Lead Human Exposure and Health Risk Assessments and Ecological Risk Assessment for Selected Areas

Pilot Phase

EXTERNAL REVIEW DRAFT TECHNICAL REPORT

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PREFACE

This document has been prepared in support of the U.S. Environmental Protection Agency's (EPA) review of the National Ambient Air Quality Standards (NAAQS) for lead (Pb). This document describes the pilot phase of t the human exposure, health risk and ecological risk assessments. The purpose of the pilot phase is to test the exposure and risk assessment methodologies for the purposes of gaining insights into their performance and to solicit comments from the Clean Air Scientific Advisory Committee (CASAC) and the public on the use of these methodologies in the full-scale analysis. Accordingly, the exposure and risk results presented in this document should be considered provisional and are subject to updating as part of the full-scale analysis.

These analyses have involved the implementation of multiple models and the evaluation of their performance is continuing beyond the release of this report. This is particularly the case for models not previously employed for applications such at those pertaining to this review (e.g., human blood Pb and risk models).

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LIST OF EXHIBITS

1. Introduction

Sections 108 and 109 of the Clean Air Act (CAA) govern the establishment and periodic review of the National Ambient Air Quality Standards (NAAQS) by the U.S. Environmental Protection Agency (EPA). These standards are established for pollutants that may reasonably be anticipated to endanger public health and welfare, and whose presence in the ambient air results from numerous or diverse mobile or stationary sources. The NAAQS are to be based on air quality criteria, which are to accurately reflect the latest scientific knowledge useful in indicating the type and extent of identifiable effects on public health or welfare that may be expected from the presence of the pollutant in ambient air. The EPA Administrator is to promulgate and periodically review "primary" (health-based) and "secondary" (welfare-based) NAAQS for such pollutants. Based on periodic reviews of the air quality criteria and standards, the Administrator is to make revisions in the criteria and standards, and promulgate any new standards, as may be appropriate. The CAA also requires that an independent scientific review committee advise the Administrator as part of this NAAQS review process, a function performed by the Clean Air Scientific Advisory Committee (CASAC).

The EPA is presently conducting a review of the NAAQS for lead (Pb). The EPA's overall plan and schedule for this NAAQS review is presented in the *Plan for the Review of the National Ambient Air Quality Standards for Lead* (USEPA 2006a). This plan discusses the preparation of two key documents in the NAAQS review process: an Air Quality Criteria Document (CD) and a Staff Paper. The CD provides a critical assessment of the latest available scientific information upon which the NAAQS is to be based (USEPA 2006b). The Staff Paper evaluates the policy implications of this information and presents staff conclusions and recommendations for standard-setting options for the Administrator to consider.

In conjunction with preparation of the first draft of the Staff Paper for Pb, EPA's Office of Air Quality Planning and Standards (OAQPS) has conducted various policy-relevant assessments, including pilot-scale quantitative human exposure and health risk assessments and an ecological risk assessment for selected areas. The methods and results of these assessments are summarized in Chapters 4 and 6 of the draft Staff Paper (USEPA 2006c) and are described in detail in this draft document.

Chapter 2 of this document provides an overview of the methods and data used for the pilot phase human exposure and health risk assessments and ecological risk assessment. An overview of the selected case studies, including descriptions of the locations, populations in the surrounding areas, and available emissions estimates and environmental measurements, is provided in Chapter 3. The methods and data used to estimate Pb concentrations in environmental media for each case study are described in Chapter 4. Chapters 5 and 6 describe the human exposure assessment and health risk characterization, respectively, including descriptions of the methods, models, and data used. The methods and results of the ecological risk assessment are provided in Chapter 7. The references for this document are provided in Chapter 8.

2. Overview of Risk Assessment

The human exposure and health risk assessments and ecological risk assessment being conducted in support of U.S. EPA's review of the Pb NAAQS focus on the estimation of risk resulting from exposure to Pb released into ambient air. The preliminary design of these assessments was described in the draft Analysis Plan (USEPA 2006d). As described in the draft Analysis Plan, these assessments are being conducted in two phases: the pilot and the full-scale phases. The pilot phase, described in this report, employed a case study based approach to estimate human exposure and health risks and ecological risks associated with the following two air quality scenarios.

- **Current Conditions.** This scenario estimated risks associated with current conditions at each case study location.

- **Attainment of Current NAAQS.** This scenario, referred to as "attainment" in this report, estimated risks assuming attainment of the current NAAQS at the case study location that is currently not in attainment.

For both air quality scenarios, human and ecological exposures to Pb in air, as well as exposures to Pb in other media (e.g., soil, indoor dust) resulting from atmospheric deposition of Pb, were assessed. The sources associated with these exposures are referred to in this report as "policy-relevant" sources. This assessment also considers policy-relevant "background" Pb exposures, which include exposures that are not linked to Pb released to air, such as ingestion of drinking water, ingestion of Pb paint, and other non-air-related exposures.

For the ecological risk assessment, the case study approach was supplemented by a national-scale screening assessment of Pb concentration measurements in surface waters and sediments across the United States.

The remainder of this chapter provides an overview of the selected case studies (Section 2.1) and summaries of the approaches taken for the human exposure and health risk assessments (Section 2.2) and the ecological risk assessment (Section 2.3).

2.1 Case Studies

The selection of case study locations reflected consideration of factors presented in the draft Analysis Plan, as well as comments received during consultation with CASAC (Henderson 2006). The selection criteria included, but were not limited to, the following:

- Coverage for a variety of ambient air sources of Pb emissions ranging from large industrial point sources (e.g., primary and secondary Pb smelters) to areas with potentially significant reentrainment of historically deposited Pb;

- Availability of site-specific measurements for key environmental media (i.e., air, soil, and indoor dust) and biomonitoring of Pb exposure (i.e., blood Pb levels); and

- Consideration of demographics and socioeconomic factors related to exposure (e.g., areas with higher density of children in likely higher risk locations).

Five case studies were selected for the pilot phase. Some of the selected case studies were "shared" by the human and ecological assessments (e.g., with one characterization of environmental conditions serving the purposes of both assessments), while other case studies were specific to one of the assessments. A brief description of each selected location is provided below.

- **Primary Pb Smelter Location.** This case study involved a location associated with a primary Pb smelting facility. This facility is located in Herculaneum, Missouri, and has been in operation for more than 100 years. Recent measured air concentrations from the surrounding area and a robust data set of recent facility configuration information, emissions estimates, measured media concentrations (soil, indoor dust, surface water, and sediment), and exposure (i.e., blood Pb) measurements were available for this location. This case study was used for both the human exposure and health risk assessments and the ecological risk assessment.

- **Secondary Pb Smelter Location.** This case study focused on a location associated with a secondary Pb smelting facility in Troy, Alabama. Recent measured air concentrations from the area surrounding the facility and facility characterization data (including emission estimates) were readily available for this location. This case study was used for both the human exposure and health risk assessments and the ecological risk assessment.

- **Near Roadway Urban Location.** This location in Houston, Texas, was selected to provide a case study that reflected the influence of historically deposited Pb near roadways in urban areas. It covers approximately 2.5 kilometers (km) of roadway near an air monitoring site and includes residences within 200 meters (m) of this section of roadway. The location is in a relatively dry region of the country where resuspension is likely to occur, there are air measurements available from a monitor located close to a roadway, there are numerous homes along the roadways near this monitor, and there is a lack of identified point sources of Pb emissions in the surrounding area. This case study was used for the human exposure and health risk assessments.

- **Near Roadway Non-Urban Locations.** Two locations were selected to evaluate the ecological risks of historically deposited Pb in soils near roads in less developed areas, where exposures to ecological receptors are expected to occur. One location was in Corpus Christi, Texas, and the other was in Atlee, Virginia, north of Richmond, both near highly traveled interstates, but less developed than the near roadway urban location.

- **Ecologically Vulnerable Location.** The Hubbard Brook Experimental Forest, located in the White Mountain National Forest, near North Woodstock, New Hampshire, was selected as an ecological case study location. This location was selected because: (1) it is located in an acidified watershed and is therefore expected to have higher bioavailability of Pb; (2) there are no identified point sources of Pb in the surrounding

area, which allows for an evaluation of impacts of regional background Pb concentrations; (3) it is located in an elevated area, which is subject to comparatively higher deposition of Pb due to wind speed and precipitation; and (4) there are available data on trends (temporal, elevation, etc.) in Pb concentrations in various environmental media. This assessment only focused on identifying this site and did not involve characterization of potential impacts associated with measured Pb concentrations.

The specific approach for each case study was tailored to the location, based on the available data, the characteristics of the emission sources, and the potentially affected populations.

For the ecological risk assessment, these case studies were supplemented by a national-scale ecological screening risk assessment, consisting of two parts: the surface water portion and the sediment portion. This screening assessment was fundamentally different from the case study analyses because it analyzed data from locations across the United States, instead of focusing on one (or two) specific location(s). An overview of this screening assessment is provided below.

- **National-Scale Screening Risk Assessment, Consisting of the Surface Water and the Sediment Portions.** This analysis utilized Pb concentration measurements for freshwaters across the United States collected by the U.S. Geological Survey as part of their National Ambient Water Quality Assessment (NAQWA) program (USGS 2004). Information on land use, Pb air emission sources, and surface water point source discharges for a subset of these locations also played a role in this analysis.

2.2 Human Exposure and Health Risk Assessments

The pilot phase of the human exposure and health risk assessments was designed to estimate neurological effects in young children (in terms of IQ loss) associated with exposures to Pb emitted into ambient air for three case studies: primary Pb smelter, secondary Pb smelter, and near roadway urban location. This section provides an overview of the approach implemented for these assessments, beginning with a description of the conceptual model (Section 2.2.1) and a summary of the spatial scale of the analysis (Section 2.2.2), followed by brief descriptions of the exposure assessment (Section 2.2.3), risk assessment (Section 2.2.4), and uncertainty/variability analysis (Section 2.2.5).

2.2.1 Conceptual Model

This section presents the conceptual model (Exhibit 2-1) intended to illustrate the elements pertinent to assessment of public health risks associated with environmental Pb exposures. Elements included in the pilot phase are indicated with gray background and solid borders. Exposure pathways involving Pb released into ambient air are shown in bold text and pathways involving background sources are shown in non-bold (regular) text. A description of how the different elements of the conceptual model were addressed in the pilot phase, including any differences from what was presented in the draft Analysis Plan, is provided below:

- **Sources.** All sources in the conceptual model were addressed, either explicitly or implicitly, in the pilot phase. As mentioned previously, the pilot phase estimated effects

for two categories of sources: policy-relevant sources (which include stationary and mobile sources and outdoor soil which contains historically deposited Pb) and background (which includes diet,[1] drinking water, indoor Pb paint, and other miscellaneous sources). To the extent possible, estimates of blood Pb levels were differentiated by the contributing exposure pathways.

- **Pathways.** Exhibit 2-1 is intended to generally illustrate the many pathways by which Pb emitted into the environment becomes available for human exposure. For the purposes of this assessment, the pathways that pass through ambient air are of particular interest; all other pathways are considered background. For the pilot phase, the contribution of reentrained outdoor soil to ambient air concentrations was not estimated separately from the contribution from other emission sources.

- **Routes.** The ingestion and inhalation routes are considered the primary routes of human exposure to environmental Pb, with the ingestion route expected to be the more significant. Both routes were included in the pilot phase.

- **Exposed Populations.** For the purposes of the pilot phase, population groups were identified based primarily on age or life stage. Exposures were characterized in the pilot phase for young children (ages 0 to 7).

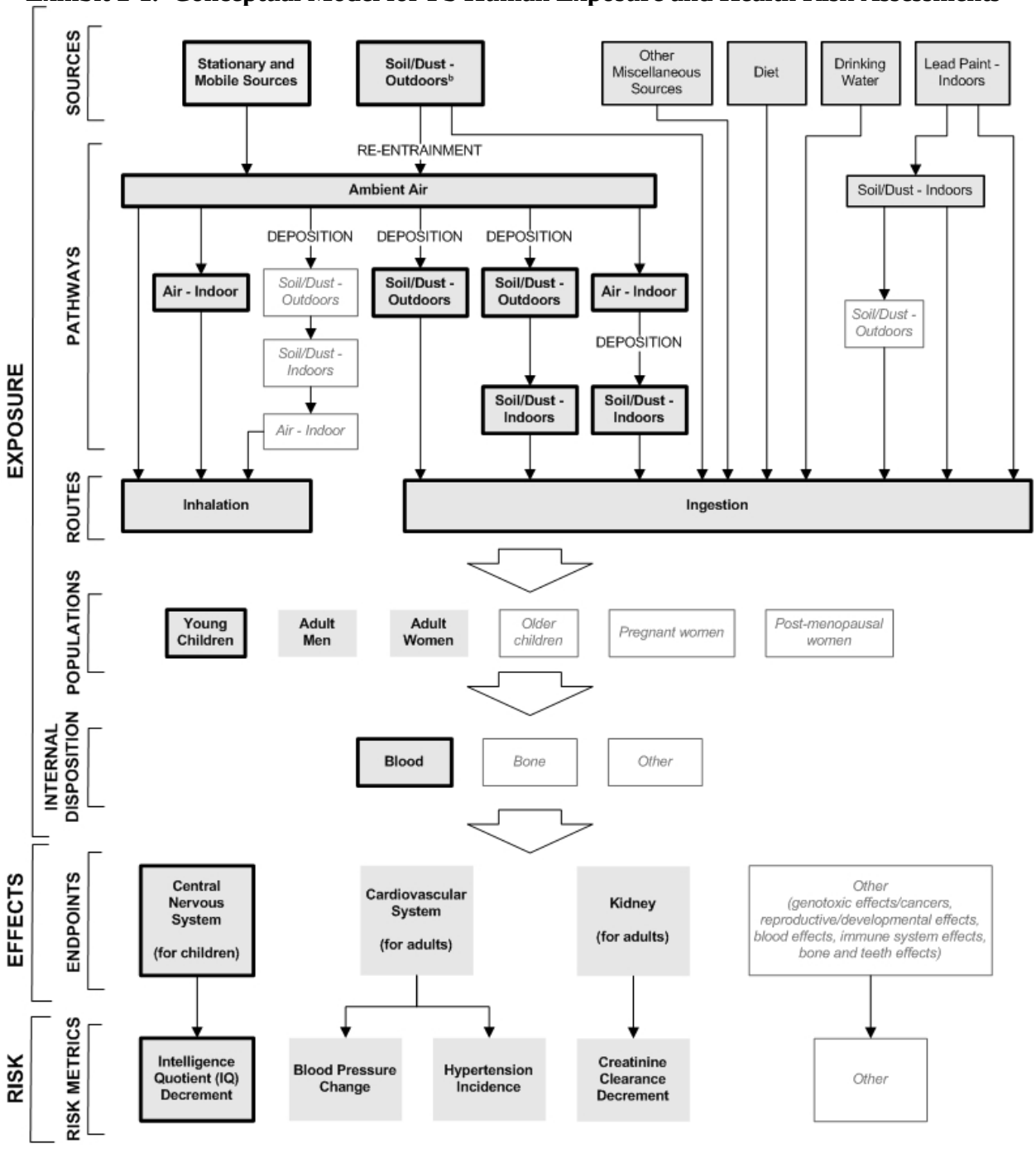
- **Internal Disposition.** While Pb is distributed throughout the body, bone is an established site of internal accumulation of Pb and blood is an established internal dose metric for purposes of both exposure and risk assessment. The pilot phase relied on blood Pb with corresponding dose-response functions to estimate risks. The biokinetic models used in this analysis recognize the role of bone as a reservoir with the potential to act as a source and storage site.

- **Endpoints.** As illustrated in Exhibit 2-1, numerous health endpoints are recognized in the Air Quality Criteria Document (CD) as associated with Pb exposures. The endpoints of interest for this assessment are those associated with the range of exposures expected to closely reflect current levels, and for which there is adequate information to develop quantitative risk assessments. Recognizing this, the pilot phase focused on neurological effects in children.

- **Metrics.** The pilot phase used IQ decrement in children as the primary risk metric.

[1] Note that exposure to Pb through the diet likely reflects a combination of: (a) air-sourced Pb impact (e.g., deposition of Pb onto crops which in turn is reflected in Pb content in processed foods); and (b) background Pb (e.g., Pb introduced into food during packaging). However, for the pilot phase, dietary Pb exposure was treated as purely background.

Exhibit 2-1. Conceptual Model for Pb Human Exposure and Health Risk Assessments[a]



[a] Components with gray background and solid borders will be included in this assessment. Components depicted in gray but without borders were not included in the pilot phase, but are being considered for inclusion in the full-scale risk assessment. Further, a distinction is made between components linked to exposure pathways involving ambient air (shown in bold) and components involving other pathways (i.e., background).

[b] Includes contributions of historical sources, including (but not limited to) emissions from the use of leaded gasoline, historical emissions from stationary sources, and exterior leaded paints.

An overview of the approach developed to implement the selected elements of the conceptual model for the pilot phase is provided in Exhibit 2-2. This exhibit shows the key types of information and models involved in each part of the pilot phase and how they are related to each other and to the other phases of the analysis. Exhibit 2-3 summarizes the use of these information types and models for each case study. As indicated in these exhibits, the specific approach for each case study differed based on the nature of the case study (e.g., type of source, locations of populations) and the site-specific measurements available. In cases where the available measurements were not sufficient to characterize the study area (e.g., due to insufficient spatial coverage), these data were used for performance evaluation. The following sections describe each part of the pilot phase in more detail.

Exhibit 2-2. Overview of Human Exposure and Health Risk Assessments' Approach for the Pilot Phase

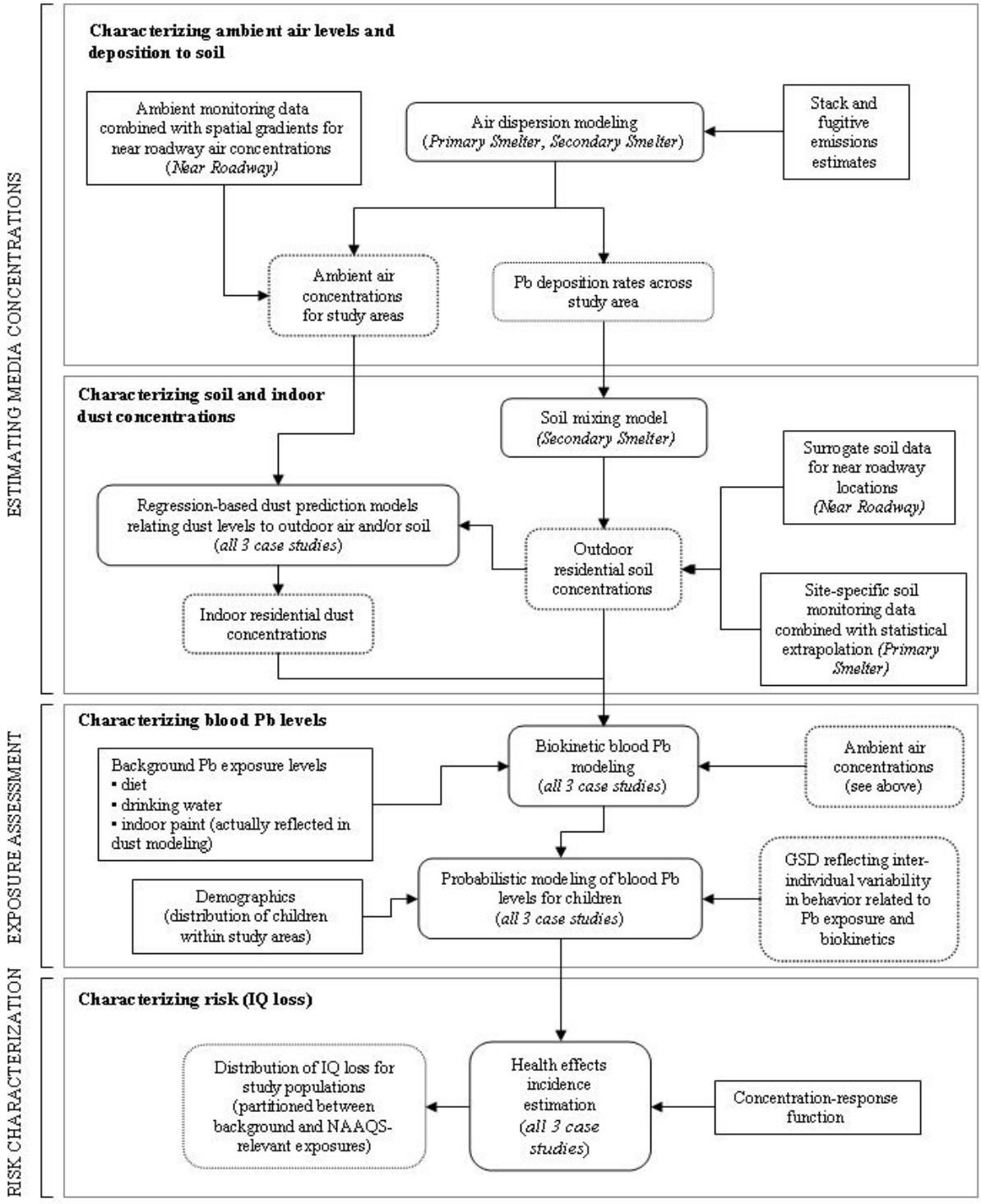


Exhibit 2-3. Models and Measurements Used for Human Exposure and Health Risk Assessments in the Pilot Phase

		Primary Pb Smelter	Secondary Pb Smelter	Near Roadway Urban
Location		Herculaneum, Missouri	Troy, Alabama	Houston, Texas
Spatial resolution		U.S. Census blocks, block groups	U.S. Census blocks	Distance ranges from roadways (0 to 12 m, 12 to 75 m, 75 to 200 m)
Exposure Assessment: Media Concentrations				
Ambient air concentrations	Models	ISC-Prime	AERMOD	n/a
	Measurements	Site-specific air monitoring (13 locations)	Site-specific air monitoring (2 locations)	Site-specific, near roadway air monitoring, combined with estimated spatial gradients from literature (1 location)
Deposition to soil	Models	ISC-Prime	AERMOD	n/a
	Measurements	Site-specific measurements (10 locations)	n/a	n/a
Soil concentrations	Models	n/a	Multiple Pathways of Exposure (MPE)	n/a
	Measurements	Site-specific soil sampling combined with statistical extrapolation (909 locations)	n/a	Surrogate soil sampling data from Corpus Christi, Texas (3 locations)
Indoor dust concentrations	Models	Combination of site-specific regression-based model and pooled analysis model (the AGG model), both using ambient air and soil concentrations as inputs	Pooled analysis model (the AGG model) using ambient air concentrations as inputs	Pooled analysis model (the AGG model) using ambient air and soil concentrations as inputs
	Measurements	Site-specific indoor dust sampling (17 locations)	n/a	n/a
Exposure Assessment: Internal Disposition				
Blood Pb levels	Models	IEUBK, Leggett		
Risk Characterization				
Risk (IQ loss)	Models	Log-linear distribution concentration-response function (Lanphear et al. 2005)		

2.2.2 Spatial Scale

To capture the spatial variability in media and exposure concentrations, a GIS-based "spatial template" was created for each case study. This template defines the outer boundary and the spatial units within this boundary for each case study. Due to the inherent differences between the case studies, such as the environmental setting, locations of populations, source types, and availability of measurements, the spatial template for each case study was different. Detailed descriptions of the templates are provided in Chapter 4. The spatial template for each case study location is summarized below:

- **Primary Pb Smelter Location.** The outer boundary of this study area was set to 10 km, which was expected to capture the population experiencing the most significant impacts of the facility's emissions. To provide sufficient spatial resolution to capture concentration gradients, while minimizing the number of computations required for estimating soil concentrations, blood Pb levels, and risks, the study area was divided into a combination of U.S. Census blocks and block groups. This was done by first modeling air concentrations at the centroid of each U.S. Census block within the study area. Based on these estimated concentrations, locations with higher gradients in air concentrations[2] used blocks as the spatial unit. All other locations used block groups. Air concentrations for block groups were estimated by calculating the area-weighted average of the block values. Soil concentrations were estimated for the same combination of blocks and block groups as the air concentrations.

- **Secondary Pb Smelter Location.** The outer boundary of this study area was also set to 10 km, which was expected to capture the population experiencing the most significant impacts of the facility's emissions. Air concentrations and deposition were modeled for the centroid of each U.S. Census block within the study area. These estimated concentrations indicated that relatively high gradients exist across the blocks in nearly all of the block groups in the study area, and thus all locations used blocks as the spatial unit. Soil concentrations were estimated for the same spatial units (i.e., blocks) as the air concentrations.

- **Near Roadway Urban Location.** The boundaries of this study area were set based on the presence of an approximately 2.5 km stretch of roadway adjacent to the selected air monitoring site. This section of roadway was selected based on data availability for roadway traffic volume and the assumption that the measured air concentrations at the monitoring site were a reasonable representation of concentrations along that length of roadway. The "depth" of the study area (i.e., how far the study area extends from the roadway) was set based on published data (ICF 2005) for other locations suggesting that emissions on roadways can contribute to ambient concentrations adjacent to the roadway out to about 200 m. This study area was then subdivided into three zones (i.e., 0 to 12 m, 12 to 75 m, and 75 to 200 m) on either side of the roadway, based on locations of soil

[2] Gradients in air concentrations were estimated by calculating the ratio of maximum U.S. Census block concentration to average block concentration for each U.S. Census block group in the study area. U.S. Census blocks were defined as the spatial units for all block groups with ratios greater than 2.0.

measurements and factors relating ambient concentrations in these regions to concentrations directly over the roadway.

2.2.3 Exposure Assessment

In the exposure assessment, blood Pb levels were estimated by first estimating Pb concentrations in environmental media (i.e., air, soil, indoor dust) and then modeling blood Pb levels for the populations exposed to these media. A summary of the timeframe associated with estimated media concentrations and population data used in the pilot phase is provided in Exhibit 2-4.

Exhibit 2-4. Timeframe Reflected in the Pilot Phase Exposure Assessment

Data	Primary Pb Smelter	Secondary Pb Smelter	Near Roadway Non-urban
Ambient outdoor air Pb concentrations	2001 to 2005 (post-2001 SIP emissions data)	1997 to 2000 (stack test emissions data)	2001 (air monitoring data)
Soil Pb concentrations	2000 to 2005 (timeframe for soil sampling at site)	1997 to 2000 (modeled based on air deposition)	1998 (soil sampling)
Indoor dust concentrations	2000 to 2005 (estimated based on ambient air and soil concentrations)	1997 to 2000 (estimated based on ambient air concentrations)	2001 (estimated based on ambient air and soil concentrations)
Population data	2000 U.S. Census		
Cumulative period reflected in data	**2000 to 2005**	**1997 to 2000**	**~2001**

The remainder of this section describes how media concentrations were estimated for each case study and how blood Pb modeling was used in the assessment.

2.2.3.1 Estimation of Media Concentrations

This section summarizes the approach for estimating media concentrations for each case study in the pilot phase. In order to model the exposure pathways identified in Exhibit 2-1, Pb concentrations in the following media are required:

- Air,
- Outdoor soil, and
- Indoor dust.

For the pilot phase, concentrations in these media were estimated for both the current conditions and attainment air quality scenarios. The approach for estimating media concentrations for these scenarios depended on the case study.

Primary Pb Smelter: For the primary Pb smelter case study, a combination of measurements and modeling was used to characterize media concentrations. Substantial analysis has been performed for this location by U.S. EPA Region 7 and the facility. As a result, a robust data set of ambient air monitoring, outdoor soil measurements, surface water and sediment measurements, and indoor dust measurements, as well as air modeling input values, were available. Ambient air concentrations for the current conditions scenario were characterized using the Industrial Source Complex – Plume Risk Model Enhancements (ISC-Prime) model (Schulman et al. 1997). This was the only case study with estimated ambient air concentrations that exceeded the current NAAQS standard, thus requiring an additional assessment of ambient air concentrations for the attainment scenario. Ambient air concentrations for the attainment scenario were estimated by first calculating quarterly average concentrations for the current conditions scenario, replacing any estimated concentrations that exceeded the NAAQS with the current standard (1.5 $\mu g/m^3$), and then calculating annual average concentrations from these quarterly averages.

Inhalation exposure concentrations were estimated from these ambient air concentrations to account for variability in daily mobility for the population of interest (young children). In this context, mobility refers to the time spent in different locations, such as outside at home, inside at school, and in vehicles. These concentrations were estimated using ratios for Pb of estimated exposure concentrations to ambient air concentrations (derived from the exposure and ambient air concentrations estimated as part of the 1999 National-Scale Air Toxics Assessment [USEPA 2006e]). These ratios, which are age- and census-tract-specific, were applied to estimated ambient concentrations for the current condition and attainment scenarios to calculate inhalation exposure concentrations specific to age and location of the study population.

Part of this study area has undergone substantial soil monitoring and remediation activities, resulting in very different spatial coverage of available soil measurements within the remediation zone compared to outside of it. For locations within the soil remediation zone, which extends approximately 1.5 km from the facility, soil concentrations were estimated using available measurements for the soil currently in those locations. Concentrations of Pb in soil outside of the soil remediation zone were estimated using a statistical regression equation based on available pre-remediation measurements. The same soil concentrations were used for the current conditions and attainment scenarios.

As with soil concentrations, indoor dust concentrations of Pb were estimated using different approaches for houses within the soil remediation zone than for those outside of the zone. For residences within the soil remediation zone, an empirical model, developed from air monitoring data and post-remediation indoor dust measurements available within the zone, was used. For locations outside the remediation zone, indoor dust concentrations were estimated using a pooled analysis model, based on data from a variety of industrial and urban/industrial studies (USEPA 1989, Appendix B). Both models estimated indoor dust concentrations for both the current conditions and attainment scenarios based on ambient air and outdoor soil concentrations, estimated as described above.

Secondary Pb Smelter: For this case study, ambient air concentrations were estimated using the American Meteorology Society/Environmental Protection Agency Regulatory Model (AERMOD) (USEPA 2004a). Based on these ambient air concentrations, inhalation exposure concentrations were estimated using the same approach as for the primary Pb smelter case study (described above). Outdoor soil concentrations were estimated using the soil modeling approach from EPA's Multiple Pathways of Exposure (MPE) methodology (USEPA 1998) and deposition outputs from AERMOD. Indoor dust concentrations were estimated using a pooled analysis model, based on data from a variety of industrial and urban/industrial studies (USEPA 1989, Appendix B), which relates ambient air concentrations to indoor dust levels (outdoor soil was not considered explicitly as an independent variable).

Near Roadway Urban Location: For this case study, air monitoring data collected adjacent to a roadway in Houston, Texas, were used to define the ambient air concentrations for the site. These air monitoring data were combined with data from the literature characterizing the spatial gradient in particulate matter concentrations near roadways to estimate air concentrations in two distance bands from the roadway: 0 to 75 m and 75 to 200 m (ICF 2005). Based on these ambient air concentrations, inhalation exposure concentrations were calculated using the same approach used for the primary Pb smelter case study (described above). No adequate site-specific outdoor soil measurements were available for this site or elsewhere in Houston, and thus Pb concentrations in outdoor soil were based on measurements obtained adjacent to a similar roadway in Corpus Christi, Texas, taking into consideration other soil Pb measurements reported in the literature and values obtained from EPA's SPECIATE data base. Based on these data and the distance bands used to characterize air concentrations, soil concentrations were estimated for three distance bands from the roadway: 0 to 12 m, 12 to 75 m, and 75 to 200 m. Indoor dust concentrations were estimated using the pooled analysis model used for the outer zone of the primary Pb smelter case study that relates ambient air and outdoor soil concentrations to indoor dust levels (USEPA 1989, Appendix B).

2.2.3.2 Blood Pb Modeling

Blood Pb levels were predicted for a child population at each case study location using two different biokinetic models: the Integrated Exposure Uptake Biokinetic Model for Children (IEUBK) (USEPA 2005a) and the International Commission for Radiation Protection model (Leggett 1993).[3] Inputs to these models included the estimated exposure media (air, soil, and indoor dust) concentrations of Pb for each case study, as well as values for other parameters that influence blood Pb levels.

The same fundamental approach was used to estimate population distributions of blood Pb levels at each of the three case study locations. This approach involved two steps. The first step involved applying the IEUBK and Leggett models to predict central tendency blood Pb levels for children within the study area for each case study location. The estimated blood Pb levels were then parsed into the fraction associated with policy-relevant background and the fraction associated with policy-relevant sources, with the latter category being further subdivided into

[3] Although emphasis was placed on biokinetic models in predicting blood Pb levels for children in the pilot phase analysis, a statistical (regression)-based blood Pb model (Lanphear et al. 1998) was included as part of the sensitivity analysis (see Section 2.2.5).

contributions from inhalation, soil ingestion and indoor dust ingestion exposure pathways. For the purposes of this pilot phase assessment, drinking water and diet exposure pathways are categorized as policy-relevant background, although it is recognized that a portion of Pb in both pathways may derive from policy-relevant sources.

The second step involved implementing a probabilistic model to generate population distributions of blood Pb levels for children in each case study location based on consideration of three key factors:

1. The central tendency blood Pb levels generated by the blood Pb model for each U.S. Census block and block group in the study area (from the first step described above);
2. Demographic data (i.e., distribution of children 0 to 7 years of age across the blocks and block groups comprising a given study area); and
3. A geometric standard deviation characterizing inter-individual variability in blood Pb level (e.g., reflecting differences in behavior and biokinetics related to Pb).

This approach produced a distribution of modeled children's blood Pb levels for each case study location. Each of the individual modeled blood Pb levels was further differentiated to show the relative contribution of policy-relevant background and sources to total blood Pb.

2.2.4 Risk Assessment

As discussed above, IQ impacts in exposed children have been selected as the endpoint for risk quantification in the pilot phase assessment, on the basis of a large body of evidence establishing the occurrence of the IQ effects at low exposure and blood Pb levels. The characterization of the IQ loss impacts for the pilot phase assessment involved generating a distribution of IQ loss estimates for the set of children simulated in the exposure assessment using the estimated blood Pb levels for these children and a blood Pb concentration-response relationship for IQ loss. The selected concentration-response relationships are the log-normal concentration-response functions for concurrent[4] and lifetime average blood Pb concentrations from a pooled analysis of epidemiology studies focusing on IQ loss in children (Lanphear et al. 2005).

For the pilot phase, cutpoints of 2.4 μg/dL (concurrent exposure metric) and 6.1 μg/dL (lifetime average exposure metric) were selected as blood Pb levels below which IQ loss was not to be predicted. The selection of these cutpoints is based on consideration for the blood Pb level at which confidence in being able to characterize the shape of the concentration-response functions diminishes significantly. For example, for the concurrent blood Pb concentration response function, less than five percent of the children from the pooled analysis had concurrent blood Pb levels below 2.4 μg /dL (Lanphear et al. 2005), which suggests an increased uncertainty in the functional form below this level of exposure.

[4] In the epidemiological studies, "concurrent" refers to a blood Pb measurement taken concurrently with the IQ assessment. The children included in the pooled analysis on which the concentration-response function was based ranged from five to 10 years old (refer to Section 6.1.2 for more details). In these studies, "lifetime average" refers to the average of blood Pb measurements taken over the child's life from six months until the IQ assessment.

As with the exposure analysis, the contributions of policy-relevant sources and policy-relevant background exposure pathways to overall IQ loss are differentiated. These results are presented using two primary risk metrics, as described below:

- **Population risk percentiles.** This metric characterizes the degree of IQ loss associated with policy-relevant sources for specific percentiles of the child population (e.g., the 50^{th}, 90^{th}, 95^{th}, 99^{th} percentile modeled child). This category of metric provides perspective on the distribution of IQ loss resulting from policy-relevant sources in each study area, ranging from the typical or average child (50^{th} percentile or mean) to children experiencing higher exposures (e.g., 90^{th} and 99^{th} percentiles).

- **Child frequency counts associated with specific risk percentiles.** This metric characterizes the number of children, for a given study area, associated with each of the population percentiles (e.g., 25 children in a particular study area population are predicted to have risk levels at or above the 99^{th} percentile). It provides a perspective on the number of children associated with various degrees of IQ loss for a particular case study.

2.2.5 Uncertainty/Variability Analysis

For the pilot phase assessment, sensitivity analysis techniques were used to examine the issue of uncertainty and its impact on exposure and risk estimates. This sensitivity analysis consisted of a series of "one-at-a-time" analyses conducted to evaluate the impacts of modeling methods and parameter inputs on estimated blood Pb and IQ loss distributions. The general approach employed was to evaluate changes in risk metrics associated with changing single models or input parameter values away from those used in a "baseline" analysis.

The baseline risk analysis focused on the current conditions scenario for the primary Pb smelter case study. Note that while the baseline models and parameter values are believed to provide a credible estimate of blood Pb and IQ changes, it is not suggested that the baseline model is uniquely the "best," or that it provides demonstrably "central tendency" risk estimates.

The sensitivity analysis completed for the pilot phase focused on those modeling elements (including input datasets and modeling steps) believed to have the greatest potential for significantly impacting exposure and risk modeling, including the following:

- **Indoor dust prediction approach**. Given its notable contribution to blood Pb levels, the sensitivity of the blood Pb and IQ change estimates to the selected approach used to estimate indoor dust concentrations was analyzed. This analysis evaluated the effect of using different regression models on distributions of estimated indoor dust Pb concentrations.

- **Blood Pb model selection.** Blood Pb metrics developed using IEUBK were compared to those based on the Leggett biokinetic model, as well as an "empirical" blood Pb model developed by Lanphear et al. (1998).

- **Exposure, intake, and update value selection.** Two variations in gastrointestinal absorption and soil and dust ingestion parameters were evaluated. In one set of sensitivity runs, the gastrointestinal absorption fraction for dietary and drinking water ingestion was varied above and below the central tendency value used in the baseline assessment. For the second set of sensitivity runs, an alternative set of inputs for soil and dust absorption and relative ingestion amounts of soil versus dust were evaluated for their impacts on estimated blood Pb and IQ loss distributions. These values for the second set of runs were derived by Von Lindern et al. (2003) from a statistical analysis of soil, dust, and blood Pb distributions at a large mining/smelting "Superfund" site.

- **Blood Pb GSD selection.** The probabilistic exposure model was run with two alternative values of the blood Pb GSD and the results were compared to the results using the baseline blood Pb GSD.

- **Concentration-response function for IQ loss.** Three alternative concentration-response functions for IQ loss were evaluated. The first two consisted of the upper and lower 95 percent confidence limits on the log-linear regression model, which were derived based on a reanalysis of the Lanphear et al. model results (USEPA 2006a). The third alternative consisted of the piecewise linear version of the Lanphear et al. (2005) model.

- **Blood Pb metrics.** This analysis evaluated the magnitude of impact on estimated IQ losses associated with using the following alternative blood Pb metrics: lifetime average and peak (defined as the highest annual average). These results were compared to the results estimated using concurrent blood Pb level distributions.

- **Blood Pb cutpoint for IQ loss**. The sensitivity of estimated IQ change to 50 percent reductions in the blood Pb cutpoints for the concurrent and lifetime average blood Pb levels was evaluated.

Ultimately, this sensitivity analysis serves two primary purposes:

1. Determining which of the modeling elements included in the sensitivity analysis has the greatest impact on risk modeling, which may help guide future efforts to refine the overall assessment; and
2. Obtaining a semi-qualitative feel for the potential magnitude of overall uncertainty in the risk results.

This type of sensitivity analysis does not, however, allow for a rigorous assessment of probabilistically-defined confidence in specific risk percentiles to be quantified (that only being possible with a formal uncertainty analysis).

2.3 Ecological Risk Assessment

The ecological risk assessment involved four case studies and a national-scale screening assessment. These analyses were designed to estimate the potential for ecological risks associated with exposures to Pb emitted into ambient air for three case studies: primary Pb smelter, secondary Pb smelter, and near roadway non-urban location. Activities for the fourth case study (ecologically vulnerable location) focused on identification and description and did not include risk analyses.

This section provides an overview of the approaches implemented for these analyses, beginning with a description of the conceptual model (Section 2.3.1) and a summary of the spatial scale of the analysis (Section 2.3.2), followed by brief descriptions of the exposure assessment (Section 2.3.3), effects assessment (Section 2.3.4), risk characterization (Section 2.3.5), and uncertainty and variability assessment (Section 2.3.6).

2.3.1 Conceptual Model

This section presents the conceptual model (Exhibit 2-5) intended to illustrate the elements pertinent to assessment of ecological risks associated with environmental Pb exposures. Elements addressed either explicitly or implicitly in the pilot phase are indicated with black text. Components with gray text were not addressed in the pilot phase due to uncertainty regarding available data and modeling tools. A description of how the different elements of the conceptual model were addressed in the pilot phase, including any differences from what was presented in the draft Analysis Plan, is provided below:

Sources. Sources associated with Pb released into ambient air were addressed, either explicitly or implicitly, in this assessment. These sources are indicated in black text in Exhibit 2-5. When measurements were used, the analysis attempted to focus on those pertaining to Pb initially released into ambient air, isolating contributions from policy-relevant background (i.e., non-air) sources to the extent possible.

Pathways. This assessment focused on policy-relevant exposure pathways involving Pb released to ambient air and subsequently transported to soil, surface water, and sediment. Direct inhalation exposures for birds and mammals were not addressed because ecological exposures are expected to be primarily through ingestion of Pb in and on their foods and through incidental ingestion of Pb in soils. Because Pb accumulates in soils, the ingestion exposures are expected to be much higher than any inhalation exposures. In addition, the human exposure and health risk assessments are expected to be protective of inhalation exposures for wildlife because the critical effect under evaluation for humans (i.e., IQ loss) is more sensitive to Pb exposure than the effects of Pb on growth, reproduction, or survival that are relevant to wildlife population risks.

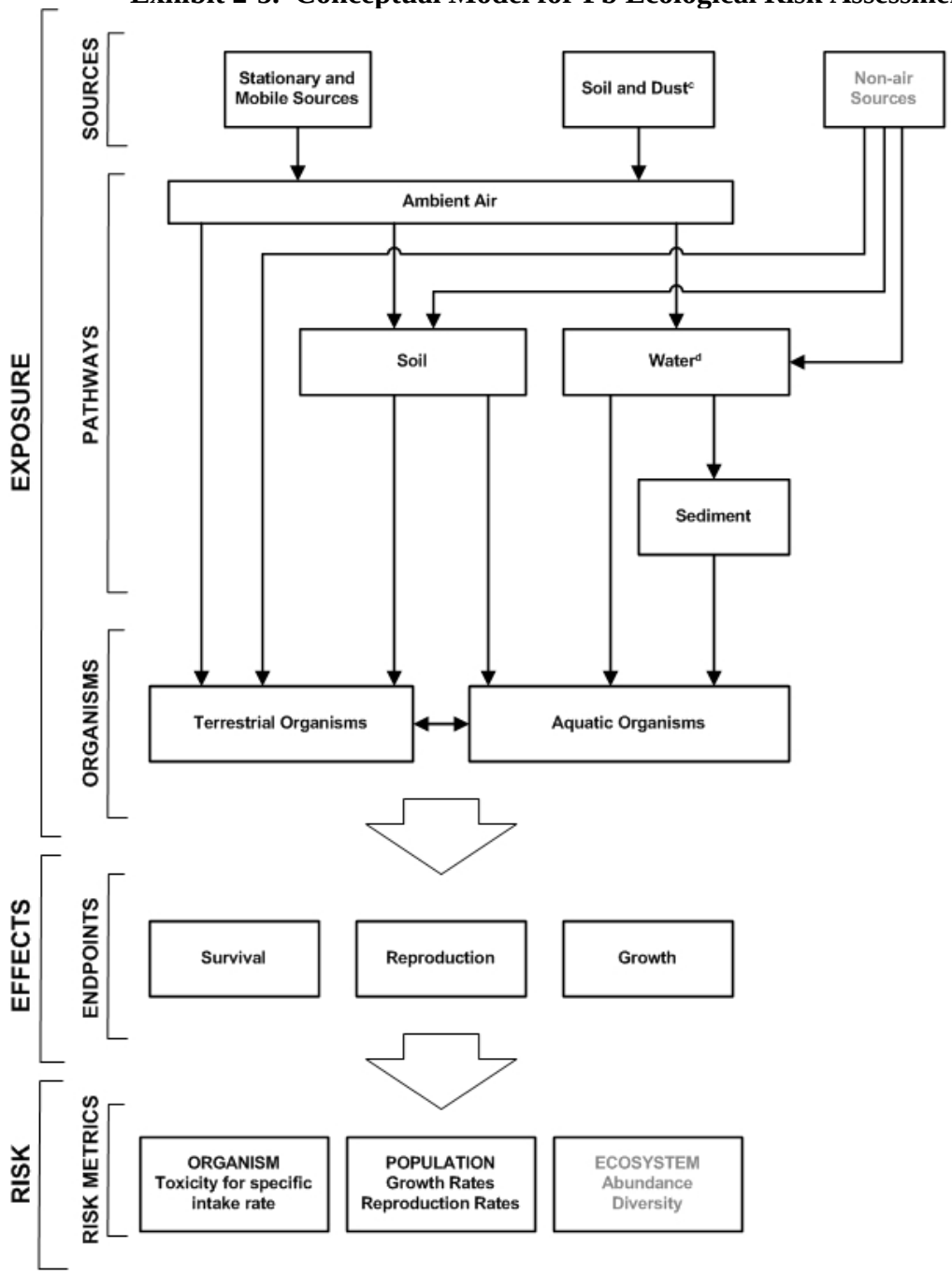
Organisms. Potential ecological impacts for both terrestrial receptors (e.g., populations of birds, mammals, soil invertebrates, plants) and freshwater aquatic organisms (e.g., benthic communities, water-column communities) were considered in the pilot phase.

Assessment Endpoints. The assessment endpoints for assessing ecological risks associated with Pb deposition from air are adverse effects on the survival, growth, and reproduction of exposed ecological receptors.

Risk Metrics. Assessment of ecological risk for this assessment involves assessing the potential for Pb exposures to be toxic to ecological receptors at the organism level and to adversely affect growth and reproductive rates at the population level. These effects will be evaluated in each case study (and in the national-scale screening assessment) using the hazard quotient (HQ) approach, in which HQs are calculated by dividing environmental medium-specific exposure estimates by the appropriate ecotoxicity screening values.

An overview of the approach developed to implement the selected elements of the conceptual model for the ecological risk assessment is provided in Exhibit 2-6. This exhibit shows the key types of information and models involved in each part of the assessment and how they are related to each other and to the other parts of the analysis. Exhibit 2-7 summarizes the use of these information types and models for each case study. As indicated in these exhibits, the specific approach for each case study differed based on the nature of the case study (e.g., type of source, locations of populations) and the site-specific measurements available. In cases where the available measurements were not sufficient to characterize the study area (e.g., due to insufficient spatial coverage), these data were used for performance evaluation. The following sections describe each of the parts of the assessment in more detail.

Exhibit 2-5. Conceptual Model for Pb Ecological Risk Assessment[a, b]



[a] Many of the processes and pathways above are circular in nature. For the clarity of the schematic, they are shown as bidirectional.
[b] Components with gray text were not addressed in the quantitative assessment due to uncertainty regarding available data and modeling tools.
[c] Includes contributions of historical sources, including (but not limited to) emissions from the use of leaded gasoline, historical emissions from stationary sources, and exterior leaded paints.
[d] Water in this schematic represents all surface water bodies; however, only freshwater was addressed in this assessment.

Exhibit 2-6. Overview of Ecological Risk Assessment Approach

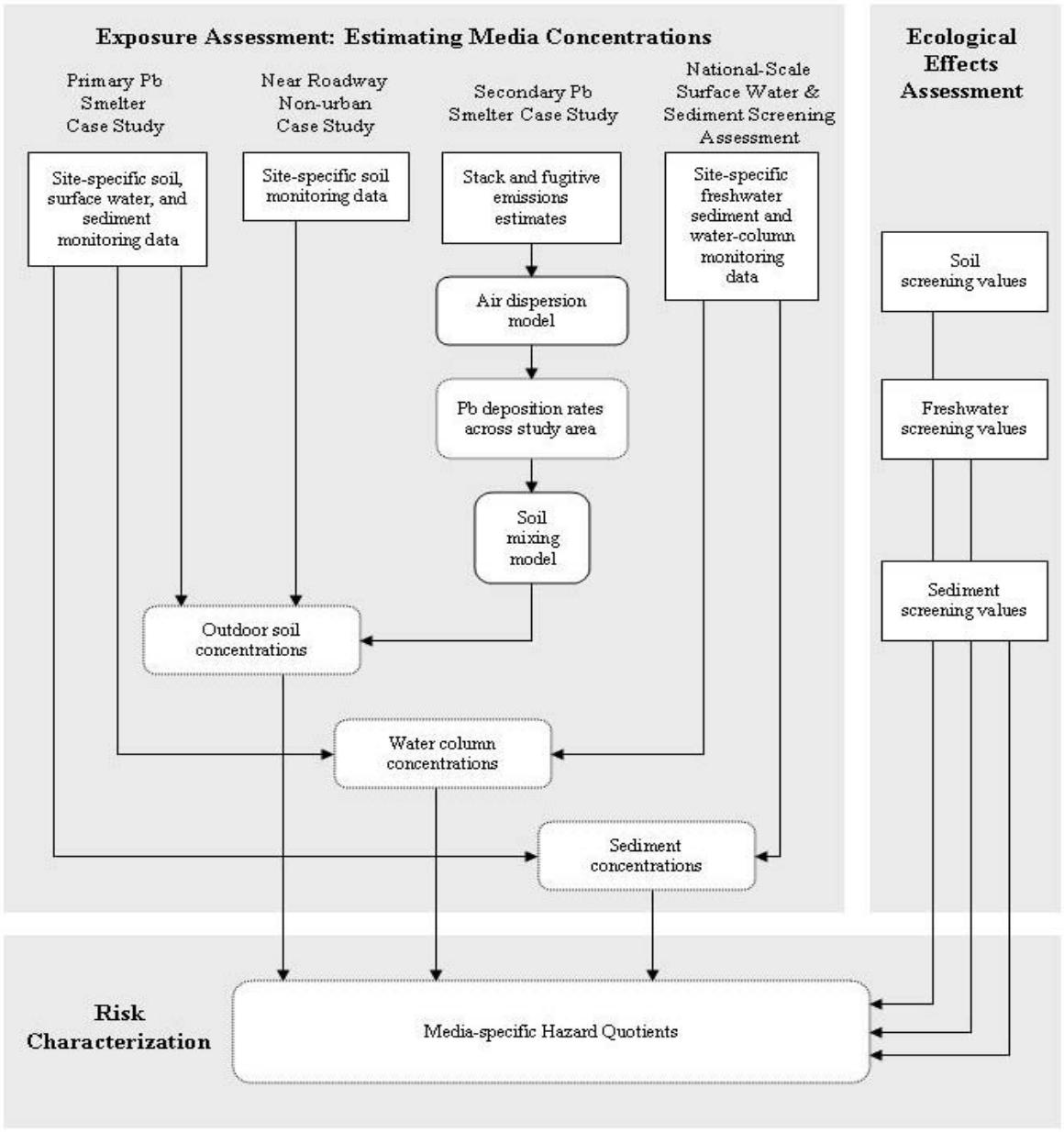


Exhibit 2-7. Models and Measurements Used for the Ecological Risk Assessment

		Primary Pb Smelter Case Study	Secondary Pb Smelter Case Study	Near Roadway Non-Urban Case Study	Ecologically Vulnerable Case Study	National-Scale Screening Assessment: Surface Water and Sediment Portions
Location		Herculaneum, Missouri	Troy, Alabama	Corpus Christi, Texas Atlee, Virginia	Hubbard Brook Experimental Forest, New Hampshire	Surface water bodies in the Unites States
Spatial extent and resolution		Approximately 6 km diameter, centered on point source	U.S. Census blocks	• Corpus Christi: single transect perpendicular to road; 0.5 to 4 m from road • Atlee: 140 m section of road; 2 to 30 m from road	oblong basin about 8 km long by 5 km wide	47 basin study units from all regions of the United States, covering approx. 50 percent of U.S. land base
Exposure Assessment: Estimating Media Concentrations						
Deposition to soil	Models	n/a	AERMOD[a]	n/a	n/a	n/a
	Measurements	n/a	n/a	n/a	12 samples	n/a
Soil conc.	Models	n/a	MPE[b]	n/a	n/a	n/a
	Measurements	Site-specific conc. of total Pb in soil samples (26 locations)	n/a	Site-specific conc. of total Pb in soil samples (Corpus Christi: 2 locations; Atlee: 26 locations)	Site-specific conc. of total Pb in soil samples (124 locations)	n/a
Surface water conc.	Models	n/a	n/a	n/a	n/a	n/a
	Measurements	Site-specific conc. of dissolved Pb in water column samples from eight water bodies/drainage areas (30 locations)	n/a	n/a	Site-specific conc. of dissolved Pb in water column samples (locations not reported)	Site-specific conc. of dissolved Pb in surface water samples (430 samples)
Sediment conc.	Models	n/a	n/a	n/a	n/a	n/a
	Measurements	Site-specific conc. of total Pb in sediment samples from five water bodies/drainage areas (69 locations)	n/a	n/a	n/a	Site-specific or nearby water body conc. of total Pb in sediment samples (15 locations)

Exhibit 2-7. Models and Measurements Used for the Ecological Risk Assessment

		Primary Pb Smelter Case Study	Secondary Pb Smelter Case Study	Near Roadway Non-Urban Case Study	Ecologically Vulnerable Case Study	National-Scale Screening Assessment: Surface Water and Sediment Portions
Ecological Risk Assessment						
Ecotoxicity Screening Levels	Soil	Soil screening values developed based on U.S. EPA Superfund Ecological Soil Screening Level (Eco-SSL) methodology [c]			n/a	n/a
	Freshwater – water column	U.S. EPA Pb freshwater AWQC for aquatic life derived based on site-specific measured water hardness (conc. of $CaCO_3$)[c]	n/a	n/a	n/a	U.S. EPA Pb freshwater AWQC for aquatic life derived based on site-specific or nearby water body measured water hardness (conc. of $CaCO_3$)[c]
	Freshwater – sediment	Sediment screening values based on MacDonald et al. (2000) sediment quality assessment guidelines [c]	n/a	n/a	n/a	Sediment screening values based on MacDonald et al. (2000) sediment quality assessment guidelines [c]

[a] American Meteorological Society/Environmental Protection Agency Regulatory Model (AERMOD) (USEPA 2004a)
[b] Multiple Pathways of Exposure (MPE) (USEPA 1998)
[c] These values are based on measured ecotoxicity data and other types of measured data (e.g., soil screening levels evaluate measured food and soil ingestion rates for birds and mammals).

2.3.2 Spatial Scale

To capture the spatial variability in media and exposure concentrations, a GIS-based "spatial template" was created for each case study. This template defines the outer boundary and the spatial units within this boundary for each case study. Due to the inherent differences between the case studies, such as the environmental setting, locations of populations, source types, and availability of measurements, the spatial template for each case study was different. Detailed descriptions of the templates are provided in Chapter 4. The spatial template for each case study location is summarized below:

- **Primary Pb Smelter Location.** Overall, the spatial template for this case study is the same as that used for the human exposure and health risk assessments of this location. For the ecological risk assessment, however, the spatial template also included a surface water body within 2.5 km of the facility.

- **Secondary Pb Smelter Location.** The spatial template for this case study is the same as that used for the human exposure and health risk assessments of this location.

- **Near Roadway Non-Urban Locations.** The characterization of the two locations selected for this case study relied upon available soil measurements. For the Corpus Christi, Texas location, the template includes a single transect of soil perpendicular to the road extending approximately 4 m from the edge of the road. For the Atlee, Virginia location, the template includes a band of soil approximately 30 m wide located adjacent to a 140 m section of I-95.

- **Ecologically Vulnerable Location.** This location was selected and described as part of the assessment, but not subjected to any further analysis. Thus, no spatial template was developed.

- **National-Scale Ecological Screening Assessment: Surface Water and Sediment Portions.** The spatial scale of this analysis included parts of watershed basins from all regions of the United States, accounting for drainage from approximately 50 percent of the United States land base.

2.3.3 Exposure Assessment

The measures of exposure for the pilot phase are total Pb concentrations in soil, dissolved Pb concentrations in fresh surface waters (water column), and total Pb concentrations in freshwater sediments. Exposure concentrations were estimated for the three case studies and the national-scale screening assessment as described below.

- For the primary Pb smelter case study, measured concentrations of total Pb in soil, dissolved Pb in surface waters, and total Pb in sediment were used to develop point estimates for locations with Pb thought to be associated with atmospheric Pb deposition, rather than with non-air sources, such as runoff from waste storage piles.

- For the secondary Pb smelter case study, soil concentrations of Pb were estimated using fate and transport modeling based on EPA's MPE methodology (USEPA 1998). The pilot phase of the assessment was limited to soil pathways for this case study.

- For the near roadway non-urban case study, soil concentration measurements collected for two locations adjacent to interstate highways, one in an area of fairly high-density development (Corpus Christi, Texas) and another in an area of medium-density development (Atlee, Virginia), were used to develop point estimates of Pb associated with historical deposition. The pilot phase of the assessment was limited to soil pathways for this case study.

- In the national-scale surface water and sediment screening assessment, measurements of dissolved Pb concentrations in surface water and total Pb concentrations in sediments for locations across the United States were used. Air emissions, water discharge, and land use data for the areas surrounding these locations were assessed to identify locations where atmospheric Pb deposition may be expected to contribute to potential ecological impacts. The exposure assessment focused on these locations.

2.3.4 Effects Assessment

The tools used in the ecological risk assessment for soils, fresh surface water bodies, and freshwater sediments included the following screening-level ecotoxicity values.

- For terrestrial organisms, soil screening values were developed for the NAAQS review based on the methodology for Ecological Soil Screening Levels (Eco-SSLs) developed by EPA's Office of Solid Waste and Emergency Response (USEPA 2005b; USEPA 2005c)

- For aquatic organisms in the water column, EPA-recommended ambient water quality criteria (AWQC), expressed as concentrations of Pb in the water column, were used.

- For benthic organisms, the sediment screening values used were sediment quality assessment guidelines developed by MacDonald et al. (2000; reported in MacDonald et al. 2003).

2.3.5 Risk Characterization

The HQ approach was used for three case studies (primary Pb smelter, secondary Pb smelter, and near roadway non-urban) and the national-scale ecological screening risk assessment, consisting of the surface water portion and the sediment portion. Measures of exposure for each location were divided by ecotoxicity screening values for each environmental medium (i.e., point estimates of Pb concentrations in soil, surface water, and sediment were divided by the corresponding ecotoxicity screening value). For each case study, as well as for the national-scale ecological screening assessment's surface water and sediment portions, HQ values were calculated where estimates of media Pb concentrations were available.

2.3.6 Uncertainty/Variability Analysis

For the ecological risk assessment, a qualitative evaluation of uncertainties was conducted and is described in Section 7.3.

3. Case Study Descriptions

As discussed in Chapter 2, a case study-based approach was employed for the pilot phase to estimate human exposure and health risks and ecological risks. This chapter, in sections 3.1 through 3.5, provides descriptions of each of the case study locations, accompanied by an overview of the available information relevant to and considered in the human exposure and health risk assessments and the ecological risk assessment. The surface water and sediment data used for the national-scale screening assessment are discussed in Section 3.6. Exhibit 3-1 provides a list of the case study and national-scale evaluations performed for the pilot phase analysis and indicates the applicability of each to the human exposure and health risk assessments and/or the ecological risk assessment.

Exhibit 3-1. Case Studies Included in the Pilot Phase

Case Study	Applicability
Primary Pb smelter location	Human and ecological
Secondary Pb smelter location	Human and ecological
Near roadway urban location	Human
Near roadway non-urban locations	Ecological
Ecologically vulnerable location	Ecological
National-scale surface water and sediment screening	Ecological

3.1 Primary Pb Smelter Case Study

The Herculaneum Lead Smelter (HLS) is one of the largest primary Pb smelters in the world and is the only currently operating Pb smelter in the United States. The HLS facility represents a relatively large point source that has been active for over a century and has a large amount of site-specific data characterizing both media concentrations (soil, indoor dust, and ambient air) and human blood Pb levels. Pb contaminant conditions for the area surrounding this facility are dominated by emissions from this facility, with older historical auto and other point source emissions being of relatively lesser importance. Environmental sampling conducted around the primary Pb smelter has shown Pb contamination throughout the community surrounding the smelter. Available environmental data are discussed in Section 3.1.5 and presented in Appendix A.

3.1.1 Description of Case Study Location

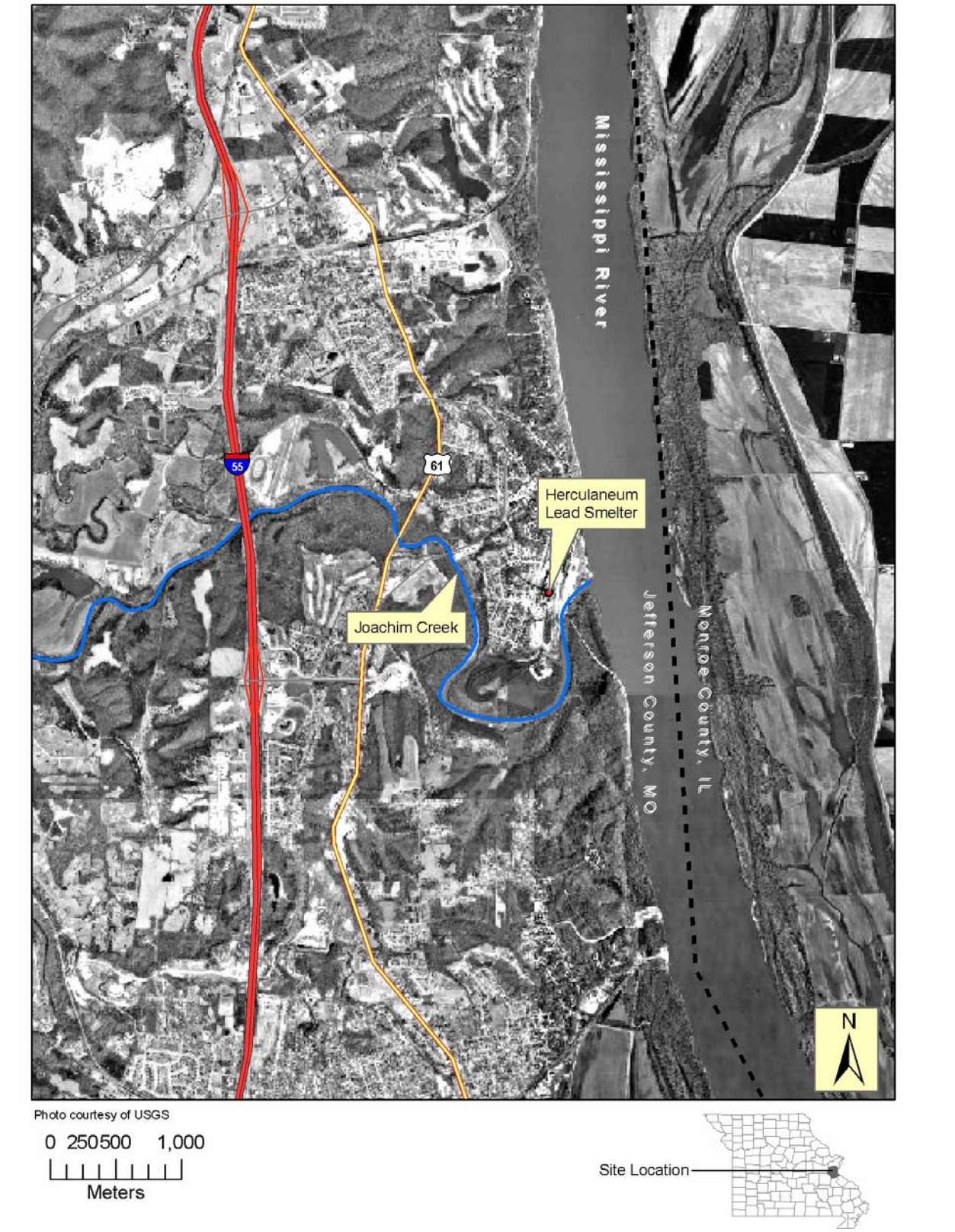
The HLS facility is located in Herculaneum, Missouri. The city of Herculaneum is in Jefferson County, about 42 kilometers (km) [26 miles (mi)] southwest of St. Louis, and its approximate area is 9 km^2. As of 2000, there were estimated to be 37,562 people living within a 10-km radius of the HLS facility (2,064 within 2 km; 14,237 between 2 and 5 km; and 21,261 between 5 and 10 km). Of this population in 2000, 3,880 were children ages 7 years old and younger (171 within 2 km; 1,545 between 2 and 5 km; and 2,164 between 5 and 10 km) (U.S. Census Bureau 2005).

3.1.2 Description of Primary Pb Smelter

The HLS facility is located at 881 Main Street in Herculaneum (see Exhibit 3-2). The HLS property is 52 acres and consists of three main areas: (1) the smelter plant, which is located on the east side of Main Street; (2) office buildings located on the west side of Main Street; and (3) a 40- to 50-feet high slag storage pile that covers 24 acres. The facility is bordered on the east by the Mississippi River, on the southeast by Joachim Creek, on the west and north-northwest by residential areas, and on the south-southwest by the slag pile. A large part of the slag pile is located in the floodplain wetlands of Joachim Creek and the Mississippi River.

The principal processing occurring at the facility includes: (1) sintering, smelting, and refining of Pb ore; (2) sulfuric acid production from waste sulfur-containing gases generated by the sintering operation; and (3) wastewater treatment. Sources at the facility include various stacks and vents from plant processes, fugitive emissions from ore handling operations, wind erosion from the slag pile, and fugitive emissions from transport of Pb concentrate over local roads. A Pb ore concentrate, consisting of approximately 80 percent Pb sulfide, is processed at the smelter. The ore is transported by truck from eight Pb mines near Viburnum, Missouri, approximately 42 km (75 mi) south-southwest of Herculaneum. The smelting operation generates a molten slag, 20 percent of which is sent to the slag storage pile as waste. Stack and fugitive emissions from the facility and deposition of these emissions to soil and surface water have resulted in elevated Pb concentrations in the surrounding areas (Missouri DNR 1999, as cited in ATSDR 2003).

Exhibit 3-2. Facility Location Map – Primary Pb Smelter



3.1.3 Human Exposure Measurements

Blood Pb levels at or above 10 µg/dL have been recorded for Herculaneum residents, including children less than 72 months of age (ATSDR 2002; ATSDR 2003). The U.S. Department of Health and Human Services (DHSS) and the Jefferson County Health Department (JCHD), in cooperation with the Agency for Toxic Substances and Disease Registry (ATSDR), have offered blood Pb (PbB) testing to the residents of Herculaneum and surrounding communities. Results of two such testing events conducted in 2001 and 2002 have been documented in DHSS/ATSDR health consultation reports (ATSDR 2002; ATSDR 2003) and are summarized here.

A total of 935 Herculaneum residents were tested in 2001. A summary of PbB results by age group is provided in Exhibit 3-3. Of the children less than 72 months old that were tested in 2001, 33 (28 percent) had PbBs of 10 µg/dL or greater. In the area closest to the smelter, 30 out of 67 (45 percent) of the children under 72 months of age who were tested in 2001 had PbBs equal to or above 10 µg/dL (ATSDR 2002).

Exhibit 3-3. Summary of 2001 PbB Measurements for Herculaneum Residents

PbB (µg/dL)	Number of Individuals
Children less than 72 Months	
0 – 9	85
10 – 19	27
20 – 29	5
30 or Higher	1
Children between 6 and 17 Years	
0 – 9	149
10 – 19	13
20 – 29	0
30 or Higher	0
Adults 18 Years or Older	
0 – 24	653
25 – 39	1
40 – 49	0
50 or Higher	1

Source: ATSDR 2002

In September 2002, DHSS and JCHD conducted a voluntary community-wide PbB testing event, and a total of 340 Herculaneum residents were tested. Results by age group for Herculaneum residents are summarized in Exhibit 3-4. As shown in Exhibit 3-4, of the children less than 72 months old that were tested in 2002, 8 (14 percent) had PbBs 10 µg/dL or higher.

Exhibit 3-4. Summary of 2002 PbB Measurements for Herculaneum Residents

PbB (µg/dL)	Number of Individuals
Children less than 72 Months	
0 – 9	50
10 – 19	6
20 – 29	2
30 or Higher	0
Children between 6 and 17 Years	
0 – 9	127
10 – 19	2
20 – 29	0
30 or Higher	0
Adults 18 Years or Older	
0 – 24	147
25 – 39	5
40 – 49	1
50 or Higher	0

Source: ATSDR 2003 (Tables 1-3)

3.1.4 Emissions

The Pb emissions estimates used for the primary Pb smelter case study were obtained from EPA Region 7 and reflect the 2000 Revision of the State Implementation Plan (SIP) developed for the facility (MDNR 2000). Cumulative Pb emissions from processes at the facility, fugitive emissions from transferring of materials, fugitive emissions from storage at the slag pile, and emissions associated with dust from roadways in the vicinity of the smelter totaled 266 tons/year. Particle sizes for emissions from road segment emission points around the primary Pb smelter ranged from 44 to 210 µm. Particle sizes for emissions from all other emission points at the primary Pb smelter ranged from 6.3 to 45.5 µm. EPA Region 7 is currently in the process of reviewing source characterization data to enhance modeling conducted as part of the SIP planning. Consequently, the dispersion model runs completed for the pilot phase using these emissions should be considered illustrative only. Emissions and release parameters, particle size inputs, and other inputs used for fate and transport modeling of the primary Pb smelter are provided in Appendix B.

3.1.5 Summary of Environmental Data

The environmental data sets available for the primary Pb smelter case study are summarized in Exhibit 3-5.

Exhibit 3-5. Summary of Environmental Data Sources for Primary Pb Smelter Case Study

Media	Data Set	Timeframe	Locations	Comments
Human Health [a]				
Ambient air	EPA-operated high volume samplers	2001 – 2003	4 locations	Located along roads; see Exhibits A-1, A-2
	EPA's Air Quality System (AQS)/AirData	2001 - 2005	9 locations	Monitors within 10 km of facility; see Exhibits A-1, A-3
Residential soil	Pre-excavation	2000 - 2004	Over 900 locations around primary Pb smelter	Locations within approximately 2.4 km (1.5 mi) of facility; see Exhibit A-4
	Post-excavation	2000 - 2004	Approximately 300 locations around primary Pb smelter	Locations within approximately 2.4 km (1.5 mi) of facility; see Exhibit A-5
	Recontamination Assessment	2002 - 2006	31 residences	Locations within approximately 1.6 km (1 mi) of facility; see Exhibits A-6, A-7
Indoor dust	Recontamination Assessment	2002 - 2006	17 residences	Locations within approximately 1.6 km (1 mi) of facility; See Exhibit A-8
Deposition to soil	Soil boxes	2003 – 2004 [b]	10 locations	See Exhibits A-9, A-10
Deposition to air	Filters	2003 – 2004 [b]	10 locations	See Exhibits A-9, A-11
Ecological [c]				
Non-residential soil	Site Characterization Investigation (ELM 2005)	2000	61 locations, 26 of which were included in the assessment	See Chapters 4 and 7 for more information
Surface water	Site Characterization Investigation (ELM 2005)	2000	62 locations, 30 of which were included in the assessment [d]	See Chapters 4 and 7 for more information
Sediment	Site Characterization Investigation (ELM 2005)	2000	82 locations, 69 of which were included in the assessment [e]	See Chapters 4 and 7 for more information

[a] Several data sources existed, including analyses conducted by EPA, the facility, ATSDR, Missouri Department of Natural Resources (DNR), and various consultants. Aside from EPA's AQS air monitoring data, the data represented in this table were obtained electronically from EPA Region 7 (2006a). These EPA data are the only environmental data discussed and summarized for the primary Pb smelter in this chapter and in the associated appendices. Attempts were made to obtain environmental data from sources outside EPA (such as those listed above), but no additional data were received within the time available for this assessment.

[b] These are the most recent data available from EPA Region 7.

[c] Some sampling locations were excluded from the ecological risk assessment because of their close proximity to the slag storage area; it was assumed that these areas had significant Pb contributions from pathways other than the air (e.g., runoff and flooding).

[d] The sampling locations were from eight water bodies/drainage areas including Joachim Creek, Mississippi River, U-shaped pond approximately 1.7 km NW of the facility.

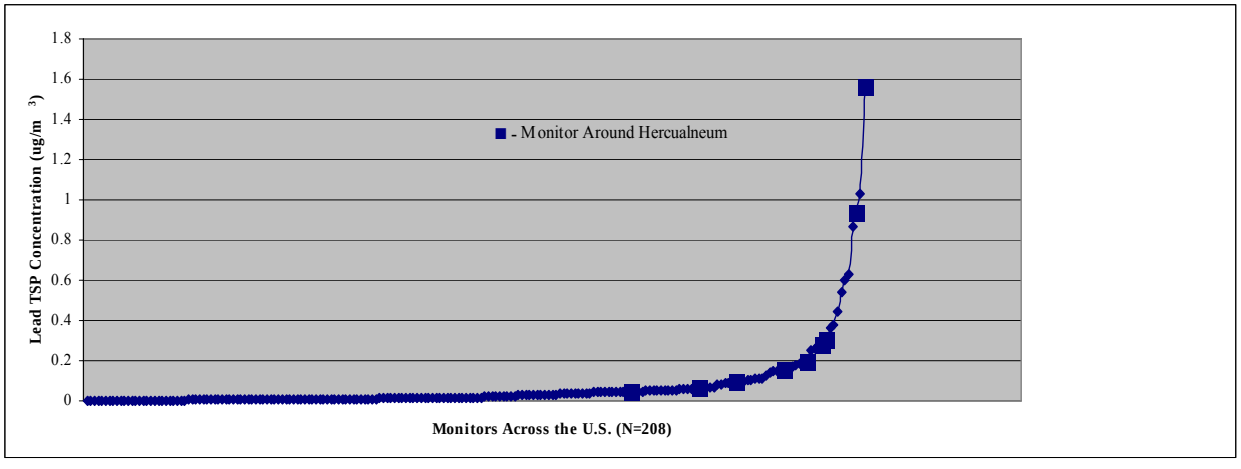
[e] The sampling locations were from five water bodies/drainage areas including Joachim Creek, Mississippi River, U-shaped pond approximately 1.7 km NW of the facility.

Air Monitoring: As shown in Exhibit 3-5, there are two air monitoring data sets available from EPA for the primary Pb smelter. Appendix A, Exhibit A-1 shows the locations of the 13 air monitoring locations relative to the facility. Air monitoring data for the four EPA high-volume sites are provided by year in Exhibit A-2. In general, average annual Pb concentrations in ambient air declined from 2001 to 2003 at these locations. Most notable are the reductions in measurements at the monitor with Pilot Analysis ID 102. The maximum Pb concentration (65 $\mu g/m^3$) occurred at this monitor in 2001. In 2003, the maximum at this monitor decreased to 10 $\mu g/m^3$ (a decrease of 85 percent). Likewise, the average annual Pb concentration at the monitor decreased 84 percent from 2001 to 2003.

Air monitoring data for the nine AirData monitors are provided by year in Exhibit A-3. Based on these data, decreases in average annual Pb concentrations have been observed since 2001. The largest difference was observed for Monitor ID 290990005, where average annual Pb concentrations decreased from 2.11 $\mu g/m^3$ in 2001 to 0.278 $\mu g/m^3$ in 2005 (a decrease of 87 percent).

For comparison purposes, the average annual Pb concentrations for the year 2005 from AirData monitors located around the primary Pb smelter were compared to AirData monitor results across the United States. Exhibit 3-6 shows the distribution of average annual Pb concentrations in total suspended particulate matter (TSP) for 208 monitors across the United States (with average annual monitored Pb concentrations sorted in ascending order). The 2005 monitor results for the nine AirData monitors located in the vicinity of the primary Pb smelter are indicated using a solid square (■). The average monitored Pb concentrations detected at the 208 monitors ranged from 0.001 to 1.56 $\mu g/m^3$. The 1.56 $\mu g/m^3$ maximum is associated with Monitor 290990015, one of the monitors identified within 10 km of the primary Pb smelter. Of the 208 locations, the nine monitors located within 10 km of the primary Pb smelter all fall in the top 30 percent of all monitored ambient levels for these 208 monitors, with four of the nine monitors in the top 10 percent.

Exhibit 3-6. Distribution of Pb TSP Measurements Across the United States Relative to Monitors Near the Primary Pb Smelter



Soil: As shown in Exhibit 3-5, there are three soil data sets available from EPA for the primary Pb smelter: pre-excavation, post-excavation, and recontamination assessment data. Pre-excavation soil samples were collected from residential locations around the smelter prior to soil removal activities. Pre-excavation soil sample results for over 900 residential locations around the primary Pb smelter are presented in Appendix A, Exhibit A-4. Average soil concentrations at these sampling locations ranged from 53 to 23,350 mg/kg.

Based on pre-excavation sampling results, Pb-contaminated soil in residential yards near the smelter was removed, replaced with clean backfill, and re-seeded with grass. Post-excavation soil data were available for over 300 residential locations. It is unclear from the available information whether post-excavation soil samples were collected immediately following excavation or after the yards were backfilled. Post-excavation results are presented in Appendix A, Exhibit A-5. Average soil concentrations at these properties ranged from 70 to 2,757 mg/kg.

The EPA has recently conducted post-remediation residential yard soil sampling at 31 locations within approximately a 1.6 km (1 mi) radius of the primary Pb smelter to determine whether residential soils previously excavated are becoming recontaminated. Results for the recontamination assessment samples are provided in Appendix A, Exhibit A-6. For most of the 31 recontamination assessment locations, average Pb concentrations in the replacement "clean" soil increased between 2002 and 2006. Refer to Appendix A, Exhibit A-7 for a summary of the pre-excavation, post-excavation, and recontamination assessment data for these 31 residential locations.

Indoor Dust: The interiors of 17 of the 31 residential properties identified for the soil recontamination assessment were also assessed for Pb levels in indoor dust. Interior dust removal (in which areas inside homes were wiped and/or vacuumed) was performed at these residences prior to recontamination sampling. Appendix A, Exhibit A-8 provides a summary of recontamination indoor dust sample results for these 17 properties. Carpet dust samples collected during recontamination sampling events at these residences contained Pb concentrations that ranged from 122 to 4,350 mg/kg. Pb loadings in window sill wipe samples ranged from 5.6 to 1,385 $\mu g/ft^2$. There were no general patterns identified at homes during successive sampling events. Pb concentrations and/or loadings may have increased, decreased, or remained generally the same (see Appendix A, Exhibit A-8). This lack of pattern may be attributed in part to inconsistent house cleaning protocols within the homes.

Deposition: As shown in Exhibit 3-5, soil boxes[1] were set up at 10 locations around the primary Pb smelter. Deposition monitoring locations are shown in Appendix A, Exhibit A-9. From 2003 to 2004, samples were collected monthly to measure Pb deposition on soil; results for these locations are presented in Appendix A, Exhibit A-10. Maximum concentrations at the nine locations (excluding the control site) ranged from 25 to 406 mg/kg in 2003 and from 25.3 to 527 mg/kg in 2004. The overall average Pb concentration in these soil boxes across all nine locations increased from 49 mg/kg in 2003 to 96.5 mg/kg in 2004, an increase of almost 100 percent.

[1] Clean soil is placed in containers that measure approximately 2 feet by 3 feet. Soil boxes were intended to provide a repeatable means of measuring Pb deposition on soil that would be less likely to be disturbed than soil in residential yards.

Air deposition monitoring data were available for the same 10 locations around the primary Pb smelter for which soil box monitoring data were available (see Appendix A, Exhibit A-9). Dry deposition samples were collected monthly at two levels (1 foot and 10 feet) above the ground surface from April 2003 through April 2004. Data collected at each level for these locations are presented in Appendix A, Exhibit A-11. The annual Pb deposition rates at a height of 1 foot for the nine monitoring locations (excluding the control site) ranged from 0.34 to 22 mg/ft^2, and the overall average Pb deposition rate across all nine locations at the height of 1 foot was 4.8 mg/ft^2. The annual Pb deposition rates at a height of 10 feet for the nine monitoring locations ranged from 0.26 to 33 mg/ft^2, and the overall average Pb deposition rate across all nine locations at the height of 10 feet was 5.0 mg/ft^2. The average annual Pb deposition rates at each level by location are provided in Exhibit A-11.

3.1.6 Ecological Characterization

3.1.6.1 Description of Site and Surroundings

Ecologically relevant features near the primary Pb smelter facility include the Mississippi River, Joachim Creek, emergent and scrub-shrub wetlands, and successional and mature bottomland hardwood forest tracts (ELM 2005). Bottomland hardwood forests and agricultural fields are present to the west, south, and east of the characterization area between the smelter's slag storage area (SSA) and Joachim Creek. The most mature bottomland hardwood forest is adjacent to Joachim Creek. Immediately south of the facility is a mixture of floodplain forest, emergent marsh, and scrub-shrub wetland habitat that is populated by willow (*Salix spp.*) trees (ELM 2005).

The Missouri Department of Conservation (MDC) lists the plant and animal species that are endangered, vulnerable, or imperiled by county in the Heritage Database (MDC 2006). Exhibit 3-7 lists the species of concern for Jefferson County; species that have acquired the state or federal status of "endangered" are identified. As shown in Exhibit 3-7, the pink mucket, scaleshell, gray bat, Indiana bat, and pallid sturgeon are on both the state and federal endangered lists.

Exhibit 3-7. Species of Concern Occurring in Jefferson County, Missouri[a]

Animals		Plants	
Lake sturgeon (SE)	*Acipenser fulvescens*	Wild leek	*Allium burdickii*
Elktoe	*Alasmidonta marginata*	Wild sarsaparilla	*Aralia nudicaulis*
Alabama shad	*Alosa alabamae*	Forked aster	*Aster furcatus*
Brown bullhead	*Ameiurus nebulosus*	A moss	*Campylium polygamum*
Western sand darter	*Ammocrypta clara*	A sedge	*Carex texensis*
Rock pocketbook	*Arcidens confragosus*	Fremont's leather flower	*Clematis fremontii*
Great egret	*Ardea alba*	Nuttall tick-trefoil	*Desmodium nuttallii*
Swamp metalmark	*Calephelis muticum*	A lichen	*Dibaeis absoluta*
Highfin carpsucker	*Carpiodes velifer*	Pale avens	*Geum virginianum*
Eastern collared lizard	*Crotaphytus collaris collaris*	Fir clubmoss	*Huperzia porophila*
Crystal darter (SE)	*Crystallaria asprella*	Weak rush	*Juncus debilis*

Exhibit 3-7. Species of Concern Occurring in Jefferson County, Missouri[a]			
Animals		**Plants**	
Blue sucker	*Cycleptus elongatus*	A moss	*Mnium thomsonii*
Eastern hellbender (SE)	*Cryptobranchus alleganiensis*	A liverwort	*Nardia lescurii*
Spectaclecase (FCE)	*Cumberlandia monodonta*	Stemless evening primrose	*Oenothera triloba*
Elephantear (SE)	*Elliptio crassidens*	A panic grass	*Panicum dichotomum var. nitidum*
Snuffbox (SE)	*Epioblasma triquetra*	Missouri cliffbrake	*Pellaea glabella var. missouriensis*
Proserpine cavesnail	*Fontigens proserpina*	Wild sweet william	*Phlox maculata ssp. Pyramidalis*
Ebonyshell (SE)	*Fusconaia ebena*	A liverwort	*Preissia quadrata*
Mooneye	*Hiodon tergisus*	Sullivantia	*Sullivantia sullivantii*
Pink mucket (SE; FE)	*Lampsilis abrupta*		
Scaleshell (SE; FE)	*Leptodea leptodon*		
Black sandshell	*Ligumia recta*		
Silver chub	*Macrhybopsis storeriana*		
Gray bat (SE; FE)	*Myotis grisescens*		
Indiana bat (SE; FE)	*Myotis sodalist*		
Ghost shiner	*Notropis buchanani*		
Hickorynut	*Obovaria olivaria*		
Belted crayfish	*Orconectes harrisonii*		
River darter	*Percina shumardi*		
Sheepnose (SE; FCE)	*Plethobasus cyphyus*		
Paddlefish	*Polyodon spathula*		
Wood frog	*Rana sylvatica*		
Eastern spadefoot	*Scaphiopus holbrookii*		
Pallid sturgeon (SE; FE)	*Scaphirhynchus albus*		

[a] Abbreviations: SE = state endangered status; FE = federally endangered status; FCE = federal candidate for endangered status; Source: MDC (2006).

Throughout much of the year, migratory birds such as the red-tailed hawk, belted kingfisher, and great blue herons utilize the habitat near the primary Pb smelter. The state and federally endangered bald eagle has been spotted on-site at the primary Pb smelter facility, which is known to be within the habitat for the bird. The facility is also within the habitat of the Indiana bat, which, as mentioned above, is also on the state and federal endangered species lists. None of these species, however, feed on soil invertebrates. In addition, the state and federally endangered pallid sturgeon has been identified in the Mississippi River adjacent to and downstream of the facility (USEPA Region 7 2000).

3.1.6.2 Description of Available Pb Environmental Data

As shown in Exhibit 3-5, the data set for the ecological risk assessment includes Pb concentration data in surface soil, surface water, and sediment collected from non-residential locations near the primary Pb smelter (ELM 2005). The data set sampling locations include

areas within a 2.1 km (1.5 mi) radius of the facility and two "reference areas" that are 6 to 7 km south of the facility (ELM 2005).

Soil: Total Pb concentration data are available for soil samples occurring in six clusters of locations in forested areas within 0.3 km of Joachim Creek that are remote from residential development (ELM 2005, Figure 7): the western and southern slopes of the slag storage area (SSA); sites along central, east, and west transects from the SSA; and sites located along the historic railroad grade on the west side of Joachim Creek. Soil samples near Joachim Creek may reflect Pb contribution from runoff from the SSA, and flooding from Joachim Creek (ELM 2005). The soil sampling locations within a 2.1 km radius were all in areas that might have been subject to Pb inputs from Joachim Creek during flooding events. As such, the stations might not represent the concentrations of Pb in soils that result from direct air emissions from the smelter. The reference soil samples were taken from areas approximately 5.6 and 6.5 km south of the facility. Soil samples were collected from 0 to 3 inches below the surface (ELM 2003).

The average total Pb soil concentration for the entire ELM (2005) data set was 269 mg/kg. The minimum concentration detected was 20.4 mg/kg, and the maximum was 752 mg/kg.

Surface water and sediment: Dissolved Pb concentration data in surface water samples and total Pb concentration data for sediment samples are available for sites along the Mississippi River, including one reference station more than 2.1 km upstream of the facility, and sampling stations within approximately 2.5 km upstream and downstream of the confluence with Joachim Creek. Data are also available for sites in Joachim Creek along the approximately 4.4 km of Joachim Creek that lie within 2.1 km of the smelter, as well as two locations farther upstream. Pb measurements are also available in nearby wetlands (ELM 2005).

Dissolved Pb was not detected by ELM in samples from surface water locations included in the pilot phase, and the detection limit for the analysis was 3 μg/L. Total Pb concentrations in surface water for the full data set averaged 14.5 μg/L and ranged from below the detection limit to 105 μg/L. Total Pb concentrations in the sediment samples for the full data set averaged 263 mg/kg and ranged from 6.2 to 3,040 mg/kg.

3.2 Secondary Pb Smelter Case Study

The secondary Pb smelter case study focused on the impacts of emissions from a smaller point source (compared to the primary Pb smelter) located in Alabama. There were less site-specific data characterizing media concentrations and human exposure levels available for this study area than for the primary Pb smelter case study. However, recent air concentration data from the area surrounding the facility and facility characterization data (including emission estimates) were readily available.

3.2.1 Description of Case Study Location

The secondary Pb smelter case study location is in Troy, Alabama. Troy is a city located in Pike County, positioned in the south central portion of the state, and its approximate area is 68 km^2. As of 2002, there were estimated to be 17,910 people living within a 10-kilometer radius of the

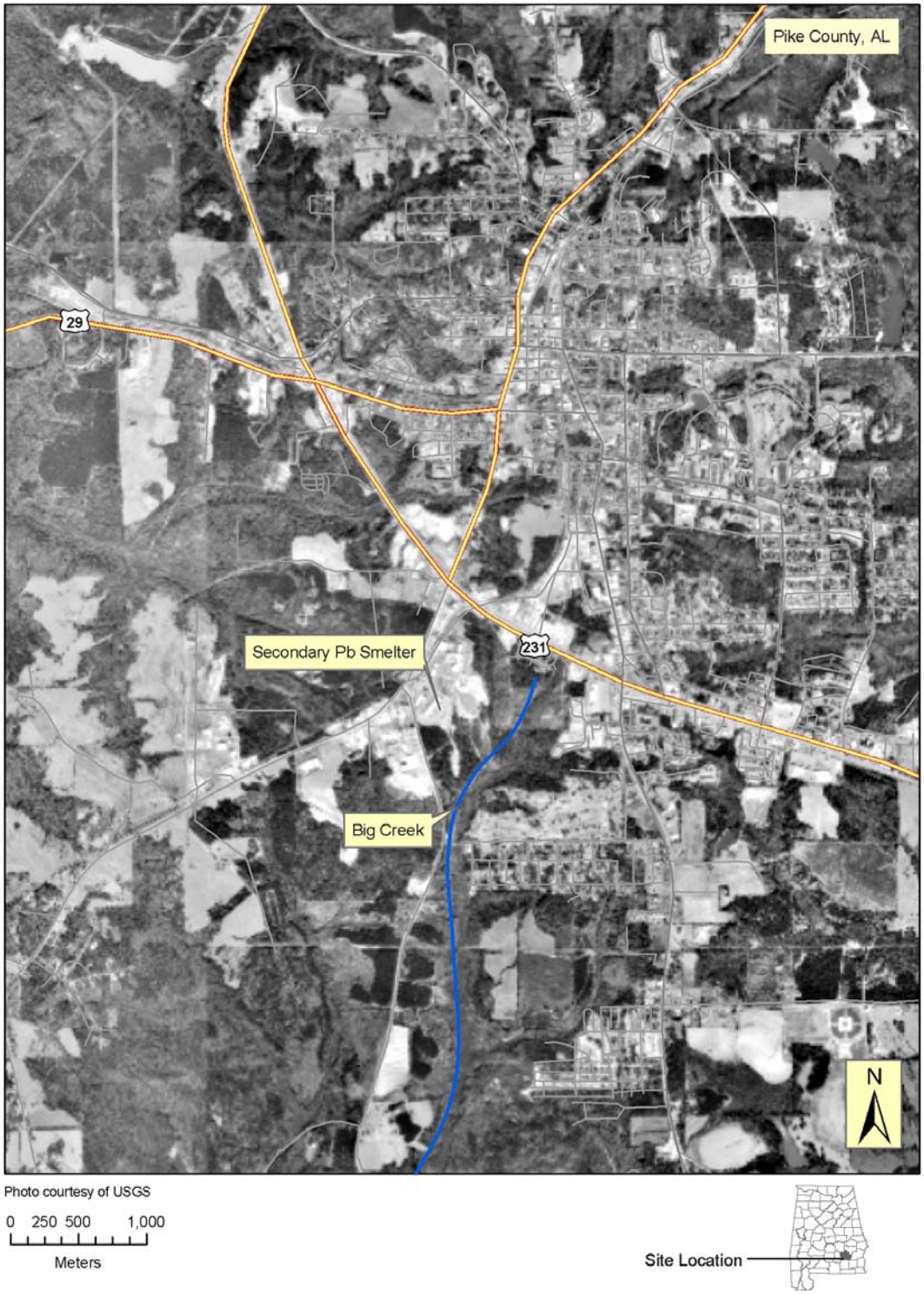
facility (2,186 within 2 kilometers; 10,634 between 2 and 5 km; and 5,090 between 5 and 10 km). Of this population, 1,672 are children ages 7 years and under (187 within 2 km; 896 between 2 and 5 km; and 589 between 5 and 10 km) (U.S. Census Bureau 2005).

3.2.2 Description of Secondary Pb Smelter

This facility is one of 15 secondary Pb smelters currently operating in the United States (ECR 2006). US-231 borders the facility to the north-northeast and a railroad line and Henderson Highway run along the north-northwest and west boundaries of the facility. The area located directly west of Henderson Highway is forested. To the south and south-southwest are other industries and businesses. Big Creek appears to be the closest major water body, located approximately 0.8 km (0.5 mi) south-southeast from the center of the facility. The City of Troy is located north and east of the facility and north of US-231 (see Exhibit 3-8).

Secondary Pb smelters produce Pb from scrap and provide the primary means for recycling Pb-acid automotive batteries. Approximately 95 percent of all Pb-acid batteries are recycled at secondary Pb smelters. Secondary Pb smelters perform three basic unit operations: battery breaking, smelting, and refining and alloying. Battery breaking is accomplished by either crushing or cutting battery cases into pieces. The plastic, spent acid, and Pb-bearing materials are then separated. Pb-bearing materials are processed in one of three types of smelting furnaces: blast, reverberatory, or rotary. Molten Pb from these furnaces is further processed in refining kettles and subsequently cast into molds. The waste stream from the furnaces (i.e., slag) is either returned to the primary smelting furnace or treated in a separate furnace dedicated to slag cleaning to recover additional Pb. There are three types of emission sources at secondary Pb facilities – process sources, process fugitive sources, and fugitive dust sources. The types of sources at the secondary Pb smelter analyzed in these assessments include: blast furnace, agglomeration furnace, alloying kettles and heating system, flue dust storage bins, and slag treatment furnace. Stack emissions from the facility and fugitive emissions associated with materials storage and handling and roadway dust have resulted in releases of Pb to the air and soil (ECR 2006).

Exhibit 3-8. Facility Location Map – Secondary Pb Smelter



3.2.3 Human Exposure Measurements

No information on children's PbBs specific to the area around the secondary Pb smelter was identified. However, the Lead Poisoning Prevention Branch of the CDC collected PbB surveillance data for children less than 72 months of age in Pike County, Alabama in 2003. Of the 407 children tested by the CDC, there were 33 (approximately 8 percent) confirmed cases of elevated PbB (i.e., PbB above 10 µg/dL). For children less than 72 months of age in the state of Alabama and in the United States as a whole, the confirmed elevated PbBs as a percent of children tested in 2003 was 2.4 percent and 1.9 percent, respectively (CDC 2003). Note, however, that the statistics for children in Pike County do not necessarily represent PbBs for children living in Troy, Alabama.

3.2.4 Emissions

As of June 9, 1994, when EPA proposed the secondary Pb smelter MACT standard (59 FR 63941), there were 23 secondary Pb smelters operating in the United States. As of 2002, there were 15 operating facilities in 11 states. Of these 15 facilities, the secondary Pb smelter analyzed in this study is the third highest emitter of Pb (ECR 2006).

The estimates for process emissions for the secondary Pb smelter analyzed in this study were calculated from Pb emissions measured during stack tests performed December 1997, November 1999, and February 2000 (ECR 2006). Fugitive emissions were estimated by comparing the modeled concentrations from the stack emissions to background Pb concentrations and monitored concentrations. The cumulative Pb emissions from this facility, including facility process and fugitive emissions, were estimated to be 4.56 tons/year. Particle sizes for emissions from point sources at the facility ranged from 0.5 to 10 µm, and particle sizes for emissions from area sources at the facility ranged from 1.25 to 22.5 µm. Emissions and release parameters, particle size inputs, and other inputs for fate and transport modeling for the facility are provided in Appendix C.

3.2.5 Summary of Environmental Data

The environmental data sets available for the secondary Pb smelter case study are summarized in Exhibit 3-9.

Exhibit 3-9. Summary of Environmental Data Sources for Secondary Pb Smelter Case Study [a]

Media	Dataset	Timeframe	Locations	Comments
Human Health				
Ambient air	EPA's AQS/AirData	1997 – 2000 [b]	2 locations	Located 400 and 680 meters from the facility; see Exhibits A-12, A-13.
Residential soil	No data identified.			
Indoor dust	No data identified.			
Deposition	No data identified.			
Ecological				
Non-reidential soil	No data identified.			
Surface water	No data identified.			
sediment	No data identified.			

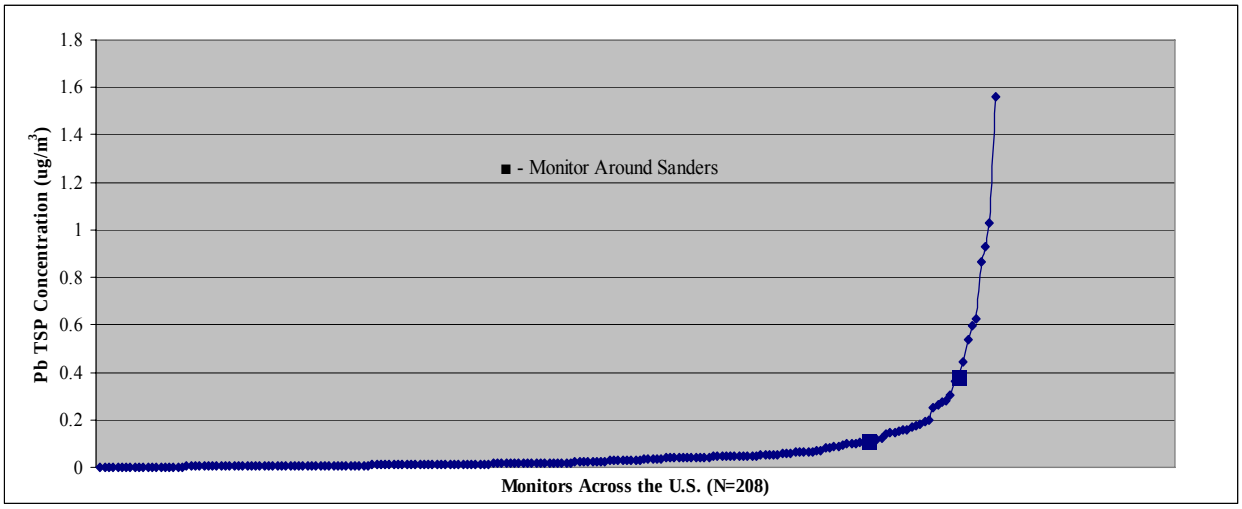
[a] In general, site characterization information was lacking for this secondary Pb smelter. Data, with the exception of limited air monitoring data, were not available based on information from EPA Region 4. Information from the Alabama Department of Environmental Management (ADEM) indicates there may be relevant soil data available from the facility (ADEM 2006); however, no data have been obtained to date.

[b] AirData monitor values from 1997 - 2000 were obtained for the purpose of comparing monitored values to modeled air concentrations (see Chapter 4). Because emissions data for the secondary Pb smelter were based on stack tests for 1997, 1999, and 2000, monitoring results from similar years were used.

Air Monitoring: As shown in Exhibit 3-9, average annual Pb concentrations in the vicinity of the secondary Pb smelter were available from EPA's AQS/AirData network (USEPA 2006f) for two air monitors located near the facility (see Appendix A, Exhibit A-12). Data from these two air monitoring sites for 1997 through 2000 are presented in Appendix A, Exhibit A-13. Over this period, average annual Pb concentrations at the monitor closer to the facility ranged from 0.383 to 0.474 $\mu g/m^3$, with the lowest average annual concentration in year 2000. Average annual Pb concentrations at the second monitor ranged from 0.132 to 0.198 $\mu g/m^3$.

For comparison purposes, the average annual Pb concentrations for the year 2005 from AirData monitors located around the secondary Pb smelter case study location were compared to AirData monitor results across the United States Exhibit 3-10 shows the distribution of average annual Pb concentrations in total suspended particulate matter (TSP) for 208 monitors across the United States (with average annual monitored Pb concentrations sorted in ascending order). The 2005 monitor results for the two AirData monitors located in the vicinity of the secondary Pb smelter are indicated using a solid square (■). The average monitored Pb concentrations from the 208 monitors ranged from 0.001 to 1.56 $\mu g/m^3$. Both of the monitors located near the secondary Pb smelter fall into the top 15 percent of the 208 locations.

Exhibit 3-10. Distribution of Pb TSP Measurements Across the United States Relative to Monitors Near the Secondary Pb Smelter



Soil: No soil measurement data for Pb were identified in the vicinity of the secondary Pb smelter case study location. For the human exposure and health risk assessments, soil concentrations were estimated using two methods – a model-only approach and a hybrid approach using model results and surrogate measurements from a similar facility. See Section 4.2.3 for details.

Indoor Dust: No indoor dust data for Pb were available from homes located in the vicinity of the secondary Pb smelter. Indoor dust concentrations were estimated using an empirical model that relates ambient air concentrations to indoor dust concentrations, as discussed in Chapter 4.

Deposition: No Pb deposition monitoring data were identified in the vicinity of the secondary Pb smelter case study location. For the human exposure and health risk assessments and the ecological risk assessment, Pb deposition arising from emissions from the secondary Pb facility was modeled using EPA's AERMOD air dispersion model, as discussed in Chapter 4.

3.2.6 Ecological Characterization

3.2.6.1 Description of Site and Surroundings

The secondary Pb smelter location falls within the Alabama Coastal Plain in Pike County, Alabama. It is located in an area of disturbed forests, and is less than 1.6 km (1 mi) from Big Creek, which is part of the Pea River watershed. Big Creek is located approximately 0.8 km (0.5 mi) south-southeast from the center of the facility. The surrounding area includes emergent and scrub-shrub wetlands, forests, freshwater creeks, ponds, rivers, croplands, pastureland, and developed urban areas. The Pea River watershed drains into the Gulf of Mexico. The watershed is underlain by coastal plain sediments, including sand, clay, and limestone; and the topography can be characterized as gentle to moderate rolling hills (CPYRWMA 2006).

Diversity of terrestrial and aquatic animal species is relatively high. The Choctawhatee and Pea River basins, in which the secondary Pb smelter is located, contain 43 species of marine, estuarine, and freshwater fish species (Cook and Kopaska-Merkel 1996). Anadromous fish species (i.e., saltwater fish that must spawn in freshwater) found in the Pea River basin include the following: the threatened Gulf sturgeon (*Acipenser oxyrhynchus desotoi*), Alabama shad (*Alosa alabamae*), striped bass (*Morone saxatalis*), and skipjack herring (*Alosa chrysochloris*). The Pea River basin also provides habitat for 20 species of freshwater mussels (Cook and Kopaska-Merkel 1996), as well as numerous species of snails, snakes, and other invertebrates.

Terrestrial species supported in this region include a variety of birds, mammals, invertebrates, and vascular plants. A recent floristic survey of the Pike County Pocosin Nature Preserve indicated that there were a total of 348 species of vascular flora (Alvin et al. 2002). The preserve is approximately 11 km (7 mi) northeast of the secondary Pb smelter facility. In a recent survey of birds spotted by citizens of Troy, Alabama, 29 species were reported, including American robins (which consume soil invertebrates), the northern cardinal, American goldfinch, and various sparrows, doves, woodpeckers, and warblers (GBBC 2003). Other terrestrial fauna found in the region include small mammals and invertebrate species. A total of 34 vascular flora from Pike County are listed by the Alabama Natural Heritage Inventory Program as endangered, threatened, or of special concern in the state (Alabama Natural Heritage Inventory 2001).

Exhibit 3-11 displays the current listing of threatened and endangered species for Pike County, Alabama (NatureServe 2006). According to NatureServe and the USFWS, no species in Pike County are on the state or federal endangered species list (Outdoor Alabama 2003). A few species, however, are candidates for the federal list, as designated in Exhibit 3-11.

Exhibit 3-11. Threatened and Endangered Species in Pike County, Alabama[a]

Animals		Plants	
Invertebrates			
Delicate spike	*Elliptio arctata*	Apalachicola Wild Indigo	*Baptisia megacarpa*
Variable spike	*Elliptio icterina*	Baltzell's Sedge	*Carex baltzellii*
Purple pigtoe	*Fusconaia succissa*	Southern Twayblade	*Listera australis*
Southern sandshell (FCE)	*Hamiota australis*	American Pinesap	*Monotropa hypopithys*
Southern fatmucket	*Lampsilis straminea claibornensis*	Alabama Black Cherry	*Prunus alabamensis*
Fuzzy pigtoe (FCE)	*Pleurobema strodeanum*	Arkansas Oak	*Quercus arkansana*
Giant floater	*Pyganodon grandis*	Eared Coneflower	*Rudbeckia auriculata*
Tapered pigtoe (FCE)	*Quincuncina burkei*	Silky Camellia	*Stewartia malacodendron*
Southern creekmussel	*Strophitus subvexus*	Sessile-leaved Warea	*Warea sessilifolia*
Lilliput	*Toxolasma parvus*		
Pondhorn	*Uniomerus tetralasmus*		
Paper pondshell	*Utterbackia imbecillis*		
Choctaw bean (FCE)	*Villosa choctawensis*		
Little spectaclecase	*Villosa lienosa*		
Southern rainbow	*Villosa vibex*		
Vertebrates- Amphibians			
Two-toed amphiuma	*Amphiuma means*		
Vertebrates-Reptiles			
Coachwhip	*Masticophis flagellum*		

[a] Abbreviation: FCE = Federal candidate for endangered status

Source: NatureServe (2006)

3.2.6.2 Description of Available Data

Pb data in environmental media other than air were not available around the vicinity of the secondary Pb smelter facility. Therefore, for estimates of Pb concentrations in outdoor soils, the ecological risk screening assessment for the secondary Pb smelter case study relied on fate and transport modeling and input data collected for the human exposure and health risk assessments.

3.3 Near Roadway Urban Case Study

The location for the near roadway case study was selected primarily based on the availability of suitable air monitoring data. Specifically, measurements were sought that were inclusive of Pb concentrations and particle sizes, reported for an urban site adjacent to a roadway, and located in a dry climate where there was historical use of leaded gasoline. Several days of air monitoring data for ambient measurements of particle-bound Pb were available for an urban site adjacent to a roadway in Houston, Texas (ICF 2006a). This location was selected for the pilot phase, and the section of land adjacent to a segment of a well-traveled road near the monitoring site was defined as the specific study area. This road has been in use as a thoroughfare since at least the 1950s; therefore, it seems likely that the historical use of leaded gasoline in automobiles during this time has resulted in an accumulation of Pb in the near roadway soil.

3.3.1 Description of Case Study Location

The setting for the near roadway case study location is a small area of mixed residential and commercial use about 14 km northwest of downtown Houston (see Exhibit 3-12). The study area is the narrow strip of land that extends 200 meters from either side of a well-traveled roadway that is located south of the selected air monitoring station. The area is focused on a 2.3 km stretch of this street that runs roughly east-west, with the monitoring station situated about 115 m north of the street. This length was defined as the segment for the case study based on the subdivision of the street into "links." Roadways are typically divided into links for the purposes of organizing data by departments of transportation and other municipal offices; the study area comprises two links of this street. Relevant vehicle traffic data for these links are presented in Exhibit 3-13.

The land adjacent to the segment of road that makes up the study area was subdivided into three zones for this case study: one extending from the edge of the roadway out to 12 meters; one that includes the region between 12 to 75 meters from the roadway edge; and one that includes the region between 75 to 200 meters from the roadway edge (see Exhibit 3-12). Additional information regarding the spatial characteristics of the study area and the rationale for subdividing the area into three zones is included in Section 4.3.1.

Exhibit 3-12. Study Area for Near Roadway Case Study, Houston, Texas

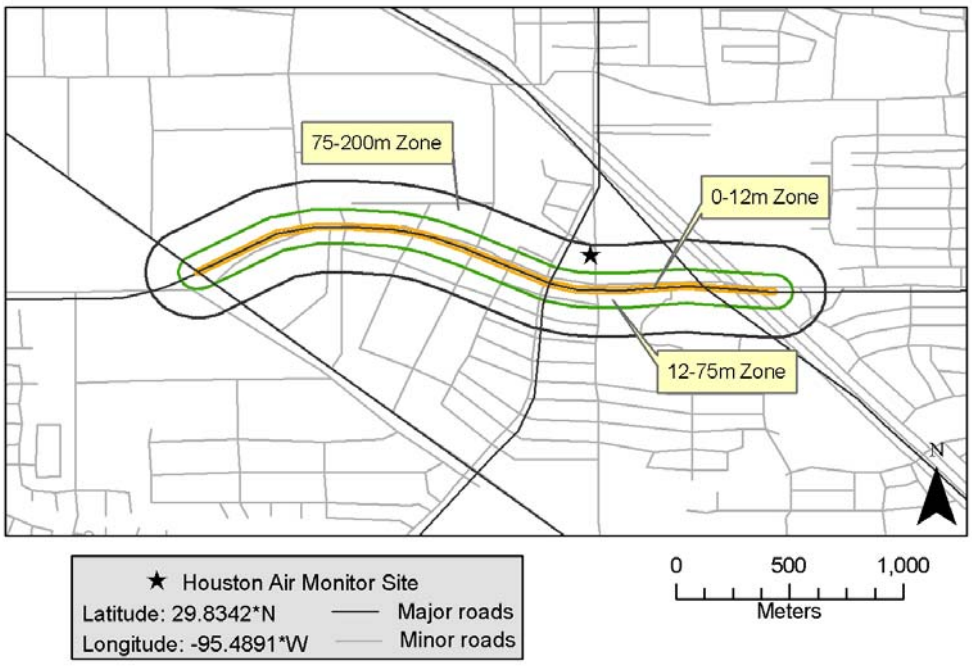


Exhibit 3-13. Vehicle Traffic Data for Road Segment of Interest

Road Link Section	Link Length (km)	Approx. Annual Average Daily Throughput (AADT), Vehicles [a]
Segment east of site	0.7	22,000
Segment west of site	1.6	17,000

[a] Length and vehicle throughput from Highway Performance Monitoring System (U.S. Department of Transportation 2006); data are representative of 2002 and 2003.

As of 2000, about 1,950 people resided in 620 households within the boundaries of this study area, including about 320 children between the ages of 0 and 7 years. Demographic data for the case study were estimated using information from the U.S. Census by area-weighting population data for census blocks overlapping with the study area. More information on these calculations and relevant demographics for each of the three zones in the study area is included in Appendix D.

3.3.2 Human Exposure Measurements

No human exposure measurements were identified for the near roadway urban study area.

3.3.3 Emissions

No emissions data were collected for the near roadway case study. Soil and air Pb concentrations were derived from measurements; and no modeling of emissions was conducted for this case study.

3.3.4 Summary of Environmental Data

The environmental data sets identified for the near roadway case study are summarized in Exhibit 3-14.

Exhibit 3-14. Summary of Environmental Data Sources for Near Roadway Urban Case Study

Media	Data Set	Timeframe	Location	Comments
Ambient air	DRI: 1 monitor ($PM_{2.5}$ and PM_{10})	February 2001; 3 daily averages (ICF 2006b)	115 m from road segment of interest	Used to characterize near roadway air concentrations
	AQS/AirData: 3 monitors ($PM_{2.5}$, PM_{10}, TSP)	1996 and 2006; multiple observations (USEPA 2006f)	Houston (15-30 km from road segment)	Distance to nearest roadway not specified
Soil	Turer and Maynard (2003): near roadway measurements	October 1998 [a]	Corpus Christi; I-37 entrance ramp near downtown; 2, 3, and 12 m from roadway	Adjacent to roadway; higher traffic volume than study area
Indoor dust	No data identified.			
Deposition	No data identified.			

[a] Sample data for soil confirmed by Maynard (2006).

Two air monitoring data sets were identified for locations in the vicinity of the road segment that is the focus of this case study. Air monitoring of Pb was conducted by Desert Research Institute (DRI) at a single monitor located about 115 m north of the road segment. Daily average air concentrations of Pb in the PM_{10} fraction collected over three days in February 2001 ranged from 0.0045 to 0.0056 $\mu g/m^3$. In addition, EPA's AirData database includes measurements of Pb for three air monitors located in Houston. None of these monitors is located within 15 km of the road segment of interest, and the distance to the nearest roadway was not specified for any monitor. Mean concentrations of Pb in the TSP fraction ranged from 0.001 to 0.02 $\mu g/m^3$ for 1996 to 2006 across the three monitors. Mean concentrations of Pb in the PM_{10} fraction ranged from 0.006 to 0.012 $\mu g/m^3$ for 1999 to 2006 for the one monitor with PM_{10} measurements.

Although no suitable soil Pb measurements were identified for locations in the vicinity of the road segment of interest or elsewhere in Houston, one set of near roadway soil measurements in Corpus Christi was identified in the literature. Turer and Maynard (2003) reported concentrations of Pb in surface soil samples taken adjacent to an interstate entrance ramp; concentrations ranged from 214 mg/kg at 12 m from the roadway to 766 mg/kg at 3 m from the roadway. See Appendix D for additional detail on the air and soil data sets. No measurements of Pb in dust inside residences or measurements of deposition were identified in the vicinity of the near roadway urban case study location.

3.4 Near Roadway Non-Urban Case Study

From the literature search for studies of Pb in near roadway soils (described in Section 3.3.4), two non-urban sites for which soil Pb levels are available were identified for use in the ecological risk assessment. These locations are: (1) Interstate 37 near oil refineries in Corpus Christi, Texas (Turer and Maynard 2003) and (2) Interstate 95 north of Richmond, Virginia, at Exit 86 which connects to a moderately traveled, two-lane road (Speiran 1998).

Land cover data from 1992 within 1.6 km (1 mi) of the Corpus Christi, Texas study location showed 59 percent industrial, 10 percent low intensity residential, and 25 percent high intensity residential (Vogelmann et al. 2001). The remaining 5 percent of the surrounding area included shrubland, row crops, pasture, grasses, and forested upland, including evergreen forest and deciduous forest. The 1992 land cover data within approximately 1.6 km (1 mi) of the Atlee, Virginia, study locations was 26 percent developed: 2 percent low-intensity residential and commercial and 24 percent industrial and transportation. The remaining 74 percent included 25 percent deciduous forest, 14 percent woody wetlands, and 12 percent pasture (Vogelmann et al. 2001). Smaller proportions of mixed forest, evergreen forest, row crops, and transitional (barren) areas were also found.

Exhibit 3-15 lists the environmental data sources for the near roadway non-urban case study.

Exhibit 3-15. Summary of Environmental Data Sources for Near Roadway Non-Urban Case Study

Media	Data Set	Timeframe	Locations	Comments
Ecological				
Soil	Corpus Christi, Texas Site No. 2 (Turer and Maynard 2003)	~1998 [a]	4 locations taken from a transect perpendicular to the road (0.5, 1.2, 2, and 4 m from the road)	See Chapters 4 and 7 for more information
	Atlee, Virginia (Speiran 1998)	1994	27 locations taken along a 140-m section of interstate, ranging from 2 to 30 m from road	

[a] Sample data for soil confirmed by Maynard (2006).

The Pb soil data for the Corpus Christi, Texas location 2 meters from the roadway ranged from 15 mg/kg at 20-cm depths to 650 mg/kg at 10-cm depths. The Pb soil concentrations at the Atlee, Virginia location ranged from 17 mg/kg 15 meters from the roadway to 540 mg/kg 2 meters from the roadway; both of these Atlee samples were collected from 7.5-to 15-cm soil depths.

3.5 Ecologically Vulnerable Case Study

The Hubbard Brook Experimental Forest (HBEF) is located in central New Hampshire within the White Mountain National Forest. HBEF is approximately 210 km north of Boston, MA, and vehicle access is limited. HBEF soils are mostly well-drained Spodosols with pH levels at or below 4.5 and feature the Berkshire, Skerry, Becket, and Lyman soil series (Hubbard Brook Information Oversight Committee 2001). A unique characteristic of the Spodosol soil order is that the soils horizons are not necessarily parallel to the soil surface, as is the case in most orders (Soil Survey Staff 1999). An important feature of HBEF surface topography is the soil layer movement caused by the uprooting of trees, which creates mixtures of the lower mineral surface soil layers as well as buried horizons where mixture does not occur. This surface disturbance influences weathering and biogeochemical cycles. HBEF is completely forested and consists of 80 to 90 percent hardwoods and 10 to 20 percent conifers. Sugar maple (*Acer saccharum*), beech (*Fagus grandifolia*), and yellow birch (*Betula allegheniensis*) are prevalent at low to mid elevations, while red spruce (*Picea rubens*), balsam fir (*Abies balsamea*), and white birch (*Betula papyrifera* var. *cordifolia*) are predominant at higher elevations. Commercial logging at HBEF ended between 1915 and 1917; the present forest is an even-aged, second-growth forest. (Hubbard Brook Information Oversight Committee 2001). HBEF consists of a number of study watersheds that have been subject to experimental manipulation; Watershed 6 (W6) serves as the reference watershed and has not been experimentally disturbed.

Pb concentration data in forest soils, surface water, and bulk precipitation are available for HBEF; Pb bulk deposition data are also available (see Exhibit 3-16). Ambient air Pb concentration data were not located. The presented data largely focus on W6 and areas proximate to W6, although data from other areas of HBEF are also included.

Exhibit 3-16. Summary of Environmental Data Sources for HBEF

Media	Data Set	Timeframe	Locations	Comments
Atmospheric Inputs	-	-	-	-
Precipitation	Wang et al. (1995)	1993	HBEF; at least 12 samples	Sample location not specified.
Bulk deposition	Johnson et al. (1995)	1976 – 1989	HBEF	Used available data (from W6 and areas immediately adjacent) to construct regression model, which was used to predict Pb inputs
Soil	Yanai et al. (2004)	1976 – 2000	HBEF (Watershed 6, Watershed 1); 100 samples were collected from elevations ranging from 490 to 790 m	See Appendix E
	Wang and Benoit (1996)	1993 – 1994	HBEF (a watershed immediately west of Watershed 6); 24 lysimeters sampled monthly at three elevations (600, 730, and 750 m)	See Appendix E
Surface water (stream)	Wang et al. (1995)	1993	Bear Brook (west of HBEF Watershed 6); samples were collected at elevations ranging from about 400 to 800 m	Sample size not reported.

Yanai et al. (2004) analyzed forest floor[2] soil samples (this data set also includes samples from Watershed 1 [W1]) and reported a 32 percent decline in forest floor Pb concentrations at HBEF from 1976 to 2000. Pb concentrations for this data set ranged from about 38 mg/kg collected in 2000 to 115 mg/kg collected in 1978. Soil profile analysis shows that Pb has become more concentrated at lower depths over time. Johnson et al. (1995) also noted a downward movement of Pb in soil layers. Wang et al. (1995) reported stream dissolved Pb concentrations ranging from about 5 to 15 parts per trillion (ppt) for samples collected in 1993.

For the pilot phase, the ecologically vulnerable case study assessment was limited to identifying the case study location (i.e., HBEF) and identifying associated available media concentration data sets (as described further in Appendix E).

[2] Yanai et al. (2004) define "forest floor" as the O-A soil horizons.

3.6 National-Scale Surface Water and Sediment Screening Assessment

The geographic coverage of the national-scale screening assessment includes locations in freshwater watershed basins from all regions of the United States. The assessment was limited to freshwaters. Monitoring data reviewed by EPA's Office of Research and Development indicated that Pb concentrations in large lakes and oceans are lower than in smaller freshwater bodies (USEPA 2006b). Exhibit 3-17 lists the data sets used in this screening assessment. Exhibit 3-18 lists the 47 basins for which the National Water Quality Assessment (NAWQA) database provided dissolved Pb surface water concentration data. Approximately 50 percent of the land base of the United States is covered by these 47 basins. The basins for which sediment Pb concentration were evaluated are listed in Exhibit 3-19.

Exhibit 3-17. Summary of Environmental Data Sources for National-Scale Surface Water and Sediment Screening Assessment

Media	Data Set	Timeframe	Locations	Comments
Ecological				
Surface water column	National Water Quality Assessment (NAWQA) Database (USGS 2004)	1991 to 2004 (Analysis limited to 1994 through 2004)	47 basin locations across the United States	See Chapters 4 and 7 and Appendix H
Sediment	National Water Quality Assessment (NAWQA) Database (USGS 2004)	1992 to 2000	12 basin locations	See Chapters 4 and 7

Exhibit 3-18. USGS NAWQA Study Units[a] Included in the National-Scale Surface Water Column Screening Assessment

Study Unit/Basin Name (Year Sampling Started)	Study Unit ID No.	Study Unit/Basin Name (year Sampling Started)	Study Unit ID No.
Northeast		**Midwest**	
New England Coastal Basins (1997)	1	Central Nebraska Basins (1991)	35
Connecticut - Housatonic - and Thames River Basins (1991)	2	South Platte River Basin (1991)	40
Hudson River Basin (1991)	3	Upper Colorado River Basin (1994)	44
Long Island and New Jersey Coastal Drainage (1994)	4	Northern Rockies Intermontane Basins (1997)	46
Delaware River Basin (1997)	5	Rio Grande Valley (1991)	45
Allegheny and Monongahela (1994)	9	**Northwest**	
Middle Atlantic		Yellowstone Basin (1997)	43
Lower Susquehanna River Basin (1991)	6	Upper Snake River Basin (1991)	48
Potomac River Basin & Delmarva Peninsula (1991, 1999)	8 7	Central Columbia Plateau & Yakima River Basin (1991, 1999)	50 51
Kananha-New River Basin (1994)	10	Puget Sound Basin (1994)	52
North-central		Williamette Basin (1991)	53
Lake Erie - Lake St. Clair Drainage (1994)	11	**Southwest**	
White River & Great and Little Miami River Basins (1991, 1997)	13 12	Great Salt Lake Basins (1997)	47
Upper Illinois River Basin (1997)	14	Central Arizona Basins (1994)	49
Lower Illinois River Basin (1994)	15	Sacramento River Basin (1994)	54
Western Lake Michigan Drainage (1991)	16	Nevada Basin and Range (1991)	55
Upper Mississippi River Basin (1994)	17	San Joaquin-Tulare Basins (1991)	56
Red River of the North Basin (1991)	18	Santa Ana River Basin (1997)	57
Albemarle-Pamlico Drainage (1991)	19	**Hawaii**	
Tennessee River Basin (upper & lower; 1994, 1997)	20 29	Oahu (1997)	58
Southeastern		**Alaska**	
Santee Basin and Coastal Drainages (1994)	21	Cook Inlet Basin	59
Apalachicola-Chattahoochee-Flint River Basin (1991)	22		
Georgia-Florida Coastal Plain (1991)	23		
Southern Florida (1994)	24		
South-central			
Mobile River and Tributaries (1997)	26		
Mississippi Embayment (1994)	27		
Acadian-Pontchartrain (1997)	28		
Eastern Iowa Basins (1994)	30		
Ozark Plateaus (1991)	31		
Trinity River Basin (1991)	33		
South Central Texas (1994)	34		

[a] See Section 4.5 for a discussion of these Study Units.

Exhibit 3-19. USGS NAWQA Study Units Included in National-Scale Sediment Screening Assessment

Study Unit/Basin Name	Study Unit ID No.
New England Coastal Basins	1
Connecticut - Housatonic - and Thames River Basins	2
Ozark Plateaus	31
Rio Grande Valley	45
Northern Rockies Intermontane Basins	46
Great Salt Lake Basins	47
Oahu	58

There are temporal trends associated with whether dissolved Pb was measured at given station. Most notably, at many stations for which multiple years of data are available, if the first year of sampling indicated all dissolved Pb measurements to be less than the quantitation limit (QL) of 1 µg/L, no further measurements of dissolved Pb were attempted (e.g., dissolved Pb measured in 1999 less than 1 µg/L; no further measurements in 2000 through 2004).

Most dissolved Pb concentrations reported in the NAWQA database are below the QL. There are only 2,558 samples analyzed for dissolved Pb in surface waters for the NAWQA database from 1994 to 2004.[3] Of the 2,558 samples examined, 83 percent (2,116) were below the QL (USEPA 2006b). Of the 442 dissolved Pb concentrations reported in the data set, 17 percent (74 of 442) were estimated.[4]

The NAWQA database had only one measurement of total Pb sediment concentrations for each of the 12 locations evaluated in the sediment portion of the screening assessment. Of these 12 total measurements, seven total Pb concentrations were between 23 and 68 mg/kg; two were at 239 and 240 mg/kg, and three were between 1,620 and 2,900 mg/kg.

[3] Note that EPA/ORD's review of the NAWQA database identified 3,445 samples; however, their review included samples from 1991 forward (USEPA 2006b, Table 7-3).

[4] Of the 74 estimated values, 68 were for concentrations above detection limit of 0.04 µg/L but below the QL of 0.08 µg/L.

4. Estimates of Media Concentrations

This chapter discusses methods, results, limitations, and uncertainties associated with the media concentrations used for exposure assessment in the human exposure and health risk assessments and ecological risk assessment. The context for these media concentrations within the risk assessments is described in Chapter 2 and illustrated in Exhibits 2-2 and 2-3 for the human exposure and health risk assessments and in Exhibits 2-5 and 2-6 for the ecological risk assessment.

4.1 Primary Pb Smelter

The media concentrations for the primary Pb smelter case study were estimated using a combination of modeling approaches and monitoring data. The timeframe of the data used in the study is from 2000 to 2005. The emissions inputs are representative of potential emissions in 2001, and the soil sample data used was collected from 2000 to 2005. Recent air monitoring data indicated that this facility is not in attainment with the current Pb NAAQS; therefore, both a current conditions and an attainment scenario were considered. The inhalation exposure and indoor dust concentrations are different between the two scenarios for the modeled locations where the current conditions air concentrations exceed the current Pb NAAQS. This is because the inhalation exposure and indoor dust concentrations are calculated from the modeled air concentrations. For the pilot phase, the same soil concentrations of Pb were used for both scenarios.

4.1.1 Spatial Template

As discussed in Section 2.2.2, the outer boundary of the study area for the primary Pb smelter case study was set to approximately 10 kilometers (km), which was expected to capture the population experiencing the most significant impacts of the facility's emissions. Modeled air concentrations of Pb from the primary Pb smelter were compared to estimated background Pb concentrations from the Interagency Monitoring of Protected Visual Environments (IMPROVE) database (IMPROVE 2006) to determine at what distance the air concentrations resulting from the facility emissions reached approximately 50 percent of the background air concentration. From this comparison, it was found that a radius greater than 50 km would be required to reach air concentrations that would be 50 percent of background. Due to model run times, this would not be feasible, so the decision was made to model the facility's impacts in a study area that extended 10 km from the facility. This smaller study area is expected to capture the highest exposures associated with the facility's emissions.

A combination of U.S. Census blocks and block groups (U.S. Census Bureau 2005) were used as the spatial units at which to model air concentrations. The 29 block groups that are predominantly within 10 km of the facility were selected to define the spatial extent of the study area. Because of the irregular shape of block groups, not all of the block groups with area within 10 km were included, and some that were included have area outside 10 km. The selection was made to create a coverage area that would include block groups whose average concentration would predominantly come from air within 10 km of the source. All census block centroids within these 29 block groups were included as receptors in the ISC model runs (i.e., air

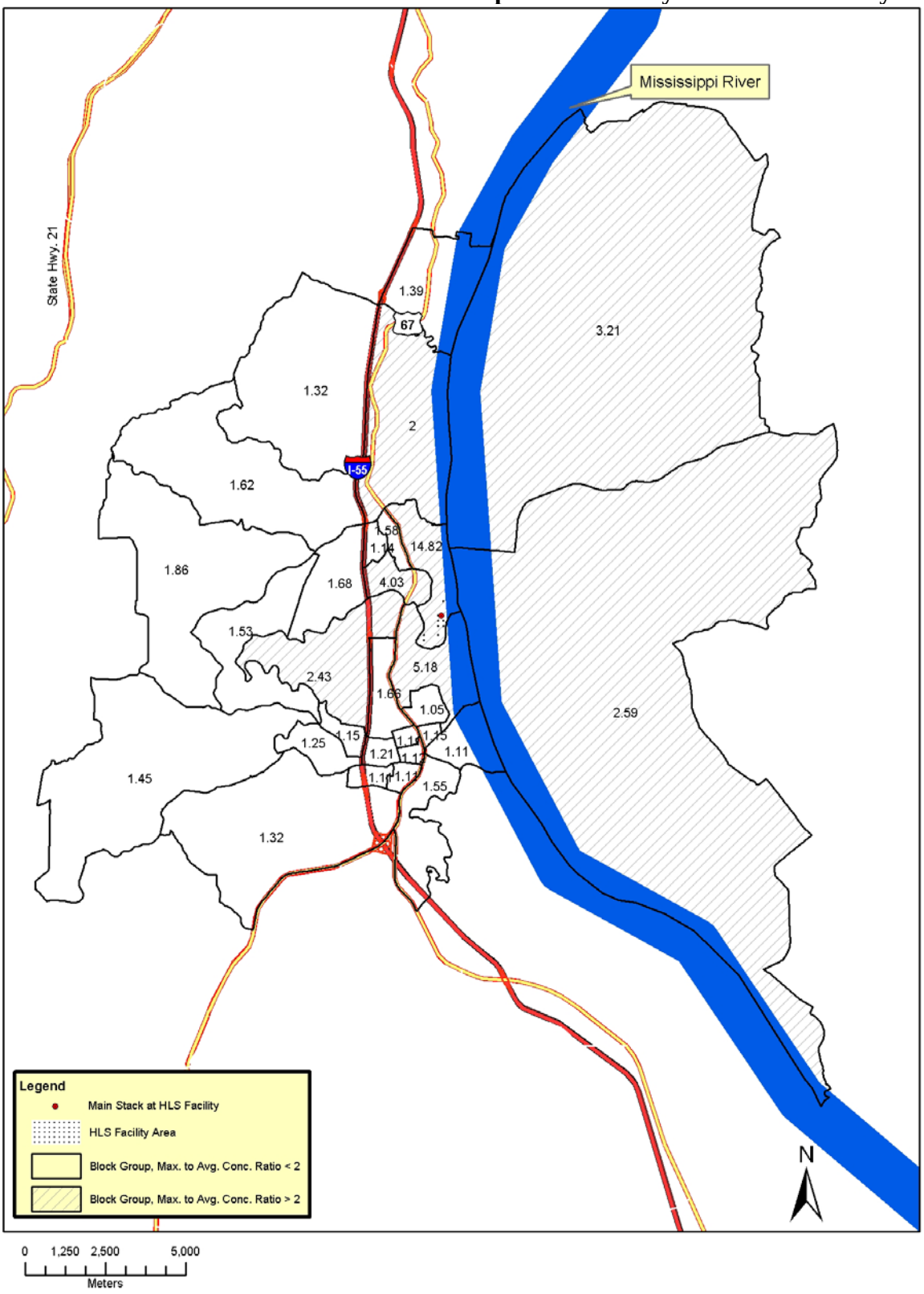
concentrations and deposition fluxes were modeled at each block centroid). There are 1,321 blocks within these block groups. Of these blocks, 14 were identified as falling within facility boundaries, or adjacent to the facility in the Mississippi River.[1] These 14 blocks were removed from the assessment; other zero population blocks within the study area were still included. A total of 1,307 census block centroids were included as receptors in the air dispersion model simulations, including blocks within the study area with zero population. The blocks with zero population were included in the modeling simulations to aid in understanding the patterns of air concentrations in the study area. These locations were not included in the exposure assessment and are not included in exhibits summarizing modeling results (with the exception of isopleths diagrams). The elevation of each block centroid was generated using USGS digital elevation model files and the AERMAP preprocessor model.

In order to limit the locations modeled for the exposure assessment, a combination of block and block group-level results were used for the exposure assessment. The average concentration and deposition in each block group was calculated by spatially-weighting the values at the block level. The area of each block was obtained from the U.S. Census (Census Bureau 2005). The decision of whether to use the block or block group was made by considering the range of block-level concentrations within a block group. If the ratio of the maximum block-level air concentration in the block group to the average air concentration in the block group was greater than 2.0, the blocks in the block group were modeled individually. All other block groups were modeled as units. This method generally resulted in blocks near the facility, as well as some blocks far from the facility that fall within very large block groups, being modeled individually (see Exhibit 4-1).

Six census blocks near the facility for which there are non-zero population counts for the year 2000 were removed from the analysis because current information indicates that as of 2004 they have zero population (Gradient Corporation 2004). A total of 22 census block groups and 201 census blocks (all with non-zero population counts) in other block groups were included in the exposure assessment.

[1] All territory in the United States is delineated into Census blocks (U.S. Census Bureau 2005). Therefore, large water bodies like the Mississippi River often contain Census blocks, although there is no population associated with these blocks.

Exhibit 4-1. Modeled Block and Block Groups near Primary Pb Smelter Facility



4.1.2 Air

The air concentrations and deposition of Pb resulting from emissions at the primary Pb smelter facility were estimated using the ISC-PRIME (Schulman et al. 1997; USEPA 1995) air dispersion model and compared to the air concentration and deposition measurements from nearby monitors. The emissions used for the air quality modeling are described in Section 3.1.4. The air concentration and deposition monitoring data used for comparison are described in Section 3.1.5.

4.1.2.1 Study Area and Air Dispersion Modeling

The air dispersion model ISC-PRIME was used for the primary Pb smelter air quality modeling. The meteorological data used for the model simulations included 25 consecutive months (April 1, 1997 to April 30, 1999) of on-site data. These meteorological data were also used for an analysis by U.S. EPA Region 7 (2006b) of the facility. Meteorological data are currently being collected at the facility, but problems due to data gaps prevented their use in this analysis (USEPA Region 7 2006b). As described in Section 3.1.4, emissions, release parameters, particle size parameters, and building downwash inputs were all provided by Region 7 (in the form of an input runstream file). The emissions and source parameters represent the 2000 Revision of the SIP developed for the facility (MDNR 2000).

Hourly air concentrations and deposition were output from the dispersion model at each receptor (i.e., block or block group, as described in Section 4.1.1) and monitor location. The hourly air concentration and deposition model results were averaged over the modeling time period to obtain annual average air concentration and annual average deposition for the current conditions scenario. The concentrations were also averaged quarterly to determine whether any of the current conditions scenario's air quarterly average concentrations were greater than the current Pb NAAQS. Modeled Pb air concentrations in some blocks exceeded the NAAQS for at least one quarter. For the current attainment scenario, these concentrations were set to the current NAAQS value (1.5 $\mu g/m^3$), assuming attainment, and the annual averages were then recalculated using the revised quarterly averages. All other modeled concentrations for the current attainment scenario were set to the same values as the current conditions scenario.[2]

4.1.2.2 Inhalation Exposure Concentrations

Inhalation exposure concentrations of Pb were estimated for the population of interest (young children) from the estimated ambient air concentrations using age group- and location-specific relationships for Pb developed from modeling performed for U.S. EPA's 1999 National-Scale Air Toxics Assessment (USEPA 2006e), one of U.S. EPA's National Air Toxics Assessment (NATA) activities. These relationships account for air concentration differences indoors and outdoors and mobility or time spent in different locations (e.g., outdoors at home, inside at home, etc.) for the population of interest.

[2] As described in Section 2.2, this method is recognized to be a simplification of how air concentrations would respond.

The NATA national-scale assessment produced air concentrations of Pb (and other hazardous air pollutants) for each U.S. Census tract (using the Assessment System for Population Exposure Nationwide model, ASPEN), and corresponding exposure concentrations of Pb for each of five age-groups at each U.S. Census tract. The relationships (or ratios) between ambient Pb concentration and Pb exposure concentration from the NATA national-scale assessment for the 0 to 4 age group (the closest age group for which outputs are available to the age group of interest for this assessment) ranged from 0.37 to 0.42 for the U.S. Census tracts within the study area for the primary Pb smelter location. The ratios are presented in Appendix F, Exhibit F-1.

Use of ratios for the 0 to 4 age group (rather than for 0 to 7) contributes some uncertainty in the estimate of children inhalation exposure concentrations. In addition, there is some uncertainty in the magnitude of the air concentrations generated using the ASPEN model for the NATA assessment. In a comparison to monitoring data across the country, the ASPEN modeled air concentrations generally underestimate monitored concentrations (see section on Comparison to Monitored Values, USEPA 2006e). However, the relationship between ambient concentrations and exposure concentrations (which is the comparison used here) is not expected to be affected by underestimated ambient concentrations from the NATA study.

4.1.2.3 Air and Deposition Results

Annual average Pb air concentrations and deposition fluxes were calculated at the centroid of each census block group, block, and monitor receptor point for the two years of meteorological data. To take into account variations in meteorological data, the annual average concentrations and depositions for each of the two years were averaged to generate one set of representative annual average concentration and deposition results for the current conditions scenario. For the attainment scenario, concentration estimates were adjusted at the eight census blocks at which one or more quarterly average air concentrations exceeded the current NAAQS. At these blocks any quarterly averages that exceeded the NAAQS were set equal to the NAAQS and the annual average was calculated from the quarterly averages. All other block and block groups for which a quarterly average air concentration did not exceed the current NAAQS were kept at the modeled values for the attainment scenario. Using this method, it is likely that the concentrations at blocks that did not exceed the current NAAQS are slightly overestimated for the attainment scenario. However, because there may be multiple methods that the facility would use to obtain attainment (i.e., different controls on different processes), the exact pattern of concentration reductions could vary depending upon which location in the facility has reduced emissions. Thus, it is likely to underestimate reductions in ambient air concentrations for the current NAAQS attainment scenario relative to the current conditions scenario.

All concentration, deposition, and inhalation exposure estimates for the census blocks and block groups modeled are presented in Appendix F, Exhibits F-2 and F-3, for both scenarios. Exhibit 4-2 presents the range of air concentration, inhalation exposure, and deposition values generated from the modeling for both the current conditions and current attainment scenarios. The values are summarized from the model results at the 223 blocks and block groups with non-zero population selected for further analysis in the exposure assessment. Exhibit 4-3 shows the isopleths of the block-level modeled air concentration results.

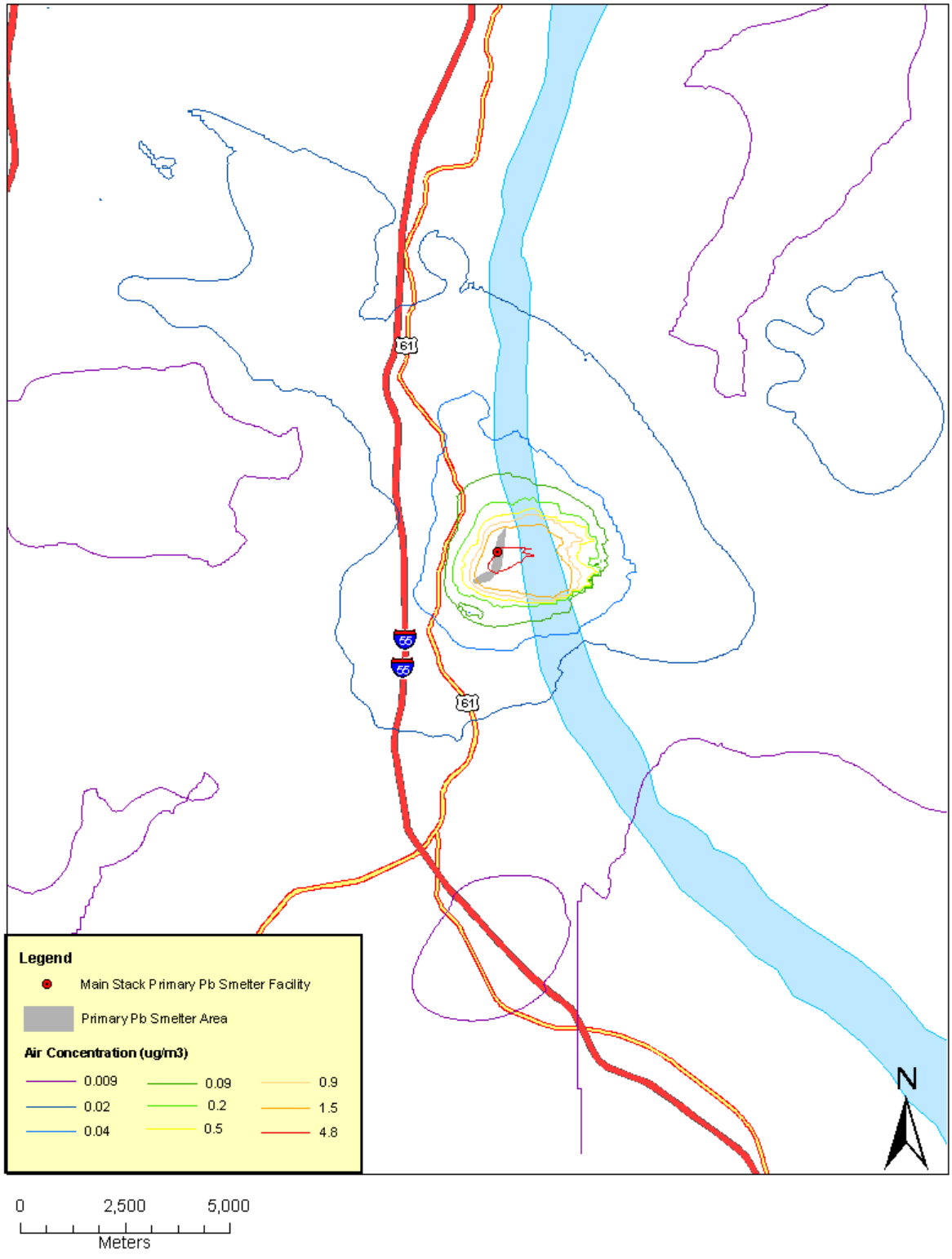
Exhibit 4-2. Primary Pb Smelter Case Study: Modeled Air Concentrations and Deposition [a]

	Current Conditions		Current Attainment		
Statistic[b]	**Average Annual Pb Air Concentration ($\mu g/m^3$)**	**Children 0 to 4 Average Annual Pb Air Exposure Concentration ($\mu g/m^3$)**	**Average Annual Pb Air Concentration ($\mu g/m^3$)**	**Children 0 to 4 Average Annual Pb Air Exposure Concentration ($\mu g/m^3$)**	**Dry Deposition (g/m^2/year)**
Maximum	2.73	1.14	1.50	0.628	5.20
95th Percentile	0.662	0.277	0.662	0.277	1.28
Median	0.0221	0.00895	0.0221	0.00895	0.0938
5th Percentile	0.00845	0.00329	0.00845	0.00329	0.0266
Minimum	0.00541	0.00210	0.00541	0.00210	0.0163

[a] The 223 blocks and block groups with non-zero population selected for analysis were used to create this summary. Note that in some of these blocks the 2000 U.S. Census indicates there are no children.

[b] The statistic (e.g., 95th percentile, median) may not be at the same location for each of the data results presented here.

Exhibit 4-3. Primary Pb Smelter Case Study: Concentration and Deposition Isopleths



4.1.2.4 Air Modeling Performance Assessment

As described in Section 3.1.5, nine air monitors from AQS/AirData (USEPA 2006f) and four EPA high-volume air monitoring stations that are not in the AQS/AirData database (USEPA Region 7 2006a) were identified within 10 km of the primary Pb smelter facility. Pb concentrations measured at all of these monitors are for Pb in total suspended particulates (TSP). Modeled air concentrations were generated from ISC-PRIME at each of these locations and are presented in Exhibit 4-4. There is no specific year of monitoring data that can be directly correlated with the emissions characterization because the emissions represent those from the 2000 Revision of the SIP developed for the facility, rather than a specific year's emissions (MDNR 2000). Therefore, the most recent 5 years of monitoring data were compared to the modeled air concentrations. The range of the ratios of monitored to modeled concentrations is presented for each monitor location in Exhibit 4-4. A more detailed comparison is presented in Appendix F, in Exhibit F-4.

As presented in Exhibit 4-4, it varies whether the model or monitor Pb air concentration is higher at each monitoring point. It appears that roughly with increasing distance from the facility, the ratio of monitored-to-modeled concentrations increases. This suggests that the model may be overestimating concentrations at close distances, and underestimating concentrations with increasing distance. This may be related to deposition parameters, emissions estimates at certain locations at the facility (the facility spans approximately 1 km), or meteorological data.

Exhibit 4-5 presents a comparison of the deposition modeled at 10 locations for which dry deposition measurements are available (USEPA Region 7 2006a). Measured deposition was available for 2003-04 at two heights (1 foot and 10 feet). The modeled deposition fluxes at those same locations are compared to the all measured values and a range of the ratio of monitored to modeled deposition is presented in Exhibit 4-5. A more detailed comparison table is provided in Appendix F, Exhibit F-5. For most locations, the predicted dry deposition is less than the measured dry deposition.

A wind rose created from the on-site meteorological data at the primary Pb smelter (Exhibit 4-6) shows that the predominant wind directions are to the east and north. The air monitors and facility are mapped in Exhibit 4-7 to show the relative locations. The shaded area on the map indicates the area with modeled emission release points, and the red circle is the main stack at the facility. Because the Mississippi River forms a natural boundary on the eastern side of the facility, the air monitors are all to the north, west, and south. There are no close-by air monitors to the east, in the predominant wind direction (the closest monitor in an eastern direction is more than 40 km to the northeast). However, because of the width of the river, the monitor network is expected to capture the high-end human inhalation exposure values from the facility emissions. In addition, the land on the eastern side of the river is farmland, and not as densely populated as the area with the monitors (see Exhibit 3-2). The highest monitored values are at monitors 102, 290990015, 290990004, and 290990005, which are all north (and slightly west) of the facility. This corresponds with what may be expected from the wind rose. Of the modeling results, one of the highest concentrations was modeled at monitor 101 to the southwest of the main stack. This likely is a result of the wide spread of emission sources modeled. Monitor 101 is located amongst modeled emission sources, resulting in the high modeled concentration. Overall, the

modeled concentrations and depositions are fairly representative of conditions over the last 5 years based on comparison with the monitors.

Exhibit 4-4. Primary Pb Smelter Case Study Air Concentration Modeled Results Compared to Monitor Values

Pilot Phase Analysis ID or Monitor ID	Distance to Main Stack (km)[a]	2-year Average Modeled Air Conc ($\mu g/m^3$)	Range of Monitored to Modeled Ratio for 2001 to 2005[b]
U.S. EPA Region 7 Database HiVols Values[c]			
100	0.39	1.06	0.7 to 1.2 (n = 3)
101	0.45	4.04	0.1 to 0.3 (n = 2)
102	0.53	2.17	0.4 to 2.8 (n = 3)
103	0.73	0.186	2.1 to 5.4 (n = 3)
AirData Monitored Values[d]			
290990015	0.29	1.71	0.8 to 1.9 (n = 5)
290990004	0.34	2.46	0.4 to 0.5 (n = 2)
290990005	0.78	0.463	0.6 to 4.6 (n = 5)
290990011	0.89	0.323	0.9 to 4.7 (n = 5)
290990016	0.92	0.151	1.3 to 2.0 (n = 2)
290990013	1.5	0.135	1.4 to 6.7 (n = 5)
290990008	1.5	0.044	1.5 to 6.2 (n = 5)
290990010	2.2	0.043	0.8 to 1.8 (n = 5)
290990009	2.7	0.048	1.0 to 7.0 (n = 5)

[a] The facility spans roughly 0.2 by 1 km. Therefore, some monitors will be closer to other emission release points than they are to the main stack.

[b] The number indicated in parentheses after each range indicates how many annual average monitored concentrations are compared to the modeled results (i.e., 2, 3, or 5 annual average values are available from the monitor).

[c] The Pb air concentration detections (which appear to be TSP Pb) reported in the HiVols data set of the U.S. EPA Region 7 database were averaged for each year (see Exhibit A-2). All non-detects were assigned half of the detection limit for averaging. Averages for 2001 are from measurements taken between October and December; averages for 2002 are from measurements taken all year; and averages for 2003 are from measurements taken between January and July.

[d] AirData values are for TSP Pb and are averages obtained from the AirData website monitor reports. In some cases, there is more than one monitor at a site.

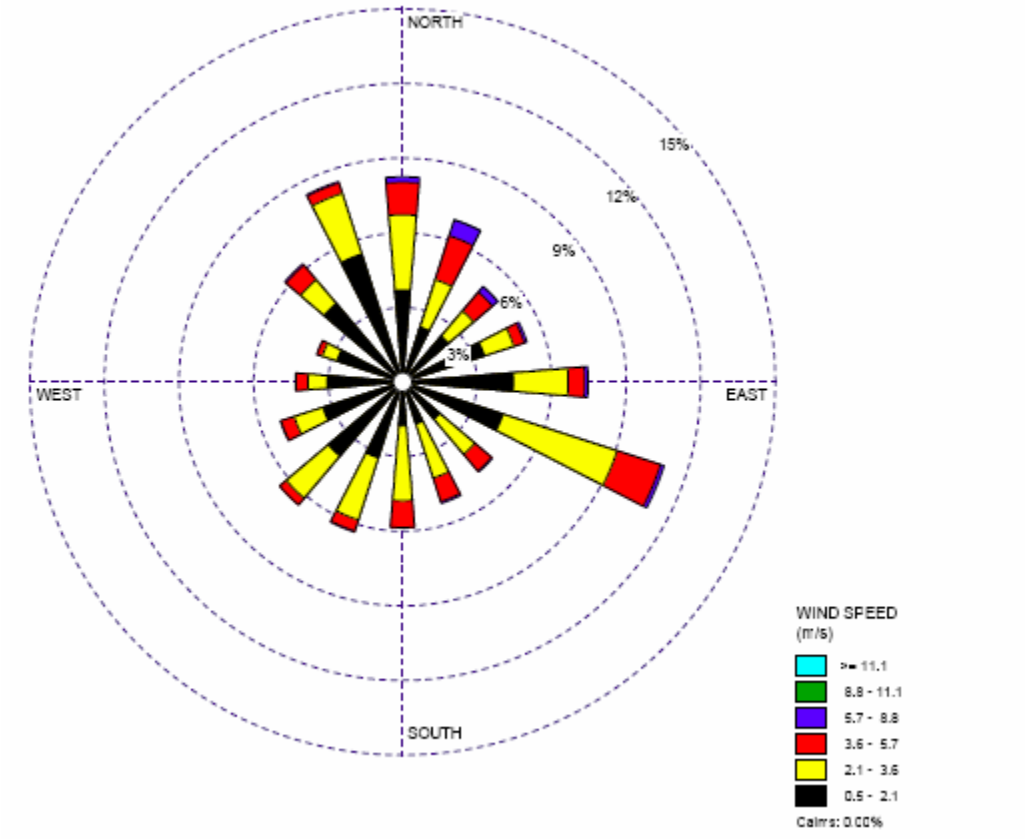
Exhibit 4-5. Primary Pb Smelter Case Study Deposition Modeled Results Compared to Measured Values

Pilot Phase Analysis ID	Distance to Main Stack (km)[a]	Modeled Average Pb Deposition ($g/m^2/yr$)	Range of Monitored to Modeled Deposition Ratio for 2003-04[b]
1071	0.38	2.62	0.53 to 0.65
181	0.49	1.23	0.71 to 0.79
286	0.56	2.55	0.30 to 0.80
1073	0.70	0.45	1.44 to 1.71
444	0.71	1.42	0.24 to 0.54
240	0.76	0.37	0.82 to 1.30
207	0.85	0.65	0.40 to 0.83
576	1.0	0.53	0.31 to 0.69
531	1.3	0.35	0.31 to 0.87
1072	1.9	0.21	0.26 to 1.04

[a] The facility spans roughly 0.2 by 1 km. Therefore, some monitors will be closer to other emission release points than they are to the main stack.

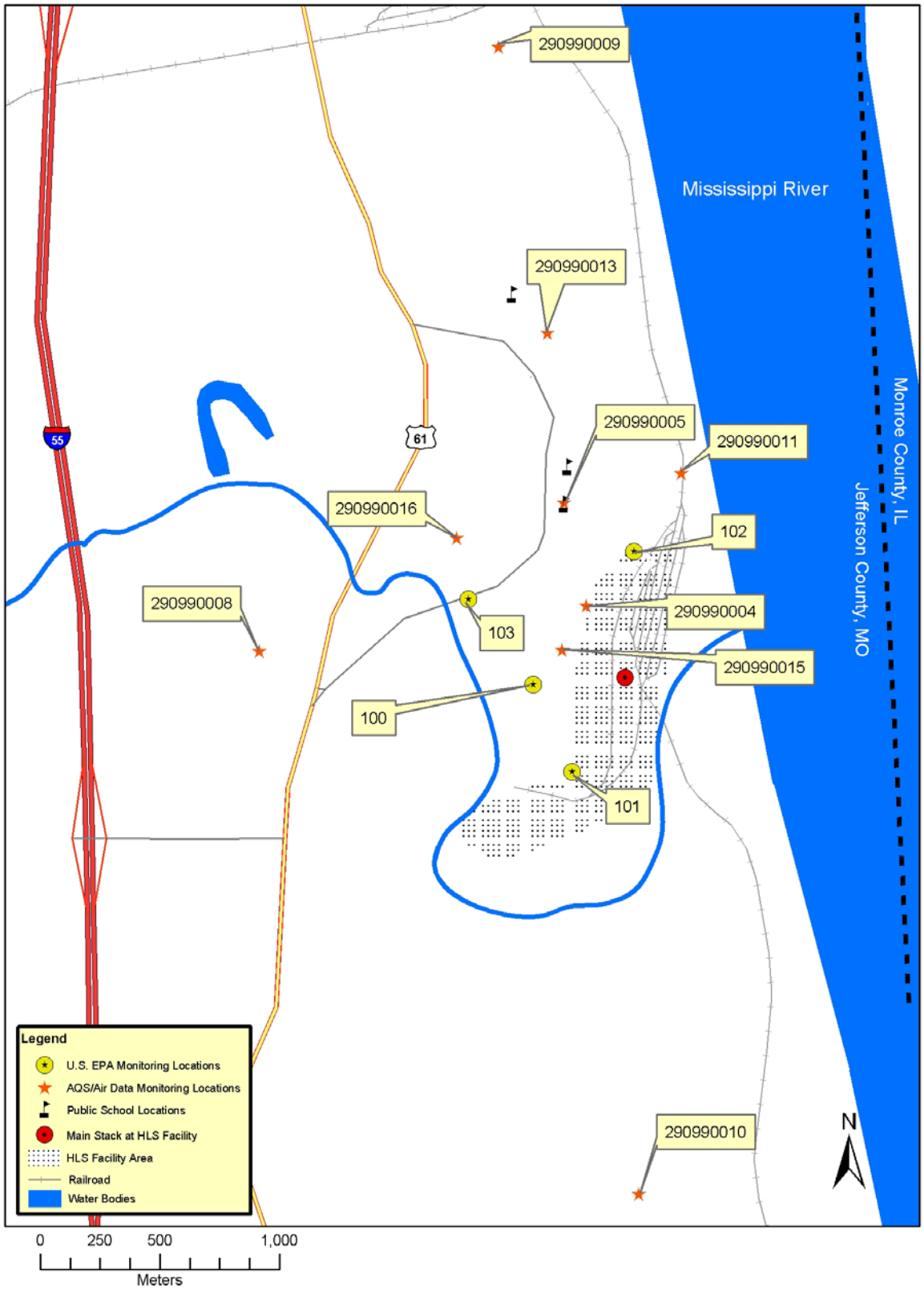
[b] For each set of ratios, four sets of averaged deposition values were compared – deposition measurements at 1 ft and 10 ft in 2003 and 2004. See Exhibit A-11 for individual deposition measurements.

Exhibit 4-6. Wind Rose of Meteorological Data used for Primary Pb Smelter Case Study (Direction in which Wind is Blowing)[a]



[a] Wind rose for 18,240 hours of data

Exhibit 4-7. Air Monitor Locations near Primary Pb Smelter



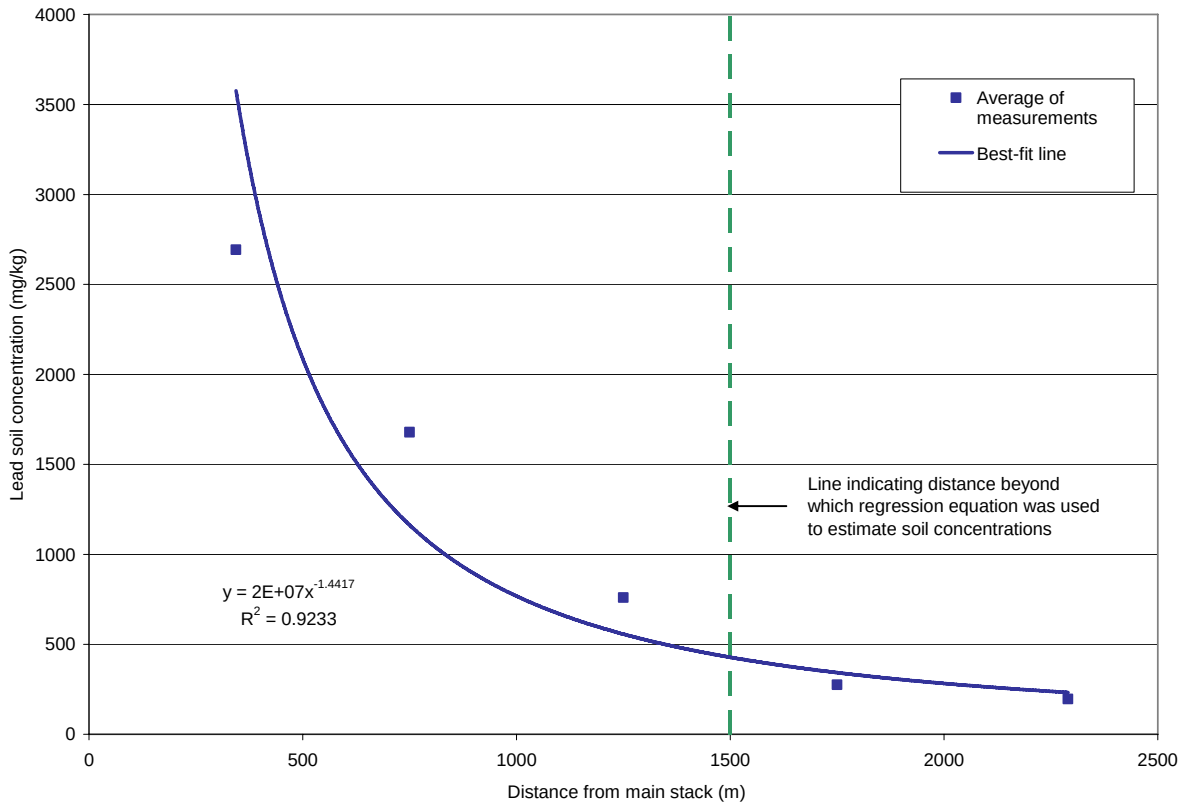
4.1.3 Outdoor Surface Soil

For the human exposure and health risk assessments, outdoor surface soil concentrations were estimated for each spatial unit (i.e., census blocks and block groups) in the study area from the soil sample measurements in the area. The extent and types of soil data sets used for the calculations are described in Section 3.1.5. Many of the yards within 1.5 km of the primary Pb smelter facility have been excavated and filled with clean soil in the last 10 years. EPA has taken soil samples from 31 of these sites since 2002. These measurements are called "recontamination" samples. The EPA database also contains soil samples for more than 900 locations labeled as "pre-excavation." These samples were taken from November 2000 to August 2004. Not all of the locations with pre-excavation samples have been filled with clean soil, only select locations within 1.5 km of the facility. Documentation available for the "recontamination" soil samples indicates that samples were taken to a depth of less than an inch (EPA 2001), and it is likely the "pre-excavation" samples were taken the same way. Depending on the location of the modeled block or block group in this study (within or outside of the soil cleanup area), the soil concentrations for this assessment were calculated using either the recontamination or pre-excavation data set.

All census blocks within the soil cleanup area (approximately 1.5 km) were identified from the Gradient Corporation report (2004). For these 28 blocks with population, soil concentrations were estimated from the recontamination soil samples taken in 2005. For blocks for which there were one or more soil measurements available, the block soil concentration was set to the average (arithmetic mean) of those measurements. For blocks for which there were no measurements, but for which there were nearby measurements (i.e., across the street), the soil concentration was set to the nearby measurement average. For other blocks, the average of all of the recontamination soil measurements within 500 m was calculated and set as the value for the block.

Outside the soil cleanup area, soil concentrations were estimated using a regression equation of the pre-excavation soil concentrations. The distance of each pre-excavation soil sample to the main stack was measured using a geographical information system (GIS). The measurements were grouped according to distance from the main stack, with separate groups for each 500 meter (m) increment. The arithmetic mean for each group was calculated, resulting in five arithmetic mean average values for soil concentration which were plotted versus distance. A regression power equation (R^2 of 0.92) was calculated from the samples (see Exhibit 4-8). Note that pre-excavation soil samples taken within 1.5 km of the facility were included to develop the regression equation; however, the equation was not used to estimate soil concentrations at blocks within the 1.5 km soil clean-up area (as indicated in Exhibit 4-8). The distance of each census block and block group centroid from the main stack was measured in GIS. Soil concentrations for the blocks and block groups outside the soil cleanup area were then calculated using the regression equations based on distance from the stack.

Exhibit 4-8. Average Pre-excavation Soil Measurements and Best-fit Trend Line



All calculated soil concentrations at the block and block group locations are summarized in Appendix F, Exhibit F-2 and F-3, with an indication of which method was used to calculate the values. A summary of the soil concentrations is presented in Exhibit 4-9. Note that because of the soil cleanup within 1.5 km of the stack, the calculated soil concentrations, as well as the measured concentrations, near the facility are in many cases lower than the soil concentrations calculated or measured in locations without soil cleanup. Soil concentrations throughout the modeled region range from 16 to 976 mg/kg.

Exhibit 4-9. Summary of Primary Pb Smelter Case Study Soil Concentrations Extrapolated from Measurements

Statistic	Calculated Average Soil Concentration (mg/kg)	Distance from Main Stack (m)[a]
Maximum	976	980
95th Percentile	426	1,700
Median	84.0	5,400
5th Percentile	23.6	13,000 [a]
Minimum	15.9	17,000 [a]

[a] Receptors at blocks greater than 10 km from the facility were included because of the irregular shape of block groups (see Section 4.1.1 and Exhibit 4-1).

For the ecological risk screening assessment, surface soil concentration data were extracted from data sets provided by ELM (2005) in its *Characterization Area Investigation Report* for the

primary Pb smelter facility in Herculaneum, Missouri. Sample results were aggregated into geographic clusters to average media concentrations that will be used in the ecological risk characterization. To ensure that data reflect contributions primarily via air deposition, the sample locations were reviewed to exclude samples designed to assess contributions from other pathways (e.g., surface drainage pathways and groundwater flow pathways). The samples on the central, east, and west transects of the smelter's slag storage area (SSA) were excluded from this assessment because the ELM (2005) report states that the floodwaters of Joachim Creek have been visually observed to come into contact with the SSA, thereby depositing slag materials into the floodplains as the water flow subsides. The soil measurements for the west and southern slope of the SSA were also excluded because the sampling transects were located close to water seeps from the SSA.

Soil samples farthest away from the SSA, where the possibility of direct contact with Pb from the SSA during flooding events was the lowest, were chosen in this analysis. This set of sampling sites was along the historic railroad grade on the west side of Joachim Creek, where each site was at a progressively greater distance from the SSA. Also included in this analysis were samples from the two "reference" areas. ELM collected samples from approximately 5.6 km and 6.5 km south of the facility, respectively, to measure "background" concentrations of Pb in the soil and to establish a baseline for comparison to the measurements closer to the facility. The closer reference area was south of Crystal City and the farther reference area was south of the Festus Memorial Airport. ELM chose these areas because they have a similar habitat to areas surrounding the facility in terms of past and current land use and regional geology. There are no known industrial sources of Pb near these references sites.

To develop soil concentration estimates for the ecological risk screening assessment, the surface soil data was grouped into three geographic clusters. Cluster 1 comprised of ten samples taken from the west side of Joachim Creek (approximately 1 km west of the facility); Cluster 2 contained six samples taken from Crystal City (approximately 5.6 km south of the facility); and Cluster 3 contained ten samples taken from near the Festus Memorial Airport (approximately 6.5 km south of the facility). Soil samples were taken from depths of 0 to 3 inches (ELM 2003). The soil sampling results for the three soil sample clusters are summarized in Exhibit 4-10 and presented in more detail in Appendix F.

Exhibit 4-10. Soil Sample Clusters Used to Estimate Ecological Receptor Exposure via Soil and Corresponding Pb Analysis Results (ELM 2005)

Sample Cluster	Mean Measured Pb Conc. (mg/kg)	Standard Deviation of Measured Pb Conc.	Mean Detection Limit (mg/kg)
Sample Cluster 1: soil samples from West of Joachim Creek	425	45.7	0.35
Sample Cluster 2: soil samples from Crystal City locations (south of facility)	64.7	10.0	0.41
Sample Cluster 3: soil samples from 'Near Festers Airport' locations (south of facility)	48.5	14.4	0.39

4.1.4 Indoor Dust

For estimating indoor dust concentrations for residences at the primary Pb smelting facility, two dust prediction models were used:

- For locations within 1.5 km of the facility: a site-specific regression model; and
- For receptors more than 1.5 km away from the facility: a pooled analysis model (referred to as the "AGG" or "aggregate" model) identified from the literature which predicts Pb dust concentrations given soil and ambient air Pb levels based on data from a variety of industrial and urban and industrial studies (USEPA 1989).

The site-specific model is based on data collected within the residential remediation zone characterizing yard soil Pb levels (post-remediation) and indoor dust levels. The AGG, or non-site-specific model, was selected for zones outside of the remediation area because available soil and indoor dust data did not extend to these more distant areas and the site-specific model derived for the remediated zone was deemed not representative for the non-remediated zone.

The site-specific regression model was developed for predicting indoor dust levels within the remediation zone immediately adjacent to the primary Pb smelter. The data set used to "fit" the model was based on indoor dust samples (dependent variable) collected for 17 houses within the remediation zone. Independent variables included:

- Modeled annual-average air Pb concentrations from census block centroids located within 200 m of each of the 17 houses;

- Road dust measurements located within 300 m of each house[3]; and

- Post-remediation residential soil Pb measurements collected from the yard of each house.

The pre-remediation soil Pb concentrations collected during 2001 were not included in regression modeling because they might not be representative of current conditions at the site (i.e., it was felt that soil samples collected post-remediation would be more representative of current conditions given the level of remediation within this zone and ongoing recontamination which was occurring). Because the focus is on predicting indoor dust Pb concentrations reflecting longer-term trends, multiple samples taken over time for a given medium at a specific house were averaged to produce a "temporally-averaged" value reflecting longer-term trends in Pb concentrations. For a more detailed summary of the regression analysis see Appendix F.

Multiple regression models were tested for this analysis, which are listed in Exhibit F-17 and F-18 of Appendix F. During the testing, the road dust samples were found to not have significant predictive power for indoor dust Pb. This may reflect the fact that the indoor dust sample data set does not provide significant coverage for homes located near to the truck haul routes, resulting in a poor correlation between road dust levels and indoor dust levels. Yard soil samples

[3] Road dust measurements were compiled by EPA Region 7 (2006a) from 2002 to 2006 at 21 locations ranging from 160 to 1,700 m from the main stack at the facility.

were found to be slightly (but statistically significantly) negatively correlated with indoor dust levels. This counter-intuitive finding may reflect the process of yard remediation which has likely resulted in a patchwork of remediated yards with varying degrees of recontamination by air-deposited Pb (i.e., remediation activity has interfered with any correlation between yard soil Pb levels, ambient air Pb levels, and indoor dust Pb levels that might have existed prior to remediation efforts). Because of this negative correlation, the relationship to soil Pb levels was excluded from the selected regression model, resulting in a model that only uses air concentrations as input but has a fairly large intercept, which likely reflects a number of factors not correlated with ambient air or distance from the facility, including generalized soil Pb contamination in the area as well as contributions of indoor Pb paint to indoor dust.

Although multiple regression models were tested, ultimately, the H6 model was selected for use in the pilot phase because it has one of the highest goodness-of-fit measures (R^2 is 0.701), it tracks non-linearities of the data even at low levels, and fits longer-term trends better than other models. The H6 indoor dust model presented as follows relates the natural log of indoor house dust to the natural log of ambient air Pb:

$$\ln(Pb_{dust}) = 8.3884 + 0.73639 \times \ln(Pb_{air})$$

$$CCC = \exp\{1.273\ [\ln(hardness)] - 4.705\} \times CF$$
$$CMC = \exp\{1.273\ [\ln(hardness)] - 1.460\} \times CF$$

where:

CF = conversion factor = 1.46203 -[ln(*hardness*) × 0.145712] *hardness* is expressed as mg/L of calcium carbonate ($CaCO_3$)
Pb_{dust} = concentration of Pb in indoor dust (mg/kg)
Pb_{air} = concentration of Pb modeled in the ambient air (µg/m3).

For the remainder of the study area, the pooled analysis AGG model based on data collected during the 1970s and 1980s at a number of active primary Pb smelters, including the primary smelter analyzed here, was used (USEPA 1989). The AGG model predicts indoor dust Pb based on both outdoor soil and ambient air Pb levels. The model is appropriate for the non-remediation portion of the primary Pb smelter case study area because this area has not been subjected to extensive remediation and is therefore likely to resemble the locations included in the pooled analysis used in deriving this model (i.e., areas not having undergone extensive yard soil remediation). Furthermore, because the non-remediation portion of the study area is likely to have soil Pb gradients reflecting long-term atmospheric deposition of Pb, it is likely that indoor dust would be partially dependent on soil Pb. Therefore, the AGG model presented here (including both the soil and air factors) was selected for this portion of the study area:

$$Pb_{dust} = 31.3 + (638 \times Pb_{air}) + (0.364 \times Pb_{soil})$$

where:

Pb_{soil} = concentration of Pb in outdoor soil (mg/kg).

Exhibit 4-11 presents a summary of the Pb indoor dust concentrations generated in the primary Pb smelting study area for the current conditions and current attainment scenarios. Indoor dust concentrations were only estimated for census blocks with children. Exhibit 4-11 shows the number of census block or block groups and children modeled with increasing levels of Pb house dust. All estimated indoor dust Pb concentrations for residences in the primary Pb smelter case study are presented in Appendix F, Exhibit F-14.

Exhibit 4-11. Primary Pb Smelter Case Study: Modeled Indoor Dust Pb Concentrations [a,b]

House dust concentration (μg/g)	Current Conditions		Current Attainment	
	Census blocks/ block groups with dust concentrations greater than value	Children living in area with dust concentration greater than value	Census blocks/ block groups with dust concentrations greater than value	Children living in area with dust concentration greater than value
30	137	3,880	137	3,880
50	61	970	61	970
100	26	104	26	104
500	23	97	23	97
1,000	14	47	14	47
3,000	5	9	4	8
5,000	2	6	0	0

[a] The 137 blocks and block groups with children aged 0 to 7 in the 2000 U.S. Census (U.S. Census Bureau 2005) were used to develop this summary. Note that Exhibits 4-2 used all blocks with population, even blocks without children.

[b] Number of children ages 0 to 7 from U.S. Census 2000 were used in this analysis (U.S. Census Bureau 2005).

Studies summarized in the 1990 review of the Pb NAAQS contained measurements of house dust ranging from 10 to 35,000 ppm. A high value of 100,000 ppm was measured in one home within 2 km of a Pb smelting facility (USEPA 1989). The Pb indoor dust concentrations for this primary Pb smelter case study fall within the range presented by U.S. EPA (1989), but they are not in the high-end of the range. Within a distance of 2 km from the modeled primary Pb smelting facility, the maximum dust concentration at a receptor location is 5,300 parts per million (ppm) at 300 m from the main stack. In a study of Pb concentrations in household dust near a facility that has operated as a secondary Pb smelter since 1972 and as a primary smelter for the previous 200 years in the Czech Republic, Rieuwerts et al. (1999) measured household dust Pb concentrations in houses in neighborhood adjacent to the facility (the neighborhood ranges from approximately 0 to 500 m away from the facility according to a figure). Measured Pb concentrations in household dust from 14 homes ranged from 861 to 5890 ppm with a geometric mean of 1668 ppm. Indoor dust concentrations in the primary Pb smelter pilot phase assessment are similar, ranging from 1,500 to 5,300 ppm out to 500 m from the facility, with a geometric mean of 3,100 ppm.

4.1.5 Surface Water and Sediment

Measurements of sediment and surface water Pb concentrations were obtained from ELM's *Characterization Area Investigation Report* (2005). Sediment samples chosen for this assessment were co-located with surface water samples at all locations. ELM sampled Joachim

Creek, the Mississippi River, a U-shaped ephemeral pond approximately 1.7 km northwest of the primary Pb smelter facility, and drainage areas in the watershed.

4.1.5.1 Surface Water

Results of all surface water samples collected from the Joachim Creek, Mississippi River, and the nearby pond and backwater flow areas showed that levels of dissolved Pb were below the detection limit for Pb of 3.0 μg/L (note that the detection limit for dissolved Pb in surface waters for the U.S. Geological Survey since 2001 is 1.0 μg/L). The surface water sample data analyzed by ELM (2005) was reviewed and aggregated, and the results for sampling locations used in this assessment are presented in Appendix F.

ELM (2005) collected surface water samples from four backwater and low flow areas within a 1.6 km radius to characterize impacts of aerial deposition of historical smelter emissions. ELM collected samples from two locations on Joachim Creek and its feeder streams (CHRDDP and DAMUP), based on the assumption that even though most of the direct deposition of particles from the emissions would be in the direction of the prevailing winds, minor gusts and winds along with aerodynamic effects may take the contaminants in any direction. Additional samples were taken from the unnamed pond north of Joachim Creek (UPOND) and from a slow flow area along the western shore of the Mississippi downstream of the facility (RRDP).

The other ELM (2005) surface water samples were reviewed and the ones that indicated likely contributions of Pb from sources other than air deposition because they bordered the storage slag area (SSA) were excluded. Sampling sites were chosen from among those taken along Joachim Creek to represent sections of the creek approximately 800 m, 1.6 km, and 3.2 km to the west and southwest of the primary Pb smelter based on aerial maps showing the sampling locations. Transects were established by ELM in the area of point bars, backwaters, and low flow areas for sediment sampling as well as locations perpendicular to stream flow at each sampling location. ELM collected three surface water samples at each transect, one at the thalweg (i.e., the deepest water area of the creek, generally in the middle) and the other two equidistant between the thalweg and the banks of the Joachim Creek. The water samples were collected in the center of the vertical water column (i.e., equidistant from the water surface and the top of the sediment) at each of the three locations of each transect.

The sampling locations identified in the Mississippi River were approximately 2.7 km upstream from the facility, in the immediate vicinity of the facility, and approximately 2.5 km downstream from the facility (data for sampling stations further downstream on the Mississippi River were not reported). ELM chose the farther downstream locations to assess the potential impact on the Mississippi River surface waters from the facility's SSA downstream, as well as deposition of smelter emissions.

Data were aggregated for the ecological risk screening assessment as follows:

- **Joachim Creek.** Samples were clustered geographically and then averaged. Cluster 1 corresponds to the set collected farthest upstream from the facility, while Cluster 5 was nearest upstream from the facility, and the remaining clusters fill the range between.

Results are provided in Exhibit F-7. Although total Pb concentrations were typically above the detection limit of 3 µg/L, with values up to 106 µg total Pb per liter of water, dissolved Pb concentrations were not above detection limits.[4] Therefore, all measured surface water concentrations of dissolved Pb at the sampling locations of interest were below the detection limits used in the study.

- **Mississippi River**. Samples from upstream, at the facility, and downstream of the facility are presented and will be evaluated separately. Sample results are presented in Exhibit F-8. Although some of the measured total Pb concentrations exceeded the detection limit, all measured dissolved Pb concentrations were below the detection limit.

- **Other surface water samples from U-shaped pond and drainage areas.** Sample results are presented in Exhibit F-9. Again, all measured concentrations of dissolved Pb in these surface waters were below the detection limit of 3 µg/L.

4.1.5.2 Sediment

Sediment sample data were extracted and aggregated into geographic clusters based on ELM's *Characterization Area Investigation Report* (2005) at the same locations as the surface water concentrations presented in the previous section. Mean Pb concentrations in dried sediment sample clusters for Joachim Creek ranged from 34.5 to 78.7 mg/kg; for the Mississippi River from 12.1 mg/kg downstream to 30.1 mg/kg near the facility; and in the U-shaped pond mean sediment concentrations were 170 mg/kg. Exhibits 4-12 through 4-14 present a summary of the clustered sediment concentrations considered in this analysis. More detailed tables showing measured sediment concentrations at each location are presented in Appendix F.

Sediment samples to 10 cm in depth from the Mississippi River were collected from the back-eddies located behind wing dams and jetties, because ELM expected significant sedimentation in these areas due to the engineering of the Mississippi River by the Army Corps of Engineers. In Joachim Creek, ELM focused sediment sampling locations at point bars and other depositional areas. Joachim Creek exhibited sinuous meanders in the vicinity of the facility, which lies within one of the large loops of one of the meanders of the creek. Since the series of meanders lead to a slower and steadier current of the water, one would expect inorganic constituents associated with facility-related operations to sorb onto silt and organic matter particles and mobilized facility-related constituents to collect at the point bars and other depositional zones. ELM collected sediment samples from these areas to assess the potential for impacts from deposition of smelter emissions, possibly from both slag piles and air emissions, on the sediments of Joachim Creek.

ELM also collected sediment samples from backwater and low flow pond areas, ephemeral ponds, and drainage pathways to characterize impacts of air deposition of historical smelter emissions. They collected surface water and sediment samples from the lowest points in the watershed and drainage pathways within 2.4 km from the facility. This choice of four sampling

[4] Appendix B-2 of ELM (2005) reports surface water summary statistics in units of mg/L; this contradicts the units reported in Appendices C-5, C-6, C-7, and C-8 which present the measured constituent concentration results for the characterization area surface water samples. The Appendix C data tables all indicate that the measured values are in µg/L unless otherwise noted; therefore, µg/L was assumed to be the correct unit.

areas represented the areas of direct deposition from smelter emissions and deposition of impacted particles by surface water run-offs.

Given the focus of this assessment on impacts from air emissions from the smelter, sediment samples that were taken very close to the water seeps at the toe of the SSA were excluded, because the ELM reports states that the waters from Joachim Creek, on occasion, had been observed to come into contact with the SSA during flooding events.

Data were aggregated for the risk screening assessment as follows:

- **Joachim Creek**. Samples were clustered geographically and then averaged. Cluster 1 corresponded to the samples collected furthest upstream from the facility, while Cluster 5 was nearest upstream from the facility. Measurements are presented in Exhibit 4-12 and Exhibit F-10.

- **Mississippi River**. Samples from upstream, at the facility, and downstream of the facility are presented and will be evaluated separately. Measurements are listed in Exhibit 4-13 and Exhibit F-11.

- **Other surface waters including U-shaped pond and watershed drainage areas**. Measurements are presented in Exhibit 4-14 and Exhibit F-12.

Exhibit 4-12. Joachim Creek Measured Sediment Pb Concentration Data for the Primary Pb Smelter Case Study (ELM 2005)

Sample Cluster	Number of Sample Sites	Mean Pb Concentration in Dry Sediment (mg/kg)	Standard Deviation of Pb Concentration in Dry Sediment
Cluster 1	3	36.5	14.9
Cluster 2	4	56.0	20.0
Cluster 3	6	78.7	55.3
Cluster 4	11	30.0	8.5
Cluster 5	3	34.5	5.2

Exhibit 4-13. Mississippi River Measured Sediment Pb Concentration Data for the Primary Pb Smelter Case Study (ELM 2005)

Location with Respect to Facility	Number of Sample Sites	Mean Pb Concentration in Dry Sediment (mg/kg)	Standard Deviation of Pb Concentration in Dry Sediment
Upstream	5	14.7	8.1
Near facility	20	30.1	25.6
Downstream	11	12.1	3.2

Exhibit 4-14. Other Surface Water—Drainage Areas and U-Shaped Pond—Measured Sediment Pb Concentration Data for the Primary Pb Smelter Case Study (ELM 2005)

Sampled Surface Water Body and Sample ID No.	Number of Sample Sites	Pb Concentration in Dry Sediment (mg/kg)	Standard Deviation of Pb Concentration in Dry Sediment
Drainage area CHRDDP	1	110	NA
Drainage area RRDP-02	1	14.8	NA
U-shaped pond	4	170.1	90.6

4.2 Secondary Pb Smelter

The media concentrations for the secondary Pb smelter case study were estimated using modeling and then compared to available measured values. For the time-period represented by the emissions (1997 to 2000), air monitoring data near the facility did not indicate quarterly average air concentrations exceeding the current NAAQS[5], and modeled quarterly air concentrations were also all less than the current NAAQS (USEPA 2006f). Therefore, only a current conditions scenario was performed for this case study.

4.2.1 Spatial Template

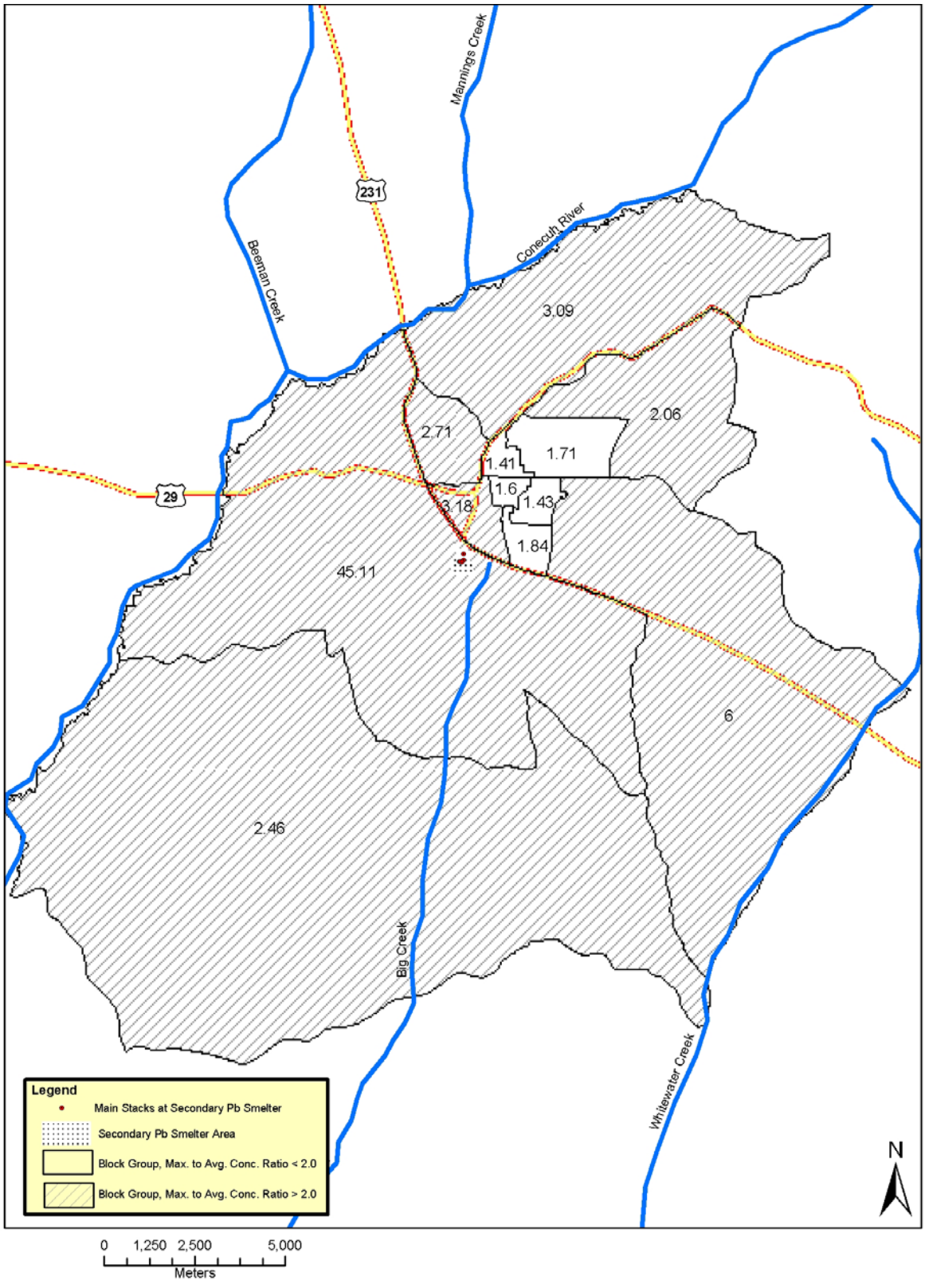
The study area extent was defined out to the distance where the concentrations associated with the secondary Pb smelter facility emissions reached approximately 50 percent of the background concentration of Pb in the area. Air concentrations generated from a preliminary AERMOD run with a radial grid of receptors placed at seven distances out to 50 km were compared to background Pb concentrations from the Interagency Monitoring of Protected Visual Environments (IMPROVE) database (IMPROVE 2006). The average (arithmetic mean) of all background Pb measurements taken between 2000 and 2004 at two nearby sites that were used for comparison to the secondary Pb smelter results were 0.0012 $\mu g/m^3$ (from 411 samples) and 0.0019 $\mu g/m^3$ (from 481 samples). Results of the preliminary air quality model run showed that at a radius of 10 km, estimated air concentrations were closest to 50 percent of the background average air concentrations. The deposition gradient was not considered for selecting the extent of the modeling region because background deposition measurements were not available.

[5] In 2003, the quarterly average Pb air concentration at one of the monitors near the secondary Pb smelter exceeded the current NAAQS (measured concentration was 1.9 $\mu g/m^3$) (USEPA 2006f). This time period was not included in this phase analysis due to availability of emissions data, but this monitor value indicates that emission rates may have changed between 2000 and 2003 at the facility.

Using GIS software, the 12 census block groups that are predominantly within 10 km of the facility were identified. Because of the irregular shape of block groups, not all of the block groups with area within 10 km were included, and some that were included have area outside 10 km. The selection was made to include block groups whose average exposure would predominantly come from air within 10 km of the source. Model receptors were placed at all block centroids within the 12 block groups of interest. This resulted in 665 block centroid points being modeled. Some of these blocks had zero population in the 2000 U.S. Census. Two air monitors from AQS/AirData (USEPA 2006f) were identified between 400 and 700 m of the facility. The locations of these 2 monitors were modeled as discrete receptors, for direct comparison purposes between model results and monitoring data.

There are 435 census blocks with non-zero population in the 2000 U.S. Census within the model region, and air dispersion model results were generated at all of these blocks. An attempt was made to limit the locations modeled for the exposure assessment, as was done for the primary Pb smelting case study (see Section 4.1.1). The average concentration and deposition in each of the 12 block groups was calculated by spatially-weighting the values at the block level. The area of each block was obtained from the U.S. Census (Census Bureau 2005). The range of block-level concentrations within each block group was considered by calculating the ratio of the maximum block-level air concentration in the block group to the average air concentration in the block group. For the primary Pb smelter case study, when these results were analyzed, the blocks groups with ratios greater than 2.0 were modeled at the block level and others modeled as block groups. For the secondary Pb smelter, although five block groups had maximum to average ratios less than 2.0, U.S. EPA and ICF decided to model these block groups as blocks because of the small size of the block groups and their proximity to the facility (see Exhibit 4-15). A total of 435 blocks with non-zero population were included in the assessment.

Exhibit 4-15. Modeled Blocks and Block Groups at Secondary Pb Smelter Facility



4.2.2 Air

The air concentrations and deposition of Pb from the secondary Pb smelter facility were modeled with AERMOD and compared to the air concentrations from nearby monitors. The emissions used for the air quality modeling are described in Section 3.2.4 and represent emissions from December 1997, November 1999, and February 2000.

4.2.2.1 Air Dispersion Modeling

The air dispersion model AERMOD was used for the secondary Pb smelter modeling (USEPA 2004a; USEPA 2004b). The meteorological data used includes five consecutive years (1990 to 1994) of nearby measurements. Surface-level and upper air meteorological data were obtained for weather stations located in Montgomery, Alabama (NOAA 1997a) and Centerville, Alabama (NOAA 1997b), respectively, and processed using the meteorological pre-processor, AERMET (USEPA 2002a). These stations represent locations close in proximity and geography to Troy, Alabama, and for which five consecutive years of surface and upper air meteorological data were available. Obtaining five consecutive years of weather observations for use in AERMOD was desirable because it allowed for the natural variability in weather conditions be captured in the air modeling.

AERMOD was configured to model the same emissions for the five years of meteorological data (1990 to 1994). Hourly air concentration and deposition results were averaged over this time period to obtain average air concentration and annual average deposition. As described in Section 3.2.4, emissions, release parameter, particle size parameters, and building downwash inputs were all obtained from the residual risk study of the facility (ECR 2006). Receptors and the study area boundary were selected specifically for this model application. Due to the relatively flat terrain in Alabama, terrain calculations were not used in this application.

4.2.2.2 Inhalation Exposure Concentrations

As in the primary Pb smelting case study, inhalation exposure concentrations of Pb were estimated for the population of interest (young children) from the estimated ambient air concentrations using age group- and location-specific relationships for Pb developed from modeling performed for U.S. EPA's 1999 National-Scale Air Toxics Assessment (USEPA 2006c), one of EPA's NATA activities. See Section 4.1.2.2 for more information.

The relationships (or ratios) between ambient Pb concentration and Pb exposure concentration from the NATA national-scale assessment for the 0 to 4 age group (the closest age group for which outputs are available to the age group of interest for this assessment) ranged from 0.44 to 0.46 for the census tracts in the study area for the secondary Pb smelter location. The ratios are presented in Appendix G, Exhibit G-1. Use of ratios for the 0 to 4 age group (rather than for 0 to 7) contributes some uncertainty in the estimate of children's inhalation exposure concentrations. Uncertainty related to the magnitude of the ASPEN modeled air concentrations is also discussed in Section 4.1.2.2.

4.2.2.3 Air and Deposition Results

Annual average Pb air concentrations and deposition fluxes were calculated at each block and monitor receptor point for the five years of meteorological data. To take into account variations in meteorological data, the annual average concentrations and depositions for each of the five years were averaged to generate one set of representative annual average concentration and deposition results for the current conditions scenario.

All concentration, deposition, and inhalation exposure estimates for the blocks modeled are presented in Appendix G, Exhibit G-2. In addition, Exhibit 4-16 presents the range of air concentration, inhalation exposure, and deposition values generated from the modeling. The values are summarized from the model results at the 435 blocks with non-zero population. Exhibit 4-17 shows isopleths of the block-level modeled air concentrations.

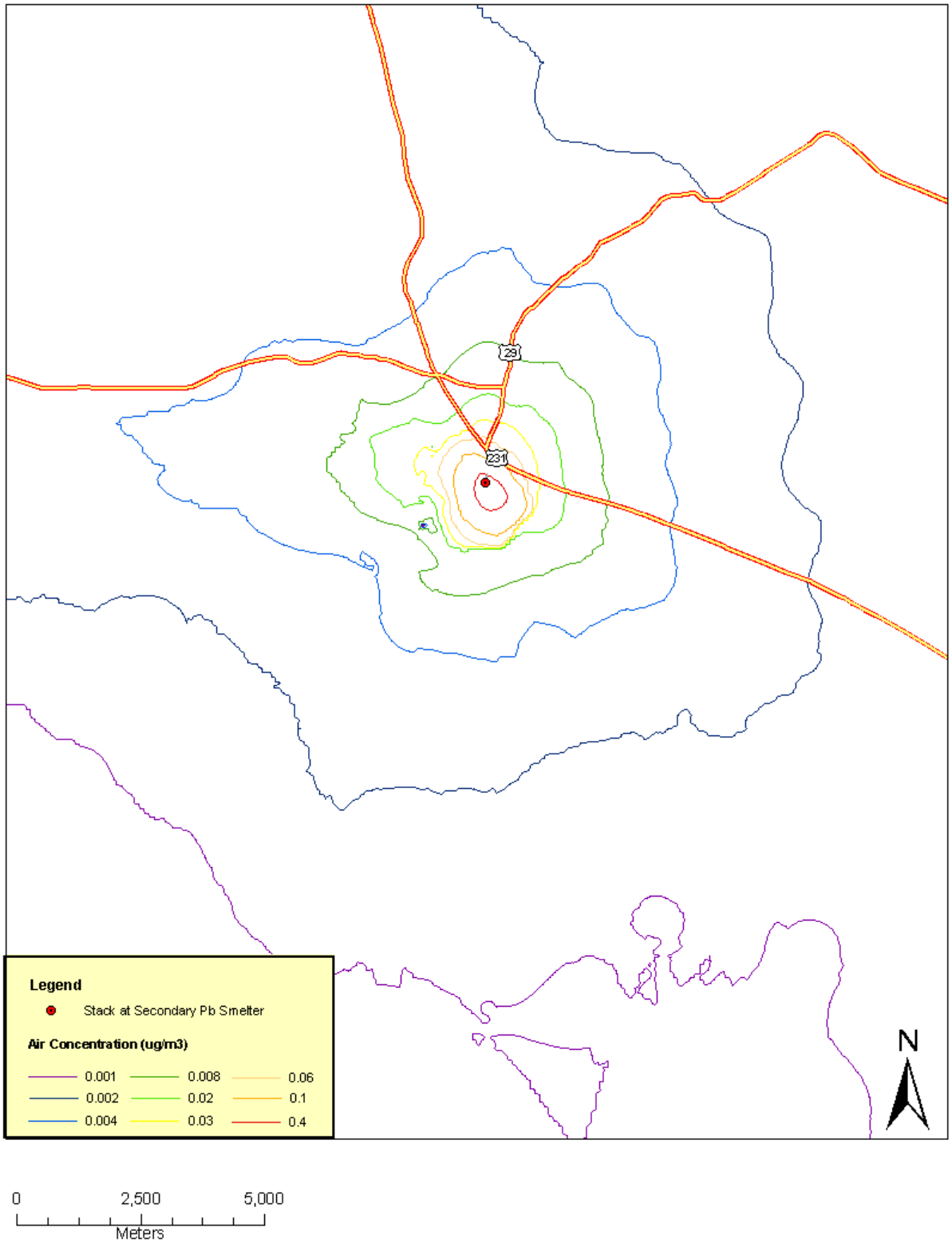
Exhibit 4-16. Secondary Pb Smelter Case Study: Summary of Modeled Air Concentrations and Deposition[a]

	Current Conditions		
Statistic[b]	**Average Annual Pb Air Concentration ($\mu g/m^3$)**	**Children Ages 0 to 4 Years Average Annual Pb Air Exposure Concentration ($\mu g/m^3$)**	**Total Deposition (g/m^2/year)**
Maximum	0.536	0.238	0.435
95th Percentile	0.0178	0.008	0.0066
Median	0.0047	0.0021	0.0013
5th Percentile	0.0009	0.0004	0.0002
Minimum	0.0005	0.0002	0.0001

[a] The 435 blocks with non-zero population selected for analysis were used to create this summary. Note that in some of these blocks the 2000 U.S. Census indicates there are no children.

[b] The statistic (e.g., 95th percentile, median) may not be at the same location for each of the data results presented here.

Exhibit 4-17. Secondary Pb Smelter Case Study: Concentration and Deposition Isopleths



4.2.2.4 Air Modeling Performance Assessment

The monitoring data at the two air monitor locations near the facility were compared to modeled concentrations at the same locations (see Exhibit 4-18). Overall, the modeled concentrations at the monitor locations are approximately three times lower than the monitor values. Both of the monitors are located to the northwest of the facility. Because the meteorological data is not site-specific, there is likely some uncertainty with its use in estimating air concentrations at specific points. It is possible that the local predominant wind direction is different from that of the meteorological data. Therefore, the monitored air concentrations were also compared to modeled air concentrations within similar distances to the facility, in all directions modeled on a radial grid (Exhibit 4-18). When compared to concentrations in all directions, the monitored values fall within the range of modeled results. A wind rose created from the five years of Montgomery, Alabama, wind data (Exhibit 4-19) shows that the predominant wind direction used in the model runs is directly to the west. The northwest (the direction of the monitors) is not a predominant wind direction in the metrological data set used for modeling; however, it is unknown whether this is the predominant wind direction at this site.

The potential difference between actual site meteorological data and the meteorological data used may help explain why the modeled concentrations are not closer to the monitored concentrations at the exact monitor locations, but modeled concentrations in all directions are within the range of monitored concentrations at similar distances. Because the monitors are both to the northwest of the facility (see Exhibit A-12), it is unknown whether modeled concentrations are underestimated in all directions. Thus, there is the potential that all air and deposition concentrations are underestimated from the modeling or that the degree of over- or under-prediction of concentrations from modeling is dependent upon direction. A directional difference between modeled and actual air concentrations can impact risk results (either under- or over-predicting) because the number of modeled children varies by census block and each census block is in a specific direction from the facility.

Exhibit 4-18. Secondary Pb Smelter Case Study Air Concentration Modeled Results Compared to Monitor Values

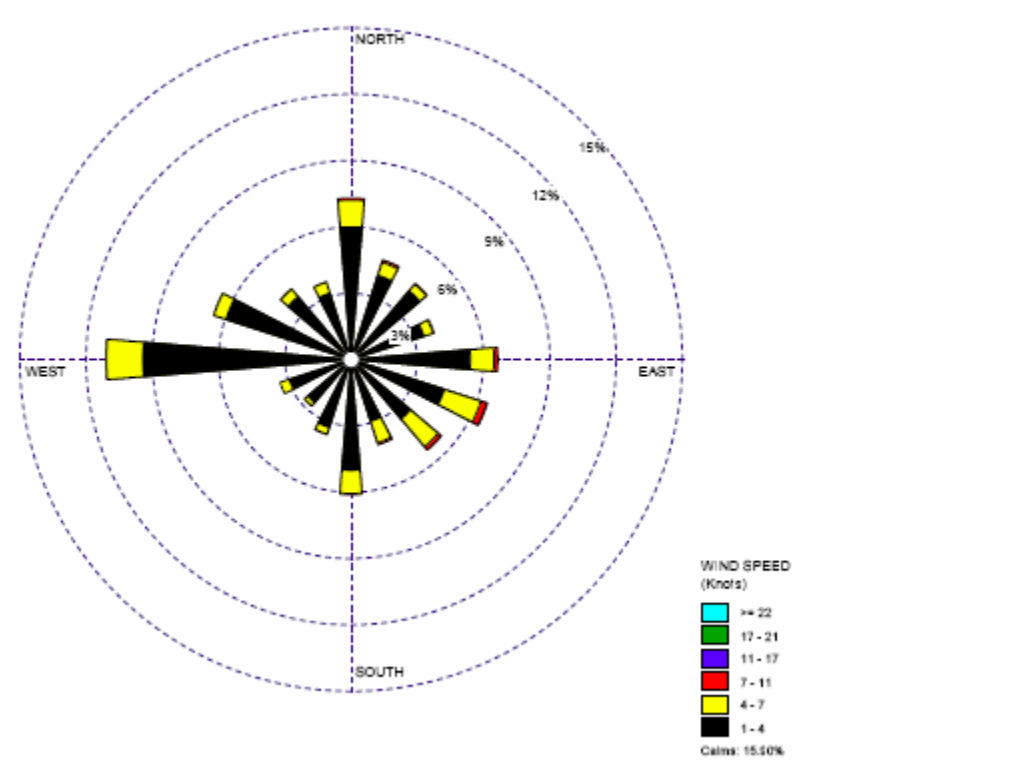
U.S. EPA AirData Monitor	Distance from Midpoint of Facility (m)		Modeled Results		
		Range of Monitor Air Concentrations from AirData 1997 to 2000 ($\mu g/m^3$)	Range of Modeled Distances for Comparison	Range of Modeled Concentrations [a] ($\mu g/m^3$)	Modeled Concentration at Monitor Location ($\mu g/m^3$)
11090003	400	0.383 to 0.474	300 to 500 m (54 Points)	0.08 to 6.0	0.11
11090006	680	0.132 to 0.198	500 to 800 m (72 Points)	0.038 to 0.17	0.050

[a] The modeled concentrations presented here were generated from a model run with a radial receptor grid. This summary is not from census block centroid results.

No local measurements of Pb deposition were found to compare to the modeled results. Studies were summarized in the U.S. EPA Pb criteria document (USEPA 2006b) that provided ranges of Pb dry deposition fluxes in various locations across the United States. None of these are specifically for deposition near a secondary Pb smelter, but they provide a range of deposition values for comparison. Exhibit 4-20 summarizes these deposition values.

The maximum annual estimated deposition flux from the secondary Pb smelter emissions at a census block centroid (0.435 g/m^2/year) is higher than any of the deposition fluxes presented in Exhibit 4-20. This is expected because none of those studies measured deposition directly next to a facility. The median estimated deposition flux (0.0013 g/m^2/year) does fall within the range of some of the measurements in Exhibit 4-20 (i.e., New York City, Detroit, and sites near Lake Michigan). The lower estimated deposition fluxes are at the low end of most of the measured ranges, which is expected because the locations of those fluxes could be described as urban background. Comparison of the modeled deposition fluxes to these measurements throughout the country provides some confidence that the modeled deposition is within the expected range.

Exhibit 4-19. Wind Rose of Meteorological Data Used for Secondary Pb Smelter Case Study (Direction in which Wind is Blowing)[a]



[a] Wind rose for 43,824 hours of data

Exhibit 4-20. Dry Deposition Fluxes from Studies in the United States

Location	Mean Value or Range (g/m^2/year)	Source
New York City, building exterior plate collecting deposition (weekly values from 2003 to 2005 averaged)	0.0098	Caravanos et al. 2006
Two sites on Chesapeake Bay in 1990 – 1991	0.00037 to 0.001	Wu et al. 1994
New York-New Jersey Harbor Bight area	0.00015 to 0.00076	Gao et al. 2002
Urban site in metropolitan Detroit 1982 – 1991	0.0004 to 0.004	Pirrone et al. 1995
Sites near Lake Michigan 1993 – 1995	0.0084 to 0.014	Yi et al. 2001
Lake Michigan	0.00095	Sweet et al. 1998
Lake Superior	0.00092	Sweet et al. 1998
Lake Erie	0.00078	Sweet et al. 1998

4.2.3 Outdoor Surface Soil

Outdoor soil concentrations were estimated using two methods for the secondary Pb smelter facility – a model-only approach and a hybrid approach using model results and surrogate measurements. The modeled soil concentrations were calculated for each census block analyzed for this case study using the AERMOD deposition estimates and EPA's Multiple Pathways of Exposure (MPE) methodology (USEPA 1998). The MPE methodology represents the update of the Indirect Exposure Methodology, or IEM (USEPA 1990a). MPE consists of a set of multimedia fate and transport algorithms developed by EPA's Office of Research and Development, including a soil mixing model which was used in this assessment to calculate the soil concentrations resulting from deposition at the secondary Pb smelter location.

In the MPE algorithms, cumulative soil concentration is calculated as a function of particle deposition, soil mixing depth, bulk density, and a soil loss constant. The soil loss constant (in this case) was set up to be a function of loss due to leaching, erosion, and runoff. Concentration in the soil was calculated in the top 1 cm of soil assuming constant deposition of Pb for the entire operating period of the facility (37 years). Although it is likely that the deposition has changed in the last 37 years, possibly even declining with reduced emissions, this method was used due to lack of soil samples and historical emissions data. With this assumption in mind, other assumptions in the soil concentration calculations were made conservatively. All input parameters used for the soil mixing model are listed in Appendix G, Exhibit G-4. Site-specific input parameters were used when feasible, but assumptions were made for some parameters, in many cases based on suggested values in the Human Health Risk Assessment Protocol (HHRAP), which contains a database of input parameters for the MPE equations (USEPA 2006g).

Soil concentrations estimated using the modeling approach for each census block are summarized in Exhibit 4-21, and presented in more detail in Appendix G, Exhibit G-2. Across the census blocks in the study area, the modeled top 1 cm soil concentrations from the facility's deposition ranged from 0.05 to 368 mg/kg. A background soil concentration of 15 mg/kg was added to these values for use in the exposure assessment (based on Gustavsson et al. 2001) to account for natural soil concentrations of Pb and general anthropogenic background.

As described in the next section (Section 4.2.3.1), a set of soil measurements at a similar facility was used to develop the hybrid soil concentration result set for the secondary Pb smelter case study (using modeled and surrogate data). Soil measurements taken in surface soil from 100 to 1000 m from a similar facility (Kimbrough and Suffet 1995) were compared to the modeled soil concentrations from this assessment. The modeled soil concentrations at the census block receptors are approximately three times lower than the measured concentrations. Thus, the modeled results were scaled up three times to account for the potential under-prediction of the soil model when the results were compared to the surrogate measurements. The hybrid soil concentrations are presented in Exhibit 4-21 along with the modeled soil concentrations. Consideration of these two approaches for characterizing soil Pb represents an important sensitivity analysis consideration for this case study and consequently, parallel sets of risk estimates (using each approach) are presented.

Exhibit 4-21. Summary of Secondary Pb Smelter Case Study Soil Concentrations[a]

Statistic	Modeled Average Soil Concentration from Deposition (mg/kg)	Background Concentration (mg/kg)	Modeled Approach Pb Soil Concentration: Modeled plus Background (mg/kg)	Hybrid Approach Pb Soil Concentration (mg/kg)
Maximum	368	15	383	1150
95th Percentile	6.16	15	21.2	63.5
Median	1.23	15	16.2	48.7
5th Percentile	0.15	15	15.2	45.5
Minimum	0.05	15	15.0	45.2

[a] Surface soil concentrations were calculated to a depth of 1 cm.

4.2.3.1 Performance Evaluation for Soil Concentration Estimates

The modeled soil concentrations were compared to measured soil concentrations near another secondary Pb smelter because soil measurements are not available near the secondary smelter used in this assessment. Kimbrough and Suffet (1995) measured Pb in soil samples out to from 100 to 1000 m from a secondary Pb smelter. Personal communication with Kimbrough indicated that the soil samples taken were from surface soil (information on the specific depth was not available) and in an urban area. Exhibit 4-22 presents the comparison of modeled soil concentrations at similar distances from the facility as presented in the Kimbrough and Suffet (1995) study.

Exhibit 4-22. Summary of Secondary Pb Smelter Case Study Soil Concentrations with Distance from Source Compared to Measurements[a]

Distance from Facility (m)	Kimbrough and Suffet (1995)		Secondary Pb Smelter Case Study	
	Measured concentrations (mg/kg)	Number of measured locations compared[b]	Modeled concentration + background (mg/kg)	Number of modeled locations compared[c]
100	140 to 1900	4		
200	770	1	190	1
250	400	1		
340 to 380			79 to 380	2
500	220 to 380	2		
620 to 680			34 to 51	5
750	150 to 490	4		
800	96	1		
830 to 880			31 to 34	2
925			31	1
1000	49 to 250	5		
1060 to 1090			25 to 29	4

[a] Soil concentrations were modeled to a depth of 1 cm, and the measurements represent surface soil (according to the study's author).
[b] Kimbrough and Suffet (1995) measured Pb concentrations in soil at different directions from the facility.
[c] Modeled soil concentrations compared here were generated from deposition estimates at census block centroid and air monitor locations. Some of the census blocks near the facility had zero population in 2000 and are therefore not included in other parts of this study.

The comparison between the modeled soil concentrations and the Kimbrough and Suffet (1995) soil concentrations varies, mostly as a result of direction from the source. On average, the measured concentrations are approximately three times higher than the modeled results, suggesting an underprediction of soil concentrations from the model results for this case study. Because of this potentially significant impact (downward bias) on exposure and risk results, an alternative soil concentration for the study area was calculated by combining the modeling results (used to characterize spatial variability in soil levels across the study area) with surrogate data from Kimbrough and Suffet (1995) to adjust the absolute soil Pb levels for the study area. The result is the "hybrid" soil concentration set presented in Exhibit 4-21, in which the modeled soil concentrations are scaled up three times.

4.2.4 Indoor Dust

Indoor dust sampling data were not available for the secondary Pb smelter case study, necessitating the use of modeling to characterize indoor dust Pb levels within the study area. A version of the AGG pooled analysis (USEPA 1989) model that uses ambient air Pb levels for predicting dust levels was chosen. This is a similar model as used for the primary Pb smelter case study at distances greater than 1.5 km from the source (see Section 4.1.4); however, in the case of the secondary Pb smelter, an "air-only" version of the model was employed reflecting the reduced overall confidence associated with soil characterization at this case study.

The air-only AGG model does reflect (implicitly) some consideration for the soil-to-indoor dust mechanism in the air signal. Specifically, the larger air factor for the air-only model (relative to the air *plus* soil dust model's air factor) reflects the fact that, in this version of the model, air measurements are used to cover both the direct loading of indoor dust from air and the loading of outdoor soil by air with subsequent impacts of that soil on indoor dust through such mechanisms as indoor tracking of soil (USEPA 1989). The AGG model used for the secondary smelter was based on a number of studies focusing mainly on primary Pb smelters. This basis carries an uncertainty into indoor dust predictions generated using this model related to the potential for differences between primary and secondary Pb smelters that may relate to indoor dust loading (e.g., particle size profiles and nature of the entrained Pb compounds). The air-only AGG model used in this analysis is presented below:

$$Pb_{dust} = 60 + (844 \times Pb_{air})$$

where:

Pb_{dust} = concentration of Pb in indoor dust (mg/kg), and Pb_{air} is the concentration of Pb modeled in the ambient air ($\mu g/m^3$).

Exhibit 4-23 presents a summary of the Pb indoor dust concentrations generated in the secondary Pb smelting study area for the current conditions scenario. Indoor dust concentrations were only estimated for census blocks with children. The numbers of census block or block groups and children modeled with increasing levels of Pb house dust are presented in Exhibit 4-23. All estimated Pb dust concentrations for the secondary Pb smelter facility are presented in Appendix G, Exhibit G-3.

Exhibit 4-23. Secondary Pb Smelter Case Study: Modeled Indoor Dust Pb Concentrations [a]

	Current Conditions	
House dust concentration (μg/g)	Census blocks/ block groups with dust concentrations greater than value [a]	Children living in area with dust concentration greater than value [b]
60	296	1,672
70	40	172
80	6	16
100	2	3
120	1	2

[a] The 296 blocks with children aged 0 to 7 in the 2000 U.S. Census were used to develop this summary. Note that Exhibits 4-21 includes all blocks with population, even blocks without children.

[b] Number of children ages 0 to 7 from the 2000 U.S. Census were used in this analysis (U.S. Census Bureau 2005).

Studies summarized in the 1990 review of the Pb NAAQS contained measurements of house dust ranging from 10 to 35,000 ppm. A high value of 100,000 ppm was measured in one home within 2 km of a Pb smelting facility (USEPA 1989). The Pb indoor dust concentrations for this case study fall within the range presented by U.S. EPA (1989), but they are at the low-end of the range. In addition, all of the concentrations are much lower than the 100,000 ppm concentration cited within 2 km of a Pb smelter. The fact that this facility is a secondary, and not primary, Pb smelter may explain some of the difference. In the study mentioned in Section 4.1.4, household dust Pb concentrations were measured at 14 homes within approximately 500 m (approximated from a figure) from a secondary Pb smelter in the Czech Republic (Rieuwerts et al. 1999). Measured Pb concentrations ranged from 861 to 5890 ppm with a geometric mean of 1668 ppm. Indoor dust concentrations in this secondary Pb smelter pilot phase assessment were not estimated within 500 m. At 590 m from the facility in this pilot phase of the analysis, the estimated indoor dust concentration is 130 ppm. In addition to the close distance, the secondary Pb smelter studied by Rieuwerts et al. (1999) used to be a primary Pb smelter, perhaps explaining why the indoor dust concentrations were higher than in this pilot phase of the analysis.

4.3 Near Roadway Urban Location Scenario

4.3.1 Spatial Template

As described in Section 3.3.1, the location of the study area for the near roadway urban case study was selected based on the availability of air monitoring data. Of the streets in the vicinity of the near roadway monitoring site in Houston, Texas, a thoroughfare to the south was selected as the street around which the scenario was designed, due to its proximity to the monitoring site and the length of the road links for that street.[6] The distance from the monitoring site to another major road is about the same as the distance from the site to the street that was selected (i.e., 100

[6] Although it was not a strict requirement that the selected roadway section used for this case study correspond to road link delineations, the road link was a convenient unit given the availability of road data, especially vehicle throughput data, by link. Matching the delineations of selected links may provide benefits in analyses beyond the pilot scale (e.g., in scaling up results to wider geographic areas).

m vs. 115 m for the selected roadway), but the roadway links for the selected street are shorter in this area. A shorter roadway segment was judged to be more desirable because it was assumed that less uncertainty would be associated with the assumption that the measured air concentrations are representative of conditions along the entire length of the roadway segment.

The subdivision of the study area into smaller regions for characterizing risks (i.e., the development of the spatial template) was based in part on research conclusions regarding the impacts of roadway proximity on the spatial gradient of pollutant concentrations near roads. Cohen et al. (2005) have conducted analyses using the CALPUFF non-steady state Gaussian puff model to estimate the dimensions of a "zone of influence" for the near-ground-level ambient air concentrations of several air pollutants, including diesel PM, in the vicinity of roadways. The zone of influence was defined as the area near a roadway in which ambient air concentrations are affected (i.e., enhanced) by emissions that occur in the roadway. Their results suggest that a steep spatial gradient exists adjacent to roadways, with a zone of influence existing out to somewhere between 200 and 400 m from the roadway.

In a related analysis of the data published by Cohen et al. (2005) as well as other air modeling results, ICF (2005) calculated the enhancement ratio (i.e., the ratio of near-roadway concentrations to concentrations more distant from the roadway) for several air pollutants, including diesel PM.[7] These enhancement ratios were developed for a 0 to 75 m zone and a 75 to 200 m zone adjacent to the roadway. Therefore, these same intervals formed the basis for the spatial template for air concentrations.

The inner 0 to 75 m zone was further divided into two areas, with one comprising the area 0 to 12 m from the roadway and the other encompassing the area between 12 and 75 m from the road. The inner region at 0 to 12 m was developed to facilitate the use of soil measurements available for a surrogate location (see below) that included measurements at and within 12 m of the roadway. This allowed an apparently steep initial decrease in near-roadway soil Pb levels to be captured by the exposure modeling for this case study. Aerial photographs of the area indicated that relatively small residential yards are situated immediately adjacent to the road, suggesting that this initial 12 m band may serve as a play area for children. Consequently, the spatial template for this case study included exposure bands parallel to the road and on either side, extending from 0 to 12 m, 12 to 75 m, and 75 to 200 m. Although six separate zones could be defined – the exposure bands are located on both sides of the road – only three sets of media concentrations were necessary for the blood Pb modeling, as soil and air concentrations were assumed to be the same on both sides of the road.

4.3.2 Air

4.3.2.1 Ambient Air Concentrations

For this scenario, 0.005 µg/m^3, the average value for Pb in the PM_{10} fraction available from the measurements collected for this location (ICF 2006b), was assumed to be a reasonable estimate

[7] Literature reviews did not identify any research specifically focused on spatial gradients associated with Pb sorbed to particulates near roadways or reentrained roadside dust. In the absence of more specific data, diesel PM was assumed to be a reasonable surrogate for Pb with regard to near-roadway concentrations.

of the ambient Pb concentration at the monitoring station site.[8] To derive media concentrations for the zones adjacent to the roadway, geometric means of the enhancement ratios for near-roadway concentrations of diesel PM from ICF (2005) were used to adjust the concentration at the monitoring station. As described in the previous description of the spatial template, enhancement ratios were derived for the intervals of 0 to 75 m from the road and 75 to 200 m from the road. Because the near roadway monitoring station in Houston was located about 115 m from the selected street, the average measured Pb concentration at this station was assumed to be representative of ambient Pb in the 75 to 200 m zone. Using the geometric mean of the enhancement ratios, the concentration for the 0 to 75 m band was then estimated to be approximately 0.008 $\mu g/m^3$. Ambient air concentrations of Pb are summarized in Exhibit 4-24.

Exhibit 4-24. Enhancement Ratios and Air Concentrations for Near Roadway Urban Scenario

Zone (Distance from Roadway)	Enhancement Ratio[a]	Estimated Ambient Pb Concentration, $\mu g/m^3$	Method for Deriving Pb Concentration
0 to 12 m *and* 12 to 75 m	2.4653	0.008	Calculated based on relative values of enhancement ratios
75 to 200 m	1.6204	0.005	Average of measured values from DRI Houston site

[a] Geometric means of enhancement ratios for diesel PM; from ICF (2005).

A comparison of the ambient air concentrations used for this scenario to other data can help evaluate the validity of the selected media concentrations. Although the average Pb concentration at the monitoring station was estimated based on only three days of measurements, the average value of 0.005 $\mu g/m^3$ is reasonably consistent with the ambient Pb measurements for Houston reported in AQS (USEPA 2006f). Specifically, the average Pb measurements for PM_{10} at the industrial/suburban site identified in AQS were 0.006 and 0.007 $\mu g/m^3$ for 1999 and 2005 (see Section 3.3.4). The average of the Houston DRI Pb measurements is also near the middle of the range of averages of ambient measurements obtained by DRI (ICF 2006b) for the other five near-roadway monitoring locations included in their study, which range from 0.0024 to 0.0164 $\mu g/m^3$.

4.3.2.2 Inhalation Exposure Concentrations

As in the primary Pb smelting case study, inhalation exposure concentrations of Pb were estimated for the population of interest (young children) from the estimated ambient air concentrations using age group- and location-specific relationships for Pb developed from

[8] It is noted that the determination of air concentrations for the other two human health scenarios used air levels associated with total suspended particulates (TSP) rather than PM10. No TSP data for Pb were available for the Houston monitoring site. As summarized in the Pb criteria document (USEPA 2006b), reentrained Pb in surface soil is generally associated with smaller particle size fractions; thus, it seems likely that the PM10 Pb measurements presented here represent the bulk of Pb exposure (if the source of Pb is indeed resuspended surface soil). It is possible, however, that actual Pb levels may be slightly underrepresented by the Pb air concentrations derived for this scenario from the Houston data. At another air monitor in Houston (labeled as commercial, urban/center city location), mean annual TSP Pb measured between 1996 and 2006 ranged from 0.02 $\mu g/m^3$ to 0.005 $\mu g/m^3$.

modeling performed for U.S. EPA's 1999 National-Scale Air Toxics Assessment (USEPA 2006e), one of U.S. EPA's NATA activities. See Section 4.1.2.2 for more information.

For the near roadway urban scenario, the study area was encompassed by one census tract (FIPS ID 48201520500). The relationships (or ratios) between ambient Pb concentration and Pb exposure concentration from the NATA national-scale assessment for the 0 to 4 age group (the closest age group to that of interest for this assessment for which outputs are available) is 0.4603 for this census tract. The exposure-to-concentration ratios for this tract for children 0 to 4 years old were used to calculate inhalation exposure concentrations for this case study (see Exhibit 4-25). Note that the use of ratios for the 0 to 4 age group (rather than for 0 to 7) contributes some uncertainty in the estimate of children inhalation exposure concentrations. Uncertainty related to the magnitude of the ASPEN modeled air concentrations is also discussed in Section 4.1.2.2

Exhibit 4-25. Estimated Inhalation Exposure Concentrations for the Near Roadway Urban Scenario

Zone	Ambient Concentration ($\mu g/m^3$)	Age Group	Exposure-to-concentration Ratio	Inhalation Exposure Concentration ($\mu g/m^3$)
0 to 12 m and 12 to 75 m	0.008	0 to 4 yrs	0.4377	0.0033
75 to 200 m	0.005	0 to 4 yrs	0.4377	0.0022

4.3.3 Soil

As described in Section 3.3.4, although no measurements of Pb in surface soil were identified in the vicinity of the case study location in Houston, a range of data collected at roadsides and similar sites at other locales were identified in the literature. Using these data, surrogate soil Pb values were developed for the Houston DRI monitoring site to estimate soil concentrations for this case study. The soil concentrations derived for this scenario are intended to represent Pb levels in surface soil that are primarily attributed to Pb emitted from vehicles during the historical use of leaded gasoline and deposited to the near roadway soils (rather than attributed to other urban sources of Pb, such as deposition of Pb from industrial sources or Pb paint). As such, the near roadway case study is intended to address the concern described in the criteria document (USEPA 2006b) that Pb deposited in surface soil over time derived from exhaust from automobile traffic burning leaded gasoline represents a potential source via direct exposure to the soil and through resuspension of dust to ambient air.

Measurements of Pb in roadside surface soil collected by Turer and Maynard (2003) in Corpus Christi, Texas, were identified as the most useful measurements for this scenario, primarily because of their proximity and geographic similarity to the case study location. Corpus Christi, located about 300 km southwest of Houston, is closer to the coast than Houston, but has roughly the same annual average air temperature and precipitation (NCDC 2006). Turer and Maynard collected samples at two locations, including a site adjacent to the entrance ramp to Interstate 37. The AADT for that site in 1998 was reported to be 48,000 vehicles per day. This is a higher traffic volume than the volume reported for the selected street in Houston near the case study (AADT about 20,000). This difference in traffic volume was judged to be acceptable for the

pilot phase. Although elevated Pb concentrations in soil have been observed to be statistically correlated with increased traffic volume on adjacent roadways in many cases (see, for example, Sanchez-Martin et al. (2000) and Fakayode and Olu-Owolabi (2003)), there appear to be other confounding variables that can result in weak correlations between traffic counts and Pb levels in near roadway soil in some cases (Hafen and Brinkman 1996). In other words, Pb is generally more elevated in soils near heavily-traveled roadways, but the exact impact of traffic volume can vary between locations and under different conditions. Total Pb concentrations in surface soil for three samples taken near Interstate 37 are summarized in Exhibit 4-26.

Exhibit 4-26. Pb Concentrations in Surface Soil, Corpus Christi, Texas[a]

Sample Location as Distance from Roadway	Sample Depth	Total Pb Concentration (mg/kg)
2 m	0 cm	731
3 m	0 cm	766
12 m	0 to 5 cm	214

[a] From Turer and Maynard (2003); sampling site near I-37 and Martin Luther King Drive at North Port Avenue close to downtown.

Before using the measurements presented by Turer and Maynard (2003) to develop media concentrations for this case study, other relevant data from the literature were reviewed. In their discussion, Turer and Maynard refer to Pb measurements by Harrison (1987) at the "edges of roadways and freeways in Corpus Christi" with concentrations of 250 mg/kg; no description of the specific sampling locations or distance from the roadway was provided. Other relevant data include records in EPA's SPECIATE database of four samples in Houston. Although these samples are in the same town as this case study, they are judged to have greater uncertainty because it is not known when the samples were taken or where the sampling occurred, especially with regard to proximity to roadways. The sampling is described as "reentrained soil dust" (no sampling depth was specified). All four of these SPECIATE samples were reported to be 0.04 percent Pb by weight, or 400 mg/kg.

In another recent study, Lejano and Ericson (2005) sampled surface soil for Pb in Pacoima, California (near Los Angeles), collecting a total of 176 measurements across eight sampling categories. Four of these categories include exclusively samples taken within 150 m of a major roadway, with each of the four categories focusing on a different road in the Pacoima area. Mean total Pb concentrations for each of these four sampling categories are reported here (see Exhibit 4-27). Traffic volume at each location as reported by Lejano and Ericson is also presented. Specific sampling locations (i.e., distance from the roadway) were not reported; however, mean Pb concentrations for three of the four roads were above 100 mg/kg. These results suggest that the near-roadway concentrations of Pb in soil within 50 m of the road could be substantially higher than the reported means.

Exhibit 4-27. Near Roadway Surface Soil Pb Concentrations in Pacoima, California[a]

Sampling Category (Adjacent Roadway)	Mean Pb Concentration (mg/kg)	AADT (Vehicles/day)
San Fernando Rd	171	10,250
I-5	119	292,000
S-118	102	142,000
S-210	43	116,000

[a] From Lejano and Ericson (2005).

There is substantial evidence that Pb concentrations in surface soil decrease rapidly with distance from the roadway. Sutherland and Tolosa (2001) reported that the relationship is approximately linear when the log of concentration is plotted against the log of distance from the roadway. Similarly, Filipelli et al. (2005) have reported an exponential decay in Pb concentration with increasing distance from the roadway. Hafen and Brinkmann (1996) surveyed results from several other studies and observed a generally exponential decrease in Pb concentration with distance from the road. In general, based on the conclusions of these studies, it appears that Pb concentrations adjacent to a road decrease down to the local background level within a distance of about 50 m from the roadway.

Local background in an urban environment, however, can vary depending on proximity to current and previous sources of Pb emissions, impacts from Pb-based paint, and other factors. No representative "urban measurements" for Houston were identified, but several studies reporting other relevant measurements were identified as follows:

- Mean concentration of total Pb in surface soil for "random" non-vehicular-impacted locations in Pacoima, California was reported as 111 mg/kg (Lejano and Ericson 2005).

- Soils at parks, schools, and roadside embankments in Corpus Christi were sampled in 1984. The arithmetic mean surface soil concentration of Pb across all 485 samples was 208 mg/kg (Harrison 1987, cited in Battelle 1996).

- Median Pb concentration in surface soil in samples from a range of land-use categories in Miami, Florida was reported to be 98 mg/kg (Chinreje et al. 2004).

- An approximate level of "background" for Pb in US soils is about 15 mg/kg, taking into account natural soil concentrations of Pb and general anthropogenic contributions (Gustavsson et al. 2001).

Based on the available data and considering the relevant details of the near roadway urban scenario, the measurements near I-37 in California reported by Turer and Maynard were selected as the primary data source for developing surrogate soil concentrations for this scenario. As described in Section 4.3.1, the zone extending out to 75 m was subdivided into two zones, a near-roadway zone from 0 to 12 m and another zone encompassing the area between 12 and 75 m. These boundaries were defined based on the availability of Pb measurements in surface soil from Turer and Maynard (2003) out to 12 m. For estimating the average concentration within

the 0 to 12 m zone, the relationship between log of Pb concentration and log of distance from the roadway within this near roadway range was assumed to be linear (Sutherland and Tolosa 2001). Based on this relationship and using the data points for samples taken 3 and 12 m from the road, intermediate concentrations between 3 and 12 m were estimated. The mean concentration for this zone was calculated using these intermediate values to be about 388 mg/kg.

For the study area between 12 and 75 m from the roadway, it seemed reasonable that Pb concentrations in surface soil would decrease more slowly until reaching local background levels. Based on the available literature (and especially Lejano and Ericson 2005), a surrogate background level of 100 mg/kg was defined for the case study location. For simplicity and consistency with the spatial template, it was assumed that the soil concentration at 75 m and beyond was at this level. In the absence of other, more specific data on near-road soil concentration patterns, the Pb surface soil levels for this case study were assumed to decrease linearly between 12 and 75 m. Given the soil concentrations defined for 12 and 75 m, the average concentration for this interval was calculated to be 157 mg/kg. Estimated soil concentrations for this case study are summarized in Exhibit 4-28.

Exhibit 4-28. Estimated Soil Exposure Concentrations for the Near Roadway Urban Scenario

Zone	Surface Soil Concentration (mg/kg)
0 to 12 m	388
12 to 75 m	157
75 to 200 m	100

4.3.4 Indoor Dust

Because indoor dust sampling data were not available for the near roadway urban case study, an empirical model was used to describe the relationship between Pb concentrations in indoor dust and in outdoor soil. Specifically, a pooled analysis model (the AGG model) identified relates indoor dust levels to ambient air and soil Pb levels was used (USEPA 1989). In this case, unlike the secondary Pb smelter case study, soil Pb levels were considered because surrogate empirical data on Pb levels near roadways were available, thereby increasing confidence in the characterization of soil Pb levels for this case study. The same version of the "AGG" model (soil *plus* air regression model) (USEPA 1989) used for the primary Pb smelter was also used for the near roadway urban case study:

$$Pb_{dust} = 31.3 + (638 \times Pb_{air}) + (0.364 \times Pb_{soil})$$

where:

Pb_{dust} = concentration of Pb in indoor dust (mg/kg)
Pb_{air} = concentration of Pb modeled in the ambient air ($\mu g/m^3$)
Pb_{soil} = concentration of Pb in outdoor soil (mg/kg).

Exhibit 4-29 presents a summary of the Pb indoor dust concentrations generated in the near roadway urban study area for the current conditions and current attainment scenarios. Indoor dust concentrations were only estimated for census blocks with children. In addition, because only three sets of soil and air concentrations were estimated for the near roadway urban case study (corresponding to the three zones surrounding the road of interest), only three indoor dust concentration estimates were derived. Exhibit 4-29 shows the number of census blocks and children modeled with increasing levels of Pb house dust. Estimated Pb dust concentrations for all census blocks included in or partially overlapping with the near roadway study area are presented in Appendix D.

Exhibit 4-29. Near Roadway Urban Case Study: Modeled Indoor Dust Pb Concentrations [a]

House Dust Concentration (μg/g)	Census Blocks with Dust Concentrations Greater than Value	Children Living in Area with Dust Concentration Greater than Value [a]
71	50	319
94	25	91
178	7	7

[a] Number of children ages 0 to 7 from U.S. Census 2000 were used in this analysis (U.S. Census Bureau 2005).

As noted in Section 4.3.2.1, Pb measurements in outdoor air were only identified for Pb associated with PM_{10} and not TSP. Because concentrations of Pb in indoor dust were estimated using outdoor ambient Pb measurements that (for this case study) were reflective of the PM_{10} fraction only, modeled indoor dust levels may be underestimated. The degree to which the modeled indoor dust levels are underpredicted (if at all) would depend on: (1) the extent that the PM_{10} fraction represents total Pb in ambient air, and (2) the sensitivity of predicted indoor dust Pb concentrations to outdoor ambient Pb concentrations (especially with respect to the relative influence of soil concentration inputs on this estimate).

4.4 Near Roadway Non-Urban Locations

For the ecological risk screening assessment, measured soil concentrations reported in the literature were extracted for use in the risk characterization for this scenario. In contrast to the human exposure and health risk assessments, data were sought for areas not located in downtown areas. Less developed locations were evaluated to increase the likelihood of proximity to ecological receptors. Soil concentrations for two non-urban locations near roadways are summarized in this section.

4.4.1 Soil Concentrations at Corpus Christi, Texas Site

Turer and Maynard (2003) analyzed samples for Pb from roadside surface soil collected in Corpus Christi, Texas. The data summarized here are from an industrial site at I-37 and Upriver Road, which had a traffic volume of 45,000 vehicles per day. Exhibit 4-30 shows the concentrations of Pb measured at various distances and sample depths from the roadside.

Exhibit 4-30. Pb Concentrations in Surface Soil, Corpus Christi, Texas, for Ecological Risk Screening Assessment[a]

Sample Location – Distance from Roadway (m)	Sample Depth (cm)	Total Pb Concentration (mg/kg)
2	2.5	340
2	10	650
2	20	15
4	2.5	140

[a] From Turer and Maynard (2003); sampling site near I-37 and Upriver Road in refinery operations area.

4.4.2 Soil Concentrations at Atlee, Virginia Site

As described in Speiran (1998), the Virginia study site is located at the northbound Atlee/Elmont exit (exit 86) from Interstate 95 (I-95), about 14.5 km north of Richmond, Virginia. The northbound exit ramp at exit 86 is separated from the highway lanes by a triangular-shaped grassy area. The grassed area has been mowed periodically, but there has been no historical application of pesticides or fertilizers. Chemical characteristics of the soil in the grassed area were investigated in 1994, prior to planned construction modifications to Interstate travel lanes and drainage ways and installation of a detention basin. Soil samples in the grassed area were collected at various depth intervals and analyzed for heavy metals and other constituents.

Samples from seven soil sampling locations were initially collected in the grassy area in June 1994. Samples were collected at 3-inch intervals up to a depth of 12 inches and were analyzed for metals. An additional 11 soil locations were sampled in the grassy area in October 1994 and were analyzed in the same manner as the June 1994 samples. Five of these 11 sampling locations were sampled at the 0 to 3 and 9 to 12-inch intervals. Two locations were sampled at the 0 to 3 and 3 to 6-inch intervals. The remaining four locations were sampled at the 0 to 3 inch interval only.

Samples used for this ecological screening risk assessment were limited to those located at a distance 2 m or greater from I-95 and the exit ramp at exit 86 and collected at a depth of 20 centimeters or less. Therefore, only samples collected at 0 to 3 inches and 3 to 6 inches and samples that appear to be at least 2 m from either I-95 or the exit ramp (using figures provided in Speiran [1998]) were used in the evaluation of potential ecological risks for this near roadway non-urban scenario. Samples at these locations were considered to be representative of potential ecological exposures. Pb analytical results for these samples are presented in Exhibit 4-31. Pb concentrations at these locations and depths ranged from 17 to 540 mg/kg, with an average concentration of 123 mg/kg.

Exhibit 4-31. Pb Concentrations in Soil Collected from Potential Ecological Exposure Area, I-95 Interchange, Atlee, VA[a]

Location ID	Pb Concentration 0 to 3 Inches in Depth (mg/kg)	Pb Concentration 3 to 6 Inches in Depth (mg/kg)
1	46	35
2	73	23
3	46	17
4	70	31
5	94	24
6	510	540
10	99	--
16	140	--
22	100	--

[a] From Speiran 1998 (see Figures 2 & 3 and Table 1).

4.5 National-Scale Surface Water and Sediment Ecological Risk Screening Assessment

This section provides a description of the media concentrations used in the national-scale surface water and sediment screening assessment. Section 4.5.1 describes the selection of a data set to represent Pb concentrations in freshwaters across the United States resulting from atmospheric sources of Pb. Section 4.5.2 provides information on the selected data set for dissolved Pb water column concentration data that were used in the surface water portion of the screening assessment. Section 4.5.3 describes the freshwater sediment data set compiled for the sediment portion of this screening assessment. These sediment data were selected based on results from the surface water portion of the screening assessment.

4.5.1 Assessment of Available Data Sources for Surface Water Data

To identify measured dissolved Pb concentrations in surface waters of the United States for use in the exposure analysis of the national-scale surface water and sediment screening assessment, readily available summaries of Pb concentrations in surface waters and database compilations of trace metal concentrations measured in surface waters were examined. The most promising databases are those developed and maintained on-line by U.S. EPA and the U.S. Geologic Survey (USGS). They include U.S. EPA's STORET database, the USGS National Water Information System (NWIS) database, and the USGS National Water-Quality Assessment (NAWQA) Data Warehouse. Each of these databases was evaluated to select the one most appropriate for the surface water portion of the national-scale surface water and sediment screening assessment. The evaluation is briefly summarized below.

STORET: The STORET database from the EPA's Office of Water includes the water quality sampling data that states develop to determine compliance with water quality standards and that states report to EPA annually. STORET also includes sampling data from Superfund sites and

other sources (e.g., U.S. Army Corps of Engineers). The EPA BASINS interactive software program uses the data from STORET. The Legacy Data Center (LDC) part of STORET is a static, archived database that contains historical water quality data dating back to the early part of the 20th century and collected up to the end of 1998. The main STORET database is an operational system actively being populated with water quality data. It contains data collected beginning in 1999, along with older data that have been properly documented and migrated from the LDC. Both the LDC and STORET contain raw biological, chemical, and physical data on surface and ground water collected by federal, state, and local agencies, Indian Tribes, volunteer groups, academics, and others. All 50 states, territories, and jurisdictions of the United States are represented; however, few stations are present in some states (e.g., Texas). For an analysis of current water conditions, the operational STORET system is more appropriate than the LDC.

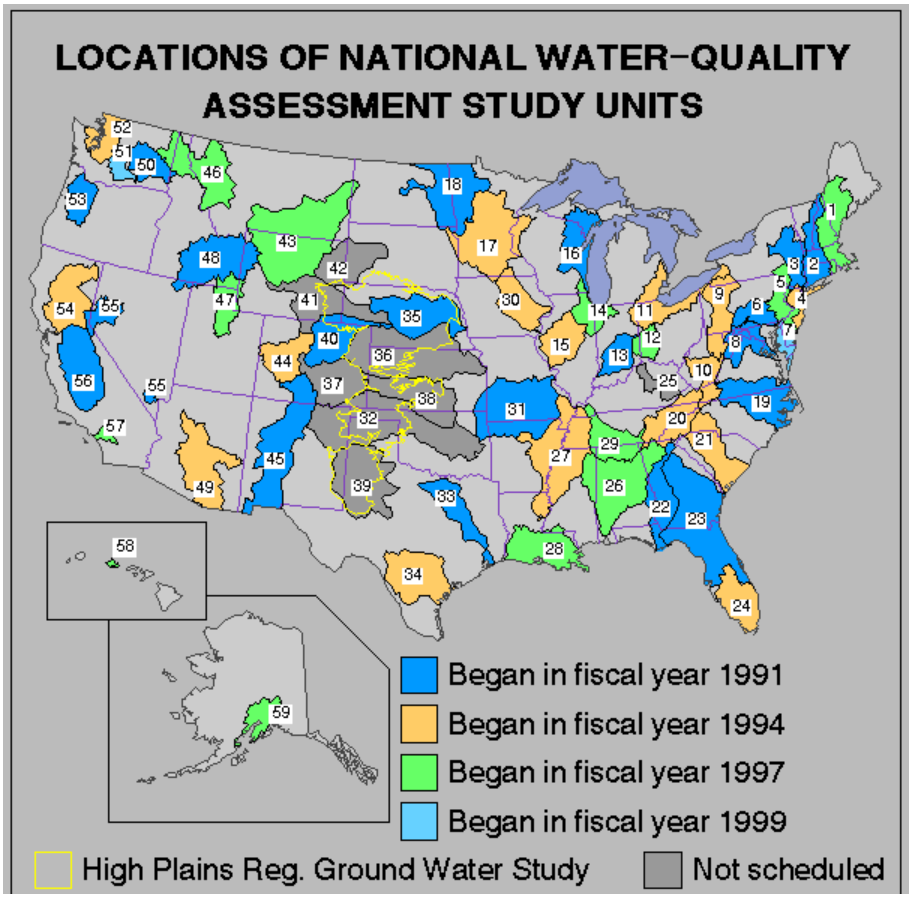
In a download of surface water concentrations for Pb from STORET for the months January 2004 to May 2006, it was observed that detection or reporting limits appeared to range from less than 0.04 to 10 µg/L based on the values reported by different states, with a quantitation limit of 1 µg/L being common. A total of 30,993 samples were found for Pb from the United States' surface waters over that time period. Of the total reported Pb concentrations included in that download, only 32 percent (9,910 of 30,993) were for dissolved Pb; the remaining concentrations were for total recoverable Pb. Of the 9,910 samples analyzed for dissolved Pb, only 21.5 percent (2,134) had dissolved Pb concentrations above the detection or reporting limit. Thus, only 7 percent of all samples analyzed for Pb (2,134 of 30,993) provided dissolved Pb measurements above the quantitation limit.

Limitations of STORET data include differences in sampling density per unit area or length of stream, differences in frequency of sampling, differences in detection and quantitation limits across states, and inconsistencies in how samples are collected, analyzed, and reported across states, particularly when estimating concentrations between the detection and quantitation limits. For Pb, another limitation is that many states do not attempt to measure dissolved Pb concentrations, only total Pb concentrations, because measurement of dissolved Pb generally results in non-detects.

USGS Databases: USGS maintains water quality data in two separate data sets: the National Water Information System (NWIS) Network and the National Water Quality Assessment (NAWQA) Data Warehouse (USGS 2004). The NWIS system was originally developed for water flow data, and includes data from 1.5 million sites across all 50 states, the District of Columbia, and Puerto Rico. Chemical quality data have been added to it in recent years, but still are limited. The data have been compiled through 2004 from a variety of projects ranging from national-level studies to small watershed projects; therefore, the sampling methods, density of sampling sites, and detection limits are variable across the data set. Data for dissolved Pb are available from 1972 to 2001; however, the vast majority of records are from the 1980s when the quantitation limit for Pb at most locations was 5 µg/L.

The NAWQA Data Warehouse includes the sampling data from the NAWQA program that started systematically collecting chemical, biological, and physical water quality data from a large number of water basins and aquifers—referred to by NAWQA as "Study Units"—across the nation in 1991. As of July 2006, data have been compiled for samples collected through September 30, 2004, and cover 47 Study Units. Basins from all regions of the United States are included; however, only approximately 50 percent of the land base is covered by these basins (see Exhibit 4-32). NAWQA Study Unit boundaries frequently cross state boundaries, and their size usually exceeds 10,000 square kilometers (about 3,900 square miles).

Exhibit 4-32. Map of NAWQA Study Units



Pb is one of the analytes in the program, but similar to the case for STORET, dissolved Pb measurements are available for fewer than 20 percent of the sampling locations. In an analysis of the data in NAWQA, U.S. EPA found a total of 3,445 measurements of dissolved Pb in surface waters, for which 86 percent were nondetects (see Exhibit 4-33) (USEPA 2006b). When looking at a subset of those data determined to be "natural" (i.e., forest or rangeland) or "ambient" (background) areas, 88 percent of the 430 samples for dissolved Pb in surface waters were non-detects. A mean concentration of 0.66 µg/L was calculated for all sites. A mean concentration of 0.52 µg/L was calculated for samples considered to represent natural areas. The relatively low dissolved Pb concentrations associated with the 95th percentile values (Exhibit 4-33) indicate that relatively few water samples are likely to exceed the initial aquatic screening

value of 1.2 µg/L. Data for dissolved Pb from NAWQA from 1994 to 2004 was examined and it was found that the quantitation limit for dissolved Pb changed for this program from 1 µg/L to 0.08 µg/L in the fall of 2000.

Exhibit 4-33. Distribution of Dissolved Pb Concentrations in Freshwater Surface Waters from the USGS NAWQA Database (USEPA 2006b)

Statistic	"Ambient" Locations	"Natural" Locations
Sample size	3445	430
% Non-detect	86	88
Minimum (µg/L)	0.04	0.04
Arithmetic mean (µg/L)	0.66	0.52
95^{th} Percentile (µg/L)	1.1	0.50
Maximum (µg/L)	29.78	8.40

It was concluded that of the three available surface water quality databases, the NAWQA data set is most appropriate for the surface water portion of the national-scale surface water and sediment screening assessment for several reasons. The inclusion of dissolved Pb as an analyte is limited in all of the databases (total Pb is measured more often). None of the databases provide the co-located measurements of water hardness in the same records as the measurements of dissolved Pb. STORET and NWIS include samples from more locations in the United States than does the NAWQA data set, but the sampling and reporting protocols represented in STORET and NWIS are less consistent from site to site. Data for dissolved Pb in NWIS are predominantly from the 1980s, and therefore do not represent current conditions. The NAWQA data set, on the other hand, provides representative (even though not complete) coverage of the United States, with samples through 2004 included in the database. The NAWQA data set also provides a consistent approach to sampling and analysis of the elements using consistent quantitation limits across the country. Given the sampling design for NAWQA and the consistency of the data across the country, it is considered to be more appropriate for a national-scale surface water and sediment screening assessment than the other two data sets.

Even with using the NAWQA database, for which the quantitation limit for dissolved Pb was 1 µg/L from 1994 to 2000, a difficulty for the assessment is that the chronic Ambient Water Quality Criterion (AWQC) for the protection of aquatic life in soft waters (i.e., for waters with $CaCO_3$ concentrations less than 45 mg/L), is less than 1 µg/L, thus the detection limit is too high to resolve the chronic AWQC for soft water. The most recent quantitation limit of 0.08 µg/L (starting in the fall of 2000) is adequate to identify concentrations of dissolved Pb that exceed the chronic AWQC benchmark in water as soft as 4.7 mg/L $CaCO_3$. Also, since the fall of 2000, dissolved Pb concentrations between the detection (0.04 µg/L) and quantitation (0.08 µg/L) limits were estimated in some cases.

4.5.2 Dissolved Pb Surface Water Concentrations in the NAWQA Database

To download dissolved Pb concentrations from the USGS NAWQA Data Warehouse, the NAWQA Study Units, trace elements in water, and the years 1994 to 2004 were specified (see Appendix H) to avoid earlier years when leaded gasoline in some locations might have still influenced air Pb concentrations. Once the data were downloaded, the values for dissolved Pb (analyte # 01049 - Pb, water, filtered, micrograms per liters) were examined.

The 49 Study Units for which sampling data currently are included in the USGS NAWQA Data Warehouse database are listed in Appendix H, Exhibit H-1. The different sampling stations within the unit were identified with an unique station ID number. Some units contained some recent series of sampling stations very close to each other along a river or stream (evident from the latitude and longitude reported for the stations); those were considered to be separate units.

It is important to note that the intensity of sampling at different locations within a Study Unit varied substantially. Sampling at some Study Units began in 1991, while sampling of others did not begin until 1999. In general, a few of the locations were sampled at least monthly for several years. In contrast, a single sample had been taken at many other locations, perhaps as an exploratory exercise. Some locations were sampled a few times a year at three year intervals, while other locations were sampled for only a couple years.

The percentage of the Study Unit sampling stations for which *dissolved* Pb was an analyte during one or more sampling times from 1994 to 2004 was 16 percent overall (Exhibit H-1), which is quite low. As is true for EPA's STORET database, measurements of total Pb are much more common than measurements of dissolved Pb. However, because the AWQC for Pb are expressed as dissolved Pb, total Pb measurements were not evaluated. The distribution of dissolved Pb measurements was not uniform over the Study Units or across sampling stations within Study Units. Dissolved Pb was not measured for any of the stations for some of the Study Units, while for other Units, dissolved Pb was measured at least once at a majority of the Sampling Stations.

There are temporal trends in whether dissolved Pb was measured at given station. Most notably, at many stations for which multiple years of data are available, if the first year of sampling indicated all dissolved Pb measurements to be less than the quantitation limit (QL) of 1 µg/L, no further measurements of dissolved Pb were attempted (e.g., dissolved Pb measured in 1999 less than 1 µg/L; no further measurements in 2000 through 2004).

Most dissolved Pb concentrations reported in the NAWQA database are below the QL. There are only 2,558 samples analyzed for dissolved Pb in surface waters for the NAWQA database from 1994 to 2004. The 3,445 samples included in EPA's review of the database (Exhibit 4-33) include samples from 1991 forward (USEPA 2006b). Of the 2,558 samples examined, 83 percent (2,116) were below the QL. Of the 442 dissolved Pb concentrations reported in the data set, 17 percent (74 of 442) were estimated.[9] Exhibit 4-34 summarizes the number of samples that were below the QL for dissolved Pb according to the reported QL.

[9] Of the 74 estimated values, 68 were for concentrations above detection limit of 0.04 µg/L but below the QL of 0.08 µg/L.

Exhibit 4-34. Number of Dissolved Pb Samples Less than the QL

QL	Number of Sample Less Than QL[a]
2	11
1	2010
0.5	1
0.2	10
0.16	1
0.08	83
Total	**2,116**

[a]A total of 2,558 samples were examined, of which 2,116 were below the QL.

Limiting the exposure assessment to the year 2000 to present was considered, given that fall 2000 was when the QL for dissolved Pb dropped to 0.08 µg/L, low enough to determine if the chronic surface water screening value is exceeded for most waters (i.e., for waters with hardness > 4.7 mg/L as $CaCO_3$). However, less than 14 percent of the total samples for dissolved Pb between 1994 and 2004 were analyzed with the lower dissolved Pb QL of 0.08 µg/L. To determine that percentage, the number of samples reported as "< 0.08" µg/L (N = 83), the number reported as estimated (E) with a value less than 0.08 µg/L (N = 68), and the number for which the measured value was equal to or greater than 0.08 but less than 1.0 µg/L (N = 205) were added. A few of the last group might have been measured with a QL between 0.08 and 1.0 (see Exhibit 4-34); however, no attempts were made to adjust for that. Thus, the percentage of samples analyzed for dissolved Pb in the NAWQA data set between 1994 to 2004 for which the QL limit was 0.08 µg/L is approximately 14 percent (i.e., 356 of 2,558). Thus, limiting the surface water and sediment water screening assessment to the years for which the QL was 0.08 µg/L would have severely limited the exposure data for the assessment.

Further descriptions of the dissolved Pb concentrations from 1994 to 2004 in the NAWQA Data Warehouse for the surface water portion of the screening assessment are provided in Section 7.2.4 (Risk Characterization) as they relate to the aquatic screening levels (AWQC).

4.5.3 Total Pb Sediment Concentrations from NAWQA Database

The surface water portion of this national-scale surface water and sediment ecological screening assessment identified 15 locations at which at least one measurement of dissolved Pb in the water column exceeded the water-hardness-specific Pb chronic AWQC. These locations are likely to represent a fraction of locations that actually exceed the Pb chronic AWQC. Nonetheless, those 15 locations were made the focus of the sediment portion of the screening assessment.

The NAWQA database also includes total Pb sediment concentration data measured for water bodies across the United States. Sediment sampling locations in the NAWQA database co-located with the surface water samples evaluated for the water column screening assessment were identified. Co-located sediment and surface water data were available for 9 of the 15 locations. There were an additional three "near-matches," with a sediment sampling station on

the same creek or river, but somewhat upstream or downstream of the surface water sampling location. For the sediment sampling sites that were categorized as near matches, however, it is important to note that the surface water samples at the same location had not exceeded the chronic AWQC in the surface water column portion of the screening assessment. Therefore, the value of the "near-match" locations is limited. Finally, there were three locations for which even a near-match was not found.

The sediment total Pb concentration data are presented in Chapter 7. Exhibit 7-16 summarizes the total Pb sediment concentrations measured at the nine locations with co-located sediment and surface water data and the three "near-match" locations. The NAWQA database appears to have only one measurement of Pb sediment concentrations for each of these 12 locations. Several of those samples were taken in the early 1990s, earlier than the bulk of the surface water Pb concentrations used in the surface water screening assessment. The concentrations ranged from 23 to 2,900 mg Pb/kg dry sediment. The high concentration was detected at a location that had an upstream NPDES permit for metals; thus the Pb contamination may have come from a source other than air.

5. Human Exposure Assessment and Blood Pb Estimation

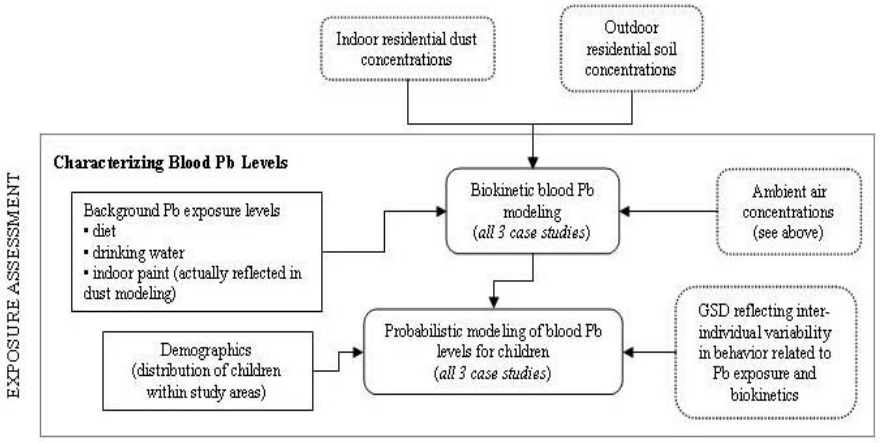
This chapter describes the approaches and methods that were used to evaluate children's Pb exposures to policy-relevant sources and policy-relevant background pathways, and to predict the changes in individual blood Pb levels and population blood Pb distributions associated with these exposures. Section 5.1 provides a description of the human exposure assessment and blood Pb modeling approaches, and presents the derivation of the model inputs. Section 5.2 reviews the results of the blood Pb model performance evaluation. Section 5.3 discusses the limitations and uncertainties associated with the human exposure assessment and blood Pb estimation, and Section 5.4 summarizes the results of the blood Pb modeling as applied to the primary Pb smelter, secondary Pb smelter, and near roadway urban case studies.

5.1 Blood Pb Modeling Methods

5.1.1 Overview of Human Exposure Assessment and Blood Pb Modeling Approach

Exhibit 5-1 (which is an excerpt from Exhibit 2-2) summarizes the human exposure assessment and blood Pb modeling methods used in this assessment. Exposure media (air, soil, and indoor dust) concentrations and Pb exposure and intake assumptions associated with both policy-relevant sources and policy-relevant background pathways serve as inputs to two biokinetic blood Pb models (discussed in Section 5.1.2). For the purposes of the pilot phase assessment, drinking water and diet pathways are categorized as policy-relevant background, although it is recognized that a portion of Pb in both pathways may derive from policy-relevant sources.

Exhibit 5-1. Schematic Representation of the Human Exposure Assessment and Blood Pb Modeling Process



Exposure concentrations are the primary inputs to the blood Pb modeling process. In each exposure pathway, the relationship between exposure concentration and Pb uptake (absorbed dose) is defined by a range of factors related to physiological processes and to the chemical and physical properties of the exposure media. These factors include respiratory volume, soil and dust ingestion rates, and gastrointestinal (GI) absorption fractions for diet, water, and soil/dust, which determine how much Pb is absorbed from each medium. Values for these factors differ with age and across exposure media, as discussed in Section 5.1.4.3.

As discussed in Chapter 4, exposure concentrations in air, soil, and indoor dust have been estimated for census blocks and block groups within the case study areas. Application of a specific set of exposure concentrations and the physiological and behavioral inputs to the blood Pb models yielded a lifetime (birth to 84 months [or 7 years] of age) blood Pb profile for the child receiving that combination of exposures, from which two blood Pb metrics have been derived. The first metric is the lifetime average (age six to 84 months) and the second is an estimate of "concurrent" blood Pb concentration, which has been defined as the average over ages 73 to 84 months).[1] The blood Pb models yield central tendency estimates of a child's blood Pb concentrations for specified simulation periods (with the temporal precision varying depending on the specific model) and for specific patterns of exposure. Estimates for a single child (representing central tendency exposure) do not provide information about how individual responses to Pb exposure might vary among the exposed children or how changes in individual blood Pb levels would affect the population exposure for a given case study. Thus, a probabilistic approach has been implemented to capture both the effects of inter-individual variability in blood Pb levels and the population distribution of exposures on the resultant population distribution of blood Pb statistics.

Development of population distributions of blood Pb levels for each of the case studies involved the following steps:

1. Blood Pb model was used to generate central tendency estimates of blood Pb per census block or block group.
2. The number of children (0 to 7 years of age) residing in each block and block group was determined from the U.S. Census Bureau data (U.S. Census Bureau 2005).
3. A lognormal distribution reflecting inter-individual variability in both behavior and biokinetics related to Pb exposure was developed using a geometric standard deviation (GSD) obtained from the literature which was centered on a value of 1.0. Values selected from this distribution represent "adjustment factors" reflecting specific patterns of behavior and biokinetics in children related to Pb exposure.
4. Population-weighted random sampling was then used to first select a blood Pb level from the results of Step 1 above. The probability for sampling each census block or block group is set proportional to the number of young children (0 to 7 years of age) in residence (obtained in Step 2). The data set generated in Step 1 was sampled 10,000 times in this way. Each of these 10,000 blood Pb levels was then multiplied by an adjustment factor selected from the lognormal distribution (described in Step 3) to produce a blood Pb level for a simulated child. These 10,000 simulated child blood Pb levels were then used to characterize (via percentiles) the distribution of blood Pb levels in the population.

The predicted blood Pb distribution in the exposed population reflects variability contributed by both the population-weighted distributions of exposure concentrations and by the inter-individual responses to Pb exposures.

[1] The rationale for defining the average blood Pb for ages 73 to 84 months as concurrent reflects the fact that the average age of IQ testing in the Lanphear et al. (2005) study of blood Pb-IQ relationships was approximately seven years (see also Chapter 6).

The following sections discuss in detail the blood Pb models used for this assessment, the selected model inputs, and how the models were implemented to estimate case study-specific blood Pb levels for children 0 to 7 years of age.

5.1.2 Selection of Blood Pb Models

Two biokinetic models were applied to predict blood Pb levels in the pilot phase assessment. These two models were the Integrated Exposure Uptake Biokinetic (IEUBK) Model for Children (hereafter referred to as the IEUBK model) and the International Commission for Radiation Protection (ICRP) model (hereafter referred to as the Leggett model). Both are well-documented, widely used, and have been subject to a range of testing and calibration exercises (see Section 4.4 of USEPA 2006b). A third "empirical" (regression-based) model developed by Lanphear et al. (1998) was included in the sensitivity analysis, as described in Chapter 6 and Appendix K.

5.1.2.1 The IEUBK Model

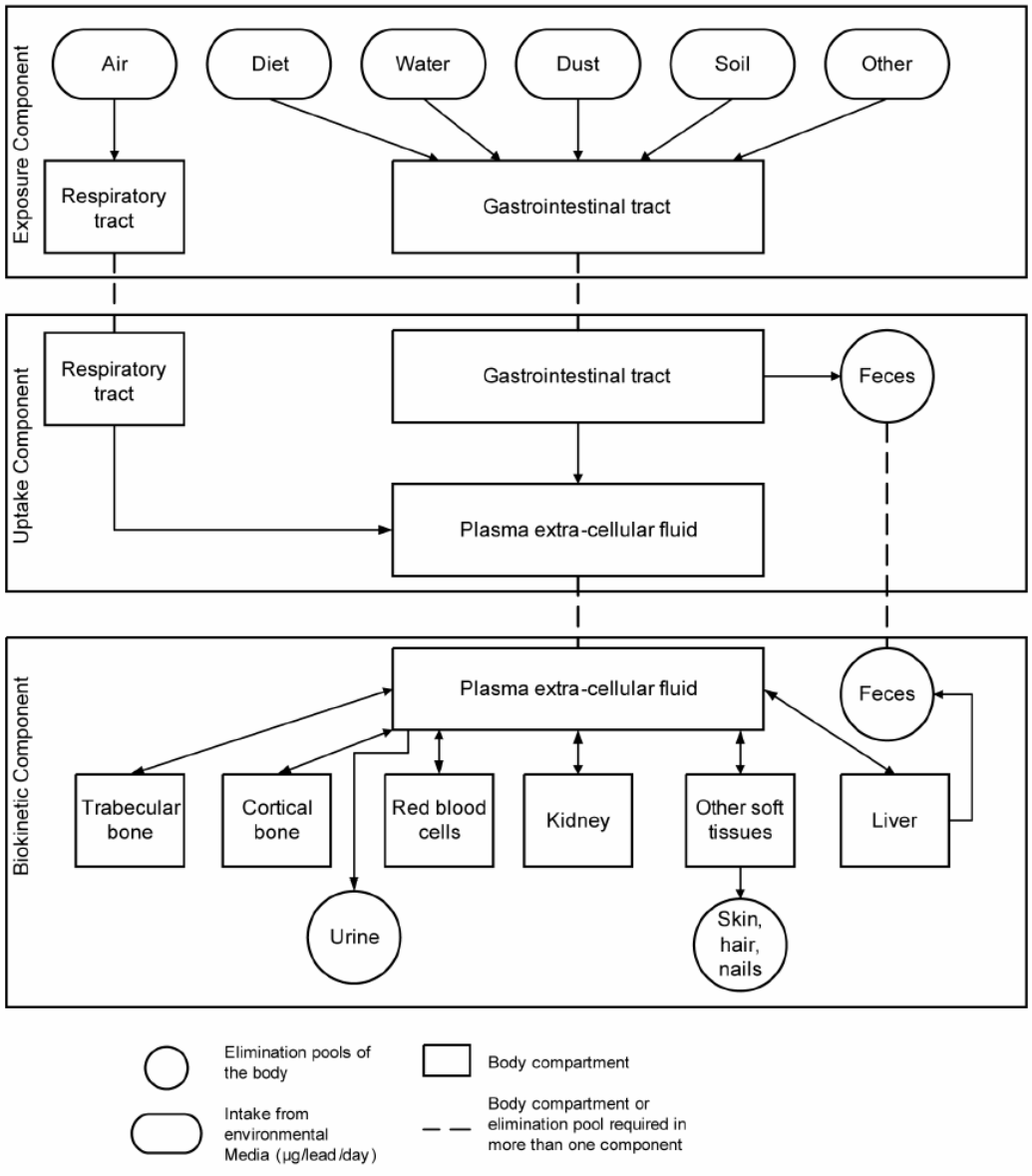
This section presents a general overview of EPA's IEUBK model, the primary blood Pb estimation tool in the pilot phase assessment. The IEUBK Lead Model Version 1.0 Build 263 (USEPA 2005a) was used to model blood Pb levels for all three case studies. The IEUBK model consists of three main modules: the exposure module, the uptake module, and the biokinetic module (see Exhibit 5-2). The exposure module accepts inputs related to six exposure media: air, food (excluding water), water, soil, indoor dust, and other. Exposure, intake, and uptake factors were combined with exposure concentrations in the various media to estimate Pb uptake through two pathways: inhalation and ingestion. The IEUBK model provides default values for the various model input parameters that can be adjusted by the user for a specific application. These parameters include those used by the model to estimate Pb uptake, including absorption and inhalation rates, water intake, dietary intakes of specific food classes, and soil and dust ingestion rates. The selection of model input parameter values for the pilot phase assessment is discussed in Section 5.1.4.

The exposure module also includes default age-specific estimates of time spent outdoors, as well as along with age-specific estimates of outdoor and indoor air Pb concentrations, inhalation rate, and respiratory tract absorption fraction, all of which are used to estimate age-specific Pb inhalation uptakes. The absorption fraction implicitly reflects both deposition of inhaled Pb in the respiratory tract and absorption of deposited Pb, either from the respiratory tract or from the GI tract. The model also contains an option for calculating indoor dust Pb concentrations based on an empirical relationship between air, soil, and dust Pb levels (a variation of the AGG models discussed in Chapter 4). Ingestion uptake is calculated using absorption fractions that are specific to the ingested media (food, water, soil, or dust). Total GI Pb uptake is modeled as being composed of a saturable and an unsaturable component.

In the biokinetic component of the model, absorbed Pb (from ingestion and inhalation) is assumed to appear immediately in the plasma-extracellular fluid (ECF) compartment. The plasma-ECF compartment constitutes the central compartment in the biokinetic model from

which exchange to all the other compartments occurs. Cortical and trabecular bone (which are not directly coupled in the IEUBK model) constitute the main long-term storage compartments, with the estimated turnover in other compartments being more rapid. The binding capacity of the red blood cell compartment is modeled as being saturable, simulating the limited capacity of ALAD and other Pb binding proteins. Pb excretion occurs through a urine pathway (distinct from the kidney compartment); hepatobiliary secretion is coupled with the liver compartment, with a minor component of excretion from "other soft tissues" (i.e., skin, hair, and nails). The model is termed "biokinetic," rather than "pharmacokinetic," because transfers between compartments are controlled by first-order transfer coefficients (equivalent to first-order rate constants), rather than being perfusion-controlled. A more complete description of the derivation and structure of the IEUBK model can be found in USEPA (1994) and White et al. (1998).

Exhibit 5-2. Structure of the IEUBK Model (USEPA 1994; White et al. 1998)



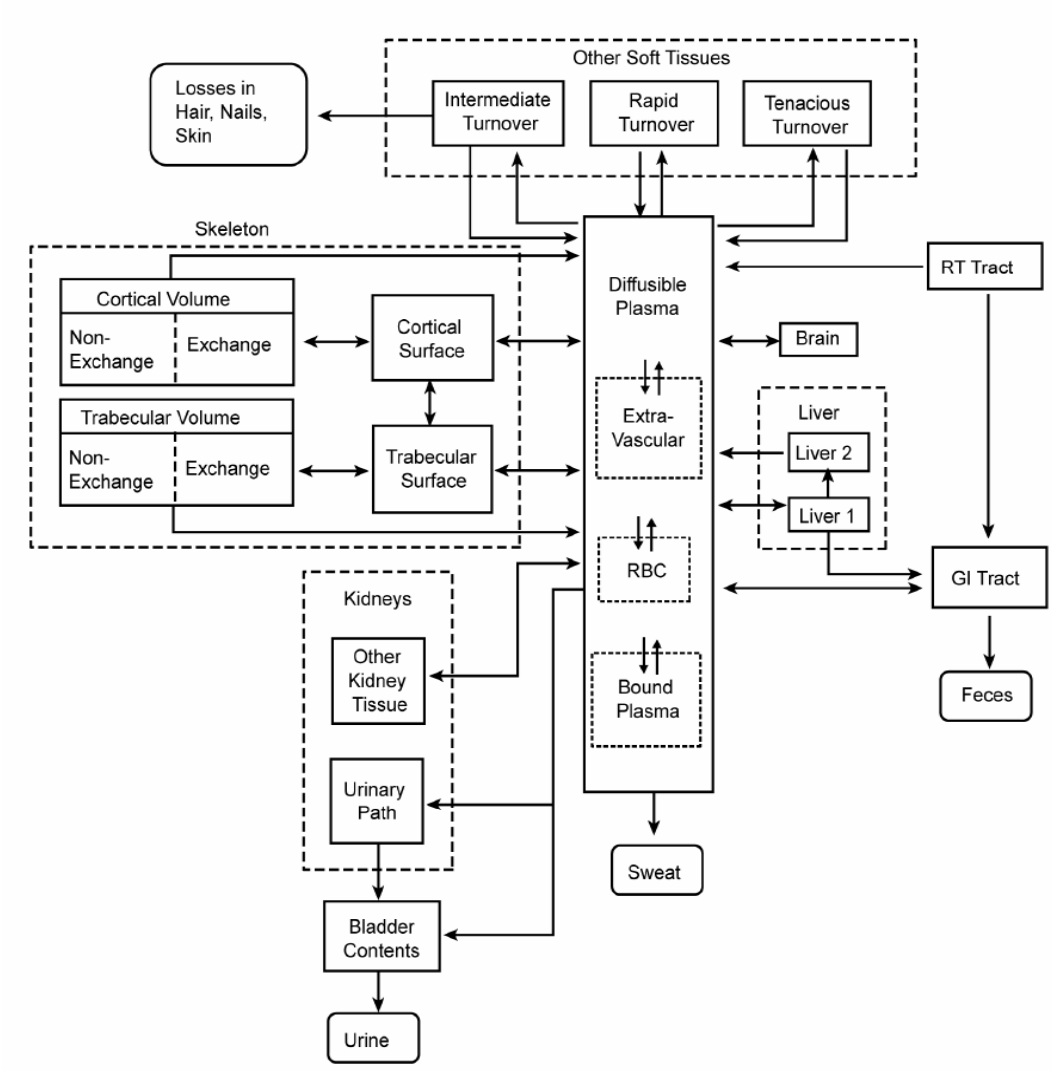
The structure of the IEUBK model is deterministic. As configured, the model generates point estimates of blood Pb concentrations in response to a given set of exposure parameters. Using the batch mode, the user may calculate the impacts of changing Pb concentrations in various media, and the model also allows the user to plot the impact of continuously changing the Pb concentration in a specified exposure medium between two values. The user can specify the age at the start of exposure, an initial blood Pb level, and the age at which the simulation ends. For each case study in the pilot phase assessment, the batch mode of the IEUBK model was used to generate blood Pb levels for the different age intervals for children living in each census block and block group.

The IEUBK model also allows the user to calculate blood Pb distributions based on the assumption of lognormality (USEPA 2002b). In this mode, the model uses the point estimate of blood Pb as the GM of a lognormal distribution with a defined GSD. Graphs of the projected blood Pb distributions can be generated, along with estimates of the proportions of exposed children with blood Pb levels above user-specified levels of concern.

5.1.2.2 The Leggett Model

Leggett (1993) published a sophisticated biokinetic model for Pb exposures, intended primarily for use in radiological dosimetry, that has since been adopted as the standard model for this purpose by the International Commission for Radiation Protection (ICRP). The model predicts changes in blood Pb levels for exposed individuals over their entire lifespan (birth to 90 years old). The compartmental structure of the Leggett model (Exhibit 5-3) is more complex than that of the IEUBK model, and was patterned after similar models developed by the ICRP to model the age-specific biokinetics of calcium-like radionuclides (Leggett 1993).

Exhibit 5-3. Structure of the Leggett Biokinetic Model (Leggett 1993)



Differences between the structures of the Leggett model and the IEUBK model include:

- The published version of the Leggett model lacks the multipathway exposure module of the IEUBK. The Leggett model accepts total respiratory and ingestion intakes (administered doses) as inputs and calculates Pb uptake using age-specific absorption factors.

- The Leggett model lacks a "probabilistic" component; all predictions are deterministic for a single individual receiving a given set of exposures, with no capability for generating graphical outputs.

- The central exchange compartment in the Leggett model is "diffusible plasma," rather than the plasma-ECF compartment used in the IEUBK model. Extra-vascular fluid, red blood cells (RBCs), and a bound plasma fraction are the other blood/fluid compartments

that exchange directly with plasma in the Leggett model, with different transfer rates reflecting differences in estimated exchange rates.

- The trabecular and cortical bone compartments in the Leggett model are divided into two subcompartments each, one exchangeable, and one "non-exchangeable." Pb in the "non-exchangeable" compartments of both types of bone can be remobilized, but only relatively slowly as a result of bone remodeling.

- Urinary excretion is modeled in the Leggett model as part of an integrated kidney compartment, rather than separately as in the IEUBK model.

- In the Leggett model, the liver is modeled as two compartments with rapid and moderately rapid Pb exchange, respectively. Other soft tissues are modeled as having three compartments with differing exchange rates. Pb in brain tissue is explicitly modeled.

On the whole, the Leggett model is somewhat more technically sophisticated than the IEUBK model, but lacks a built-in facility to convert exposure concentrations into Pb uptake and to integrate uptakes from multiple exposure media.

Like the IEUBK model, the Leggett model is biokinetic, with exchange between compartments modeled using first-order "transfer rates" (equivalent to rate constants). The values for the transfer rates were estimated using a range of data from adult human radioactive tracer studies, autopsy data from adults and children, and data from animal studies related to the absorption, deposition, and excretion of Pb and chemically similar elements (Leggett 1993). The Leggett model differs from the IEUBK model in that data from short-term studies (on the time-scale of hours to days) are used to estimate parameter values for the most rapid of the uptake and exchange processes and thus the time resolution of the Leggett model is much finer than that of the IEUBK model. The user may specify any step length, depending on the degree of time resolution required in the blood Pb predictions. Like the IEUBK model, the binding capacity of Pb in RBCs is assumed to be saturable.

Leggett developed estimates of biokinetic parameters for six age categories: newborn (0 to 100 days), 1 year, 5 years, 10 years, 15 years, and 25 years and older, with age-specific transfer parameters for children estimated by interpolation between the nearest values. Transfer factors for children were adjusted to take into account the more rapid bone turnover (calcium/Pb addition and resorption) in children compared with adults.

Predictions from the Leggett model have been compared with the deterministic predictions of blood Pb levels generated by the IEUBK model, using the IEUBK default inputs (Pounds and Leggett 1998). In that comparison, the Leggett model predictions were substantially higher than those of the IEUBK model.[2]

[2] Blood Pb levels generated using the Leggett model for this exposure analysis (Section 5.4) are systematically lower than the IEUBK-based blood Pb levels. This trend is the opposite of that reported in the Pounds and Leggett (1998) article. Given that a customized version of the Leggett model was developed for the pilot phase assessment, additional model evaluation activities are being completed for the pilot phase (see Section 5.1.3.3).

5.1.3 Application of Blood Pb Models

5.1.3.1 Key Modeling Assumptions

For all of the case studies, exposure concentrations were assumed to be constant throughout the seven-year duration of the exposure scenario. Exposure concentrations associated with policy-relevant sources were estimated and modeled based on data from the recent past (1997 to 2005), as summarized in Exhibit 2-4.

Data from the U.S. Census provided estimates for the year 2000 of the numbers of children living in each block or block group (U.S. Census Bureau 2005; see Chapter 3); the numbers of exposed children were assumed to be constant through the entire exposure period (to 7 years). In- and out-migration to and from the case study areas were not considered. To the extent that the case study exposures were higher than might occur at typical residences outside the study areas, this assumption may have overestimated blood Pb impacts compared to those that might actually occur, and vice versa.

As discussed further below, maternal blood Pb levels during pregnancy were assumed to be identical for all children at a level consistent with nationally representative values for women of childbearing age (see Section 5.1.4.3). Thus, all children were assumed to start with the same body burden of Pb at birth. Similarly, all exposed children were assumed to receive the same pattern of nationally representative policy-relevant background (non-air related) exposures throughout the exposure period.

5.1.3.2 Adaptation of IEUBK Model

As discussed above, the IEUBK model was used in batch mode to generate blood Pb estimates at different ages for children exposed from birth in each block or block group for each case study location. Inputs to the IEUBK model included exposure parameters and intake and uptake factor values (see Section 5.1.4 and Appendix I) and the exposure concentrations for each pathway. The input data also include age-specific Pb exposure concentrations for policy-relevant background pathways (e.g., drinking water and non-water diet), which were assumed to be the same for children at all locations.

As described in Section 5.1.1, for the pilot phase assessment, blood Pb metrics were calculated to match the lifetime average and concurrent blood Pb estimates used in deriving the concentration-response (blood Pb-IQ loss) functions that are used in this analysis (described in Section 6.1.2). For this analysis, lifetime average is derived as the average blood Pb level for the exposure interval 6 months through 84 months, and concurrent is derived as the average blood Pb level for the exposure interval 73 through 84 months.[3] To derive these metrics, first IEUBK blood estimates were generated for nine specific ages (see Exhibit 5-4) for each block or block group; these estimates represented the central-tendency blood Pb levels experienced by children of those

[3] The rationale for defining the average blood Pb for ages 73 to 84 months as concurrent reflects the fact that the average age of IQ testing in the Lanphear et al. (2005) study of blood Pb-IQ relationships was approximately seven years (see Chapter 6).

ages in each block or block group. The lifetime average blood Pb metric is derived as the time-weighted average of the blood Pb values for the nine ages. The concurrent blood Pb metric is derived as the average of the last two ages (6 years 4 months and 6 years 10 months).

Exhibit 5-4. Ages for IEUBK-Derived Blood Pb Estimates

Ages (Months) for Output IEUBK Blood Pb Estimates	Age Period (Months) Represented by IEUBK Blood Pb Estimate[a]
9	7 to 12
15	13 to 18
21	19 to 24
31	25 to 36
43	37 to 48
55	49 to 60
67	61 to 72
76	73 to 78
82	79 to 84

[a] Modeling periods run from the first day of the first month to the last day of the last month.

The nine age periods for which the point estimates were obtained using IEUBK were selected to capture those periods of childhood exposure expected to produce significant variability in blood Pb (i.e., exposures occurring under 2 years of age). Consequently, exposure intervals covering the first two years of life (i.e., 7 to 12 months, 13 to 8 months, and 19 to 24 months) are six months long, while the remainder of the simulation period, up to the last year (see below), are simulated with year-long exposure intervals.

The lifetime average and concurrent (average during age 6) estimates were stored in spreadsheets to serve as inputs to the probabilistic population blood Pb model (see below). Each time the Monte Carlo sampling algorithm chose a particular block or block group, the appropriate lifetime average and concurrent blood Pb levels served as the GM values for that block or block group from which the individual blood Pb estimates were derived.

5.1.3.3 Adaptation of Leggett Model

For the pilot phase assessment, rather than using the existing FORTRAN-based Leggett model, a version of the Leggett model implemented in Visual Basic® was developed. The need for this new version of the Leggett model was based on two factors:

1. The need for a version of the model capable of batch-mode runs to support estimating blood Pb levels for a relatively large number of blocks and block groups associated with each case study location (see Section 5.1.1), and
2. The need for the ability to efficiently link the Leggett model (by coding the Leggett model in Visual Basic®) to the multipathway intake and uptake module that was developed for tracking pathway-specific contributions to total Pb uptake.

The development of a Visual Basic® version of the Leggett model involved contacting Dr. Joel Pounds (Pounds 2005) and obtaining the FORTRAN code for the Leggett model (DOS version) together with input files containing the biokinetic parameter values. The biokinetic algorithms of the model were recoded in Visual Basic® and a user interface and multipathway intake and uptake module were added (as noted above). The intake and uptake module was designed to

accept the same exposure, intake, and uptake factors as the IEUBK, and to calculate inhalation and ingestion Pb uptake in the same way as the IEUBK model.

Inhalation pathway uptake was calculated as follows:

$$BRTCRN = RSVOL \times BRETH \times AFLNG$$

where:

BRTCRN = Pb uptake by inhalation (µg/day)
RSVOL = age-specific respiratory volume (m^3/day)
BRETH = modeled exposure concentration (µg/m^3)
AFLNG = inhalation absorption fraction (unitless)

Ingestion pathway Pb uptake was calculated as:

$$EATCRN = FOODUP + H2OUP + SOILUP + DUSTUP$$
$$FOODUP = EAT \times FI$$
$$H2OUP = H2OCNS \times H2OCONC \times H2OA$$
$$SOILUP = SLING \times SOILPB \times SLAF$$
$$DUSTUP = DSTING \times DUSTPB \times DSTAF$$

where:

EATCRN = total uptake through ingestion (µg/day)
FOODUP = uptake from food (µg/day)
EAT = daily Pb intake from non-water diet (µg/day)
FI = GI absorption fraction for diet (unitless)
H2OUP = uptake from drinking (µg/day)
H2OCNS = direct residential drinking water consumption (l/day)
H2OCONC = average Pb concentration in drinking water (µg/day)
H2OA = GI absorption fraction for water (unitless)
SOILUP = daily Pb uptake from soil (µg/day)
SLING = daily soil ingestion (g/day)
SOILPB = average soil Pb concentration (µg/g)
SLAF = absolute GI absorption fraction for soil Pb (unitless)
DUSTUP = daily Pb uptake from indoor dust (µg/day)
DSTING = daily indoor dust ingestion (g/day)
DUSTPB = average indoor dust Pb concentration (µg/g)
DSTAF = absolute GI absorption fraction for dust Pb (unitless)

Model code was tested for logical correctness against the code supplied in Pounds (2005). The model algorithms were tested for numerical stability and precision, and the impacts of different iteration time step durations on predicted blood Pb levels were evaluated. In addition, the results of test exposure scenarios were compared to population blood Pb statistics (as discussed in Section 5.2). Age-specific parameter values used to estimate Pb input parameters were specified

to conform as closely as possible to those used in the IEUBK modeling; that is, so that they yielded the same daily Pb uptake (absorbed dose) from each pathway as went into the IEUBK biokinetic component. Although comprehensive model evaluation of the version of the Leggett model developed for the pilot phase assessment was not completed in time for this report, the Leggett-based exposure analysis has been included here to allow consideration for the use of this particular blood Pb model as part of the full-scale assessment.[4] The results (both exposure and risk) generated using Leggett should, as is the case with all of the results in the pilot phase assessment, be considered provisional and subject to updating as part of the full-scale assessment and in light of additional model evaluation findings.

For the case study blood Pb modeling, exposure concentrations for each block or block group and exposure scenario were input to the model in batch mode. Lifetime (6 to 84 months) blood Pb profiles were generated from the daily Pb model outputs, and lifetime average and concurrent blood Pb estimates were calculated using the appropriate averaging periods (i.e., averaging the individual daily values from 6 to 84 months for the lifetime average metric and averaging daily values for the last year (months 73 to 84) for the concurrent metric). The model iteration time was set at 0.1 day throughout the modeling period. Test runs indicated that modeled daily, concurrent, and lifetime average blood Pb concentrations from age six months onward were identical to those obtained using much shorter time steps.

Outputs from the blood Pb modeling (lifetime average and concurrent blood Pb estimates for each census block or block group) were saved in spreadsheets (as were the IEUBK results) and used as inputs to the probabilistic model, as described in Section 5.1.3.4.

5.1.3.4 Implementation of the Probabilistic Population Blood Pb Model

The probabilistic population blood Pb and health risk model was implemented in an Excel® spreadsheet using spreadsheet functions and Visual Basic for Applications® code. The input sheets for the case studies and exposure scenarios contained one row per block or block group having one or more resident children. The data elements in each row included: the block or block group number; the number of resident children 0 to 7 years old; the exposure concentrations in air, soil, and indoor dust; and the estimated lifetime average and concurrent GM blood Pb concentrations for each block or block group, as derived using either the IEUBK or Leggett models described above. Also included in the table was a weighting factor value for each block or block group that indicated the cumulative proportion of exposed children residing in the block or block groups and all of the blocks or block groups in the preceding rows of the spreadsheet. The model used the cumulative weights to select census blocks or block groups in a population-weighted fashion during the sampling iterations, as discussed below.

The model included a spreadsheet and code algorithm that calculated the approximate proportions of total Pb uptake (absorbed dose) from each pathway for each census block or block group. This algorithm was run before the main simulation and the results were placed in the appropriate rows of the input sheet. Proportional contributions to total Pb uptake were calculated for inhalation, soil ingestion, indoor dust ingestion, and combined policy-relevant background

[4] Evaluation of the Leggett model as applied for this exposure and risk assessment is continuing beyond completion of this report. Additional findings of that evaluation will be conveyed by separate cover.

pathways. The model calculated indoor dust contributions separately for the intercept of the indoor dust model (assumed to be policy-relevant background from Pb paint and other exposure sources not explicitly modeled for the pilot phase), and for air- and soil-related terms in the AGG model, as appropriate. The source contributions to total Pb uptake were calculated deterministically for each block or block group using the same intake and uptake factors as used in the blood Pb models. It is important to note that the model calculates the contributions of the various exposure pathways to total Pb uptake for each block or block group, *not* for individuals. Thus, the results discussed in Section 5.3.1 represent only approximate central tendency estimates of pathway-specific contributions associated with various blood Pb percentile estimates from the probabilistic model.

The probabilistic blood Pb/IQ loss model was implemented by sampling blood Pb levels 10,000 times in a population-weighted manner from the blocks and block groups included in each case study, adding a lognormally distributed term representing individual variability to each of the 10,000 GM lifetime average and concurrent blood Pb values, and extracting population blood Pb statistics from the resulting individual blood Pb estimates. Individual variability was modeled by sampling from a 0 to 1 uniform variate and using the Excel® LOGINV function to generate a lognormal variate with the appropriate block or block group GM, and a defined population GSD (see Section 5.1.4.3).

The pathway contributions to total Pb uptake associated with each block or block group were also carried through the analysis. Each iteration of the model was implemented on a single row of a spreadsheet, and standard spreadsheet functions were used to extract population blood Pb statistics (percentile values) after the sampling was complete. For the primary Pb smelter and near roadway case studies, diet and drinking water were the only policy-relevant background pathways. In the secondary Pb smelter case study, diet, water, and an estimated 15 μg/g natural soil exposure were identified as policy-relevant background pathways. As noted in Chapter 4, the "AGG" models used to estimate house dust concentrations include intercepts that presumably represent the contribution to house dust from paint and other indoor sources of exposure. When the AGG (Air-only) model is used, the intercept contributes 60 μg/g to the total estimated house dust Pb levels. When the AGG (Air + Soil) version was used, the house dust intercept concentration was 30.5 μg/g. In all of the case studies, the "intercept" contributions were included as being associated with policy-relevant sources because it was not possible to unconditionally rule out these elements of the house dust Pb models as being indirectly related to air Pb levels.

5.1.4 Inputs to the Blood Pb Models

5.1.4.1 Exposure Concentration Estimates for Policy-Relevant Sources

Exposure concentrations in media impacted by policy-relevant sources were estimated for each census block or block group in each of the three case studies as described in Chapter 4. In the primary Pb smelter case study, two scenarios were evaluated: the current conditions scenario, representing estimates of exposure based on recent site-related data and modeling; and the attainment scenario, wherein all of the exposure concentration estimates were the same as under current conditions, with the exception of seven census blocks near the facility, where all

estimated quarterly averaged ambient air concentrations above the NAAQS (1.5 μg/m^3) were reduced to the level of standard and annual average air concentrations then derived from these adjusted quarterly averages. Exposure concentrations that were estimated from the ambient air concentrations (for example, indoor dust Pb concentrations) were likewise reduced in these seven high-exposure blocks. In the secondary Pb smelter case study, two sets of soil and indoor dust exposure concentrations were developed, one based on EPA's MPE methodology (see Section 4.2.3) and one based on a hybrid method in which the MPE estimates were scaled to be more congruent with environmental sampling data from similar facilities.

For the inhalation exposure modeling, as described above, each block or block group was characterized by an estimated inhalation exposure concentration and by an estimated number of resident children. The inhalation exposure concentrations together with exposure concentrations for other key media (e.g., soil, indoor dust, diet) and numbers of children exposed in each block or block group define the distribution of population exposures that go into the calculation of individual blood Pb levels in the probabilistic model. Exhibit 5-5 shows the cumulative population distribution of inhalation pathway exposure concentrations for the primary Pb smelter case study under the current conditions and attainment scenarios.[5] Important features of this graph include the wide range of exposures (note that the y-axis is logarithmic) and the fact that the great majority of the exposed populations experience relatively low inhalation exposures (less than approximately 0.015 μg/m^3). Only a small "tail" of the distribution (a small proportion of the exposed children) experiences much higher exposures. Note that the cumulative exposure distributions are identical for the two exposure scenarios, except for the seven highest exposure blocks, as discussed above.

[5] The cumulative distribution is derived by ranking the census blocks from lowest to highest based on inhalation exposure concentration and plotting the exposure concentrations versus the number of children exposed at or below that concentration.

Exhibit 5-5. Population Distribution of Inhalation Exposure Concentrations for the Primary Pb Smelter Case Study

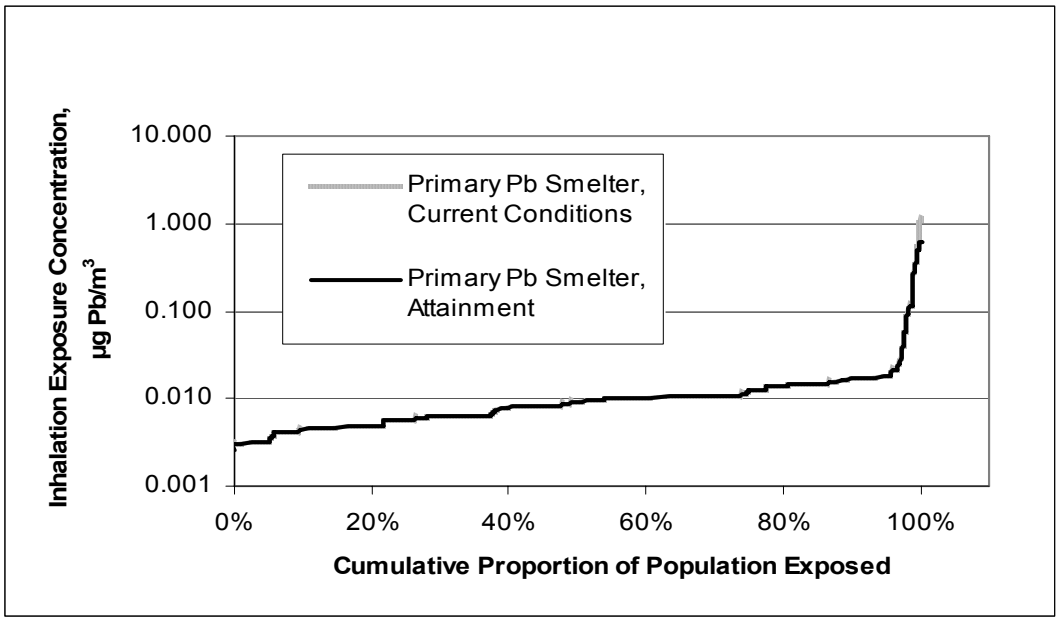
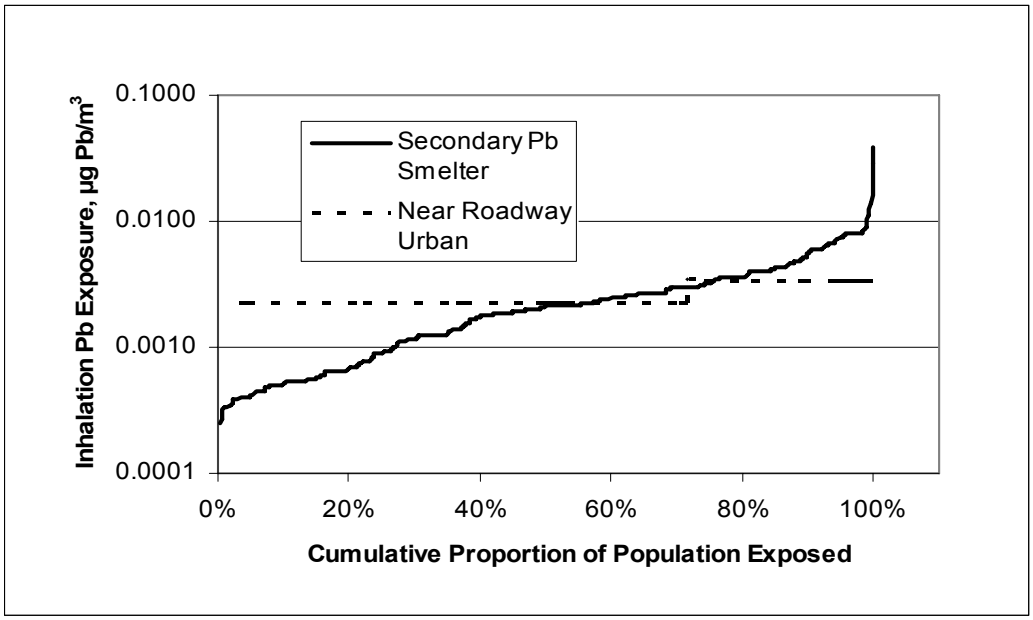


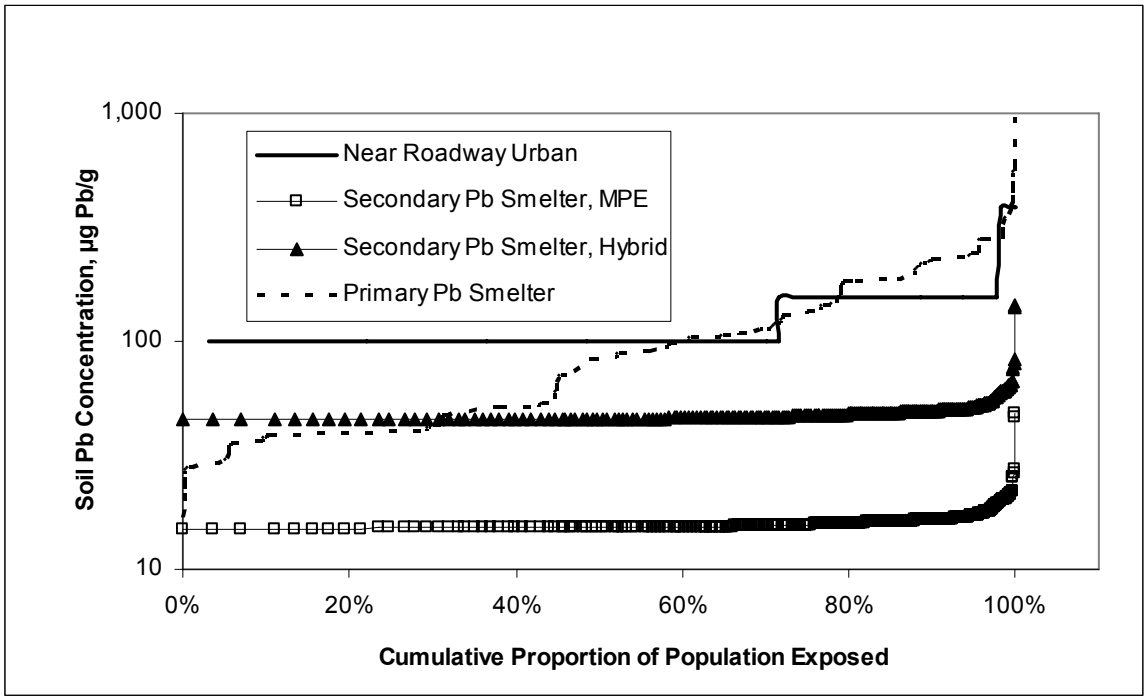
Exhibit 5-6 shows the population distributions of inhalation exposure concentrations for the secondary Pb smelter and near roadway urban case studies. For the secondary Pb smelter case study, the basic shape is the same, although less extreme, than that seen for the primary Pb smelter. Estimated inhalation exposures are very low in the secondary Pb smelter case study, and increase gradually, for most of the population. There is a small "hook" at the high end of the curve representing a relatively small proportion of the population in census blocks near the facility experiencing higher exposures. In contrast, the exposure distribution for the near roadway urban case study is a step function; this pattern shows the single-value estimated inhalation concentrations for census blocks in the 75 to 200 meters (m) and 0 to 75 m bands, respectively.

Exhibit 5-6. Population Distribution of Inhalation Exposure Concentrations for the Secondary Pb Smelter and Near Roadway Urban Case Studies



The distribution of soil exposure concentrations for all three case studies (including both the MPE and hybrid approaches for soil for the secondary Pb smelter case study) is shown in Exhibit 5-7. In this case, it is the exposure concentration distributions for the secondary Pb smelter case study that are flatter, with more extreme high-end "tails" than the primary Pb smelter. The distribution of exposure concentrations for the near roadway urban case study is again a step function, in this case with three "steps" representing the soil concentrations in the 75 to 200 m, 12 to 75 m, and 0 to 12 m exposure "bands," respectively.

Exhibit 5-7. Population Distribution of Soil Exposure Concentrations for the Three Case Studies



The population distributions of indoor dust exposure concentrations are shown for the primary Pb smelter case study in Exhibit 5-8, and for the secondary Pb smelter and near roadway urban case studies in Exhibit 5-9. The pattern of indoor dust exposure concentrations closely matches the pattern seen for inhalation exposure concentrations, a consequence of the fact that the air Pb concentration in air is a major determinant of indoor dust Pb concentrations estimates by the AGG models (described in Chapter 4 and 6). The estimated indoor dust concentrations for the secondary Pb smelter and near roadway urban case studies also follow the general patterns seen for the other media, again because of the correlation introduced between air, soil, and indoor dust exposure by the use of the AGG model. The skewed nature of the exposure concentration distributions for the primary and secondary Pb smelter case studies strongly affects the nature of the estimated blood Pb distributions, as will be discussed in Section 5.4.

Exhibit 5-8. Population Distribution of House Dust Exposure Concentrations for the Primary Pb Smelter Case Study

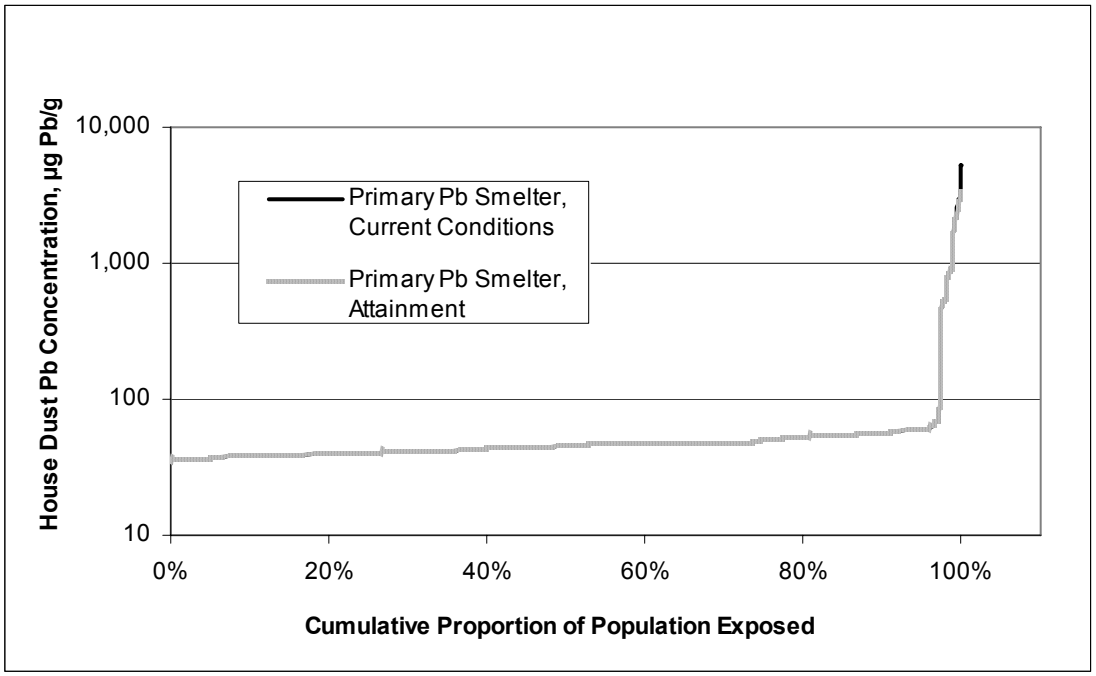
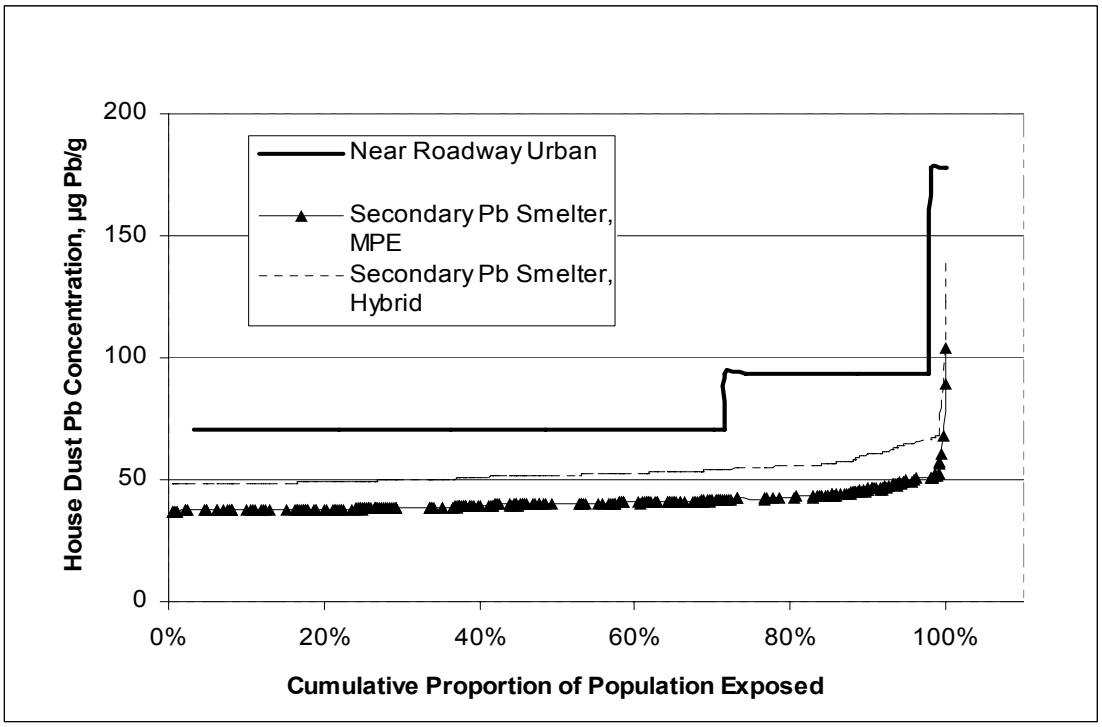


Exhibit 5-9. Population Distribution of House Dust Exposure Concentrations for the Secondary Pb Smelter and Near-Roadway Urban Case Studies



5.1.4.2 Policy-Relevant Background Exposure Pathway Concentrations and Pb Intake Estimates

As noted above, the exposure concentrations and Pb intake from policy-relevant background pathways (drinking water and non-water diet) were also parameter inputs to the blood Pb model. For the purposes of the pilot phase assessment, as stated in Section 5.1.1, exposure through these pathways was categorized as policy-relevant background, and all of the exposed children were assigned the same age-specific estimates.

All of the exposed populations were assigned the same Pb concentration in drinking water. While there is a rather large amount of data in the literature, in many cases, the data are from "first-draw" samples, non-random ("priority") samples, or from communities where Pb levels were known to be elevated. After reviewing the literature, the average drinking water concentration was estimated to be 4.61 µg/L, based on data from two recent studies of residential water concentrations in homes and apartments in the United States and Canada (Moir et al. 1996, Clayton et al. 1999). The range of values seen in these studies (0.84 to 16 µg/L) was considered to be representative of randomly sampled residential water in houses constructed since Pb pipe and solder were banned for residential use. The selected value is close to the "default" value (4.0 µg/L) provided with the IEUBK model (USEPA 1994). Much higher values have been encountered in homes with Pb piping and/or very corrosive water.

In addition to drinking water, it is expected that young children will be exposed to Pb in the foods they consume. In this assessment, all exposed children were assumed to receive the age-specific estimates of dietary Pb intake developed by EPA's Office of Solid Waste and Emergency Response (USEPA 2006h). EPA developed these estimates by analyzing food consumption data from the third National Health and Nutrition Examination Survey (NHANES III) conducted by the National Center for Health Statistics (CDC 1997), and food residue data from the U.S. Food and Drug Administration's (FDA) Total Dietary Study (USFDA 2001). The daily intake values shown in Exhibit 5-10 are considerably lower than those developed using the same methodology in the 1980s and 1990s. Pb concentrations in food have decreased dramatically since the prohibition of Pb solder in food containers in 1982 (USEPA 2006b, Section 3.4).

Exhibit 5-10. Summary of Non-Water Dietary Pb Intake Estimates

Age Category (months)	Updated Dietary Pb Intake Estimate (µg/day)
0 to 11	3.16
12 to 23	2.6
24 to 35	2.87
36 to 47	2.74
48 to 59	2.61
60 to 71	2.74
72 to 84	2.99

There is some potential for double-counting of water and dietary Pb intake because some food categories (e.g., baby formula, soup) are prepared using domestic water. As discussed in Section 5.1.4.3, this double-counting is minimized by limiting the estimated intake of domestic water to "direct ingestion" (i.e., consumption directly from the tap).

5.1.4.3 Factors Determining Pb Exposure, Intake, and Uptake

As discussed previously, there is a range of model inputs that govern how the exposure concentrations are converted to absorbed Pb dose (uptake). These factors represent the physiological and behavioral characteristics of the exposed population and the chemical and physical properties of the exposure media that govern exposure and absorption by inhalation and ingestion.

Because substantial data have become available since the IEUBK default values were last updated, a literature review was conducted to identify and evaluate recent information related to Pb exposures, absorption, and bioavailability (USEPA 2006b). Experts in OSWER and OAQPS were also consulted in an effort to derive exposure, intake, and uptake values for the pilot phase assessment. Exhibit 5-11 presents the parameter values that were selected as inputs to the IEUBK in the pilot phase assessment.

A number of the values in Exhibit 5-11 differ from the suggested default values in the most current version of the IEUBK (USEPA 2005a). Children's daily ventilation rate estimates were based on the ICRP Publication 89 (ICRP 2002). Pending a re-analysis pertaining to the child respiratory absorption fraction values, the values used in the pilot phase for the primary and secondary Pb smelter case studies were those developed for point source locations for the assessment described in the 1990 Staff Paper (USEPA 1990b; USEPA 1989). For the near roadway urban case study, the value used in the pilot phase assessment was 0.32, which falls within the range of 0.25 to 0.45 estimated in the past for areas not influenced by point sources (USEPA 1989), and is the IEUBK default.

Exhibit 5-11. Input Parameter Values for the IEUBK Model

Parameter	IEUBK Parameter Name	Parameter Value							Basis/Derivation
		IEUBK Default Age Ranges (Years)							
		0.5 to 1	1 to 2	2 to 3	3 to 4	4 to 5	5 to 6	6 to 7	
Inhalation									
Daily ventilation rate (m3/day)	Ventilation rate	4	5.1	6	6.8	7.8	8.8	10	ICRP (2002), with interpolation for intermediate ages.
Absolute inhalation absorption fraction (unitless)	Lung absorption	0.42 (Primary, secondary smelter), 0.32 (Near Roadway Urban)							Smelter value: USEPA (1989), Appendix A IEUBK value: USEPA (1989), central tendency value from analysis of deposition data.
Indoor air Pb concentration	Indoor air Pb concentration (percentage of outdoor)	100%							Time spent indoors/outdoors is not considered because the input air concentrations are already long-term weighted averages of indoor and outdoor concentrations.
Time spent outdoors	Time spent outdoors (hours/day)	Not used							
Drinking Water Ingestion									
Water consumption (L/day)	Water consumption (L/day)	0.34	0.31	0.31	0.33	0.36	0.39	0.42	Based on value for infants, 1 to 3 yr olds, 1 to10 yr olds (with trend lines used to interpolate intermediate age ranges) (USEPA 2002c).
Water Pb concentration (µg/L)	Pb concentration in drinking water (µg/L)	4.61							GM of values reported in studies of United States and Canadian populations (residential water) (Moir et al. 1996, Clayton et al. 1999, as cited in USEPA (2006b), Section 3.3 Table 3-10).
Absolute absorption (unitless)	Total percent accessible	50% (Single value used across all age ranges)							Assumed similar to dietary absorption (see "Total percent accessible" under Ingestion-Diet below).

Exhibit 5-11. Input Parameter Values for the IEUBK Model

Parameter	IEUBK Parameter Name	Parameter Value							Basis/Derivation
		IEUBK Default Age Ranges (Years)							
		0.5 to 1	1 to 2	2 to 3	3 to 4	4 to 5	5 to 6	6 to 7	
Diet									
Dietary Pb intake (µg/day)	Daily Pb intake (µg/day)	3.16	2.6	2.87	2.74	2.61	2.74	2.99	Estimates based on (a) Pb food residue data from U.S. Food and Drug Administration Total Diet Study (USFDA 2001), and (b) food consumption data from NHANES III (CDC 1997).
Absolute absorption (unitless)	Total percent accessible	50%							Alexander et al. (1974) and Ziegler et al. (1978), as cited in USEPA 2006 (Section 4.2.1). These two dietary balance studies suggest that 40-50% of ingested Pb is absorbed by children (2 weeks to 8 years of age).
Soil/Indoor Dust Ingestion									
Soil/dust weighting factor (unitless)	Soil/dust ingestion weighting factor (percent soil)	45%							This is the percent of total ingestion that is soil. Value reflects best judgment and consideration (results published by van Wijnen et al. (1990), as cited in (USEPA 1989). The van Wijnen et al. study looked at tracer studies of ingestion rates for rainy days and non –rainy days. It was assumed that rainy days were associated with all soil ingestion and non-rainy days were associated with a combination of soil and dust with the delta representing soil.
Total dust + soil ingestion (mg/day)	Amount of soil/dust ingested daily (mg)	85	135	135	135	100	90	85	USEPA 1989, which was based on multiple studies focusing on children.
Absolute gastrointestinal absorption (soil and dust) (unitless)	Total percent accessible	· Primary Pb smelter case study: 0.48 for soil and 0.26 for dust · Secondary Pb smelter and near roadway: 0.30 for both soil and dust							· Site specific absorption factors for soil and indoor dust were derived for the primary Pb smelter case study using relative bioavailability (RBA) estimates generated based on swine studies involving soil and dust samples collected in the study area (Casteel et al. 2005). These RBAs were converted to absolute bioavailability factors (i.e., total percent accessible values) by applying the absolute bioavailability factor for the control material (Pb acetate water solution also fed to the animals). · Secondary Pb smelter and near roadway values: USEPA (1989) reflects evidence that Pb in dust and soil is as accessible as dietary Pb and that dust/soil ingestion may occur away from mealtimes (resulting in enhanced absorption relative to exposure during meal events).

Exhibit 5-11. Input Parameter Values for the IEUBK Model

Parameter	IEUBK Parameter Name	Parameter Value							Basis/Derivation
		IEUBK Default Age Ranges (Years)							
		0.5 to 1	1 to 2	2 to 3	3 to 4	4 to 5	5 to 6	6 to 7	
Other									
Maternal blood Pb (µg/dL)	Maternal blood Pb concentration at childbirth, µg/dL	1.94							NHANES IV, national GM for adult women – all nationalities (CDC 2004).

Estimated children's direct water ingestion values were interpolated from values in EPA's Children-Specific Exposure Factors Handbook (USEPA 2002c); the GI absorption fraction of Pb from water (and diet) was retained at the IEUBK default value of 50 percent, and is consistent with OAQPS previous analyses of Pb uptake (USEPA 1989). As noted in Section 5.1.4.2, age-specific dietary intake values for Pb were revised to reflect the latest analyses of FDA and NHANES III data on food consumption pattern and Pb residue levels. (USEPA 2006h).

Age-specific soil and indoor dust ingestion rates were left at the IEUBK default values. Similarly, the weighting factor for soil and indoor dust ingestion was also left at 45 percent soil, despite limited data supporting this specific value (USEPA 1989; USEPA 1994). The impacts of changes in the weighting factor and other variables related to soil and dust ingestion were investigated through sensitivity analyses, as discussed in Section 6.3.

Casteel et al. (2005) evaluated the GI absorption of Pb and other metals from soil samples taken from the primary Pb smelter study area in juvenile swine. Results of these experiments were used to derive estimates of absolute GI absorption fractions (the IEUBK inputs are called "Percent Available") of 0.48 (48 percent) for soil and 0.26 (26 percent) for indoor dusts. For the other case studies, the IEUBK generic default value for GI absorption of Pb from soil and indoor dust (0.30, or 30 percent) was used. This value is generally consistent with more recently reported values, although estimates vary widely. As was the case with the soil-indoor dust weighting factor, the impacts of changes in absorption fractions for soil and indoor dust were investigated in sensitivity analyses.

For the case study blood Pb modeling, the IEUBK default value for maternal blood Pb level was updated using data from the most recent NHANES survey. NHANES IV data from 2002 to 2004 indicate that the GM blood Pb value for reproductive age women has dropped to about 1.94 µg/dL (Maddaloni et al. 2005).

5.1.4.4 Inter-Individual Variability

The last major input to the probabilistic blood Pb model is the GSD reflecting individual variability in response to Pb exposures. A GSD value of 1.6 has been used as the central tendency estimate for individual blood Pb variability in the pilot phase assessment. This value

was provided by EPA (USEPA 1994) based on the distribution of blood Pb levels measured in children exposed to smelter emissions at the Midvale, UT, smelter (White et al. 1998). Subsequent studies suggest that this is a reasonable central tendency GSD for populations of children living in relatively small, defined areas where the sources and relative importance of different exposure pathways and media are similar across the exposed population. The selection of this value is supported by the fact that exposure concentrations used in this analysis are expected to be relatively consistent within each block and block group. Similarly, using a GSD reflecting the distribution of blood Pb levels for children residing near a primary Pb smelter (i.e., a dominant Pb source such as the Midvale, UT, smelter) seems appropriate for modeling blood Pb levels near primary and secondary Pb smelters. It is also worth noting that the pre-remediation blood Pb GSD for children participating in the Baltimore Urban Pb Soil Abatement Project was estimated at 1.5 (White et al. 1998), which adds support to using the GSD of 1.6 for the near roadway urban case study, which can be interpreted as being somewhat similar to a general urban scenario in terms of Pb exposure.

Recent surveys of blood Pb levels in children at the national-level (NHANES IV data for years 1999 to 2002) have found GSD values in the range of 2.03 to 2.23 (CDC 2004). These GSDs, which are considerably larger than the values used in the pilot phase assessment, likely reflect the fact that, while blood Pb levels for the majority of children in the United States have decreased significantly over the last one to two decades, a small fraction of children still retain relatively elevated blood Pb levels due to continued exposure to Pb paint and other artifact sources. Consequently, as the median and mean blood Pb levels have dropped, the extreme upper tail of the distribution is still somewhat anchored by these high-exposure children, resulting in an increased GSD for the overall population (CDC 2004). It is expected that the variation in exposure concentrations within the case study blocks or block groups is much smaller than that seen in these national population surveys, further supporting the use of the smaller GSD values.

The small amount of publicly available blood Pb data from children living near the primary Pb smelter site (see Exhibits 3-3 and 3-4), while not sufficient by itself to establish reliable GSD estimates for this group, appear to be consistent with a low to moderate GSD value (less than 2). This conclusion was based on a review of summary blood Pb data from the site (numbers of samples within specified concentration ranges); individual blood Pb measurements were not available. The impact of assuming different GSD values has a major impact on the blood Pb distribution results; the results of sensitivity analysis involving this variable are provided in Section 6.4.

5.2 Blood Pb Model Performance Evaluation

5.2.1 Evaluation Versus General Population and Site-Specific Blood Pb Measurements

As noted previously, the IEUBK and Leggett blood Pb models were evaluated with regard to their predictions relative to each other and relative to population blood Pb statistics from well-studied populations believed to have experienced exposures similar to those estimated for populations in the case studies. In addition, the upper percentile blood Pb estimates from the primary Pb smelter case study were compared to children's blood Pb sampling results taken near the facility in 2001. Exhibit 5-12 provides a summary of these analyses.

Exhibit 5-12. Summary of Blood Pb Model Evaluation

Case Study	Description of Performance Evaluation	Results of the Performance Evaluation	Implications for Overall Analysis
All Case Study Populations	For both the lifetime average and average at age 6 (concurrent) blood metrics, the 50th percentile (i.e., median) value for each of the three study populations was compared to the GM value for the 0 to 5 year old age group from NHANES IV (2001-2002) (CDC 2004).	Modeled median lifetime average blood Pb modeled with IEUBK ranged from 1.2 to 1.9 µg/dL (average 1.7) for the three case studies. Median concurrent blood Pb modeled with IEUBK ranged from 1.0 to 1.4 µg/dL (average 1.2) for the three case studies. Leggett median lifetime average blood Pb ranged from 0.8 to 1.2 µg/dL (average 1.0) for the three case studies Median concurrent blood Pb ranged from 0.5 to 0.8 µg/dL (mean 0.7) for the three case studies.	The relatively close match between the modeled median lifetime average levels from IEUBK and the national GM value from NHANES IV (for children ages 0 to 5 years old) suggests that the IEUBK model with the given set of exposure, intake, and uptake factors is neither significantly over- or underestimating exposures for the study population. Because concurrent blood Pb in this analysis is defined as the average blood Pb from age 6 to 7 years, when blood Pb levels are known to decline from values seen in younger children, the median concurrent IEUBK values appear to also generally consistent with population data. The median blood Pb estimates are insensitive to the relatively small number of high-exposure block groups in the primary and secondary Pb smelter case studies; thus the lower exposure experienced by the large majority of the exposed populations are dominating this metric. The Leggett model, however, with the selected exposure, intake, and uptake factors, appears to be underestimating the GM blood Pb statistics somewhat, compared to the national population. The reason for this is not clear, although it is possible that the Pb exposure levels of the NHANES population are actually lower than the combined policy-relevant background and policy-relevant source levels used in the case studies. Recent studies (USEPA 2006b) indicate that the soil and house dust Pb concentrations seen in all but the most exposed blocks in the primary Pb smelter case study fall well within the range of typical national values.

Exhibit 5-12. Summary of Blood Pb Model Evaluation

Case Study	Description of Performance Evaluation	Results of the Performance Evaluation	Implications for Overall Analysis
Primary Pb smelter	Comparison of *upper-bound end of range* of modeled blood Pb levels against the set of site-specific measured blood Pb levels collected for children <6 years of age in 2002	58 site-specific blood Pb level measurements from children ages 0 to 72 months old in 2001 yield the following percentiles: > 95th is 20 to 29 µg/dL > 90th is 10 to 19 µg/dL > 50th is 0 to 9 µg/dL Percentile results generated from our site-specific modeling (Leggett and IEUBK combined): > 99.9th% is 13 to 29 µg/dL > 99.5th% is 6 to 17 µg/dL > 99th% is 4 to 11 µg/dL > 95th% approaches 5 µg/dL	It should be noted that the blood Pb levels measured at the primary Pb smelter location correspond to highly-contaminated areas and were focused on children believed to be at risk of adverse effects. Thus, it is to be expected that the percentile values seen in the blood Pb data should correspond to higher percentile estimated blood Pb values at the primary Pb smelter location, because the latter include many children far from the site, who experience relatively low exposures. Numerically, the highest percentile values measurable from the data are in the same range as the highest percentiles in the modeled data. These results indicate that the model-predicted higher percentile blood Pb values from the secondary Pb case study are generally consistent with the high-end measured values.

With regard to the median blood Pb levels, the IEUBK lifetime average blood Pb predictions from the three case studies rather closely match the GM estimate from the NHANES survey. As expected, the IEUBK concurrent blood Pb estimates are somewhat lower than the NHANES GM values, likely because the former data are derived from younger children, who are expected to have somewhat higher blood Pb levels. Median blood Pb estimates from the Leggett model are systematically lower than those from the IEUBK, and lower than the NHANES GM.[6] The reason for this difference is not clear, although it is possible that the NHANES population may have experienced lower exposures than the exposed populations in the case studies; this would imply that the IEUBK values were slight overestimates and would support the argument that the Leggett model "underestimates" were not as large as they appear.

The upper percentile blood Pb estimates from the IEUBK and Leggett models appear to be in the same general range as the upper percentile data from the blood Pb samples taken near the facility in 2001. This suggests that the models are not dramatically overestimating blood Pb levels for this subpopulation that would be most strongly affected by Pb exposures from policy-relevant sources.

[6] As noted earlier, the trend between IEUBK-based and Leggett-based blood Pb levels generated for the pilot phase is opposite to the trend reported in the Pounds and Leggett (1998) study. This apparent contradiction elevates the importance of model evaluation and performance evaluation related to the IEUBK and Leggett models, which was not completed in time for inclusion in this report. Additional findings from that evaluation will be conveyed by separate cover.

5.2.2 Stability of Percentile Blood Pb Estimates

In addition to the performance evaluation of the blood Pb model described above, the stability of the various blood Pb percentile estimates was also evaluated. Because these estimates are probabilistic, they can be expected to vary as the model is run multiple times or with different numbers of iterations.

To examine the variability in the percentile blood Pb estimates, the probabilistic blood Pb model for the primary Pb smelter current conditions scenario was run 100 times, with 10,000 iterations per run. The distributions of the blood Pb percentile estimates are summarized in Exhibit 5-13.

Exhibit 5-13. Distributions of Blood Pb Percentile Estimates, 100 Repetitions of the Probabilistic Blood Pb Model

Blood Pb Percentile	Mean Estimate of Statistic, µg/dL	Standard Deviation	Coefficient of Variation	Average Difference[a]	Average Difference/Mean Estimate
99.9th	20.9	2.02	10%	2.24	11%
99.5th	11.2	0.87	8%	0.98	9%
99th	7.6	0.41	5%	0.46	6%
95th	3.7	0.065	2%	0.074	2%
90th	2.9	0.034	1%	0.038	1%
75th	2.0	0.018	1%	0.020	1%
Mean	1.7	0.018	1%	0.021	1%
Median	1.3	0.010	1%	0.012	1%
25th	0.93	0.007	1%	0.008	1%
First	0.40	0.007	2%	0.008	2%

[a] Average difference = expected value of difference between two successive simulation estimates (10,000 iterations).

The results indicate that the precision of the model is quite good for the mid-range blood Pb percentile estimates; the coefficient of variation (the standard deviation divided by the mean, multiplied by 100) is one percent or less for the median and all percentiles between the 25th and the 90th, inclusive. Above the 95th percentile, the precision of the estimates drops rapidly, such that the coefficient of variation is about five percent for the 99th percentile, 8 percent for the 99.5th percentile, and 10 percent for the 99.9th percentile. These statistics should be kept in mind when interpreting the blood Pb distribution results presented in Section 5.4.

5.3 Limitations and Uncertainties in the Human Exposure Assessment and Blood Pb Modeling

A number of factors affect the degree of uncertainty associated with the human exposure assessment and blood Pb modeling. These factors, discussed in the following subsections, include the estimated exposure concentrations associated with policy-relevant sources and policy-relevant background, the exposure, intake, and uptake factor values, the differences in the blood Pb model themselves, the approach used to characterize individual variability, and the demographics of the exposed population.

5.3.1 Exposure Concentrations and Intake Estimates

Exposure concentration estimates have a varying affect on the results of the blood Pb modeling, depending on the extent to which they contribute to total Pb uptake (absorbed dose). Exhibit 5-14 summarizes the general pattern of how policy-relevant sources and policy-relevant background pathways contribute to total Pb intake for different percentile estimates of blood Pb for the three case studies. As discussed in Section 5.1.3.4, the values in the exhibit are the Pb uptake contribution estimates for the block or block group whose geometric mean blood Pb was used to estimate the specified individual blood Pb percentiles. In addition, the values represent the averages of the pathway contributions calculated from the distributions of lifetime average and concurrent blood Pb percentile values, and the averages of the percentile values calculated from the different exposure scenarios for each case study location. For the most part, the pathway contributions to Pb uptake do not differ substantially across the exposure scenarios or blood Pb metrics. Exceptions to this pattern are discussed in Sections 6.1 and 6.2. Detailed tables of pathway contributions are provided in Appendix J.

Exhibit 5-14. Average Contributions of Policy-Relevant Sources and Policy-Relevant Background Pathways to Overall Pb Uptake

Blood Pb Percentile Estimate	Primary Pb Smelter (Current Conditions)		Secondary Pb Smelter		Near Roadway Urban	
	Policy-Relevant Background	Policy-Relevant Sources	Policy-Relevant Background	Policy-Relevant Sources	Policy-Relevant Background	Policy-Relevant Sources
Maximum	2%	98%	68%	32%	20%	80%
99.9th	2%	98%	65%	35%	44%	56%
99.5th	4%	96%	67%	33%	44%	56%
99th	4%	96%	66%	34%	44%	56%
95th	22%	78%	68%	32%	44%	56%
90th	34%	66%	67%	33%	44%	56%
75th	45%	55%	67%	33%	44%	56%
Median	56%	44%	68%	32%	35%	65%
25th	56%	44%	68%	32%	35%	65%
1st	54%	46%	68%	32%	44%	56%

As illustrated in Exhibit 5-14, the policy-relevant background pathways tend to contribute a relatively large proportion of total Pb uptake for individuals with lower blood Pb levels, and the importance of policy-relevant background pathways then drops with increasing blood Pb. This pattern is especially pronounced for the primary Pb smelter because air, soil, and indoor dust exposures are relatively low for the bulk of the exposed population, and then become extremely high for the highest exposure blocks and block groups. For the secondary Pb smelter case study, the contribution of policy-relevant background exposure pathways (diet, water, and background soil) remains above 50 percent for all of the exposure percentiles. In the near roadway urban case study, the contribution from policy-relevant background pathways is lower for less exposed groups than for the other case studies, but it is noted that 100 μg/g "urban background" soil Pb is included as "policy-related" in this tabulation.

Overall, among the pathways for which individual contributions were derived (see Appendix J), the inhalation pathway contributes the least to total Pb uptake, ranging from 0.01 percent in the lowest exposure blocks for the secondary Pb smelter case study to 4.3 percent in the highest exposure blocks for the primary Pb smelter. As noted previously, however, the impact of ambient air is assumed to be amplified by its impact on indoor dust Pb concentrations through the AGG models. The proportional contribution from indoor dust to total Pb intake is consistently in the range of 25 to 35 percent in the secondary Pb smelter and near roadway case studies; however, for the primary Pb smelter case study, it ranges from about 10 percent in the low-exposure blocks and block groups to 87 to 88 percent in the highest exposure block groups. As noted previously, the contribution from house dust includes an "intercept" which accounts for a large, sometimes dominant, proportion of the total dust contribution in census blocks with low air and/or soil concentrations.

Soil exposure contributes about 6 to 11 percent of total Pb intake in the secondary Pb smelter case study, 30 to 53 percent in the near roadway urban case study, and from about 7 to 63 percent in the primary Pb smelter case study. Because of the relatively widespread soil contamination at the primary Pb smelter location, soil contributions to total Pb uptake tend to be greater for individuals in the lower blood Pb percentiles, whose air and indoor dust exposure concentrations are relatively low.

The factors that contribute to uncertainties in the exposure concentrations are discussed below for the primary Pb smelter, the secondary Pb smelter, and the near roadway urban case study locations:

Primary Pb Smelter

- Emissions estimates from the facility processes were based on maximum permitted emissions. This may not appropriately represent current emissions (could be higher or lower). In addition, source characteristics of area and volume sources are uncertain, and emissions from roadways not directly adjacent to the facility were not included.

- The particle size distributions of the emissions from the primary Pb smelter are uncertain. These distributions impact air and dust concentrations (but not soil, as soil concentrations were estimated from measurements for this case study).

- Inhalation exposure concentrations for the 0 to 7 age group were estimated from the pilot phase assessment ambient air concentrations using location-specific ratios of ambient concentration to exposure concentrations for the 0 to 4 age group from the NATA national-scale assessment (see Section 4.1.2.2). This contributes to uncertainty, particularly because the 5 to 7 age group is school-aged and may have different activity patterns from the 0 to 4 group.

- Soil concentration measurements were available out to approximately 2.5 km from the facility. Beyond that distance, the regression equation used to calculate soil concentration was extrapolated, leading to some uncertainty.

- To generate air concentrations for the attainment scenario, all quarterly average air concentrations exceeding the NAAQS were set to the attainment level (1.5 $\mu g/m^3$). All other air concentrations were kept at the current conditions modeled concentrations. This is a simplification of how air concentrations would be affected by emission changes necessary to meet the NAAQS, and likely results in an overestimation of air concentrations for the attainment scenario.

- Soil concentration estimates for the attainment scenario are the same as those used for the current conditions scenario. This may contribute to a more conservative soil concentration, although soil remediation activities make this difference difficult to predict.

Secondary Pb Smelter

- Facility Pb emissions estimates were based on stack tests from 1997, 1999, and 2000. Current emissions may be different. In addition, estimating annual emissions from single-day stack tests may not appropriately account for true trends in emissions over an entire year.

- The particle size distributions of the emissions from the secondary Pb smelter are uncertain and were based on general process information from AP-42. These distributions lead to uncertainties in air, soil, and dust concentrations.

- The meteorological data used for the air dispersion modeling were not local, but from a nearby station. The use of these data result in less precision in directional estimates of air, soil, and dust concentrations, which is consistent with the results of the comparison to air monitoring data at specific locations (see Section 4.2.2.4).

- Inhalation exposure concentrations for the 0 to 7 age group were estimated from the pilot phase analysis ambient air concentrations using location-specific ratios of ambient concentration to exposure concentrations for the 0 to 4 age group from the NATA national-scale assessment (see Section 4.1.2.2). This leads to uncertainty, particularly because the 5 to 7 age group is school-aged and may have different activity patterns from the 0 to 4 group.

- Soil concentrations of Pb were calculated assuming the deposition (derived from the recent emissions estimates [see Appendix L]) has been constant for the last 37 years (facility operating time). This is likely not true, and it is quite possible this results in an underestimation of soil concentrations. In addition, because of lack of site-specific data, modeled soil concentrations are uncertain. In a comparison to measured soil concentrations at a similar facility, the modeled results were much lower (Section 4.2.3.1). Thus, a second soil scenario was created with soil concentrations scaled up by a factor of three. There is still uncertainty over whether this scaled-up set of Pb soil concentrations is an appropriate upper-bound because the measurements were not taken at the same location.

Near Roadway Urban Case Study

- Compared to the other case studies, there were relatively little site-specific data available upon which to base estimates of air, soil, and indoor dust concentrations. These estimates are therefore very uncertain, and only indicative of potential exposure magnitudes.

5.3.2 Exposure, Intake, and Uptake Parameters and Individual Variability

The exposure, intake, and uptake factors used in this analysis, particularly those affecting exposure pathways accounting for a large proportion of total Pb uptake, may strongly affect the results of the blood Pb modeling for specific individuals. Section 6.4 presents the results of a sensitivity analysis in which the variables evaluated were selected based on the relative importance of the various pathways. The variables evaluated included dietary and drinking water GI absorption fractions, soil and dust ingestion rates, and soil and GI absorption fractions. As discussed in Chapter 6, variation of these values over credible ranges tended to have much less impact on the blood Pb distributions than differences in the blood Pb models that were used, and between the blood Pb metrics (lifetime average, concurrent, and peak).

The single variable with by far the largest impact on blood Pb distributions was, not unexpectedly, the GSD estimate representing individual responses to Pb exposures.

5.4 Blood Pb Distribution Estimates for the Three Case Studies

5.4.1 Primary Pb Smelter

The results of the blood Pb modeling for the primary Pb smelter case study are summarized in Exhibit 5-15. The median concurrent blood Pb estimates are 1.3 μg/dL and 0.7 μg/dL, respectively, when estimated with the IEUBK and Leggett models. The median lifetime average blood Pb estimates from the two models are 2.7 and 1.6 μg/dL, respectively. Comparing the two sets of estimates, it can be seen that the reduction of exposure in the highest exposure blocks under the attainment scenario has essentially no impact on the estimated blood Pb distributions except for the very highest percentiles. The 99.9^{th} percentile values for the four blood Pb metrics are reduced between about eight percent and 18 percent under the attainment scenario, compared to the current conditions estimates.

Exhibit 5-15. Projected Blood Pb Levels (µg/dL) for Primary Pb Smelter Case Study

Statistic	Concurrent Blood Pb		Lifetime Average Blood Pb		Average Proportion from Policy-Relevant Sources [a]
	IEUBK	Leggett	IEUBK	Leggett	
Current Conditions Exposure Scenario					
99.9th	22	14	29	23	98%
99.5th	12	6.7	17	11	95%
99th	7.4	4.2	11	6.8	91%
95th	3.7	2	5.3	3.1	67%
90th	2.9	1.5	4.1	2.3	70%
75th	2.0	1.0	2.7	1.6	59%
Median	1.3	0.7	1.8	1.1	54%
25th	0.9	0.5	1.2	0.7	49%
1st	0.4	0.2	0.5	0.3	46%
Attainment Exposure Scenario					
99.9th	18	13	25	21	96%
99.5th	11	6.4	16	11	93%
99th	7.9	4.3	11	7.1	93%
95th	3.7	2	5.4	3.1	76%
90th	2.9	1.5	4.1	2.3	68%
75th	2.0	1.0	2.8	1.6	58%
Median	1.4	0.7	1.9	1.1	52%
25th	0.9	0.5	1.2	0.7	45%
1st	0.4	0.2	0.5	0.3	44%

[a] Pathways impacted by policy-relevant sources include inhalation, soil and indoor dust ingestion (and exclude policy-relevant background sources, such as diet and drinking water).

The last column of Exhibit 5-15 summarizes the estimated average proportions of Pb uptake contributed by policy-relevant sources at the various percentile blood Pb estimates. Under both exposure scenarios, it can be seen that the proportion of Pb uptake from policy-relevant sources increases from around 40 percent for the lowest blood Pb percentiles, to around 98 percent for the very highest percentiles. The contributions of individual exposure pathways to total blood uptake in the three case studies are discussed in more detail in Section 6.2 and tabulated in Appendix J.

5.4.2 Secondary Pb Smelter

Exhibit 5-16 summarizes the blood Pb modeling results for the secondary Pb smelter case study. The median blood Pb levels (and the higher percentile estimates) are lower than those calculated for the primary Pb smelter, for both sets of soil estimates. Median concurrent estimates are 0.9 µg/dL and 0.5 µg/dL, based on the IEUBK and Leggett models, respectively. The median lifetime average blood Pb estimates are 1.1 and 0.7 µg/dL. It can be seen from the data in Exhibit 5-16 that the higher soil concentrations estimated using the hybrid approach result in increases in both the blood percentile estimates and in the estimated proportions of Pb uptake from policy-relevant sources.

Exhibit 5-16. Projected Blood Pb Levels (µg/dL) for Secondary Pb Smelter Case Study

Statistic	Concurrent Blood Pb		Lifetime Average Blood Pb		Average Proportion from Policy-Relevant Sources [a]
	IEUBK	Leggett	IEUBK	Leggett	
MPE Approach for Characterizing Soil Concentrations					
99.9th	3.7	2.1	4.7	3.0	25%
99.5th	3.0	1.6	3.9	2.3	27%
99th	2.7	1.4	3.5	1.9	24%
95th	1.9	1.0	2.5	1.4	28%
90th	1.7	0.9	2.1	1.2	23%
75th	1.2	0.6	1.6	0.9	24%
Median	0.9	0.5	1.1	0.7	25%
25th	0.7	0.3	0.8	0.5	24%
1st	0.3	0.2	0.4	0.2	24%
Hybrid Approach (Model + Surrogate Data) for Characterizing Soil Concentrations					
99.9th	4.7	2.3	6.3	3.3	38%
99.5th	3.7	1.9	4.9	2.7	39%
99th	3.3	1.7	4.4	2.5	40%
95th	2.4	1.2	3.2	1.8	39%
90th	2.0	1.0	2.7	1.5	37%
75th	1.5	0.8	2.0	1.1	39%
Median	1.1	0.6	1.5	0.8	38%
25th	0.8	0.4	1.1	0.6	38%
1st	0.4	0.2	0.5	0.3	37%

[a] Pathways impacted by policy-relevant sources include inhalation, soil and indoor dust ingestion (and exclude policy-relevant background sources, such as diet and drinking water).

5.4.3 Near Roadway Urban Case Study

The near roadway urban case study blood Pb estimates are shown in Exhibit 5-17. As discussed in Section 5.2.1, the median blood Pb estimates generated by the IEUBK model fall in the same range as those for the primary Pb smelter case study. However, the upper percentile estimates are substantially lower than for the primary Pb smelter because of the lower maximum exposures in the near roadway case. The estimated proportion of Pb uptake from policy-relevant sources stays relatively constant across the blood Pb percentiles.

Exhibit 5-17. Projected Blood Pb Levels (µg/dL) for Near Roadway Urban Case Study

Statistic	Concurrent Blood Pb		Lifetime Average Blood Pb		Average Proportion from Policy-Relevant Sources [a]
	IEUBK	Leggett	IEUBK	Leggett	
99.9th	6.5	4.2	9.1	6.9	62%
99.5th	5.0	3.1	7.0	4.9	56%
99th	4.4	2.7	6.2	4.1	58%
95th	3.1	1.9	4.4	2.9	56%
90th	2.6	1.6	3.6	2.4	58%
75th	1.9	1.2	2.7	1.7	56%
Median	1.4	0.8	1.9	1.2	63%
25th	1.0	0.6	1.4	0.9	60%
1st	0.4	0.3	0.6	0.4	56%

[a] Pathways impacted by policy-relevant sources include inhalation, soil and indoor dust ingestion (and exclude policy-relevant background sources, such as diet and drinking water).

6. Estimation of Human Health Risk

This chapter presents the methods and results for the characterization of health impacts associated with Pb exposures for the three human health case studies. As discussed in Chapter 2, IQ changes in exposed children have been selected as the endpoint for risk quantification in the pilot phase assessment, on the basis of a large body of evidence establishing the occurrence of the IQ effects at low exposure and blood Pb levels. In addition, the estimation of these impacts is facilitated by the availability of recent high-quality studies of the relationship between children's blood Pb and IQ loss. It is noted, however, that low-level Pb exposures have been found to be associated with a wide range of adverse effects in children and adults beyond those quantified here (USEPA 2006b).

6.1 Methods

6.1.1 Overview of Approach

The estimation of IQ loss is performed using the distribution of blood Pb estimates for exposed populations derived via the probabilistic model described in Section 5.1.3.4. This model estimates a population-weighted distribution of estimated individual blood Pb concentrations by first taking 10,000 population-weighed random samples from the GM blood Pb concentrations estimated for each block or block group in the case studies. Each of these 10,000 GM blood Pb levels is then combined with an adjustment factor (sampled from a lognormal distribution reflecting inter-individual variability in behavior and biokinetics related to Pb exposure) to generate a blood Pb level for a simulated child. This procedure ultimately generates blood Pb levels for 10,000 simulated children for each case study location.

To calculate the IQ loss impacts of the case study Pb exposures, exposure-response functions for IQ loss differentiated for the two blood metrics considered in the pilot phase (i.e., concurrent and lifetime average) are used to convert each of the blood Pb levels for the 10,000 simulated children into resulting IQ loss. As described below, the blood Pb-IQ relationships are derived from the results of a recent analysis of pooled epidemiological data by Lanphear et al. (2005). Individual IQ loss estimates are calculated for each of the lifetime average and concurrent blood Pb estimates using two forms of the blood Pb-IQ model from Lanphear et al. (2005) matched to these blood Pb metrics. The outputs of the risk assessment are the distributions of estimated IQ loss across the exposed populations, characterized as percentile values.

6.1.2 Blood Pb-IQ Relationships

As noted above, the blood Pb-IQ models were derived from the results of a large epidemiological analysis by Lanphear et al. (2005), in which the relationships between blood Pb concentrations and IQ test results in seven populations were analyzed. The 1,333 subjects in the study population included subjects of four studies in the United States, and one each from Mexico, Australia, and Kosovo. A number of statistical techniques were used to characterize the relationships between several blood Pb metrics and test results for individual cohorts and for the pooled study population. For the pilot phase assessment, model forms were selected that related IQ change to two blood Pb metrics, lifetime average and concurrent. These metric were meant to

be analogous to the metrics defined in the Lanphear et al. (2005) study, where lifetime average was defined as the average of all blood Pb measurements taken from a given individual, and concurrent as the blood Pb measurement taken closest to the time of IQ testing. For purposes of the pilot phase of the assessment, lifetime average blood Pb was the estimated time-weighted average blood Pb concentration from age 6 months to 84 months, calculated as described in Section 5.1.3, using either the IEUBK or Leggett model. Concurrent blood Pb was estimated, for purposes of the pilot phase assessment, as the average modeled blood Pb concentration from ages 73 to 84 months, inclusive (see Section 5.1.3), the rational being a desire to have a blood Pb metric matching as closely as possible to the average age at IQ testing (6.9 years) in the Lanphear et al. study.

The log-linear model for IQ loss derived from the Lanphear et al. pooled analysis was adapted for use in the pilot phase assessment. The coefficients in the model are different for lifetime average and concurrent blood Pb, as shown below:

$$\text{Concurrent:} \quad IQ = 99.3 - 2.70 \times \ln(\text{Concurrent Blood Pb})$$

$$\text{Lifetime Average:} \quad IQ = 100.9 - 3.04 \times \ln(\text{Lifetime Average Blood Pb})$$

The form of these models is consistent with the observed greater slope of the blood Pb-IQ relationship seen at low blood Pb levels. Lanphear et al. (2005) also investigated piecewise linear models to capture this relationship and found that the slope of the blood Pb-IQ relationship at blood Pb concentrations less than 7.5 µg/dL was significantly greater than the linear slope estimated above that level.

Applying the models shown above to the estimation of IQ changes involves a degree of uncertainty. First, because of the log linear form of the models, they predict very large changes in IQ for small changes in blood Pb at low concentrations. Thus, it is important to stay within the range of the exposure data used in fitting the functions when predicting IQ loss for hypothetical exposed populations.

In addition, the constant terms in the models represent central tendency estimates of "background" (low exposure) IQ distributions for the study populations. These values may not be applicable to the population for which IQ is being estimated. Thus, applying the models "as is" to the case study population would involve making possibly unjustifiable assumptions about baseline IQ levels.

For these reasons, the Lanphear et al. (2005) models were adjusted in the following manner for application in the pilot phase assessment. For each model, a blood Pb "cutpoint" was derived that corresponded to the approximate 5th percentile blood Pb levels seen in the Lanphear et al. pooled study population. Adverse effects were not estimated for predicted individual blood Pb levels below the cutpoints. The cutpoints vary for the blood Pb metrics; for concurrent blood Pb, the cutpoint is 2.4 µg/dL, while for lifetime average blood Pb, the cutpoint is 6.1 µg/dL (Lanphear et al. 2005, Table 4).

The use of the blood Pb cutpoints in IQ loss estimation is not intended to imply that these concentrations are thresholds below which adverse effects do not occur. Rather, they are intended to reflect practical limits on IQ prediction based on the low numbers of Lanphear et al. (2005) study subjects in these concentration ranges, and the corresponding large uncertainties in the estimates of the blood Pb-IQ relationships below the cutpoints.

Changes in IQ associated with predicted individual blood Pb levels above the cutpoints were estimated relative to the corresponding cutpoint as:

$$\text{Concurrent:} \quad \text{IQ loss (relative to cutpoint)} = -2.70 \times \ln(\text{concurrent blood Pb} / 2.4)$$

$$\text{Lifetime Average:} \quad \text{IQ loss (relative to cutpoint)} = -3.04 \times \ln(\text{lifetime average blood Pb} / 6.1)$$

The outputs of the IQ modeling were thus estimates of changes in IQ relative to an individual exposed at the cutpoint level, rather than absolute IQ loss estimates.

6.2 Risk Assessment Results

As noted in Section 6.1, the primary outputs of the pilot phase risk assessment are population distributions of estimated blood Pb concentrations and IQ loss, along with evaluations of the proportional contribution of individual exposure pathways to total lifetime Pb uptake. This section presents the results for each of the three case studies. Detailed results of the risk assessment can be found in Appendix J.

Risk results presented in this section and in Appendix J reflect both IEUBK- and Leggett-based blood Pb modeling. As noted earlier in Chapter 5 (see Sections 5.1.2.2 and 5.1.3.3) and as evident in the risk results exhibits presented in this section, modeling completed for the pilot phase has consistently shown a trend of Leggett-derived blood Pb levels being lower than IEUBK-derived levels for the same exposure scenario. This trend is the opposite of findings presented in Pounds and Leggett (1998). This apparent discrepancy elevates the importance of model evaluation focused on IEUBK and Leggett (as applied in the pilot phase) which is currently ongoing. It also emphasizes the fact that results of the pilot phase should be considered provisional and subject to update in the full-scale assessment. Note that the results of model evaluation, once completed, will be conveyed by separate cover.

6.2.1 Primary Pb Smelter Case Study

Exhibit 6-1 summarizes the blood Pb and IQ loss distributions for the primary Pb smelter current conditions scenario, calculated using the IEUBK and Leggett blood Pb models, with IQ loss values based on lifetime average and concurrent values. All other input assumptions and values used in the assessment are described in Chapter 5. The first column of this exhibit labels the percentile values of the outputs from the probabilistic model, and the second column shows the number of children who would have blood Pb and IQ loss values above the percentile value, given the estimated population at risk in the study area.

Exhibit 6-1. Projected IQ Losses from the Primary Pb Smelter Current Conditions Scenario

Percentile	Population Above	IEUBK Model				Leggett Model			
		Concurrent Blood Pb		Lifetime Average Blood Pb		Concurrent Blood Pb		Lifetime Average Blood Pb	
		Total Blood Pb (µg/dL)	IQ Loss (Log-Linear Model)	Total Blood Pb (µg/dL)	IQ Loss (Log-Linear Model)	Total Blood Pb (µg/dL)	IQ Loss (Log-Linear Model)	Total Blood Pb (µg/dL)	IQ Loss (Log-Linear Model)
99.9th	4	21.9	6.0	28.6	4.7	13.9	4.7	22.9	4.0
99.5th	19	12.4	4.4	16.9	3.1	6.7	2.8	11.1	1.8
99th	39	7.4	3.0	10.6	1.7	4.2	1.5	6.8	<1
95th	194	3.7	1.2	5.3	--	2.0	--	3.1	--
90th	388	2.9	<1	4.1	--	1.5	--	2.3	--
75th	970	2.0	--	2.7	--	1.0	--	1.6	--
Median	1,940	1.3	--	1.8	--	0.7	--	1.1	--
25th	2,910	0.9	--	1.2	--	0.5	--	0.7	--
1st	3,841	0.4	--	0.5	--	0.2	--	0.3	--

It can be seen that the large majority of exposed children are predicted to have blood Pb levels below both the concurrent and lifetime average cutpoint values (i.e., no IQ loss associated with Pb exposure is estimated). No IQ loss is predicted for the median or 75th percentile child by either the IEUBK or Leggett models, based on either the lifetime average or concurrent blood Pb values. IQ losses are predicted for the largest proportion of children by the IEUBK model (because it predicted higher blood Pb concentration than Leggett) when the concurrent blood Pb metric is used as the input to the blood Pb-IQ model. Estimated IQ loss values range from less than1 for the 90th percentile child to 6.0 for the 99.9th percentile. Under the three other modeling options, IQ losses are predicted starting between the 95th and 99th percentiles and range from less than 1 to 4.7.

Predicted blood Pb and IQ loss distributions for the primary Pb smelter attainment scenario are essentially identical (Exhibit 6-2) to those for the current conditions scenario, except for the very highest percentiles. This is as expected because the exposure concentrations change between the two air quality scenarios only for the seven highest exposure blocks (see Section 5.1.4), where only about 0.5 percent of the exposed children reside. Thus, the estimated IQ losses are slightly smaller for the 99.5th and 99.9th percentile children under the attainment scenario, while the remaining estimates are the same, within simulation error, to estimates derived for the current conditions scenario.

Exhibit 6-2. Projected IQ Losses from the Primary Pb Smelter Attainment Scenario

Percentile	Population Above	IEUBK Model				Leggett Model			
		Concurrent Blood Pb		Lifetime Average Blood Pb		Concurrent Blood Pb		Lifetime Average Blood Pb	
		Total Blood Pb (µg/dL)	IQ Loss (Log-Linear Model)	Total Blood Pb (µg/dL)	IQ Loss (Log-Linear Model)	Total Blood Pb (µg/dL)	IQ Loss (Log-Linear Model)	Total Blood Pb (µg/dL)	IQ Loss (Log-Linear Model)
99.9th	4	18.4	5.5	24.9	4.3	12.7	4.5	20.7	3.7
99.5th	19	11.2	4.2	15.5	2.8	6.4	2.6	10.6	1.7
99th	39	7.9	3.2	11.3	1.9	4.3	1.6	7.1	0.5
95th	194	3.7	1.2	5.4	--	2.0	--	3.1	--
90th	388	2.9	0.5	4.1	--	1.5	--	2.3	--
75th	970	2.0	--	2.8	--	1.0	--	1.6	--
Median	1,940	1.4	--	1.9	--	0.7	--	1.1	--
25th	2,910	0.9	--	1.2	--	0.5	--	0.7	--
1st	3,841	0.4	--	0.5	--	0.2	--	0.3	--

In addition to calculating blood Pb and IQ loss distributions, the probabilistic model also records the contribution of each exposure pathway to total lifetime average Pb uptake at each percentile of the blood distributions. Pb uptake is defined as the total amount of Pb that is absorbed into the blood compartments of the biokinetic models. The probabilistic model calculates pathway-specific Pb uptakes for each census block and block group independently of the biokinetic models, using the same uptake calculations equations as used in the IEUBK and Leggett models (Section 5.1.3.3). When a census block or block group is chosen in the simulation model, the uptake values are saved. When percentiles of the blood Pb/IQ loss distributions are calculated, the associated pathway contributions for the block group are retrieved. These values give an approximate picture of the importance of specific exposure pathways at specific percentile values; they do not, however, reflect individual variability in Pb uptake within the blocks or block groups.

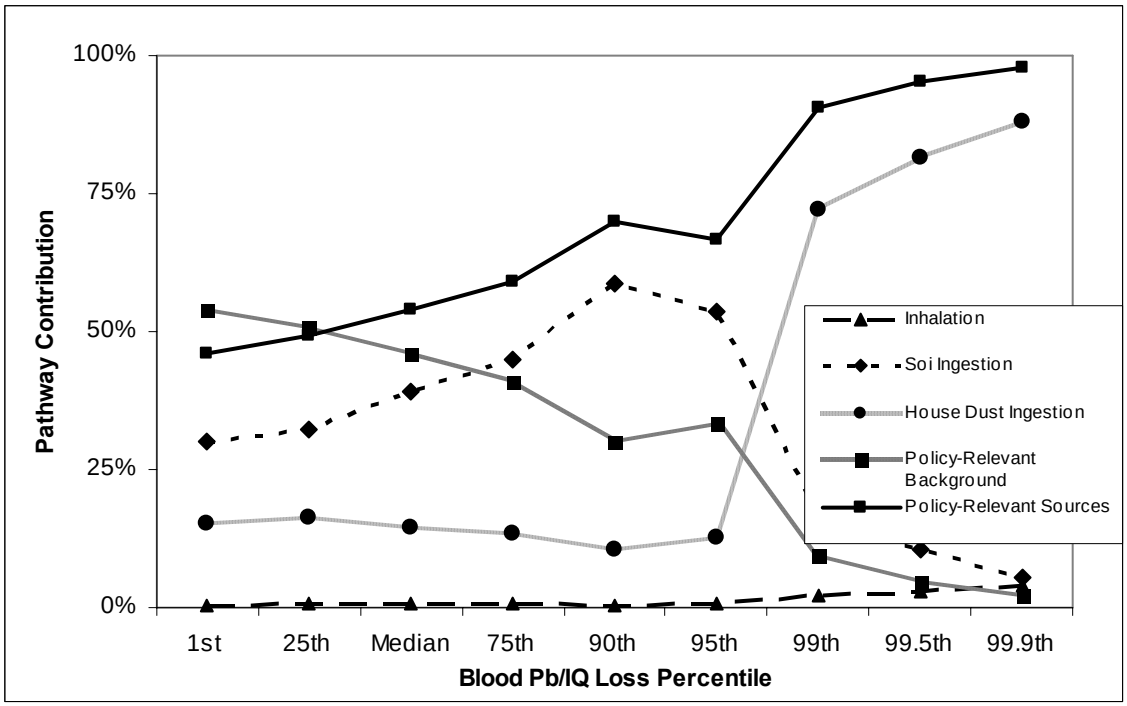
The average proportion of total Pb uptake contributed by policy-relevant sources and policy-relevant background pathways differs widely across the blood Pb/IQ loss percentiles (Exhibit 6-3). At the lowest percentiles, policy-relevant background (in this case study, they are diet and drinking water) contributes the majority (about 54 percent) of total Pb uptake. The proportional contribution from policy-relevant background then decreases rapidly until it accounts for only about two percent of the total Pb uptake at the 99.9th percentile of the simulation blood Pb.

Among the exposure pathways associated with policy-relevant sources, inhalation exposure accounts for a very small proportion (less than five percent) of Pb uptake across all blood Pb percentiles. Ingestion of soil provides about 30 percent of total uptake for the low percentiles, increasing to just more than half of the total (59 percent) at the 90th percentile, then decreasing for the highest blood Pb percentiles. As might be expected, the blocks and block groups with the

highest percentile (99th and above) individual blood Pb and IQ estimates appear to have very high house dust Pb levels, which accounts for the bulk of total Pb uptake.

The proportions graphed in Exhibit 6-3 represent the average of the contributions from all eight exposure scenario/blood Pb model/blood Pb metric combinations for the primary Pb smelter case study.[1] The estimated contributions from the individual pathways for specific percentile blood Pb values differ only slightly (generally, a few percent) across the various modeling approaches.

Exhibit 6-3. Average Exposure Pathway Contributions to Total Pb Uptake, Primary Pb Smelter Case Study[a]



[a]" Policy-Relevant Sources" exposures include air, soil, and house dust. Diet and water exposures constitute "Policy-Relevant Background" in this case study.

6.2.2 Secondary Pb Smelter Case Study

Exhibit 6-4 summarizes the estimated blood Pb and IQ loss distributions for the secondary Pb smelter case study when the modeled (MPE) estimates of soil concentrations are used as inputs. In this scenario, as well as when the hybrid soil concentration model is used, the AGG (Air + Soil) model (as described further in Section 4.1) was used to estimate house dust Pb concentrations.

As expected, due to the lower exposure concentrations, the estimated blood Pb levels and IQ losses are smaller than those estimated for the primary Pb smelter. Based on the concurrent blood Pb distributions obtained from the IEUBK model, IQ losses are predicted only for children

[1] The eight estimates come from two exposure scenarios (current conditions and attainment), two blood Pb models (IEUBK and Leggett), and two blood Pb metrics (concurrent and lifetime average).

above the 95th percentile of the individual blood Pb distribution. Based on the estimated number of resident children near the secondary Pb smelter, approximately 84 children would be at the 95th percentile blood Pb or above.

Exhibit 6-4. Projected IQ Losses from the Secondary Pb Smelter – Modeled (MPE) Soil Concentration Estimates

Percentile	Population Above	IEUBK Model				Leggett Model			
		Concurrent Blood Pb		Lifetime Average Blood Pb		Concurrent Blood Pb		Lifetime Average Blood Pb	
		Total Blood Pb (µg/dL)	IQ Loss (Log-Linear Model)	Total Blood Pb (µg/dL)	IQ Loss (Log-Linear Model)	Total Blood Pb (µg/dL)	IQ Loss (Log-Linear Model)	Total Blood Pb (µg/dL)	IQ Loss (Log-Linear Model)
99.9th	2	3.7	1.8	4.7	--	2.1	--	3.0	--
99.5th	8	3.0	1.1	3.9	--	1.6	--	2.3	--
99th	17	2.7	<1	3.5	--	1.4	--	1.9	--
95th	84	1.9	<1	2.5	--	1.0	--	1.4	--
90th	167	1.7	--	2.1	--	0.9	--	1.2	--
75th	418	1.2	--	1.6	--	0.6	--	0.9	--
Median	836	0.9	--	1.1	--	0.5	--	0.7	--
25th	1,254	0.7	--	0.8	--	0.3	--	0.5	--
1st	1,655	0.3	--	0.4	--	0.2	--	0.2	--

For this case study, predicted IQ changes range from less than 1 point for the 99th percentile to 1.8 points for the 99.9th percentile child. No IQ losses are predicted using the lifetime blood Pb distribution from the IEUBK model or using either the lifetime average or concurrent metric generated by the Leggett model. All of the 99.9th percentile blood Pb estimates are less than the lifetime average (6.1 µg/dL) or concurrent (2.4 µg/dL) cutpoints.

Exhibit 6-5 shows the blood Pb and IQ distributions for the secondary Pb smelter case study when the hybrid soil Pb estimation model is used. As discussed in Section 5.1.4, hybrid soil concentration estimates are three times the MPE modeled values. Because of the higher soil concentrations, it can be seen that the estimated Pb and IQ loss percentiles are slightly higher than when the modeled (MPE) soil concentrations are used. IQ losses are predicted for children above about the 90th percentile (corresponding to 167 children); based on the IEUBK concurrent blood Pb estimates, IQ losses range from less than 1 at the 95th percentile to 2.3 at the 99.9th percentile. A very small IQ loss (0.1 point) is also predicted by the IEUBK lifetime average blood Pb metric. As was the case for the modeled soil estimates, no IQ losses are predicted when the Leggett model is used to predict either lifetime average or concurrent blood Pb distributions.

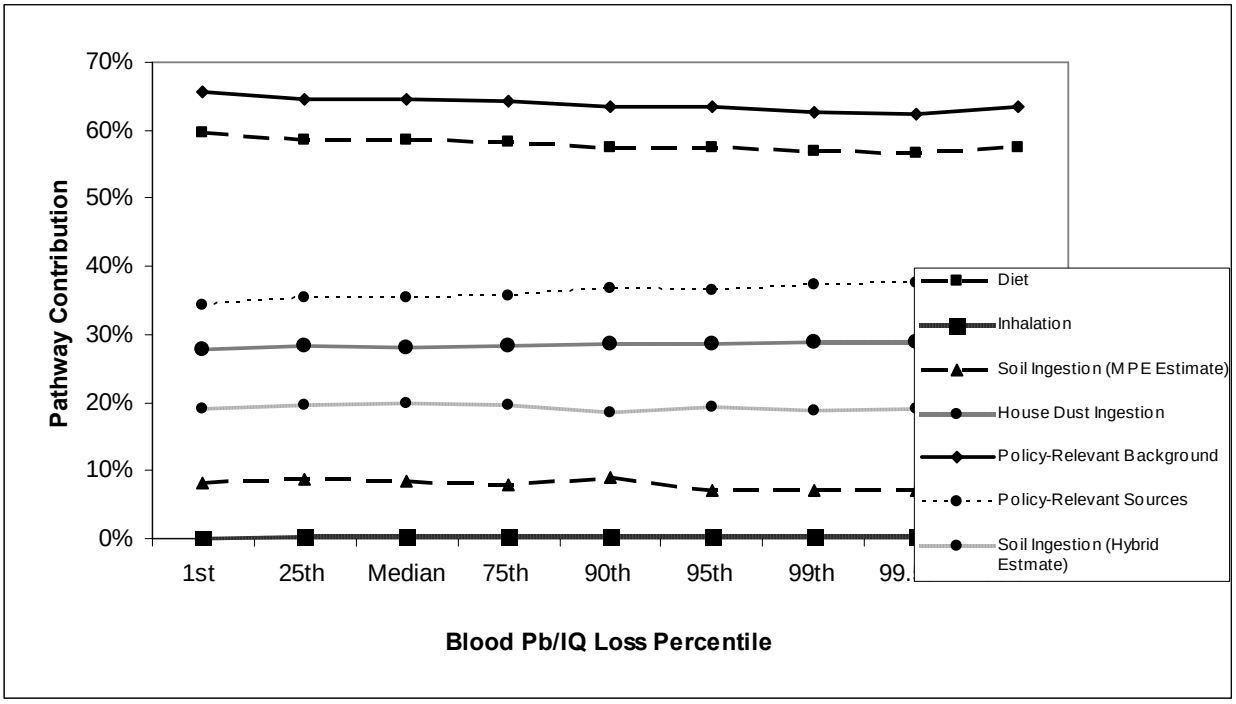
Exhibit 6-5. Projected IQ Losses from the Secondary Pb Smelter Case Study – Hybrid Soil Concentration Estimates

Percentile	Population Above	IEUBK Model				Leggett Model			
		Concurrent Blood Pb		Lifetime Average Blood Pb		Concurrent Blood Pb		Lifetime Average Blood Pb	
		Total Blood Pb (µg/dL)	IQ Loss (Log-Linear Model)	Total Blood Pb (µg/dL)	IQ Loss (Log-Linear Model)	Total Blood Pb (µg/dL)	IQ Loss (Log-Linear Model)	Total Blood Pb (µg/dL)	IQ Loss (Log-Linear Model)
99.9th	2	4.7	2.3	6.3	<1	2.3	--	3.3	--
99.5th	8	3.7	1.4	4.9	--	1.9	--	2.7	--
99th	17	3.3	1.0	4.4	--	1.7	--	2.5	--
95th	84	2.4	<1	3.2	--	1.2	--	1.8	--
90th	167	2.0	--	2.7	--	1.0	--	1.5	--
75th	418	1.5	--	2.0	--	0.8	--	1.1	--
Median	836	1.1	--	1.5	--	0.6	--	0.8	--
25th	1,254	0.8	--	1.1	--	0.4	--	0.6	--
1st	1,655	0.4	--	0.5	--	0.2	--	0.3	--

The proportional contribution of the individual exposure pathways to total Pb uptake in the secondary Pb smelter case study is very different from that seen for the primary Pb smelter case study. Unlike the primary Pb smelter case study, the average pathway contributions remain approximately constant across the blood Pb/IQ loss percentiles, all varying less than about two percent around their mean values. This is due primarily to the relatively flat exposure distributions; most census blocks have very low ambient air and house dust Pb concentrations, and policy-relevant background exposure pathways (diet, water, and naturally occurring soil Pb) contribute a large percentage of total Pb uptake across all the blood Pb percentiles (Exhibit 6-6).

As expected, the relative pathway contributions differ when the two different sets of soil concentration estimates are used as inputs to the blood Pb/IQ loss models. The soil pathway contribution, averaged across the four combinations of exposure scenario and blood Pb model, increases from 7.8 percent to about 19 percent when hybrid, rather than modeled, soil concentration estimates are used. The contributions from the other pathways decrease in compensation for the higher soil Pb levels.

Exhibit 6-6. Pathway Contributions to Total Pb Uptake, Secondary Pb Smelter Case Study[a]



[a] "Policy-Relevant Background" includes diet and water intake, plus a contribution from naturally occurring Pb in soil of 15 μg/g).

6.2.3 Near Roadway Urban Case Study

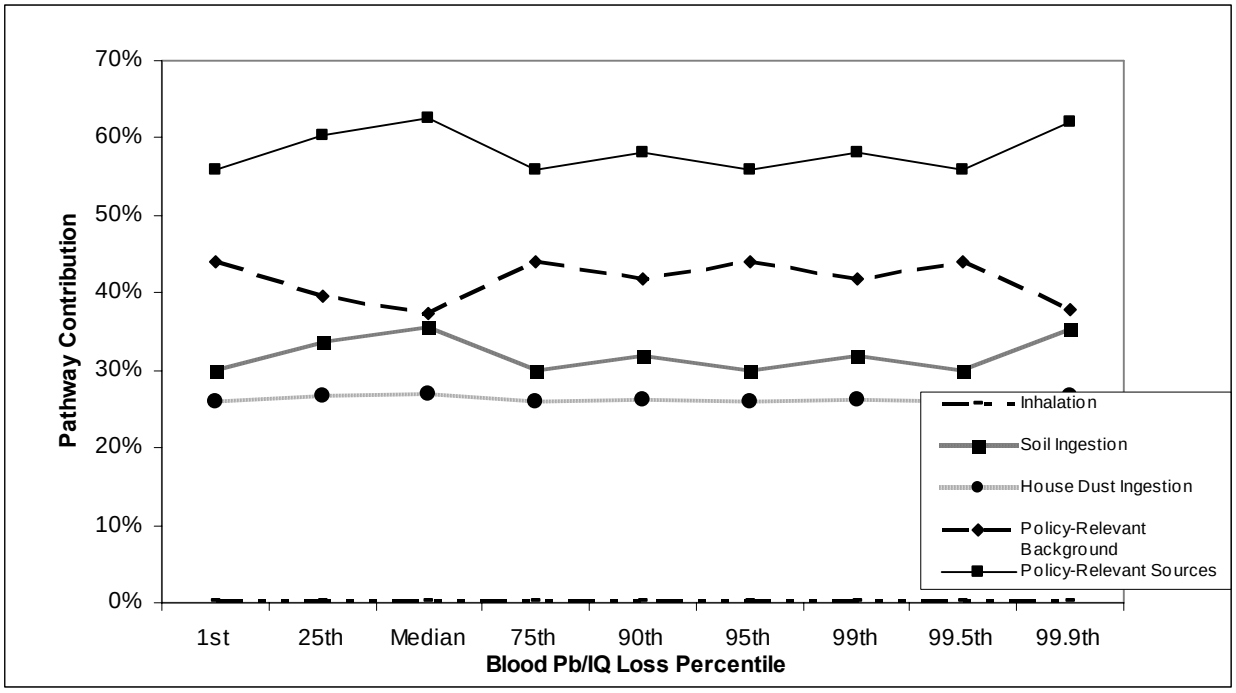
Exhibit 6-7 summarizes the results of the pilot phase risk assessment for the near roadway urban case study. IQ losses are predicted to occur for children above the 75th percentile IEUBK concurrent blood Pb estimate, ranging from less than 1 point at the 90th percentile to 2.7 points at the 99.9th percentile. Based on the estimated 319 children living in the census blocks included in this case study, 32 children are predicted to be exposed at or above the 90th percentile blood Pb level, and approximately three children above the 99th percentile blood/IQ level. IQ losses are predicted for smaller proportions of children when the IEUBK model is used to estimate lifetime average blood Pb values and for both blood Pb metrics when estimated with the Leggett model.

Exhibit 6-7. Projected IQ Losses from the Near Roadway Urban Case Study

Percentile	Population Above	IEUBK Model				Leggett Model			
		Concurrent Blood Pb		Lifetime Average Blood Pb		Concurrent Blood Pb		Lifetime Average Blood Pb	
		Total Blood Pb (µg/dL)	IQ Loss (Log-Linear Model)	Total Blood Pb (µg/dL)	IQ Loss (Log-Linear Model)	Total Blood Pb (µg/dL)	IQ Loss (Log-Linear Model)	Total Blood Pb (µg/dL)	IQ Loss (Log-Linear Model)
99.9th	0	6.5	2.7	9.1	1.2	4.2	1.5	6.9	<1
99.5th	2	5.0	2.0	7.0	<1	3.2	<1	4.9	--
99th	3	4.4	1.6	6.2	--	2.7	<1	4.1	--
95th	16	3.1	0.7	4.4	--	1.9	--	2.9	--
90th	32	2.6	<1	3.6	--	1.6	--	2.4	--
75th	80	1.9	--	2.7	--	1.2	--	1.7	--
Median	159	1.4	--	1.9	--	0.8	--	1.2	--
25th	239	1.0	--	1.4	--	0.6	--	0.9	--
1st	316	0.4	--	0.6	--	0.3	--	0.4	--

As was the case for the secondary Pb smelter case study, the contribution of the individual exposure pathways is relatively constant across blood Pb/IQ loss percentiles (Exhibit 6-8). In contrast to the secondary Pb smelter, however, policy-relevant sources contribute the majority of total Pb uptake (56 to 63 percent) across the blood Pb percentile distribution. Policy-relevant background exposures (diet and drinking water, for this case study) contribute between 37 and 44 percent to total Pb uptake.

Exhibit 6-8. Average Exposure Pathway Contributions to Total Pb Uptake, Near Roadway Urban Case Study[a]



[a] "Policy-Relevant Sources" exposures include air, soil, and house dust. Diet and water are the "Policy-Relevant Background" exposures in this case study.

6.3 Uncertainty and Variability in the IQ Loss Estimates

6.3.1 Sources of Uncertainty

There are two sets of factors that contribute the overall uncertainty in the IQ loss estimates. The first, discussed in Section 5.3, are those factors which affect the blood Pb estimates from which the IQ loss distributions are calculated. These include the choice of the model used to estimate blood Pb levels, the GSD describing the individual variability in response to Pb exposures, uncertainties in exposure concentration estimates, and uncertainties in the exposure, intake, and uptake factors used to estimates of Pb uptake (absorbed dose).

The second set of factors that needs to be considered includes those that are associated specifically with the estimation of IQ loss from the individual blood Pb estimates. In the sensitivity analysis described in Section 6.4, the impact of using two different models (log-linear and piecewise linear) derived from the Lanphear et al. (2005) epidemiological study are examined. Blood Pb-IQ models have been developed from a number of other high-quality studies (Canfield et al. 2003, for example), and such models could also be applied to the individual blood Pb levels calculated in the probabilistic model.

The nature of the blood Pb metric used to estimate IQ loss also strongly affects the results of the analysis. Both the slope parameters and the cutpoints are very different in the concurrent and lifetime average blood Pb-IQ model equations summarized above. In the sensitivity analyses,

described in the following section, the effect of using these two metrics, as well as models for the "peak" blood Pb, on the modeled IQ loss distributions was examined.

The decision to use the blood Pb cutpoints also introduces uncertainty into the assessment. Using different cutpoints (i.e., making different judgments about the predictive power of the Lanphear et al. (2005) blood Pb-IQ models at low blood Pb concentrations) changes the both the numbers and proportions of children for whom adverse effects (IQ loss) are predicted to occur, and the magnitude of the predicted IQ changes. The impact of different assumptions about the blood Pb cutpoints is also investigated in the sensitivity analysis described below.

6.3.2 Sensitivity Analysis Methodology

For the pilot phase risk assessment, a series of "one-at-a-time" analyses has been conducted to evaluate the impacts of individual modeling methods and parameter inputs on estimated blood Pb and IQ loss distributions. The general approach employed was to evaluate changes in risk metrics associated with varying single models or input parameter values away from those used in a "baseline" analysis. The set of models and input parameter values associated with the baseline analysis are summarized in Exhibit 6-9. This exhibit also indicates the analytical steps on which a sensitivity analysis was performed and the alternate model or parameter values that were examined in the analysis.

The baseline risk analysis focused on the current conditions scenario for the primary Pb smelter case study. In this analysis, inhalation and soil exposure concentrations were estimated as described in Section 4.1. House dust Pb concentrations were derived based on the H6 site-specific regression model for the census block and block groups near the facility, and with the AGG (air + soil) model for the remainder of the exposed population. The baseline individual blood Pb concentrations were derived using the IEUBK model estimates for each census block or block group, assuming individual blood Pb concentrations were lognormally distributed with a GSD of 1.6. Individual IQ loss was estimated based on concurrent blood Pb concentration using the Lanphear et al. (2005) log-linear model with IQ loss not predicted for individuals with blood Pb concentrations below the cutpoint value of 2.4 µg/dL. Note that while we believe the baseline models and parameter values are believed to provide a credible estimate of blood Pb and IQ changes, it is not suggested that the baseline model is uniquely the "best," or that it provides demonstrably "central tendency" risk estimates.

The bottom row of Exhibit 6-9 summarizes the models and parameters that were varied (individually) during the sensitivity analysis. Changes in soil and air exposure concentrations were not changed in the course of the sensitivity analysis. In contrast, the impact of different models for estimating house dust concentrations (AGG [air only] and AGG [air + soil]) was evaluated. GM blood Pb concentrations for the blocks and block groups were calculated based on the Leggett biokinetic model, as well as an "empirical" blood Pb model developed by Lanphear et al. (1998) (see Appendix K), and the impacts of using the different models on predicted changes in IQ was also evaluated. Variations in parameters related to Pb exposure, intake, and uptake estimation were also evaluated. In one set of sensitivity runs, the GI absorption fractions for dietary and drinking water ingestion were varied above and below their estimated central tendency values used in the baseline assessment. For the second set of

sensitivity runs, alternative estimates of daily soil and dust ingestion and soil and dust absorption fractions for soil and dust were evaluated for their impacts on estimated blood Pb and IQ loss distributions. These values were derived by Von Lindern et al. (2003) from a systematic statistical analysis of soil, dust, and blood Pb distributions at a large mining/smelting "Superfund" site. In addition, the probabilistic model was run with two alternative values of the blood Pb GSD.

Exhibit 6-9. Baseline and Sensitivity Analysis Blood Pb and Risk Estimation Models and Assumptions

Modeling Element	Case Study/ Scenario	Air, Soil Exposure Conc.	House Dust Pb Conc. Model	Blood Estimation Model	Exposure/ Intake/Uptake Factors	Individual Blood Pb Variability (GSD)	Blood Pb-IQ Model	Blood Pb Metric	Blood Pb Cutpoint
Baseline Estimate	Primary Pb Smelter/ Current Conditions	Modeled (ISC-PRIME), regression (soil)	Regression (H6 and AGG [Air + Soil])	IEUBK (Batch mode age profile)	Baseline (Exhibit 5-11)	1.6	Lanphear et al. (2005), log-linear	Concurrent (age 73-84 months)	2.4 (concurrent), 6.1 (lifetime average)
Sensitivity Analysis	Unchanged	Unchanged	Exclusively AGG (Air + Soil), AGG (Air)	Leggett, Lanphear et al. (1998)	Low, high dietary absorption; Von Lindern et al. (2003) soil, dust exposure factor estimates	1.3, 1.9; lower and upper bound values on GSD for children around exposure sources	Lanphear et al. (2005), piece-wise linear, upper and lower confidence limit slope parameter estimates	Lifetime, (6 - 84 months), peak (highest annual average)	1.2 (concurrent), 3.1 (lifetime average)

Alternative approaches to IQ loss estimation were also investigated. The results obtained from the piecewise linear form of the Lanphear et al. (2005) model were compared to the baseline (log-linear) modeling approach. Alternative estimates of IQ loss were also derived using the upper and lower confidence limit slope parameters from the Lanphear et al. (2005) model.

As noted above, the distribution of IQ loss was also estimated based on lifetime average and peak individual blood Pb values (instead of the baseline concurrent metric). Finally, the effect of reducing the Pb cutpoint value for estimating IQ loss was also tested. Detailed descriptions of the changes in models and parameter values are provided in Section 6.4.

6.4 Sensitivity Analysis Results

As discussed in 6.3, sensitivity analyses were conducted by varying one model or set of parameter values at a time from the "baseline" assumptions and examining the impact of the changes on blood Pb and IQ loss distributions. This section summarizes the results of these sensitivity analyses. In all of the following exhibits, the sensitivity analysis models differ from the baseline only with regard to the single model or input parameter value that is being varied.

6.4.1 Alternative House Dust Pb Concentration Models

As shown in Section 6.2.1, house dust Pb exposures can contribute a large proportion of total Pb uptake, particularly for the highly exposed blocks and block groups in the primary Pb smelter case study. Thus, the impacts of employing alternative models for estimating house dust Pb concentrations were examined as part of the sensitivity analysis.

In the baseline assessment, house dust Pb concentrations were estimated using two different models. The "H6" site-specific regression model was used for blocks and block groups within 1.5 kilometers of the facility, while the AGG (Air + Soil) model was used to estimate house dust Pb levels for areas more distant from the smelter (see Section 4.1.4). In the sensitivity analysis, two variants of the AGG model were employed (Exhibit 6-10), one in which the Air + Soil variant was applied to all of the blocks and block groups, and the other where the Air Only form of the model was used. In all cases, the IEUBK model was used to estimate blood Pb distributions. The resultant blood Pb distributions are only mildly affected by the choice of the house dust Pb model – the median values are nearly the same (between 1.4 and 1.5 μg/dL). At the higher percentile values, the model estimates diverge more significantly because the H6 model predicts much higher house dust Pb concentrations for the blocks and block groups near the facility, relative to the other models.

Exhibit 6-10. Effects of Alternative House Dust Pb Concentration Models on Estimated Blood Pb Distributions [a,b]

Percentile	House Dust Pb Model			Change versus Baseline	
	Baseline (Composite Model), µg/dL	AGG Model (Air Only), µg/dL[c]	AGG Model (Air + Soil), µg/dL[d]	AGG Model (Air Only)	AGG Model (Air + Soil)
99.9th	17.0	13.6	11.2	-20%	-34%
99.5th	10.5	7.8	7.2	-25%	-32%
99th	7.3	6.1	6.2	-15%	-15%
95th	3.6	3.8	3.9	5%	7%
90th	2.8	3.1	3.1	8%	7%
75th	1.9	2.1	2.1	9%	9%
Median	1.3	1.5	1.4	12%	10%
25th	0.9	1.0	1.0	13%	9%
1st	0.4	0.4	0.4	15%	9%

[a] See Chapter 4, Section 4.1, for further discussion of the AGG regression model.
[b] Concurrent blood Pb estimates from the IEUBK model and the log-linear form of the Lanphear et al. (2005) blood Pb-IQ model are used for all estimates.
[c] AGG Model (Air Only) House Dust (µg/g) = 60 + 844 * Ambient Air Pb (µg/m^3)
[d] AGG Model (Air +Soil)House Dust (µg/g) = 31.1 + 638 * Ambient Air Pb (µg/m^3) + 0.364 * Soil Pb (µg/g)

This pattern is reflected in the estimated IQ loss distributions shown in Exhibit 6-11. Again, the decreasing slope of the log-linear model reduces the differences in estimated IQ loss at the higher percentiles.

Exhibit 6-11. Effects of Alternative House Dust Pb Models on Estimated IQ Distributions[a]

Percentile	House Dust Pb Model			Change versus Baseline	
	IQ Change Based on Composite Model House Dust Estimates	IQ Change Based on AGG Model (Air Only)	IQ Change Based on AGG Model (Air + Soil)	AGG Model (Air Only)	AGG Model (Air + Soil)
99.9th	-5.3	-4.7	-4.1	-11%	-22%
99.5th	-4.0	-3.2	-3.0	-20%	-26%
99th	-3.0	-2.5	-2.6	-15%	-14%
95th	-1.1	-1.2	-1.3	12%	17%
90th	-0.5	-0.7	-0.6	46%	43%
75th	--	--	--	--	--
Median	--	--	--	--	--
25th	--	--	--	--	--
1st	--	--	--	--	--

[a] Concurrent blood Pb estimates from the IEUBK model and the log-linear form of the Lanphear et al. (2005) blood Pb-IQ model are used for all estimates.

6.4.2 Alternative Blood Pb Models

To evaluate the sensitivity of IQ loss estimates to the selection of a blood Pb model, the IEUBK model was selected as the baseline and used to generate the concurrent baseline blood Pb distribution and IQ loss distributions, using as inputs the media concentration estimates from the primary Pb smelter current conditions scenario. The IEUBK input parameter values for this model run are presented in Exhibit 5-11. The results from this baseline IEUBK simulation were compared to results generated using two alternative blood Pb models.

First, the Leggett model was applied, with the results summarized in Section 6.2.1.[2] In addition, an "empirical" blood Pb model developed by Lanphear et al. (1998) was applied, which predicts blood Pb levels for 16 month-old children. This model was derived by statistical analyses of the relationships between multimedia Pb exposures (air, soil, house dust Pb loading, drinking water, and presence/absence of Pb paint), socioeconomic variables, and blood Pb measurement in young children obtained as part of a national survey conducted in 1997. The best fitting models from the analysis predict children's long-term blood Pb levels as a function of soil Pb concentrations and house dust Pb loading (the amount of Pb present per unit area of household surfaces; in this case, on the floor.) To apply the Lanphear et al. (1998) model, it was necessary to convert the estimated house dust exposure concentrations in the primary Pb smelter case study into estimates of dust Pb loading. As discussed in Appendix K, this was done by conducting a regression analysis of house dust Pb loading and concentration measurements from 305 housing units from the 1997 survey. The resulting relationship between house dust loading and concentration was then used to generate equivalent dust loading estimates for each block and block group, which were then used as inputs to the Lanphear et al. (1998) model (Appendix K).

The blood Pb distribution estimates generated by the IEUBK and Leggett models are summarized in Exhibit 6-12. As illustrated in this exhibit, the median blood Pb estimates from the models differ substantially, with the Leggett median estimate (0.7 µg/dL) being about one-half that from the baseline IEUBK model (1.3 µg/dL). The higher percentiles also differ greatly, although on a proportional basis, the estimates converge slightly across the three blood Pb models. The Lanphear et al. (1998) model predicts blood Pb levels for 16 month-old children, when exposures and blood Pb levels tend to be considerably higher than for older children, and thus the blood Pb levels that it predicts (which are considerably higher than the IEUBK concurrent estimates) at all percentiles are not directly comparable to the values presented in Exhibit 6-12. The IQ loss estimates predicted based on the Lanphear et al. (1998) model, however, should be roughly comparable, since they predict the same persistent, possibly permanent, decrements in function due to exposures at different ages.

[2] As noted in Section 6.2, modeling completed for the pilot phase has consistently shown a trend of Leggett-derived blood Pb levels being lower than IEUBK-derived levels for the same exposure scenario. This trend is the opposite of findings presented in Pounds and Leggett (1998). This apparent discrepancy elevates the importance of model evaluation focused on IEUBK and Leggett (as applied in the pilot phase) which is currently ongoing. It also emphasizes the fact that results of the pilot phase (including sensitivity analysis results) should be considered provisional and subject to update in the full-scale assessment. Note that the results of model evaluation, once completed, will be conveyed by separate cover.

Exhibit 6-12. Blood Pb Distributions Generated by the Leggett and IEUBK Models

Percentile	Blood Pb Model		Change versus Baseline
	Leggett (1993) Model[a,b]	Baseline (IEUBK)[a]	Leggett Model
99.9th	13.9	17.0	-18%
99.5th	6.7	10.5	-36%
99th	4.2	7.3	-42%
95th	2.0	3.6	-46%
90th	1.5	2.8	-47%
75th	1.0	1.9	-47%
Median	0.7	1.3	-46%
25th	0.5	0.9	-46%
1st	0.2	0.4	-46%

[a] Exposure factor values for the Leggett model were selected so as to produce equivalent Pb uptake for each exposure pathway as those used in the IEUBK model, concurrent blood Pb (6-7 years).
[b.] Blood Pb estimate for 16-month old child, based on soil and dust pathways only.

Exhibit 6-13 summarizes the differences in estimated IQ losses when the three different blood Pb models are used. IQ losses are predicted above the 95th percentile by the Leggett model, above the 75th percentile by the IEUBK model, and by the Lanphear et al. (1998) model.[3] The maximum (99.9th percentile) IQ losses predicted are not that different across the three models because the slope of the log-linear IQ model decreases rapidly with increased blood Pb.

[3] The IQ loss blood Pb cutoff (4.0 µg/dL) and slope factor (-2.85) corresponding to the "peak" Pb blood model of Lanphear et al. (2005) were used to calculate IQ loss based on the Lanphear et al. (1998) blood Pb model for 16-month old children. The model for concurrent blood Pb (cutoff = 2.4 µg/dL, slope factor = -2.7) was used to estimate IQ losses associated with blood Pb predictions from the IEUBK and Leggett models.

Exhibit 6-13. IQ Loss Estimates for Primary Pb Smelter Current Conditions Derived Using the Leggett, IEUBK, and Lanphear et al. (1998) Blood Pb Models[a]

Percentile	Blood Pb Model			Change versus Baseline	
	IQ Change Based on Leggett (1993) Model	IQ Change Based on IEUBK Blood Pb Estimates	IQ Change Based on Lanphear et al. (1998) Empirical Model	Effect of Using Leggett Model Blood Pb	Effect of Using Lanphear et al. (1998) Model Blood Pb
99.9th	-4.7	-5.3	-5.1	-11%	-4%
99.5th	-2.8	-4.0	-3.7	-31%	-8%
99th	-1.5	-3.0	-2.9	-50%	-2%
95th	--	-1.1	-1.4	-100%	27%
90th	--	-0.5	-0.8	-100%	55%
75th	--	--	--	--	--
Median	--	--	--	--	--
25th	--	--	--	--	--
1st	--	--	--	--	--

[a] Log-linear model IQ loss estimates are reported throughout.

6.4.3 Alternative Diet and Water Absorption Fractions

As noted in Section 6.2, policy-relevant background pathways (specifically, diet and drinking water) contribute a substantial proportion of total Pb uptake in all of the case studies. Under the primary Pb smelter current conditions scenario, diet and drinking water account for more than half of total Pb uptake at the low end of the blood Pb distribution, and lesser proportions at higher percentiles as the amount of uptake from other pathways increases.

Dietary and drinking water Pb uptake is determined by several sets of model inputs, including age-specific dietary intakes, drinking water Pb concentrations, and drinking water consumption. In addition, the GI uptake fractions (i.e., absorption fraction for dietary Pb [AFDiet] and absorption fraction for Pb in drinking water [AFWater]) determine the proportion of ingested Pb that is absorbed. For the pilot phase, the simplest approach to varying these policy-relevant background pathway Pb uptakes was determined to be adjusting AFDiet and AFWater values in parallel for both pathways. In the baseline analysis, AFDiet and AFWater are both 0.5 (50 percent). For the sensitivity analysis, the values were varied downward to 40 percent and upward to 60 percent. These values were thought to be reasonable lower- and upper-end estimates based on a review of recent literature (USEPA 2006b; Maddaloni et al. 2005).

Increasing or decreasing the drinking water and dietary absorption fractions to these values has moderate impacts on the blood Pb distributions (Exhibit 6-14). Median concurrent blood Pb estimates for the low, baseline, and high estimates are 1.0, 1.3, and 1.5 µg/dL, respectively. For the higher blood Pb percentiles, the estimates derived with the low AFDiet and AFWater values are about 70 percent of those estimated using the baseline values, and the estimates derived with the high-end values are approximately 15 percent higher than the baseline.

Exhibit 6-14. Impacts of Changing Diet and Drinking Water GI Absorption Fractions on Estimated Blood Pb Distributions[a]

Percentile	GI Absorption Fractions for Drinking Water, Diet			Change Relative to Baseline	
	AFDiet, AFWater = 40%	Baseline (AFDiet, AFWater = 50%)[1]	AFDiet, AFWater = 60%	AFDiet, AFWater = 40%	AFDiet, AFWater = 60%
99.9th	12.4	17.0	21.7	-27%	28%
99.5th	6.3	10.5	11.9	-40%	13%
99th	4.5	7.3	8.5	-38%	17%
95th	2.5	3.6	4.0	-30%	11%
90th	2.1	2.8	3.1	-28%	9%
75th	1.5	1.9	2.2	-25%	12%
Median	1.0	1.3	1.5	-23%	12%
25th	0.7	0.9	1.0	-20%	13%
1st	0.3	0.4	0.4	-20%	14%

[a]All estimates are for the primary Pb smelter current conditions scenario using IEUBK concurrent blood Pb metrics.

Exhibit 6-15 shows the corresponding impacts of variations to AFDiet and AFWater on IQ loss distributions for the primary Pb smelter current conditions scenarios, based on concurrent blood Pb estimates from the IEUBK model. When low AF values are used, IQ loss is predicted for children above the 90th percentile. When the baseline and higher values are used, IQ losses are predicted for children above the 75th percentile. The 99.9th percentile IQ losses range from 4.4 (low AF values) to 5.9 points (high AF values).

Exhibit 6-15. Impacts of Changing Diet and Drinking Water GI Absorption Fractions on Estimated IQ Loss Distributions[a]

Percentile	GI Absorption Fractions for Water, Diet			Change Relative to Baseline	
	AFDiet, AFWater = 40%	Baseline (AFDiet, AFWater = 50%)[1]	AFDiet, AFWater = 60%	AFDiet, AFWater = 40%	AFDiet, AFWater = 60%
Maximum	-6.6	-7.0	-8.8	-5%	27%
99.9th	-4.4	-5.3	-5.9	-16%	13%
99.5th	-2.6	-4.0	-4.3	-35%	9%
99th	-1.7	-3.0	-3.4	-44%	14%
95th	-0.1	-1.1	-1.4	-88%	25%
90th	--	-0.5	-0.7	-100%	52%
75th	--	--	--	--	--
Median	--	--	--	--	--
25th	--	--	--	--	--
1st	--	--	--	--	--

[a] All estimates are derived using IEUBK concurrent blood Pb distributions, log-linear blood Pb-IQ model.

6.4.4 Alternative Values for Soil and House Dust Ingestion and for Soil and Dust Absorption Fractions

Soil and house dust exposures play an important role in determining Pb uptake and blood Pb distributions in the primary Pb smelter case study, particularly in the highly-exposed blocks and block groups near the facility. Thus, alternative assumptions relating to soil and dust ingestion rates and GI absorption fractions were examined to determine their impacts on blood Pb and IQ loss distributions.

Von Lindern et al. (2003) developed a combination of parameter estimates related to soil and dust Pb contributions to blood Pb based on a comprehensive statistical analyses of environmental and blood Pb measurements taken over a period of 12 years near a large mining and smelting "Superfund" site in Idaho. Based on their analyses, Von Lindern et al. estimated that dust ingestion accounted for about 42 percent and soil ingestion for approximately 58 percent of the total soil plus dust influence on blood Pb levels. Therefore, the "soil dust weighting factor" input to the IEUBK model was adjusted to 58 percent from the IEUBK default value. Von Lindern et al. distinguished between "community," "neighborhood," and "yard soil" contributions to total Pb intake. No such distinction was made in this assessment – the estimated soil and dust concentrations for each block and block group were assumed to be representative of all three types of contributions.

Von Lindern et al. (2003) also estimated that the average (absolute) GI absorption fraction for Pb uptake from soil and dust was 18 percent more than the period covered by their analysis. In the sensitivity analysis, this value (18 percent) was substituted for both the site-specific soil and house dust GI absorption fraction values used in the baseline modeling, which were 26 percent and 48 percent, respectively.

Exhibit 6-16 illustrates that using the alternative soil and dust intake and uptake factors from Von Lindern et al. (2003) results in lower blood Pb estimates than the when the baseline values are used. While the higher proportion of house dust ingestion might be expected to result in higher Pb uptake, especially in the most highly exposed block groups, this potential increase is offset by the lower estimated soil and dust ingestion fractions (18 percent for both media versus 48 and 26 percent, respectively, in the baseline analysis). The median blood Pb estimate is thus reduced to 1.0 µg/dL, compared to 1.4 for the baseline parameter values, and the blood Pb percentile estimates above the median are 25 to 40 percent lower than the baseline estimates.

Use of the Von Lindern et al. (2003) soil-dust weighting factor and GI absorption fractions has a corresponding effect on the estimated IQ loss distributions (Exhibit 6-17). IQ losses are predicted above the 90^{th} percentile when the Von Lindern et al. parameters are used. When the baseline parameter estimates are used, IQ losses are predicted for children above the 75^{th} percentile. The 99.9^{th} percentile IQ change under the baseline is 5.3 points, whereas, when the alternative values for the soil-dust weighting factor and absorption fraction are used, the corresponding IQ loss estimate is 4.4 points.

Exhibit 6-16. Effects of Von Lindern et al. (2003) Soil and Dust Intake Parameters on Estimated Blood Pb Distributions, Primary Pb Smelter Case Study

Percentile	Soil, House Dust Exposure Factor Values		Change from Baseline
	Baseline Exposure Factors[a]	Von Lindern et al. Exposure Factors[b]	Alternative Exposure Factors
99.9th	17.0	12.4	-27%
99.5th	10.5	6.3	-40%
99th	7.3	4.5	-38%
95th	3.6	2.5	-30%
90th	2.8	2.1	-28%
75th	1.9	1.5	-25%
Median	1.3	1.0	-23%
25th	0.9	0.7	-20%
1st	0.4	0.3	-20%

[a]Soil-Dust Weighting factor = 45%, AFSoil = 26%, AFDust = 48 %
[b]Soil-Dust Weighting factor = 58%, AFSoil = AFDust = 18 %

Exhibit 6-17. Effects of Von Lindern et al. (2003) Soil and Dust Intake Parameters on Estimated IQ Loss Distributions, Primary Pb Smelter Case Study[a]

Percentile	Soil, House Dust Exposure Factor Values		Change from Baseline
	IQ Change Derived from Baseline Exposure Factors[a,b]	IQ Change Derived from Von Lindern et al. Exposure Factors[a,b]	Alternative Exposure Factors
99.9th	-5.3	-4.4	-16%
99.5th	-4.0	-2.6	-35%
99th	-3.0	-1.7	-44%
95th	-1.1	-0.1	-88%
90th	-0.5	--	-100%
75th	--	--	--
Median	--	--	--
25th	--	--	--
1st	--	--	--

[a] Log-linear blood Pb-IQ loss model
[b] See notes to Exhibit 6-16

6.4.5 Alternative Blood Pb Geometric Standard Deviations

To this point, the sensitivity analysis has addressed models and parameters that affect the GM blood Pb estimates for the individual blocks and block groups. The parameter to be evaluated next, the GSD for individual variability, affects the distribution of individual blood Pb values derived from the block and block group GM values.

The baseline (central tendency) estimate for the blood Pb GSD used in all three case studies is 1.6 (see Section 5.1.4.4). Based on a review of the literature conducted for this assessment, GSD values for children in limited geographical areas were found to range from approximately 1.3 to 1.9. Larger variability has been seen in national and regional data (CDC 2004; Maddaloni 2005), but it is likely that the children in these populations faced more varied exposures than those in smaller populations. The values 1.3 and 1.9 were thus considered to be plausible "lower" and "upper" estimates of the GSD for individual variability, with no implication intended that they represent absolute limits or defined percentiles of actual population variability.

As expected, changing the GSD value leaves the median blood Pb estimates unchanged (Exhibit 6-18) and reducing the GSD value to 1.3 results in the upper percentile values that are lower than the corresponding baseline estimates, with the proportional decrease increasing toward the "tail" of the analysis. The pattern is clear up to the 99th percentile, but the 99.5 and 99.9th percentile blood Pb estimates calculated with the lower GSD value are similar to, or greater than, those calculated with the baseline values. This is certainly an artifact due to simulation uncertainty (see Exhibit 5-13).

Exhibit 6-18. Effect of Varying Individual Variability Estimates on Predicted Blood Pb Distributions[a]

Percentile	Blood Pb Individual Variability			Change Relative to Baseline	
	GSD = 1.3	Baseline (GSD = 1.6)	GSD = 1.9	GSD = 1.3	GSD = 1.9
99.9th	16.4	17.0	22.8	-3%	34%
99.5th	10.9	10.5	13.5	4%	29%
99th	6.8	7.3	9.8	-6%	35%
95th	2.9	3.6	4.7	-19%	31%
90th	2.3	2.8	3.5	-18%	23%
75th	1.7	1.9	2.2	-11%	13%
Median	1.3	1.3	1.3	0%	3%
25th	1.0	0.9	0.8	13%	-8%
1st	0.6	0.4	0.3	62%	-32%

[a] Concurrent blood Pb estimates are calculated for the primary Pb smelter current conditions scenario using the IEUBK model.

Blood Pb estimates above the median are consistently elevated compared to the baseline when the higher estimate for population variability (1.9) is used to calculate individual blood Pb levels. The estimates for the upper percentiles derived using the upper GSD estimate are on the order of 30 to 35 percent higher than the corresponding baseline values.

Exhibit 6-19 shows the differences in estimates of IQ loss when the alternative GSD values are used. For the low GSD estimate, IQ losses are predicted for children above the 90th percentile, while for the baseline and high-end GSDs, IQ losses are predicted for children above the 75th percentile. The highest predicted IQ losses derived using the three GSD estimates are quite similar (5.2, 5.3, and 6.1, respectively), the closeness of the estimates again reflecting the shallow slope of the log-linear blood Pb-IQ loss model at high blood Pb levels.

Exhibit 6-19. Effect of Changing the GSD for Individual Variability on Predicted IQ Loss Distributions[a]

Percentile	Blood Pb Individual Variability			Change Relative to Baseline	
	GSD = 1.3	Baseline (GSD = 1.6)	GSD = 1.9	GSD = 1.3	GSD = 1.9
Maximum	-6.7	-7.0	-10.6	-3%	53%
99.9th	-5.2	-5.3	-6.1	-2%	15%
99.5th	-4.1	-4.0	-4.7	3%	17%
99th	-2.8	-3.0	-3.8	-6%	27%
95th	-0.5	-1.1	-1.8	-51%	66%
90th	--	-0.5	-1.0	-100%	124%
75th	--	--	--	--	--
Median	--	--	--	--	--
25th	--	--	--	--	--
1st	--	--	--	--	--

[a] IQ loss estimates are derived based on concurrent IEUBK blood Pb estimates for the primary Pb smelter current conditions scenario.

6.4.5 Alternative Blood Pb-IQ Models and Uncertainty Related to Statistical Fit of Models

To this point, all of the IQ loss estimates have been calculated based on the log-linear version of the Lanphear et al. (2005) model. For the sensitivity analysis, three alternative parameterizations of the blood Pb-IQ models were used to calculate IQ changes. The first two sets of estimates were derived using the estimated upper and lower confidence limits on the log linear slope parameters from the concurrent blood Pb-IQ model derived by Lanphear et al. (2005). The equations for these model were:

$$\text{IQ loss (relative to cutpoint)} = -3.74 \times \ln(\text{concurrent blood Pb} / 2.4)$$

$$\text{IQ loss (relative to cutpoint)} = -1.66 \times \ln(\text{concurrent blood Pb} / 2.4)$$

with the first being the "upper" confidence limit (the value that predicts the largest change in IQ), and the second being the lower confidence limit.

The third alternative was the piecewise linear form of the model presented in Lanphear et al. (2005):

$$\text{Concurrent Blood Pb} < 2.4\ \mu g/dL: \quad \text{IQ change not calculated}$$

$$\text{Concurrent Blood Pb} < 10\ \mu g/dL: \quad \text{IQ} = 97.2 - 0.80 \times \text{Blood Pb}$$

$$\text{Concurrent Blood Pb} > 10\ \mu g/dL: \quad \text{IQ} = 89.2 - 0.13 \times \text{Blood Pb}$$

The intercept in the third equation is simply the IQ loss estimated from the second equation at a blood Pb concentration of 10 µg/dL. Thus, the form of this model is a "hockey-stick" with a

slope of -0.80 below 10 µg/dL, and a slope of -0.13 above 10 µg/dL. As for the log-linear model, IQ changes were not estimated if individual blood Pb levels were below the concurrent blood Pb cutpoint discussed above.

Exhibit 6-20 summarizes the IQ loss distributions derived using the alternative blood Pb-IQ models with the concurrent blood Pb metric. In all cases, block and block group blood Pb concentrations were calculated with IEUBK, and a GSD value of 1.6 was used to describe individual variability in response to Pb exposures. Of the alternatives evaluated, the predictions generated by the piecewise linear model differ the most from the baseline estimates. The 90th and 95th percentile IQ losses predicted by the two models are quite similar, but the 99th and higher percentile IQ loss predictions from the piecewise model are on the order of twice those from the log-linear model. This is due to difference in form and the much higher slope of the piecewise linear model in the range from the blood Pb cutpoint to 10 µg/dL.

The lower and upper confidence limit model predictions parallel the predictions from the baseline model, as expected. The lower confidence limit (LCL) model predicts IQ losses for children above the 75th percentile, as high as 1.2 points at the 99.9th percentile. The upper confidence limit (UCL) model, like the baseline estimate, predicts IQ losses for children above the 75th percentile, but the maximum (99.9th percentile) IQ loss is predicted to be on the order of 8.1 points, compared to about 5.3 points for the baseline model.

Exhibit 6-20. Effects of Alternative Blood Pb-IQ Models on Predicted IQ Distributions

Percentile	Blood Pb-IQ Model				Change Relative to Baseline		
	95% LCL on Log-Linear Model (1.66)	Baseline (Log-Linear Model)[a]	95% UCL on Log-Linear Model (3.72)	Piecewise Linear Model	95% LCL	95% UCL	Piecewise Linear
99.9th	-1.2	-5.3	-8.1	-8.9	-78%	53%	69%
99.5th	-0.8	-4.0	-5.9	-8.1	-80%	49%	103%
99th	-0.6	-3.0	-4.3	-3.9	-80%	44%	30%
95th	-0.2	-1.1	-1.6	-1.0	-77%	48%	-12%
90th	-0.1	-0.5	-0.7	-0.4	-76%	49%	-23%
75th	--	--	--	--	--	--	--
Median	--	--	--	--	--	--	--
25th	--	--	--	--	--	--	--
1st	--	--	--	--	--	--	--

[a] All IQ Loss estimates are for the primary Pb smelter current conditions scenario, based on IEUBK concurrent blood Pb estimates.

6.4.7 Alternative Blood Pb Metrics

In Section 6.2, the concurrent and lifetime average blood Pb distributions, and IQ loss estimates generated from these distributions, were presented for each of the case studies. In addition, the impact of using the "peak" blood Pb estimates (the highest annual average, which was found to occur at ages 6 through 18 months) on the magnitude of IQ loss predictions was also evaluated. The Lanphear et al. (2005) log-linear model form was used to derive all of these results. The concurrent, lifetime average, and peak blood Pb-based models, while sharing the same mathematical form, differ in two respects. First, they have different slope parameters (2.7, 3.04, and 2.85, respectively for the concurrent, lifetime average, and peak blood Pb metrics). In addition, the cutpoint values below which IQ losses are not calculated varies across the models (see Section 6.1.2). The cutpoint for concurrent blood Pb is 2.4 µg/dL, the cutpoint for lifetime average blood Pb is 6.1 µg/dL, and the cutpoint for peak exposures is 4.0 µg/dL, corresponding to the 5th percentile blood Pb values measured in the Lanphear et al. (2005) pooled cohort.

Exhibit 6-21 shows that both the lifetime average and peak blood Pb values generated using the IEUBK model are substantially greater than the baseline (concurrent) estimates. This is because the lifetime average metric contains a substantial contribution from younger ages, when blood Pb is estimated to be higher than for the concurrent children (aged 6 to 7 years). The peak (highest annual) blood Pb is, by definition, taken from the exposure period when blood Pb levels are highest. Across the percentiles, the lifetime average blood Pb estimates are approximately 35 to 40 percent higher than the corresponding baseline values, and the peak estimates are about 80 percent higher. These observations are consistent with published findings in the literature regarding varying blood Pb levels with age (USEPA 2006a).

Exhibit 6-21. Distributions of Concurrent, Lifetime Average, and Peak Blood Pb Distributions from the Primary Pb Smelter Case Study

Percentile	Blood Pb Metric		
	Lifetime Average Blood Pb, µg/dL	Baseline (Concurrent Blood Pb), µg/dL[a]	Peak (Highest Annual) Blood Pb, µg/dL[b]
99.9th	23.2	17.0	33.4
99.5th	14.3	10.5	18.0
99th	10.4	7.3	13.4
95th	5.1	3.6	6.7
90th	4.0	2.8	5.3
75th	2.7	1.9	3.5
Median	1.8	1.3	2.4
25th	1.2	0.9	1.7
1st	0.5	0.4	0.7

[a]IEUBK concurrent (age 6 to 7 years) blood Pb estimates.
[b]IEUBK highest 12-month average (age 6 to 18 months) blood Pb estimates.

These differences in blood Pb predictions strongly affect the magnitude of predicted IQ losses (Exhibit 6-22), as do the differences in the model parameters. Although the lifetime average blood Pb percentiles are higher than the baseline concurrent estimates, the magnitude of predicted IQ loss is lower. When interpreting the results in Exhibit 6-22, it is important to remember that IQ changes are calculated using different blood Pb cutpoints and slope factors for each blood Pb metric.

When the lifetime average blood Pb distribution is input into the lifetime average blood Pb-IQ model, only children above the 95th percentile are predicted to experience IQ changes, and the greatest (99.9th percentile) estimate is an IQ loss of 4.1 points. These IQ changes are significantly lower than the baseline predictions based on the concurrent blood Pb distributions. The primary reason for this pattern is the relatively high lifetime average blood Pb cutpoint (6.1 µg/g), compared to the concurrent cutpoint of 2.4 µg/dL.

Exhibit 6-22. Effects of Using Different Blood Pb Metrics to Predict IQ Loss

Percentile	Blood Pb Metric			Difference from Concurrent	
	IQ Change Based on Lifetime Average Blood Pb[a]	IQ Change Based on Concurrent Blood Pb[b]	IQ Change Based on Highest Annual Blood Pb[c]	Lifetime Average	Peak
99.9th	-4.1	-5.3	-7.5	-23%	42%
99.5th	-2.6	-4.0	-5.7	-35%	44%
99th	-1.6	-3.0	-4.9	-46%	64%
95th	--	-1.1	-2.9	-100%	165%
90th	--	-0.5	-2.3	-100%	395%
75th	--	--	-1.1	--	--
Median	--	--	--	--	--
25th	--	--	--	--	--
1st	--	--	--	--	--

[a] IQ estimates derived using Lanphear et al. (2005) log-linear model using parameter values for lifetime average blood Pb levels.
[b] Derived using Lanphear et al. (2005) parameter values for concurrent blood Pb levels.
[c] Derived using Lanphear et al. (2005) parameter values for peak blood Pb exposures.

In contrast, the IQ losses predicted based on the distribution of peak blood Pb levels are considerably higher than the baseline estimates. IQ losses are predicted for children above the 50th percentile blood Pb, and range from 1.1 points (75th percentile) to 7.5 points (99.9th percentile). In the case of the peak blood Pb values, the difference between the blood Pb cutoff values is not great enough to overcome the effect of much greater peak blood Pb estimates.

6.4.8 Alternative Blood Pb Cutpoint for IQ Loss Estimation

Because of the impact on IQ loss estimates of the blood Pb cutoff values used as the "floor" for estimating IQ loss, the impact of changing these values was also investigated. The data available from the Lanphear et al. (2005) study does not provide a foundation for alternative cutpoints for use in sensitivity analysis. For the pilot phase, the simple expedient of reducing the cutpoints for

concurrent and lifetime average blood Pb by 50 percent was adopted. The slope parameters for the two models remained unchanged.

Exhibit 6-23. Effect of Reducing Blood Pb Cutoff Values for IQ Estimation[a]

Percentiles	Blood Pb Cutpoint		Change from Baseline
	Baseline Blood Pb Thresholds	Thresholds = 1/2 Baseline Values	Reduced Cutpoint
99.9th	-5.3	-8.3	57%
99.5th	-4.0	-6.3	58%
99th	-3.0	-5.0	68%
95th	-1.1	-3.0	171%
90th	-0.5	-2.3	413%
75th	--	-1.3	--
Median	--	-0.3	--
25th	--	--	--
1st	--	--	--

[a] All IQ loss estimates are derived from IEUBK blood Pb estimates for the primary Pb smelter current conditions scenario.

As shown in Exhibit 6-23, when the concurrent blood Pb cutpoint is reduced from 2.4 to 1.2 µg/dL, children above the 25th (rather than the 75th) percentile in the primary Pb smelter case study are predicted to experience IQ loss, and the maximum (99.9th percentile) IQ loss using IEUBK and concurrent blood Pb blood Pb-IQ function is increased from 5.3 to 8.3 points. A change in the cutpoint for the lifetime blood Pb metric would be expected to have similar impacts on the IQ changes.

6.4.9 Summary of Sensitivity Analysis

The preceding discussion demonstrates a wide range of impacts on blood Pb and IQ loss distributions associated with alternative model selection and parameter values. Exhibit 6-24 summarizes the impacts of the variations in models and input assumptions on predicted blood Pb and IQ distributions. Changes in the median 90th, 95th, and 99th percentile values relative to the corresponding baseline estimates are displayed for each set of models and parameter values that were varied.

As expected, the predicted changes in blood Pb and IQ values in Exhibit 6-24 tend to move in parallel, although with a reverse sign. However, variations in models or parameter values affect the predicted blood Pb or IQ to varying degrees owing to the non-linearity of the statistical blood Pb-IQ models and variations in the blood Pb-IQ loss cutpoints when different blood Pb metrics (concurrent, lifetime average, or peak) are used. Many changes in assumptions that result in significant decreases in predicted blood Pb levels accordingly result in a lesser predicted IQ change, or no predicted IQ change at some percentiles. Some relatively small changes in blood Pb associated with the sensitivity analysis abolish the estimated IQ loss entirely (identified as "-100%" changes in Exhibit 6-24) because the predicted blood Pb falls below the cutpoint. In such cases, both the numbers of children for which IQ losses are predicted and the magnitude of IQ

losses among the affected children are reduced. It is also important to remember that the proportional changes displayed in the exhibit apply to much smaller absolute blood Pb and IQ loss values at the median and 90th percentile than at the 95th and 99th, owing to the "skewed" nature of the blood Pb and IQ loss distributions.

The factors most strongly affecting the magnitude of IQ loss estimates are the nature of the blood Pb metric from which they are estimated and assumptions related to the blood Pb cutpoint for IQ loss estimation. The estimated GSD values for individual variability in response to Pb exposures also strongly affect the predicted IQ loss distributions, particularly at the upper percentiles. Among the factors related to blood Pb modeling, the selection of blood Pb model has the most impact on estimated IQ changes. Relatively modest changes in GI absorption changes for diet and water Pb surprisingly produce relatively large proportional changes in the predicted blood Pb and IQ distributions, as do changes in soil/dust exposure factor values. As noted, above, however, the highest proportional changes associated with these modeling alternatives are seen at the lower percentile values, where the absolute magnitudes of the change from the baseline can be rather small.

Exhibit 6-24. Impacts of Alternative Models and Parameters on Blood Pb and IQ Loss Estimates in the Sensitivity Analysis.

Percentile	Change Versus Baseline							
	House Dust Pb Concentration Model	**Blood Pb Estimation Model**	**Water, Diet Absorption Fractions**	**Soil, Dust Exposure Intake/Uptake Factors**	**Individual Blood Pb Variability (GSD)**	**Blood Pb-IQ Model**	**Blood Pb Metric**	**Reduced Blood Pb Cutpoint**
Blood Pb								
99th	-15%	-42% to +58%	-38% to +17%	-38%	-6% to +35%	--	+43% to +84%	--
95th	-5% to -7%	-46% to 84%	-30% to +11%	-30%	-19% to +31%	--	+42% to +85%	--
90th	-8% to -7%	-47% to 84%	-28% to +9%	-28%	-18% to +23%	--	+41% to +86%	--
Median	-12% to -10%	-46% to +108%	-23% to +12%	-23%	0% to 3%	--	+37% to +83%	--
IQ Loss								
99th	-14% to -15%	-50% to -2%	-44% to +14%	-44%	-6% to +27%	-80% to +44%	-46%to +64%	+68%
95th	12% to 17%	-100% to +27%	-88% to +25%	-88%	-51% to +66%	-77% to +48%	-100% to +165%	+171%
90th	43% to 46%	-100% to +55%	-100% to +52%	-100%	-100% to +124%	-76% to +49%	-100% to +365%	+413%
Median	--	--	--	--	--	--	--	--

7. Ecological Risk Assessment

The ecological risk assessment for this NAAQS review consisted of three case study screening assessments and a national-scale surface water and sediment screening assessment. The case study screening assessments were designed to estimate the potential for ecological risks associated with exposures to Pb emitted into ambient air for three case studies: primary Pb smelter, secondary Pb smelter, and near roadway non-urban location. The national-scale screening assessment evaluated the potential for ecological risks associated with the atmospheric deposition of Pb released into ambient air at surface water and sediment monitoring locations across the United States. This chapter describes the methods applied for this assessment (Section 7.1), presents the exposure and risk results (Section 7.2), and discusses the uncertainties and limitations of the assessment (Section 7.3).

7.1 Methods

This section describes the screening methods used to determine the potential for risks of adverse effects on ecological receptors associated with exposures of Pb emitted into ambient air and subsequently deposited to other environmental media. Section 7.1.1 summarizes the risk characterization methods used to characterize ecological risks for the case studies; Section 7.1.2 describes the methods used to conduct the national-scale surface water and sediment screening assessment; and Section 7.1.3 describes the ecological effects characterization methods and the ecotoxicity screening values used in this assessment.

7.1.1 Case Studies

The Hazard Quotient (HQ) approach was used to compare estimated media concentrations with ecotoxicity screening values in soils, surface waters, and sediments around the primary Pb smelter, and for soils only around the secondary Pb smelter case study location and the near roadway non-urban case study locations. The HQ is calculated as the ratio of the media concentration to the ecotoxicity screening value. The HQ is represented by the following equation:

$$HQ = \text{(estimated media concentration)} / \text{(ecotoxicity screening value)}$$

For each case study, HQ values were calculated for each location where either modeled or measured media concentrations were available. Separate soil HQ values were calculated for each ecological receptor group for which an ecotoxicity screening value has been developed (i.e., birds, mammals, soil invertebrates, and plants). HQ values less than 1.0 suggest that Pb concentrations in a specific medium are unlikely to pose significant risks to ecological receptors whereas HQ values greater than 1.0 indicate that the expected exposure exceeds the ecotoxicity screening value.

7.1.2 National-Scale Surface Water and Sediment Screening Assessment

The HQ method was used to characterize risk for the surface water portion of the national-scale surface water and sediment screening assessment, where measures of dissolved Pb

concentrations in surface waters at monitoring locations across the United States were compared with both acute and chronic water column (surface water) ecotoxicity screening values. To identify which sites might pose a risk to aquatic communities, the frequencies with which dissolved Pb concentrations exceeded the selected chronic and acute surface water ecotoxicity screening values for the protection of aquatic life (described in Section 7.1.3.2) were examined. Because of uneven sampling frequency across sampling locations, a screen was implemented to exclude from further analysis sites expected to pose negligible risks to aquatic communities. Locations were eliminated if the frequency with which a sample exceeds the chronic surface water screening value for dissolved Pb was 5 percent or less of available data, and the acute screening value was never exceeded. The sampling locations that were dropped from the assessment using this screen are identified in Appendix H with the code "na" in the column indicating the number of Pb measurements that exceeded the screening value out of the total number of samples analyzed for dissolved Pb.

As described in Section 7.1.3.2, a review of the data on water hardness in the NAWQA data set indicated that the initial chronic screening value was too high to identify all locations where the chronic surface water screening value for the protection of aquatic life was exceeded. As a result, a second screen was conducted. In the second screen, for each sampling location with one or more measured dissolved Pb concentrations above the QL but less than 1.2 µg/L (the initial screening value assuming a water hardness of 50 µg/L $CaCO_3$), measurements of $CaCO_3$ were used to calculate a location-specific chronic surface water screening value. If none of the dissolved Pb measurements at a particular location exceeded this location-specific value, the location was not considered further. If one or more of the dissolved Pb measurements at a particular location did exceed the location-specific value, the location was retained for further analysis.

Where dissolved Pb concentrations were found to exceed the chronic surface water screening values at a frequency of greater than 5 percent of available data or where the acute surface water screening value was exceeded by at least one measurement, the next step involved identifying the most likely sources of Pb input to the surface waters to focus the screen on those locations for which air is a major contributor to the Pb concentrations in the water column. The following considerations were used to exclude sites for which air contributions could not be apportioned.

- Where the surrounding land use was for mining, the Pb in the surface water could not be apportioned between runoff/erosion and air emissions. Therefore, these sites were categorized separately.

- If there is a major permitted effluent discharge upstream of a sampling location with Pb specified in the permit, it is possible that this discharge was responsible for some of the Pb measured in the surface water. Therefore, air contributions could not be readily apportioned.

- If there were no facilities known to emit Pb to the air within a 50-km radius of the sampling location, the Pb in the surface water was assumed to come largely from non-air sources.

Based on this screen of monitoring locations, 15 non-mining surface water locations were identified with elevated Pb concentrations possibly attributable to policy-relevant sources. Risk characterization for the sediment portion of the screening assessment was conducted for these 15 locations. This risk characterization was performed using the HQ method, where measures of total Pb concentrations in sediments were compared with the sediment ecotoxicity screening values for the protection of benthic communities. Concentrations that exceeded the sediment screening values were identified and possible sources of Pb input to these sediments were examined. The first step involved identifying sediment sampling locations in the NAWQA database that were co-located with the surface water samples identified for the surface water portion of the screening assessment. Three of the 15 sites did not have co-located sediment and surface water results. In these cases, sediment sampling results from nearby sampling locations were identified on the basis of latitude, longitude, and name of the site location. For the sediment sampling sites that were categorized as near matches, it is important to note that the surface water samples at the same location had not exceeded the chronic surface water screening value in the surface water portion of the screening assessment. Therefore, the value of the "near-match" locations is limited.

7.1.3 Ecotoxicity Screening Values

The sections that follow describe the ecotoxicity screening values used in this ecological risk assessment to evaluate the soil, surface water, and sediment concentrations of Pb identified in the case studies and in the national-scale screening assessment. Section 7.1.3.1 describes the soil ecotoxicity screening values developed for this assessment. Section 7.1.3.2 describes the surface water quality screening values, and Section 7.1.3.3 describes the sediment screening values.

7.1.3.1 Soil Screening Values

In identifying soil screening values (SSVs) for this assessment, the Ecological Soil Screening Levels (Eco-SSLs) developed by EPA's Superfund program (USEPA 2003; USEPA 2005c) were considered. These values are defined as "concentrations of contaminants in soil that are protective of ecological receptors that commonly come in contact with soil or ingest biota that live in or on soil." They were derived separately for four categories of ecological receptors: plants, soil invertebrates, birds, and mammals. For the purposes of Superfund screening level ecological risk assessments at individual hazardous waste sites, Eco-SSLs are used to exclude from further investigation those receptor/exposure pathway combinations that pose negligible risks. Eco-SSLs are therefore derived using conservative assumptions and parameter values.

The development of Eco-SSLs for each receptor category involved the assessment of several species within each category. Available toxicity test data for these species were evaluated. In the case of plants and soil invertebrates, these data are expressed as contaminant concentrations in soil (mg contaminant/kg soil), are evaluated with consideration of characteristics affecting bioavailability (e.g., pH, organic content, etc), and a single Eco-SSL is derived for each category. Unlike Eco-SSLs for plants and soil invertebrates, those for the avian and mammalian wildlife categories involve a two step process: (1) derivation of a toxicity reference value (TRV) in mg contaminant per kg body weight per day, and (2) application of the TRV and exposure parameter values to derive an Eco-SSL in mg contaminant per kg soil for three different species in each category. In the first step, a single toxicity reference value (TRV) was developed (e.g., the

reference dose for the most sensitive of the adverse effects on birds that might cause population-level effects) that applied to all species in each wildlife category. In the second step, however, species-specific exposure assumptions regarding incidental soil ingestion, diet composition, and contaminant uptake from soil by prey generally resulted in different Eco-SSL values, expressed as soil concentrations, for the different species in each receptor category. The receptor category Eco-SSL was then set equal to the lowest species-specific Eco-SSL.

In developing SSVs for use in the Pb NAAQS review, the exposure assumptions and parameter values used in the second step of the derivation of the Superfund Eco-SSLs for birds and mammals were examined, and as appropriate, augmented or replaced with alternatives considered more appropriate for the Pb NAAQS review (see Appendix L). SSVs for birds and mammals for this assessment were derived using the Eco-SSL methodology and TRVs and alternative values for some of the exposure parameters. For purposes of this assessment, the SSV for avian wildlife of 38 mg/kg dry weight was developed using data for the woodcock, the species on which the avian Eco-SSL was based (USEPA 2003; USEPA 2005c). A more nationally representative value was used for woodcock diet composition (i.e., 75 percent earthworms, 25 percent soil arthropods), and alternative values, supported by available data, were used for food ingestion rate (i.e., 0.20 kg food dry weight/kg-day), incidental soil ingestion rate (i.e., 11.4 percent of the dry weight diet), and absorption of ingested Pb (i.e., 50 percent). The same 50 percent absorption efficiency was used to calculate an SSV for mammals of 56 mg/kg dry weight. This absorption efficiency assumption was based on studies that suggest that the oral bioavailability of Pb in soils or food for mammals is between 3 and 50 percent of its oral bioavailability when added as soluble Pb acetate to food or water (ATSDR 2005; Ruby et al. 1999). The mammalian SSV developed for the Pb NAAQS review remains somewhat more conservative than the avian SSV because the proportion of earthworms in the diet of shrews (the mammalian species with the highest exposure) is likely less than the assumed 100 percent. Shrews also ingest arthropods, and an arthropod model for Pb uptake from soil predicts lower concentrations in the arthropods than in earthworms at the same soil concentrations.

The SSVs developed for this assessment are shown in Exhibit 7-1. For plants and soil invertebrates, the Eco-SSLs were adopted as the SSVs for the Pb NAAQS review.

Exhibit 7-1. Soil Screening Values (SSVs) for Pb

Receptor	SSV (mg Pb/kg Soil Dry Weight)	Source
Plants	120	USEPA 2005c
Soil invertebrates	1700	
Avian wildlife	38	Appendix L
Mammalian wildlife	112	

7.1.3.2 Water Quality Criteria

The AWQC were developed by U.S. EPA to provide guidance to states and authorized tribes to use in adopting water quality standards and are based on toxicity testing in aquatic organisms, including fish, invertebrates and algae. AWQC values for chronic exposures are called the criterion continuous concentration (CCC) and for acute exposures are called the criterion maximum concentration (CMC), and they are available for freshwater and marine environments. For a CCC to be exceeded, a 4-day average water concentration must exceed the CCC more than once every three years (USEPA 1984).

The freshwater AWQC, both the CCC and CMC, for Pb are derived to vary with water hardness, and when initially developed pertained to total recoverable Pb. During a re-evaluation of AWQC for metals in 1993, EPA concluded that it would be more accurate to base the AWQC on that portion of the metal in the water column that is readily bioavailable, that is, the dissolved fraction (USEPA 1993). As a result, EPA examined the relationship of total to dissolved concentrations for a series of metals to develop a conversion factor for each metal. For Pb, the conversion varies with water hardness and is the same for both the CCC and CMC (USEPA 1993). The current equations/values may be reissued based on pH at the conclusion of the current revisions to the Pb AWQC scheduled to be completed in 2007. The CCC and *CMC* are calculated using the following formulas (USEPA 1984):

$$CCC = \exp\{1.273\ [ln(hardness)] - 4.705\} \times CF$$
$$CMC = \exp\{1.273\ [ln(hardness)] - 1.460\} \times CF$$

where:

CF = conversion factor = 1.46203 -[*ln(hardness)* x 0.145712]
Hardness = mg/L of calcium carbonate ($CaCO_3$).

For the primary Pb smelter case study, freshwater AWQC were calculated based on water body-specific measured hardness values for the water bodies located in the vicinity of the facility that were sampled and analyzed for Pb (ELM 2005). Exhibit 7-2 lists these site-specific AWQC.

Exhibit 7-2. Calculated Pb AWQC for Water Bodies Analyzed for the Primary Pb Smelter Case Study

Sample Cluster or Area	Hardness[a] (mg $CaCO_3$/L)	Chronic AWQC for Pb - CCC (μg/L)	Acute AWQC for Pb - CMC (μg/L)
Joachim Creek			
Cluster 1	283.3	7.7	197
Cluster 2	276.7	7.5	192
Cluster 3	281.7	7.6	195
Cluster 4	272.2	7.3	188
Cluster 5	273.3	7.4	189
Mississippi River			
Upstream	210	5.6	143
Near facility	227.5	6.1	156
Downstream	235	6.3	161
Pond and Drainage Areas			
U-shaped pond cluster	166.67	4.4	112
CHRDDP	465	12.8	328
RRDP-02	240	6.4	165
DAMUP	280	7.6	195

[a] Hardness measures were taken from ELM (2005).
CCC = criterion continuous concentration (chronic criterion), U.S. EPA's National AWQC for Pb
CMC = criterion maximum concentration (acute criterion), U.S. EPA's National AWQC for Pb

To simplify the national-scale surface water screening assessment, a more conservative freshwater CCC of 1.2 μg/L dissolved Pb associated with a low water hardness of 50 mg/L $CaCO_3$ was used as an initial screening step. At sampling locations where that value was exceeded, the next step was to find co-located measurements of water hardness (expressed as mg/L of $CaCO_3$) and to use those values to calculate a location-specific CCC and CMC for the identified sampling location.

Location-specific CCC and CMC values for the national-scale screening assessment were derived based on water hardness measurements available for those locations. The number of water hardness measurements at a given sampling station in the NAWQA data set ranged from zero to more than 60. Where there were two or more measurements of water hardness, the arithmetic mean of the available measurements across all years was used to represent the water hardness at the sampling location. Where there were no analyses of water hardness at a given sampling location, the next closest sampling station (from station ID number and latitude and longitude) was identified and water hardness associated with this station was used for the sampling location in question. Where there was a single value reported for water hardness, that value was used without comparing it to values for any next nearest sampling locations.

Rather than discard any sampling stations from the screening assessment, the use of single or nearby water hardness measurements to calculate a CCC and CMC for a location was deemed acceptable. For sampling stations with more than two measurements of water hardness, the

arithmetic mean of the measurements is the measure most consistent with the limited data at other stations.

A review of the data on water hardness in the NAWQA data set for 1994 to 2004 indicated that the initial screening value of 1.2 µg/L was too high to identify all locations for which dissolved Pb concentrations exceeded the CCC for the protection of aquatic life. Many waters in the United States are softer than originally thought (i.e., measured $CaCO_3$ concentrations were as low as 1 mg/L). A second screening was therefore conducted in which dissolved Pb measurements greater than the quantitation limit (QL) but less than 1.2 µg/L were reviewed.

In the second screen, for each sampling location with one or more measured dissolved Pb concentrations above the QL but less than 1.2 µg/L, measurements of $CaCO_3$ were used to calculate a location-specific CCC as described above.[1] If none of the dissolved Pb measurements at a particular location exceeded its location-specific CCC, the location was not considered further. If one or more of the dissolved Pb measurements at a particular location did exceed the location-specific CCC, the location was retained for further analysis. The Pb measurements for the additional locations identified in the second screening are provided in Appendix H.

7.1.3.3 Sediment Screening Values

A number of sediment ecotoxicity values were considered in identifying chronic sediment screening values for this assessment. These included several types of criteria for freshwater sediments derived using different approaches; Exhibit 7-3 lists the candidate sediment screening values; Appendix M reviews the details of their derivation. The EPA approach for sediment quality criteria (i.e., equilibrium partitioning sediment benchmarks) (USEPA 2005d) utilizes EPA's AWQC with site-specific information on acid volatile sulfides and organic carbon. As such site-specific information was not available for the case study location at which sediment was assessed, or for the national-scale screening assessment, sediment quality assessment guidelines developed by MacDonald et al. (2000; reported in MacDonald et al. 2003) were identified for use in this ecological risk assessment at all case study locations and in the national-scale screening assessment.

The MacDonald et al. (2000; 2003) guidelines include two values for Pb: (1) a threshold effect concentration (TEC) of 35.8 mg/kg and (2) a probable effect concentration (PEC) of 128 mg/kg. The TEC was derived as the geometric mean of literature values described as being at the lower end of the effects continuum. The PEC was derived as the geometric mean of literature values described as more prevalently associated with adverse effects. Appendix M provides more details on their basis. These values are considered to be applicable to the United States (and Canada).

[1] For each sampling site, the arithmetic mean of all $CaCO_3$ samples, the single hardness measurement, or the water hardness from a nearby sampling site was used.

Exhibit 7-3. Pb Sediment Screening Values Evaluated for Use in This Ecological Risk Assessment

Description	Value (mg Pb/kg Dry Sediment)
EPA Region 4 Sediment Screening Value (derived by MacDonald 1994)	30.2
EPA Region 5 Ecological Screening Level for Sediment (adopted consensus TEC from MacDonald et al. 2000; see below)	35.8
EPA Region 6 Ecological Benchmark for Sediment – Freshwater (TNRCC 2001)	35
Consensus TEC for freshwater sediments (MacDonald 2000)	35.8
Consensus PEC for freshwater sediments (MacDonald 2000)	128
Equilibrium partitioning (USEPA 2005d) - Low risk of adverse biological effects	Requires site-specific measures of acid volatile sulfide and organic carbon fraction in the sediment
Equilibrium partitioning (USEPA 2005d) - May have adverse biological effects	
Equilibrium partitioning (USEPA 2005d) - Adverse biological effects expected	

Abbreviations:
TNRCC = Texas National Resource Conservation Commission; TEC = Threshold Effect Concentration; Consensus PEC = Probable effect concentration (sediment concentration above which harmful effects are likely to be observed, MacDonald et al. [2000]); Consensus TEC = Threshold effect concentration (sediment concentration below which harmful effects are unlikely to be observed, MacDonald et al. [2000])

The consensus-based TEC for Pb was calculated as the geometric mean of five literature values, ranging from a threshold effect level of 35 mg/kg from Smith et al. (1996) to a minimal effect threshold of 42 mg/kg from Environment Canada (EC) and Ministere de l'Environnement du Quebec (MENVIQ) (EC and MENVIQ 1992). The consensus-based PEC for Pb was calculated as the geometric mean of five values, ranging from a probable effect level of 91.3 mg/kg (Smith et al. 1996) to a severe effect level of 250 mg/kg (Persaud et al. 1993). Moreover, they were derived via collaboration of many partners in the *Freshwater Sediment Quality Assessment Initiative* (MacDonald et al. 2003) which included review of a wide range of options before selecting an approach.

Note that there are some limitations to using the TEC and PEC values for assessing risk to the benthic community. A major limitation is that site-specific differences in the bioavailability of Pb are not taken into account.

7.2 Exposure and Risk Estimates

Chapter 4 presents the methods and data used to develop the media concentrations used as the measures of exposure in the ecological risk assessment. These media concentrations of Pb are summarized and the associated risk HQs are presented in the sections that follow for the case studies: primary Pb smelter (Section 7.2.1), secondary Pb smelter (Section 7.2.2), and near

roadway non-urban (Section 7.2.3). The national-scale surface water and sediment screening assessment Pb concentrations and HQs are presented in Section 7.2.4.

7.2.1 Primary Pb Smelter Case Study

Sampling clusters based on likely exposure patterns were created for this case study location by pooling empirical data collected from various sampling locations within the case study location. Section 4.1.3 summarizes the total Pb soil concentrations, and Section 4.1.5 discusses the surface water concentrations and sediment Pb concentrations for the primary Pb smelter case study. Appendix F presents a detailed description of the aggregates that form each cluster. Soil, surface water, and sediment average concentrations and HQ results are presented in Exhibits 7-4, 7-5, and 7-6, respectively.

Exhibit 7-4. HQs Calculated for Soils for Primary Pb Smelter Case Study[a]

Location of Sample Cluster	Average Pb Concentration (mg/kg)	HQ for Plants	HQ for Soil Invertebrates	HQ for Avian Wildlife	HQ for Mammalian Wildlife
1 - West Bank of Joachim Creek	425	**3.55**	0.25	**11.19**	**3.80**
2 - Crystal City	62.6	0.54	0.04	**1.70**	0.58
3 - Near Fester's Airport	48.5	0.40	0.03	**1.28**	0.43

[a] HQ values greater than 1.0 are highlighted in bold type.

As shown in Exhibit 7-4, all of the soil sampling clusters had HQs that exceeded 1.0 for birds, including those "control" locations outside the assumed impact area of the primary Pb smelter. The west bank of the Joachim Creek samples also had HQs greater than 1.0 for plants and mammals. The west bank of Joachim Creek, which is inside the zone of influence of the primary smelter, had HQ values that were substantially greater than those for the control areas.

As discussed in Section 3.1.6, there is wildlife habitat in the vicinity of Joachim Creek, and birds and mammals often use riparian (stream-side) areas for feeding and for moving between habitats.

Exhibit 7-5. HQs Calculated for Surface Waters for Primary Pb Smelter Case Study

Sample Location and Cluster ID	Dissolved Pb Concentration	HQ using CCC (Chronic)	HQ using CMC (Acute)
Joachim Creek			
Cluster 1	ND[a]	0.39	0.02
Cluster 2	ND	0.40	0.02
Cluster 3	ND	0.39	0.02
Cluster 4	ND	0.41	0.02
Cluster 5	ND	0.41	0.02
Mississippi River			
Upstream	ND	0.54	0.02
Near facility	ND	0.49	0.02
Downstream	ND	0.48	0.02
Emission Deposition			
Cluster 1	ND	0.69	0.03
CHRDDP	ND	0.24	0.01
RRDP-02	ND	0.47	0.02
DAMUP	ND	0.40	0.02

[a] ND = not detected; method detection limit was 3 µg/L; assumed for this assessment that surface water concentration level was at the detection limit.

Exhibit 7-6. HQs Calculated for Sediments in Surface Waters for Primary Pb Smelter Case Study

Location and Cluster Sample ID	Average Measured Pb Concentration in Sediment (mg/kg)	HQ
Joachim Creek		
Cluster 1	36.5	**1.0**
Cluster 2	56.0	**1.6**
Cluster 3	78.7	**2.2**
Cluster 4	30.0	0.84
Cluster 5	34.5	0.96
Mississippi River		
Upstream	14.7	0.41
Near facility	30.1	0.84
Downstream	12.1	0.34
Pond and Drainage Areas		
U-shaped ephemeral pond cluster	170	**4.8**
ED1	110	**3.1**
ED2	14.8	0.41

[a] HQ values greater than 1.0 are highlighted in bold type.

The surface water sampling clusters, which assumed a Pb concentration equal to the detection level, had HQs less than 1.0. However, three sediment sample clusters in Joachim Creek (1, 2, and 3) had HQs ranging from 1.0 to 2.2, and the U-shaped ephemeral pond and one drainage area had sediment HQs greater than 3 but less than 5.

7.2.2 Secondary Pb Smelter Case Study

For the secondary Pb smelter case study, as described in Section 4.2.3, two sets of modeled average Pb soil concentrations were developed for the human exposure and health risk assessments and also used as exposure estimates for the ecological risk assessment. The averages for 1-, 5-, or 10-km interval distances from the secondary Pb smelter facility are summarized in Exhibit 7-7. The associated soil HQs calculated for each interval are also presented in Exhibit 7-7.

Exhibit 7-7. HQs Calculated for Soils for Secondary Pb Smelter Case Study[a]

Distance Range (m)		Modeled Soil Concentration Data Sets (mg/kg)	HQ for Plants	HQ for Soil Invertebrates	HQ for Birds	HQ for Mammals
		Total Pb Soil Conc. with Background				
0	1000	86.6	0.72	0.05	**2.28**	0.77
1000	2000	20.7	0.17	0.012	0.54	0.18
2000	3000	17.3	0.14	0.010	0.46	0.15
3000	4000	16.3	0.14	0.010	0.43	0.15
4000	5000	15.8	0.13	0.009	0.42	0.14
5000	10000	15.4	0.13	0.009	0.41	0.14
10000	20000	15.1	0.13	0.009	0.40	0.13
		Scaled 3x Total Pb Soil Conc. with Background				
0	1000	260	**2.17**	0.15	**6.84**	**2.32**
1000	2000	62.3	0.52	0.037	**1.64**	0.56
2000	3000	51.8	0.43	0.030	**1.36**	0.46
3000	4000	48.9	0.41	0.029	**1.29**	0.44
4000	5000	47.3	0.39	0.028	**1.24**	0.42
5000	10000	46.2	0.39	0.027	**1.22**	0.41
10000	20000	45.4	0.38	0.027	**1.19**	0.41

[a] HQ values greater than 1.0 are highlighted in bold type.

The modeled soil concentrations within 1 km of the facility showed HQs of greater than 1.0 for avian wildlife. All soil concentrations for locations greater that 1 km from the facility were associated with HQs less than 1.0 for this data set. The scaled soil concentration data set resulted in avian HQs greater than 1.0 for all distance intervals evaluated, including the farthest interval modeled, 10 to 20 km from the facility. The scaled soil concentrations within 1 km of the facility also showed HQs greater than 1.0 for plants, birds, and mammals.

As for the previous case study, there is wildlife habitat near the secondary Pb smelter facility. However, as discussed in Section 3.2.6, none of the species residing in the county where this facility is located are currently on state or federal threatened and endangered species lists.

7.2.3 Near Roadway Non-Urban Case Study

Average Pb soil concentrations and risk characterization results for the two non-urban case study locations—Corpus Christi, Texas and Atlee, Virginia—are presented in this section. Exhibit 7-8 presents the average Pb soil concentrations (which are detailed in Section 4.4.1) and HQ calculated for the Corpus Christi, Texas near roadside soil concentration data.

Exhibit 7-8. Soil HQs Calculated for the Non-Urban Near Roadway Case Study Location in Corpus Christi, Texas [a]

Sample Location – Distance from Roadway (m)	Sample Depth (cm)	Total Pb Concentration (mg/kg)	Soil HQs			
			Plants	Invertebrates	Avian Wildlife	Mammalian Wildlife
2	2.5	340	**2.83**	0.20	**8.95**	**3.04**
2	10	650	**5.42**	0.38	**17.1**	**5.80**
2	20	15	0.13	0.019	na[b]	0.13
4	2.5	140	**1.17**	0.082	**3.68**	**1.25**

[a] HQ values greater than 1.0 are highlighted in bold type.

[b] na = not applicable. Woodcock are specialized for feeding on earthworms by probing into soils, and may be able to reach a depth of 6 to 7 cm, which is the approximate length of the bill (Keppie and Whiting 1994).

HQs for birds and mammals were greater than 1.0 at both 2 and 4 meters (m) from the road. Plants also had HQs ranging from 2.83 to 5.42 at the 2-m distance, depending on depth.

Exhibit 7-9 presents the location-specific Pb soil concentration data and HQs calculated for the ecological exposure area in Atlee, Virginia, for the four ecological receptor groups. The soil concentration data for this location are described in Section 4.4.2. The HQs associated with the overall average Pb concentration for the potential ecological exposure area are also presented.

Exhibit 7-9. Soil HQs Associated with Non-Urban Near Roadway Location in Atlee, Virginia [a]

Location ID (Approximate Distance from Nearest Road)	Average Total Pb Concentration (mg/kg)[b]	Soil HQs			
		Plants	Soil Invertebrates	Avian Wildlife	Mammalian Wildlife
6 (2 m)	525	**4.4**	0.31	**14**	**4.7**
5 (2 m)	59	0.49	0.035	**1.6**	0.53
16 (10 m)	140	**1.2**	0.082	**3.7**	**1.3**
22 (10 m)	100	0.83	0.059	**2.6**	0.89
10 (15 m)	99	0.83	0.058	**2.6**	0.88
4 (15 m)	50.5	0.42	0.030	**1.3**	0.45
2 (15 m)	48	0.40	0.028	**1.3**	0.43
1 (30 m)[c]	40.5	0.34	0.024	**1.1**	0.36
3 (30 m)[c]	31.5	0.26	0.019	0.83	0.28
Average	**123**	**1.0**	**0.072**	**3.2**	**1.1**

[a] HQ values greater than 1.0 are highlighted in bold type.
[b] Sampling depths of 0 to 15 inches.
[c] Sampled soils in a surface water runoff detention basin.

HQs for birds were greater than 1.0 at all locations except one (Location ID 3) at 30 meters from the interstate. Plant and mammal HQs were greater than 1.0 at two of the nine locations (Location IDs 6 and 16) within 2 and 10 meters of the road, respectively, although the HQs associated with Location ID 16, 10 m from the interstate, were only slightly above 1.0. HQs for the soil invertebrate receptor group were below 1.0 at all locations.

7.2.4 National-Scale Surface Water and Sediment Screening Assessment

This section presents the Pb exposure measures and risk characterization results of the national-scale surface water and sediment screening assessment. The water column and sediment Pb concentration data used to estimate ecological exposures are described in Section 4.5.2 and 4.5.3, respectively. Section 7.2.4.1 presents the surface water dissolved Pb concentrations and associated risk HQs, and Section 7.2.4.2 presents the total Pb sediment concentrations and associated risk HQs.

7.2.4.1 Surface Water Column Portion of the Screening Assessment

The initial screen of dissolved Pb concentrations for measurements equal to or greater than 1.2 µg/L identified 42 sampling locations for which one or more measurements exceeded that screening value. Data for each measurement of dissolved Pb at these locations are provided in Appendix H. The number of samples analyzed for dissolved Pb at each station ranged from 1 to 60, and the number of measurements that exceeded 1.2 µg/L ranged from 1 to 50. For purposes of this assessment, given the limited analyses for dissolved Pb, all 42 sampling locations were retained for analysis. Eight additional locations were added during a second screen, which identified sampling locations with one or more measured dissolved Pb concentrations above the QL, but less than 1.2 µg/L, and that exceed site specific CCC values (see Appendix H).

Chronic Risks: Exhibit 7-10 presents the chronic HQs for the locations identified in the initial screen (i.e., those locations not specified as "na" in Appendix H). This exhibit is organized in order of increasing water hardness, and therefore, increasing values of the site-specific CCCs. The land use is as reported in the NAWQA database.

For the measures of $CaCO_3$ at each sampling location, the mean (± one standard deviation) concentration and sample size are presented. For stations for which there were no water hardness data (ND), an endnote to the exhibit describes how the CCC was estimated.

The dissolved Pb measurements for each location were characterized in four ways:

1. The number of measured dissolved Pb concentrations that exceeded the CCC out of the total number of samples analyzed for dissolved Pb;
2. The number of samples that were less than the QL where the QL is greater than the CCC for dissolved Pb (i.e., the number of samples that might or might not exceed the CCC);
3. The mean of the measured Pb concentrations that exceeded the CCC; and
4. The maximum measured dissolved Pb concentration.

Two types of HQs are presented in Exhibit 7-10. The first type of HQ represents an average HQ for the measurements that exceeded the CCC. This HQ answers the question "when the CCC is exceeded, on average, what is the magnitude of the HQ?" The second HQ represents a maximum HQ based on the maximum dissolved Pb concentration. This HQ assumes that the maximum value represents at least a four-day average concentration. For many sampling sites, both HQs are the same because only a single measurement of dissolved Pb exceeded the CCC. Exhibit 7-10 illustrates that many of the larger chronic HQ values are associated with mining sites.

Exhibit 7-10. Results of Chronic Aquatic Risk Screening Assessment –HQs for Measured Dissolved Pb Concentrations Compared to Chronic AWQC (i.e., Criterion Continuous Concentration or CCC) for the Protection of Aquatic Life for Pb[a]

Basin ID		State	Station ID	Land Use	Calcium Carbonate		Pb CCC (µg/L)	Dissolved Pb Measurements				Hazard Quotients	
					Mean ± SD (mg/L)	N		No. > CCC / Total N	No. < QL, which is > CCC	Mean of [Pb]s > CCC (µg/L)	Max [Pb] (µg/L)	Mean [Pb] / CCC	Max [Pb] / CCC
44	UCOL	CO	393557105512400	Mining	1 [(b)]	1	0.013	1/1	0	1.26	1.26	**96.92**	**96.92**
59	COOK	AK	600715152572800	Reference	6	1	0.11	1/1	0	1.64	1.64	**14.91**	**14.91**
46	NROK	ID	12392155	Forest	8.8 ± 2.8	18	0.17	4/17	10	0.224	0.261	**1.32**	**1.54**
58	OAHU	HA	16212700	Mixed	9	2	0.17	1/2	1	0.337	0.337	**1.98**	**1.98**
2	CONN	CT	1119375	Mixed	9.5 ± 2.9	19	0.18	5/20	13	0.302	0.377	**1.68**	**2.09**
2	CONN	CT	1124000	Mixed	14.7 ± 4.9	30	0.30	11/23	9	0.759	1	**2.53**	**3.33**
44	UCOL	CO	375732107394000	Mining	ND [(c)]	0	0.35	2/2	0	21.6	23	**61.71**	**65.71**
44	UCOL	CO	385437107015600	Mining	17	1	0.35	1/1	0	6.02	6.02	**17.20**	**17.20**
2	CONN	CT	1127000	Mixed	17.4 ± 8.7	29	0.36	3/22	14	0.408	0.472	**1.13**	**1.31**
46	NROK	ID	12419000	Mixed	17.9 ± 1.5	29	0.37	2/26	16	0.995	1.579	**2.69**	**4.27**
1	NECB	RI	1112900	Mixed	21 ± 10	13	0.44	3/3	0	1.103	1.551	**2.51**	**3.53**
46	NROK	ID	12413470	Mining	25 ± 7.2	34	0.54	33/33	0	4.62	11.62	**8.56**	**21.52**
44	UCOL	CO	9046530	Mining	31.6 ± 3.3	62	0.70	53/60	7	3.39	8	**4.84**	**11.43**
44	UCOL	CO	392907106013900	Mining	35.3 ± 4.0	3	0.80	2/3	1	1.50	2	**1.88**	**2.50**
46	NROK	ID	12413123	Mining	36.3	1	0.82	1/1	0	29.78	29.78	**36.32**	**36.32**
44	UCOL	CO	385240106583600	Other/Mixed	39	2	0.89	1/4	2	0.971	0.971	**1.09**	**1.09**
44	UCOL	CO	392947106024500	Mining	41.3 ± 4.0	3	0.95	2/3	1	1.50	2	**1.58**	**2.11**
46	NROK	WA	12422500	Urban	43 ± 27	20	0.99	4/28	24	1.131	1.233	**1.14**	**1.25**
46	NROK	WA	12422000	Urban	43.1 ± 25.7	18	0.99	2/20	18	1.158	1.158	**1.17**	**1.17**
46	NROK	ID	12413150	Mining	50.2	1	1.2	1/1	0	10.518	10.518	**8.77**	**8.77**
44	UCOL	CO	9146200	Mining	84.7 ± 28	52	2.1	0/3	0	2	2	0.95	0.95
31	OZRK	AR	7050500	Mixed	102 ± 39	13	2.6	1/8	0	9	9	**3.46**	**3.46**
45	RIOG	NM	8331000	Mixed	113 ± 18	37	2.9	1/12	0	3	3	**1.03**	**1.03**
50-51	CCYK	WA	461850120005800	Other/Mixed	ND [(d)]	0	3.3	0/1	0	1.2	1.2	0.36	0.36
31	OZRK	MO	7061155	Mining	136	1	3.5	0/1	0	3	3	0.86	0.86
31	OZRK	MO	7018100	Forest	142	1	3.7	1/2	0	7	7	**1.89**	**1.89**
31	OZRK	MO	7061135	Mining	160	1	4.2	1/1	0	11	11	**2.62**	**2.62**
55	NVBR	NV	94196783	Urban	175 ± 57	106	4.6	0/5	0	2.2	3	0.48	0.65

Exhibit 7-10. Results of Chronic Aquatic Risk Screening Assessment –HQs for Measured Dissolved Pb Concentrations Compared to Chronic AWQC (i.e., Criterion Continuous Concentration or CCC) for the Protection of Aquatic Life for Pb[a]

Basin ID		State	Station ID	Land Use	Calcium Carbonate		Pb CCC (µg/L)	Dissolved Pb Measurements				Hazard Quotients	
					Mean ± SD (mg/L)	N		No. > CCC / Total N	No. < QL, which is > CCC	Mean of [Pb]s > CCC (µg/L)	Max [Pb] (µg/L)	Mean [Pb] / CCC	Max [Pb] / CCC
40	SPLT	CO	6752260	Mixed	ND [e]	0	4.9	0/4	0	4	4	0.82	0.82
40	SPLT	CO	6752270	Mixed	ND [e]	0	4.9	0/4	0	2	2	0.41	0.41
40	SPLT	CO	6752280	Mixed	ND [e]	0	4.9	0/4	0	3	3	0.61	0.61
49	CAZB	AZ	9517000	Agriculture	190 ± 36	48	5.0	0/1	0	2	2	0.40	0.40
47	GRSL	UT	404026111273001	Rangeland	195	1	5.2	0/5	0	2.34	2.828	0.45	0.54
47	GRSL	UT	403938111300201	Urban	ND [f]	0	5.2	0/2	0	1.432	1.432	0.28	0.28
45	RIOG	NM	8317200	Urban	196 ± 68	14	5.2	0/3	0	2.33	3	0.45	0.58
47	GRSL	UT	404431111282901	Urban	ND [g]	0	5.3	0/2	0	3.754	4.271	0.71	0.81
47	GRSL	UT	404847111240501	Rangeland	ND [h]	0	5.8	0/2	0	5.495	8.403	0.95	**1.45**

Abbreviations: SD = standard deviation; N = Total number of samples analyzed for specified analyte; [Pb] = dissolved Pb concentration; QL = quantitation limit (where QL = 0.08 µg/L dissolved Pb, some measured concentrations that were greater than the detection limit, but less than the QL were estimated and are considered ≥ QL for purposes of this exhibit); No. = Number of samples analyzed for dissolved Pb; Basin ID four-letter codes are described in Appendix H; State codes = state postal abbreviations; ND = no data

[a] Records are presented in order of increasing mean water hardness (mg/L as $CaCO_3$) and hardness-specific CCC. HQ values greater than 1.0 are highlighted in bold type.

[b] This measurement of water hardness seems unreasonably low.

[c] No water hardness data for this sampling station or nearby stations; used CCC determined for nearest station (i.e., 385437107015600).

[d] No water hardness data for this sampling station; used CCC based on a value of 128 mg/L as $CaCO_3$ from nearest station (i.e., 461517119402500).

[e] No water hardness data for this sampling station; used CCC based on a value of 187 mg/L as $CaCO_3$ from nearest station (i.e., 6754000).

[f] No data for this station; used CCC based on water hardness value for the other single value for Silver Creek (station No. 404026111273001).

[g] No data for this station; one nearby station had values of 234 and 223 mg/L as $CaCO_3$; another nearby station had values of 195 and 176 mg/L as $CaCO_3$. This analysis therefore assumed a water hardness of 200 mg/L to estimate the CCC for this location.

[h] No water hardness data for this station; used CCC based on a value of 219 mg/L as $CaCO_3$ from nearest station (i.e., 404751111423501).

Acute Risks: Exhibit 7-11 presents the acute aquatic risk HQs for the locations identified in the initial screen (i.e., those locations not specified as "na" in Appendix H). This exhibit is organized in order of increasing water hardness, and therefore, increasing values of the site-specific CMCs. In this exhibit, dissolved Pb is characterized in two ways:

1. The number of measured dissolved Pb concentrations that exceed the CMC out of the total number of samples analyzed for dissolved Pb; and
2. The maximum measured dissolved Pb concentration.

Given that the CMCs are between 20 and 25 times higher than the CCCs, the only HQ considered here is the maximum dissolved Pb concentration compared with the CMC. Exhibit 7-11 shows that only three sampling sites had acute HQ values that exceeded 1.0 (in bold), and all three of these are located near mining sites.

Exhibit 7-11. Results of Acute Aquatic Risk Screening Assessment –HQs for Measured Dissolved Pb Concentrations Compared to Acute AWQC (i.e., Criterion Maximum Concentration or CMC) for the Protection of Aquatic Life for Pb[a]

Basin ID		State	Station ID	Land Use	Pb CMC (µg/L)	Dissolved Pb Measurements		HQ: Max [Pb] / CMC
						No. > CMC / Total N	Max [Pb] (µg/L)	
44	UCOL	CO	393557105512400	Mining	0.34	1/1	1.26	**3.71**
59	COOK	AK	600715152572800	Reference	2.7	0/1	1.64	0.61
46	NROK	ID	12392155	Forest	4.2	0/17	0.261	0.06
58	OAHU	HA	16212700	Mixed	4.4	0/2	0.337	0.08
2	CONN	CT	1119375	Mixed	4.6	0/20	0.377	0.08
2	CONN	CT	1124000	Mixed	7.6	0/23	1	0.13
44	UCOL	CO	375732107394000	Mining	9.0	0/1	6.02	0.67
44	UCOL	CO	385437107015600	Mining	9.0	2/2	23	**2.56**
2	CONN	CT	1127000	Mixed	9.2	0/22	0.472	0.05
46	NROK	ID	12419000	Mixed	9.5	0/26	1.579	0.17
1	NECB	RI	1112900	Mixed	11	0/3	1.551	0.14
46	NROK	ID	12413470	Mining	14	0/33	11.62	0.83
44	UCOL	CO	9046530	Mining	18	0/60	8	0.44
44	UCOL	CO	392907106013900	Mining	20	2/3	2	0.10
46	NROK	ID	12413123	Mining	21	1/1	29.78	**1.42**
44	UCOL	CO	385240106583600	Other/Mixed	23	0/4	0.971	0.04
44	UCOL	CO	392947106024500	Mining	24	0/3	2	0.08
46	NROK	WA	12422500	Urban	25	0/28	1.233	0.05
46	NROK	WA	12422000	Urban	25	0/20	1.158	0.05
46	NROK	ID	12413150	Mining	30	0/1	10.518	0.35
44	UCOL	CO	9146200	Mining	54	0/3	2	0.04
31	OZRK	AR	7050500	Mixed	66	0/8	9	0.14
45	RIOG	NM	8331000	Mixed	74	0/12	3	0.04
50-51	CCYK	WA	461850120005800	Other/Mixed	84	0/1	1.2	0.01
31	OZRK	MO	7061155	Mining	90	0/1	3	0.03
31	OZRK	MO	7018100	Forest	94	0/2	7	0.07
31	OZRK	MO	7061135	Mining	110	0/1	11	0.10

Exhibit 7-11. Results of Acute Aquatic Risk Screening Assessment –HQs for Measured Dissolved Pb Concentrations Compared to Acute AWQC (i.e., Criterion Maximum Concentration or CMC) for the Protection of Aquatic Life for Pb[a]

Basin ID		State	Station ID	Land Use	Pb CMC (µg/L)	Dissolved Pb Measurements		HQ: Max [Pb] / CMC
						No. > CMC / Total N	Max [Pb] (µg/L)	
55	NVBR	NV	94196783	Urban	120	0/5	3	0.03
47	GRSL	UT	404026111273001	Rangeland	130	0/5	2.828	0.02
49	CAZB	AZ	9517000	Agriculture	130	0/1	2	0.02
45	RIOG	NM	8317200	Urban	130	0/3	3	0.02
47	GRSL	UT	403938111300201	Urban	130	0/2	1.432	0.01
40	SPLT	CO	6752270	Mixed	130	0/4	2	0.02
40	SPLT	CO	6752280	Mixed	130	0/4	3	0.02
40	SPLT	CO	6752260	Mixed	130	0/4	4	0.03
47	GRSL	UT	404431111282901	Urban	140	0/2	4.271	0.03
47	GRSL	UT	404847111240501	Rangeland	150	0/2	8.403	0.06

Abbreviations: Same as Exhibit 7-10.

[a] Records are presented in order of increasing hardness-specific CMC. HQ values greater than 1.0 are highlighted in bold type.

Aquatic Risks at Mining Sites Compared with other Sites: The locations of interest for the Pb NAAQS review are those locations for which deposition from air might be a significant input of Pb into the surface water. It may be assumed that air inputs are not easily distinguishable from other non-air inputs near mining sites. Therefore, the results for the mining sites (Exhibit 7-12) were separated from the other types of locations (Exhibit 7-13). Locations for which the HQ was less than 1.0 were eliminated to summarize the results for the aquatic chronic risk screen.

Exhibit 7-12 summarizes the two chronic and one acute risk HQs for the 10 mining sites for which the chronic HQ exceeded 1.0. The exhibit is presented in order of increasing HQ.

Exhibit 7-12. Results of Aquatic Risk Screening Assessment – Mining Sites at which Dissolved Pb Measurements Exceed AWQC [a]

Basin ID		State	Station ID	Land Use	Pb CCC (µg/L)	Pb Measurements		Hazard Quotient		
						No. > CCC / Total N	No. < QL, which is > CCC	Mean [Pb] / CCC	Max [Pb] / CCC	Max [Pb] / CMC
44	UCOL	CO	392947106024500	Mining	0.95	2/3	1	**1.58**	**2.11**	0.08
44	UCOL	CO	392907106013900	Mining	0.80	2/3	1	**1.88**	**2.50**	0.10
31	OZRK	MO	7061135	Mining	4.2	1/1	0	**2.62**	**2.62**	0.10
44	UCOL	CO	9046530	Mining	0.70	53/60	7	**4.84**	**11.43**	0.44
46	NROK	ID	12413470	Mining	0.54	33/33	0	**8.56**	**21.52**	0.83
46	NROK	ID	12413150	Mining	1.2	1/1	0	**8.77**	**8.77**	0.35
44	UCOL	CO	385437107015600	Mining	0.35	1/1	0	**17.20**	**17.20**	0.67
46	NROK	ID	12413123	Mining	0.82	1/1	0	**36.32**	**36.32**	**1.42**
44	UCOL	CO	375732107394000	Mining	0.35	2/2	0	**61.71**	**65.71**	**2.56**
44	UCOL	CO	393557105512400	Mining	0.013	1/1	0	**96.92**	**96.92**	**3.71**

Abbreviations: Same as for Exhibits 7-10; CCC = criterion continuous concentration (or chronic AWQC); CMC = criterion maximum concentration (or acute AWQC).

[a] In order of increasing Hazard Quotient for the CCC aquatic toxicity benchmark. HQ values greater than 1.0 are highlighted in bold type.

Many of the sampling sites near mining land uses are characterized by soft water (i.e., $CaCO_3$ less than 50 mg/L and CCC less than 1.2 µg/L). Note that some of the largest HQs are associated with sites that had one or two samples for dissolved Pb and no or few samples for $CaCO_3$. Thus, the uncertainty associated with the chronic HQ values for these sites is higher than for the two locations that have been well characterized (i.e., with 33 and 60 samples analyzed for dissolved Pb). For the mining sites, it was assumed that drainage from the mines provides the majority of the Pb input to the surface waters, and these sites were not examined further.

Exhibit 7-13 summarizes the HQs for the 15 non-mining sites for which the chronic HQs exceed 1.0 in order of increasing HQ. These locations are in areas classified in the NAWQA database as urban and mixed, but also include forest, rangeland, and a "reference" site in Alaska. The highest HQ is for the Alaska reference site and is based on one measurement of dissolved Pb and one measurement of $CaCO_3$, which was extremely low (i.e., 6 mg/L). Thus, the uncertainty associated with this HQ is high.

Exhibit 7-13. Results of Aquatic Risk Screening Assessment - Locations at which Dissolved Pb Measurements Exceed AWQC, Excluding Mining Sites [a]

Basin ID		State	Station ID	Land Use	Pb CCC (µg/L)	Pb Measurements		Hazard Quotient		
						No. > CCC / Total N	No. < QL, which is > CCC	Mean [Pb] / CCC	Max [Pb] / CCC	Max [Pb] / CMC
45	RIOG	NM	8331000	Mixed	2.9	1/12	0	**1.03**	**1.03**	0.04
44	UCOL	CO	385240106583600	Other/Mixed	0.89	1/4	2	**1.09**	**1.09**	0.04
2	CONN	CT	1127000	Mixed	0.36	3/22	14	**1.13**	**1.31**	0.05
46	NROK	WA	12422500	Urban	0.99	4/28	24	**1.14**	**1.25**	0.05
46	NROK	WA	12422000	Urban	0.99	2/20	18	**1.17**	**1.17**	0.05
46	NROK	ID	12392155	Forest	0.17	4/17	10	**1.32**	**1.54**	0.06
47	GRSL	UT	404847111240501	Rangeland	5.8	1/2	0	**1.45**	**1.45**	0.06
2	CONN	CT	1119375	Mixed	0.18	5/20	13	**1.68**	**2.09**	0.08
31	OZRK	MO	7018100	Forest	3.7	1/2	0	**1.89**	**1.89**	0.07
58	OAHU	HA	16212700	Mixed	0.17	1/2	1	**1.98**	**1.98**	0.08
1	NECB	RI	1112900	Mixed	0.44	3/3	0	**2.51**	**3.53**	0.14
2	CONN	CT	1124000	Mixed	0.30	11/23	9	**2.53**	**3.33**	0.13
46	NROK	ID	12419000	Mixed	0.37	2/26	16	**2.69**	**4.27**	0.17
31	OZRK	AR	7050500	Mixed	2.6	1/8	0	**3.46**	**3.46**	0.14
31	COOK	AK	600715152572800	Reference	0.11	1/1	0	**14.91**	**14.91**	0.61

Abbreviations: Same as Exhibit 7-11; CCC = criterion continuous concentration (or chronic AWQC); CMC = criterion maximum concentration (or acute AWQC).

[a] In order of increasing Hazard Quotient for the CCC aquatic toxicity benchmark. HQ values greater than 1.0 are highlighted in bold type. Additional information characterizing these locations is provided in Appendix H.

Contribution of Air to Pb Concentration at Non-Mining Site Locations: For the sites in Exhibit 7-13, three additional sources of information were examined in an attempt to determine whether air deposition is likely to be a significant source of Pb for the surface water. These sources are as follows:

1. **Land Use.** More extensive data on the land uses surrounding the sampling sites were obtained from the 1992 National Land Cover Database for land uses within 20 km of each sampling site, if available.
2. **Air Pb Emissions.** The number of facilities that emit Pb to air, and the total annual quantity of Pb emitted to air, within 20 km and 50 km of the sampling site were determined from the 2002 National Emissions Inventory (USEPA 2006i). Facility emissions to the air within 20 km are likely to be much more important than the emissions between 20 and 50 km away. Data on non-point and mobile emissions of Pb at the county level were obtained from the same source.
3. **NPDES.** For each sampling site, the watershed was examined to identify facilities in the National Pollutant Discharge Elimination System (NPDES) database upstream of the sampling site for which Pb was a constituent identified for the facility. For each facility identified, available data were examined on permitted limits for Pb releases and on measured Pb releases.

The detailed results of these analyses are presented in Appendix H, and the result highlights are presented in Exhibit 7-14.

Exhibit 7-14. Results of National-Scale Surface Water Risk Screening Assessment - Locations at which Dissolved Pb Measurements Exceed AWQC, Excluding Mining Sites [a]

Basin ID (and State)	Station ID	Pb CCC (µg/L)	Pb Emissions (tons/year)			% Strip Mines, etc.	No. NPDES upstream (metals)	No. [Pb] > CCC / N	HQ Max [Pb] / CCC
			Facil-ities < 20 km	Facil-ities < 50 km	Mobile & Non-point in County				
45 RIOG NM	8331000	2.9	0.068	0.095	0.029	0.19	3 (0)	1/12	**1.03**
44 UCOL CO	385240106583600	0.89	0.0	0.00029	0.036	0.0	ND	1/4	**1.09**
46 NROK WA	12422000	0.99	0.41	0.43	0.24	0.0	3 (0)	2/20	**1.17**
46 NROK WA	12422500	0.99	0.39	0.43	0.24	0.0	3 (0)	4/28	**1.25**
2 CONN CT	1127000	0.36	**6.1**	7.0	0.14	0.11	15 (0)	4/22	**1.31**
47 GRSL UT	404847111240501	5.8	0.0	0.36	0.0011	0.06	1 (1)	1/2	**1.45**
46 NROK ID	12392155	0.17	0.0	0.0014	0.22	0.0	1 (1)	4/17	**1.54**
31 OZRK MO	7018100	3.7	0.0	0.34	0.018	0.05	ND	1/2	**1.89**
58 OAHU HA	16212700	0.17	**4.9**	4.9	0.49	0.0	ND	1/2	**1.98**
2 CONN CT	1119375	0.18	0.68	13.7	0.014	0.07	5 (0)	5/20	**2.09**
2 CONN CT	1124000	0.3	0.081	11.3	0.92	0.09	1 (0)	11/23	**3.33**
31 OZRK AR	7050500	2.6	0.0	0.0051	0.0064	0.02	1 (0)	1/8	**3.46**
1 NECB RI	1112900	0.44	**4.1**	11.7	0.15	0.44	14 (6)	3/3	**3.53**
46 NROK ID	12419000	0.37	0.34	0.43	0.6	0.0	11 (4)	2/26	**4.27**
31 COOK AK	60071515257280	0.11	0.0	0.0	0.1	0.0	ND	1/1	**14.91**

Abbreviations: Same as Exhibit 7-10; CCC = Criterion Continuous Concentration (or chronic AWQC); CMC = Criterion Maximum Concentration (or acute AWQC).

[a] In order of increasing Hazard Quotient (HQ) for the CCC aquatic toxicity benchmark. HQ values greater than 1.0 are highlighted in bold type. Additional information characterizing these locations is provided in Appendix H.

Of the 15 sampling locations in Exhibit 7-14, only three appear to be within 20 km of facilities emitting relatively large quantities of Pb to the atmosphere (i.e., more than 1 ton per year): one is in Oahu, Hawaii, one in Jewett City, Connecticut, and one in Manville, Rhode Island (see bold entries in column "Facilities < 20 km"). Two additional sampling locations appear to be within 50 km of facilities emitting relatively large quantities of Pb, both in Connecticut; however, whether these facilities are close enough to influence the Pb concentrations in the water column at these sampling sites is unknown.

Of the three sampling locations within 20 km of facilities emitting more than 1 ton of Pb per year, the location in Rhode Island might also be receiving a large fraction of its Pb from upstream discharges from metal ore processing facilities (i.e., there are six such discharges out of fourteen NPDES permitted facilities upstream of this sampling location).

7.2.4.2 Sediment Portion of the Screening Assessment

The sediment sampling locations used to represent each of the 15 non-mining sites are identified in Exhibit 7-15. Co-located sediment and surface water data were available for 9 of the 15 sites. In addition, there were three "near matches," with a sediment sampling station on the same creek or river, but somewhat upstream or downstream of the surface water sampling location.

Exhibit 7-15. Results of Sediment Risk Screening Assessment – Stations With Exact or Near Match to Surface Water Stations at which Dissolved Pb Measurements Exceed AWQC, Excluding Mining Sites [a]

Basin ID		State	Station ID	Land Use	Match	Name of Location
45	RIOG	NM	SW + Sed 8331000	Mixed	Yes	Rio Grande at Isleta
44	UCOL	CO	SW 3852401-06583600	Other/Mixed	No – nearest location is in different water body	Slate River above Coal Creek near Crested Butte
			Sed 3854371-07015600	Mining		Oh-be-joyful Creek at mouth near rested Butte
2	CONN	CT	SW 1127000	Mixed	Near match with location on same river	Quinebaug River at Jewett City
			Sed 1126850	Mixed		Quinebaug River (Aspinock Pond) at Clayville
46	NROK	WA	SW 12422500	Urban	No – nearest location is in different water body	Spokane River at Spokane
			Sed 12424000	Cropland		Hangman Creek at Spokane, Washington
46	NROK	WA	SW 12422000	Urban	Near match with location on same river	Spokane River below Green Street at Spokane
			Sed 12424500	Urban		Spokane River at 7 Mile Bridge Near Spokane, Washington
46	NROK	ID	SW + Sed 12392155	Forest	Yes	Lightning Creek at Clark Fork, Idaho
47	GRSL	UT	SW + Sed 4048471-11240501	Rangeland	Yes	Silver Creek at Wanship, Utah
2	CONN	CT	SW 1119375	Mixed	No other station in county	Willimantic River at Merrow, Connecticut
			No Sed Match	na		na
31	OZRK	MO	SW + Sed 7018100	Forest	Yes	Big River Near Richwoods, Missouri
58	OAHU	HA	SW + Sed 16212700	Mixed	Yes	Waikakalaua Street Near Wahiawa, Oahu, Hawaii
1	NECB	RI	SW + Sed 1112900	Mixed	Yes	Blackstone River at Manville, Rhode Island
2	CONN	CT	SW 1124000	Mixed	Near match – see also SW 1127000	Quinebaug River at Quinebaug, Connecticut
			Sed 1126850	Mixed		Quinebaug River (Aspinock Pond) at Clayville
46	NROK	ID	SW + Sed 12419000	Mixed	Yes	Spokane River Near Post Falls, Idaho
31	OZRK	AR	SW 7050500	Mixed	Yes	Kings River near Berryville
31	COOK	AK	6007151-52572800	Reference	Yes	EF ORE C Near MTH near Johnson Glacier Near Tuxedni Bay, Alaska

Abbreviations: SW = surface water sampling station; Sed = sediment sampling station.

[a] Locations listed in order of increasing Hazard Quotient for the CCC aquatic toxicity benchmark.

Measured Total Pb Sediment Concentrations: Exhibit 7-16 summarizes the total Pb sediment concentrations measured at 12 of the locations in Exhibit 7-15: nine locations with co-located sediment and surface water data and the three "near-match" locations in Exhibit 7-15. The

NAWQA database appears to have only one measurement of Pb sediment concentrations for each of these 12 locations. Several of those samples were taken in the early 1990s, earlier than most of the surface water Pb concentrations that were summarized in the water column risk screen. Nonetheless, assuming that sediment concentrations change slowly over time, which may be one reason for the single samples in the NAWQA database, these data are of some value. For comparison purposes, Exhibit 7-16 also provides the surface water HQ for Pb estimated by dividing the maximum dissolved Pb concentration by the CCC. For the 12 sites in Exhibit 7-16, the total Pb concentrations in sediments seemed to be trimodal. Seven total Pb concentrations were between 23 and 68 mg/kg; two were at 239 and 240 mg/kg, and three were between 1620 and 2900 mg/kg.

Chronic Risks to Benthic Communities and Contribution of Air Pb Emissions to those Risks: This section presents the HQs associated with measured total Pb concentrations in sediments at the 12 sites presented in Exhibit 7-16. The HQs were calculated by dividing potential Pb exposure concentrations by the TEC and PEC benchmarks for benthic communities (MacDonald et al. 2000; 2003). The possible contributions of air Pb emissions to risks at these locations are also discussed.

As shown in Exhibit 7-16, 9 of 12 sites had TEC-based HQs that exceeded 1.0; TEC-based HQs at the remaining three sites were less than 1.0. The three sites with HQs less than 1.0 are unlikely to pose risks to benthic aquatic communities based on the available data. None of these three sites were those likely to be affected by air emissions of Pb from point sources (i.e., Pb emissions were less than 0.07 tons per year at all three sites).

PEC-based HQs at five sites exceeded 1.0, indicating probable adverse effects to ecological receptors at these sites. Three of these five exceeded a PEC-based HQ of 10, indicating a very high probability of adverse effects to ecological receptors at these sites, and possibly higher severity of effects than at the locations with lower HQ values. It is believed that none of these three sites were likely to be affected, however, by air emissions. One in Idaho was downstream from several NPDES-permitted discharges of metals to surface waters (station ID 12419000). The other two sites are located in Utah and Montana, suggesting possible historic sediment contamination from mining operations.

Exhibit 7-16. Concentrations of Total Pb in Sediments at Locations Near or Co-Located with the 15 Sites at which Dissolved Pb Concentrations Exceeded the AWQC, Excluding Mining Sites [a]

Basin ID		State	Station ID	Land Use	Match	Date	Total [Pb] (mg/kg dry sediment)	SW HQ [b]: max [Pb]/CCC	Pb Emissions (tons/year)		No. Upstream NPDES permits for metals[b]	Sediment Hazard Quotients	
									Fac < 20 km	Fac < 50 km		[Pb]/TEC	[Pb]/PEC
45	RIOG	NM	SW + Sed 8331000	Mixed	Yes	02/25/93	23	**1.03**	0.068	0.095	0	0.64	0.18
2	CONN	CT	Sed 1126850	Mixed	Near	07/20/93	68	**1.13**	6.1	7.0	0	**1.9**	0.53
46	NROK	WA	Sed 12424500	Urban	Near	08/05/98	47.3	**1.14**	0.39	0.43	0	**1.3**	0.37
46	NROK	ID	SW + Sed 12392155	Forest	Yes	07/29/98	24.9	**1.32**	0.0	0.0	1	0.70	0.19
47	GRSL	UT	SW + Sed 4048471-11240501	Rangeland	Yes	07/22/99	2900	**1.45**	0.0	0.36	1	**81**	**23**
31	OZRK	MO	SW + Sed 7018100	Forest	Yes	08/25/92	2300	**1.89**	0.0	0.34	ND	**64**	**18**
58	OAHU	HA	SW + Sed 16212700	Mixed	Yes	07/12/00	59	**1.98**	4.9	4.9	ND	**1.6**	0.46
1	NECB	RI	SW + Sed 1112900	Mixed	Yes	06/30/99	240	**2.51**	4.1	11.7	6	**6.7**	**1.9**
2	CONN	CT	Sed 1126850	Mixed	Near	07/20/93	68	**2.53**	0.081	11.3	0	**1.9**	0.53
46	NROK	ID	SW + Sed 12419000	Mixed	Yes	08/03/98	1620	**2.69**	0.34	0.43	4	**45**	**13**
31	OZRK	AR	SW + Sed 7050500	Mixed	Yes	11/02/93	28	**3.46**	0.0	0.01	0	0.78	0.22
31	COOK	AK	SW + Sed 6007151-52572800	Reference	Yes	07/25/00	239	**14.91**	0.0	0.0	0	**6.7**	**1.9**

[a] Exhibited in increasing order of the surface water (SW) column risk hazard quotient (HQ). HQs exceeding 1.0 are highlighted in bold type.

[b] Data from Exhibit 7-14 for corresponding locations.

Abbreviations:[Pb] = total Pb concentration in sediments (mg/kg dry sediment). CCC = Criterion Continuous Concentration (or chronic AWQC). For Study Basin IDs, see Appendix H. TEC = threshold-effect concentration, and PEC = probable-effect concentration, both from the consensus-based sediment quality criteria approach published by MacDonald et al. (2000; 2003).

Of the three locations in Exhibit 7-16 for which air emissions of Pb from point sources might be contributing to ongoing Pb contamination of surface water and sediments (i.e., the 2nd, 7th, and 8th rows for locations in Connecticut, Hawaii, and Rhode Island, respectively), only one, the Blackstone River in Manville, Rhode Island, is likely also to receive significant Pb inputs from upstream NPDES-permitted sites. That site also might have received historic inputs of Pb from the use of leaded gasoline and subsequent erosion of soils to the river. The other two, the Quinebaug River in Connecticut (a near match between the Jewett City and Clayville locations) and the water body at Waikakalaua Street near Wahiawa, Oahu, Hawaii, had no other obvious inputs of Pb in our assessment other than the point sources of air emissions within 20 km. Both of those sites, however, are in "mixed" urbanized areas, and therefore could quite possibly also have historic Pb deposition from leaded gasoline and ongoing inputs of Pb to sediments from erosion of soils contaminated by leaded gasoline.

7.3 Limitations and Uncertainties

This section summarizes primary limitations and uncertainties for each of the case studies and the national-scale screening assessment. Section 7.3.1 addresses uncertainties and limitations associated with the primary Pb smelter case study; Section 7.3.2 describes them for the secondary Pb smelter facility case study; Section 7.3.3 discusses them for the near roadway non-urban case studies; and Section 7.3.4 addresses limitations and uncertainties associated with the national-scale screening assessment for risks to aquatic organisms from Pb deposition from air to surface waters.

Uncertainties that apply across the case studies and national-scale screening assessment include but are not limited to the following:

- The case study analyses and national-scale surface water and sediment screening assessment are limited to specific case study locations and other locations nationwide for which dissolved Pb data in surface water were available. Efforts were made to ensure that the exposure estimates were attributable to background Pb concentrations and air emissions of Pb; however, it is uncertain whether other sources might have actually contributed to the Pb exposure estimates.

- The selected ecotoxicity screening values for surface waters and sediments might not be sufficient to identify risks to some threatened or endangered species or unusually sensitive aquatic ecosystems.

- The database supporting the current AWQC for Pb is over 20 years old. There are data to indicate that Pb bioconcentrates to some extent in invertebrates (e.g., bioconcentration factors, or BCFs, of 500 to 1,700), and, to a lesser extent, in fish (e.g., BCFs of 42 to 45 in two species) in freshwater ecosystems. However, in 1984, data were insufficient to estimate Final Tissue Residue Levels associated with adverse effects in fish, and thus the BCFs did not influence the CCC value (USEPA 1984). Also, EPA is evaluating whether pH may also be an indicator of bioavailability in addition to water hardness.

- No adjustments were made for sediment-specific characteristics that might affect the bioavailability of Pb in sediments in the derivation of the sediment quality criteria used for this ecological risk assessment. Similarly, characteristics of soils for the case study locations were not evaluated for measures of bioavailability.

- Although the SSV calculated for birds is based on realistic parameter values for average diet composition, 90th percentile soil and diet ingestion rates, and a high-end Pb assimilation efficiency, a recent site-specific determination of a soil concentration protective of soil-invertebrate-consuming birds suggested that the value of 38 mg/kg dry soil is still overly conservative. Use of 90th percentile values for two parameters and the maximum Pb assimilation efficiency suggested by available data may be overly conservative.

7.3.1 Primary Pb Smelter Case Study

The ELM *Sampling and Analysis Plan* was designed to investigate possible ecological risks from all sources of Pb (and other contaminants) attributable to the primary Pb smelter without a need to attribute the source of Pb in ecologically sensitive areas (ELM 2003; ELM 2005). For purposes of the Pb NAAQS review, it is important to distinguish areas impacted primarily by air deposition of Pb from areas impacted primarily by Pb from other sources (e.g., soil runoff, groundwater discharge to surface water).

The soil sampling locations within a 2.1-km radius were all in areas that might have been subject to Pb inputs from Joachim Creek during flooding events. As such, the stations might not represent the concentrations of Pb in soils that result from direct air emissions from the smelter. This limitation may overstate the risks from deposition of Pb emitted from the facility.

7.3.2 Secondary Pb Smelter Case Study

The ecological risk assessment used modeled rather than soil concentration measurements to estimate exposure because measured data were not available. Fate and transport modeling limitations and uncertainties are described in Section 4.2.3.1. As discussed in this section, the modeled soil concentrations were compared to measured soil concentrations from areas surrounding another secondary Pb smelter and were found to be approximately three times lower than the measured concentrations. Therefore, the modeled results were scaled up three times to address the potential under-prediction of the soil model when the results were compared to the surrogate measurements.

7.3.3 Near Roadway Non-Urban Case Study

Few measurements were available to evaluate ecological risk associated with contaminated soils near roadways in less developed areas where presence of ecological receptors may be anticipated. It is also uncertain how representative of other roadways these data are.

It is unlikely that there is significant avian and mammalian use of habitats within 2 meters of heavily traveled roads; therefore, HQs above 1.0 at locations at 2 meters might not be associated with ecological impacts beyond those associated with traffic on the road.

For the Atlee, Virginia location, soil concentrations were measured at sampling locations between 2 to 30 meters of intensely traveled roads. This assessment did not evaluate avian and mammalian wildlife habitat use within 30 meters of roadways. Without this evaluation, the Atlee, Virginia assessment is likely to overestimate the ecological risks of Pb in roadway soils.

The assessment did not address surface water ecosystem impacts of Pb from near roadway runoff of Pb contaminated soils. This may underestimate risks to aquatic receptors via this exposure pathway.

7.3.4 National-Scale Surface Water and Sediment Screening Assessment

7.3.4.1 Surface Water Column Screen

The analysis revealed only two or three NAWQA sampling locations nationwide where there appear to be potential risks to the aquatic community from Pb that might have originated from atmospheric deposition. However, this is likely to be a large underestimate of the true number of such sites for several reasons:

- The NAWQA Study Units cover less than 50 percent of the land area of the United States.
- Dissolved Pb was an analyte at only 16 percent of all NAWQA sampling locations.
- Dissolved Pb was measured only once or twice at many locations.
- For waters with a hardness of less than 47 mg/L as $CaCO_3$, the CCC for dissolved Pb is less than the quantitation limit for dissolved Pb that was used until the fall of 2000 (i.e., 1 µg/L).
- Fewer than 15 percent of samples analyzed for dissolved Pb between 1994 and 2004 were assessed with the lower quantitation limit of 0.08 µg/L, which is a value that is sufficiently low to match the CCC for waters with a hardness as low as 4.7 mg/L $CaCO_3$.

The first two bullet points alone suggest that the number of such sites nationwide might easily be at least ten times higher than what was represented in the NAWQA database. In addition, where the land use around a sampling location was classified as "mining," no investigation was conducted to determine whether air emissions from a nearby smelter might also be contributing to the Pb in the water.

There are many sources of uncertainty in the results presented for the sampling locations for which there were some data, including the following:

- Many sampling locations are represented by only one or two measurements of dissolved Pb.
- The water hardness for some sampling locations was not measured or is represented by only one or two measurements.

- Where there are multiple measures of both dissolved Pb and water hardness at a given location, no attempt was made to match sampling dates and times to develop time-specific CCC values.[2]
- The water hardness measured at some locations was less than the lowest value of 20 mg/L of $CaCO_3$ used to develop the equation to calculate a CCC. The CCC equation is not necessarily valid at values less than 20 mg/L $CaCO_3$.
- It is not known how quickly dissolved Pb concentrations changed at any of the locations.
- The database supporting the current AWQC for Pb is over 20 years old; new AWQC for Pb should be available in 2007.

7.3.4.2 Sediment Screen

Results of this analysis cannot conclusively link any of the locations with probable adverse effects of Pb in sediments on benthic communities to ongoing air emissions of Pb. There is no clear correlation between HQs for the surface water column and HQs for the sediments. Some locations with the highest Pb concentrations in sediments have no obvious air or NPDES-permitted discharge sources of Pb. Some locations (e.g., Utah and Montana) suggest historical drainage of Pb from mining activities. Other locations with high HQs for benthic communities (e.g., near or in cities) suggest historical deposition of Pb from the use of leaded gasoline. Erosion of Pb-contaminated soils from near roadways to these surface waters may be a significant ongoing source of Pb input in such areas.

This analysis was limited to those 15 locations from the NAWQA database at which dissolved concentrations of Pb in surface waters exceeded the chronic AWQC for Pb. Those 15 locations are believed to represent a small fraction of surface waters in the United States for reasons described in Section 7.3.4.1.

An additional limitation is that where the land use around a sampling location was classified as "mining," no investigation was conducted to determine whether air emissions from a nearby smelter might also be contributing to the Pb in sediments. It was assumed that direct runoff and erosion from the mining sites to the surface waters would have contributed the bulk of the Pb in sediments.

Further limitations accrue from the sediment sampling data. There were only nine exact matches and three near matches between the 15 surface water sampling locations of interest and locations at which sediment samples also were analyzed. Furthermore, there was a single sediment sample at each of the locations of interest, some of which were taken in the early 1990s.

Finally, no adjustments were made for sediment-specific characteristics that might affect the bioavailability of Pb in sediments in the derivation of the sediment quality criteria used for this risk screen.

[2] The coefficient of variation (i.e., the standard deviation divided by the mean – see Appendix H for standard deviation and mean values for $CaCO_3$ measurements) for water hardness measurements was less than 10 or 20 percent for many stations; however, at some locations, the coefficient of variation exceeded 50 percent, indicating higher fluctuations in water hardness measurements.

8. References

Agency for Toxic Substances and Disease Registry (ATSDR). (2002) *Health Consultation; Public Health Implications from Attending or Working at Herculaneum Schools; Herculaneum Lead Smelter Site; Herculaneum, Jefferson County, Missouri.* June.

Agency for Toxic Substances and Disease Registry (ATSDR). (2003) *Health Consultation; Blood Lead Results for 2002 Calendar Year; Herculaneum Lead Smelter Site; Herculaneum, Jefferson County, Missouri, EPA facility ID MOD006266373.* August.

Agency for Toxic Substances and Disease Registry (ATSDR). (2005) *Draft Toxicological Profile for Lead.* Atlanta, GA: U.S. Department of Health and Human Services, Pubic Health Service. September.

Alabama Department of Environmental Management (ADEM). (2006) *Conversations with representative of ADEM regarding available measured data from media in and around the secondary Pb smelter in Troy, AL.* October 31 and November 1, 2006.

Alabama Natural Heritage Inventory. (2001) *Alabama Inventory List: The Rare, Threatened and Endangered Species, Animals and Natural Communities of Alabama.* Montgomery, AL: Privately printed by the Alabama Natural Heritage Program.

Alexander, F.W., Clayton, B.E., and Delves, H.T. (1974) Mineral and trace-metal balances in children receiving normal and synthetic diets. *Q. J. Med.* 43: 89-111.

Alvin, R., Diamond, J.R., Woods, M., Hall, J.A., and Martin, B.H. (2002) The vascular flora for the Pike County Poscosin Nature Preserve, Alabama. *Southeastern Naturalist.* 1(1):45-54.

Battelle Memorial Institute. (1996) *Sources of Lead in Soil: A Literature Review.* Prepared for by Battelle for EPA/OPPTS. EPA 747-R-98-001b.

Canfield, R.L., Henderson, C.R., Jr., Cory-Slechta, D.A., Cox, C., Jusko, T.A., and Lanphear, B.P. (2003) Intellectual impairment in children with blood lead concentrations below 10 µgrams per deciliter. *N. Engl. J. Med.* 348: 1517-1526.

Caravanos, J., Weiss, A.L., and Jaeger, R.J. (2006) An exterior and interior leaded dust deposition survey in New York City: Results of a 2-year study. *Environmental Research.* 100:159-164.

Casteel, S.W., Tessman R., Brattin W.J. and Wahlquist A.M.. (2005) Relative Bioavailability of Lead in House Dust and Soil from the Herculaneum Lead Smelter Site in Herculaneum, Missouri. Prepared for Black & Veatch Special Projects Corporation. May 2005.

Centers for Disease Control and Prevention (CDC). (1997) Third National Health and Nutrition Examination Survey (NHANES III), 1988-94, Dietary Recall Data Available at http://www.cdc.gov/nchs/products/elec_prods/subject/nhanes3.htm.

Centers for Disease Control and Prevention (CDC). (2003) *CDC Surveillance Data, 1997-2004.* Available at http://www.cdc.gov/nceh/lead/surv/stats.htm.

Centers for Disease Control and Prevention (CDC). (2004) Children's Blood Lead Levels in the United States. Childhood Lead Poisoning Prevention Branch, National Center for Environmental Health, CDC. Available at http://www.cdc.gov/nceh/lead/research/kidsBLL.htm#National%20surveys.

Chirenje, T., Ma, L.Q., Reeves, M., and Szulczewski, M. (2004) Lead distribution in near-surface soils of two Florida cities: Gainesville and Miami. *Geoderma.* 119(2):113-120.

Choctawhatchee Pea and Yellow Rivers Watershed Management Authority (CPYRWMA). (2006) *About the Watersheds: The Rivers.*

Clayton, C.A., Pellizzari, E.D., Whitmore, R.W., Perritt, R.L., and Quackenboss, J. J. (1999) National human exposure assessment survey (NHEXAS): distributions and associations of lead, arsenic, and volatile organic compounds in EPA Region 5. *J. Exposure Anal. Environ. Epidemiol*. 9: 381-392.

Cohen, J., Cook, R., Bailey, C.R., and Carr, E. (2005) Relationship between motor vehicle emissions of hazardous pollutants, roadway proximity, and ambient concentrations in Portland, Oregon. *Environmental Modeling & Software.* 20:7-12.

Cook, M.R. and Kopaska-Merkel, D.C. (1996) *Hydrologic Characterization of the Water Resources of the Choctawhatchee-Pea Rivers Watershed - Phase I.* Tuscaloosa, AL: Geological Survey of Alabama. 137p.

EC/R Incorporated. (2006) *Secondary Lead Smelter Industry – Source Characterization for Residual Risk Assessment.* Prepared for USEPA Office of Air and Radiation, Office of Air Quality Planning and Standards, Research Triangle Park, NC. November.

ELM Consulting LLC. (2003) *Sampling Analysis Plan: Characterization Area Investigation: The Doe Run Company Lead Smelter, Herculaneum, Missouri (Version 1).* Prepared for the Doe Run Company. March.

ELM Consulting LLC. (2005) *Characterization Area Investigation Report.* July.

Environment Canada and Ministere de l-Environnement du Quebec (EC and MENVIQ). (1992) *Interim Criteria for Quality Assessment of St. Lawrence River Sediment.* Ottawa, Ontario: Environment Canada. ISBN 0-662-19849-2.

Fakayode, S.O., and Olu-Owolabi, B.I. (2003) Heavy metal contamination of roadside topsoil in Osogbo, Nigeria: its relationship to traffic density and proximity to highways. *Environmental Geology*. 44:150-157.

Filippelli, G.M., Laidlaw, M.A.S., Latimer, J.C., and Raftis, R. (2005) Urban lead poisoning and medical geology: An unfinished story. *GSA Today.* 15(1):4-11.

Gao, Y., Nelson, E.D., Field, M.P., Ding, Q., Li, H., Sherrell, R.M., Gigliotti, C.L., Van Ry, D.A., Glenn, T.R., and Eisenreich, S.J. (2002) Characterization of atmospheric trace elements on PM2.5 particulate matter over the New York-New Jersey harbor estuary. *Atmospheric Environment.* 36:1077-1086. (as cited in EPA 2006, criteria document)

Gradient Corporation. (2004) *RAGS Part D Interim Deliverables Report for Community Risk Assessment; Herculaneum, Missouri (Draft).* Prepared for the Don Run Company. October.

Great Backyard Bird Count (GBBC). (2003) *Great Backyard Bird Count; 2003 Results: Troy, Alabama.*

Gustavsson, N., Bolviken, B., Smith, D.B., and Severson, R.C. (2001) *Geochemical Landscapes of the Conterminous United States -- New Map Presentations of 22 Elements.* U.S. Department of the Interior, U.S. Geological Survey. Professional Paper 1648. Available at http://pubs.usgs.gov/pp/p1648.

Hafen, M.R., and Brinkmann, R. (1996) Analysis of lead in soils adjacent to an interstate highway in Tampa, Florida. *Environmental Geochemistry and Health.* 18:171-179.
Hanson, R.L. (1991) *Evapotranspiration and Droughts in National Water Summary 1988-1989 - Hydrologic Events and Floods and Droughts.* U.S. Geological Survey Water-Supply Paper 2375. Available at http://geochange.er.usgs.gov/sw/changes/natural/et.

Harrison, G. (1987) A survey of the lead distribution in soil of Corpus Christi, Texas. *Texas Journal of Science.* 39(1):15-22.

Henderson, R. (2006) *Clean Air Scientific Advisory Committee (CASAC) Lead Review Panel's Consultation on EPA's Draft Analysis Plan for Human Health and Ecological Risk Assessment for the Review of the Lead National Ambient Air Quality Standards.* Memorandum to Stephan Johnson, EPA Administrator, from Dr. Rogene Henderson. July. Available at http://www.epa.gov/sab/pdf/casac-con-06-006.pdf.

Hubbard Brook Information Oversight Committee. (2001) *Overview.* Available at http://www.hubbardbrook.org/research/overview/hbguidebook.htm.

ICF International. (2005) *Task 2: Near Roadway Concentrations (Revised): A Memorandum to Chad Bailey and Rich Cook, EPA from Jonathan Cohen and Arlene Rosenbaum.* September.

ICF International. (2006a) *Refinement of the Lead Eco-SSLs for Birds and Mammals for the Lead NAAQS Review: A Memorandum to USEPA from Meg McVey.* September.

ICF International. (2006b) *Ambient PM2.5 and PM10 Sampling Conducted by Desert Research Institute (DRI) at Seven Locations – Pb Results Used in Pb NAAQS Pilot Phase Risk Assessment: A Memorandum to Zach Pekar, EPA, from David Burch and Arlene Rosenbaum.* November.

Interagency Monitoring of Protected Visual Environments (IMPROVE). (2006) *IMPROVE Database.* Available at http://vista.circa.colostate.edu/improve/.

International Commission on Radiological Protection (ICRP). (2002) ICRP Publication 89: Basic anatomical and physiological data for use in radiological protection. Annals of the ICRP 32 (3-4):100.

Johnson, C., Siccama, T., Driscoll, C., Likens, G., and Moeller, R. (1995) Changes in lead biochemistry in response to decreasing atmospheric inputs. *Ecological Applications.* 5(3):813-822.

Keppie, D. M. and R. M. Whiting, Jr. 1994. *American Woodcock (Scolopax minor).* In The Birds of North America, No. 100 (A. Poole and F. Gill, Eds.). Philadelphia: The Academy of Natural Sciences; Washington, D.C.: The American Ornithologists' Union.

Kimbrough, D.E. and Suffet, I.H. (1995) Off-site forensic determination of airborne elemental emissions by multi-media analysis: A case study at two secondary lead smelters. *Environmental Science & Technology.* 29:2217-2221.

Lanphear, B.P., Matte, T.D., Rogers, J., Clickner, R.P., Dietz, B., Bornschein, R.L., Succop, P., Mahaffey, K.R., Dixon, S., Galke, W., Rabinowitz, M., Farfel, M., Rohde, C., Schwartz, J., Ashley, P., and Jacobs, D.E. (1998) The Contribution of Lead-Contaminated House Dust and Residential Soil to Children's Blood Lead Levels: A Pooled Analysis of 12 Epidemiologic Studies. *Environmental Research* 79:51-68.

Lanphear, B.P., Hornung, R., Khoury, J., Yolton, K., Baghurst, P., Bellinger, D.C., Canfield, R.L., Dietrich, K.N., Bornschein, R., Greene, T., Rothenberg, S.J., Needleman, H.L., Schnaas, L., Wasserman, G., Graziano, J., and Robe, R. (2005) Low-level environmental lead exposure and children's intellectual function: An international pooled analysis. *Environmental Health Perspectives.* 113(7).

Leggett,R.W. (1993) An Age-Specific Kinetic Model of Lead Metabolism in Humans. *Environ Health Perspect* 101:598-616.

Lejano, R.P. and Ericson, J.E. (2005) Tragedy of the temporal commons: Soil-bound lead and the anachronicity of risk. *Journal of Environmental Planning and Management.* 48(2):301-320.

MacDonald, D. D. 1994. *Approach to the Assessment of Sediment Quality in Florida Coastal Waters, Volume I.* Tallahasse, FL: Florida Department of Environmental Protection.

MacDonald, D.D., Ingersoll, C.G., and Berger, T.A. (2000) Development and evaluation of consensus-based sediment quality guidelines for freshwater ecosystems. *Archives of Environmental Contamination and Toxicology.* 39:20-31.

MacDonald, D.D., Ingersoll, C.G., Smorong, D.E., Lindskoog, R.A., Sloane, G., and Biernacki, T. (2003) *Development and Evaluation of Numerical Sediment Quality Assessment Guidelines for Florida Inland Waters.* British Columbia: MacDonald Environmental Sciences, Lt. Columbia, MO: U.S. Geological Survey. Prepared for: Florida Department of Environmental Protection, Tallahassee, FL. January.

Maddaloni,M., Bellew,M., Diamond,G., Follansbee,M., Gefell,D., Goodrum,P., Johnson,M., Koporec,K., Khoury,G., Luey,J., Odin,M., Troast,R., Van,L.P., and Zaragoza,L. (2005) Assessing Lead Risks at Non-Residential Hazardous Waste Sites. *Human and Ecological Risk Assessment* 11:967-1005

Maynard, J.B. (2006) *Conversation with J. B. Maynard, University of Cincinnati, regarding sampling date for soil sampling performed in Corpus Christi, TX, for research on heavy metal contamination in highway soils.* (See Turer and Maynard 2003 for discussion of sampling results.) November 3, 2006.

Missouri Department of Conservation (MDC). (2006) *MDC Heritage Database Results for Jefferson County.* Available at http://www.mdc.mo.gov/cgi-bin/heritage/search.cgi?county=Jefferson.

Missouri Department of Natural Resources (MDNR). (2000) *2000 Revision of the State Implementation Plan for Lead for the Doe Run Resources Corporation Primary Lead Smelter Herculaneum, Missouri, Public Hearing – October 26, 2000.* Jefferson City, MO: Air Pollution Control Program, Division of Environmental Quality. October.

Missouri Department of Natural Resources, Division of Environmental Quality. (1999) *Preliminary Assessment: Herculaneum Lead Smelter Site, Jefferson County*, Missouri. March 30, 1999.

Moir, C. M., Freedman, B., and McCurdy, R. (1996) Metal mobilization from water-distribution systems of buildings serviced by lead-pipe mains. *Can. Water Resour. J.* 21: 45-52.

National Climatic Data Center (NCDC). (2006) *Climatography of the U.S., Maps of Annual 1961-1990, Normal Temperature, Precipitation and Degree Days.* Compiled by Owenby, J., Heim, Jr., R., Burgin, M., and Ezell, D. No. 81 - Supplement #3. Available at http://www.ncdc.noaa.gov/oa/documentlibrary/clim81supp3/clim81.html.

National Oceanic and Atmospheric Administration (NOAA). (1997a) *Hourly United States Weather Observations 1990-1995.* Asheville, NC: National Climatic Data Center (NCDC).

National Oceanic and Atmospheric Administration (NOAA). (1997b) *Rediosonde Data of North America.* Asheville, NC: National Climatic Data Center.

NatureServe. (2006) *NatureServe Explorer: An Online Encyclopedia of Life [Web Application]. Version 5.0.* Arlington, VA and Pike County, AL: NatureServe. Available at http://www.natureserve.org/explorer.

Outdoor Alabama. (2003) *Listing of Alabama Species on the Federal List for Threatened and Endangered Species or Whose Status is a Concern.* Available at http://www.outdooralabama.com/watchable-wildlife/regulations/endangered-species.cfm.

Persaud, D., Jaagumagi, R., and Hayton, A. (1993) *Guidelines for the Protection and Management of Aquatic Sediment Quality in Ontario.* Canada: Ontario Ministry of the Environment and Energy. August.

Pirrone, N., Keeler, G.J., and Warner, P.O. (1995) Trends of ambient concentrations and deposition fluxes of particulate trace metals in Detroit from 1982 to 1992. *Science of Total Environment.* 162(43):61. (as cited in EPA 2006, criteria document).

Pounds J.G. and Leggett J. (1998) The ICRP Age-Specific Biokinetic Model for Lead: Validations, Empirical Comparisons, and Explorations. *Environ. Health Pers.* 106 (Suppl 6):1505-1511.

Pounds JG. (2005) Personal communication with William Mendez, ICF International, including the ICRP (Leggett) model FORTRAN code and User's Manual.

Rieuwerts, J.S., Farago M., Cikrt M., and Bencko V.. (1999) Heavy metal concentrations in and around households near a secondary lead smelter. *Environmental Monitoring and Assessment* 58: 317-335.

Ruby, M.V., Schoof, R., Brattin, W., Goldade, M., Post, G., Harnois, M., Mosby, D.E., Casteel, S.W., Berti, W., Carpenter, M., Edwards, D., Cragin, D., and Chappell, W. (1999) Advances in evaluating the oral bioavailability of inorganics in soil for use in human health risk assessment. *Environ. Sci. Technol*. 33(21): 3697-3705.

Sánchez-Martin, M.J., Sánchez-Camazano, M. and Lorenzo, L.F. (2000) Cadmium and lead contents in suburban and urban soils from two medium-sized cities of Spain: influence of traffic intensity. *Bulletin of Environmental Contamination and Toxicology*. 64:250-257.

Schulman, L.L., Stimaitis, D.G., and Scire, J.S. (1997) *Addendum to ISC3 User's Guide: The Prime Plume Rise and Building Downwash Model.* Palo Alto, CA: Electric Power Research Institute. Earth Tech Document A287. A-99-05, II-A-12. November. Available at http://www.epa.gov/scram001/7thconf/iscprime/useguide.pdf.

Smith, S.L., MacDonald, D.D., Keenleyside, K.A., Ingersoll, C.G., and Field, L.J. (1996) A preliminary evaluation of sediment quality assessment values for freshwater ecosystems. *Journal of Great Lakes Research.* 22(3):624-638.

Soil Survey Division Staff. (1999) *Soil Taxonomy.* Soil Conservation. U.S. Department of Agriculture Handbook 436, Second Edition.

Speiran, G.K. (1998) *Selected heavy metals and other constituents in soil and stormwater runoff at the Interstate 95 Interchange near Atlee, Virginia, April 1993-May 1997.* U.S. Geological Survey. Water-Resources Investigations Report 98-4115, 39p.

Sutherland, R.A. and Tolosa, C.A. (2001) Variation in total and extractable elements with distance from roads in an urban watershed, Honolulu, Hawaii. *Water, Air, and Soil Pollution.* 127(4):315-338.

Sweet, C.W., Weiss, A., and Vermette, S.J. (1998) Atmospheric deposition of trace metals at three sites near. *Water, Air, and Soil Pollution.* 103:423-439. (as cited in EPA 2006, criteria document).

Texas Natural Resource Conservation Commission (TNRCC). 2001. *Guidance for Conducting Ecological Risk Assessments at Remediation Sites in Texas.* Austin, TX: Toxicology and Risk Assessment Section, Texas Natural Resource Conservation Commission. RG-263 (revised).

Turer, D.G. and Maynard, J.B. (2003) Heavy metal contamination in highway soils. Comparison of Corpus Christi, Texas and Cincinnati, Ohio shows organic matter is key to mobility. *Clean Technologies and Environmental Policy.* 4(4):235-245.

U.S. Census Bureau. (2005) *United States Census 2000: Summary File 1.* Public Information Office. Available at http://www.census.gov/Press-Release/www/2001/sumfile1.html.

U.S. Department of Transportation. (2006) *Highway Performance Monitoring System.* Federal Highways Administration. Available at http://www.fhwa.dot.gov/policy/ohpi/hpms/index.htm.

U.S. Environmental Protection Agency (USEPA). (1984) *Ambient Water Quality Criteria for Lead - 1984.* Washington, DC: Office of Water Regulations and Standards, Criteria and Standards Division. EPA 440/5-B4-027. Available at http://yosemite.epa.gov/water/owrccatalog.nsf/065ca07e299b464685256ce50075c11a/05edf009dcde1d7d85256b060072307e!OpenDocument.

U.S. Environmental Protection Agency (USEPA). (1989) *Review of the National Ambient Air Quality Standard for Lead: Exposure Analysis Methodology and Validation.* Research Triangle Park, NC: Office of Air Quality Planning and Standards. EPA-450/2-89-011. June.

U.S. Environmental Protection Agency (USEPA). (1990a) Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions. Office of Health and Environmental Assessment. EPA 600/6-90/003. Available at http://www.ntis.gov/help/ordermethods.asp?loc=7-4-0.

U.S. Environmental Protection Agency (USEPA). (1990b) Review of the national ambient air quality standards for lead: assessment of scientific and technical information: OAQPS staff paper. Research Triangle Park, NC: Office of Air Quality Planning and Standards; report no. EPA-450/2-89/022. Available from: NTIS, Springfield, VA; PB91-206185. Available at http://www.epa.gov/ttn/naaqs/standards/pb/data/rnaaqsl_asti.pdf.

U.S. Environmental Protection Agency (USEPA). (1993) *Office of Water Policy and Technical Guidance on Interpretation and Implementation of Aquatic Life Metals Criteria: A Memoradum by Martha G. Prothro.* Washington, DC: U.S. Environmental Protection Agency, Office of Water. October. Available at http://www.epa.gov/waterscience/library/wqcriteria/metalsinterpret.pdf.

U.S. Environmental Protection Agency (USEPA). (1994) Technical Support Document: Parameters and Equations Used in the Integrated Exposure Uptake Biokinetic Model for Lead in Children (v.099d). Office of Solid Waste. EPA 540/R-94/040.

U.S. Environmental Protection Agency (USEPA). (1995) *User's Guide for the Industrial Source Complex (ISC3) Dispersion Models, Volume 1- User Instructions.* Washington, DC: Office of Air Quality Planning and Standards. Available at

http://www.epa.gov/scram001/userg/regmod/isc3v1.pdf#search=%22%22user's%20guide%20for%20the%20industrial%20source%20complex%22%22.

U.S. Environmental Protection Agency (USEPA). (1998) *Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions.* Cincinnati, OH: Update to EPA/600/6-90/003, EPA/NCEA (EPA 600/R-98/137). Available at oaspub.epa.gov/eims/eimscomm.getfile?p_download_id=427339.

U.S. Environmental Protection Agency (USEPA). (2001) *Quality Assurance Project Plan for a Site Characterization at the Herculaneum Lead Smelter.* Herculaneum, Missouri, CERCLIS ID No.: MOD 006266373. Prepared for USEPA, Region 7, Superfund Division by USEPA Region 7 Superfund Technical Assessment and Response Team 2. September.

U.S. Environmental Protection Agency (USEPA). (2002a) *Addendum to User's Guide for the AERMOD Meteorological Preprocessor.* Research Triangle Park, NC: Office of Air Quality Planning and Standards; Emissions, Monitoring and Analysis Division. EPA-454/B-02-002b.

U.S. Environmental Protection Agency (USEPA). (2002b) User's Guide for the Integrated Exposure Uptake Biokinetic Model for Lead in Children (IEUBK) Windows version – 32 bit version. EPA 540-K-01-005. OSWER #9285.7-42, Office of Solid Waste and Emergency Response, Washington, DC.

U.S. Environmental Protection Agency (USEPA). (2002c) Children-Specific Exposure Factors Handbook Interim Draft. National Center for Environmental Assessment, Office of Research and Development. EPA-600-P-00-002B.

U.S. Environmental Protection Agency (USEPA). (2003) *Guidance for Developing Ecological Soil Screening Levels (Eco-SSLs), Attachment 1-4: Exposure Factors and Bioaccumulation Models for Derivation of Wildlife Eco-SSLs.* Washington, DC: Office of Solid Waste and Emergency Response. OSWER Directive 9285.7-55. November. Available at http://rais.ornl.gov/homepage/ecossl.pdf.

U.S. Environmental Protection Agency (USEPA). (2004a) *User's Guide for the AMS/EPA Regulatory Model - AERMOD.* Office of Air Quality Planning and Standards. EPA-454/B-03-001. September.

U.S. Environmental Protection Agency (USEPA). (2004b) *Addendum User's Guide for the AMS/EPA Regulatory Mode - AERMOD.* Office of Air Quality Planning and Standards. Available at http://www.epa.gov/ttn/scram/7thconf/aermod/aerguide_addm.pdf.

U.S. Environmental Protection Agency (USEPA). (2005a) Integrated Exposure Uptake Biokinetic Model for Lead in Children, Windows® version (IEUBKwin v1.0 build 263). Available at http://www.epa.gov/superfund/lead/products.htm.

U.S. Environmental Protection Agency (USEPA). (2005b) *Guidance for Developing Ecological Soil Screening Levels.* Washington, DC: Office of Solid Waste and Emergency Response. OSWER Directive 9285.7-55. November.

U.S. Environmental Protection Agency (USEPA). (2005c) *Ecological Soil Screening Levels for Lead, Interim Final.* Washington, DC: Office of Solid Waste and Emergency Response. OSWER Directive 9285.7-70. Available at http://www.epa.gov/ecotox/ecossl/pdf/eco-ssl_lead.pdf.

U.S. Environmental Protection Agency (USEPA). (2005d) *Procedures for the Derivation of Equilibrium Partitioning Sediment Benchmarks (EBSs) for the Protection of Benthic Organisms: Metal Mixtures (Cadmium, Copper, Lead, Nickel, Silver, and Zinc).* Washington, DC: Office of Research and Development. EPA-600-R-02-011.

U.S. Environmental Protection Agency (USEPA). (2006a) *Plan for the Review of the National Ambient Air Quality Standards for Lead.* Research Triangle Park, NC: Office of Air Quality Planning and Standards. February. Available at http://www.epa.gov/ttn/naaqs/standards/pb/s_pb_cr_pd.html

U.S. Environmental Protection Agency (USEPA). (2006b) *Air Quality Criteria for Lead, Volume I of II.* Research Triangle Park, NC: National Center for Environmental Assessment - RTP Office. EPA/600/R-5/144aF. October. Available at: http://cfpub.epa.gov/ncea/cfm/recordisplay.cfm?deid=158823

U.S. Environmental Protection Agency (USEPA). (2006c) *Review of the National Ambient Air Quality Standards for Lead: Policy Assessment of Scientific and Technical Information. OAQPS Staff Paper - First Draft.* Research Triangle Park, NC: Office of Air Quality Planning and Standards. December. Reference should be updated when the title is finalized.

U.S. Environmental Protection Agency (USEPA). (2006d) *Analysis Plan for Human Health and Ecological Risk Assessment for the Review of the Lead National Ambient Air Quality Standards. Draft.* Research Triangle Park, NC: Office of Air Quality Planning and Standards. May.

U.S. Environmental Protection Agency (USEPA). (2006e) *1999 National-Scale Air Toxics Assessment.* Available at http://www.epa.gov/ttn/atw/nata1999/nsata99.html.

U.S. Environmental Protection Agency (USEPA). (2006f) *AirData: Access to Air Pollution Data.* Available at http://www.epa.gov/air/data.

U.S. Environmental Protection Agency (USEPA). (2006g) *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities.* Office of Solid Waste and Emergency Response. EPA530-R-05-006. September. Available at http://www.epa.gov/epaoswer/hazwaste/combust/risk.htm.

U.S. Environmental Protection Agency (USEPA). (2006h) Specific estimates of dietary Pb intake developed by EPA's Office of Solid Waste and Emergency Response. Available at http://www.epa.gov/superfund/lead/ieubkfaq.htm#FDA.

U.S. Environmental Protection Agency (USEPA). (2006i). 2002 National Emissions Inventory Data & Documentation, Final Version 1 Release. February. Available at: http://www.epa.gov/ttn/chief/net/2002inventory.html

U.S. Environmental Protection Agency (USEPA) Region 7. (2000) *Final DOE Run Resources Corporation Administrative Order on Consent.*

U.S. Environmental Protection Agency (USEPA) Region 7. (2006a) *Herculaneum Lead Master 090804 (Microsoft Access Database).* Obtained from Region 7 and Updated with 2005 and 2006 Data.

U.S. Environmental Protection Agency (USEPA) Region 7. (2006b) *Predicted Lead Deposition, Herculaneum, Missouri (April 1, 1997 through March 21, 1999). Memorandum from Richard L. Daye to Gene Gunn.* January.

U.S. Food and Drug Administration (FDA). (2001) Total Diet Study. U. S. Food and Drug Administration Center for Food Safety and Applied Nutrition Office of Plant and Dairy Foods and Beverages (June 2001). Available at http://www.cfsan.fda.gov/~comm/tds-toc.html.

U.S. Geological Survey. (2004) *National Water Quality Assessment Program.* Available at http://water.usgs.gov/nawqa/data.

van Wijnen JH, Clausing P, and Brunekreef B. (1990) Estimated soil ingestion by children. *Environ Res.* 51(2):147-62.

Vogelmann, J.E., Howard, S.M., Yang, L., Larson, C.R., Wylie, B.K., and Van Driel, J.N. (2001) Completion of the 1990's National Land Cover Data Set for the conterminous United States. *Photogrammetric Engineering and Remote Sensing.* 67:650-662. Available at http://edc.usgs.gov/products/landcover/nlcd.html.

Von Lindern, I., Spalinger, S. Petroysan, V., and Von Braun, M. (2003) Assessing Remedial Effectiveness Through Blood Lead:Soil/Dust Relationship at the Bunker Hill Superfund Site in the Silver Valley of Idaho. *Science of the Total Environment* 303: 139–170

Wang, E., Bormann, F., and Benoit, G. (1995) Evidence of complete retention of atmospheric lead in the soils of northern hardwood forested ecosystems. *Environmental Science & Technology.* 29(3):735-739.

Wang, E. and Benoit, G. (1996) Mechanisms controlling the mobility of lead in the spodosols of a northern hardwood forest ecosystem. *Environmental Science & Technology.* 30(7):2211-2219.

White, P.D., Van Leeuwen, P., Davis, B.D., Maddaloni, M., Hogan, K.A., and Marcus, A.H. (1998) The conceptual structure of the integrated exposure uptake biokinetic model for lead in children. *Environmental Health Perspectives.* 106(S6):1513-1530.

Wu, Z.Y., Han, M., Lin, Z.C., and Ondov, J.M. (1994) Chesapeake Bay atmospheric deposition study, year 1: Sources and dry deposition of selected elements in aerosol particles. *Atmospheric Environment.* 28:1471-1486. (as cited in EPA 2006, criteria document).

Yanai, R., Ray, D., and Siccama, T. (2004) Lead reduction and redistribution in the forest floor in New Hampshire northern hardwoods. *Journal of Environmental Quality.* 33(1):141-148.

Yi, S.M., Shahin, U., Sivadechathep, J., Sofuoglu, S.C. , and Holsen, T.M. (2001) Overall elemental dry deposition velocities measured around Lake Michigan. *Atmospheric Environment.* 35:1133-1140. (as cited in EPA 2006, criteria document).

Ziegler, E. E., Edwards, B. B., Jensen, R. L., Mahaffey, K. R., and Fomon, S. J. (1978) Absorption and retention of lead by infants. *Pediatr. Res.* 12: 29-34.