (i.e., water that emerges after the standing water has been fully flushed from the pipes). This is potentially important because lead levels in first-flush water tends to be higher than in post-flush samples in homes where lead is present in pipes, soldered joints, or plumbing fixtures.

In the original data set (Black and Veatch 2008a), one location (property ID 29349) yielded values of 48.8 µg/L for first-flush water and 635 µg/L for post-flush water. These values were considered unusual for two reasons: 1) the values were much higher than any other samples collected, and 2) the post-flush sample was much higher than the first-flush sample. Because these levels (if actual) would be of acute health concern to residents at the location, USEPA re-sampled the tap water at this residence and found the levels of lead to be considerably lower that the initial estimates, with both values below the national drinking water standard (15 µg/L). Because the high concentrations of lead were not confirmed by the second set of tap water samples, the first tap water results collected at this residence are considered invalid and were excluded from the risk assessment, and only the results from the second round of sampling at this property were used.

The resulting data are shown in Table 2-7. The average concentration in water ingested by residents was calculated by assuming that 50% of the water ingested is first-flush and 50% is post-flush (USEPA 1994a). Based on this assumption, the mean concentration of lead in tap water at the OLS is 1.36 µg/L.

Outdoor Air

Data on the mean concentration of lead in outdoor ambient air in the vicinity of the site was obtained from USEPA's AQS database (USEPA 2008b). This database contains ambient air monitoring results for contaminants that are considered hazardous air pollutants or criteria air pollutants by the Clean Air Act, including lead. Table 2-8 lists 13 stations located within Douglas county, and the availability of lead data from each station. As shown in Figure 2-7, 11 of these stations are located within the Final Focus Area of the OLS. Table 2-9 presents the annual average lead concentrations measured at these stations for the years 1997 to 2002. As seen, data are only available at 5 of the 11 stations during this time period.

Inspection of the data in Table 2-9 reveals that lead levels in ambient air have tended to decline over the time interval from 1997 to 2002. The reason for the decline is not certain, but one factor that is likely to be important is the cessation of ASARCO smelter operations in 1997. Because the goal of the risk assessment is to evaluate risks under current site conditions, only the most recent data were used in calculating the EPC for air. The available data are not adequate to determine with confidence if downward trends are continuing in the time frame of 2000 or later, so all data from 2000-2002 were used. If downward trends are still continuing, this approach could lead to an overestimate of lead exposure from ambient air under current site conditions.

It should be noted that the three stations for which data are available in the 2000-2002 time frame are all located in central Omaha near the former smelter site. The data are not adequate to reveal if there is a tendency for air concentrations to decrease as a function of distance from the former smelter site, so these data were used to represent all locations within the Final Focus Area. If there is a tendency for lead concentrations to decrease as a function of distance from the smelter, it is possible that the use of these data may tend to overestimate exposure from ambient air at locations that are further removed from the site.

Based on these considerations, the concentration value selected to represent lead in ambient air in the Final Focus Area is 0.036 $\mu g/m^3$.

Diet

For food items purchased from a grocery store, concentrations of lead were based on measured values reported by the Food and Drug Administration's (FDA) Total Diet Study (FDA 2006). These values are shown in Table 4-1. Exposure of area residents by ingestion of lead taken up from soil into home-grown garden vegetables was not evaluated in this risk assessment because no site-specific data were available on the concentration of lead in home grown produce. This is not considered to be a substantial source of uncertainty because data from other sites indicates that lead does not tend to strongly accumulate in home grown garden vegetables (Life Systems 1995, USEPA 2001d).

4.4.2 Lead Bioavailability

Accurate assessment of the human health risks resulting from oral exposure to lead requires knowledge of the amount of lead absorbed from the gastrointestinal tract into the body. This information on gastrointestinal absorption may be described either in absolute or relative terms:

> *Absolute Bioavailability (ABA)* is the ratio of the amount of lead absorbed to the amount ingested:
>
> ABA = (Absorbed Dose) / (Ingested Dose)
>
> This ratio is also referred to as the oral absorption fraction (AFo).
>
> *Relative Bioavailability (RBA)* is the ratio of the absolute bioavailability of lead in a test material to the absolute bioavailability of lead in an appropriate reference material:
>
> RBA = ABA(test) / ABA(reference)
>
> Usually the form of lead used as the reference material is a compound that is readily soluble in water (e.g., lead acetate) and that is expected to readily dissolve in gastrointestinal fluids when ingested.

For example, if 100 µg of lead dissolved in drinking water were ingested and a total of 50 µg were absorbed into the body, the ABA would be 0.50 (50%). Likewise, if 100 µg of lead contained in soil were ingested and 30 µg were absorbed into the body, the ABA for soil would be 0.30 (30%). If the lead dissolved in water was used as the frame of reference for describing the relative amount of lead absorbed from soil, the RBA would be 0.30/0.50, or 0.60 (60%).

Site-specific estimates of RBA of lead in soil may be obtained from either *in vivo* or *in vitro* methods. The *in vivo* approach measures the gastrointestinal absorption of lead in animals dosed with site soils compared to that for lead acetate. This ratio is the RBA for lead. The *in vitro* approach measures the fraction of lead that is solubilized from soil into an extraction fluid that simulates natural gastrointestinal conditions. The fraction solubilized is referred to as the *in vitro* bioaccessibility (IVBA). For lead, IVBA results are well correlated with the *in vivo* RBA results, and USEPA has determined that this particular IVBA method is appropriate for estimating the RBA for lead (USEPA 2007a). The relationship between a measured IVBA value and the corresponding RBA value is given by the following equation (USEPA 2007b):

$$RBA = 0.878 \bullet IVBA - 0.028$$

USEPA Region 7 conducted both *in vivo* and IVBA testing on soil samples collected from the OLS, and IVBA testing was conducted on indoor dust samples (see Table 4-2). The results are presented in the following sections.

4.4.2.1 Lead RBA in Soil

In Vivo Results

USEPA measured the RBA of lead in juvenile swine using two test materials collected from properties at the OLS (Casteel *et al.* 2004). Each test material consisted of a composite surface soil sample collected from 2 residential yards. These materials were sieved before testing to isolate the fine fraction (< 250 µm). This is because the fine fraction is generally considered to be more likely to adhere to the hands and be ingested than coarse soil particles. The original report of the *in vivo* study is provided in Appendix B, and the resulting RBA values are shown in Panel A of Table 4-3. As seen, the suggested point estimate and the plausible range of RBA of lead measured in these two surface soils was 1.01 (1.01-1.03) and 0.74 (0.68-0.76).

More recently, USEPA (2007a) described a new statistical approach for estimating the RBA of lead in soil from *in vivo* studies. In accord with this guidance, the RBA data were reanalyzed using USEPA's new recommended statistical method. In addition, the concentrations of lead in the two test soils were revised based on quintuplicate measurements of each soil. The revised results are provided in Appendix B and are summarized in Panel B of Table 4-3. As seen, the new statistical method yields point estimates and 90% confidence intervals for RBA of 0.96 (0.64 – 1.29) and 0.83 (0.58 – 1.10) for Test Materials 1 and 2, respectively.

IVBA Results

USEPA measured the IVBA of lead in the fine fraction (< 250 µm) of 47 different surface soil samples collected at the OLS (Drexler 2004, 2008a). These data are provided in Appendix B. Table 4-4 presents the IVBA results[1] and the estimated RBA values derived from the IVBA results using the equation presented above. As seen, the estimated RBA values range from 0.27 – 0.95, with a mean of 0.68. Figure 4-2 plots the estimated RBA values as a function of soil concentration. This plot indicates there are two data points with estimated RBA values that are considerably lower than the rest of the results. These two data points were identified as outliers using Rosner's Test (Gilbert 1987), and were excluded from the data set. The relationship between soil concentration and estimated RBA (excluding outliers) was examined using linear regression and the slope of the line was not statistically different from zero ($p = 0.45$). Based on this, it is concluded that the RBA of lead in soil is independent of concentration, and the best estimate of RBA based on the IVBA approach is equal to the mean of the estimated RBA values (outliers excluded). The resulting value is 0.70.

[1] Note: The values shown in this table are different than originally reported in Drexler (2004). This is because the 2004 report was published before the ICP results for the concentrations of lead in the fine fraction (<250 µm) were available, so the initial IVBA estimates presented in Drexler (2004) were based on XRF estimates of lead in the bulk fraction (<2 mm) (Drexler 2008a, personal communication). The values shown in Table 4-4 are based on the concentration of lead in the fine fraction measured by ICP.

Correlation Between RBA and IVBA

The two test materials used in the *in vivo* RBA study (Test Material 1 and Test Material 2) were also analyzed by IVBA (Drexler 2008a). Table 4-5 compares the RBA values derived from the IVBA results and from the *in vivo* study. For both test materials, the RBA values estimated from the *in vivo* test were somewhat higher than estimated using the IVBA approach, although the uncertainty bounds of the estimates overlapped in both cases.

In making this comparison, it should be recognized that a) *in vivo* RBA values are generally subject to more measurement error than IVBA values, and b) that extrapolation from IVBA to RBA yields the average RBA value associated with that IVBA, and that actual RBA values may be either higher or lower than the calculated mean. Thus, exact agreement is not expected.

Selection of Weight-of-Evidence RBA Value for Soil

Both the *in vivo* results for two samples and the *in vitro* results for 47 samples clearly indicate that the RBA of lead in soil at the site is higher than the USEPA default value of 0.60. The *in vitro* data suggest a mean value of about 0.70, while the two *in vivo* values are 0.83 and 0.96. In general, greatest weight is placed on the *in vivo* data, suggesting a point estimate of about 0.9. However, as noted above, *in vivo* values are more difficult to measure and have wider uncertainty than *in vitro* values, and in this case, there are 47 *in vitro* values compared to only two *in viv*o values. Based on the weight of evidence, a point estimate of 0.8 is selected to represent the mean RBA of lead in soil at the site. This value is slightly above the average for the *in vitro* estimates, is approximately equal to the *in vivo* point estimate for Test Material 2, and is only somewhat lower than the *in vivo* point estimate for Test Material 1.

4.4.2.2 Lead RBA in Indoor Dust

IVBA Results

USEPA measured the IVBA of lead in the fine fraction (< 250 µm) of 94 different indoor dust samples collected at the OLS in 2007 (Drexler 2008b). Table 4-6 presents the IVBA results and the estimated RBA values derived from the IVBA results. As seen in Table 4-6, the estimated RBA values for indoor dust range from 0.32 – 4.44, with a mean of 0.79. Figure 4-3 presents the estimated RBA values plotted as a function of the dust concentration. As seen, there are three data points with estimated RBA values that were considerably higher than the rest of the results (RBA > 1.5), and one data point has an estimated RBA that is considerable lower (RBA < 0.3). These four data points were identified as outliers using Rosner's Test (Gilbert 1987), and were excluded from the data set. The relationship between indoor dust and estimated RBA (excluding outliers) was examined using linear regression and the slope of the line was not statistically different from zero ($p = 0.18$). Based on this, it is concluded that the RBA of lead in indoor dust

is independent of concentration, and the best estimate is given by the average of the values (outliers excluded). The resulting value is 0.73.

Selection of Weight-of-Evidence RBA Value for Dust

In the case of soil, two data sets were available for consideration: two *in vivo* RBA values and 47 IVBA values. As discussed above, the two *in vivo* values tended to be somewhat higher than the corresponding RBA values computed for those samples from the IVBA values, and were also higher than the average of the other IVBA-based RBA values. On this basis, the RBA for soil was set to a value somewhat higher than the mean IVBA-based RBA. In the case of dust, only IVBA-based RBA estimates are available, and there are no direct *in vivo* measurements. However, there is no reason to expect that the RBA for dust should be lower than for soil. To the contrary, because dust particles tend to be smaller than some soil particles, it is expected that RBA in dust should be the same or higher than in soil. On this basis, and by extension of the rationale used for soil, the RBA value selected to represent the mean for dust is 0.8.

4.4.3 Age

When the IEUBK model is run for a particular property, the resulting output is intended to represent the distribution of long-term average (0-84 month) blood lead values in a population of children exposed at that location. When the IEUBK model must be run for a large number of properties (approximately 40,000 at the OLS), calculations are not run one by one, but are performed using a "batch mode". In the batch mode, the output is the blood lead level for a child of a specific age, which must be specified as input. In order to ensure that the output of the batch run is comparable to that of a single property run, an age of 50 months was selected for use. This is because the blood lead at age 50 months is nearly the same as the average blood from 0-84 months.

4.5 Results

The IEUBK model was run in batch mode to calculate the probability that a child would have a blood lead level over 10 µg/dL at each property in the OLS for which data are available and which has not yet undergone soil cleanup activities. The results are summarized in Table 4-7, stratified by zip code.

Inspection of Table 4-7 and Figure 4-4 reveals the following main findings:

- Of the 28,478 properties evaluated, a total of 19,445 homes (68%) are predicted to have P10 values at or below the health-based goal of 5%, and 9,033 properties (32%) have values that exceed the goal.

- Of these 9,033 properties, 3,177 have P10 values between 5% and 10%, 3,051 properties have P10 values between 10% and 20%, and 2,805 properties have P10 values greater than 20%.
- The location of properties with P10 values greater than the health-based goal of 5% were widespread across the OLS final focus area and were found within all zip codes with the exception of 68117 (which only had 2 properties).

These results indicate that a number of homes or parcels within the final focus area have soil lead levels that are of potential health concern to children who may reside there, now or in the future.

4.6 Uncertainties

It is important to recognize that the exposure and risk calculations for lead presented in this section are subject to uncertainty that arises from a number of different sources. Each of the major sources of uncertainty in the lead risk assessment are discussed below. It is important for risk managers and the public to take these uncertainties into account when interpreting the risk conclusions for lead at this site.

4.6.1 Uncertainty in Lead Concentration Estimates

Evaluation of human health risk from lead at any particular location (property) requires accurate information on the average concentration level of lead in each environmental medium at that location.

Uncertainty in Soil Concentration Values

For soil, lead concentrations were usually measured using XRF for four or more composite samples representing different areas of the residential property. Each measured XRF value is uncertain, due both to analytical error and to sampling variability. Consequently, the mean across the multiple values is also uncertain. The effect of analytical error is likely to be relatively small. The effect of sampling error is somewhat larger than the effect of analytical error, but is still relatively small because each sample is a composite of 5 sub-samples. That is, the variability in the mean of five samples is smaller than the variability in each of the individual samples.

Additional uncertainty in the measured concentration of lead in soil arises from two adjustments to the measured data that are needed. First, XRF values of lead in soil at this site tend to be biased somewhat low compared to paired ICP measures (see Appendix C), so it was necessary to adjust for this bias using a factor of 1.16. This adjustment factor itself is uncertain (see Appendix F), so corrected values may be either higher or lower than the true mean. Second, XRF measures of lead were performed on "bulk" soils (< 2 mm) while human exposure is believed to be primarily to "fine" soils (< 250 µm). At some sites, the concentration of lead in

the fine fraction can be up to 30% or more higher than in the bulk fraction, which would result in an underestimate of exposure if not accounted for. However, at this site, the ratio of the concentration of lead in the fine fraction to that in the bulk fraction is only slightly higher than 1 (1.04). Thus, although this adjustment is a source of uncertainty, the magnitude of the uncertainty is small.

Uncertainty in Dust Concentration Values

Lead in soil may serve as a source of contamination of indoor dust. The USEPA generally assumes that 70% of dust is derived from soil (i.e., $C_{dust} = 0.7 \cdot C_{soil}$). At this site, data on the relationship between lead levels in soil and dust were collected at 98 different properties in order to support the derivation of a site-specific equation to relate lead levels in dust to those in soil. These data and the resulting equation are detailed in Appendix F.

As discussed in Appendix F, quantification of the relationship is complicated by the fact that there is measurement and sampling error in both the soil and the dust values, and that both errors tend to increase as the concentration of lead increases. A number of different statistical strategies were investigated to determine which best accounted for the errors, and the resulting equation utilizes the most successful method. However, the parameters of the regression equation are uncertain, and dust levels at any given property may be either higher or lower than computed using the equation.

4.6.2 Uncertainty in Lead Absorption from Soil and Dust

Another important source of uncertainty regarding the risk from lead in soil and dust is the degree of absorption (RBA) of lead within the gastrointestinal tract. For this risk assessment, a site-specific relative bioavailability factor for lead of 0.80 has been applied to both soil and dust. This value is based on two types of site-specific data: *in vivo* estimates derived by dosing an appropriate animal model (juvenile swine) with two different site soils, and *in vitro* measures of bioaccessability (IVBA) derived for 47 soils and 94 dust samples. This is a very substantial data set upon which to estimate a site-specific RBA value, and the data are in general agreement that the value is higher than the default. However, there remains some difference between the *in vivo* and the IVBA estimates, and although the value selected for use reflects a weight-of evidence consideration of both types of data, the selected value is uncertain, and might be either higher or lower than the true value.

4.6.3 Uncertainty in the IEUBK Model

All predictive models, including the IEUBK model, are subject to a number of limitations. First, there is inherent difficulty in providing the model with reliable estimates of human exposure to lead-contaminated media. For example, exposure to soil and dust is difficult to quantify because human intake of these media is likely to be highly variable, and it is very difficult to derive

accurate measurements of actual intake rates. Second, it is often difficult to obtain reliable estimates of key pharmacokinetic parameters in humans (e.g., absorption rates, distribution and clearance rates from various tissues), since direct observations in humans are limited. Finally, the absorption, distribution, and clearance of lead in the human body is an extremely complicated process, and any mathematical model intended to simulate the actual processes is likely to be an over-simplification. Consequently, model calculations and predictions are uncertain.

A second source of uncertainty arises from the approached used by the IEUBK model to estimate the probability of a random child at a property having a blood level above 10 µg/dL (P10). As discussed in Section 3.3, the IEUBK model estimates this value by assuming that the distribution of blood lead values at a location is lognormal with a specified geometric standard deviation (GSD). The value of P10 is quite sensitive to the assumed value of GSD, with P10 increasing rapidly as GSD increases. The true value of GSD at the OLS is not known, and the lead risk calculations are based on a default value of 1.6, which is estimated from national blood lead statistics (USEPA 1994a). Use of this assumed GSD is an important source of uncertainty, and actual P10 values might be either higher or lower than calculated.

5.0 EVALUATION OF RISKS FROM NON-LEAD CONTAMINANTS

Although lead was the primary contaminant released to the environment from the historic operation of the smelters in the OLS, other metal and metalloid contaminants may also have been released. The Interim Risk Assessment (NHHS 2004) included an initial evaluation of the risks from non-lead contaminants, and concluded that adverse non-cancer and cancer effects were not anticipated as a result of incidental ingestion of soil (NHHS 2004). The purpose of this section is to provide an updated evaluation of the risks from non-lead contaminants that utilizes new data that have been collected since the date of the Interim Risk Assessment.

5.1 Exposure Assessment

The generalized CSM for exposure of residents to smelter-related contaminants was presented above in Figure 3-1. The detailed CSM for exposure of residents to non-lead COPCs is presented in Figure 5-1. The following sections describe which of these pathways are likely to be potentially significant and that have been selected for quantitative evaluation in the risk assessment.

Incidental Ingestion of Surface Soil and Indoor Dust

Even though few people intentionally ingest soil, residents (especially children) who have direct contact with soil at the site might ingest small amounts that adhere to their hands during outdoor activities. In addition, soil can be brought into residences by direct transport on shoes, clothing, pets, etc., or may be deposited in residence by airborne dust where it becomes mixed into indoor dust, which may also be ingested by hand to mouth activities. Incidental ingestion of outdoor soil and indoor dust are often the most important routes of human exposure at a site, so these exposure pathways are evaluated quantitatively.

Dermal Contact with Soil and Indoor Dust

Residents may get soil on their skin during outdoor activities involving direct contact with soil. Although most metals do not readily cross the skin into the body, dermal exposure to soil is a complete exposure pathway and is evaluated quantitatively in the risk assessment when data are sufficient. Little is known about dermal contact with indoor dust while in a residence, but it is generally assumed this is minor compared to dermal contact with soil. Therefore, dermal contact with dust is not evaluated quantitatively.

Inhalation of Airborne Soil Particulates

Whenever contaminated soil is exposed at the surface, particles of contaminated surface soil may become suspended in air by wind disturbance, and humans in the area could inhale those

particles. Screening level calculations (see Appendix D) indicate that exposure to particulates suspended by wind erosion is very small compared to oral exposure, so this pathway is evaluated qualitatively rather than quantitatively in this risk assessment.

Ingestion of Homegrown Produce Items

Area residents may grow garden vegetables or fruit in smelter-impacted soil, and some contaminants in the soil may be taken up by the vegetables or fruit and ingested. Because no data are available on the levels of metals in locally grown vegetables or fruit, this exposure pathway cannot be evaluated quantitatively in this risk assessment. However, uptake of metals into plants is usually not extensive, so omission of this pathway is not likely to be a major source of uncertainty.

Exposure to Surface Water and Sediment

Contaminants in soil may be eroded into the Missouri River from surface runoff events, and area residents who have contact with the river might be exposed by incidental ingestion or dermal contact with surface water or sediment. However, sampling results of water and sediment in the Missouri River immediately adjacent to the ASARCO and Gould facilities did not detect elevated levels of lead or any other smelter-related contamination (Black and Veatch 2008c). This is not unexpected because the amount of dilution provided by the flow of the Missouri River is expected to reduce any potential release of transported site contaminants to very low levels (USEPA 2005a). Therefore, exposures to surface water and sediment are not evaluated quantitatively in this assessment.

Exposure to Groundwater

In some cases, smelting-related releases to soil may lead to increased levels of contaminants in groundwater due to infiltration of the contaminants from rain and snow-melt. If the groundwater were used for drinking, area residents could be exposed by ingestion of the contaminants. However, at the OLS site, area residents receive their drinking water from the municipal public water system, so it is not believed that the groundwater pathway is complete at present.

Exposure to Drinking Water

Although drinking water at the site is derived from the Missouri River, because no significant increases in metal concentrations have been observed in the river near the site, and because the water is treated in accord with the requirements of the Safe Drinking Water Act, this exposure pathway is not considered to be of significant concern and is not evaluated quantitatively in this risk assessment.

Exposure to Fish Tissue

Residents who fish in the Missouri River may consume locally caught fish and may indirectly ingest metals that are taken up from surface water or sediment into edible portions of fish. Because metals in surface water and sediment samples collected from the Missouri River were not observably elevated, and because the dilution factor of the Missouri River is so great (Black and Veatch 2008c), site-related contaminants are not expected to accumulate in fish tissue. Thus, exposure to metals ingested in fish tissue is likely to be a minor exposure pathway which is not be evaluated quantitatively in this risk assessment.

Summary of Exposure Pathways for Quantitative Assessment

Based on the evaluation of potential exposure pathways presented above, the following exposure pathways to non-lead contaminants are evaluated quantitatively in this risk assessment:

Population	**Exposure Pathway**
Residents	Ingestion of outdoor soil Ingestion of indoor dust Dermal contact with outdoor soil

5.2 Quantification of Human Exposure

5.2.1 Basic Equations

Quantification of Oral Exposure

The amount of a chemical which is ingested from an environmental medium is quantified in terms of the average daily intake (DI, expressed in units of mg/kg-day), calculated using an equation of the following form:

$$DI = C * (IR / BW) * (EF * ED / AT)$$

where:

DI = Daily intake of chemical (mg of chemical per kg of body weight per day).

C = Concentration of the chemical in the contaminated environmental medium (soil, water) to which the person is exposed. The units are mg/L for water and mg/kg for soil.

IR = Intake rate of the contaminated environmental medium. The units are kg/day for soil and dust.

BW = Body weight of the exposed person (kg).

EF = Exposure frequency (days/year). This describes how often a person is likely to be exposed to the contaminated medium over the course of a typical year.

ED = Exposure duration (years). This describes how long a person is likely to be exposed to the contaminated medium during their lifetime.

AT = Averaging time (days). This term specifies the length of time over which the average dose is calculated. Usually, two different averaging times are considered:

> "Chronic" exposure includes averaging times on the scale of years (typically ranging from 7 years to 70 years). This exposure duration is used when assessing the non-cancer risks from chemicals of potential concern.
>
> "Lifetime" exposure employs an averaging time of 70 years. This exposure interval is selected when evaluating cancer risks.

Note that the factors EF, ED, and AT combine to yield a factor between zero and one. Values near 1.0 indicate that exposure is nearly continuous over the specified averaging period, while values near zero indicate that exposure occurs only rarely.

For mathematical convenience, the general equation for calculating dose can be written as:

$$DI = C * HIF$$

where:

HIF = Human Intake Factor. This term describes the average amount of an environmental medium contacted by the exposed person each day. The value of HIF is typically given by:

$$HIF = (IR / BW) * (EF * ED / AT)$$

The units of HIF are kg/kg-day for soil and dust.

Because one or more exposure parameters (e.g., intake rates, body weight, and exposure frequency) may change as a function of age, exposure calculations are often performed separately for children and adults. In the case of residents, non-cancer risks were calculated separately for a child resident and an adult resident. However, for estimating excess cancer risks from exposure to a chemical, because the same individual may be exposed beginning as a child and extending into adulthood, exposure is calculated as the time-weighted average (TWA) exposure:

$$TWA\ DI = C *[(IRc / BWc) * (EFc * EDc / AT) + [(IRa / BWa) * (EFa * EDa / AT)]$$

where the subscripts “c” and “a” refer to child and adult, respectively.

Quantification of Dermal Exposure

The amount of a chemical which is absorbed across the skin is referred to as the dermal absorbed dose (DAD). Procedures for estimation of the DAD as outlined in USEPA (2004a) were used in this assessment and are described below. For chemicals except lead (which is evaluated as described in Section 4.4.2), exposure is quantified using an equation of the following general form:

$$DAD = DA_{event} * EF * ED * EV * SA / (BW * AT)$$

where:

DAD = Dermal absorbed dose (mg of chemical per kg of body weight per day).

DA_{event} = Absorbed dose per event (mg of chemical per square centimeter of skin surface area per event). This is media-specific and is further described below.

EF = Exposure frequency (days/year). This describes how often a person is likely to be exposed to the contaminated medium over the course of a typical year.

ED = Exposure duration (years). This describes how long a person is likely to be exposed to the contaminated medium during their lifetime.

EV = Event frequency (events/day). This describes the number of times per day a person comes in contact with a contaminant in soil.

SA = Surface area (cm^2). This describes the amount of skin exposed to the contaminated media.

BW = Body weight of the exposed person (kg).

AT = Averaging time (days). This term specifies the length of time over which the average dose is calculated.

For contaminants in soil, DA_{event} is estimated as follows:

$$DA_{event} = C_{soil} * CF * AF * ABS_d$$

where:

C_{soil} = Chemical concentration in soil (mg of chemical per kg of soil).

CF = Conversion factor (10^{-6} kg/mg).

AF = Adherence factor (mg of soil per square centimeter of skin surface area per event). This describes the amount of soil that adheres to the skin per unit of surface area.

ABS_d = Dermal absorption fraction (unitless). This value is chemical-specific and represents the contribution of absorption of a chemical across a person's skin from soil to the systemic dose.

Combining these equations yields the following:

$$DAD = C_{soil} * CF * AF * ABS_d * EF * ED * EV * SA / (BW * AT)$$

For mathematical convenience, the general equation for calculating DAD can be written as:

$$DAD = C_{soil} * ABS_d * HIF_{soil}$$

where:

$$HIF_{soil} = (SA * AF * EF * ED * EV * CF) / (BW * AT)$$

The units of HIF_{soil} are kg/kg-day.

As described above, in the case of residents, because the same individual may be exposed beginning as a child and extending into adulthood, exposure is calculated as the time-weighted average (TWA) exposure for evaluating cancer risks. For non-cancer risks, children and adults were evaluated separately.

5.2.2 Human Exposure Parameters

For every exposure pathway of potential concern, it is expected that there will be differences between different individuals in the level of exposure at a specific location due to differences in intake rates, body weights, exposure frequencies, and exposure durations. Thus, there is normally a wide range of average daily intakes between different members of an exposed population. Because of this, all daily intake calculations must specify what part of the range of doses is being estimated. Typically, attention is focused on intakes that are "average" or are otherwise near the central portion of the range, and on intakes that are near the upper end of the range (e.g., the 95th percentile). These two exposure estimates are referred to as Central Tendency Exposure (CTE) and Reasonable Maximum Exposure (RME), respectively.

The USEPA has collected a wide variety of data and has performed a number of studies to help establish default values for most residential exposure parameters. The chief sources of these standard default values are the following documents:

1. Risk Assessment Guidance for Superfund (RAGS). Volume I. Human Health Evaluation Manual (Part A). USEPA 1989a.

2. Human Health Evaluation Manual, Supplemental Guidance: "Standard Default Exposure Factors." USEPA 1991a.

3. Exposure Factors Handbook. USEPA 1997a.

4. Supplemental Guidance for Developing Soil Screening Levels for Superfund Sites. USEPA 2002a.

5. Risk Assessment Guidance for Superfund (RAGS). Volume I. Human Health Evaluation Manual (Part E, Supplemental Guidance for Dermal Risk Assessment). USEPA 2004a.

Exposure parameters for evaluation of area residents are summarized in Table 5-1 and are described below.

Body Weight. For children, a body weight of 15 kg was assumed for a CTE and RME resident (USEPA 1991a). This value is the average body weight for young children (boys and girls combined) under the age of 6 years. For adults, a body weight of 70 kg was assumed for a CTE and RME resident. This value is the average body weight for adult men and women (USEPA 1991a and 1997a). These body weights are USEPA (1991a) recommended default values.

Exposure Frequency. For both children and adult residents, a CTE exposure frequency of 245 days per year and an RME exposure frequency of 350 days per year were assumed. The CTE value is based on the mean percent of time men and women spend at home (67%) reported in Table 15A-3 of the Exposure Factors Handbook (USEPA 1997a). The RME value is the USEPA (1991a) recommended default value for exposure frequency for a resident.

Exposure Duration. For both children and adult residents, a CTE exposure duration of 9 years and an RME exposure duration of 30 years were used. The CTE value is based on the average residency occupancy of a home, as reported in EPA's Exposure Factors Handbook (USEPA 1997a). Of the 9 years of exposure, it is assumed that 2 years of exposure occur as a child and that 7 years of exposure occur as an adult, based on professional judgment. The RME value is the USEPA (1991a) recommended default value for a resident. This value represents the 90th percentile for time spent at one residence (USEPA 1991a).

Averaging Time. As recommended by USEPA (1989a), an averaging time of 70 years was used for cancer for both CTE and RME residents. For non-cancer, the averaging time was set equal to the exposure duration (USEPA 1989a).

Soil and Dust Ingestion Rate. For children, an intake rate of 200 mg/day of soil plus dust (combined) was used for an RME individual (USEPA 1991a), and an intake rate of 100 mg/day was used for a CTE individual (USEPA 1997a). For adults, the USEPA (1991a) recommended default intake rate of 100 mg/day was used for an RME individual and an intake rate of 50 mg/day was used for a CTE individual.

Exposed Surface Area. USEPA (2004a) recommended default values for both child and adult residents were used for the exposed surface area. An exposed surface area of 2,800 cm^2 was used for both the CTE and RME child resident and an exposed surface area of 5,700 cm^2 was used for the CTE and RME adult resident. The values assume that the exposed skin surface

includes the head, hands, forearms, and lower legs. The exposed surface area for children also assumes exposure of the feet.

Soil Adherence Factor. USEPA (2004a) recommended default values for child and adult residents were used for the soil adherence factor exposure parameter. For children, the CTE adherence factor of 0.04 mg/cm^2 is the geometric mean of children playing with dry soil. The RME adherence factor of 0.2 mg/cm^2 for a child resident represents the high end (95th percentile) of a typical soil contact activity (children playing at a day care center) and the central tendency estimate (50th percentile) of a high-end soil contact activity (children playing with wet soil). For adults, the CTE adherence factor of 0.01 mg/cm^2 represents the geometric mean adherence factor for a low-end soil contact activity (grounds keepers). The RME adherence factor of 0.07 mg/cm^2 is based on the 50th percentile of a high-end soil contact activity (gardening) (USEPA 2004a).

Dermal Absorption Fraction. USEPA (2004a) recommended default values were available for 2 of the 11 COPCs (arsenic and cadmium). The values used in the risk assessment were 0.03 for arsenic and 0.001 for cadmium (USEPA 2004a).

5.2.3 Selection of Exposure Points

An exposure point (also referred to as an exposure unit or exposure area) is an area where a human receptor is likely to be exposed to a contaminated environmental medium. Selection of the bounds of an exposure point is based mainly on a consideration of the likely activity patterns of the exposed receptors. That is, an exposure point is an area within which a receptor is likely to spend most of their time and to move about more or less at random.

For exposure of residents to outdoor soil, it is expected that most contact will occur when a resident is working or playing in their own yard. Likewise, exposure to indoor dust is expected to occur mainly within the home of the resident. For these reasons, the exposure point selected for evaluation of residents is the property where the resident lives.

5.2.4 Exposure Point Concentrations

Because of the assumption of random exposure over an exposure area, risk from a chemical is related to the arithmetic mean concentration of that chemical averaged over the entire exposure area. Since the true arithmetic mean concentration cannot be calculated with certainty from a limited number of measurements, the USEPA recommends that the upper confidence limit (UCL) of the arithmetic mean at each exposure point be used when calculating exposure and risk at that location (USEPA 1992a). The approach that is most appropriate for computing the UCL of a data set depends on a number of factors, including the number of data points available, the shape of the distribution of the values, and the degree of censoring (i.e., samples below the detection limit) (USEPA 2002b). To facilitate this process, the USEPA has developed a

specialized software package (ProUCL v4.0) that calculates UCLs for a data set using several different strategies and recommends which UCL is considered preferable based on the properties of the data set (USEPA 2007c).

At this site, reliable data for all non-lead COPCs except zinc are restricted to measurements performed by ICP. Because ICP measurements were performed on only about 5% of all soil samples, many properties have no reliable measurements, and when a value is available, there is usually only one (maximum = 2) per property. Because it is not possible to compute a reliable UCL based on only one or two samples, the maximum values for each property were used to compute exposure and risk.

For zinc, because XRF data are available in addition to the ICP data, there are usually 4-6 reliable samples per property. While ProUCL can compute a UCL based on 4-6 samples, it is generally recognized that a UCL based on a sample set of this relatively small size is statistically unstable, and it is expected that the majority of UCL values would exceed the maximum detected value. For this reason, calculations of exposure and risk from zinc were also performed using the maximum concentration values for each property.

Because the toxicity of chromium depends on valence state (Cr^{III} *vs*. Cr^{VI}), but the data report only the concentration of total chromium (Cr^{III} + Cr^{VI}), the fraction of total chromium in each valence state was estimated based on data summarized in ATSDR (2000), USEPA (1998c) and USEPA (1998d), as follows:

Medium	Valence of Chromium	
	Cr^{III}	Cr^{VI}
Soil	86%	14%

Detailed exposure point concentrations for all non-lead COPCs are provided in Appendix G.

5.2.5 Exposure Calculations

Detailed exposure calculations are provided in Appendix H.

5.3 Toxicity Assessment

The objective of a toxicity assessment is to identify what adverse health effects a chemical causes, and how the appearance of these adverse effects depends on exposure level. In addition, the toxic effects of a chemical frequently depend on the route of exposure (oral, inhalation, dermal) and the duration of exposure (subchronic, chronic, or lifetime). Thus, a full description of the toxic effects of a chemical includes a listing of what adverse health effects the chemical may cause, and how the occurrence of these effects depends upon dose, route, and duration of exposure.

5.3.1 Basic Methods

The toxicity assessment process is usually divided into two parts: the first characterizes and quantifies the non-cancer effects of the chemical, while the second addresses the cancer effects of the chemical. This two-part approach is employed because there are typically major differences in the time-course of action and the shape of the dose-response curve for cancer and non-cancer effects.

Non-Cancer Effects

Essentially all chemicals can cause adverse health effects if given at a high enough dose. However, when the dose is sufficiently low, typically no adverse effect is observed. Thus, in characterizing the non-cancer effects of a chemical, the key parameter is the threshold dose at which an adverse effect first becomes evident. Doses below the threshold are considered to be safe, while doses above the threshold are likely to cause an effect.

The threshold dose is typically estimated from toxicological data (derived from studies of humans and/or animals) by finding the highest dose that does not produce an observable adverse effect, and the lowest dose which does produce an effect. These are referred to as the "No-observed-adverse-effect-level" (NOAEL) and the "Lowest-observed-adverse-effect-level" (LOAEL), respectively. The threshold is presumed to lie in the interval between the NOAEL and the LOAEL. However, in order to be conservative (protective), non-cancer risk evaluations are not based directly on the threshold exposure level, but on a value referred to as the Reference Dose (RfD). The RfD is an estimate (with uncertainty spanning perhaps an order of magnitude) of a daily exposure to the human population (including sensitive subgroups) that is likely to be without an appreciable risk of deleterious effects during a lifetime.

The RfD is derived from the NOAEL, LOAEL or benchmark dose by dividing by an "uncertainty factor" that reflects the limitations of the data used. If the data are from studies in humans, and if the observations are considered to be very reliable, the uncertainty factor may be as small as 1.0. However, the uncertainty factor is normally at least 10, and can be much higher if the data are limited. Uncertainty factors are assigned to account for uncertainty arising from extrapolation of animal data to humans, the use of a LOAEL instead of a NOAEL, the use of less than chronic exposure, and other limitations in the available data (e.g., lack of reproductive data, etc.). The effect of dividing the NOAEL or the LOAEL by an uncertainty factor is to ensure that the RfD is not higher than the threshold level for adverse effects. Thus, there is always a "margin of safety" built into an RfD, and doses equal to or less than the RfD are nearly certain to be without any risk of adverse effect. Doses higher than the RfD may carry some risk, but because of the margin of safety, a dose above the RfD does not mean that an effect will necessarily occur.

Cancer Effects

For cancer effects, the toxicity assessment process has two components. The first is a qualitative evaluation of the weight of evidence (WOE) that the chemical does or does not cause cancer in humans. Typically, this evaluation is performed by the USEPA (1986), using the system summarized below:

WOE Group	Meaning	Description
A	Known human carcinogen	Sufficient evidence of cancer in humans.
B1	Probable human carcinogen	Suggestive evidence of cancer incidence in humans.
B2	Probable human carcinogen	Sufficient evidence of cancer in animals, but lack of data or insufficient data in humans.
C	Possible human carcinogen	Suggestive evidence of carcinogenicity in animals.
D	Cannot be evaluated	No evidence or inadequate evidence of cancer in animals or humans.
E	Not carcinogenic to humans	Strong evidence that it does not cause cancer in humans.

Note that USEPA has revised the weight of evidence descriptions and its approach for evaluating the carcinogenic potential of environmental contaminants. This is described in USEPA (2005b).

For chemicals which are classified in Group A, B1, B2, or C, using the 1986 guidelines, the second part of the toxicity assessment is to describe the carcinogenic potency of the chemical. This is done by quantifying how the number of cancers observed in exposed animals or humans increases as the dose increases. Typically, it is assumed that the dose-response curve for cancer has no threshold, arising from the origin and increasing linearly until high doses are reached. Thus, the most convenient descriptor of cancer potency is the slope of the dose-response curve at low doses (where the slope is still linear). This is referred to as the Slope Factor (SF), which has dimensions of risk of cancer per unit dose.

Estimating the cancer SF is often complicated by the fact that observable increases in cancer incidence usually occur only at relatively high doses, frequently in the part of the dose-response curve that is no longer linear. Thus, it is necessary to use mathematical models to extrapolate from the observed high dose data to the desired (but unmeasurable) slope at low dose. In order

to account for the uncertainty in this extrapolation process, USEPA typically chooses to employ the upper 95th confidence limit of the slope as the Slope Factor. That is, there is a 95 percent probability that the true cancer potency is lower than the value chosen for the SF. This approach ensures that there is a margin of safety in cancer as well as non-cancer risk estimates.

5.3.2 Toxicity Values

Toxicity values (RfD and SF values) are often estimated by a number of different groups or agencies. USEPA (2003b) identifies a recommended hierarchy for selecting toxicity values for use in human health risk assessment at Superfund sites. The first preference is for USEPA consensus values as listed in the Integrated Risk Information System (IRIS), an electronic database containing human health assessments for various chemicals (available online at http://www.epa.gov/iris/). If values are not available from IRIS, the next preference is to seek Provisional Peer Reviewed Toxicity Values for Superfund (PPRTVs) developed by USEPA's Superfund Health Risk Technical Support Center (STSC). If PPRTVs are not available, toxicity values may be obtained from other sources, such as the Agency for Toxic Substances and Disease Registry's (ATSDR) Minimal Risk Levels (MRLs) (available online at http://www.atsdr.cdc.gov/mrls.html), California EPA's Toxicity Criteria Database (available online at http://www.oehha.ca.gov/risk/ChemicalDB/index.asp), and USEPA's Health Effects Assessment Summary Tables (HEAST) (USEPA 1997b). Most of these values are also compiled in the Regional Screening Level Tables developed and maintained by USEPA (USEPA 2008c).

Table 5-2 summarizes the toxicity values used for quantitative evaluation of human health risks from COPCs at this site. Values were selected in accordance with USEPA (2003b). Points to note regarding the data in this table are listed below:

- Two oral RfD values are available for cadmium, depending on exposure medium (water, food). The value for food is assumed to apply to soil.

- The RfD for manganese (0.023 mg/kg-day) is based on the oral RfD of 1.4E-01 mg/kg-day in the diet. In accord with recommendations in IRIS, this value is modified by dividing by a Modifying Factor of 3 for application to exposures from soil.

- The RfD for thallium (6.5E-05 mg/kg-day) is based on the oral RfD for thallium sulfate of 8E-05 mg/kg-day. As recommended by USEPA (2008c), this value is adjusted to factor out the molecular weight of the sulfate ion by multiplying by the percent thallium in thallium sulfate (81%).

- The RfD for vanadium (5E-03 mg/kg-day) is based on the oral RfD for vanadium pentoxide of 9E-03 mg/kg-day. As recommended by USEPA (2008c), this value is

adjusted to factor out the molecular weight of the oxide ion by multiplying by the percent vanadium in vanadium pentoxide (56%).

5.3.3 Adjustments for Relative Bioavailability

As discussed above for lead, accurate assessment of human exposure to metals in soil and dust requires knowledge of the amount of metal absorbed into the organism following contact with a contaminated medium.

No site-specific data were available on the relative bioavailability of any non-lead COPCs in soil or dust. In the absence of site-specific data, the RBA for all non-lead chemicals in all media was assumed to be 1.0 (USEPA 1989a). This is expected to be a conservative assumption, tending to overestimate the true degree of absorption and risk.

5.3.4 Adjustments for Evaluating Dermal Exposures

In the absence of dermal toxicity factors, USEPA (1989a, 2004a) recommends that dermal toxicity factors be estimated by extrapolation from oral toxicity factors. Because dermal exposures are expressed in terms of absorbed dose, while oral exposures are expressed in terms of ingested dose, this requires an adjustment of the oral toxicity factors as follows:

$$SF_{ABS} = SFo/ABS_{GI}$$
$$RfD_{ABS} = RfD_o * ABS_{GI}$$

where:

SF_o	=	Oral slope factor based on ingested dose $(mg/kg\text{-}day)^{-1}$
SF_{ABS}	=	Dermal slope factor based on absorbed dose $(mg/kg\text{-}day)^{-1}$
RfD_o	=	Oral reference dose based in ingested dose (mg/kg-day)
RfD_{ABS}	=	Dermal reference dose based on absorbed dose (mg/kg-day)
ABS_{GI}	=	Fraction of contaminant absorbed in gastrointestinal tract (dimensionless) in the critical toxicity study

USEPA (2004a) provides recommended ABS_{GI} values for a number of chemicals, and these values were used when available. In accord with USEPA (2004a) guidance, if no ABS_{GI} value was recommended, a factor of 1.0 was used. Table 5-2 presents the dermal toxicity values (SF_{ABS} and RfD_{ABS}) used in the risk assessment and the values used in their derivation.

5.4 Risk Characterization

5.4.1 Basic Methods

Non-Cancer

The potential for non-cancer effects is evaluated by comparing the estimated daily intake of the chemical over a specific time period with the RfD for that chemical derived for a similar exposed period. This comparison results in a non-cancer Hazard Quotient (HQ), as follows (USEPA 1989a):

$$HQ = DI / RfD$$

where:

HQ = Hazard Quotient
DI = Daily Intake (mg/kg-day)
RfD = Reference Dose (mg/kg-day)

If the HQ for a chemical is equal to or less than one (1E+00), it is believed that there is no appreciable risk that non-cancer health effects will occur. If an HQ exceeds 1E+00, there is some possibility that non-cancer effects may occur, although an HQ above 1E+00 does not indicate an effect will definitely occur. This is because of the margin of safety inherent in the derivation of all RfD values. However, the larger the HQ value, the more likely it is that an adverse effect may occur.

If an individual is exposed to more than one chemical, a screening-level estimate of the total non-cancer risk is derived simply by summing the HQ values for that individual. This total is referred to as the Hazard Index (HI). If the HI value is less than 1E+00, non-cancer risks are not expected from any chemical, alone or in combination with others. If the screening level HI exceeds 1E+00, it may be appropriate to perform a follow-on evaluation in which HQ values are added only if they affect the same target tissue or organ system (e.g., the liver). This is because chemicals which do not cause toxicity in the same tissues are not likely to cause additive effects.

Cancer

The excess risk of cancer from exposure to a chemical is described in terms of the probability that an exposed individual will develop cancer because of that exposure by age 70 which is considered a lifetime. For each chemical of concern, this value is calculated from the daily intake of the chemical from the site, averaged over a lifetime (DI_L), and the slope factor (SF) for the chemical, as follows (USEPA 1989a):

$$\text{Excess Cancer Risk} = 1 - \exp(-DI_L * SF)$$

In most cases (except when the product of $DI_L \cdot SF$ is larger than about 0.01), this equation may be accurately approximated by the following:

$$\text{Excess Cancer Risk} = DI_L * SF$$

Excess cancer risks are summed across all chemicals of concern and all exposure pathways that contribute to exposure of an individual in a given population.

The level of total cancer risk that is of concern is a matter of personal, community, and regulatory judgment. In general, the USEPA considers excess cancer risks that are below about 1E-06 to be so small as to be negligible, and risks above 1E-04 to be sufficiently large that some sort of remediation is desirable. Excess cancer risks that range between 1E-04 and 1E-06 are generally considered to be acceptable (USEPA 1991b), although this is evaluated on a case by case basis, and USEPA may determine that risks lower than 1E-04 are not sufficiently protective and warrant remedial action.

5.4.2 Results: Estimated Risks to Residents from Non-Lead COPCs

Detailed cancer and non-cancer risk calculations for non-lead COPCs are provided in Appendix H, and the results are summarized in Tables 5-3 to 5-6. As mentioned above, not all 13 non-lead COPCs were analyzed in every soil sample, so risk estimates are not available for every COPC at every residential property.

Estimated Cancer Risks

The only COPC at this site that is carcinogenic by the oral or dermal route is arsenic. Table 5-3 presents a summary of the estimated cancer risks to CTE residents and RME residents from arsenic in soil, including both the ingestion and dermal pathways. As seen, estimated cancer risks to CTE residents are within USEPA's target risk range (1E-06 to 1E-04) at all properties. Estimated risks to RME residents are also within USEPA's target risk range at most properties, although risks exceed 1E-04 at 141 locations (5% of the properties with data). The excess individual lifetime cancer risks at these 141 properties range from 1E-04 to 1E-03.

Non-Cancer Risk

By Chemical

Table 5-4 shows the estimated non-cancer Hazard Quotients (HQs) for residential CTE (Panel A) and RME (Panel B) scenarios, including both children (age 0-6 years) and adults (age 7-30 years). As shown, risks from most COPCs in surface soil are below a level of potential concern

(HQ ≤ 1E+00) for both child and adult residents with CTE and RME scenarios. An exception is arsenic, which results in an HQ > 1E+00 for a RME child at about 10% of the properties Some HQ values above 1E+00 also occur for antimony, mercury, and/or thallium, although the frequency is quite low (< 0.6% of the properties). Maximum HQ values are summarized below:

COPC	Maximum HQ			
	CTE Child	RME Child	CTE Adult	RME Adult
Antimony	0.9	2	0.09	0.3
Arsenic	8	24	0.8	3
Mercury	6	17	0.6	2
Thallium	4	11	0.4	1

Combined Risks Across Chemicals

To evaluate the potential additive effects of exposure to more than one non-lead COPC in soil, the chemical-specific non-cancer HQs were summed to estimate the total non-cancer HI for each property. Based on data availability, the number of COPC HQs included in the HI calculations for each property varied from 7 to 13. The results are detailed in Appendix H and are summarized in Table 5-5. As seen, total non-cancer HIs exceed a level of potential concern (HI > 1E+00) for an RME child resident at many properties (up to 97% when data are available for 13 COPCs). These risks are attributed primarily to the ingestion of arsenic in surface soil and from the ingestion of thallium in surface soil at a few locations. Risks to CTE and RME adults are below a level of concern at most properties evaluated, with the exception of 11 properties (0.4% of the properties evaluated). This exceedence is due mainly to arsenic.

Risks Combined By Target Organ

It is important to recognize that HI values derived by summing HQ values across all chemicals is a screening-level approach that may tend to overestimate non-cancer risks. This is because not all chemicals act on the same target tissues or through the same mode of action, so not all HQ values are actually additive.

Therefore, in cases where screening-level HI values exceed 1E+00, it is appropriate to segregate HQ values according to target organ, and compute organ-specific HI values (USEPA 1989a).

Table 5-6 presents organ-specific HI values based on the critical target organs identified in Table 5-2. As seen, for most of the target organs evaluated, combined non-cancer risks are below a level of potential concern (HI ≤ 1E+00) for both CTE and RME child and adult residents. However, non-cancer effects to both the liver and skin are possible for child residents due to the ingestion of arsenic and thallium in soil at some locations, and there are a few properties where concentrations of arsenic in surface soil might pose an unacceptable health risk to the skin of an RME adult resident.

Summary of Risks from Non-Lead COPCs

Based on the evaluations described above, the frequency of properties with chemical-specific HQ values or organ-specific HI values above 1E+00 is very low except for arsenic. At some properties, the concentrations of arsenic are sufficient that, if the soil is ingested by child and/or adult residents over a long period of time, the risk of both cancer and adverse non-cancer effects would be higher than normally considered acceptable by EPA

5.5 Uncertainties

Quantitative evaluation of the risks to humans from environmental contamination is frequently limited by uncertainty regarding a number of key data items, including concentration levels in the environment, the true level of human contact with contaminated media, and the true dose-response curves for non-cancer and cancer effects in humans. This uncertainty is usually addressed by making assumptions or estimates for uncertain parameters based on whatever limited data are available. Because of these assumptions and estimates, the results of risk calculations are themselves uncertain, and it is important for risk managers and the public to keep this in mind when interpreting the results of a risk assessment. The following sections review the main sources of uncertainty in the risk calculations performed at this site.

5.5.1 Uncertainties in Exposure Assessment

As described above, the risk assessment process begins with estimation of human exposure to potentially toxic chemicals in environmental media. There are multiple sources of uncertainty in these exposure estimates, as discussed below.

Uncertainties from Exposure Pathways Not Evaluated

As discussed in Section 3, humans may be exposed to site-related chemicals by a number of pathways, but not all of these pathways were evaluated quantitatively in this risk assessment. For example, at this site, the following pathways were omitted: inhalation of dust in air and ingestion of home-grown garden vegetables.

In the case of inhalation of dust in air, omission of this pathway will result in a small underestimation of exposure and risk, but the magnitude of this underestimation is not expected to be significant. This is because the amount of soil inhaled from air is much smaller than the amount ingested.

With regard to exposure from home-grown vegetables, data are available that show that lead and cadmium are taken up from soils into vegetables (Hemphill *et al.* 1973, Finster *et al.* 2003, Samsoe-Peterson 2002). However, risk assessments at other mining, milling and smelting sites indicate that exposure from ingestion of washed garden vegetables is likely to be a minor source

of exposure compared with direct ingestion of soil. For example, a 1995 study at the Kennecott Mining site found no significant uptake of lead and arsenic into fruit or leafy and root vegetables (Life Systems 1995). At the Murray Smelter site in Utah, estimated exposure to arsenic from leafy and root vegetables, legumes, and garden fruits was two orders of magnitude less than that from soil and indoor dust (Weston 1997). Additionally, there was no statistically significant relationship between the concentration of lead in garden soil and the concentration of lead in garden vegetables collected at the Vasquez Boulevard and Interstate 70 Site in Colorado (USEPA 2001d). Based on this, it is concluded that omission of the home grown garden vegetable pathway is not likely to result in a significant underestimation of exposure and risk from metal contaminants in soil.

Uncertainties From Chemicals Not Evaluated

As discussed in Section 3.3, exposure and risk were quantified only for a selected subset (the COPCs) of chemicals detected in environmental media. In most cases, omission of other (non-COPC) chemicals is likely to be a minor source of uncertainty because the highest level of the chemical detected did not exceed a level of potential concern. For chemicals for which toxicity data are not available, (e.g., bismuth), omission of the chemical may tend to underestimate total risk, but it is not possible to estimate the magnitude of the risk.

Uncertainties in Exposure Point Concentrations

In all exposure calculations, the desired input parameter is the true mean concentration of a contaminant within a medium, averaged over the area where random exposure occurs (the exposure point). However, because the true mean cannot be calculated based on a limited set of measurements, the USEPA (1989a, 1992a) recommends that the exposure estimate for non-lead COPCs be based on the upper confidence limit (UCL) of the mean. When data are plentiful and inter-sample variability is not large, the UCL may be only slightly higher than the mean of the data. However, when data are sparse or are highly variable, the UCL may be far greater than the mean of the available data. In cases when the UCL exceeds the highest value detected, the maximum detected concentration is used. Such EPCs (substantially higher than the sample mean) reflect the substantial uncertainty that exists when data are sparse or highly variable, and in general are likely to result in an overestimate of risk.

Soil

At this site, only 1-2 samples were available for most non-lead COPCs in surface soil at residential properties, except for zinc, which usually had 4-6 soil samples per property. Because of these small data sizes, the maximum concentration was used as the EPC for soil in all cases. Use of the maximum value to characterize risk at a residence will usually tend to overestimate risk. However, when the data set is small (1-2 samples) the maximum value may not exceed the true mean, and risks might be underestimated in some cases.

Dust

For exposure to non-lead COPCs in dust, the concentration at each home was assumed to be equal to the concentration in outdoor soil at that home. Data from other mining and smelting sites suggest that concentrations of metals in indoor dust are often lower (e.g., 15-40%) than in outdoor soil (Weston 1995, 1997 and USEPA 2001d , 2002d), so this assumption is likely to result in an overestimation of risks to residents.

Uncertainties in Human Exposure Parameters

Accurate calculation of risk values requires accurate estimates of the level of human exposure that is occurring now or may occur in the future. In general, the exposure parameters were chosen in a way that was intended to be conservative or health-protective. Therefore, the values selected are thought to more likely overestimate than underestimate actual exposure and risk.

Uncertainties in Chemical Absorption (RBA)

The risk from an ingested chemical depends on how much of the ingested chemical is absorbed from the gastrointestinal tract into the body. This issue is especially important for metals in soil, because some of the metals may exist in poorly absorbable forms, and failure to account for this may result in a substantial overestimation of exposure and risk.

In the absence of data for non-lead COPCs, the approach followed in this document is to assume that the RBA is 100%. Use of this default assumption is likely to overestimate the true risk, with the magnitude of the error depending on the true RBA value. For example, studies at other sites (Weston 1997; USEPA 2001d) suggest the RBA for arsenic in soil is often about 20-40% when measured in young swine. Similarly, the RBA of arsenic in 14 soil samples from 12 different sites ranged from 0.05 to 0.31 (5-31%) when measured in monkeys (Roberts *et al.* 2007). Assuming the RBA of arsenic in soil at this site might be in the range of 10-30%, the calculations presented in this assessment could be overestimating both cancer and non-cancer risks by a factor of 3-10..

5.5.2 Uncertainties in Toxicity Values

Toxicity information for many chemicals is limited, and uncertainty in toxicity factors is one of the largest sources of uncertainty in risk estimates at a site. For example, uncertainties can arise from extrapolation from animal studies to humans, extrapolation from high dose to low dose, and extrapolation from continuous exposure to intermittent exposure. In addition, in some cases, only a few studies are available to characterize the toxicity of a chemical, and uncertainties exist not only in the dose-response curve, but also in the nature and severity of the adverse effects which the chemical may cause. For non-cancer effects, USEPA typically deals with this

uncertainty by applying an uncertainty factor of 10 - 100 to account for limitations in the database. For carcinogens, this is accounted for by using the upper bound on the slope factor. Because the USEPA seeks to be conservative in the development of both cancer and non-cancer toxicity values, risk estimates based on these values are more likely to be high than low.

In this document, non-cancer risks to child residents were quantified using chronic oral RfDs. This is conservative because the exposure duration of a child resident (6 years) is considered a subchronic exposure. Thus, using chronic RfDs to evaluate subchronic exposures may tend to overestimate non-cancer risks to child residents for some COPCs.

5.5.3 Uncertainties in Risk Characterization

A number of uncertainties occur in the risk characterization approach for cancer and non-cancer health effects.

First, because risk estimates for a chemical are derived by combining uncertain estimates of exposure and toxicity (see above), the risk estimates for each chemical are more uncertain than either the exposure estimate or the toxicity estimate alone. However, even if the risk estimates for individual chemicals were quite certain, there is considerable uncertainty in how to combine risk estimates across different chemicals. In some cases, the effects caused by one chemical do not influence the effects caused by other chemicals. In other cases, the effects of one chemical may interact with effects of other chemicals, causing responses that are approximately additive, greater than additive (synergistic), or less than additive (antagonistic). In most cases, available toxicity data are not sufficient to define what type of interaction is expected, so the USEPA generally assumes effects are additive for non-carcinogens that act on the same target tissue and for carcinogens (all target tissues). Because documented cases of synergistic interactions between chemicals are relatively uncommon, this approach is likely to be reasonable for most chemicals.

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TABLE 2-1. RESIDENTIAL-TYPE PROPERTIES IN THE FINAL FOCUS AREA

CURRENT LAND USE		NUMBER OF PROPERTIES
RESIDENTIAL[1]	single family	32,121
	multi-family	3,172
VACANT LOTS		4,012
RESIDENTIAL AND VACANT LOTS		**39,305**

[1] Includes child-care facilities.

Table 2-2. Summary of Available Soil Data

Investigation	Date	Medium	Sample Description	Analytes
PA/SI (Black and Veatch 2001)	1998 - April 2001	Surface Soil	Five-point composite samples from yards, gardens, play areas and drip zones; collected from 0-1inches bgs and sieved through number 10 (2 mm) or number 60 (250 um) mesh sieve. Composite drip zone sample, consisting of four aliquots (one from each side of the house), collected from 0-6 inches bgs and sieved through number 10 (2 mm) mesh sieve.	XRF: XRF metals ICP: TAL metals
		Subsurface Soil	Grab samples collected at every other property along the North/South/East/West sampling corridors at depths of 0-8" bgs, 8-16" bgs, and 16-24" bgs.	
RI/FS (Black and Veatch 2008)	May 2001 - September 2008	Surface Soil	Five-point composite samples from yards, gardens, play areas and drip zones; collected from 0-1inches bgs and sieved through number 10 (2 mm) or number 60 (250 um) mesh sieve. Composite drip zone sample, consisting of four aliquots (one from each side of the house), collected from 0-6 inches bgs and sieved through number 10 (2 mm) mesh sieve.	XRF: XRF metals ICP: TAL metals
Omaha Lead Site Investigatoin Background Summary Report (Jacobs 2000).	November 1999	Surface Soil	5-point composite samples collected from 0-1 inches bgs and sieved through number 10 (2mm) mesh sieve.	XRF: XRF metals ICP: TAL metals (excluding mercury (Hg)) plus bismuth (Bi)
		Subsurface Soil	Grab samples collected at 2 properties at depths 0-8" bgs (2 samples), 8-16" bgs (3 samples), and 16-24" bgs (2 samples). Sieved through a number 10 (2mm) mesh sieve.	

bulk = bulk fraction of soil (sieved to 2 mm, using a number 10 sieve).

fine = fine fraction of soil (sieved to 250 µm using a number 60 sieve).

XRF = X-Ray Fluorescence

ICP-AES = Inductively Coupled Plasma Atomic Emission Spectroscopy

TAL (Target Analyte List) Metals: aluminum (Al), antimony (Sb), arsenic (As), barium (Ba), beryllium (Be), cadmium (Cd), calcium (Ca), chromium (Cr), cobalt (Co), copper (Cu), iron (Fe), lead (Pb), magnesium (Mg), manganese (Mn), mercury (Hg), nickel (Ni), potassium (K), selenium (Se), silver (Ag), sodium (Na), thallium (Th), vanadium (V), zinc (Zn).

XRF Metals: antimony (Sb), arsenic (As), cadmium (Cd), chromium (Cr), cobalt (Co) copper (Cu), iron (Fe), lead (Pb), manganese (Mn), mercury (Hg), molybdenum (Mo), nickel (Ni), rubidium (Rb), selenium (Se), silver (Ag), strontium (Sr), tin (Sn), titanium (Ti), uranium (U), Zinc (Zn), zirconium (Zr).

Table 2-3. Summary of Available Indoor Dust, Tap Water and Ambient Air Data

Data Type	Investigation	Medium	Date	Number of Stations	Number of Field Samples	Sample Description	Analyses	Analytes	sieved
Indoor Dust (Paired with outdoor soil)	Remedial Investigation (Black and Veatch 2004a and 2004b)	Surface Soil	December 2003 - January 2004	159	636	Five-point composite samples from yards, gardens, play areas and drip zones; collected from 0-1 inches bgs and sieved through number 10 (2 mm) or number 60 (250 µm) mesh sieve. Composite drip zone sample, consisting of four aliquots (one from each side of the house), collected from 0-6 inches bgs and sieved through number 10 (2 mm) mesh sieve.	XRF - 100% (159)	XRF: XRF Metals	bulk - 100% (636)
		Indoor Dust	December 2003 - January 2004	159	475	Up to 3 samples collected at each property from entryways, living areas and windowsills. Samples sieved using a number 100 mesh screen (150 µm).	ICP - 100% (477)	Lead (Pb)	fine[1] - 100% (477)
	Omaha Lead Site OU2 Field Investigation (Black and Veatch 2007)	Surface Soil	October - December 2007	98	98	One 20-point composite surface soil (0-1 inches bgs) sample was collected from each property, collected by dividing up the property into 4 quadrants and collecting five grab samples from each quadrant (e.g., from the four corners and a center of each quadrant). Samples were sieved using a number 60 mesh screen (250 µm).	ICP - 100% (98)	23 TAL Metals, with the addition of titanium, silicon and zirconium	fine - 100% (98)
		Indoor Dust	October - December 2007	98	98	One composite dust sample was collected from each property, comprised of dust collected from 3 locations in the home: entryway, living area and bedroom. Samples were sieved using a number 60 mesh screen (250 µm).	ICP - 100% (98)	23 TAL Metals, with the addition of titanium, silicon and zirconium	fine - 100% (98)
Tap Water	Omaha Lead Site OU2 Field Investigation (Black and Veatch 2007)	Tap Water	December 2007	77	154	Grab samples from residential taps. Two grabs were collected at each residence: a first-flush sample and a post-flush sample.	ICP - 100%	Lead (Pb)	--
Ambient Air	USEPA Air Quality Database	Ambient Air	1997 - 2002	11	24	Mean ambient air concentration in total suspended particulates recorded at a monitoring station.	NA	Lead (Pb)	--

-- = Not applicable.
bulk = bulk fraction of soil (sieved to 2 mm, using a number 10 sieve).
fine = fine fraction of soil (sieved to 250 µm using a number 60 sieve).
First - flush = tap water sample collected first thing in the morning, after water has set in pipes overnight.
ICP-AES = Inductively Coupled Plasma Atomic Emission Spectroscopy
NA = Not available.
Post-flush = tap water sample collected after runing the tap for at least 5 minutes.
XRF = X-Ray Florescence

TAL (Target Analyte List) Metals: aluminum (Al), antimony (Sb), arsenic (As), barium (Ba), beryllium (Be), cadmium (Cd), calcium (Ca), chromium (Cr), cobalt (Co), copper (Cu), iron (Fe), lead (Pb), magnesium (Mg), manganese (Mn), mercury (Hg), nickel (Ni), potassium (K), selenium (Se), silver (Ag), sodium (Na), thallium (Th), vanadium (V), zinc (Zn).

XRF Metals: antimony (Sb), arsenic (As), cadmium (Cd), chromium (Cr), cobalt (Co) copper (Cu), iron (Fe), lead (Pb), manganese (Mn), mercury (Hg), molybdenum (Mo), nickel (Ni), rubidium (Rb), selenium (Se), silver (Ag), strontium (Sr), tin (Sn), titanium (Ti), uranium (U), Zinc (Zn), zirconium (Zr).

[1] Fine fraction in this investigation is < 150 µm.

Table 2-4. Summary Statistics for Metals in Residential Soil

ANALYTE	SOIL CONCENTRATION (mg/kg)[1]				
	Detection Frequency		Minimum	Maximum	Average
ANTIMONY	320 / 89,605	0.4%	9	2,142	45
ARSENIC	56,882 / 162,903	35%	1.9	4,107	21
BARIUM	14,563 / 89,580	16%	66.3	6,896	332
CADMIUM	6,351 / 89,838	7%	2	1,057	24
CHROMIUM	2,951 / 116,450	3%	4.9	1,602,400	154
COBALT	38,764 / 111,330	35%	23	8,867	162
COPPER	55,200 / 111,243	50%	1.7	17,783	33
IRON	111,894 / 111,899	100%	1,155	2,211,200	19,982
LEAD	166,661 / 168,076	99%	3	54,219	337
MANGANESE	95,095 / 111,895	85%	30	103,900	497
MERCURY	2,136 / 116,493	2%	1	6,746	9
MOLYBDENUM	5,180 / 107,386	5%	1	17,245	5
NICKEL	19,107 / 110,518	17%	7	599,650	77
RUBIDIUM	111,706 / 111,878	100%	4	415	72
SELENIUM	225 / 116,112	0.2%	1	833	3
SILVER	104 / 89,600	0.1%	2.4	2,718	21
STRONTIUM	111,861 / 111,878	100%	5	3,224	138
TIN	18,687 / 89,526	21%	0.3	1,604.0	50
TITANIUM	89,845 / 90,250	100%	155.7	44,045	3,090
URANIUM	19 / 2,226	1%	4	68	14
ZINC	160,422 / 163,401	98%	6.6	4,444,444	436.1
ZIRCONIUM	111,881 / 111,896	100%	4	3,240	251

[1] Results for samples analyzed by XRF are presented. Concentrations qualified as not detected were adjusted to 1/2 of the quantitation limit.

Table 2-5. Summary Statistics for Metals in Background Soil

ANALYTE	SOIL CONCENTRATION (mg/kg)[1]																			
	Surface Soil (0-2" bgs)					Subsurface Soil (0-8" bgs)					Subsurface Soil (8-16" bgs)					Subsurface Soil (16-24" bgs)				
	DF		Min	Max	Avg	DF		Min	Max	Avg	DF		Min	Max	Avg	DF		Min	Max	Avg
Aluminum	22/22	100%	2,600	6,200	4,682	2/2	100%	2,600	6,100	4,350	2/2	100%	2,800	5,100	3,950	3/3	100%	2,600	5,300	3,767
Antimony	2/22	9%	3.0	6.8	3.3	0/2	0%	3	3	3	0/2	0%	3.0	3.0	3.0	0/3	0%	3.0	3.0	3.0
Arsenic	22/22	100%	3.1	11	7.2	2/2	100%	8	11	9	2/2	100%	6.8	8.2	7.5	3/3	100%	6.5	8.6	7.3
Barium	22/22	100%	110	190	153	2/2	100%	130	160	145	2/2	100%	130	160	145	3/3	100%	130	170	147
Beryllium	7/22	32%	0.3	0.6	0.3	1/2	50%	0.3	0.6	0.4	1/2	50%	0.3	0.5	0.4	1/3	33%	0.3	0.6	0.4
Bismuth	0/22	0%	0.5	0.5	0.5	0/2	0%	0.5	0.5	0.5	0/2	0%	0.5	0.5	0.5	0/3	0%	0.5	0.5	0.5
Cadmium	20/22	91%	0.3	1.3	0.7	1/2	50%	0.3	0.8	0.5	1/2	50%	0.3	0.8	0.5	3/3	100%	0.5	0.8	0.7
Calcium	22/22	100%	2,700	68,000	12,473	2/2	100%	11,000	12,000	11,500	2/2	100%	8,200	11,000	9,600	3/3	100%	7,900	15,000	10,967
Chromium	22/22	100%	5	17	9	2/2	100%	5.4	10	7.7	2/2	100%	5.4	15	10	3/3	100%	5.3	15.0	8.9
Cobalt	22/22	100%	4	8	6	2/2	100%	5.4	7.3	6.4	2/2	100%	5.1	6.6	5.8	3/3	100%	5.1	6.9	5.7
Copper	22/22	100%	6	18	11	2/2	100%	7.1	19	13	2/2	100%	8.5	15	12	3/3	100%	7.6	15	11
Iron	22/22	100%	6,000	12,000	9,095	2/2	100%	7,400	12,000	9,700	2/2	100%	7,400	10,000	8,700	3/3	100%	6,900	11,000	8,733
Lead	22/22	100%	12	40	20	2/2	100%	10	16	13	2/2	100%	11	17	14	3/3	100%	15	19	17
Magnesium	22/22	100%	1,600	6,300	3,700	2/2	100%	4,400	4,500	4,450	2/2	100%	3,700	4,400	4,050	3/3	100%	3,600	5,100	4,333
Manganese	22/22	100%	250	620	487	2/2	100%	330	590	460	2/2	100%	350	510	430	3/3	100%	330	560	417
Nickel	22/22	100%	7.6	14	11	2/2	100%	10	14	12	2/2	100%	10	13	11	3/3	100%	10	13	11
Potassium	22/22	100%	1,300	2,700	2,014	2/2	100%	1,300	2,500	1,900	2/2	100%	1,300	2,100	1,700	3/3	100%	1,400	2,200	1,700
Selenium	16/22	73%	0.3	2.6	0.7	2/2	100%	1.0	1.0	1.0	2/2	100%	0.8	1.2	1.0	2/3	67%	0.3	0.7	0.5
Silver	0/22	0%	0.5	0.5	0.5	0/2	0%	0.5	0.5	0.5	0/2	0%	0.5	0.5	0.5	0/3	0%	0.5	0.5	0.5
Sodium	9/22	41%	50	220	102	0/2	0%	50	50	50	0/2	0%	50	50	50	0/3	0%	50	50	50
Thallium	0/22	0%	0.3	0.3	0.3	0/2	0%	0.3	0.3	0.3	0/2	0%	0.3	0.3	0.3	0/3	0%	0.3	0.3	0.3
Vanadium	22/22	100%	10	19	14	2/2	100%	11	19	15	2/2	100%	11	17	14	3/3	100%	10	17	13
Zinc	22/22	100%	38	68	53	2/2	100%	36	67	52	2/2	100%	37	59	48	3/3	100%	37	62	48

[1] Results for samples analyzed by ICP are presented. Concentrations qualified as not detected were adjusted to 1/2 of the quantitation limit.

Avg = Arithmetic average concentraiton

DF = Detection Frequency

Max = Maximum concentration

Min = Minimum concentration

Table 2-6. Summary Statistics for Metals in Indoor Dust

ANALYTE	INDOOR DUST CONCENTRATION (mg/kg)[1]				
	Detection Frequency		Minimum	Maximum	Average
Aluminum	98/98	100%	1,120	23,900	4,874
Antimony	13/98	13%	3.0	226.0	10.0
Arsenic	92/98	94%	0.5	47	7.4
Barium	98/98	100%	29	814	226
Beryllium	1/98	1%	0.3	0.8	0.3
Cadmium	95/98	97%	0.3	27.7	2.9
Calcium	98/98	100%	6,730	245,000	35,548
Chromium	98/98	100%	6.6	94	29
Cobalt	27/98	28%	2.5	13	3.7
Copper	98/98	100%	24	1,110	144
Iron	98/98	100%	770	20,400	7,530
Lead	569/573	99%	1.0	15,900	427
Magnesium	98/98	100%	800	13,700	4,098
Manganese	98/98	100%	20	406	199
Mercury	98/98	100%	0.02	5	0.7
Nickel	98/98	100%	4.7	197	26
Potassium	98/98	100%	1,230	214,000	7,497
Selenium	2/98	2%	1.8	5.3	1.8
Silicon	98/98	100%	432	2,240	1,047
Silver	26/52	50%	0.5	14	2.7
Sodium	98/98	100%	1,370	466,000	45,675
Thallium	0/98	0%	1.3	1.3	1.3
Titanium	98/98	100%	20	1,090	119
Vanadium	88/98	90%	2.5	47	11
Zinc	98/98	100%	103	2,980	718
Zirconium	44/98	45%	42	1,130	137

[1] Results for samples analyzed by ICP are presented. Concentrations qualified as not detected were adjusted to 1/2 of the quantitation limit.

Table 2-7. Lead Concentrations in Tap Water

Data Set	Metric	Results	
		First Flush	Post Flush
ALL DATA [1]	Total Number of Samples	97	97
	Minimum Concentration (µg/L)	0.5	0.5
	Maximum Concentration (µg/L)	48.8	634
	Average Concentration (µg/L)	1.9	7.7
	Average Non-Detect Concentration (µg/L)	0.51	0.50
	Detection Frequency	26%	26%
	Number of Samples > Action Level (15 µg/L)	1 (1% of samples)	1 (1% of samples)
POTENTIAL OUTLIER REPLACED [2]	Total Number of Samples	97	97
	Minimum Concentration (µg/L)	0.5	0.5
	Maximum Concentration (µg/L)	13.4	7.42
	Average Concentration (µg/L)	1.5	1.2
	Average Non-Detect Concentration (µg/L)	0.51	0.50
	Detection Frequency	25%	25%
	Number of Samples > Action Level (15 µg/L)	0	0

[1] Original results from Fall 2007 for property ID 29349 were included. Results from the resampling event for this property are not included.
[2] Original results from Fall 2007 results form property ID 29349: First Flush = 48.8 µg/L; Post Flush = 634 µg/L. were not included and were replaced with results from a resampling event (April 2008).

Table 2-8. Air Monitoring Stations in Douglas County, Nebraska

MONITORING STATION	LOCATION	LEAD MONITORING DATES
0011	11th and Nicholas	1982 - 2000
0019	42nd and Woolworth	1982 - 1993
0030	1723 Harney	1977
0034	19th Ave and Howard	1977 -1988
0038	7759 Dodge	1982 -1983
0040	7717 Dodge	1984 - 1997
0041	4th and Jones	1985 - 2000
0042	700 Abbott Drive	1985 - 2002
0045	46th and Farnam	--
0046	Riverfront	1990 - 2002
0047	2300 South 26th Street	1992 - 1995
0049	Union Pacific	1995 - 2000
1001	18th and Dodge	1959 - 1962; 1975; 1965 - 1975

-- = Lead not measured at this station.

Source: USEPA Air Quality System database, Accessed 6/2008.

Table 2-9. Concentrations of Lead in Ambient Air
at 11 Monitoring Stations Located at the Omaha Lead Site

YEAR	AVERAGE LEAD CONCENTRATION (µg/m3)											
	0011	0019	0030	0034	0041	0042	0045	0046	0047	0049	1001	ALL
1997	0.057	--	--	--	0.054	0.319	--	0.433	--	2.271	--	0.524
1998	0.02	--	--	--	0.012	0.076	--	0.028	--	0.346	--	0.096
1999	0.026	--	--	--	0.022	0.169	--	0.172	--	0.893	--	0.256
2000	0.031	--	--	--	0.012	0.05	--	0.024	--	0.087	--	0.041
2001	--	--	--	--	--	0.041	--	0.031	--	--	--	0.036
2002	--	--	--	--	--	0.027	--	0.018	--	--	--	0.023
2000 - 2002 AVERAGE												**0.036**

Source: USEPA Air Quality System database, accessed 6/2008.

-- = Lead not measured at this station during this reporting period.

Table 3-1. Summary of Quantitative COPCs in Surface Soil

CHEMICALS OF POTENTIAL CONCERN	
Aluminum	Iron
Antimony	Lead
Arsenic	Manganese
Cadmium	Mercury
Chromium	Thallium
Cobalt	Vanadium
Copper	Zinc

Table 4-1 IEUBK Model Inputs

A. AGE-INDEPENDENT MODEL INPUTS:

PARAMETER	VALUE	BASIS
Soil concentration (mg/kg)	Property-specific	Yard-wide average concentration[1] (excluding drip zone samples)
Indoor dust concentration (mg/kg)	Property-specific	Calculated using site-specific M_{sd} equation: $C_{dust} = 42 + 0.74 \bullet C_{soil}$
Air concentration (µg/m^3)	0.036	Average concentration in air at the Site (2000 – 2002) (USEPA 2008b)
Indoor air concentration (ug/m^3)	30% of outdoors	USEPA (1994a) default
Drinking water concentration (µg/L)	1.36	Average concentration in tap water at the Site. Assumes water consumed is 50% first draw and 50% post flush
Absorption Fractions: Air Diet Water Soil Dust	 32% 50% 50% 40% 40%	 USEPA (1994a) default USEPA (1994a) default USEPA (1994a) default Site-specific value Site-specific value
Fraction soil	45%	USEPA (1994a) default
GSD	1.6	USEPA (1994a) default

[1] Fine fraction, ICP-equivalent concentration of lead in soil (see Section 3.4.1):
Fine ICP = Coarse XRF · 1.16 · 1.04

B. AGE-DEPENDENT MODEL INPUTS:

	AIR		DIET	WATER	SOIL
Age	Time Outdoors (hrs)	Ventilation Rate (m^3/day)	Dietary Intake [2] (µg/day)	Intake (L/day)	Intake (mg/day)
0-1	1.0	2.0	2.26	0.20	85
1-2	2.0	3.0	1.96	0.50	135
2-3	3.0	5.0	2.13	0.52	135
3-4	4.0	5.0	2.04	0.53	135
4-5	4.0	5.0	1.95	0.55	100
5-6	4.0	7.0	2.05	0.58	90
6-7	4.0	7.0	2.22	0.59	85

[2] Revised USEPA (2008a) recommended dietary intake parameters, based on updated dietary lead estimates from the Food and Drug Administration's Total Diet Study (FDA 2006) and food consumption data from NHANES III (CDC 1997) .

Table 4-2. Summary of Lead Relative Bioavailability (RBA) Data

Medium	Estimation Method	Reference	Date	Number of Field Samples	Sample Description	Soil Fraction
Surface Soil	*In Vivo*	Casteel *et al.* (2004)	2004	2	Two test materials consisted of 2 surface soil samples collected from residential yards at the site	fine
	IVBA	Drexler (2004)	2002	28	Composite surface soil samples collected from residential yards	fine
			2004	19	Composite surface soil samples collected from residential yards	fine
		Drexler (2008a)	2004	2	Samples consisted of the 2 *in vivo* test materials evaluated by Casteel *et al.* (2004)	fine
Indoor Dust	IVBA	Drexler (2008b)	2008	94	Composite indoor dust samples collected in 2007	fine

Table 4-3. *In Vivo* RBA Results for Lead in Soil

A. ORIGINAL RESULTS

Measurement Endpoint	Estimated Soil RBA	
	Test Material 1	Test Material 2
Blood Lead AUC	1.01	0.76
Liver Lead	1.07	0.74
Kidney Lead	1.04	0.66
Bone Lead	0.98	0.64

Uncertainty Range	Estimated Soil RBA	
	Test Material 1	Test Material 2
Plausible Range	1.01 - 1.03	0.68 - 0.76
Preferred Range	1.01 - 1.02	0.72 - 0.76
Suggested Point Estimate	1.01	0.74

Source: Black and Veatch (2004c)

B. REANALYSIS USING EPA 2007 METHOD

Measurement Endpoint	Estimated Soil RBA (90% Confidence Interval)	
	Test Material 1	Test Material 2
Blood Lead AUC	1.04 (0.58 - 1.43)	0.87 (0.57 - 1.24)
Liver Lead (a)	0.91 (0.62 - 1.36)	0.82 (0.55 - 1.22)
Kidney Lead	0.83 (0.62 - 1.10)	0.75 (0.56 - 1.00)
Bone Lead	1.07 (0.85 - 1.37)	0.89 (0.70 - 1.15)
Combined	0.96 (0.64 - 1.29)	0.83 (0.58 - 1.10)

(a) Results exclude one outlier data point

Table 4-4. IVBA Results for Lead in Soil

A. IVBA AND ESTIMATED RBA RESULTS:

Index	Sample ID	Concentration (Fine Fraction) (mg/kg) [2]	IVBA [1]	Estimated RBA [1,2]	Outlier? [3]
1	5083 DZ	2,568	90%	76%	
2	5083 B1	6,193	34%	27%	Yes
3	5082 B1	695	104%	89%	
4	5080 B2	480	96%	82%	
5	5048 B2	498	78%	66%	
6	5079 B2	1,717	89%	76%	
7	5081 F2	796	76%	64%	
8	5046 DZ	2,785	63%	53%	
9	5046 B1	399	78%	66%	
10	5063 F2	1,524	68%	57%	
11	5061 F2	503	93%	79%	
12	5058 F2	490	100%	85%	
13	5044 B1	419	86%	73%	
14	5055 F1	669	77%	65%	
15	5088 F2	746	69%	58%	
16	5034 B1	514	87%	74%	
17	5017 F1	1,040	78%	66%	
18	5086 B1	787	112%	95%	
19	5087 G	1,188	66%	55%	
20	5060 B2	2,214	78%	66%	
21	5020 B2	941	68%	57%	
22	5098 B2	58	70%	59%	
23	5008 B1	524	108%	92%	
24	5056 F2	2,251	35%	28%	Yes
25	5030 B1	133	75%	63%	
26	5007 F2	534	85%	72%	
27	5059 B1	743	99%	84%	
28	5041 B1	134	82%	69%	
29	13271	1,303	93%	79%	
30	18343	664	77%	65%	
31	22258	858	74%	62%	
32	22412	1,172	96%	82%	
33	29478	259	77%	65%	
34	33449	1,115	89%	75%	
35	34544	1,433	84%	71%	
36	36276	206	73%	61%	
37	37666	697	80%	67%	
38	38573	1,362	77%	65%	
39	38775	861	78%	66%	
40	40182	498	70%	59%	
41	40299	930	80%	67%	
42	41449	964	87%	73%	
43	41488	466	82%	69%	
44	44481	85	94%	80%	
45	44837	510	84%	71%	
46	47483	605	79%	67%	
47	47618 [4]	385	75%	63%	

Source: Drexler (2004 and 2008a).

IVBA = *In vitro* bioaccessibility

RBA = Relative bioavailability

B. SUMMARY STATISTICS:

DATA SET	N	MEAN (RANGE)	
		IVBA	RBA
All	47	81% (34% - 112%)	68% (27% - 95%)
Outliers Excluded	45	83% (63% - 112%)	70% (53% - 95%)

[1] Note: The values shown in this table are different than originally reported in Drexler (2004). This is because the 2004 report was published before the ICP results for the concentrations of lead in the fine fraction (<250 µm) were available, so the initial IVBA estimates were based on XRF estimates of lead in the bulk fraction (<2 mm) (Drexler 2008a, personal communication).

[2] RBA estimated in accord with USEPA (2007a): $[RBA] = 0.878 \cdot [IVBA] - 0.028$

[3] Identified using Rosner's Test for Outliers (Gilbert 1987).

[4] Corrected value, based on personal communication with Drexler (2008a).

Table 4-5.
In Vivo and In Vitro RBA Estimates for Lead in Soil

Soil	Concentration (mg/kg)	IVBA mean (range)	RBA point estimate (range)	
			estimated from IVBA	measured *In Vivo*
Test Material 1	2,003	0.90 (0.88 - 0.92)	0.76 (0.74 - 0.78)	0.96 (0.64 - 1.29)
Test Material 2	1,613	0.84 (0.83 - 0.85)	0.71 (0.70 - 0.72)	0.83 (0.58 - 1.10)

Table 4-6. IVBA Results for Lead in Indoor Dust

A. IVBA AND ESTIMATED RBA RESULTS:

Index	Sample ID	Concentration (fine fraction) (mg/kg)	IVBA	RBA [1]	Outlier? [2]
1	3662-265	18.1	40%	32%	Yes
2	3662-275	259	64%	53%	
3	3662-254	65.8	66%	55%	
4	3662-225	67.3	67%	56%	
5	3662-236	270	67%	56%	
6	3662-205	233	68%	57%	
7	3662-235	159	68%	57%	
8	3662-249	124	70%	58%	
9	3662-213	499	71%	59%	
10	3662-215	58.3	71%	59%	
11	3662-206	331	71%	60%	
12	3662-284	152	72%	60%	
13	3662-204	246	72%	60%	
14	3662-289	634	73%	61%	
15	3662-201	32	73%	61%	
16	3662-216	56.2	73%	62%	
17	3662-269	33.9	74%	62%	
18	3662-203	101	75%	63%	
19	3662-202	158	75%	63%	
20	3662-252	106	77%	65%	
21	3662-288	151	78%	66%	
22	3662-246	178	79%	67%	
23	3662-238	206	79%	67%	
24	3662-285	930	80%	68%	
25	3662-272	144	80%	68%	
26	3662-214	224	81%	68%	
27	3662-207	174	81%	68%	
28	3662-295	13.2	81%	68%	
29	3662-264	354	83%	70%	
30	3662-292	374	83%	70%	
31	3662-240	319	83%	70%	
32	3662-274	159	84%	71%	
33	3662-227	176	84%	71%	
34	3662-296	784	85%	72%	
35	3662-293	153	85%	72%	
36	3662-217	3180	85%	72%	
37	3662-221	755	85%	72%	
38	3662-244	146	86%	73%	
39	3662-267	406	86%	73%	
40	3662-211	254	86%	73%	
41	3662-233	538	87%	73%	
42	3662-291	591	87%	73%	
43	3662-232	530	87%	74%	
44	3662-268	410	87%	74%	
45	3662-237	64.2	87%	74%	
46	3662-220	261	88%	74%	
47	3662-299	249	88%	74%	
48	3662-298	230	88%	74%	
49	3662-255	85.9	88%	75%	
50	3662-270	829	88%	75%	
51	3662-208	194	88%	75%	
52	3662-239	202	88%	75%	
53	3662-302	177	89%	75%	
54	3662-231	246	89%	75%	
55	3662-229	181	89%	75%	
56	3662-241	705	89%	75%	
57	3662-210	206	90%	76%	
58	3662-222	354	90%	76%	
59	3662-247	264	90%	76%	
60	3662-245	35.5	90%	76%	

Table 4-6. IVBA Results for Lead in Indoor Dust

(Continued)

A. IVBA AND ESTIMATED RBA RESULTS:

Index	Sample ID	Concentration (fine fraction) (mg/kg)	IVBA	RBA [1]	Outlier? [2]
61	3662-223	74.8	90%	77%	
62	3662-287	242	91%	77%	
63	3662-209	3810	91%	77%	
64	3662-234	304	92%	78%	
65	3662-224	281	92%	78%	
66	3662-219	171	92%	78%	
67	3662-266	2720	93%	79%	
68	3662-263	44.9	94%	80%	
69	3662-282	420	94%	80%	
70	3662-262	441	94%	80%	
71	3662-297	271	95%	80%	
72	3662-294	446	95%	80%	
73	3662-290	374	95%	80%	
74	3662-228	216	95%	80%	
75	3662-283	527	95%	81%	
76	3662-261	106	96%	81%	
77	3662-251	306	96%	82%	
78	3662-242	527	96%	82%	
79	3662-212	200	97%	82%	
80	3662-243	187	99%	84%	
81	3662-257	193	100%	85%	
82	3662-277	259	100%	85%	
83	3662-218	9.44	102%	87%	
84	3662-301	686	102%	87%	
85	3662-281	128	103%	87%	
86	3662-300	69.1	103%	88%	
87	3662-258	94.6	103%	88%	
88	3662-276	372	109%	93%	
89	3662-250	98.4	113%	96%	
90	3662-260	758	115%	98%	
91	3662-259	1120	117%	100%	
92	3662-230	910	180%	155%	Yes
93	3662-253	289	261%	227%	Yes
94	3662-226	79.5	509%	444%	Yes

B. SUMMARY STATISTICS:

DATA SET	N	MEAN (RANGE)	
		IVBA	RBA
All	94	94% (40% - 509%)	79% (32% - 444%)
Outliers Excluded	90	87% (64% - 117%)	73% (53% - 100%)

[1] RBA estimated in accord with USEPA (2007): $[RBA] = 0.878 \cdot [IVBA] - 0.028$

[2] Identified using Rosner's Test for Outliers (Gilbert 1987).

IVBA = *In vitro* bioaccessibility

RBA = Relative bioavailability

Table 4-7. Summary of Risks to Residents from Exposure to Lead in Surface Soil

ZIP CODE	NUMBER OF PROPERTIES EVALUATED	ESTIMATED NUMBER AND PERCENT OF PROPERTIES WITHIN THE SPECIFIED P10 RANGE					TOTAL NUMBER AND PERCENT OF PROPERTIES WHERE P10 EXCEEDS 5%
		≤ 5%	>5% to ≤10%	>10% to ≤20%	>20% to ≤50%	>50%	
68102	71	45	6	9	11	0	26
		63%	8%	13%	15%	0%	37%
68104	27	21	3	3	0	0	6
		78%	11%	11%	0%	0%	22%
68105	4,953	3,585	518	463	343	44	1,368
		72%	10%	9%	7%	1%	28%
68106	165	147	7	7	3	1	18
		89%	4%	4%	2%	1%	11%
68107	7,069	5,762	587	424	258	38	1,307
		82%	8%	6%	4%	1%	18%
68108	3,324	1,468	499	653	625	79	1,856
		44%	15%	20%	19%	2%	56%
68110	2,170	1,011	332	384	381	62	1,159
		47%	15%	18%	18%	3%	53%
68111	7,295	5,422	766	634	421	52	1,873
		74%	11%	9%	6%	1%	26%
68112	162	115	17	20	8	2	47
		71%	10%	12%	5%	1%	29%
68117	2	2	0	0	0	0	0
		100%	0%	0%	0%	0%	0%
68131	1,955	1,019	286	312	269	69	936
		52%	15%	16%	14%	4%	48%
68132	1,285	848	156	142	110	29	437
		66%	12%	11%	9%	2%	34%
ALL	28,478	19,445	3,177	3,051	2,429	376	9,033
		68%	11%	11%	9%	1%	32%

P10 = Probability of exceeding a blood lead value of 10 µg/dL (%).

Table 5-1. Exposure Parameters for Residents

Exposure Pathway	Exposure Parameter	Units	CTE				RME			
			Adult	Source	Child	Source	Adult	Source	Child	Source
General	Body Weight	kg	70	[1]	15	[1]	70	[1]	15	[1]
	Exposure frequency	days/yr	245	[3, a]	245	[3, a]	350	[1]	350	[1]
	Exposure duration	years	7	[3, 4, d]	2	[3, 4, d]	24	[1]	6	[1]
	Averaging Time, Cancer	days	25,550	[2, c]	25,550	[2, c]	25,550	[2, c]	25,550	[2, c]
	Averaging Time, Noncancer	days	2,555	[2, c]	730	[2, c]	8,760	[2, c]	2,190	[2, c]
Ingestion of Soil and Indoor Dust	Ingestion rate	mg/day	50	[3]	100	[3, b]	100	[1]	200	[1]
	Conversion factor	kg/mg	1.00E-06	--	1.00E-06	--	1.00E-06	--	1.00E-06	--
Dermal Exposure to Soil	Exposed Surface Area (SA)	cm^2/event	5,700	[5]	2,800	[5]	5,700	[5]	2,800	[5]
	Adherence Factor (AF)	mg/cm^2	0.01	[5]	0.04	[5]	0.07	[5]	0.2	[5]
	Event Frequency	events/day	1	[5]	1	[5]	1	[5]	1	[5]
	Dermal Absorption Fraction (ABS_d)	unitless	CS	[5, e]	CS	[5, e]	CS	[5, e]	CS	[5, e]
	Conversion factor	kg/mg	1.00E-06	--	1.00E-06	--	1.00E-06	--	1.00E-06	--

CTE = Central Tendency Exposure
RME = Reasonable Maximum Exposure

Sources:

[1] USEPA 1991a. Human Health Evaluation Manual, Supplemental Guidance: Standard Default Exposure Factors. Office of Emergency and Remedial Response. Washington, DC. OSWER Directive 9285.6-03. March.

[2] USEPA 1989. Risk Assessment Guidance for Superfund, Volume I, Human Health Evaluation Manual (Part A). Office of Emergency and Remedial Response, Washington, D.C. EPA/540/1-89/002. December.

[3] USEPA 1997a. Exposure Factors Handbook.

[4] Professional judgment.

[5] USEPA 2004a. Risk Assessment Guidance for Superfund, Volume 1, Human Health Evaluation Manual (Part E). Office of Solid Waste and Emergency Response, Washington, D.C. EPA/540/R/99/005. July.

Notes:

[a] Estimated by taking the percent of time men and women spend at home (67% of time spent at home, see Table 15A-3) and multiplying by 365 days (0.67*365 days = 245 days).

[b] Table 4-23. Mean recommended values for soil ingestion.

[c] Averaging time expressed as days. Noncancer averaging time calculated by multiplying the exposure duration in years by 365 days/year and cancer averaging time calculated by multiplying a 70 year lifetime for cancer effects by 365 days/year.

[d] Table 15-176. Average residency/occupancy of a home (9 years). Assumes 2 years as a child and 7 years as an adult.

[e] Dermal absorption fraction values were available for arsenic and cadmium, only. The values used in the risk assessment are as follows:

cadmium = 0.001; all other chemicals = not evaluated.

arsenic = 0.03,

Table 5-2. Oral and Dermal Human Health Toxicity Values

CHEMICAL	INGESTION				DERMAL						Primary Target Organ (noncancer effects)	Weight of Evidence (cancer)
	SFo $(mg/kg\text{-}day)^{-1}$	Source	RfDo mg/kg-day	Source	Absorption Fraction [8]	Adjust? [8]	SF_{ABS} $(mg/kg\text{-}day)^{-1}$	Source	RfD_{ABS} mg/kg-day [9]	Source		
Aluminum	--	--	1.0E+00	P [1]	NA	NA	--	--	1.0E+00	--	CNS	NA
Antimony	--	--	4.0E-04	I	0.15	Yes	--	--	6.0E-05	I [9]	Liver	NA
Arsenic	1.5E+00	I	3.0E-04	I	0.95	No	1.5E+00	I [9]	3.0E-04	I [9]	Skin	A
Cadmium	--	--	1.0E-03	I	0.025	Yes	--	--	2.5E-05	I [9]	Kidney	NA
Chromium III	--	--	1.5E+00	I	0.013	Yes	--	--	2.0E-02	I [9]	No Effect	NA
Chromium VI	--	--	3.0E-03	I	0.025	Yes	--	--	7.5E-05	I [9]	GI	NA
Cobalt	--	--	3.0E-04	P [10]	NA	NA	--	--	3.0E-04	P [10]	Thyroid	NA
Copper	--	--	4.0E-02	H [2]	NA	NA	--	--	4.0E-02	H [2, 9]	GI	NA
Iron	--	--	7.0E-01	P [3]	NA	NA	--	--	7.0E-01	P [3, 9]	GI	NA
Manganese	--	--	4.7E-02	I [4]	0.04	Yes	--	--	1.9E-03	I [4, 9]	CNS	NA
Mercury	--	--	3.0E-04	I [5]	0.07	Yes	--	--	2.1E-05	I [5, 9]	Kidney	NA
Thallium	--	--	6.5E-05	I [2, 6]	NA	NA	--	--	6.5E-05	I [2, 6, 9]	Liver	NA
Vanadium	--	--	5.0E-03	I [2, 7]	0.026	Yes	--	--	1.3E-04	I [2, 7, 9]	Kidney	NA
Zinc	--	--	3.0E-01	I	variable	No	--	--	3.0E-01	I [9]	Blood	NA

-- = A USEPA Recommended toxicity value is not available for this chemical

Abbreviations:

A = Human carcinogen
CNS = Central Nervous System
GI = Gastrointestinal System
H = HEAST
I = IRIS
NA = not available
P = EPA Provisional Peer-Reviewed Value
RfD_{ABS} = Absorbed Reference Dose
RfDo = Oral Noncancer Reference Dose
SF_{ABS} = Absorbed Slope Factor
SFo = Oral Cancer Slope Factor

Notes:

[1] Provisional Peer Reviewed Toxicity Values for Aluminum (October 2006).
[2] As cited in Regional Screening Level Tables (9/2008 update):
http://www.epa.gov/reg3hwmd/risk/human/rb-concentration_table/Generic_Tables/index.htm
[3] Provisional Peer Reviewed Toxicity Values for Iron and Compounds (September 2006).
[4] RfDo (1.4E-01 mg/kg-day) adjusted by a modifying factor of 3, in accord with IRIS recommendations.
[5] Toxicity data for mercuric chloride.
[6] RfDo for thallium sulfate (8E-05) adjusted by multiplying by 81% (the percent molecular weight of thallium in thallium sulfate) in order to factor out the molecular weight (MW) of the sulfate ion.
[7] RfDo for vanadium pentoxide (9E-03) adjusted by multiplying by the percent molecular weight of vanadium in vanadium pentoxide in order to factor out the molecurlar weight of the oxide ion.
[8] USEPA 2004. Supplemental Guidance for Dermal Risk Assessment. Exhibit 4-1.
[9] Derivation based on absorption fraction, in accord with USEPA 2004
[10] Provisional Peer Reviewed Toxicity Values for Cobalt (August 2008).

Table 5-3. Summary of Total Cancer Risks to Residents (Children and Adults) from Ingestion and Dermal Exposure to Arsenic in Surface Soil

ZIP CODE	NUMBER OF PROPERTIES EVALUATED	ESTIMATED NUMBER AND PERCENT OF PROPERTIES WITHIN THE SPECIFIED RISK RANGE							
		Estimated Total Cancer Risk							
		CTE				RME			
		≤1E-06	>1E-06 to ≤1E-05	>1E-05 to ≤1E-04	>1E-04	≤1E-06	>1E-06 to ≤1E-05	>1E-05 to ≤1E-04	>1E-04
68102	6	1	5	0	0	0	1	5	0
		17%	83%	0%	0%	0%	17%	83%	0%
68104	7	1	5	1	0	0	1	5	1
		14%	71%	14%	0%	0%	14%	71%	14%
68105	466	49	373	44	0	1	51	372	42
		11%	80%	9%	0%	0%	11%	80%	9%
68106	33	6	22	5	0	0	6	23	4
		18%	67%	15%	0%	0%	18%	70%	12%
68107	800	94	674	32	0	0	105	664	31
		12%	84%	4%	0%	0%	13%	83%	4%
68108	276	15	257	4	0	0	15	257	4
		5%	93%	1%	0%	0%	5%	93%	1%
68110	259	15	241	3	0	0	16	240	3
		6%	93%	1%	0%	0%	6%	93%	1%
68111	800	69	697	34	0	0	79	690	31
		9%	87%	4%	0%	0%	10%	86%	4%
68112	6	0	5	1	0	0	0	5	1
		0%	83%	17%	0%	0%	0%	83%	17%
68131	172	19	145	8	0	0	24	140	8
		11%	84%	5%	0%	0%	14%	81%	5%
68132	221	15	188	18	0	1	20	184	16
		7%	85%	8%	0%	0%	9%	83%	7%
ALL	3046	284	2612	150	0	2	318	2585	141
		9%	86%	5%	0%	0%	10%	85%	5%

Table 5-4. Estimated Total Non-Cancer Risks to Residents from Ingestion and Dermal Exposure from Surface Soil

Panel A. CTE Receptor

ANALYTE	TOTAL NUMBER OF PROPERTIES	NUMBER OF PROPERTIES WITHIN THE SPECIFIED RISK RANGE							
		Total Non-Cancer Hazard Index (HI)							
		Child				Adult			
		≤1	2 - 5	>5	Max	≤1	2 - 5	>5	Max
Aluminum	219	214	0	0	0.08	214	0	0	0.01
Antimony	2,843	2,843	0	0	0.9	2,843	0	0	0.09
Arsenic	3,046	2,970	71	5	8	3,046	0	0	0.8
Cadmium	3,046	3,046	0	0	0.06	3,046	0	0	0.01
Cobalt	214	214	0	0	0.2	214	0	0	0.02
Chromium III	3,046	3,046	0	0	0.001	3,046	0	0	0.0001
Chromium VI	3,046	3,046	0	0	0.08	3,046	0	0	0.01
Copper	214	214	0	0	0.1	214	0	0	0.02
Iron	214	214	0	0	0.3	214	0	0	0.03
Manganese	219	214	0	0	0.1	214	0	0	0.01
Mercury	2,832	2,830	1	1	6	2,832	0	0	0.6
Thallium	3,046	3,040	6	0	4	3,046	0	0	0.4
Vanadium	214	214	0	0	0.04	214	0	0	0.00
Zinc	27,737	27,737	0	0	0.1	27,737	0	0	0.01

Panel B. RME Receptor

ANALYTE	TOTAL NUMBER OF PROPERTIES	NUMBER OF PROPERTIES WITHIN THE SPECIFIED RISK RANGE							
		Total Non-Cancer Hazard Index (HI)							
		Child				Adult			
		≤1	2 - 5	>5	Max	≤1	2 - 5	>5	Max
Aluminum	219	214	0	0	0.2	214	0	0	0.03
Antimony	2,843	2,841	2	0	2.5	2,843	0	0	0.3
Arsenic	3,046	2,736	253	57	24	3,038	8	0	3
Cadmium	3,046	3,046	0	0	0.2	3,046	0	0	0.02
Cobalt	214	214	0	0	0.5	214	0	0	0.1
Chromium III	3,046	3,046	0	0	0.003	3,046	0	0	0.0003
Chromium VI	3,046	3,046	0	0	0.2	3,046	0	0	0.03
Copper	214	214	0	0	0.4	214	0	0	0.04
Iron	214	214	0	0	0.8	214	0	0	0.1
Manganese	219	214	0	0	0.3	214	0	0	0.04
Mercury	2,832	2,829	1	2	17	2,831	1	0	1.8
Thallium	3,046	3,029	16	1	11	3,046	0	0	1.1
Vanadium	214	214	0	0	0.1	214	0	0	0.01
Zinc	27,737	27,737	0	0	0.4	27,737	0	0	0.04

-- = Not applicable

Table 5-5. Summary of Non-Cancer Risks to Residents from Ingestion and Dermal Exposure to Surface Soil
Total Non-Cancer Hazard Index (HI)

Zip Codes	Chemical-Specific HQs Included in the HI Calculations[1]			Number of Properties Evaluated[2]	Number of Properties with Total Non-Cancer HI > 1 (percent of properties with Total Non-Cancer HI >1)				Primary Contributor(s) to Estimated Risks
					CHILD		ADULT		
	Number	Chemical Names			CTE	RME	CTE	RME	
ALL	13	aluminum antimony arsenic cadmium chromium III chromium VI cobalt copper	iron manganese thallium vanadium zinc	213	7 (3%)	207 (97%)	0 (0%)	2 (0.9%)	·Arsenic and/or Thallium (ingestion)
	8	antimony arsenic cadmium chromium III chromium VI	mercury thallium zinc	2,630	104 (4%)	1,412 (53%)	0 (0%)	8 (<0.3%)	·Arsenic and/or Thallium (ingestion)
	7	arsenic cadmium chromium III chromium VI	mercury thallium zinc	202	7 (4%)	23 (11%)	0 (0%)	1 (0.5%)	·Arsenic (ingestion)

[1] The number of chemicals that could be summed across properties varies from chemical-to-chemical and property to property, based on the available data used in the risk assessment.

[2] Total number of properties with data available for all chemicals included in the HI calculations (e.g., n=7, n=8 or n=13). HIs not calculated for cases where the number of chemicals evalauted at a property equals 1 (e.g., HQ results for zinc).

CTE = Central Tendency Estimate
HI = Hazard Index
HQ = Hazard Quotient
RME = Reasonable Maximum Estimate

Table 5-6. Summary of Non-Cancer Risks to Residents from Ingestion and Dermal Exposure to Surface Soil
Target Organ Hazard Index (HI)

Target Organ	Chemicals Evaluated	Number of Properties Evaluated[1]	Number of Properties with Target Organ Non-Cancer HI > 1			
			CHILD		ADULT	
			CTE	RME	CTE	RME
Central Nervous Sysem	aluminum, manganese	214	0	0	0	0
Kidney	cadmium, mercury, vanadium	0	--	--	--	--
Gastrointestinal Tract	chromium VI, copper, iron	214	0	0	0	0
Liver	antimony, thallium	2,843	6 (0.2%)	26 (0.9%)	0	0
Blood	zinc	27,737	0	0	0	0
Thyroid	cobalt	214	0	0	0	0
Skin	arsenic	3,046	76 (2.5%)	310 (10.2%)	0	8 (0.3%)

[1] Properties with results for all chemicals acting on a target organ were evaluated, only.

-- = Could not be quantified. No residential properties had results for all 3 chemicals (cadmium, mercury and vanadium).

CTE = Central Tendency Exposure
RME = Reasonable Maximum Exposure

Figure 2-1 Site Location
Omaha, Nebraska

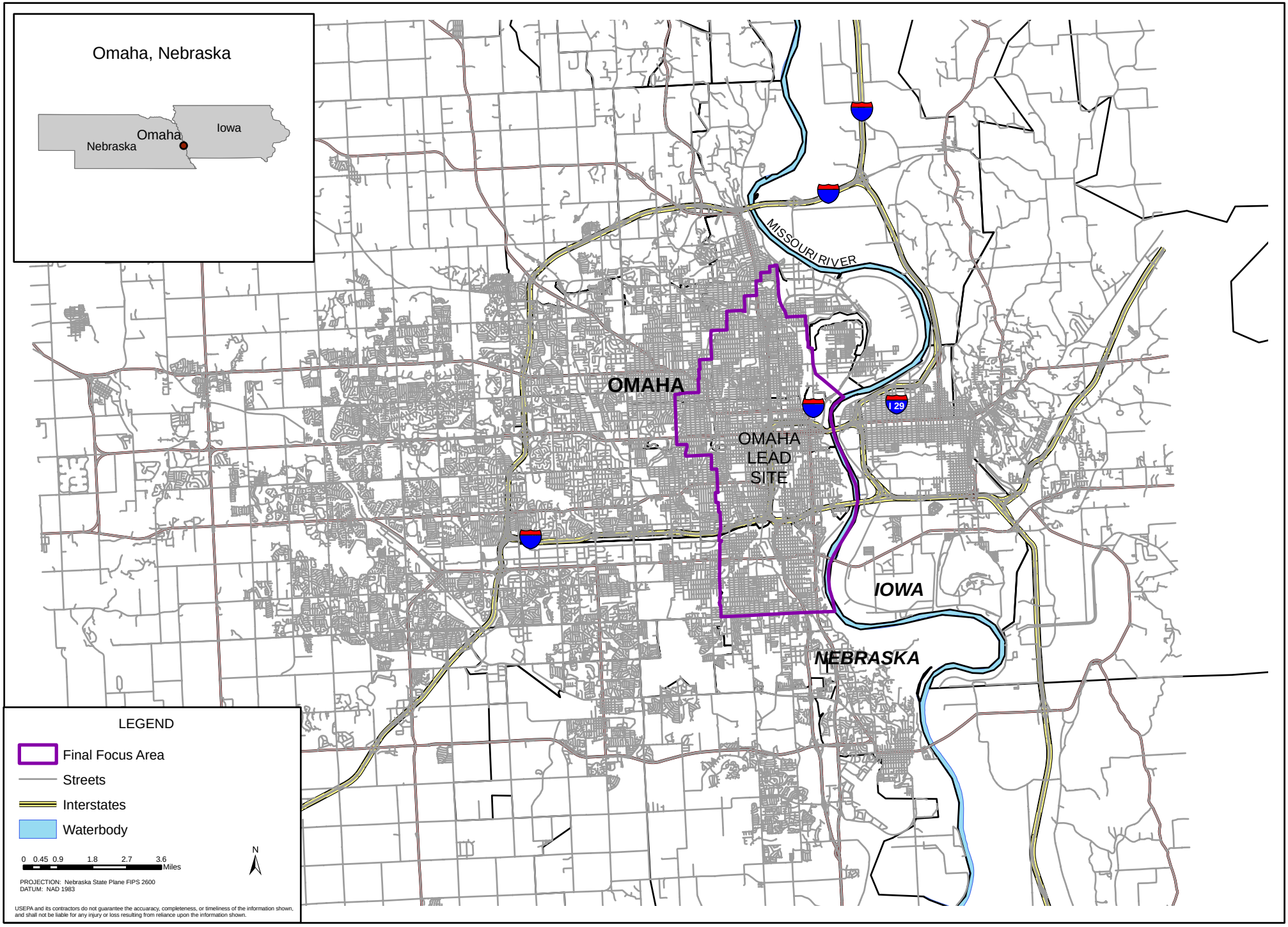


Figure 2-2. Highest Non-Drip Lead Concentration

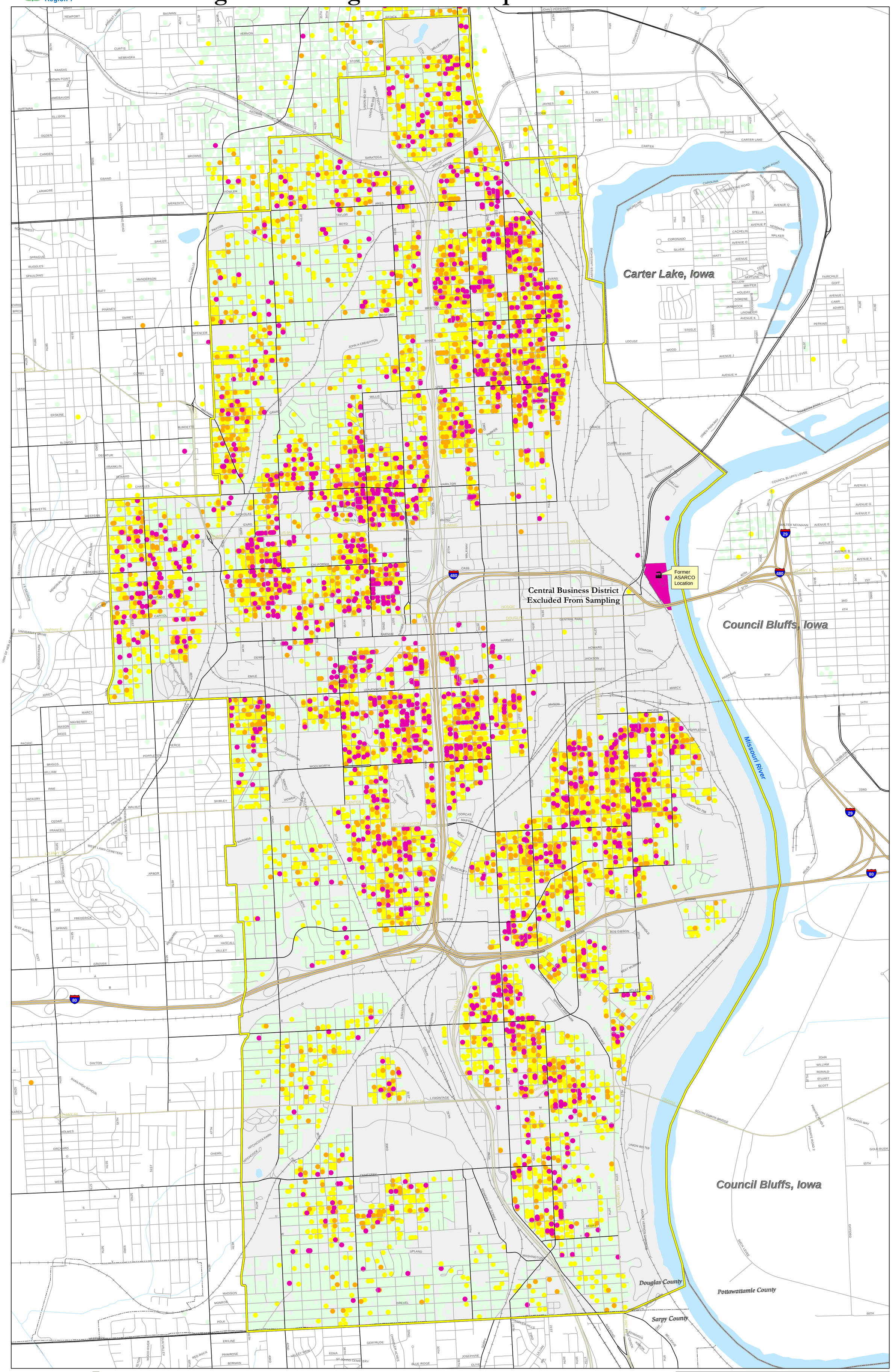
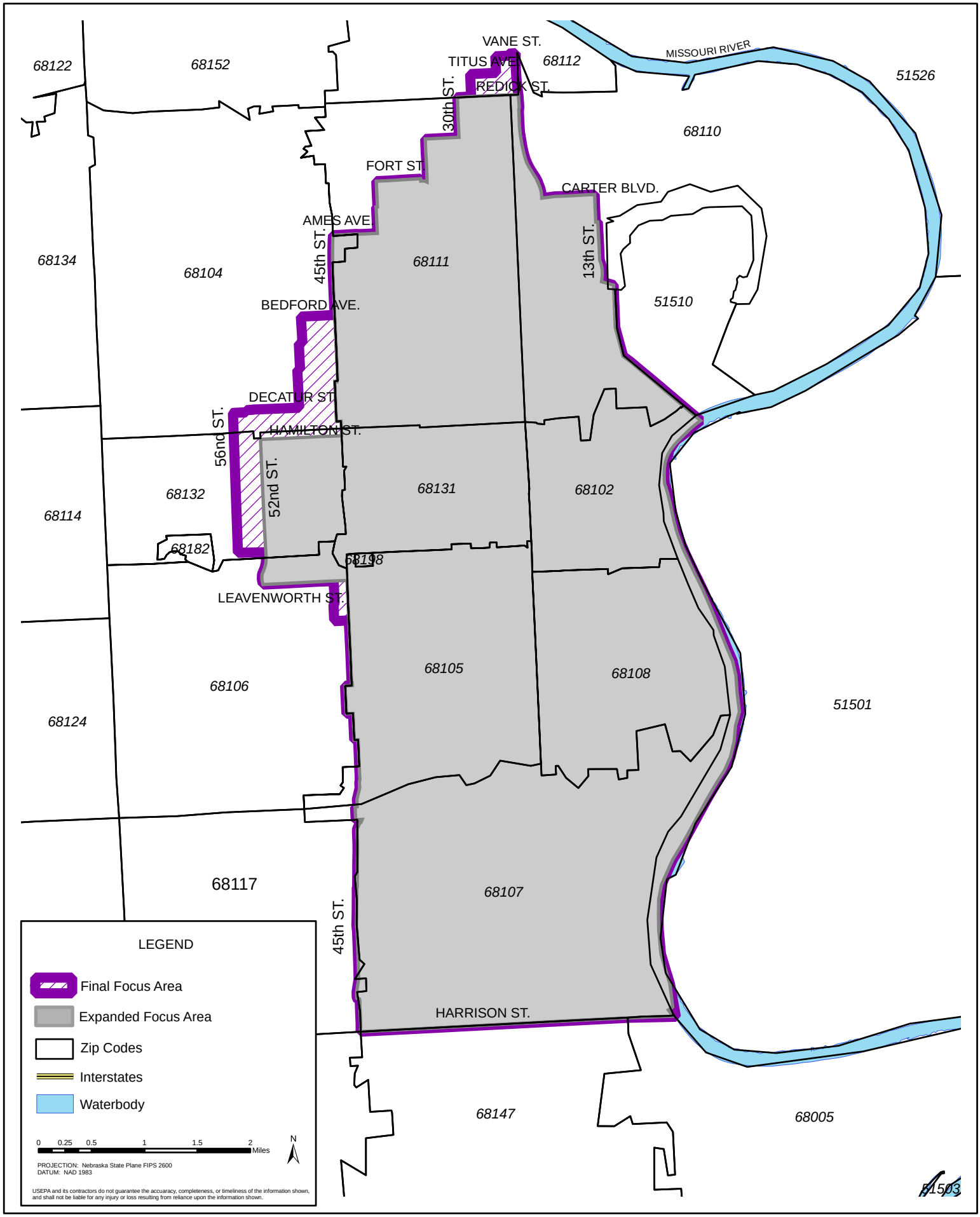


Figure 2-3 Focus Area Boundaries
Omaha Lead Site

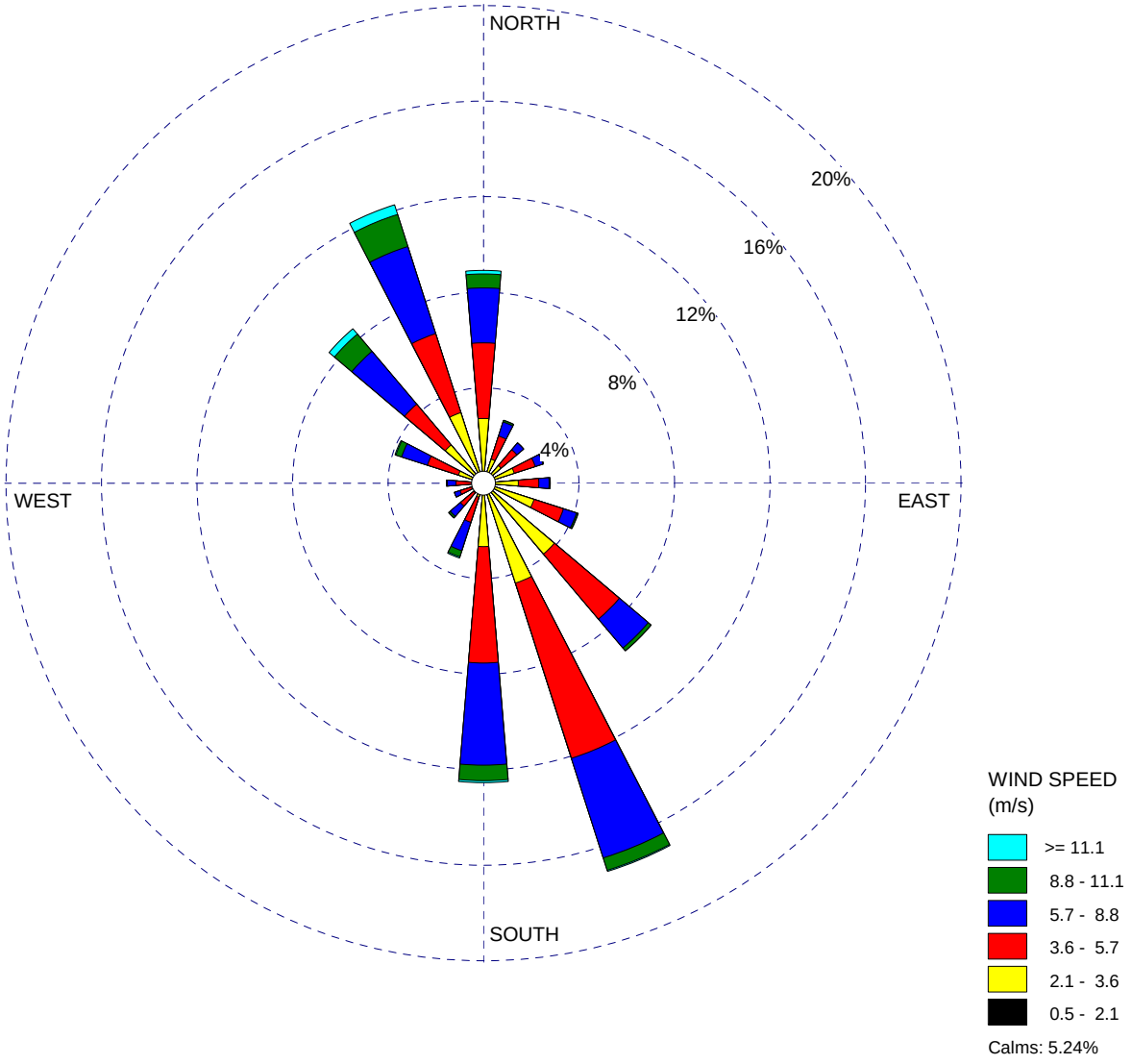


WIND ROSE PLOT:

Figure 2-4. Wind Rose Plot for Omaha/Eppley Airfield (Station#14942)
Wind Direction (blowing from)

DISPLAY:

Wind Speed
Direction (blowing from)



COMMENTS:	DATA PERIOD:	COMPANY NAME:	
	1984 1985 1986 1987 1988 1989 1990 1991 1992 **Jan 1 - Dec 31** **00:00 - 23:00**	MODELER:	
	CALM WINDS: **5.24%**	TOTAL COUNT: **78912 hrs.**	
	AVG. WIND SPEED: **4.54 m/s**	DATE: **8/13/2008**	PROJECT NO.:

WRPLOT View - Lakes Environmental Software

Figure 2-5. Historical Locations of Former Sources
Omaha Lead Site

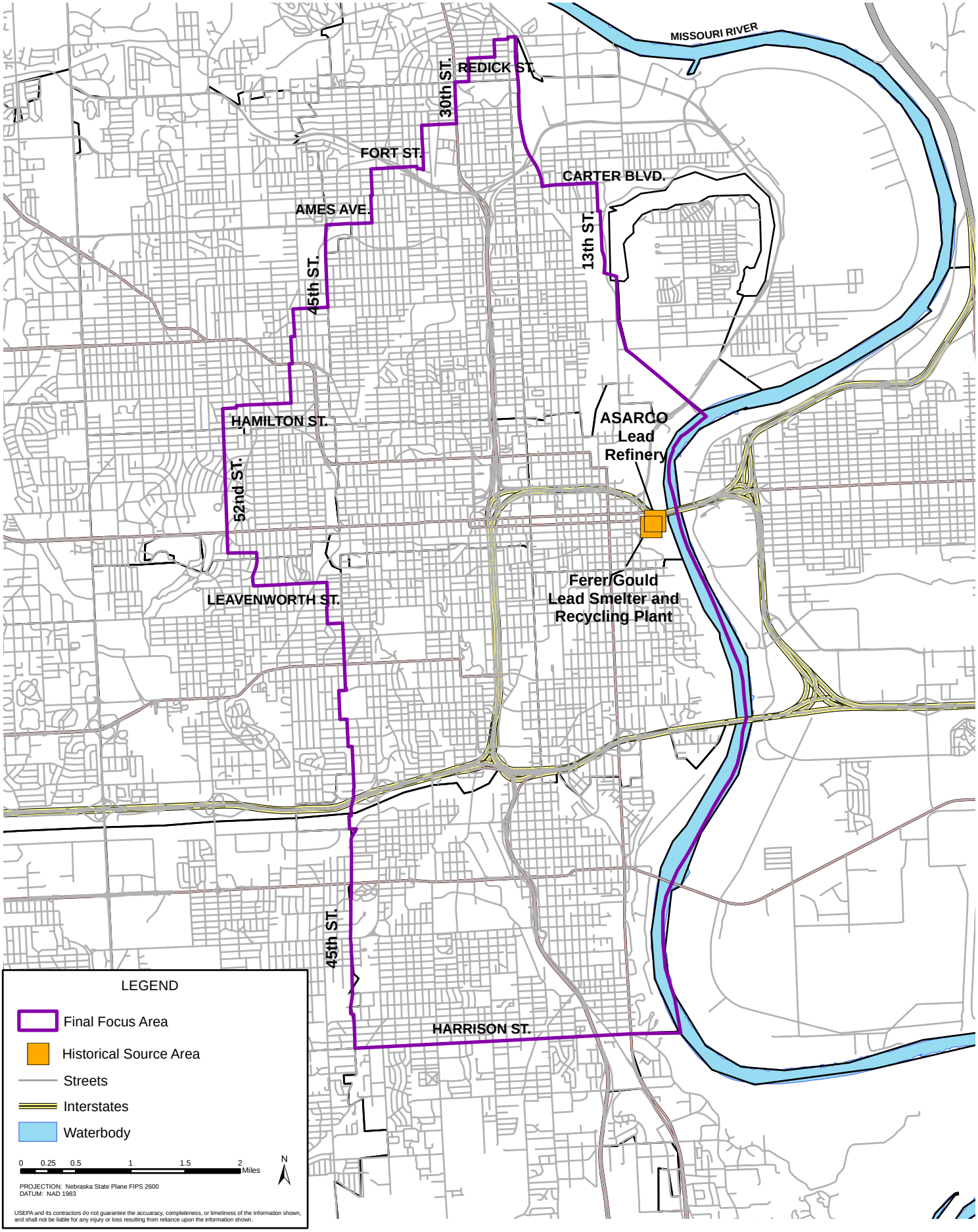


FIGURE 2-6 ELEVATED BLOOD LEAD LEVELS BY YEAR

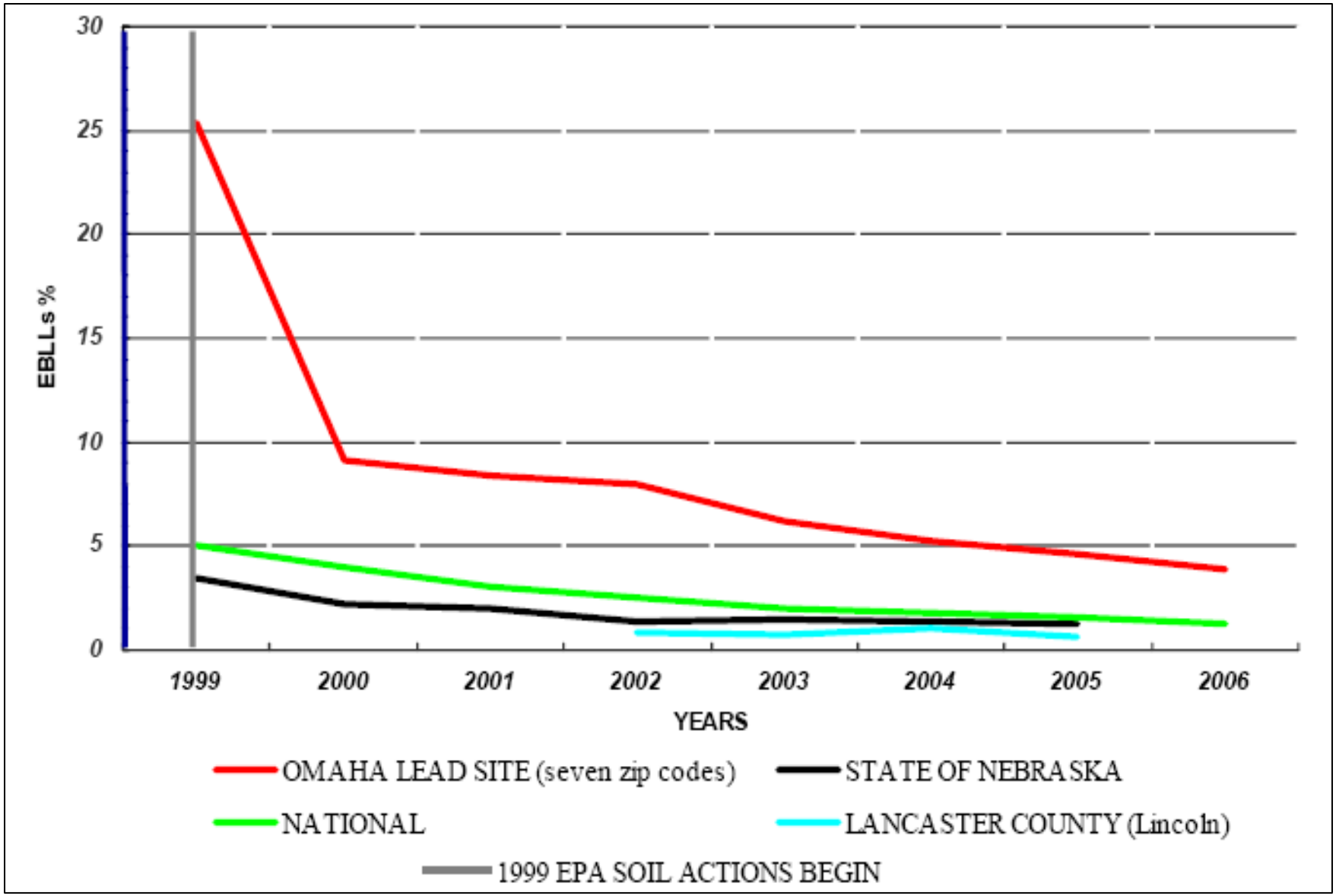


Figure 2-7. USEPA Ambient Air Monitoring Stations in Douglas County, Nebraska

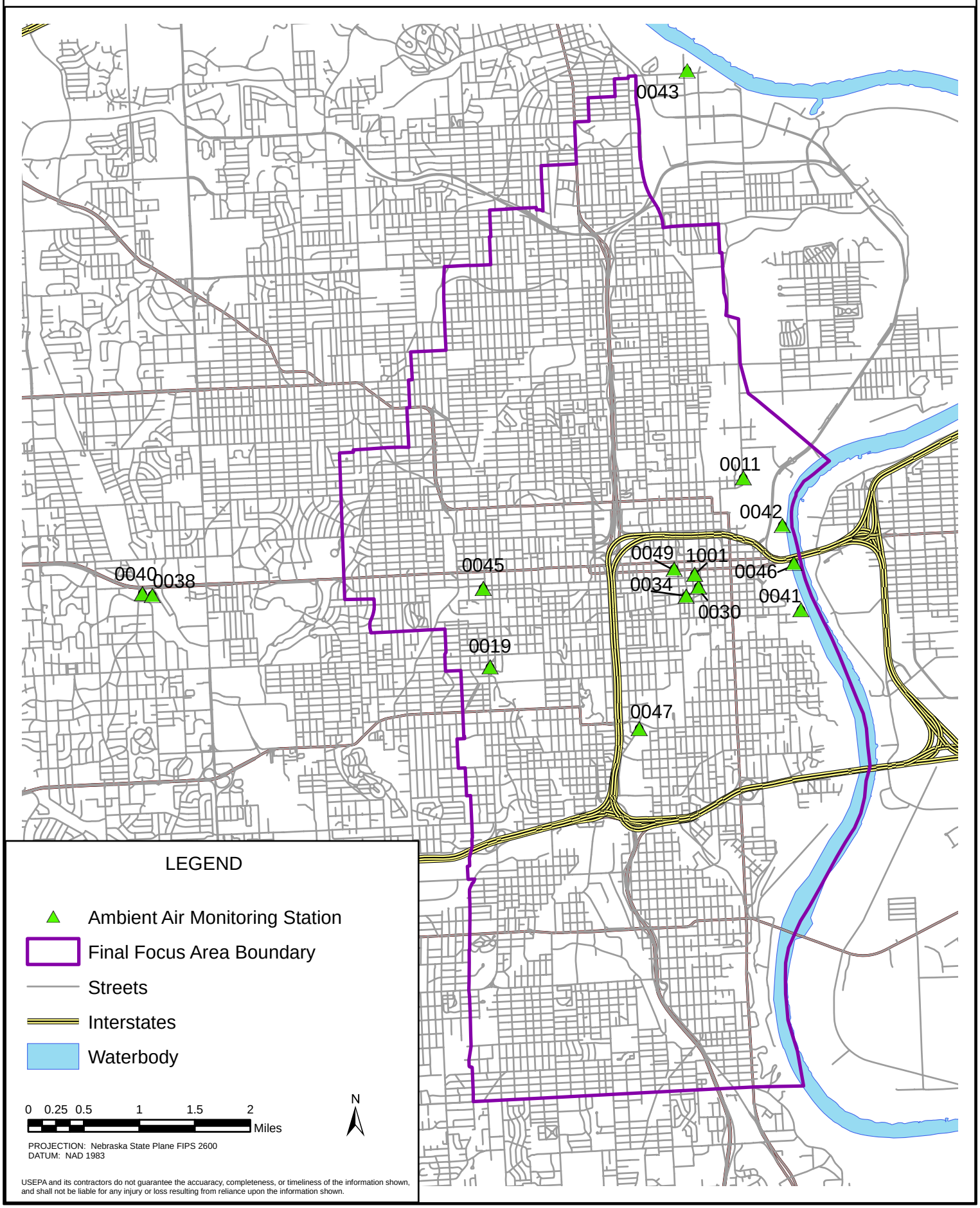


Figure 3-1. General Site Conceptual Model for Human Exposure at the Omaha Lead Site

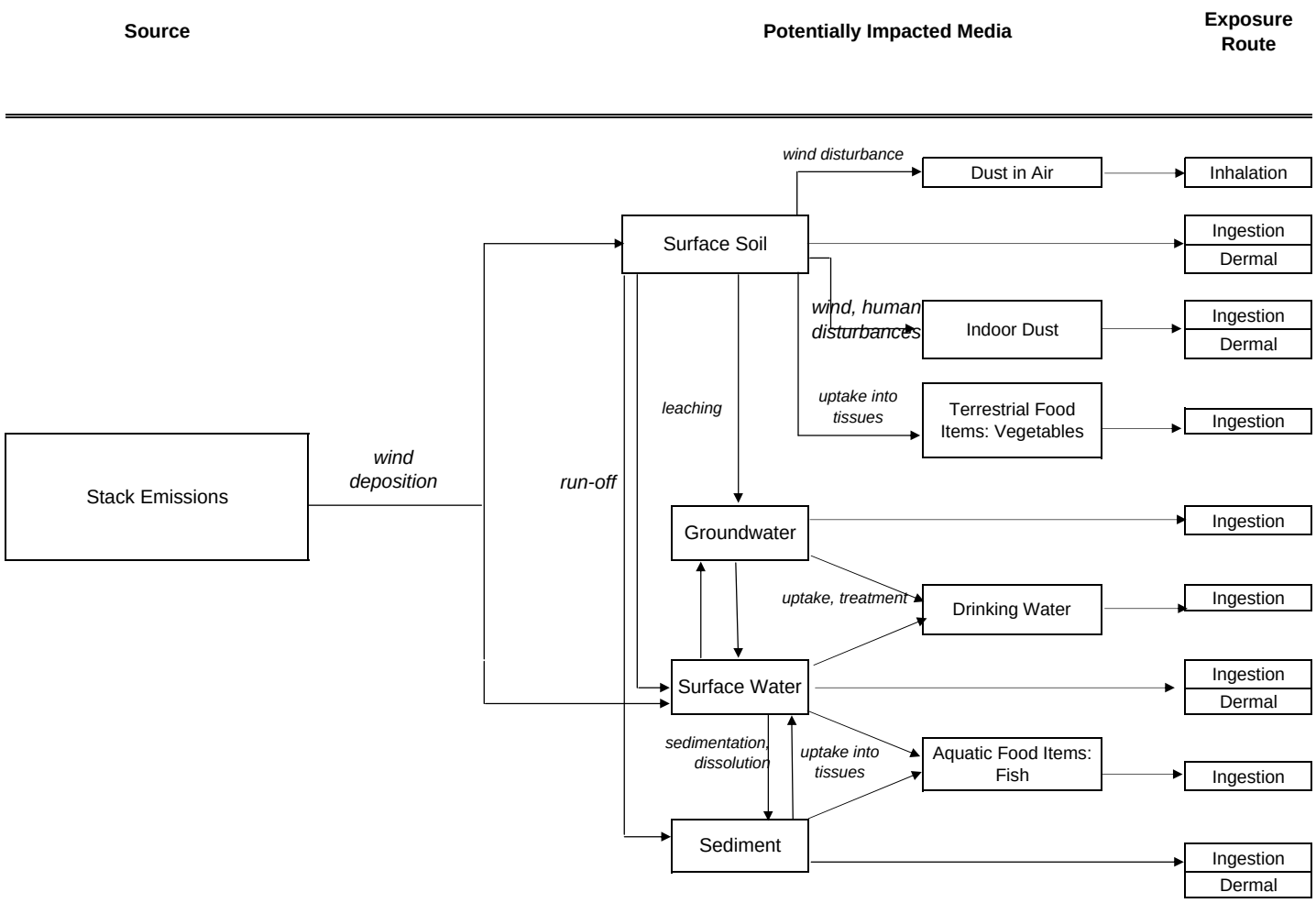
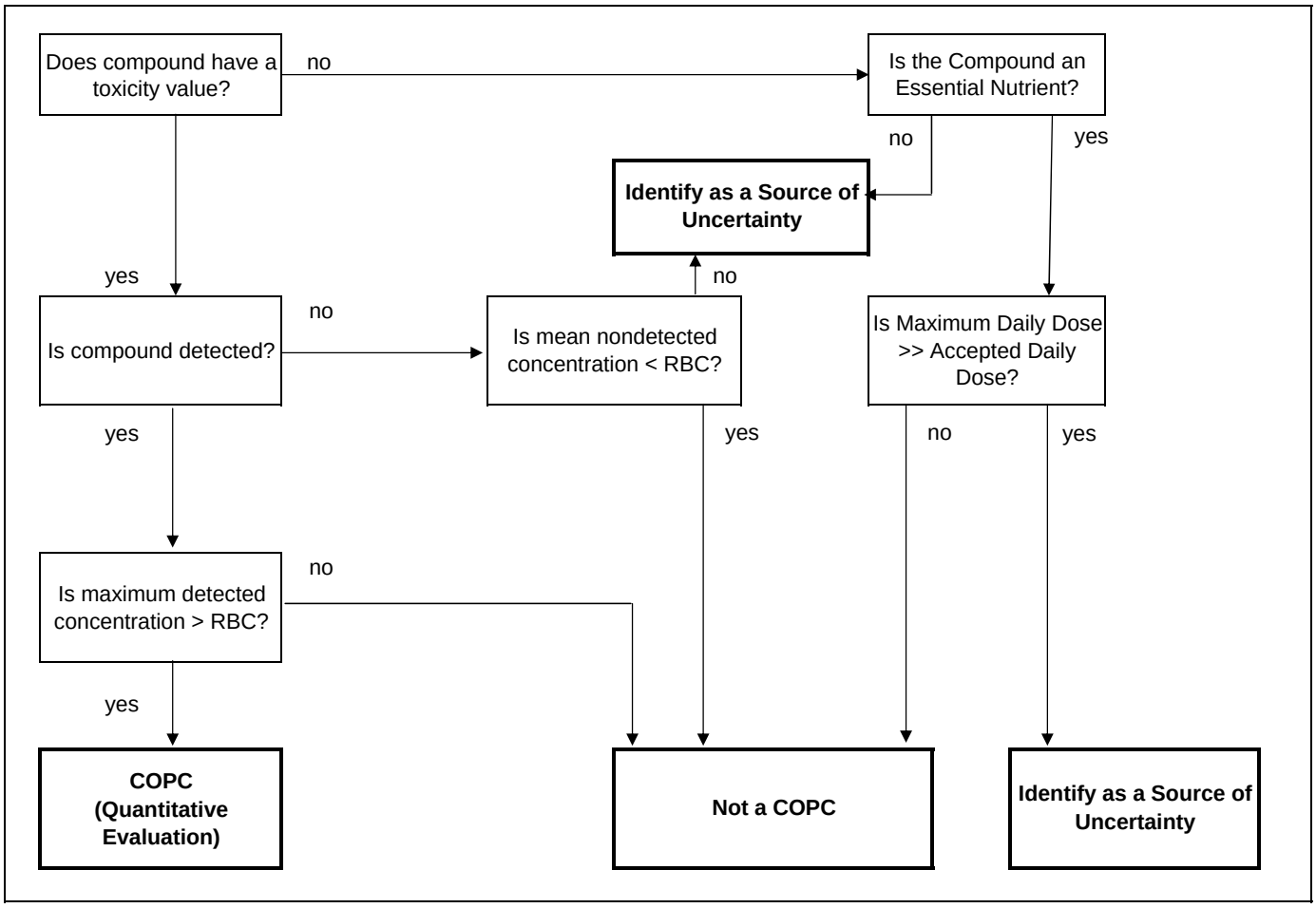


Figure 3-2 COPC Selection Procedure

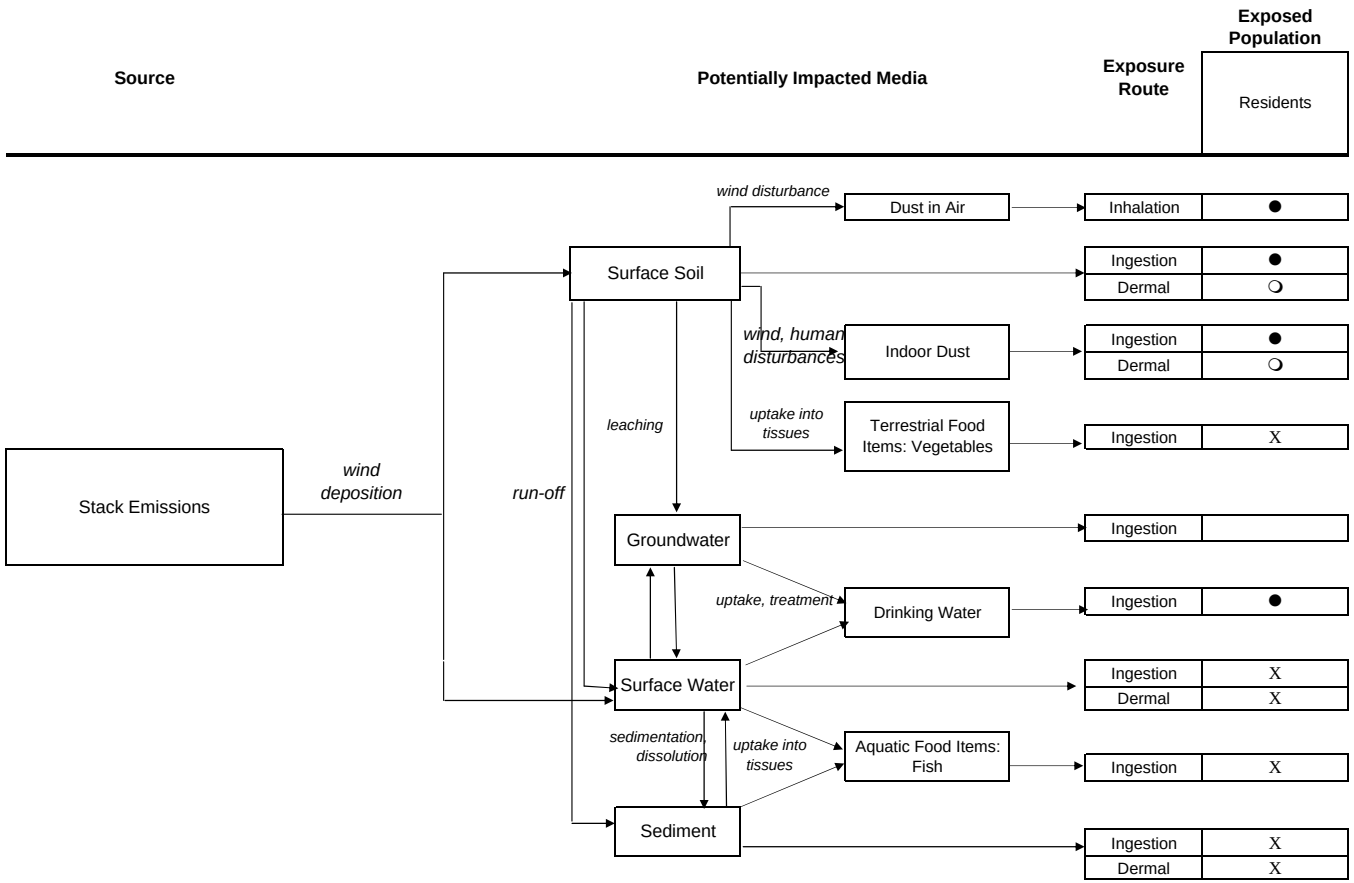


Notes:

RBC = Risk-Based Concentration. A risk-based concentration based on a non-cancer Hazard Quotient (HQ) = 0.1 or Cancer risk = 1E-06. Values are the USEPA (2008c) Regional Screening Levels for residential soil.

COPC = chemical of potential concern

Figure 4-1. Site Conceptual Model for Human Exposure to Lead at the Omaha Lead Site



LEGEND

●	Pathway is complete and might be significant; sufficient data are available for quantitative evaluation
❍	Pathway is complete and might be significant; insufficient data are available for quantitative evaluation
X	Pathway is or may be complete, however, risk is low or data are lacking. Qualitative evaluation only.
	Pathway is not complete; no evaluation required

FIGURE 4-2. RBA VALUES ESTIMATED FROM IVBA MEASURES IN SOIL

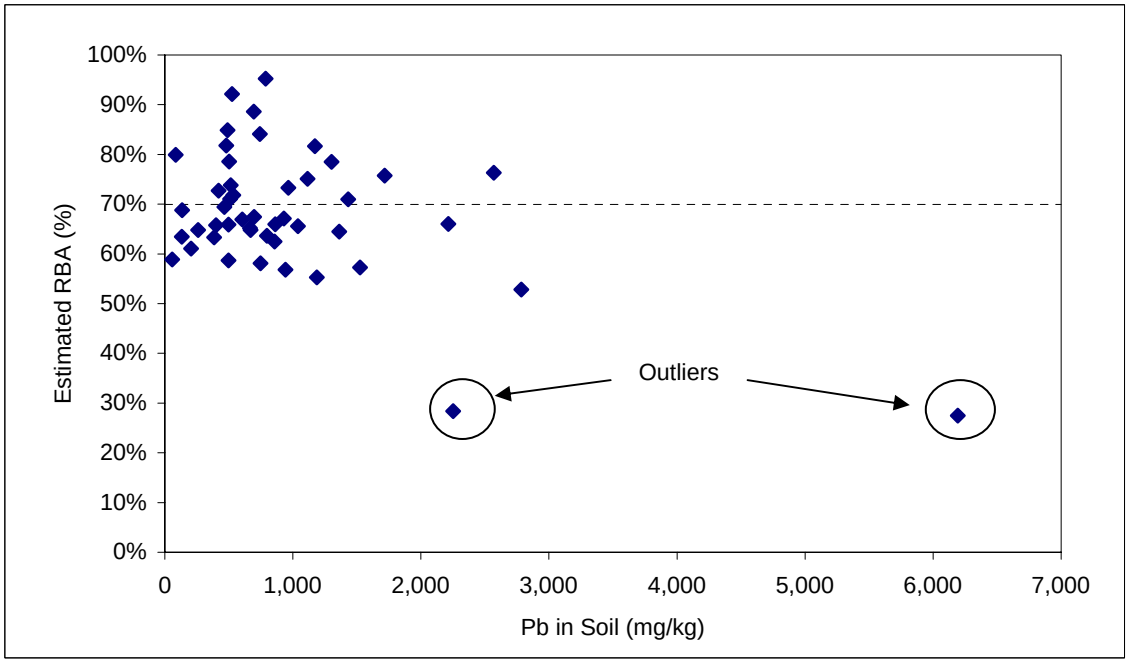


FIGURE 4-3 RBA VALUES ESTIMATED FROM IVBA MEASURES IN DUST

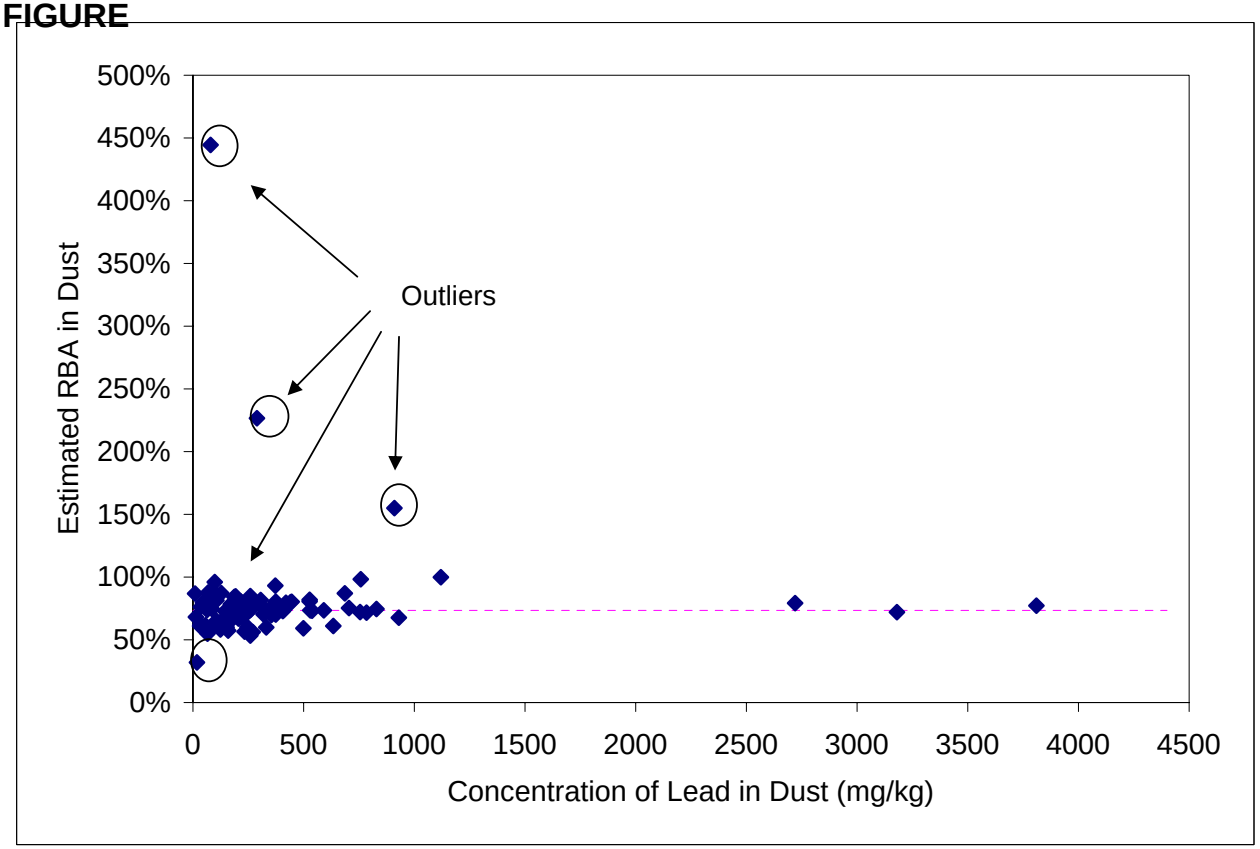
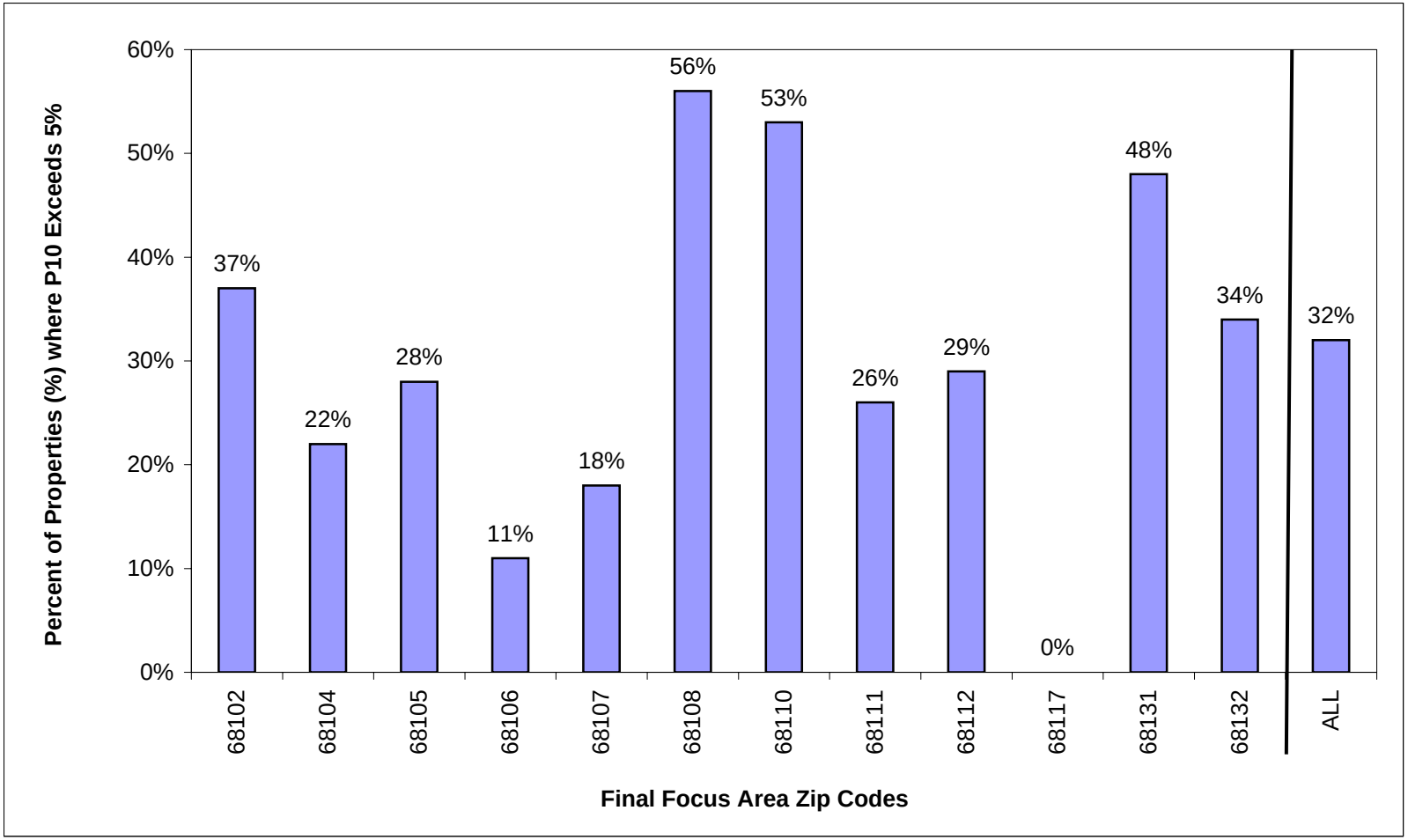
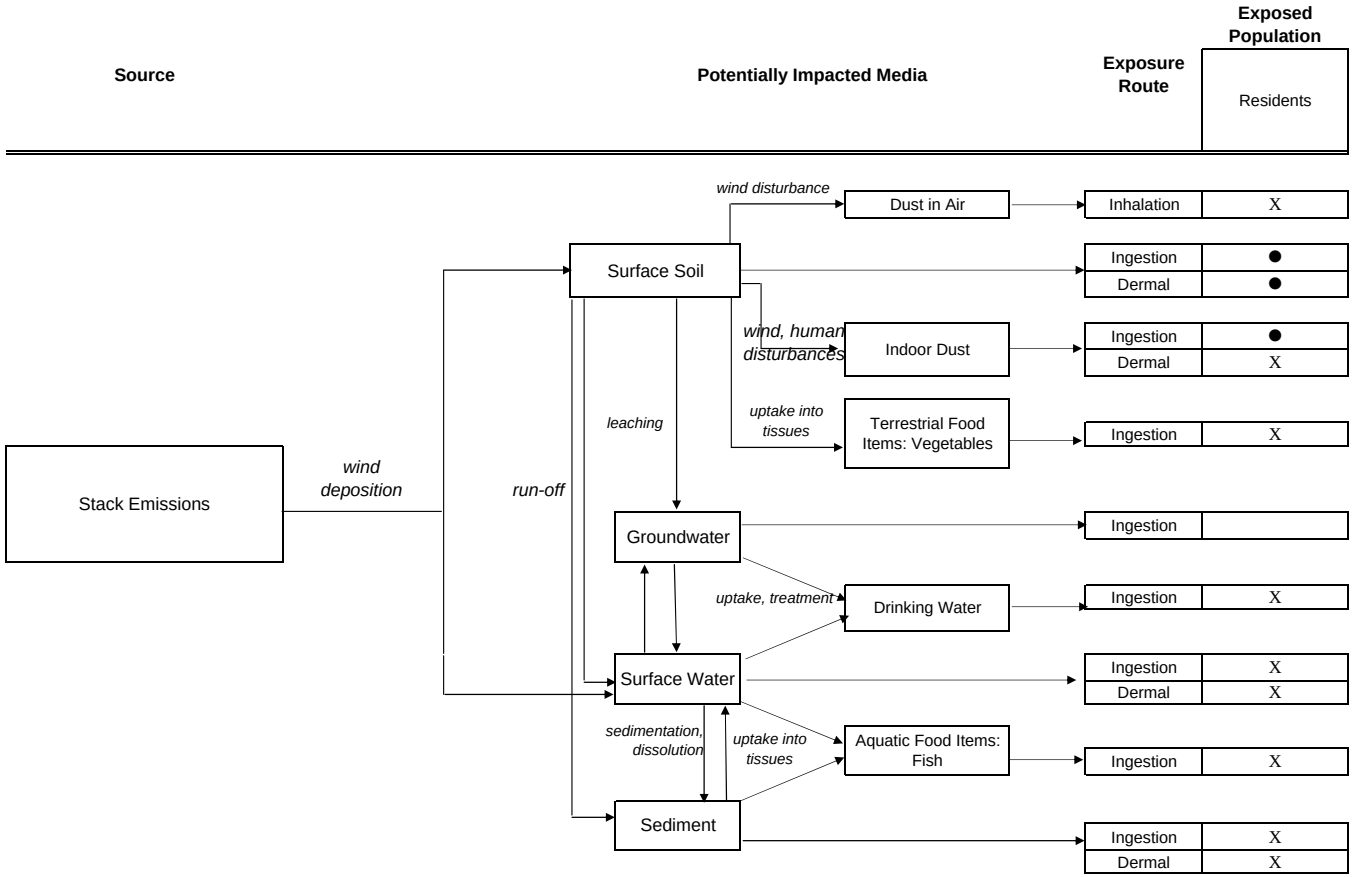


Figure 4-4. Percent of Properties in the Final Focus Area that Exceed EPA's Health Based Goal for Lead (P10>5%)



P10 = probability of having a blood lead level that exceeds 10 µg/dL

Figure 5-1. Site Conceptual Model for Human Exposure to Non-Lead COPCs at the Omaha Lead Site



LEGEND

●	Pathway is complete and might be significant; sufficient data are available for quantitative evaluation
❍	Pathway is complete and might be significant; insufficient data are available for quantitative evaluation
X	Pathway is or may be complete, however, risk is low or data are lacking. Qualitative evaluation only.
	Pathway is not complete; no evaluation required

APPENDIX A

DERIVATION OF THE FINAL FOCUS AREA BOUNDARY USING GEOSPATIAL ANALYSIS

APPENDIX A
DERIVATION OF THE FINAL FOCUS AREA BOUNDARY USING GEOSPATIAL ANALYSIS

1.0 INTRODUCTION

As discussed in the risk assessment, when there has been a release of a hazardous substance at a Superfund site, the EPA seeks to define the nature and extent of the environmental contamination by performing a Remedial Investigation (RI). The spatial bounds of the RI are not set by any *a priori* definition, but are determined by the results of the RI. That is, EPA generally continues to sample and collect data until a) there is no longer an observable increment of the contaminant in the environment, or b) the concentration levels drop below a level of concern.

At the Omaha Lead Site (OLS), the RI has focused on characterizing the extent of lead contamination in surface soil of residential or vacant properties located in the vicinity of two former smelters. As of December 9, 2007, data were available for the level of lead in soil at 28,685 residential or vacant properties at the site. The purpose of this Appendix is to describe how these data were evaluated to help establish a boundary for the RI, beyond which there does not appear to be a need for continued sampling and investigation.

2.0 OVERVIEW OF THE STRATEGY

EPA considered two strategies for defining the boundary of the RI. In the first approach, the boundary would be defined as the area beyond which the increase in soil lead concentrations due to lead emissions from the former smelters was sufficiently small (e.g., 10 mg/kg) that it would be negligible compared to naturally occurring levels of lead in soil. This approach is very difficult to implement, however, because of the inherent variability in lead levels in soil from naturally occurring and other non-smelter sources. For this reason, EPA chose to implement a second approach that focuses on whether or not the average concentration of lead in soil at a property is above 400 mg/kg (the default level of concern and soil screening level for residential properties). If a property has a mean concentration in yard soil above 400 mg/kg, it is considered likely that smelter releases are the principal reason, although other sources of lead (naturally occurring background levels, releases from paint, automobile exhaust, etc.) may also contribute in some cases.

Ideally, the study area boundary would be selected to include every property with a mean soil concentration above 400 mg/kg. However, since it would not be possible to identify such properties without testing, this would, in essence, extend the study boundary indefinitely. However, the area requiring investigation can be limited by focusing on the <u>density</u> (frequency) of properties above 400 mg/kg. That is, it is expected that the density of impacted properties will

be highest near the former smelters, and then tend to decrease as a function of distance away from the smelters. Based on this, the boundary of the study area may be defined as the distance at which the density of properties with mean concentrations above 400 mg/kg drops below 5%. It is possible that a small number of properties beyond the boundary may have concentrations above 400 mg/kg, and that smelter releases may be a contributing factor in some cases. However, such properties are expected to be infrequent, and the relative contribution of the smelters to the soil lead level is expected to be low.

3.0 MATHEMATICAL METHODS

3.1 Calculating the Mean Concentration at Each Property

The mean concentration of lead at each property was computed as detailed in Appendix B and Section 4.4.1. In brief, the mean was computed as the average of all surface soil (0-1 inch) composite samples from the yard, garden, or play area. Samples from the drip zone were not included.

3.2 Calculating the Density of Properties Above 400 mg/kg

The density, or percentage, of properties with mean concentrations above 400 mg/kg was calculated on a *neighborhood* basis. The *neighborhood* of a given property consists of all other properties that are located within 500 meters of the property. This calculation was implemented using the *neighborhood statistics* function of ESRI's Spatial Analyst extension with a 500 meter radius moving circular window. The result is that each property is assigned a value that represents the density (frequency) of properties in the *neighborhood* of that property (including the property itself) with a mean soil lead concentration above 400 mg/kg. The interpolation function of the Spatial Analyst extension was then used with this grid of values to generate a contour line that connected points on the grid with values of 5%.

4.0 RESULTS

The resulting contour line is the green boundary line shown in the enclosed map. The green boundary includes all properties that are located in *neighborhoods* in which 5% or more of the properties have a mean concentration exceeding 400 mg/kg.

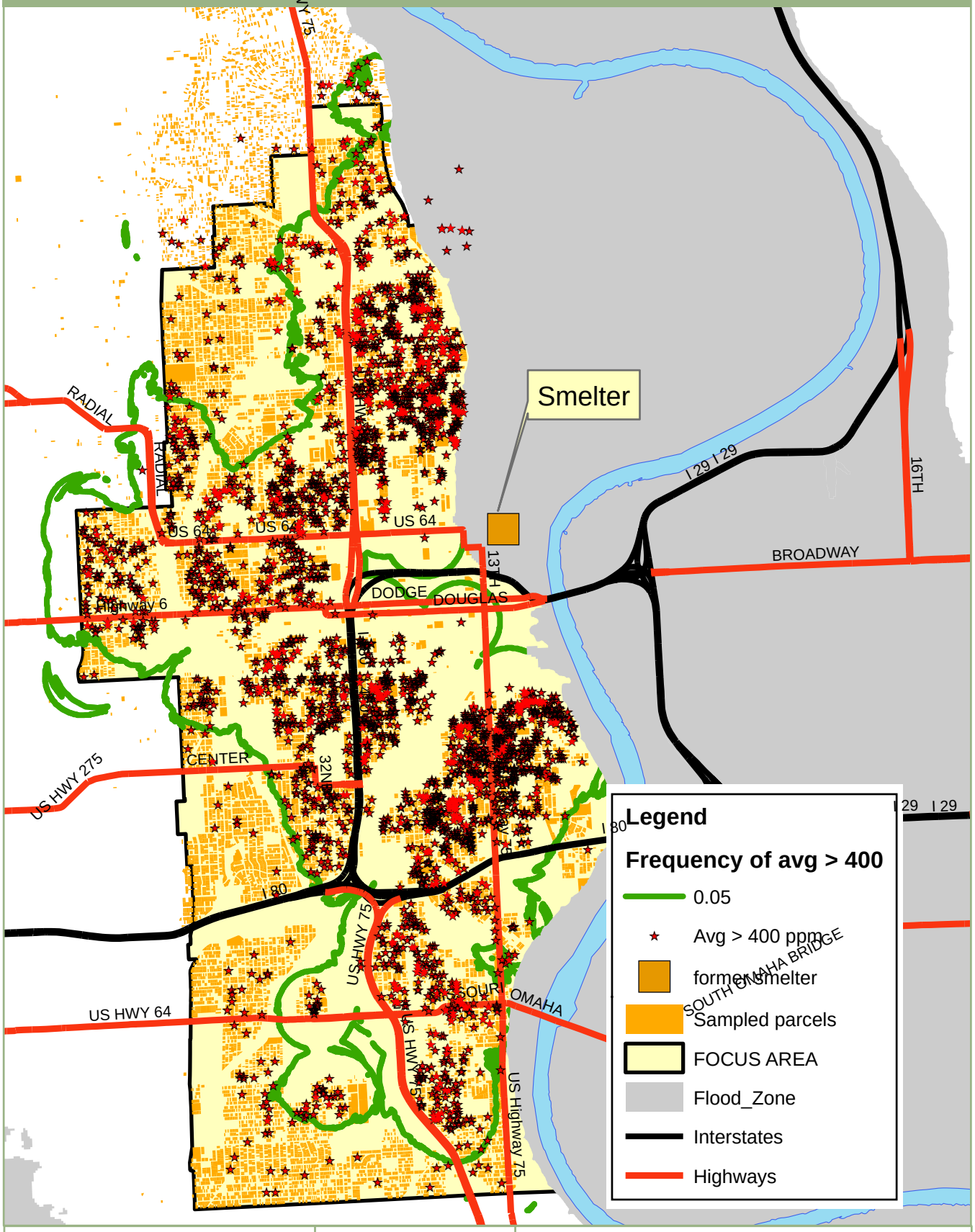
The isolated shapes or "islands" which can be found northwest and southwest of the estimated site boundary are artifacts that occur when the number of properties in a neighborhood is small, or there is a large gap between sampled parcels (or both). For example, the crescent-shaped island that is between Highway 6 and Highway 275 is due to the combination of the isolated parcels that are located near the crescent shape and the two parcels located closest to, but outside of, the estimated site boundary, which have mean concentrations exceeding 400 mg/kg. The neighborhoods of these isolated parcels include one or both of the parcels with mean

concentrations exceeding 400 mg/kg. However, the combination of the gap between these isolated parcels, and the isolation of the two parcels with mean concentrations greater than 400 mg/kg (relative to other parcels with mean concentrations greater than 400 mg/kg), provides insufficient evidence to conclude that the neighborhoods located within the crescent are actually impacted by smelter releases. Similar arguments can be made for the other isolated shapes that are present in the map. While the algorithm could have been modified to eliminate neighborhoods with less than a selected minimum number of parcels, and those that included isolated parcels with mean concentrations exceeding 400 mg/kg, the selection of the criteria that excluded neighborhoods would have been arbitrary. It seemed more transparent to include all neighborhoods in the estimation of the boundary location and then consider how the boundary was estimated when interpreting the results.

5.0 UNCERTAINTY

The large amount of data that are available at this site to estimate the location of the study area boundary provides a generally high level of confidence in the results. However, several sections of boundary along the western edge are located near the geographic extent of the available data, which increases the uncertainty somewhat in that area. The uncertainty in the location of the boundary along the western edge of the site may be lessened by the collection of additional data in this area.

The green boundary includes parcels that are located within 500 meter overlapping circular windows in which the frequency of parcels with average concentrations exceeding 400 ppm is 5% or more.



Miles

0 0.5 1 2 3

Map Notes

Prepared by: BCT Date: 5/11/2008

\Region 7\Omaha\GIS\Projects\PhaseII\OmahaGeostatistics_Boundary_FreqAvgGT400_5_11_08.mxd

APPENDIX B

ELECTRONIC DATA AND DATABASE QUERIES

SEE ATTACHED CD

APPENDIX C

XRF Data Adequacy Evaluation

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C-4 XRF Detection Limit Evaluation
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APPENDIX C
XRF Data Adequacy Evaluation

1.0 OVERVIEW

Soil samples collected from the Omaha Lead Site (OLS) were analyzed for lead and other metals using X-ray Fluorescence Spectroscopy (XRF). Approximately 5% of these soil samples were also analyzed by Inductively Coupled Plasma Spectroscopy (ICP). Of these two methods, ICP is considered to be the most reliable, while XRF may be prone to measurement error and/or bias for some analytes.

In order to determine if the data obtained by XRF are reliable for use in the risk assessment, a data quality assessment of the XRF data was conducted. The evaluation approaches and the results are presented in the following sections.

2.0 METHODS FOR EVALUATING DATA QUALITY

The data adequacy evaluation approach is presented in Figure C-1, and is discussed below.

2.1 Detection Limit Evaluation

The first step in the data adequacy evaluation was to evaluate whether the XRF detection limit was adequate. In order for a detection limit to be considered adequate, either (1) the detection frequency had to be high (>80%), or (2) the detection limit had to be less than the concentration of potential concern to humans. The concentration of potential concern to humans is referred to as the Risk Based Concentration (RBC) or Preliminary Remediation Goal (PRG). Values of RBC were based on the Regional Screening Tables USEPA for residential soil screening levels (USEPA 2008), using a cancer risk of 1E-06 or a non-cancer Hazard Quotient (HQ) of 0.1, whichever was lower.

2.2 Correlation with ICP

If the XRF detection limit was considered adequate, the XRF data were then evaluated by assessing the strength of the correlation between the XRF values and the paired ICP values. If either the XRF or the ICP value was non-detect, the data pair was excluded from the correlation analysis. A minimum of 10 pairs of ICP/XRF data were required to perform the analysis.

The strength of the correlation was based on the R^2 value for the best fit straight line derived using ordinary linear regression. If the R^2 value was less than 0.7, it was concluded that the precision of the XRF method for analysis of that chemical was unacceptably low. The value of 0.7 is based on the USEPA Region VIII Superfund Technical Assessment and Response Team (START) Generic Quality Assurance Project Plan (QAPP) which specifies that XRF is usually considered adequate when the coefficient of determination is above 0.70 (UOS 1999).

3.0 RESULTS

Results presented in this section are based on the OLS soil database as it existed on 06/21/2007.

Table C-1 presents summary statistics for soil samples analyzed by XRF. As seen, there are XRF data for 22 different metals, although not all metals were analyzed in every sample. The most data are available for lead (N = 160,035), arsenic (N = 155,366) and zinc (N = 155,882).

Table C-2 shows summary statistics for soil samples analyzed by ICP. As seen, data are available for 25 different metals. As above, not all metals were analyzed in every sample. The most data are available for lead (N = 8099), with over 4000 samples for 11 other metals.

Table C-3 presents the list of metals evaluated by both XRF and ICP. Note that there are a number of metals evaluated by XRF that were not evaluated by ICP (rubidium, strontium, tin, titanium, uranium, and zirconium), and that there were a number of metals analyzed by ICP that were not evaluated by XRF (aluminum, beryllium, bismuth, calcium, magnesium, potassium, sodium, thallium, and vanadium).

3.1 Detection Frequency Evaluation

Table C-4 presents an evaluation of the XRF detection frequencies. Of the 22 chemicals analyzed by XRF, the detection frequency was considered adequate for only 9 analytes: iron, lead, manganese, rubidium, strontium, tin, titanium, zinc, and zirconium.

3.2 Correlation with ICP Concentrations

Figures C-2 through C-5 present the correlation of XRF and ICP concentrations for four of these chemicals (iron, lead, manganese, and zinc). As noted above, ICP analysis was not performed for the other 5 chemicals (rubidium, strontium, tin, titanium, and zirconium), so no correlation analysis can be performed for these analytes.

Table C-5 summarizes the results. The correlation of the XRF to the ICP data was first evaluated using the full data set. Based on this, data points whose studentized residual was more than 3 standard deviations were identified as potential outliers (Canavos 1984), and the analysis was repeated after exclusion of these data pairs. Examination of Table C-4 indicates that, of the 5 chemicals evaluated, two (lead and zinc) yielded R^2 value > 0.7.

3.3 Data Adequacy and Useability

Table C-6 combines the results of these three data quality evaluations and provides a conclusion of the overall adequacy of the XRF data. As shown, the results for only 2

analytes (lead and zinc) met the specified data quality criteria and are considered reliable for use in the risk assessment.

3.4 Adjustment of Useable XRF Data

Although the XRF data for lead and zinc are considered adequate for use, it is still necessary to adjust each XRF value to an estimated ICP-equivalent concentration to account for biases in the XRF data:

$$\text{ICP-equivalent} = a + b \cdot \text{XRF}$$

For lead, a detailed evaluation was performed to identify the best way to estimate the parameters for this adjustment. This evaluation is presented in Appendix F. In brief, the parameters were derived using the One Group (zero intercept approach) for a sub-set of the data where the concentration by ICP was ≤ 1000 mg/kg). The results are presented in Figure C-6.

For zinc, data were not available to support an assessment similar to that performed for lead in Appendix F, so the parameters were derived using ordinary linear regression.

Table C-7 summarizes the parameters used to adjust XRF values to ICP-equivalent concentrations for lead and zinc.

4.0 REFERENCES

Canavos, G.C. 1984. Applied Probability and Statistical Methods. Little, Brown and Company, Boston.

UOS. 1999. Emergency Response Program (ERP) Generic Quality Assurance Project Plan (QAPP) for the Superfund Technical Assessment and Response Team (START), USEPA Region VIII. March 11.

USEPA. 2008. Regional Screening Level Tables. Residential Soil Screening Level Table (*ressoil_sl_table_run_12SEP2008.xls)*. Available online at: http://www.epa.gov/reg3hwmd/risk/human/rb-concentration_table/Generic_Tables/index.htm (accessed September 2008).

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Table C-1. XRF Concentration Summary Statistics (mg/kg)

Panel A: Bulk Soil Fraction

ANALYTE	# Detects/ # Samples	Detection Frequency	Non-Detects			Detects			All		
			Avg	Min	Max	Avg	Min	Max	Avg	Min	Max
Antimony	223 / 78,072	0.3%	47	16	2,142	109	9	433	47	9	2,142
Arsenic	51,314 / 155,366	33.0%	14	2	270	36	4	3,330	21	2	3,330
Barium	5,459 / 51,479	10.6%	197	66	6,896	560	150	1,709	236	66	6,896
Cadmium	6,473 / 78,315	8.3%	24	9	1,057	52	2	194	26	2	1,057
Chromium	1,670 / 108,704	1.5%	125	26	2,985	3,186	5	1,602,400	172	5	1,602,400
Cobalt	37,489 / 99,309	37.7%	113	24	2,743	268	23	8,867	171	23	8,867
Copper	48,059 / 99,222	48.4%	27	6	442	40	2	17,783	34	2	17,783
Iron	99,883 / 99,888	100.0%	2,879	2,549	3,300	20,062	1,155	2,211,200	20,061	1,155	2,211,200
Lead	158,006 / 160,325	98.6%	19	3	378	336	10	52,995	332	3	52,995
Manganese	83,002 / 99,884	83.1%	512	30	2,702	493	82	103,900	496	30	103,900
Mercury	2,067 / 108,749	1.9%	9	1	324	41	1	6,746	10	1	6,746
Molybdenum	2,842 / 95,265	3.0%	4	1	141	30	1	17,245	5	1	17,245
Nickel	17,602 / 98,477	17.9%	31	7	685	323	7	599,650	84	7	599,650
Rubidium	99,670 / 99,849	99.8%	29	4	180	73	6	415	72	4	415
Selenium	237 / 108360	0.2%	4	1	111	30	1	833	4	1	833
Silver	84 / 78063	0.1%	24	7	753	107	2	2,718	24	2	2,718
Strontium	99832 / 99849	100.0%	19	5	46	137	10	3,224	137	5	3,224
Tin	18604 / 77381	24.0%	40	17	1,604	97	0	1,364	54	0	1,604
Titanium	77717 / 78123	99.5%	547	156	9,200	3,155	392	44,045	3,142	156	44,045
Uranium	19 / 2261	0.8%	14	4	68	31	24	38	14	4	68
Zinc	151663 / 155882	97.3%	33	7	516	306	16	12,701	299	7	12,701
Zirconium	99870 / 99885	100.0%	38	4	165	246	21	3,240	246	4	3,240

* Non-detects were evaluated at 1/2 the Quantitation limit

NA = value not available

Table C-2. ICP Concentration Summary Statistics (mg/kg)

Panel A: Bulk Soil Fraction

ANALYTE	# Detects/ # Samples	Detection Frequency	Non-Detects			Detects			All		
			Avg	Min	Max	Avg	Min	Max	Avg	Min	Max
Aluminum	1,420 / 1,421	99.9%	2.1	2.1	2.1	5,949	891	18,400	5,945	2.1	18,400
Antimony	968 / 4,265	22.7%	2.2	0.4	5.5	5.9	0.9	200	3.0	0.4	200
Arsenic	4,596 / 5,003	91.9%	3.5	0.3	7.5	21	1.7	1,600	19	0.3	1,600
Barium	4,998 / 5,003	99.9%	0.8	0.1	1.0	251	20	3,360	251	0.1	3,360
Beryllium	1,517 / 4,931	30.8%	0.4	0.2	0.7	0.8	0.2	2.0	0.5	0.2	2.0
Bismuth	296 / 1,248	23.7%	1.0	0.3	2.5	3.9	1.0	80	1.6	0.3	80
Cadmium	3,309 / 4,838	68.4%	0.5	0.1	1.9	2.1	0.3	22	1.6	0.1	22
Calcium	1,420 / 1,421	99.9%	6.4	6.4	6.4	9,140	1400	150,000	9,133	6.4	150,000
Chromium	4,906 / 5,002	98.1%	5.6	0.1	10	17	2.2	280	17	0.1	280
Cobalt	1,420 / 1,421	99.9%	0.5	0.5	0.5	6.9	0.3	20	6.9	0.3	20
Copper	1,420 / 1,421	99.9%	0.1	0.1	0.1	30	3.8	2,300	30	0.1	2,300
Iron	1,420 / 1,421	99.9%	6.7	6.7	6.7	11,004	1970	44,100	10,996	6.7	44,100
Lead	8,091 / 8,099	99.9%	2.0	1.1	4.1	559	0.52	54,800	558	0.5190	54,800
Magnesium	1,419 / 1,421	99.9%	17	2.7	31	3,043	420	21,000	3,039	2.7	21,000
Manganese	1,420 / 1,421	99.9%	0.1	0.1	0.1	519	91	8,300	519	0.1	8,300
Mercury	2,533 / 3,589	70.6%	0.1	0.005	0.3	0.7	0.01	395	0.5	0.01	395
Molybdenum	0 / 226	0.0%	1.0	1.0	1.0	NA	NA	NA	1.0	1.0	1.0
Nickel	4,998 / 5,003	99.9%	1.0	0.1	2.0	20	2.0	122	20	0.1	122
Potassium	1,416 / 1,421	99.6%	250	250	250	1,838	120	3,900	1,832	120	3,900
Selenium	489 / 4,988	9.8%	3.2	0.1	6.4	3.0	0.5	27	3.1	0.1	27
Silver	156 / 5,004	3.1%	0.7	0.1	1.3	1.4	0.2	30	0.8	0.1	30
Sodium	1,128 / 1,421	79.4%	46	26	60	547	2	14,100	444	2	14,100
Thallium	131 / 5,004	2.6%	2.8	0.1	6.4	6.0	0.9	54	2.9	0.1	54
Vanadium	1,420 / 1,421	99.9%	2.5	2.5	2.5	18	0.4	130	18	0.4	130
Zinc	4,999 / 5,003	99.9%	1.9	0.1	2.5	315	12	9,530	314	0.1	9,530

* Non-detects were evaluated at 1/2 the Quantitation limit
NA = value not available

Table C-3. Summary of Chemicals Analyzed by XRF and ICP

Analyte	XRF	ICP	Both
Aluminum		x	
Antimony	x	x	x
Arsenic	x	x	x
Barium	x	x	x
Beryllium		x	
Bismuth		x	
Cadmium	x	x	x
Calcium		x	
Chromium	x	x	x
Cobalt	x	x	x
Copper	x	x	x
Iron	x	x	x
Lead	x	x	x
Magnesium		x	
Manganese	x	x	x
Mercury	x	x	x
Molybdenum	x	x	x
Nickel	x	x	x
Rubidium	x		
Potassium		x	
Selenium	x	x	x
Silver	x	x	x
Strontium	x		
Sodium		x	
Thallium		x	
Tin	x		
Titanium	x		
Vanadium		x	
Uranium	x		
Zinc	x	x	x
Zirconium	x		

Table C-4. XRF Detection Limit Evaluation

ANALYTE	# Detects/ # Samples	Detection Frequency	Detection Limit (mg/kg)	Soil Screening Level [1] (mg/kg)		Detection Limit Adequate?
Antimony	223 / 78,072	0.3%	16 - 2,100	3.1		No
Arsenic	51,314 / 155,366	33%	1.9 - 270	0.4		No
Barium	5,459 / 51,479	11%	66 - 6,900	1,500		No
Cadmium	6,473 / 78,315	8%	9.3 - 1,100	7.0		No
Chromium	1,670 / 108,704	2%	26 - 3,000	280	[2]	No
Cobalt	37,489 / 99,309	38%	24 - 2,700	2.00		No
Copper	48,059 / 99,222	48%	5.8 - 440	310		No
Iron	99,883 / 99,888	100%	2,500 - 3,300	5,500		**Yes**
Lead	158,006 / 160,325	99%	3.3 - 380	400		**Yes**
Manganese	83,002 / 99,884	83%	30 - 2,700	180	[3]	**Yes**
Mercury	2,067 / 108,749	2%	1 - 320	2.3	[4]	No
Molybdenum	2,842 / 95,265	3%	1.3 - 140	39		No
Nickel	17,602 / 98,477	18%	7.2 - 680	160		No
Rubidium	99,670 / 99,849	100%	4 - 180	--		**Yes**
Selenium	237 / 108,360	0.2%	0.77 - 110	39		No
Silver	84 / 78,063	0.1%	6.9 - 750	39		No
Strontium	99,832 / 99,849	100%	4.6 - 46	--		**Yes**
Tin	18,604 / 77,381	24%	17 - 1,600	4,700		**Yes**
Titanium	77,717 / 78,123	99%	160 - 9,200	--		**Yes**
Uranium	19 / 2,261	1%	4.4 - 68	23	[5]	No
Zinc	151,663 / 155,882	97%	6.6 - 520	2,300		**Yes**
Zirconium	99,870 / 99,885	100%	3.9 - 160	--		**Yes**

-- = A soil screening level was not available for this chemical.

NOTES:

[1] USEPA. 2008c. Regional Screening Level Tables for Residential Soil. Values are based on a target cancer risk of 1E-06 or a noncancer HI of 0.1, whichever value is lower.

[2] RBC is based on Cr (VI), the most conservative screening level for chromium.

[3] Value is from IRIS and is based on the oral RfD of 1.4E-01 mg/kg-day in the diet. In accord with recommendations in IRIS, this value is modified by dividing by a Modifying Factor of 3 for application to exposures from soil or water.

[4] Merchloric chloride and inorganic salts.

[5] RBC is for insoluble salts.

Table C-5. Summary of XRF/ICP Correlation

Analyte	Outliers	Number of Data Pairs	R^2	Correlation Adequate?[1]
Iron	Included	401	0.19	No
	Excluded	397	0.22	No
Lead	Included	7,094	0.52	No
	Excluded	7,041	**0.87**	**Yes**
Manganese	Included	227	0.002	No
	Excluded	224	0.006	No
Rhubidium	Included	0	--	cannot be evaluated
	Excluded			
Strontium	Included	0	--	cannot be evaluated
	Excluded			
Tin	Included	0	--	cannot be evaluated
	Excluded			
Titanium	Included	0	--	cannot be evaluated
	Excluded			
Zinc	Included	3,959	**0.84**	**Yes**
	Excluded	3,896	**0.93**	**Yes**
Zirconium	Included	0	--	cannot be evaluated
	Excluded			

Boldface indicates analyte with an adequate XRF/ICP correlation (R^2 > 0.7).

ABBREVIATIONS:

XRF = X-Ray Fluorescence

-- = Not evaluated. A minimum of 10 data pairs are required for correlation analysis.

NOTES:

[1] Data are considered adequate if R^2 > 0.70.

Table C-6. XRF Data Quality Evaluation Summary

Analyte	Detection Frequency Adequate?	Correlation Adequate?	XRF Dataset Reliable?
Antimony	No	--	No
Arsenic	No	--	No
Barium	No	--	No
Cadmium	No	--	No
Chromium	No	No	No
Cobalt	No	--	No
Copper	No	--	No
Iron	**Yes**	No	No
Lead	**Yes**	**Yes**	**Yes**
Manganese	**Yes**	No	No
Mercury	No	--	No
Molybdenum	No	--	No
Nickel	No	--	No
Rubidium	**Yes**	unknown	--
Selenium	No	--	No
Silver	No	--	No
Strontium	**Yes**	unknown	--
Tin	**Yes**	unknown	--
Titanium	**Yes**	unknown	--
Uranium	No	--	No
Zinc	**Yes**	**Yes**	**Yes**
Zirconium	**Yes**	unknown	--

Boldface indicates an analyte that is considered adequate, and that is retained for further evaluation.

-- = Not evaluated.

Table C-7. Estimation of ICP-Equivalent Concentrations from XRF Data

Equation for Estimating ICP-Equivalent Concentrations:

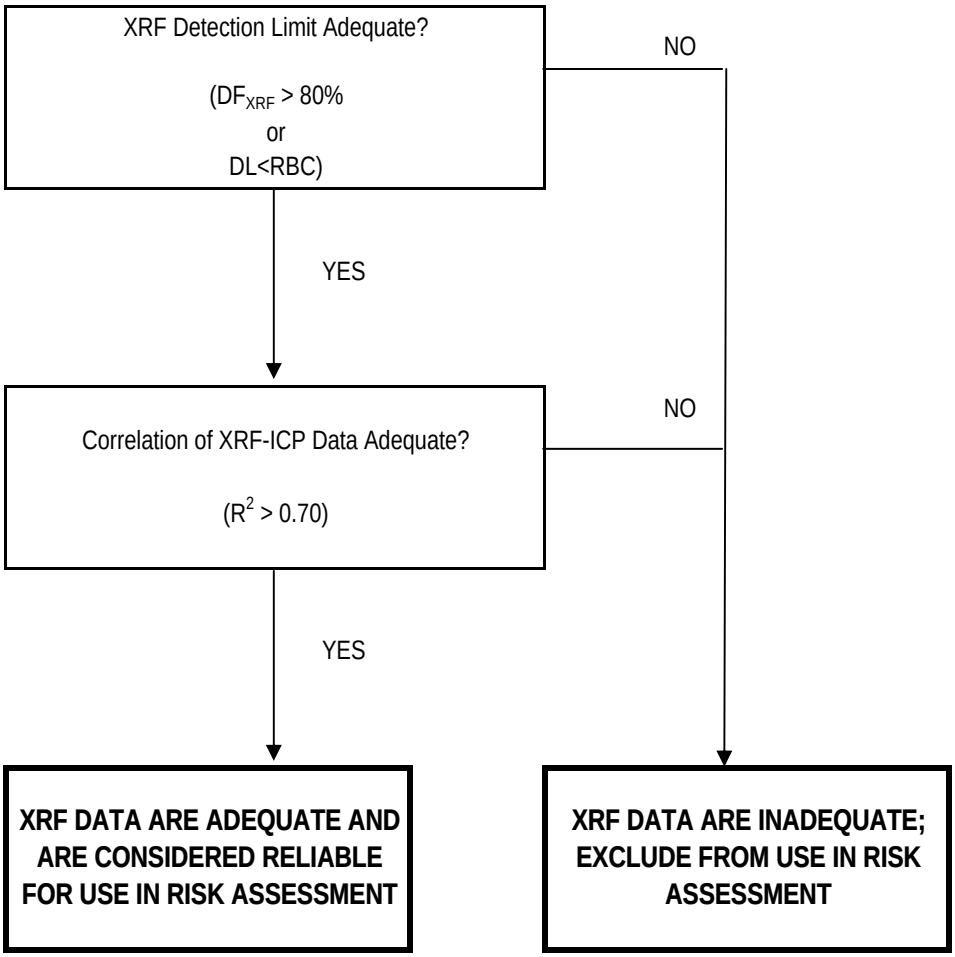
[ICP-equivalent concentration] = a + b * [XRF Concentration]

Parameters:

Analyte	Data Set	Outliers	Intercept (a)	Slope (b)
Lead	Full (all data)	Included	-3	1.47
		Excluded	-25	1.40
	Restricted (0 -1,000 mg/kg)	Included	0	1.14
		Excluded	**0**	**1.16**
Zinc	Full (all data)	Included	27	0.89
		Excluded	**23**	**0.89**

Boldface indicates parameters used to estimate ICP-equivalent concentrations in the risk assessment.

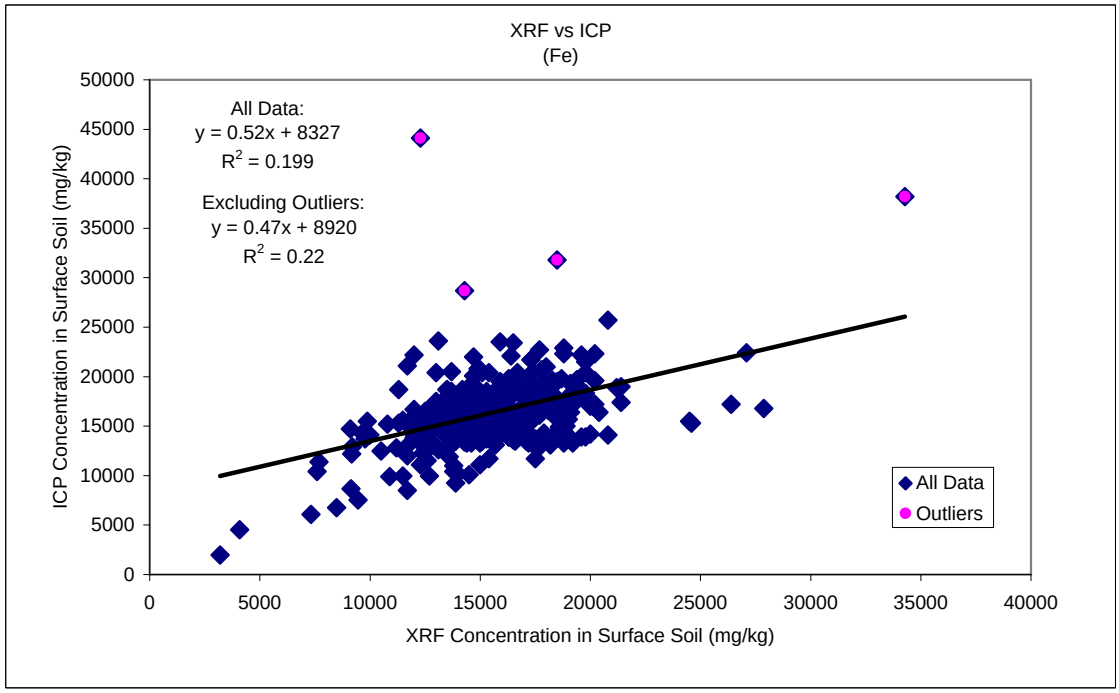
Figure C-1. Procedure for Evaluating the Adequacy of XRF Data for Use in the Risk Assessment



ABBREVIATIONS:

CV = Coefficient of Variation
DF = Detection Frequency
ICP = Inductively Coupled Plasma Atomic Emission Spectroscopy
RBC = Risk-Based Concentration
XRF = X-Ray Fluorescence

Figure C-2. Correlation of XRF and ICP Results for Iron



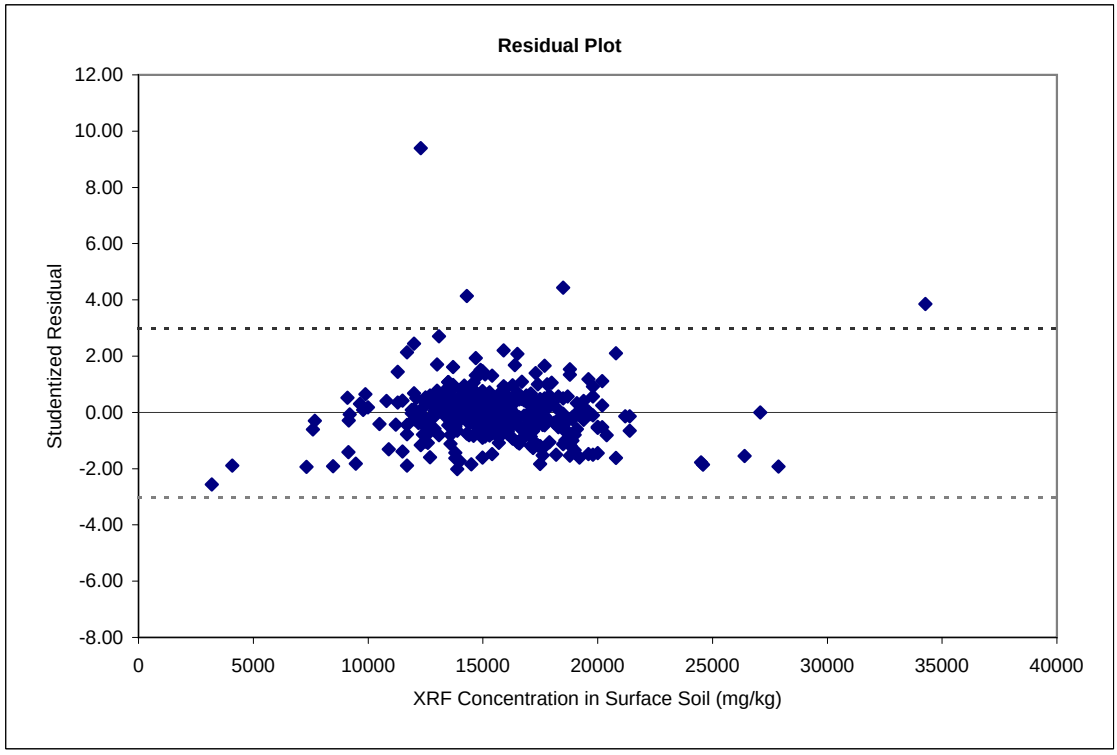
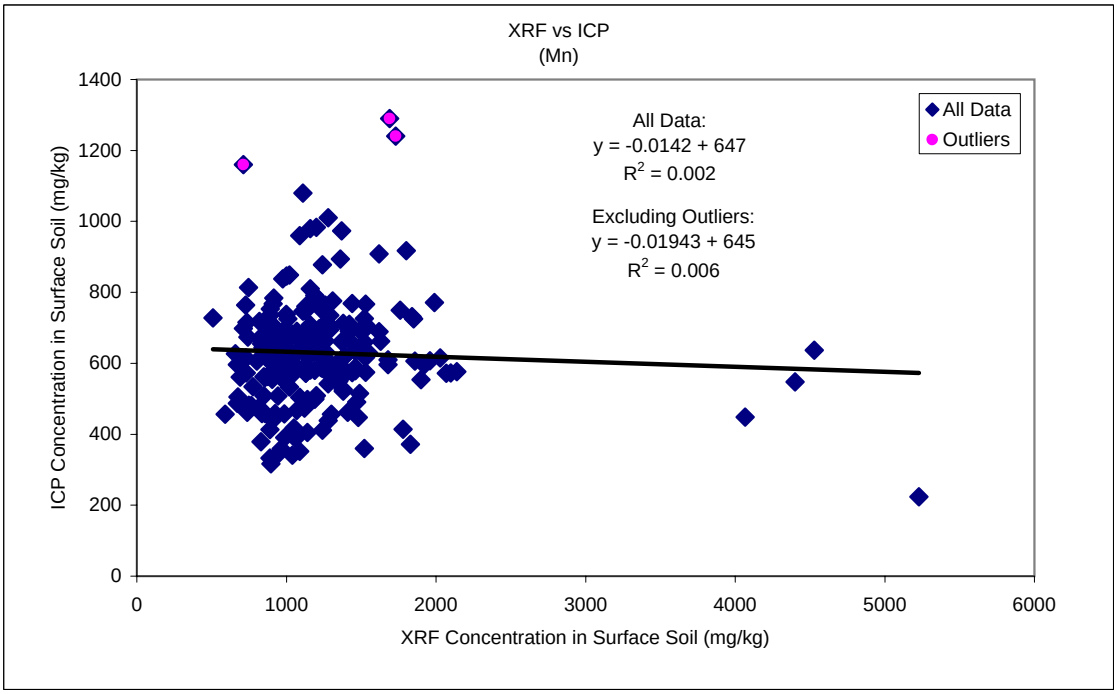


Figure C-3. Correlation of XRF and ICP Results for Manganese



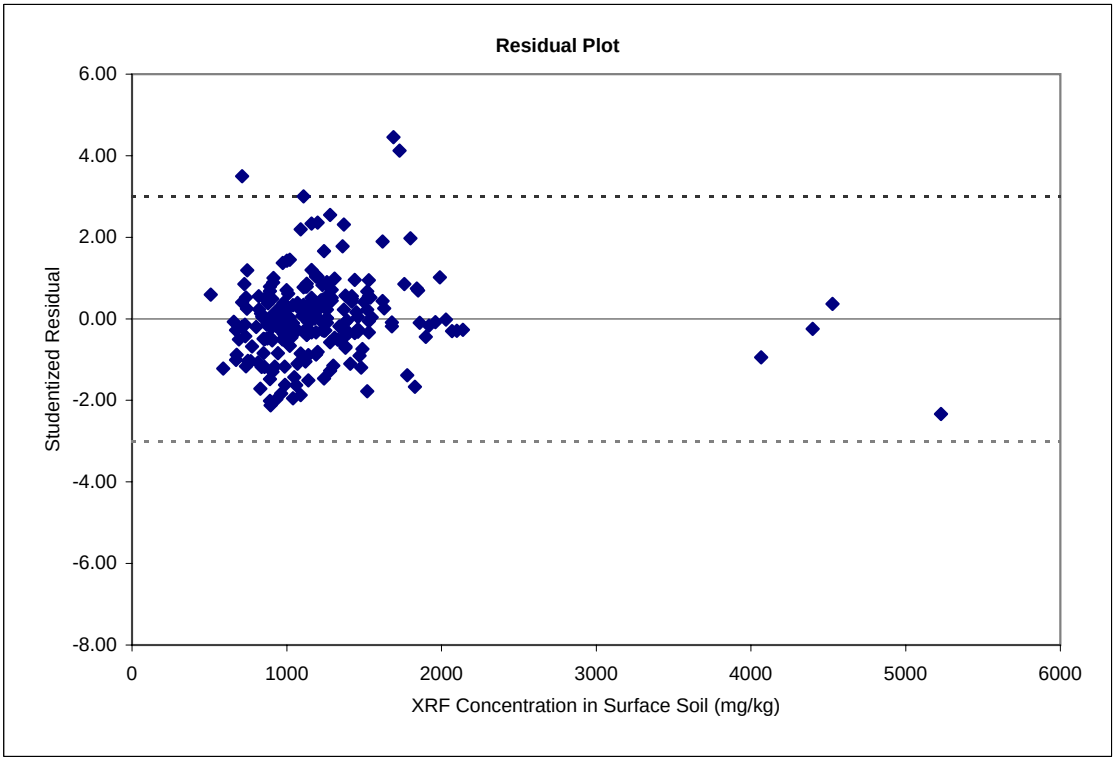
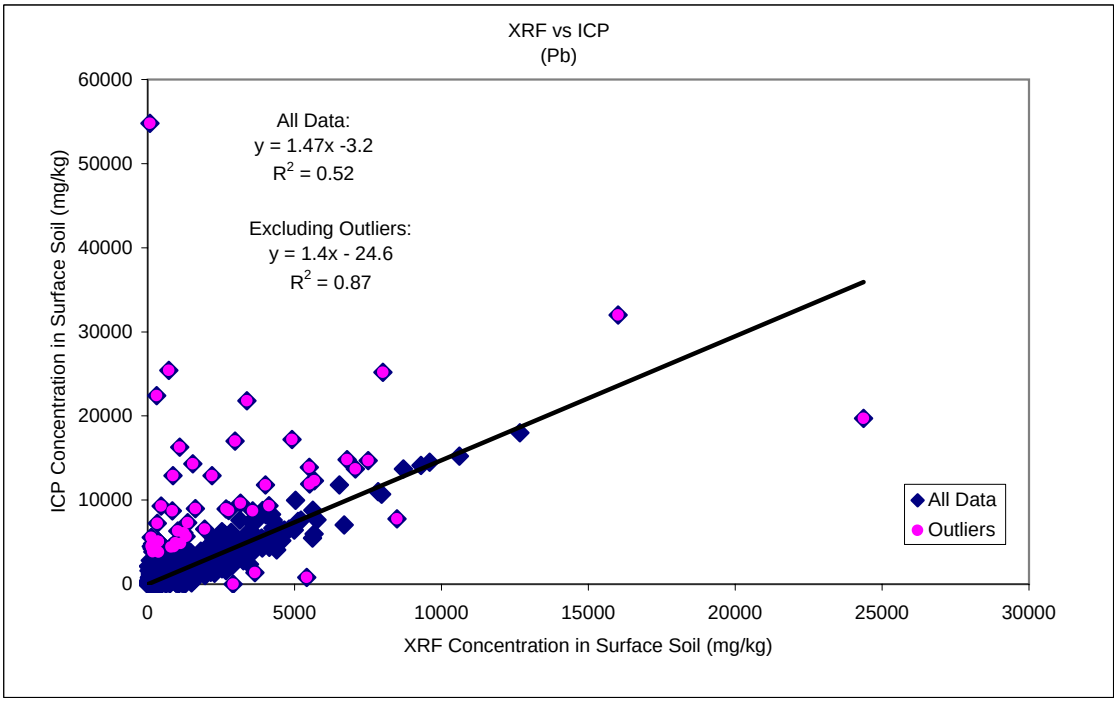


Figure C-4. Correlation of XRF and ICP Results for Lead



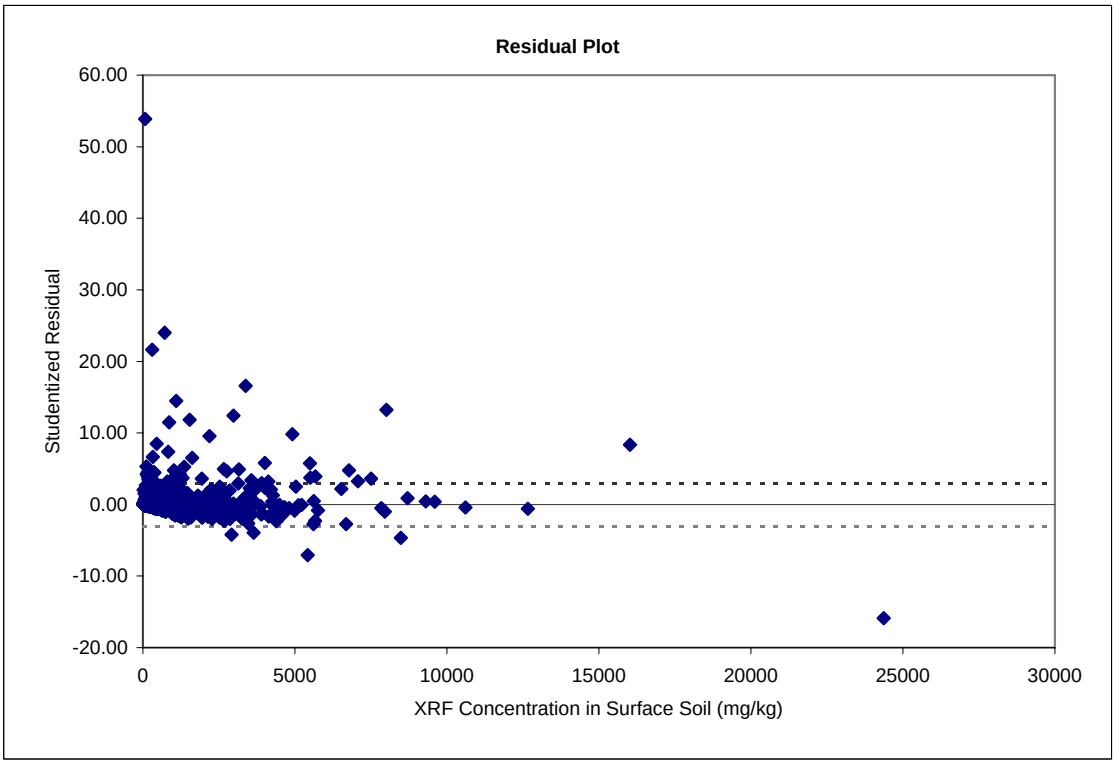
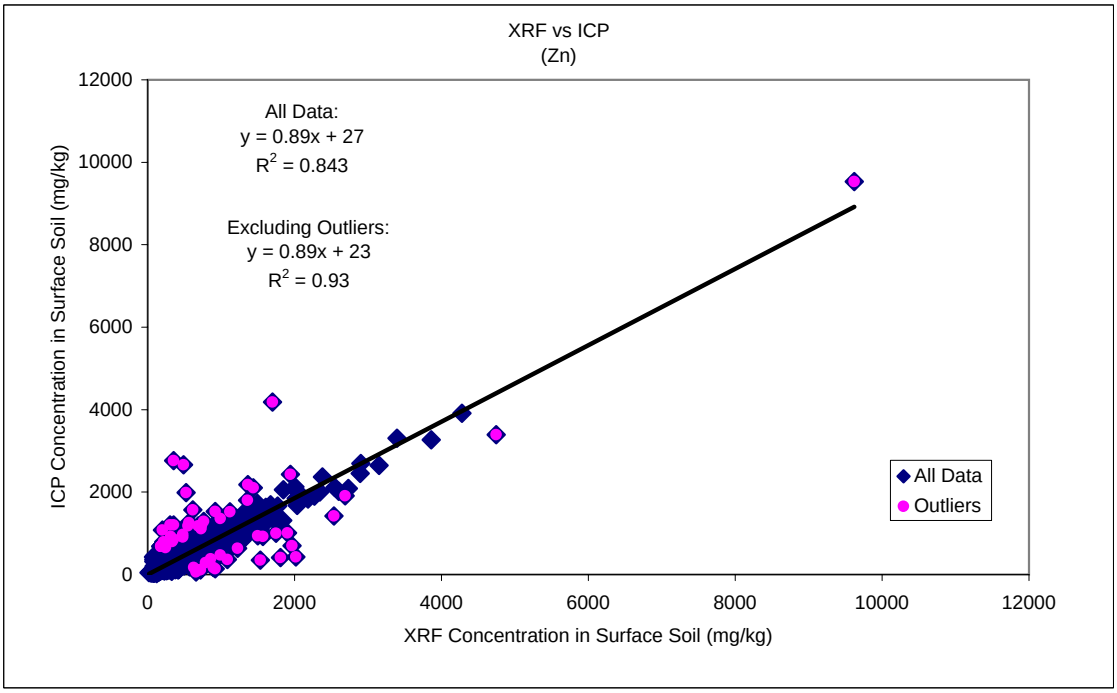


Figure C-5. Correlation of XRF and ICP Results for Zinc



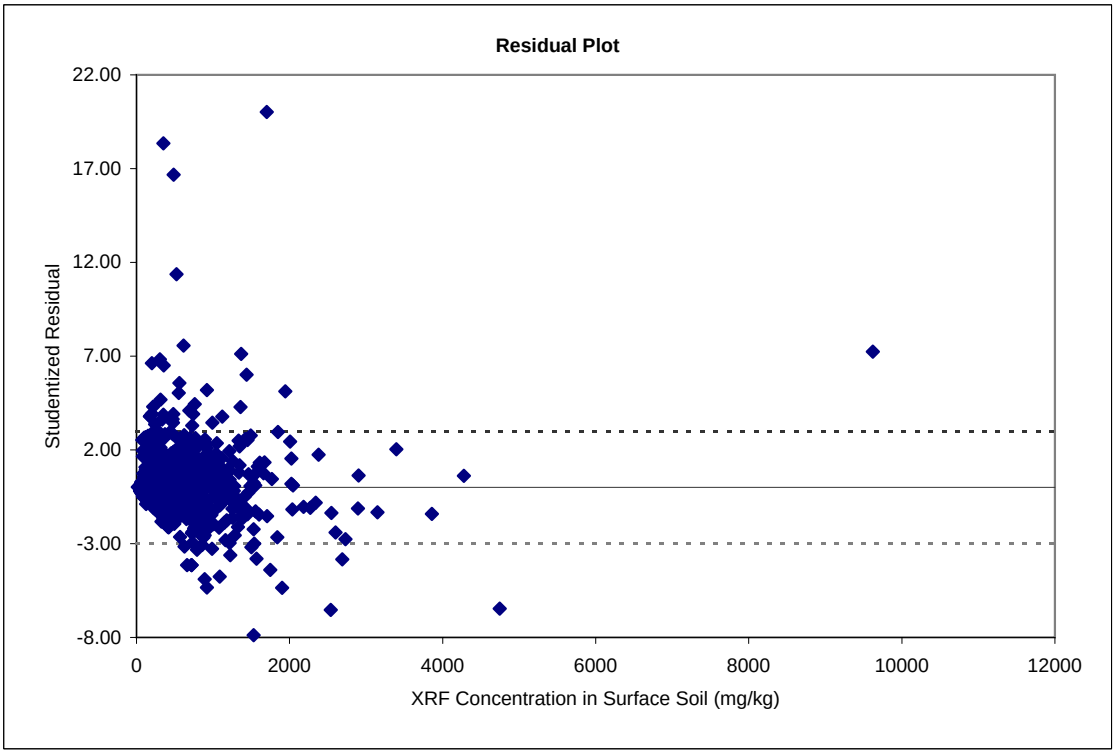
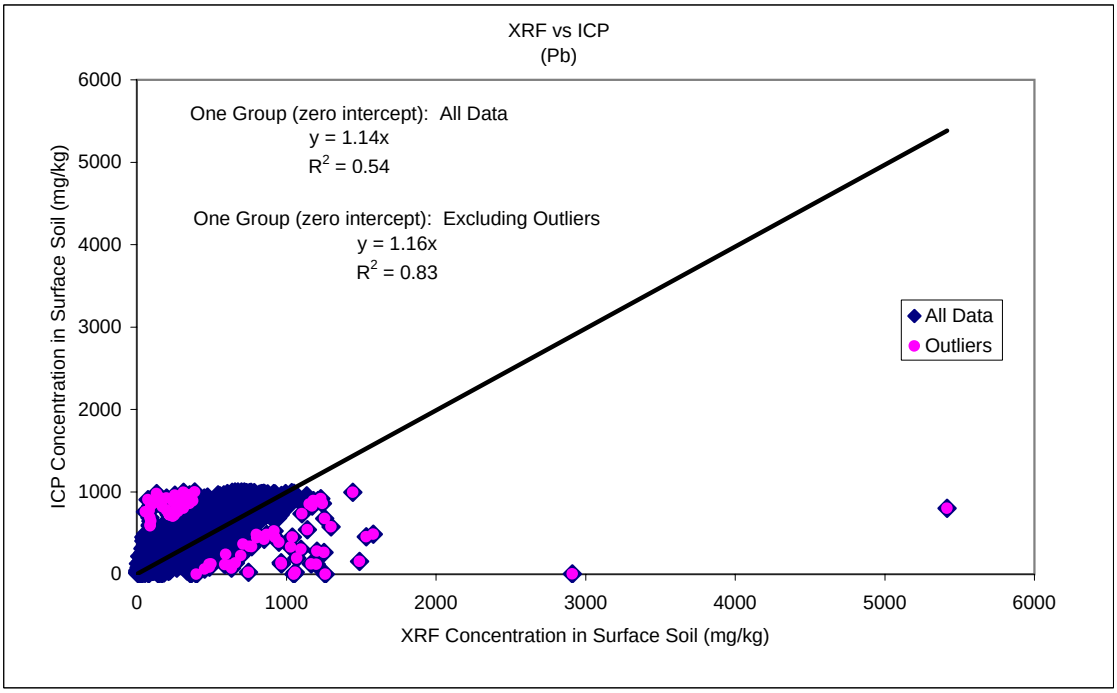
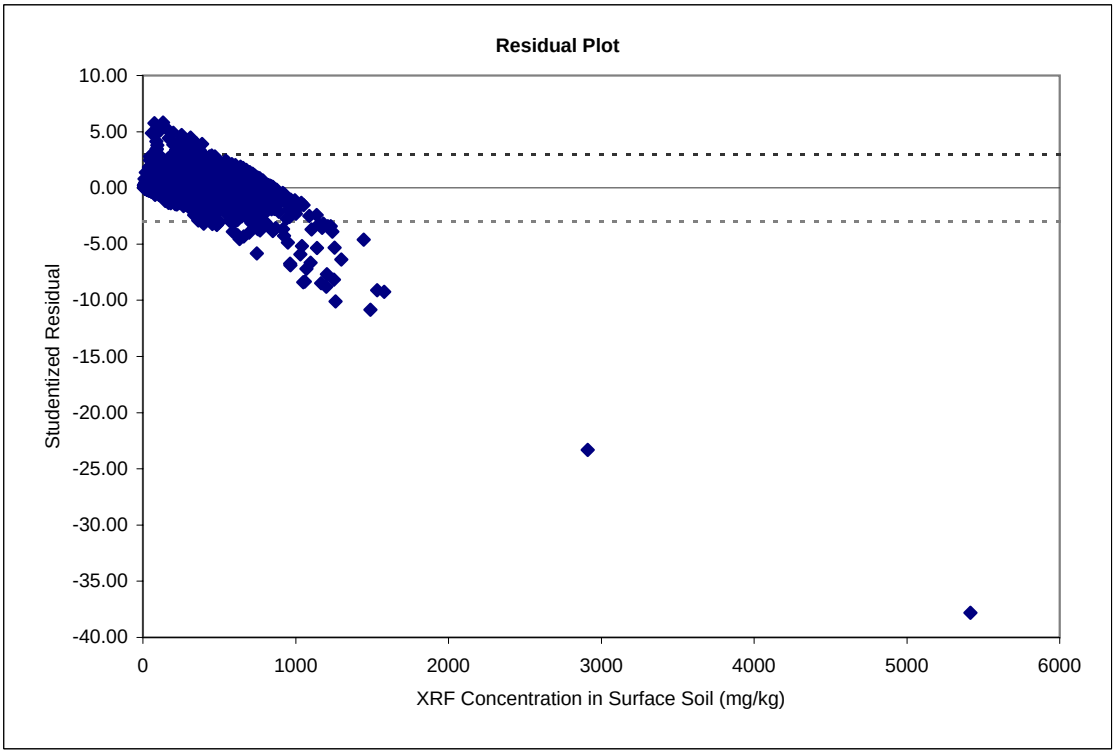


Figure C-6. Correlation of XRF and ICP Results for Lead
(restricted data set: 0 - 1,000 mg/kg ICP)





APPENDIX D

SCREENING LEVEL EVALUATION OF INHALATION OF DUST EXPOSURE PATHWAY

1.0 INTRODUCTION

This appendix presents a screening level evaluation of the inhalation of particulates from the air exposure pathway identified in the conceptual site model to determine if this pathway requires further evaluation in the risk assessment.

The screening level approach quantifies the dose of metals inhaled from particulates in air relative to the dose of metals ingested from soil.

The basic equation recommended by EPA (1989) for evaluation of inhalation exposure is:

$$DI_{air} = C_a \times BR_a \times EF \times ED/(BW \times AT)$$

where:

DI_{air}	=	Daily intake from air (mg/kg-d)
C_a	=	Concentration of substance in air (mg/m^3)
BR_a	=	Breathing rate of air (m^3/day)
EF	=	Exposure frequency (days/yr)
ED	=	Exposure duration (yrs)
BW	=	Body weight (kg)
AT	=	Averaging time (days)

and

$$C_a = PEF \times C_{soil}$$

where:

C_{soil}	=	Concentration of substance in soil (mg/kg)
PEF	=	Particulate Emission Factor (PEF), or soil to air transfer factor (kg/m^3)

The basic equation recommended by EPA (1989) for evaluation of soil ingestion is given by:

$$DI_{soil} = C_s \times IR_s \times EF \times ED/(BW \times AT)$$

where:

DI_{soil}	=	Daily intake from soil (mg/kg-d)
C_s	=	Concentration of substance in soil (mg/kg)
IR_s	=	Ingestion rate for soil (kg/day)
EF	=	Exposure frequency (days/yr)
ED	=	Exposure duration (yrs)

BW = Body weight (kg)
AT = Averaging time (days)

Based on the above equations, the relative magnitude of the inhaled dose of a COPC from air can be compared to the ingested dose from soil as follows:

Ratio (inhalation / ingestion) = PEF $\times$ BR_d / IR_s

Values for these parameters for each of the receptors identified in the conceptual model are summarized in Table D-1.

2.0 RESULTS

Table D-1 summarizes the ratio of the mass of soil inhaled to that ingested for each of the receptors identified in the conceptual model. As seen, the inhaled dose of soil from wind erosion is very small (<<1%) compared to the ingested dose, so the wind erosion pathway is not considered significant at this site.

3.0 REFERENCES

USEPA. 1989. Risk Assessment Guidance for Superfund. Volume I: Human Health Evaluation Manual Part A. Interim Final. Office of Solid Waste and Emergency Response (OSWER), Washington, DC. OSWER Directive 9285.701A.

USEPA. 1991. Human Health Evaluation Manual, Supplemental Guidance: "Standard Default Exposure Factors." Office of Emergency and Remedial Response. Washington, DC. OSWER Directive 9285.6-03.

USEPA. 1996. Soil Screening Guidance: User's Guide. Second Edition. Office of Solid Waste and Emergency Response. Washington, D.C. EPA/540/R-96/018. July.

USEPA. 1997. Exposure Factors Handbook, Volumes I, II, and III. U.S. Environmental Protection Agency, Office of Research and Development. EPA/600/P-95/002Fa.

TABLE D-1.
PATHWAY SCREENING INHALATION OF PARTICULATES RELATIVE TO SOIL INGESTION

Basic Equation: $DI_{air}/DI_{soil} = PEF \bullet BR_d/IR_s$

Air Source	Receptor	Input Parameters			Ratio	
		PEF[1] (kg/m^3)	BR_d[2] (m^3/day)	IR_s[3] (kg/day)	DI_{air}/DI_{soil}	DI_{air}/DI_{soil} (%)
wind erosion	Adult Resident	8E-10	20	1E-04	2E-04	0.015%
	Child Resident	8E-10	10	2E-04	4E-05	0.0038%

Note: RME exposure parameters are used in the calculations

PEF = Particulate Emission Factor (see Appendix F for derivation)
BR_d = Breathing rate of dust
IR_s = Soil Ingestion Rate
DI = Daily Intake (mg/kg-day)

[1] USEPA (1996) default PEF of 1.32E+09 m^3/kg, expressed as kg/m^3.
[2] USEPA (1991) recommended default inhalation rate for adults; USEPA (1997) recommended inhalation rate for children 6-8 years.
[3] USEPA (1991) recommended default soil ingestion rate for residents.

APPENDIX E

SELECTION OF CHEMICALS OF POTENTIAL CONCERN (COPCS)

Table E-1. Soil COPC Selection

CHEMICAL	DATA[1]												Detection Limit	Essential Nutrient		SOIL COPCs		
	Detection Frequency (%)	Maximum Detected Concentration (mg/kg)	Mean Nondetected Concentration (mg/kg)	Essential Nutrient w/o Toxicity Data (Yes/No) [2]	Max Daily Dose (mg/day) [3]	Accepted Daily Dose (mg/day) [4]	Soil RBC (mg/kg) [5]	Basis	Source	Does chemical have a toxicity value?	Is chemical detected?	Is Max Detect > RBC?	Is mean nondetected concentration < RBC?	Is compound a non-toxic essential nutrient?	Does Max Dose >> Accepted	Quant-itative COPC	Source of Uncertainty	Not a COPC
Aluminum	99.5%	18,400	2.1	No	--	--	7,700	NC	USEPA	Yes	Yes	Yes		--	--	X		
Antimony	23.6%	77	2.1	No	--	--	3.1	NC	USEPA	Yes	Yes	Yes		--	--	X		
Arsenic	91%	513	3.5	No	--	--	0.39	C	USEPA	Yes	Yes	Yes		--	--	X		
Barium	99.9%	815	2.9	No	--	--	1,500	NC	USEPA	Yes	Yes	No		--	--			X
Beryllium	21%	1.7	0.4	No	--	--	16	NC	USEPA	Yes	Yes	No		--	--			X
Bismuth	0%	--	2.5	No	--	--	--	--	--	No	--	--		No	--		X	
Cadmium	60%	12	0.5	No	--	--	7.0	NC	USEPA	Yes	Yes	Yes		--	--	X		
Calcium	99.5%	75,400	6.4	Yes	15	1,000	--	--	--	No	--	--		Yes	No			X
Chromium	97.9%	400	5.3	No	--	--	280	C	USEPA	Yes	Yes	Yes		--	--	X		
Cobalt	100%	13	--	No	--	--	2	NC	USEPA	Yes	Yes	Yes		--	--	X		
Copper	99.5%	1,310	0.1	No	--	--	310	NC	USEPA	Yes	Yes	Yes		--	--	X		
Iron	99.5%	44,100	6.7	No	--	--	5,500	NC	USEPA	Yes	Yes	Yes		--	--	X		
Lead	99.9%	6,570	1.9	No	--	--	400	NC	USEPA	Yes	Yes	Yes		--	--	X		
Magnesium	99.5%	10,600	2.7	Yes	2.1	400	--	--	--	No	--	--		Yes	No			X
Manganese	99.5%	1,270	0.1	No	--	--	180	NC	USEPA	Yes	Yes	Yes		--	--	X		
Mercury	71.8%	395	0.1	No	--	--	2.3	NC	USEPA	Yes	Yes	Yes		--	--	X		
Molybdenum	0%	--	1.0	No	--	--	39	NC	USEPA	Yes	No	--	Yes	--	--			X
Nickel	99.9%	122	1.0	No	--	--	160	NC	USEPA	Yes	Yes	No		--	--			X
Potassium	100%	3,710	--	Yes	0.74	3,500	--	--	--	No	--	--		Yes	No			X
Selenium	4%	27	3.8	No	--	--	39	NC	USEPA	Yes	Yes	No		--	--			X
Silver	2.5%	19	0.8	No	--	--	39	NC	USEPA	Yes	Yes	No		--	--			X
Sodium	100%	1,430	--	Yes	0.29	2,400	--	--	--	No	--	--		Yes	No			X
Thallium	3%	54	3.6	No	--	--	0.5	NC	USEPA	Yes	Yes	Yes		--	--	X		
Vanadium	100%	48	--	No	--	--	39	NC	USEPA	Yes	Yes	Yes		--	--	X		
Zinc	99.9%	2,760	1.7	No	--	--	2,300	NC	USEPA	Yes	Yes	Yes		--	--	X		

[1] Surface soil samples collected from resdiential properties (single family, multi-family and vacant lots) within the final focus area, excluding remediated properties. Data include all ICP results and XRF results for lead and zinc that have been adjusted, according to the XRF-ICP relationships presented in Appendix C.

[2] Based on USEPA (1994c), Table 1. Chemicals identified by USEPA as essential nutrients for which toxicity data were not available were assigned a value of "Yes". All other chemicals were assigned values of "No".

[3] Maximum expected dose for the maximally exposed receptor (child resident), see Table E-2 for calculations.

[4] Values are either Reference Daily Intake (RDI) or Daily Reference Value (DRV). RDIs replace the term "U. S. Recommended Daily Allowances" (introduced in 1973 as a reference value for vitamins, minerals, and protein). DRVs are for nutrients for which no set of standards previously existed. Values obtained from http://www.fda.gov/fdac/special/foodlabel/dvs.html.

[5] USEPA (2008c) default soil screening level for residential soil, based on a target cancer risk of 1E-06 and a target noncancer Hazard Quotient of 0.1 .

C = Cancer

COPC = Chemical of Potential Concern

NC = Noncancer

RBC = Risk Based Concentration

Table E-2. Evaluation of Essential Nutrients

Media	Maximally Exposed Receptor	Essential Nutrient	Maximum Concentration (C_{max})		RME Intake Rate (IR)		Maximum Daily Intake [1] (mg/day)	Accepted Daily Intake [2] (mg/day)		Ratio
			value	units	value	units		value	Source	
Residential Soil	Resident (child)	Calcium	75,400	mg/kg	200	mg/day	15.08	1,000	RDI	0.015
		Magnesium	10,600	mg/kg	200	mg/day	2.12	400	RDI	0.0053
		Potassium	3,710	mg/kg	200	mg/day	0.74	3,500	DRV	0.00021
		Sodium	1,430	mg/kg	200	mg/day	0.29	2,400	DRV	0.00012

[1] Calculated from maximum concentration and RME intake rate for the maximally exposed receptor (highest intake rate).
Max Daily Intake = C_{max} * IR. Conversion factors applied (as necessary) to yield daily intake in units of mg/day. Phosphorus in environmental media assumed to be present as phosphate. Maximum site concentration converted to phosphorus by multiplying by 0.316 (mass phosphorus/mass of phosphate).

[2] Valies are Reference Daily Intake (RDI) or Daily Reference Value (DRV). RDIs replace the term "U. S. Recommended Daily Allowances" (introduced in 1973 as a reference value for vitamins, minerals, and protein). DRVs are for nutrients for which no set of standards previously existed. Values obtained from http://www.fda.gov/fdac/special/foodlabel/dvs.html.

APPENDIX F

STATISTICAL METHODS USED TO FIT LINEAR MODELS FOR LEAD DATA

APPENDIX F
STATISTICAL METHODS USED TO FIT LINEAR MODELS FOR LEAD DATA

1.0 OVERVIEW

At the OLS site, data have been collected to evaluate three site-specific relationships that are needed for use in the lead risk assessment:

- The concentration of lead in soil when measured by XRF compared to that when measured by ICP ("XRF vs ICP")
- The concentration of lead in the fine fraction (< 250 µm) of soil compared to that in the bulk (< 2 mm) fraction ("Bulk vs Fine")
- The concentration of lead in indoor dust at a property compared to the average level in soil for that property ("Soil vs Dust")

All three of these relationships are expected to be characterized by a linear relationship:

- ICP = a1 + b1·XRF
- Fine = a2 + b2·Bulk
- Dust = a3 + b3·Soil

The parameters (slope and intercept) for linear models of this type are often estimated using ordinary linear regression (OLR). Use of OLR assumes that two conditions are satisfied: 1) the errors in y are normal and independent of the value of y (or x) (i.e., y is homoscedastic), and 2) there are no errors in the x variable. However, neither of these assumptions is likely to be true for the three data sets considered here. That is, the magnitude of the errors in y are likely to depend on the value of y (i.e., y is heteroscedastic), and there are likely to be errors in the value of x (measurement error). The occurrence of measurement error in x is especially important, because application of OLS to data sets that include measurement error will generally tend to underestimate the slope and overestimate the intercept.

There are a number of statistical methods that have been developed to deal with data that are either heteroscedastic and/or that include measurement error in x. These methods differ from each other in the degree to which they are statistically rigorous or are simplifications and approximations, and in the amount of information that is needed to implement the methods. The purpose of this appendix is to describe a set of seven alternative methods that were considered for use, describe an approach that was followed to evaluate which method was best for each data set, and give the results of the application of the selected method to each of the three data sets described above.

2.0 DESCRIPTION OF METHODS EVALUATED

A total of seven different approaches were considered in this evaluation, as described below.

2.1 Ordinary Linear Regression (OLR)

The parameters of the best fit OLR line to a data set are obtained by computing the squared error for each data pair and minimizing the sum of the squared errors. The solution for the intercept and slope are given by:

Slope = S_{xy} / S_{xx}
Intercept = $\overline{Y} - slope \cdot \overline{X}$

where:

$$S_{xy} = \sum_{i=1}^{N} \left(X_i - \overline{X}\right)\left(Y_i - \overline{Y}\right)$$

$$S_{xx} = \sum_{i=1}^{N} \left(X_i - \overline{X}\right)^2$$

$$\overline{Y} = \frac{\sum_{i=1}^{N} Y_i}{N}$$

$$\overline{X} = = \frac{\sum_{i=1}^{N} X_i}{N}$$

As noted above, while easy to implement, OLR is expected to yield slope estimates that are too low and intercepts that are too high when there is significant measurement error in x.

2.2 95% UCL on the OLR Slope

As described in (USEPA 1998), one potential approach for mitigating the tendency of OLR to underestimate slope is to use the 95% UCL on the OLR slope rather than the best estimate. The 95% UCL on slope is given by:

95UCL(slope) = Slope(best est) + $t_{n-2,0.95} \cdot (MSE / S_{xx})^{0.5}$

where:

$$MSE = \frac{\sum_{i-1}^{N} \left(y_i - a - b \cdot x_i \right)^2}{N-2}$$

The intercept is then computed as follows:

$$\text{Intercept} = \overline{Y} - slope \cdot \overline{X}$$

2.3 Chi-Square Merit Function

The most rigorous approach for dealing with data sets that may be either heteroscedastic and/or have measurement error in x is the Chi-Square Merit Function (Press *et al.* 1992):

$$X_i^2 = \frac{(y_i - a - b \cdot x_i)^2}{\sigma_{yi}^2 + b^2 \cdot \sigma_{xi}^2}$$

This approach weights the squared error for each data pair (i) in inverse proportion to the magnitude of the errors in both x(i) and y(i). The parameters of the regression line (slope, intercept) are found by minimizing the sum of the X_i^2 terms.

The difficulty in implementing this approach is the requirement for the $\sigma^2(x_i)$ and $\sigma^2(y_i)$ terms. Ideally, these would be derived by multiple measures of x(i) and y(i) in each sample "i". For example, this would involve collecting multiple rounds of indoor dust and outdoor soil samples at each of the study properties. However, such replicate measure are rarely performed at a property, and no data of this type are available for the three data sets of interest from the OLS site. Therefore, to implement the Chi-Square Merit Function approach, it is necessary to estimate the values of $\sigma^2(x_i)$ and $\sigma^2(y_i)$, as follows.

For all three data sets, the error in x was modeled as:

$$\sigma_{x(i)} = MAX\left[\sigma_x \min, k1 + k2 \cdot x(i)\right]$$

Note that, in this error model, the value of $\sigma_{x(i)}$ can not be smaller than $\sigma_x$min. This is important because true values of $\sigma_{x(i)}$ are unlikely to approach zero as the value of x approaches zero, and because the merit function would assign unreasonably high weights to data pairs with small x if $\sigma_{x(i)}$ were allowed to approach zero.

For the first two data sets (XRF vs ICP and Bulk vs Fine), the errors in x are simply the measurement errors of lead in soil when measured by XRF. The values of k1 and k2 were estimated from a data set of triplicate XRF measures of 6128 soil samples from a different site (Madison County Mines Site in Missouri). For each sample, the standard error of the mean of the three replicates is given by:

SEM(i) = Stdev(i) / sqrt(3)

A plot of SEM(i) vs Mean(i) is shown in Figure F-1. As seen, the data are clearly heteroscedastic (i.e., the error in x increases as a function of x), and the relationship is well characterized by a straight line with parameters:

k1 = 2.7
k2 = 0.0063

Based on this, the value of $\sigma_x$min is also set to 2.7.

For the third data set (Dust vs Soil), the errors in x arise both from measurement error and also from sampling error. The values of k1 and k2 were estimated from the OLS data set as follows. For each property that had four or more 5-point composite soil samples available, the mean and standard deviation were computed from the composite values. The standard deviation of the 5-point composites was adjusted by multiplying by the square root of 5/20 = 0.5 to account for the fact that the error being modeled is based on a 20-point composite rather than on 5-point composites. A plot of this estimated sampling/measurement error in the 20-point composite as a function of the mean soil level is shown in Figure F-2. As seen, there is a clear tendency towards heteroscedasticity, with a slope of about 0.28. However, the intercept based on the data is negative (an impossibility), so the intercept term was set to a value of 5 mg/kg. This is approximately the SEM that would be expected for a composite of 20 samples drawn from a lognormal distribution with a GSD of about 2. Based on this intercept, the slope is 0.22.

In all three cases, the error in y was modeled as follows:

$$\sigma_{y(i)} = MAX[\sigma_y \min, k3 + k4 \cdot y(i)]$$

As above, the error in y is limited by the value $\sigma_y$min to avoid unreasonable values of $\sigma_{y(i)}$ and to prevent inappropriate weights for low values of y.

The parameters k3 and k4 were estimated by dividing the data into a series of bins based on the value of x, with approximately 10 data pairs in each bin. For each bin, the standard deviation of the y values was computed and plotted as a function of the mean value of y for the bin. These data points were then fit to a linear model to estimate the values of k3 and k4.

stdev y = k3 + k4· $\bar{y}$

The value of $\sigma_y$min is set to 5 for XRF-ICP, 1 for Bulk-Fine, and 5 for Soil-Dust.

2.4 Geometric Mean Functional Relationship (GMFR)

Draper and Smith (1998) discuss the problem of fitting linear regression lines in the face of measurement error, and recommend an approach referred to as the geometric mean functional relationship. In this approach, the slope of the line is computed as:

$$\text{Slope} = \text{sqrt}(S_{yy}/S_{xx})$$

where:

$$S_{yy} = \sum_{i=1}^{N}\left(Y_i - \overline{Y}\right)^2$$

$$S_{xx} = \sum_{i=1}^{N}\left(X_i - \overline{X}\right)^2$$

Values of S_{xx} and S_{yy} are derived from the data. Note that this approach assumes homoscedasticity in both x and y.

2.5 Three-Group Approach

Draper and Smith (1998) also describe a simple alternative approximation method for estimation of the slope, as follows:

1. Divide the data into three groups of approximately equal number of samples, based on the values of x (low, medium, high).
2. Compute the center of mass (defined as $\overline{Y_i}, \overline{X_i}$) for both the first group) (i = 1) and the third group (i = 3).
3. Compute the slope of the line by drawing a line through these two centers of mass:
   $$slope = \left(\overline{Y}_3 - \overline{Y}_1\right)/\left(\overline{X}_3 - \overline{X}_1\right)$$
4. Calculate the intercept by requiring the line to pass through the overall center of mass $\left(\overline{Y}, \overline{X}\right)$:
   $$Intercept = \overline{Y} - Slope \cdot \overline{X}$$

2.6 One-Group Approach (zero intercept)

This approach is suggested by EPA Region 8 (USEPA 1995). This approach assumes the intercept is zero, and defines the slope as the line that passes through the overall center of mass:

$$slope = \frac{\overline{Y}}{\overline{X}}$$

2.7 One-Group Approach (non-zero intercept)

This approach, also identified as an alternative by Region 8 (USEPA 1995), first estimates the intercept by finding the average value of y for the first 10% of the x values:

$$Intercept = \frac{10}{N}\sum_{i=1}^{N/10} y_i$$

The slope is the found by requiring the line to pass through the overall center of mass $\left(\overline{Y}, \overline{X}\right)$:

$$Slope = \frac{\overline{Y} - Intercept}{\overline{X}}$$

3.0 EVALUATION OF METHOD PERFORMANCE

In order to evaluate which of these methods was most appropriate for evaluating the three data sets from the OLS, Monte Carlo simulations were performed as follows.

1. For each data set, define a set of assumed "true" values for N (number of data pairs), slope, intercept, and error terms in x and y that are selected to be similar to the actual site data set.
2. Draw a random data set of size N, and compute the slope using each of the 7 methods above. Divide the estimated slope by the "true" slope to obtain the "slope ratio" for that simulated data set.
3. Repeat step 2 many times, and plot the distribution of slope ratios for each of the methods.

The preferred method is the method that yields a distribution of slope ratios most nearly centered on 1.0, and that has the narrowest distribution.

3.1 Monte Carlo Simulation of ICP vs XRF Data Sets

For the ICP vs XRF data set, the Monte Carlo simulation was based on the following model of "truth":

$$Csoil_{true} \sim LN(\mu,\sigma)$$
$$IPC_{true} = Csoil_{true}$$
$$XRF_{true} = a + b \cdot ICP_{true}$$
$$ICP_{true} = (XRF_{true} - a) / b$$
$$XRF_{obs} \sim XRF_{true} + N(0,\sigma_x)$$
$$ICP_{obs} = ICP_{true} + N(0,\sigma_y)$$
$$\sigma_x = k1 + k2 \cdot x_{true}$$
$$\sigma_y = k3 + k4 \cdot y_{true}$$

Values for use in the Monte Carlo simulation were selected to be similar to the authentic site data set as follows:

Parameters	Values	Basis
N	9072	Number of paired XRF vs ICP data set for lead in soil at the OLS
a, b	0, 1.47	OLR parameters of the XRF vs ICP data set at the OLS
μ,σ	5.3, 1.29	Log-probability regression coefficients of the lead in soil data set for ICP samples with matching XRF values.
k1, k2	2.7, 0.0063	Parameter estimates for the Standard Error of the Mean (SEM) of lead measured by XRF, based on triplicate XRF analyses of soil at the Madison County Mines Site (see Figure F-1).
k3, k4	0, 0.1	Parameter estimates for the Standard Deviation of lead measured by ICP, based on Matrix Spike/Matrix Spike Duplicate (MS/MSD) analysis of lead in soil at the Omaha Lead Site (see Figure F-3).

3.2 Monte Carlo Simulation of Fine vs Bulk Data Sets

For the Fine vs Bulk data set, the Monte Carlo simulation was based on the following model of "truth":

$$Csoil_{true} \sim LN(\mu,\sigma)$$
$$Bulk_{true} = Csoil_{true}$$
$$Fine_{true} = a + b \cdot Bulk_{true}$$
$$Bulk_{obs} \sim Bulk_{true} + N(0,\sigma_x)$$
$$Fine_{obs} = Fine_{true} + N(0,\sigma_y)$$
$$\sigma_x = k1 + k2 \cdot x_{true}$$
$$\sigma_y = k3 + k4 \cdot y_{true}$$

Values were selected to be generally similar to the authentic site data as follows:

Parameters	Values	Basis
N	380	Number of paired Bulk vs Fine data set for lead in soil at the OLS.
a, b	0, 1.08	OLR parameters of the Bulk vs Fine data set at the OLS.
μ,σ	5.3, 0.96	Log-probability regression coefficients of the lead in soil data set measured by XRF for bulk samples with matching fine values.
k1, k2	2.7, 0.0063	Parameter estimates for the Standard Error of the Mean (SEM) of lead measured by XRF, based on triplicate XRF analyses of soil at the Madison County Mines Site (see Figure F-1).
k3, k4	2.7, 0.0063	Parameter estimates for the Standard Error of the Mean (SEM) of lead measured by XRF, based on triplicate XRF analyses of soil at the Madison County Mines Site (see Figure F-1).

3.3 Monte Carlo Simulation of Dust vs Soil Data Sets

For the Indoor Dust vs Outdoor Soil data set, the Monte Carlo simulation was based on the following model of "truth":

$$Csoil_{true} \sim UNIFORM(min, max)$$
$$Cdust_{true} = a + b \cdot Csoil_{true}$$
$$a \sim LN(\mu,\sigma)$$
$$b \sim BETA(p1,p2)$$
$$Csoil_{obs} \sim Csoil_{true} + N(0,\sigma_x)$$
$$Cdust_{obs} = Cdust_{true} + N(0,\sigma_y)$$
$$\sigma_x = k1 + k2 \cdot x_{true}$$
$$\sigma_y = k3 + k4 \cdot y_{true}$$

Values were selected to be generally similar to the authentic site data as follows:

$N = 98$
$min = 50$
$max = 2000$
$\mu = 4.62$
$\sigma = 0.57$
$p1 = 4$
$p2 = 3$
$k1 = 20$
$k2 = 0.24$
$k3 = 20$
$k4 = 0.36$

Parameters	Values	Basis
N	9072	Number of paired Soil vs Dust data set for lead in soil at the OLS
min, max	50, 2,000	Minimum and maximum XRF concentration of lead in soil in the soil-dust data set at the OLS
μ,σ	4.62, 0.57	Lognormal distribution parameters for the intercept parameter (a) at the soil-indoor dust relationship at the OLS. Values are based on professional judgment and correspond to linear space values of 120 and 75.
p1, p2	4, 3	Beta distribution parameters for the b parameter in the soil-indoor dust relationship at the OLS. Values are based on professional judgment and yield a distribution with mean = 0.57 and 90% of values between 0.27 and 0.85.
k1, k2	5, 0.22	Parameter estimates for the Standard Error of the Mean (SEM) of lead in soil. Estimated from OLS soil dataset from the yard-wide average concentration of lead in soil vs the SEM of lead in soil (see Figure F-2).
k3, k4	5, 0.36	Parameter estimates for the SEM lead in indoor dust. Estimated from OLS indoor dust dataset (see Figure F-4)

4.0 RESULTS

4.1 Preferred Method and Parameter Estimates for ICP vs XRF Data Set

Figure F-5 plots the distribution of slope ratios for each of the seven alternative statistical methods applied to data sets that are similar to the site-specific ICP vs XRF data set. As seen, Approach 6 (one group, zero intercept) yields the best results, although Approach 2 is also reasonable.

Based on this, the authentic ICP vs XRF data from the OLS site were fit using Method 6. The results are shown in Figure F-6. Based on the best fit, data points whose studentized residual was larger than 3 or less than -3 were identified as potential outliers (Canavos 1984), and the analysis was repeated after exclusion of these data pairs. The resulting equation after exclusion of outliers is:

$$\text{ICP-Equivalent} = 1.46 \cdot \text{XRF}$$

This process was then repeated for a sub-set of the data where the concentration by ICP was ≤ 1000 mg/kg). This evaluation was performed because this is the range of soil lead concentrations of primary concern for decision-making at the site. That is, soil concentrations > 1000 mg/kg (whether by XRF or ICP) are above a level of concern for residential exposures and will be remediated. Thus, it is most important to accurately characterize the correlation in the range of 0-1000 mg/kg. These results are shown in Figure F-7.

As seen, after exclusion of outliers, the best fit equation is:

$$\text{ICP-Equivalent} = 1.16 \cdot \text{XRF}$$

This equation is selected for use in this risk assessment.

4.2 Preferred Method and Parameter Estimates for Bulk vs Fine Data Set

Figure F-8 (Panel A) plots the distribution of slope ratios for each of the seven alternative statistical methods applied to data sets that are similar to the site-specific Bulk vs Fine data set. An expanded view is provided in Panel B. As seen, Approach 6 (one group, zero intercept) yields the best results, although Approach 2 is also reasonable.

Based on this, the authentic Bulk vs Fine data from the OLS site were fit using Method 6. The results are shown in Figure F-9. Based on the best fit, data points whose studentized residual was larger than 3 or less than -3 were identified as potential outliers (Canavos 1984), and the analysis was repeated after exclusion of these data pairs. The resulting equation after exclusion of outliers is:

$$\text{Fine} = 1.04 \cdot \text{Bulk}$$

4.3 Preferred Method and Parameter Estimates for Soil vs Dust Data Set

Figure F-10 plots the distribution of slope ratios for each of the seven alternative statistical methods applied to data sets that are similar to the site-specific Dust vs Soil data set. As seen, none of the approaches yields totally un-biased results, but the best result is achieved using Approach 2 (95% UCL of the OLR slope).

Based on this, the authentic ICP vs XRF data from the OLS site were fit using Method 2. The results are shown in Figure F-11. Based on the best fit, data points whose studentized residual was larger than 3 or less than -3 were identified as potential outliers (Canavos 1984), and the analysis was repeated after exclusion of these data pairs. The resulting equation after exclusion of outliers is:

$$\text{Dust} = 42 + 0.74 \cdot \text{Soil}$$

5.0 REFERENCES

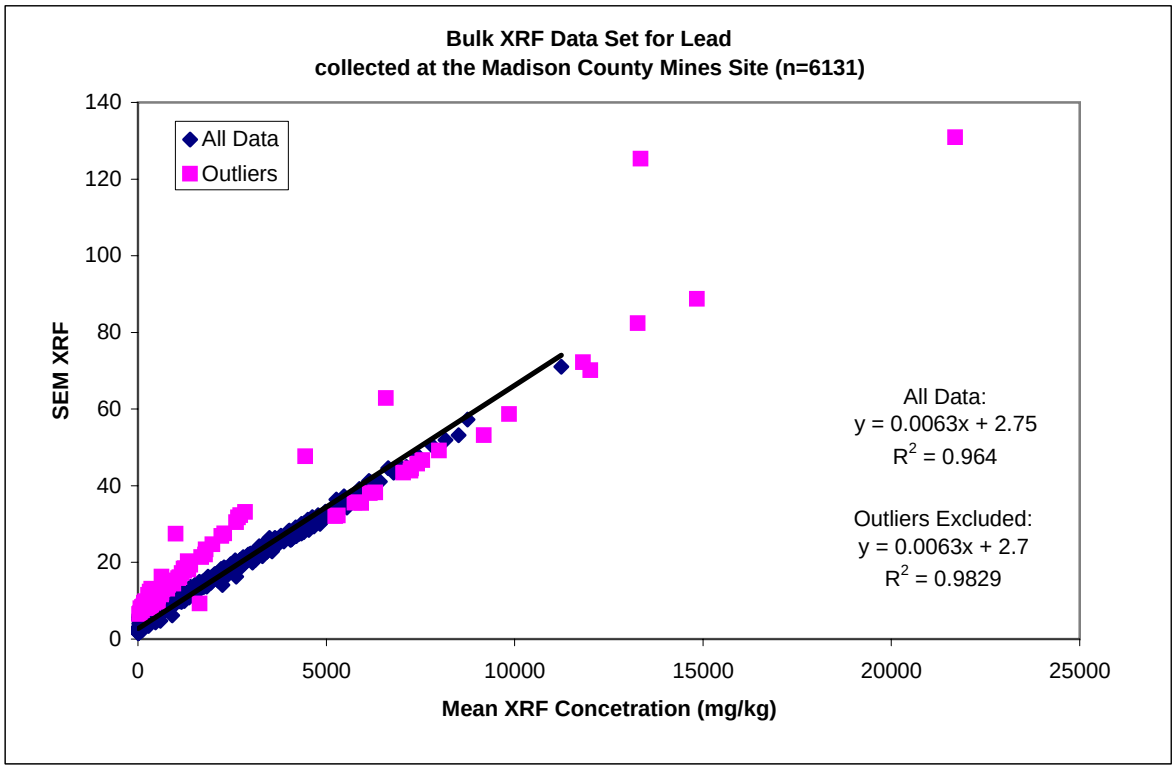
Canavos G.C. 1984. Applied Probability and Statistical Methods. Little, Brown and Company, Boston.

Press W.H., Teukolsky S.A., Vettering, W.T., and Flannery B.P. 1992. Numerical Recipes in C. The Art of Scientific Computing. Second Edition. Cambridge University Press.

USEPA. 1995. Estimating Site-Specific Exposure to Contaminants in Indoor Dust. Region 8 Superfund Technical Guidance No. FA-06: Exposure to Indoor Dust. September.

USEPA. 1998. Short Sheet: IEUBK Model Mass Fraction of Soil in Indoor Dust (Msd) Variable. Office of Solid Waste and Emergency Response. Washington, DC. EPA#540-F-00-008. June.

FIGURE F-1. SIGMA X ESTIMATE FOR LEAD IN SOIL MEASURED BY XRF



SUMMARY OUTPUT - Excluding Outliers

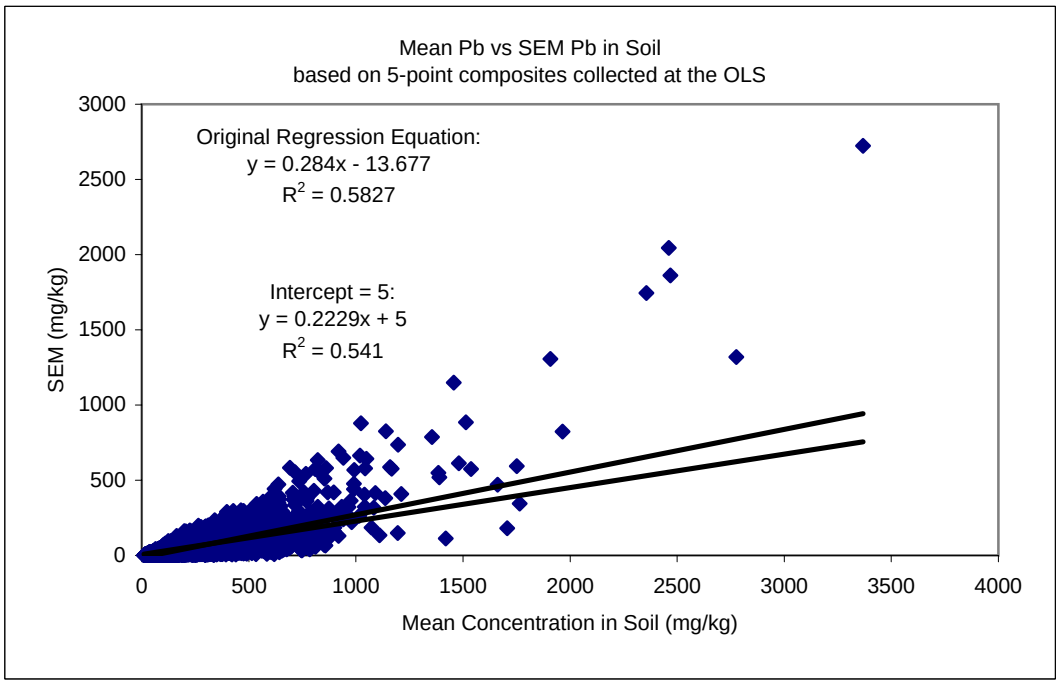
Regression Statistics	
Multiple R	0.991438
R Square	0.982949
Adjusted R	0.982947
Standard E	0.619337
Observatio	6128

ANOVA

	df	*SS*	*MS*	*F*	*ignificance F*
Regressior	1	135463.4	135463.4	353157.1	0
Residual	6126	2349.8	0.383578		
Total	6127	137813.2			

	Coefficients	*andard Err*	*t Stat*	*P-value*	*Lower 95%*	*Upper 95%*	*ower 95.0%*	*Jpper 95.0%*
Intercept	2.692332	0.008526	315.7671	0	2.675618	2.709047	2.675618	2.709047
X Variable	0.006348	1.07E-05	594.2702	0	0.006327	0.006369	0.006327	0.006369

FIGURE F-2 - ESTIMATE OF SIGMA X FOR LEAD IN BULK SOIL



SUMMARY OUTPUT - Original Regression

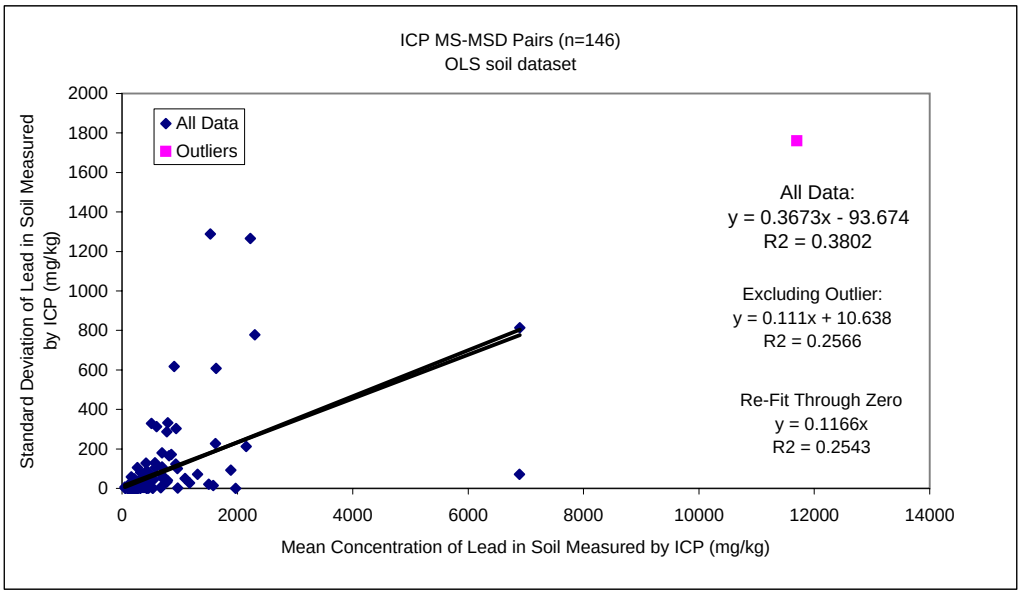
Regression Statistics	
Multiple R	0.763318
R Square	0.582655
Adjusted R	0.582638
Standard E	35.12674
Observatio	24673

ANOVA

	df	*SS*	*MS*	*F*	*ignificance F*
Regressior	1	42499017	42499017	34443.18	0
Residual	24671	30441248	1233.888		
Total	24672	72940265			

	Coefficients	*andard Err*	*t Stat*	*P-value*	*Lower 95%*	*Upper 95%*	*ower 95.0%*	*Jpper 95.0%*
Intercept	-13.67668	0.376451	-36.33054	1.4E-281	-14.41455	-12.93881	-14.41455	-12.93881
X Variable	0.284023	0.00153	185.5887	0	0.281024	0.287023	0.281024	0.287023

FIGURE F-3. SIGMA Y ESTIMATE FOR LEAD IN SOIL MEASURED BY ICP



SUMMARY OUTPUT

Regression Statistics	
Multiple R	0.506604
R Square	0.256648
Adjusted R	0.251821
Standard E	161.5376
Observatio	156

ANOVA

	df	*SS*	*MS*	*F*	*Significance F*
Regressior	1	1387430.4	1387430.4	53.16966073	1.50336E-11
Residual	154	4018537.7	26094.401		
Total	155	5405968.2			

	Coefficient	*standard Erro*	*t Stat*	*P-value*	*Lower 95%*	*Upper 95%*	*ower 95.0%*	*Upper 95.0%*
Intercept	10.63848	15.189852	0.7003674	0.484754639	-19.36889369	40.64585	-19.36889	40.64585
X Variable	0.110985	0.0152206	7.2917529	1.50336E-11	0.080916661	0.141053	0.080917	0.141053

REFIT THROUGH ZERO

SUMMARY OUTPUT

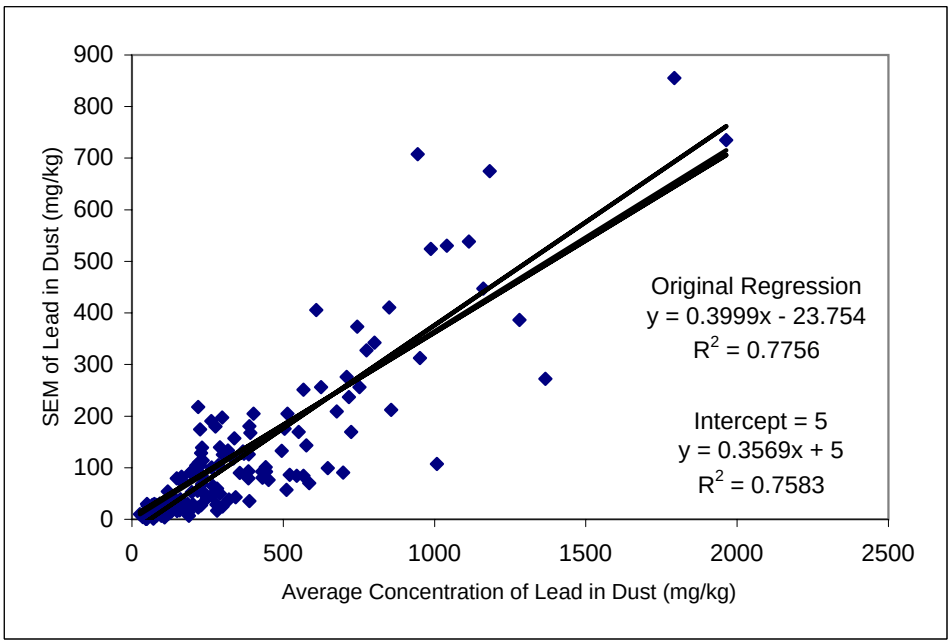
Regression Statistics	
Multiple R	0.586284
R Square	0.343729
Adjusted R	0.337277
Standard E	161.2719
Observatio	156

ANOVA

	df	*SS*	*MS*	*F*	*Significance F*
Regressior	1	2111455.4	2111455.4	81.18287983	7.48463E-16
Residual	155	4031337.4	26008.629		
Total	156	6142792.8			

	Coefficient	*standard Erro*	*t Stat*	*P-value*	*Lower 95%*	*Upper 95%*	*ower 95.0%*	*Upper 95.0%*
Intercept	0	#N/A	#N/A	#N/A	#N/A	#N/A	#N/A	#N/A
X Variable	0.116575	0.0129382	9.0101543	7.19099E-16	0.0910173	0.142133	0.091017	0.142133

FIGURE F-4 - SIGMA Y ESTIMATE FOR LEAD IN INDOOR DUST



SUMMARY OUTPUT - Original Regression

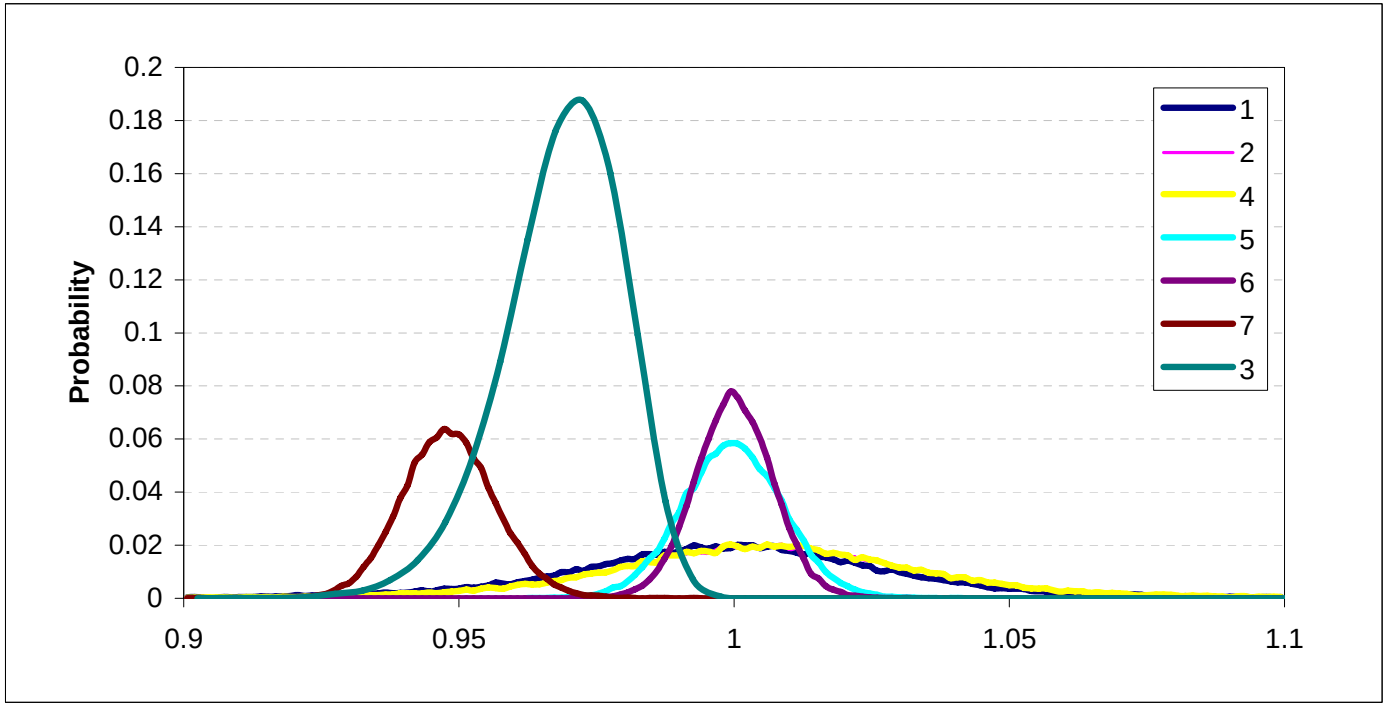
Regression Statistics	
Multiple R	0.880661
R Square	0.775564
Adjusted R	0.774106
Standard E	72.35289
Observatio	156

ANOVA

	df	*SS*	*MS*	*F*	*ignificance F*
Regression	1	2785842	2785842	532.1631	7.87E-52
Residual	154	806180.8	5234.94		
Total	155	3592023			

	Coefficients	*tandard Erro*	*t Stat*	*P-value*	*Lower 95%*	*Upper 95%*	*ower 95.0%*	*Upper 95.0%*
Intercept	-23.75433	8.343464	-2.847059	0.005015	-40.23674	-7.271919	-40.23674	-7.271919
X Variable	0.399934	0.017337	23.06866	7.87E-52	0.365685	0.434182	0.365685	0.434182

FIGURE F-5. XRF vs ICP SIMULTION RESULTS FOR LEAD

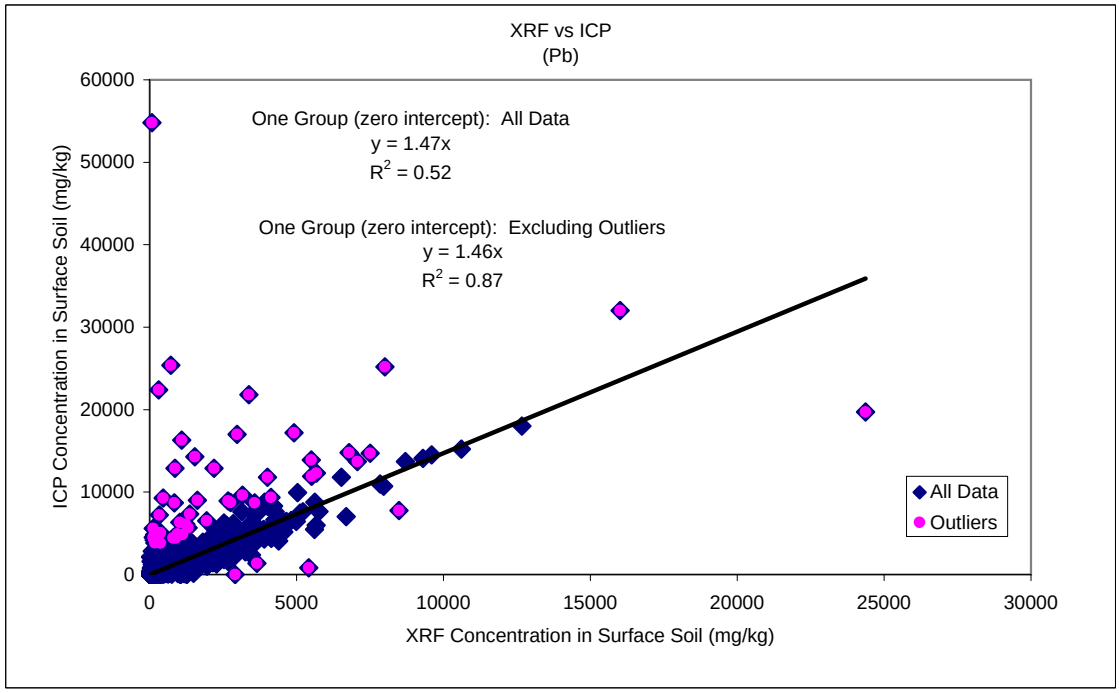


1 = Ordinary Linear Regression (OLR)
2 = 95th UCL of the OLR Mean
3 = Chi Square Merit Function
4 = Geometric Mean Functional Relationship
5 = Three Group
* 6 = One Group (zero intercept)
7 = One Group (non-zero intercept)

**recommended for use in the risk assessment*

Case1_XRF-ICP_lognormal x_v7.xls: CHART

Figure F-6. Correlation of XRF and ICP Results for Lead



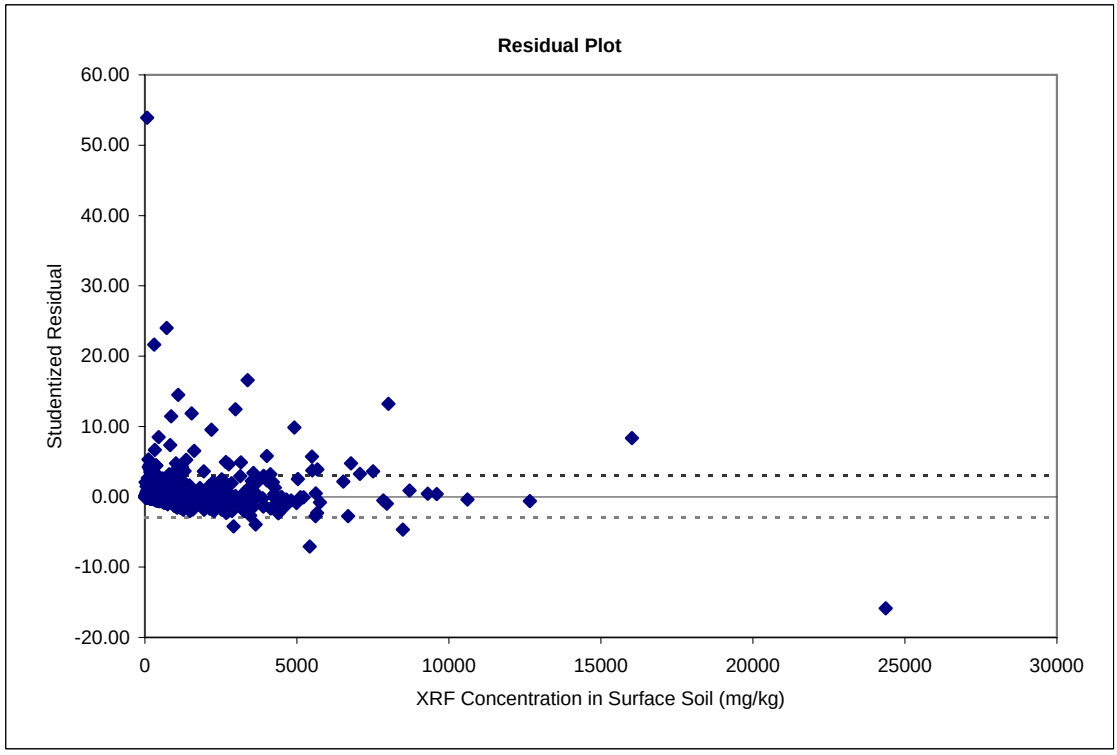
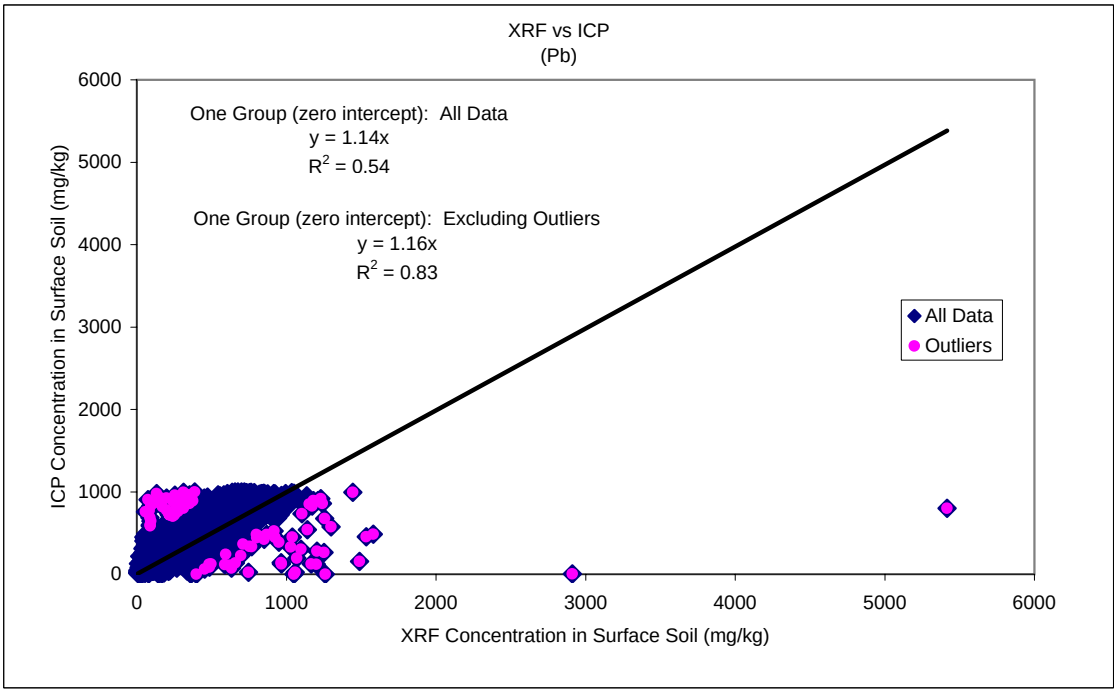


Figure F-7. Correlation of XRF and ICP Results for Lead
(restricted data set: 0 - 1,000 mg/kg ICP)



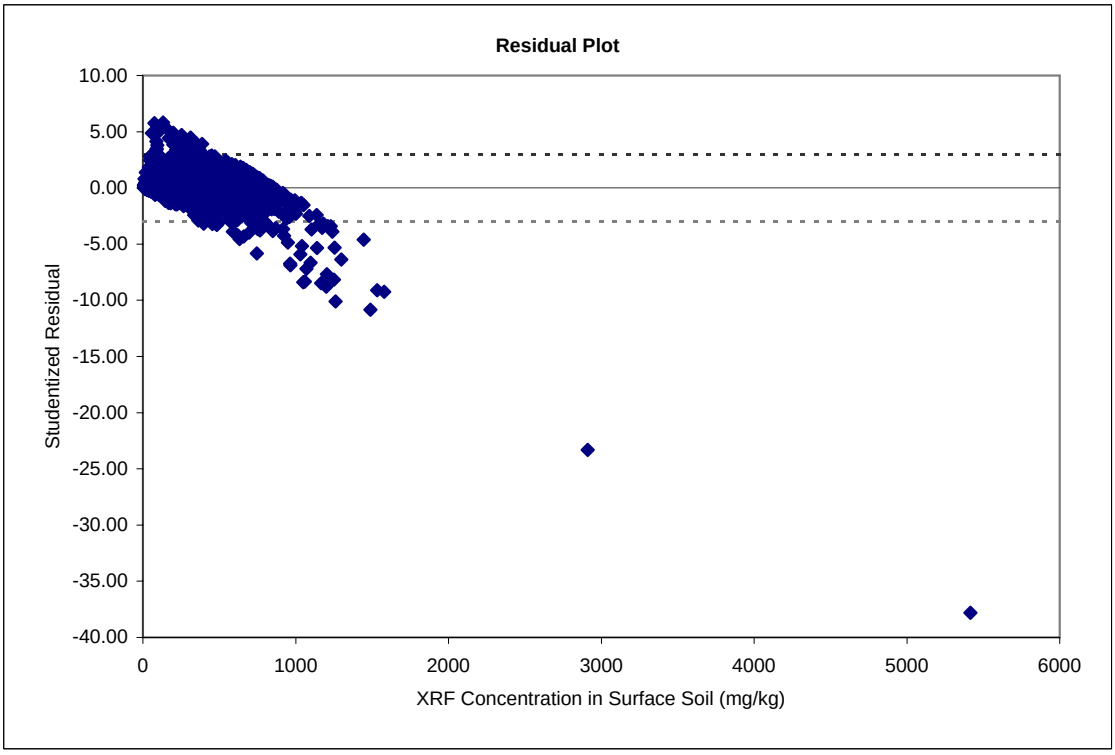
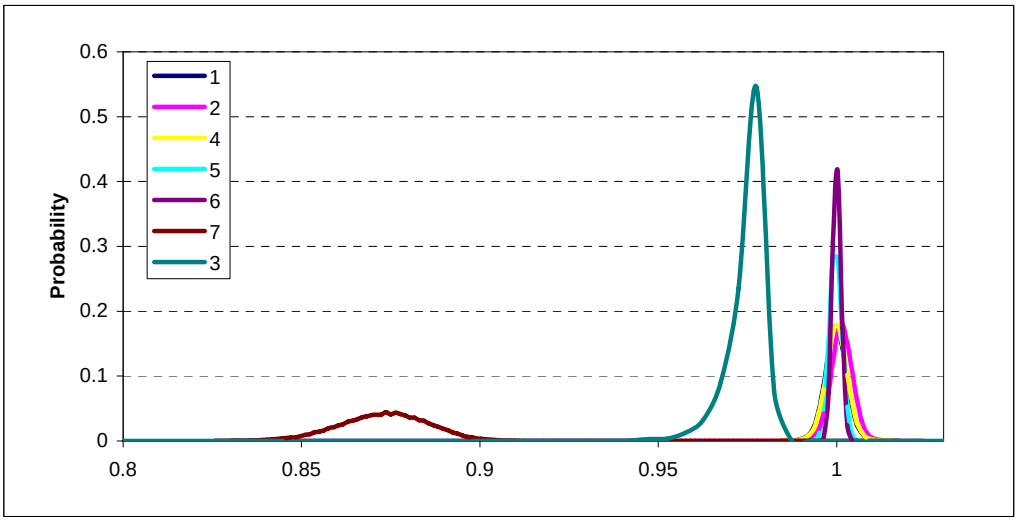
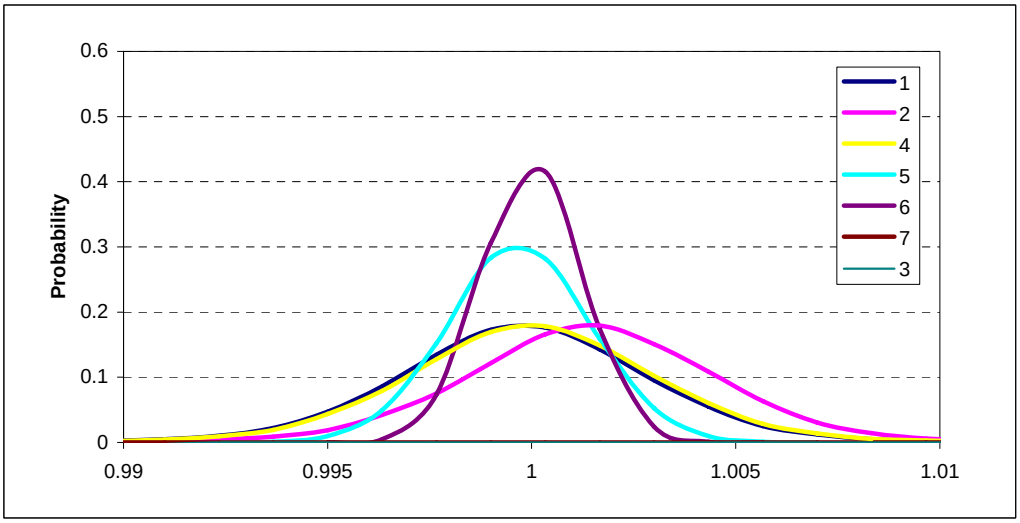


FIGURE F-8. BULK vs FINE SIMULATION RESULTS FOR LEAD

PANEL A.



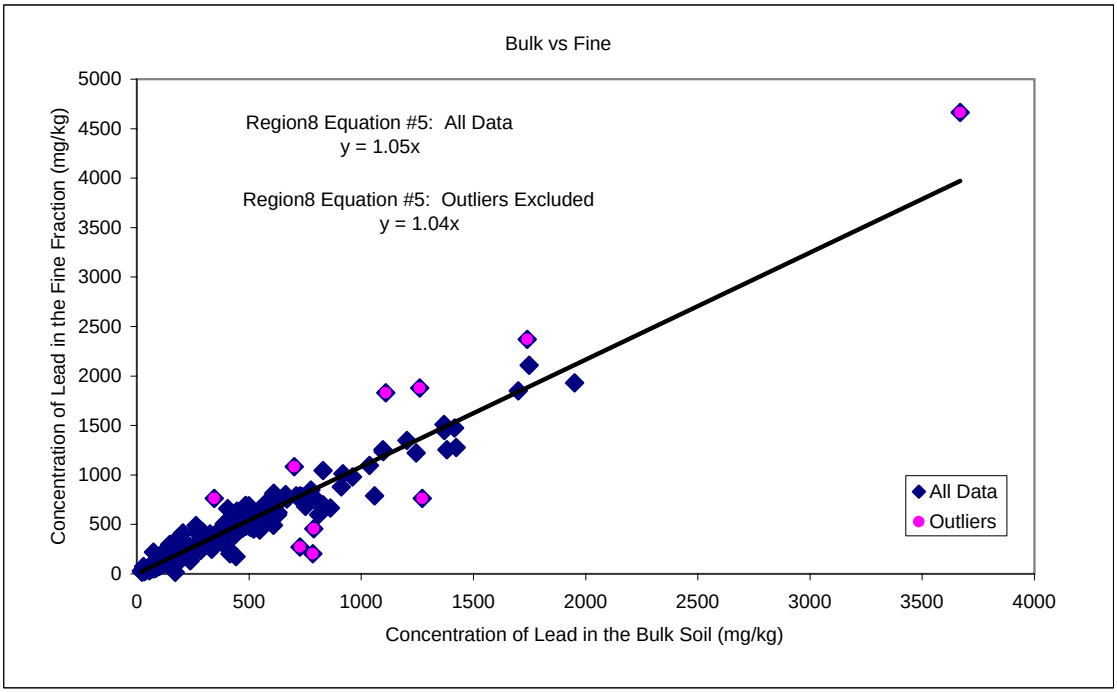
PANEL B.



1 = Ordinary Linear Regression (OLR)
2 = 95th UCL of the OLR Mean
3 = Chi Square Merit Function
4 = Geometric Mean Functional Relationship
5 = Three Group
*6 = One Group (zero intercept)
7 = One Group (non-zero intercept)

* recommended for use in the risk assessment

FIGURE F-9. BULK VS. FINE RELATIONSHIP FOR LEAD



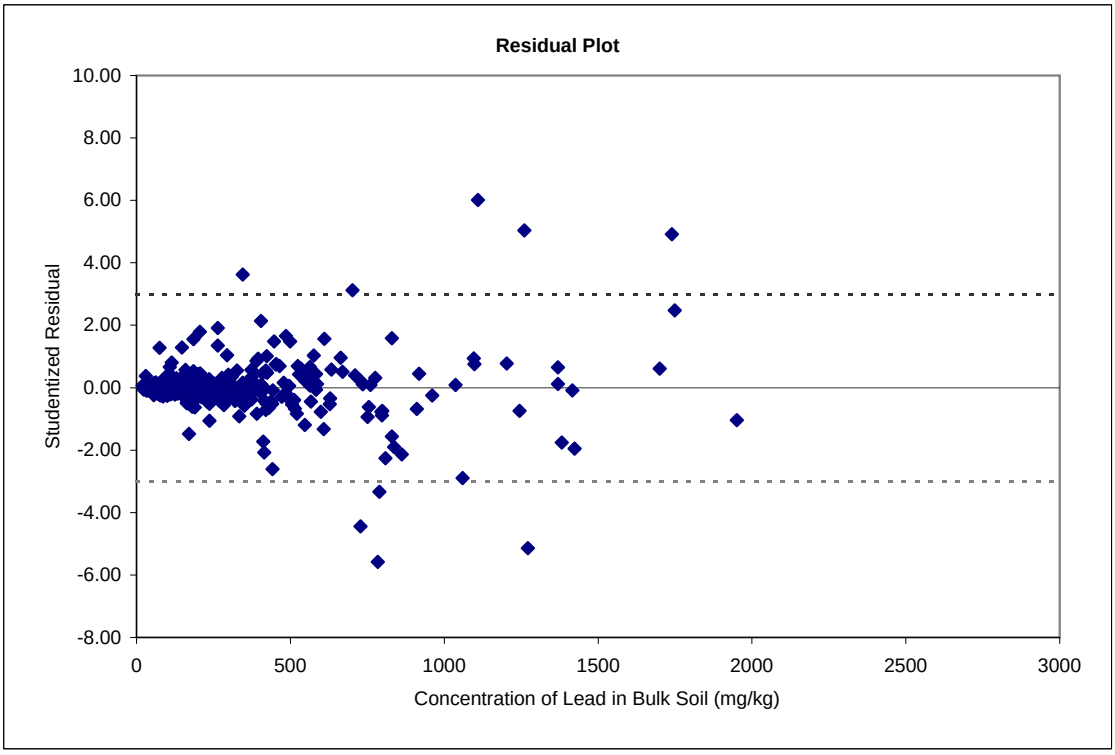
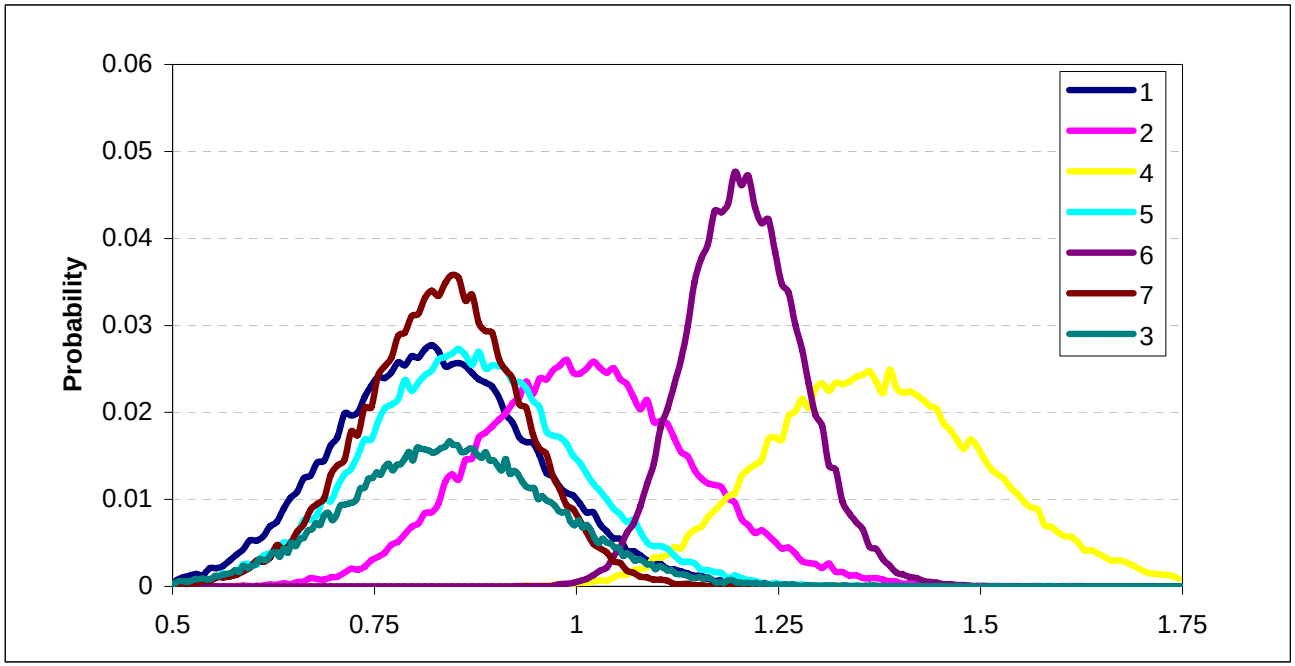


FIGURE F-10. SOIL VS DUST SIMULATION RESULTS FOR LEAD

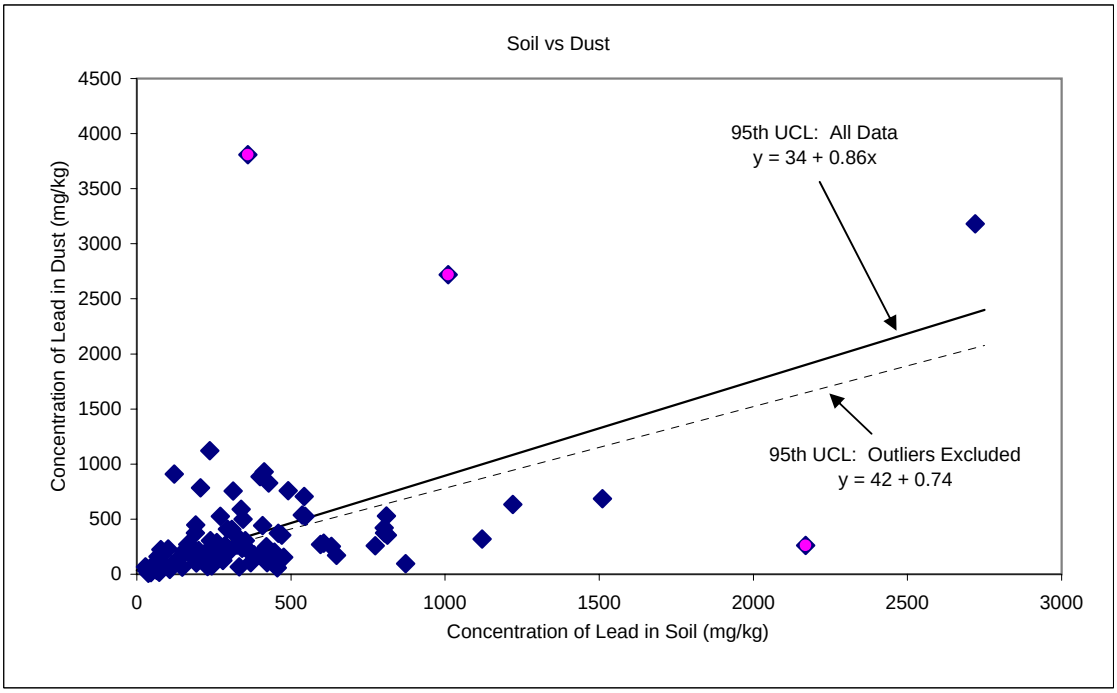


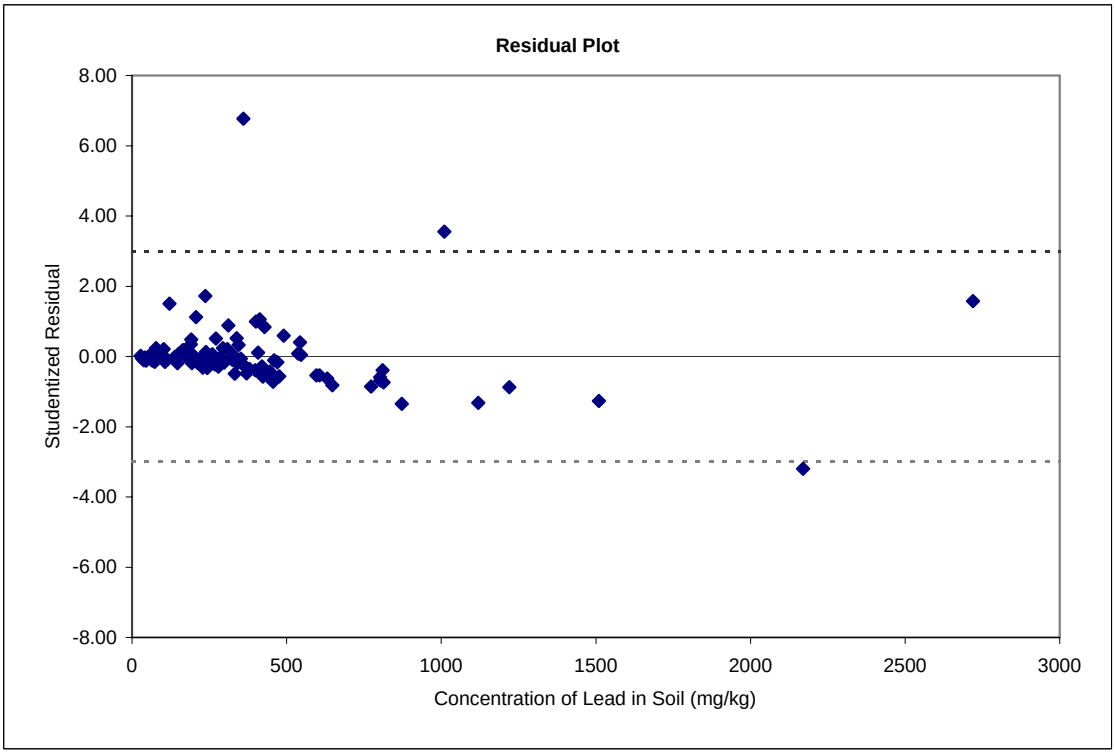
1 = Ordinary Linear Regression (OLR)
*2 = 95th UCL of the OLR Mean
3 = Chi Square Merit Function
4 = Geometric Mean Functional Relationship
5 = Three Group
6 = One Group
7 = One Group (non-zero intercept)

**recommended for use in risk assessment*

Case3_Soil-Dust_uniform x_v6.xls: CHART

FIGURE F-11. SOIL VS. DUST RELATIONSHIP FOR LEAD





APPENDIX G

Exposure Point Concentrations

Table G-1. **Exposure Point Concentrations for Lead in Soil and Dust**
Table G-2. **Exposure Point Concentrations for Non-Lead COPCs in Soil (except for Zinc)**
Table G-3. **Exposure Point Concentrations for Zinc in Soil**

see attached CD

APPENDIX H

DETAILED RISK CALCULATIONS
AND
RAGS PART D TABLES

SEE ATTACHED CD