4.0 DOSE-RESPONSE ASSESSMENT

CHAPTER 4 SUMMARY

Chapter 4 presents the approach for characterizing the relationship between environmental lead exposure and the resulting adverse health effects. The relationship is established in two stages. First, the relationship between environmental lead levels and blood-lead concentration is characterized. Two different models, the IEUBK and empirical models, are used to characterize this relationship. Then the relationship between blood-lead concentration and specific elevated blood-lead concentration and health effect endpoints is established. This two-stage relationship is applied in this risk analysis (Chapter 5), using environmental data from the HUD National Survey, to estimate the number of children who will benefit from the §403 rule.

This chapter describes the two models that are used to relate environmental-lead levels to blood-lead concentration and establishes the relationship between blood lead and the specific elevated blood-lead concentration and health effect endpoints. Methods for converting environmental lead levels measured by different sampling methods are also presented.

Figure 4-1 outlines the approach for the dose-response assessment. The conclusions from the dose-response assessment are presented in Section 4.5.

This chapter seeks to answer the following questions:

1. What is the dose-response relationship between environmental-lead exposure and the blood-lead concentration and health effect endpoints evaluated in this risk analysis?

    1a. What is the dose-response relationship between environmental-lead exposure and childhood blood-lead concentration?

    1b. What is the dose-response relationship between childhood blood-lead concentration and health effects?

2. Can lead loadings in dust samples collected using a vacuum sampler be converted to wipe-equivalent dust-lead loadings?

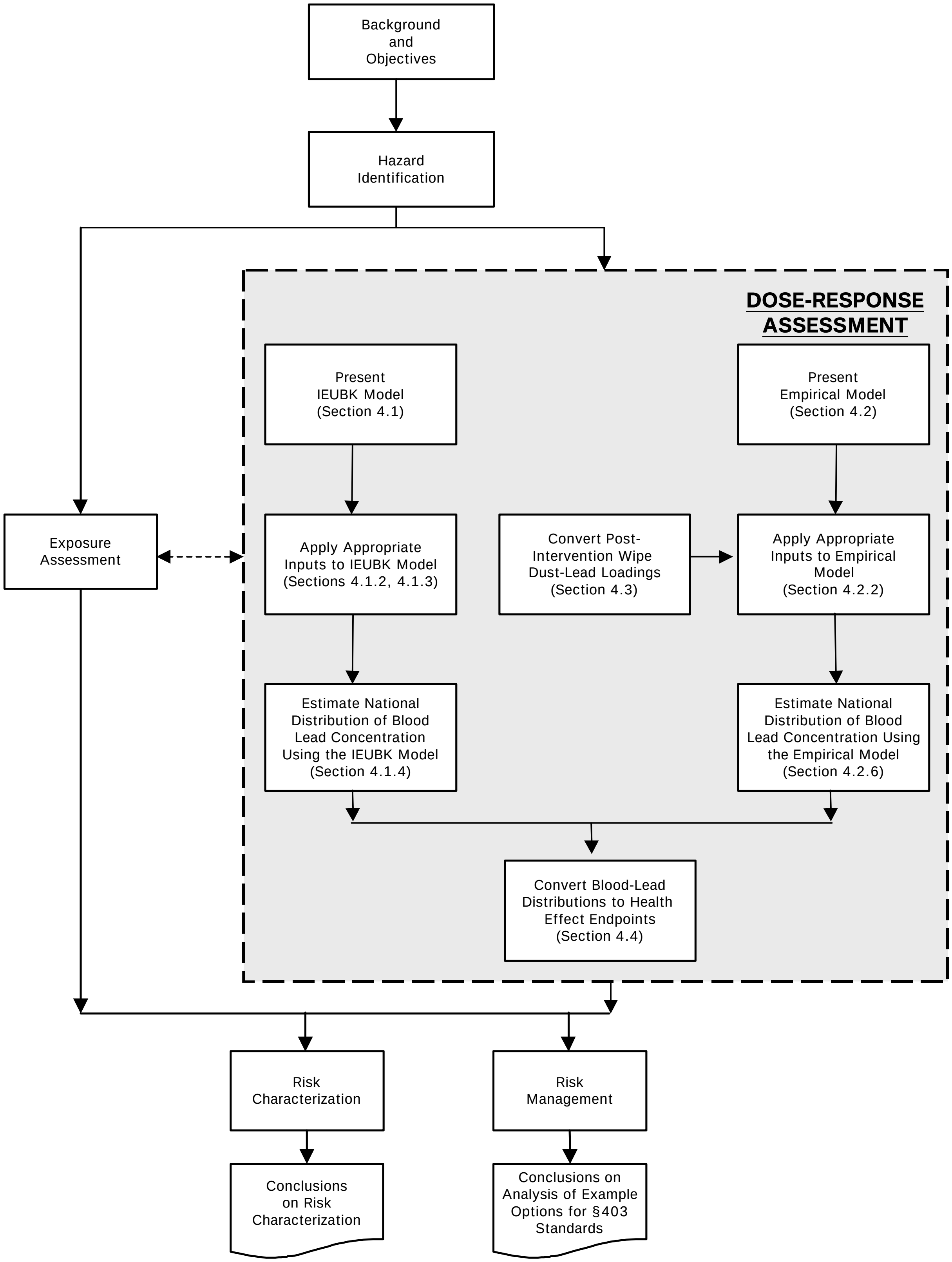

Figure 4-1. Detailed Flowchart of the Approach to Dose-Response Assessment.

Answering question 1, the primary question addressed in this chapter, is problematic, as only limited data exist for relating health outcomes directly to environmental-lead levels. The link between lead exposure and health effects is usually studied in terms of a measure of body-lead burden, such as blood-lead concentration, rather than environmental-lead levels. Therefore, the answer to question 1 is obtained by addressing questions 1a and 1b. The relationship between environmental-lead levels and health outcomes is computed in a two stage process. This two stage dose-response relationship is used to characterize the risk due to lead exposure under present environmental conditions (Chapter 5) and to estimate the risk under environmental conditions predicted to occur for various examples of options for the §403 standards (Chapter 6). The specific health effect and blood-lead concentration endpoints utilized in this risk analysis were identified in Chapter 2. Question 2 is necessary as §403 dust-lead standards are expected to be defined in terms of a wipe dust-lead loading, and dust samples in the HUD National Survey (the primary source of data on environmental-lead levels in the nation's housing stock used in this risk analysis) were collected via vacuum sampling.

Figure 4-1 illustrates the relationship between material presented in this chapter and other key elements of the risk analysis. Two models are utilized to relate environmental-lead levels to blood-lead concentrations: the Integrated Exposure, Uptake, and Biokinetic (IEUBK) Model for Lead in Children (USEPA, 1994a, 1995d) and an empirical model that was developed specifically for this risk analysis. The application of the IEUBK model in this risk analysis is described in Section 4.1. The development of the empirical model and its application in this risk analysis are described in Section 4.2. Briefly, each model is applied to characterize the national distribution of blood-lead concentrations of children aged 1-2 years both prior to and following implementation of §403 rulemaking ("pre-§403" and "post-§403"). Data collected in the HUD National Survey serve as inputs to both models for the estimation of the pre-§403 distribution. Estimation of post-§403 environmental lead distributions is discussed in Chapter 6. Section 4.3 presents conversion equations developed to relate dust-lead loadings under different dust sampling methods (e.g., Blue Nozzle vacuum) to wipe dust-lead loadings. These conversions are used to compare environmental levels to standards and to prepare the post-intervention data for input to the models. The national distributions of blood-lead concentrations predicted by each model are used as input to the second stage models relating blood-lead concentrations to health outcomes. Section 4.4 presents the approach for relating blood-lead concentration to the elevated blood-lead concentration and health effects endpoints identified in Chapter 2. The dose-response characterization (Section 4.5) provides summary answers to the above questions and addresses the strengths and weaknesses of the scientific evidence and decisions made, as they are relevant to this risk analysis.

4.1 IEUBK MODEL

This section describes how EPA's Integrated Exposure, Uptake, and Biokinetic (IEUBK) Model for Lead in Children (USEPA, 1994a, 1995d) is used in this risk analysis to model the dose-response relationship between environmental-lead levels in the nation's housing stock and blood-lead concentration in children aged 1-2 years.

The precursor to the biokinetic part of the IEUBK model was developed in 1985 by EPA's Office of Air Quality Planning and Standards (OAQPS) as a tool for setting air lead standards. The version used by the Air program was peer reviewed and found acceptable by EPA’s Clean Air Science Advisory Committee of the Science Advisory Board (USEPA, 1990b). The IEUBK model has been recommended as a risk assessment tool to support the implementation of the July 14, 1994 Office of Solid Waste and Emergency Response (OSWER) Interim Directive on Revised Soil Lead Guidance for CERCLA Sites and RCRA Facilities. The most current version, Version 0.99D, of the IEUBK model is used in this risk analysis.

4.1.1 Description of the IEUBK Model

The IEUBK model employs exposure, uptake, and biokinetic information to predict a distribution of blood-lead levels in children corresponding to a specific combination of environmental-lead levels. The predicted distribution may be used to predict the probability of elevated blood-lead levels in children exposed to similar environmental-lead levels. The model addresses three components of environmental risk assessment: 1) multimedia nature of exposures to lead, 2) the differential bioavailability of various sources of lead, 3) the pharmacokinetics of internal distribution of lead to bone, blood, and other tissues, and 4) inter-individual variability in blood-lead levels.

Specifically, the model uses lead concentrations measured in dust, soil, air, water, diet, and other ingested media to estimate a longitudinal exposure pattern from birth to seven years of age (USEPA, 1995d). The model then estimates a distribution of blood-lead levels for a population of children receiving similar exposures. The center of this distribution, the geometric mean, is predicted by the model. A constant empirical estimate is used by the model to represent the variability about the geometric mean. In statistical terminology, this variation is referred to as the geometric standard deviation (GSD). The GSD characterizes the inter-individual and biological variability in blood-lead levels of children exposed to similar environmental-lead levels. The IEUBK model is not intended to predict the blood-lead level of an individual child and cannot substitute for a medical evaluation of an individual child.

It is beyond the scope of this document to describe the IEUBK model in detail. Very briefly, the model has three distinct functional components that work together in series: exposure, uptake, and biokinetic components. Each model component is a set of complex equations and parameters. The Technical Support Document (USEPA, 1995d) provides the scientific basis of the parameters and equations used in the model, while the Guidance Manual (USEPA, 1994a) includes a detailed description of the exposure pathways, absorption mechanism, and biokinetic compartments and associated compartmented transfers of lead.

4.1.2 Inputs to the IEUBK Model

This section describes the inputs to the IEUBK model used in this risk analysis. Three sets of parameters are used in the IEUBK model equations. (1) Exposure parameters are used to estimate the amount of environmental lead that is taken into the body, through breathing or ingestion. (2) Uptake parameters estimate the amount of lead that is absorbed from environmental sources. (3) Biokinetic parameters characterize the transfer of lead between compartments of the body (for example, between blood and bone) and the elimination of lead from the body. The IEUBK model allows the user to input values for most exposure and uptake parameters. The biokinetic parameter values, however, are not accessible.

For this risk analysis, soil- and dust-lead concentrations from the HUD National Survey (Section 3.3.1.1) are used as inputs to the IEUBK model to predict national distribution of blood-lead concentrations that represents baseline (pre-§403) conditions, while adjusted concentrations are used to predict a blood-lead concentration distribution for post-§403 conditions. IEUBK model default values are applied for all other parameters. The default parameter values for the IEUBK model and the calculation of input values based on the HUD National Survey are described in this section.

IEUBK Model Default Parameters

When exposure and uptake parameter values are not specified, the IEUBK model program provides default values. Table 4-1 presents the default values for the exposure and uptake parameters. The default parameter values are based on various studies and are considered the best available estimates for urban residents with no unusual lead exposure (USEPA, 1994a, 1995d). For example, the default air lead concentration is 0.1 $\mu g/m^3$, which is approximately the average 1990 urban air lead concentration (USEPA, 1991). Thus, blood-lead concentrations estimated using the default parameter values for exposure other than dust and soil represent the 'background' blood-lead levels that cannot be avoided (USEPA, 1994a). The use of default parameter values is documented in detail in the IEUBK Guidance Manual (USEPA, 1994a). While the Guidance Manual encourages the use of site-specific estimates, the default parameter values are appropriate for assessment of national risk. In addition, site-specific estimates for the default parameter values utilized in this risk analysis were not available.

Data from many different scientific studies of lead biokinetics, contact rates of children with environmental media, and data on the presence and behavior of environmental lead were utilized in developing the IEUBK model default parameter values. Details on these data sources and the derivation of the default parameter values are provided in the Technical Support Document (USEPA, 1995d). In brief, default values fall into five general categories: exposure rates, exposure concentrations, uptake of ingested lead, biokinetic parameters, and variability in blood-lead levels. Key (to this risk analysis) default parameter values in each category are described briefly below.

Exposure rates: The age-weighted dust and soil ingestion rates used as defaults in the model (85-135 mg/day) represent central tendency values within the range of values seen in different studies. The default proportion (45%) of total dust and soil ingested that is derived from soil is based primarily on a study of Dutch children in day care centers (USEPA, 1994a), contrasting dust plus soil ingestion on days with good weather with dust ingestion on days with rainy weather (presumably little outdoor activity on those days).

Table 4-1. Summary of Default Parameter Values Used in the IEUBK Model (Version 0.99D).

Air Parameters

Parameter	Vary air concentration by year?	Outdoor air lead concentration	Indoor air lead concentration
Setting*	No	0.10 µg/m^3	30% of outdoor value

* All air parameters use default values

Diet Intake Parameters

Parameter	Lead intake in diet, by age of child						
	0-1 yrs	1-2 yrs	2-3 yrs	3-4 yrs	4-5 yrs	5-6 yrs	6-7 yrs
Setting*	5.53 µg/day	5.78 µg/day	6.49 µg/day	6.24 µg/day	6.01 µg/day	6.34 µg/day	7.00 µg/day

* All diet intake parameters use default values

Water Intake Parameters

Parameter	Lead Conc. in Water	Drinking water consumption, by age of child						
		0-1 yrs	1-2 yrs	2-3 yrs	3-4 yrs	4-5 yrs	5-6 yrs	6-7 yrs
Setting*	4 µg/L	0.20 L/day	0.50 L/day	0.52 L/day	0.53 L/day	0.55 L/day	0.58 L/day	0.59 L/day

* All water intake parameters use default values

Soil and Dust Intake Parameters

Parameter	Soil/Dust Ingestion Weighting Factor	Total soil + dust intake, by age of child						
		0-1 yrs	1-2 yrs	2-3 yrs	3-4 yrs	4-5 yrs	5-6 yrs	6-7 yrs
Setting*	45% soil; 55% dust	0.085 g/day	0.135 g/day	0.135 g/day	0.135 g/day	0.1 g/day	0.09 g/day	0.085 g/day

* Soil and dust lead concentrations are input. All other parameters use default values.

Absorption Method Parameters

Parameter	Half Saturation Level	Total Absorption					Fraction of Total Assumed Passive Absorption				
		Soil	Dust	Water	Diet	Alt.	Soil	Dust	Water	Diet	Alt.
Setting	100 µg/day	30%	30%	50%	50%	0%	0.20	0.20	0.20	0.20	0.20

Blood Lead Parameter

Parameter	Geometric Standard Deviation (GSD)
Setting	1.6

Exposure concentrations: The default dust- and soil-lead concentrations are not used in this risk analysis. Default diet values (5.53-7.00 μg/day) are based on data from the Food and Drug Administration. No other data were available. Model default water values were considered adequate for communities without a particular water-lead problem. The default air lead concentration (0.1 µg/m^3) is approximately the average 1990 urban air lead concentration.

Uptake of ingested lead: Lead bioavailability varies across the chemical forms in which lead can exist. Many factors complicate the estimation of bioavailability, including nutritional status and timing of meals relative to lead intake (lead uptake generally increases as dietary levels of calcium, iron, phosphate, vitamin D, fats, etc. decrease), age, and magnitude of exposure. The default media-specific bioavailabilities in the IEUBK model are central tendency estimates.

Biokinetic parameters: The data on which these parameter values are based originate from a variety of separate investigations, including as much clinical data as were available (USEPA, 1995d). The biokinetic parameters cannot be changed by the user.

Variability in blood leads (GSD): A variety of factors may cause children exposed to similar environmental-lead concentrations to have varying blood-lead concentrations. These include differences in children's tendency to ingest soil or dust, hygiene habits, the potential for soil or dust to be deposited on food, and biological factors that may affect the absorption and processing of lead. The complexity of these factors suggests that the overall variability encompassed by the GSD cannot be determined by aggregating the variability in each of these factors into an overall GSD estimate. Instead, an empirical estimate of the variability in blood-lead concentrations, a GSD of 1.6, was estimated from residential community blood-lead studies (USEPA, 1995d). This estimate is applied for predictions of the national distribution of blood-lead concentrations utilizing both the IEUBK and empirical models (Section 4.2).

Figure 4-2 illustrates the relationship between blood-lead concentration predicted by the IEUBK model and specified soil- or dust-lead concentration, for children aged 24 months. The solid line illustrates the relationship for a fixed dust-lead concentration of 200 ppm and varying soil-lead concentrations. For the dashed line, the soil-lead concentration was fixed at 100 ppm and dust-lead concentration varied. These fixed values are similar to the geometric mean dust-lead concentration (192 ppm) and soil-lead concentration (78 ppm), reported in the HUD National Survey. From the dashed line in Figure 4-2, the predicted blood-lead concentration is 3 µg/dL for a dust-lead concentration of 100 ppm and a soil-lead concentration of 100 ppm. Similarly, from the solid line, the predicted blood-lead concentration for 200 ppm soil- and dust-lead concentrations is 4.5 µg/dL. It is important to recognize that each point on the predicted curve represents a geometric mean blood-lead level for children exposed to similar environmental-lead levels. The blood-lead levels for individual children will vary.

Utilizing the HUD National Survey Data

The IEUBK model is used in this risk analysis to predict a national distribution of children's blood-lead concentrations. A nationally representative sample of environmental-lead levels in housing is required to provide inputs to the IEUBK model for this purpose. The HUD National Survey is a recent nationally representative study that assessed environmental-lead levels in paint, dust and soil in residential housing.

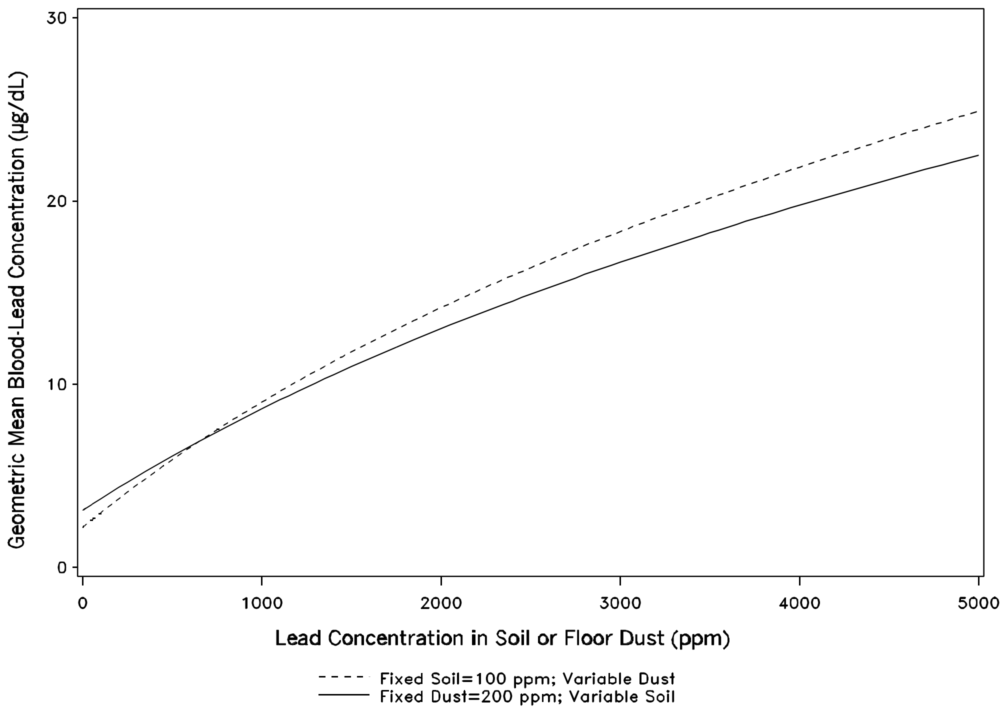

Figure 4-2. IEUBK Model Predicted Geometric Mean Blood-Lead Concentration for Children Aged 24 Months Plotted Separately Against Soil-Lead Concentration and Dust-Lead Concentration for Fixed Default Values of the Remaining Model Parameters

In the HUD National Survey (Section 3.3.1.1), one floor-dust sample was collected from each of three locations (wet, dry, and entry rooms) using a Blue Nozzle vacuum sampler. The mass weighted dust-lead concentration of these three samples is input to the IEUBK model, to represent the dust-lead concentration to which a child is exposed. Three soil samples were collected, one each from dripline, entryway, and remote locations. A factor weighted soil-lead concentration (0.25 * dripline measurement + 0.25 * entryway measurement + 0.5 * remote measurement) is used to represent the average soil-lead concentration in the yard. This weighting scheme avoids double counting the concentration near the house (i.e., the dripline and entryway samples) and does not estimate the amount of time children spend in specific areas of the yard.

There are some limitations inherent in using the HUD National Survey data to provide inputs to the IEUBK model. For example, the IEUBK model uses the specified lead concentrations in conjunction with soil and dust ingestion rates and bioavailability factors to determine the dose of lead absorbed by the body. This dose is then used to predict the geometric mean blood-lead concentration for children exposed to the specified lead concentrations. An important assumption is that the dust- and soil-lead concentrations input to the IEUBK model are representative of the actual lead concentration to which a child is exposed. Thus, risk assessors typically use children's activity patterns to guide the selection of dust and soil samples. It is not possible, however, to determine whether children are actually exposed to soil- and dust-lead levels represented by the samples collected in each HUD National Survey home. Therefore, it is uncertain whether the mass weighted dust-lead concentrations and the factor weighted soil-lead concentrations are typical of childhood lead exposures. A number of factors affect the actual exposure scenario for an individual child, including the number of hours the child spends playing inside and outside the residence, the amount of time that the child spends away from home, the presence of pets that spend time inside and outside, the frequency and thoroughness of house cleaning, air conditioning, parental occupation, and a host of similar factors. Such factors can differ among neighborhoods, communities, and time periods (Stark et al., 1982; Bornschein et al., 1985b)

4.1.3 Estimating the Effect of Pica for Paint on Childhood Blood-Lead Levels

The exposure pathway from lead-based paint to childhood blood-lead concentration can be both direct and indirect. Indirect exposure takes place when deteriorated lead-based paint contaminates residential dust or soil, which is then ingested by the child. Direct exposure takes place through the ingestion of paint chips. While the IEUBK model estimates the geometric mean blood-lead concentration for children receiving indirect exposure to lead-based paint through the soil- and dust-lead concentrations used as model inputs, it does not include a direct mechanism for estimating the contribution of paint chip ingestion to childhood blood lead. This section describes how this risk analysis accounts for the effect of pica for paint on the geometric mean blood-lead concentrations predicated by the IEUBK model. Note that this approach was developed specifically for this risk analysis and is not a component of the IEUBK model.

As described in Section 4.1.2, environmental conditions observed in the HUD National Survey are used as input to the IEUBK model. For each home in the HUD National Survey, the IEUBK model is used to predict the geometric mean blood-lead concentration of children exposed to those environmental conditions. The distribution of blood-lead levels in the population of children aged 1-2 years is then characterized by allowing each home in the HUD National Survey to represent a proportion of the total number of children aged 1-2 years in the country. For homes without damaged lead-based paint, the predicted geometric mean blood-lead concentration and the assumed geometric standard deviation of 1.6 are used to model the distribution of blood-lead levels in children represented by each home.

For homes with damaged lead-based paint (defined as greater than 0 ft^2 of interior or exterior deteriorated lead-based paint), adjustments are made to the IEUBK model predictions to account for the effect of pica for paint on children's blood-lead concentrations. In this adjustment procedure, the children represented by each of these homes are assigned into three groups: 1) children who have recently ingested paint chips (0.03%), 2) children who ingested paint chips at some time (8.97%), and 3) children who do not ingest paint chips (91%). The distribution of blood-lead levels for children in the three groups is estimated as follows:

1. Children who have recently ingested paint chips (0.03%) – Blood-lead concentration is assigned the value 63 µg/dL with no variation.

2. Children who ingested paint chips at some time (8.97%) – Geometric mean blood-lead concentration is 3.0 µg/dL greater than the geometric mean blood-lead concentration predicted by the IEUBK model. The adjusted geometric mean blood-lead concentration and the assumed geometric standard deviation of 1.6 µg/dL are used to model the distribution of blood-lead levels for these children.

3. Children who do not ingest paint chips (91.0%) – The IEUBK model predicted geometric mean blood-lead concentration and the assumed geometric standard deviation of 1.6 µg/dL are used to model the distribution of blood-lead levels for these children.

The scientific evidence and assumptions used to select percentages of children assigned to each group and the adjustments to blood-lead concentrations for children who have ingested paint chips are described in Appendix D1.

4.1.4 Estimating the National Distribution of Blood-Lead Using the IEUBK Model

For prediction of the pre-§403 national distribution of children's blood-lead concentrations, estimates of soil- and dust-lead concentrations observed in the HUD National Survey are used as inputs to the IEUBK model, as described in Section 4.1.2. If the input values for the IEUBK model were missing for a home in the HUD National Survey, then an imputed value is used in the risk analysis. The imputed values are summarized in Table 3-14 of Section 3.3.1.1, with more details provided in Appendix C1. The IEUBK model is then used to predict the geometric mean blood-lead concentration associated with each home in the HUD National Survey.

To predict a post-§403 national distribution of children's blood-lead concentrations, the following method was used to prepare soil- and dust-lead concentrations in the HUD National Survey data for input into the IEUBK model:

1. Observed levels of lead in environmental variables in the HUD National Survey were compared to candidate §403 standards. Blue-nozzle vacuum floor and window sill dust-lead loadings were converted to wipe dust-lead loadings before comparison to the §403 standards. Although sill dust-lead levels are not provided as input to the IEUBK model, they are used to determine which homes require an intervention.

2. §403 interventions were triggered in HUD National Survey residential units that had levels of lead in environmental variables that were above the candidate standard. If an intervention was triggered, assumed post-intervention lead levels in environmental variables were substituted for observed levels. Post intervention dust-lead concentrations for use in the prediction are determined from methods documented in Section 4.3.

The geometric mean blood-lead concentrations predicted by the IEUBK model, an assumed geometric standard deviation of 1.6, and population weights adjusted to the 1997 population of children (aged 1-2 years), are used to predict the pre- and post-§403 national distributions of blood-lead concentrations.

Although the IEUBK model simulates a longitudinal exposure pattern from birth to seven years of age, in order to simplify calculations for this risk analysis, a specific age was selected at which blood-lead concentrations are estimated. The representative population for the risk analysis is children aged 1-2 years. IEUBK model-predicted blood-lead concentrations were examined for each month over the 12-35 month period. Predicted blood-lead concentrations at age 24 months were found to be approximately equal to the mean predicted values over the entire two-year period. Thus, IEUBK model predictions at age 24 months are utilized in the risk analysis to characterize blood-lead concentrations of children aged 1-2 years.

4.2 EMPIRICAL MODEL

This section describes the development and application of an empirical model in this risk analysis. The empirical model was developed using data from the Rochester Lead-in-Dust Study to estimate the relationship between blood-lead levels in young children and observed levels of lead in environmental media (paint, dust and soil) from their primary residences. The purpose of this model is to serve as a basis for predicting a national distribution of children’s blood-lead concentrations as a function of environmental lead-levels observed in the HUD National Survey. Variables were selected for the model from among those that were measured in both studies, or could be constructed in both studies using the available data. The mathematical form of the model, variables included in the model, and parameter estimates based on the Rochester study are presented in Sections 4.2.1 through 4.2.3. The model was then adjusted to account for systematic differences and differences in error structure between the Rochester study variables and the analogous HUD National Survey variables (Section 4.2.4). The final form of the empirical model is presented in Section 4.2.5. The application of the empirical model to predict the national distribution of blood-lead concentrations is described in Section 4.2.6.

The choice and construction of variables, the mathematical form of the empirical model, assessment of goodness of fit and influential points, and the treatment of measurement error in predictor variables are described in detail in Appendix G. The empirical model has not yet undergone formal peer review or model evaluation, and is based on data from only one source (the Rochester Lead-in-Dust Study). It is not intended as a general dose-response model, but rather as a predictive model developed specifically for use in this risk analysis and specifically to predict a national distribution of blood-lead concentrations from estimates of environmental lead as measured in the HUD National Survey.

4.2.1 Form of the Model

The empirical model is log-linear in nature, expressing natural-log transformed blood-lead concentration as a linear combination of natural-log transformed exposure variables and select covariates. A typical multimedia exposure log-linear model for blood-lead concentrations might appear as follows:

$$\ln(PbB_i) = \beta_0 + \beta_1 \cdot \ln(Dust_i) + \beta_2 \cdot \ln(Soil_i) + \beta_3 \cdot \ln(Paint_i) + \gamma \cdot Covariate_i + e_i$$

where PbB_i is the observed blood-lead concentration of the i^{th} child, $Dust_i$, $Soil_i$, and $Paint_i$ are the environmental lead levels in the home of the i^{th} child, $Covariate_i$ represents one or more variables with strong predictive value, and e_i (the residual error) is assumed to follow a normal distribution with mean zero and variance σ^2_{Error}.

When translated back into the original scale of observed blood-lead concentrations, the log-linear model yields a multiplicative relationship between environmental-lead levels and blood-lead concentration:

$$PbB_i = \exp(\beta_0) \cdot Dust_i^{\beta_1} \cdot Soil_i^{\beta_2} \cdot Paint_i^{\beta_3} \cdot Covariate_i^{\gamma} \cdot \exp(e_i)$$

Thus, for example, the effect of dust lead on blood lead is dependent on the combined effects of all of the other variables included in the model. Furthermore, the difference between predicted blood-lead concentrations for children exposed to dust-lead loadings of 5 and 50 µg/ft² is the same as that between children exposed to dust-lead loadings of 500 and 5000 µg/ft² if the values of the other variables are constant. Although the multiplicative interpretation of the log-linear model is not considered biologically or physically plausible, for low to moderately exposed children, the log-linear model often fits the data better than statistical models with a more plausible, biological/physical basis (Rust et al., 1996; Jiang and Succop, 1996).

4.2.2 Variable Selection

The criteria used for the selection of predictor variables in the empirical model emphasized use of measures of environmental lead and other factors observed in both the Rochester Lead-in-Dust Study and the HUD National Survey. Variables whose translation between the two studies was straightforward, whose statistical relationship with blood-lead concentration in the Rochester study was significant, and whose values in the HUD National Survey covered a wide range, were used in the empirical model.

The predictor variables selected for the final model are described below. Each variable is first defined as in the Rochester study for model development purposes. Next, the definition of an analogous variable based on the HUD National Survey data is presented. This latter definition was employed when applying the empirical model to the environmental data from the HUD National Survey to estimate a national distribution of children's blood-lead concentrations.

Floor Dust-Lead Loading

The empirical model was developed using the natural logarithm of the area-weighted arithmetic average (wipe) dust-lead loading from carpeted and uncarpeted floors in the Rochester study. For this risk analysis, the natural logarithm of the area-weighted arithmetic average floor dust-lead loading from 3 sample locations (wet, dry and entry rooms) in HUD National Survey homes (as measured using Blue Nozzle vacuum techniques) is used as the measure of lead in floor dust.

Window Sill Dust-Lead Loading

The empirical model was developed using the natural logarithm of the area-weighted arithmetic average (wipe) dust-lead loading from window sills in the Rochester study. For this risk analysis, the natural logarithm of the area-weighted arithmetic average (Blue Nozzle Vacuum) dust-lead loading from window sills in 2 sample locations (wet and dry rooms) in HUD National Survey homes is used as the measure of lead in window sill dust.

Soil-Lead Concentration

The empirical model was developed using the natural logarithm of the dripline soil-lead concentration (fine soil fraction) in the Rochester study. Dripline soil samples in the Rochester study were thoroughly homogenized and sieved into coarse and fine fractions using a 2 mm mesh sieve followed by a 250 µm mesh sieve. These two soil fractions were chemically analyzed separately, and results from the fine soil fraction were selected for statistical analysis. For this risk analysis, the natural logarithm of the weighted average concentration of samples collected from dripline, entryway, and remote locations (with weights of 25%, 25%, and 50%, respectively) from HUD National Survey homes is used as the measure of soil-lead concentration.

Extent of Paint/Pica Hazard

The empirical model was developed using a paint/pica variable that took into account the presence and condition of lead-based paint (LBP) in the home and the tendency of the child to ingest paint chips. The following question in the Rochester study questionnaire was designed to measure mouthing behavior or pica tendencies in resident children:

How often does the child put paint chips in his/her mouth?

The possible responses to this question were: 0 = Never, 1 = Rarely, 2 = Sometimes, 3 = Often, and 4 = Always. For the empirical model, a categorical variable (paint/pica) was constructed that was nonzero when the home contained some damaged or deteriorated interior lead-based paint (determined by whether any paint had a condition of fair or poor) and the response to the above pica question was 1 or greater (i.e., at least rarely). This variable was defined to have values of 0, 1, and 2, which were defined as follows:

0 No LBP present (maximum XRF reading < 1 mg/cm^2) <u>or</u> condition of paint is rated as Good <u>or</u> child does not exhibit pica;

1 LBP present (maximum XRF reading ≥ 1 mg/cm^2) <u>and</u> paint condition is Fair or Poor <u>and</u> child exhibits pica rarely;

2 LBP present (maximum XRF reading $\geq$ 1 mg/cm²) and paint condition is Fair or Poor and child exhibits pica at least sometimes.

For the Rochester study, condition of the paint was characterized as Good when less than 5% of the surface was deteriorated, Fair when 5% to 15% of the surface was deteriorated, and Poor when more than 15% of the surface was deteriorated.

For this risk analysis, the value of the paint/pica variable for each home in the HUD National Survey is determined by a combination of the presence of deteriorated lead based paint and an assumed 9% of children aged 1-2 years who exhibit pica for paint (Appendix D1). For homes with no deteriorated lead-based paint, the value of the paint-pica variable is set to zero. For homes that were found to contain deteriorated lead-based paint, it was assumed that 9% of children living in a similar environmental would ingest paint chips at some time. For those children, the value of the paint/pica variable is set at 1.5, which is the average response to the paint pica question in Rochester among children who exhibited pica for paint. The remaining 91% of the children living in homes with deteriorated lead-based paint are assumed to exhibit no pica for paint. Thus, the paint/pica variable is set equal to zero for 91% of children living in homes with deteriorated lead-based paint.

The development of the empirical model for this risk analysis is complicated by the fact that the sampling methodology used to measure lead exposures in the HUD National Survey is different from that used in the Rochester Lead-in-Dust Study. Specifically, two of the lead exposure measurements from the HUD National Survey are blue nozzle vacuum floor dust-lead loading and blue nozzle vacuum window sill dust-lead loading, compared to floor dust-lead loading and window still wipe dust-lead loading in Rochester. Thus, these variables have different interpretations in the two studies.

In addition, the soil variable from the HUD National Survey is the weighted average of samples collected from dripline, entryway and remote locations (with weights of 25%, 25%, and 50%, respectively), whereas the soil variable from the Rochester study is based on a composite sample from the dripline area only. Also, the paint/pica variable from the HUD National Survey data was based on the measures of paint on both interior and exterior surfaces, whereas the variable from the Rochester study was based on measures of paint on only interior surfaces. Lead-based paint on deteriorated exterior surfaces was not considered in the estimation of the paint/pica model parameter based on Rochester data, because nearly every home surveyed in the Rochester study had deteriorated lead-based paint on exterior surfaces. The differences in paint/pica variable construction between the Rochester study and HUD National Survey are considered minor in comparison to the differences in the dust-lead loading and soil variables.

4.2.3 Rochester Multimedia Model

As a first step in developing the empirical model, a multi-media predictive model was developed using data from the Rochester Lead-in-Dust Study which explained children's blood-lead concentration as a function of dust-lead loadings from floors and window sills, drip-line soil-lead concentration and the paint/pica variable. The Rochester multimedia model was log-linear in nature, and specific details on model development are found in Appendix G. (The model is referred to as "the Multi-Media Predictive model based on Rochester data" in Appendix G.) Table 4-2 provides parameter estimates and associated standard errors for the multimedia predictive model.

Table 4-2. Parameter Estimates and Associated Standard Errors for the Rochester Multimedia Model

Parameter	Variable Description	Estimate (Standard Error)
β_0	Intercept	0.418 (0.240)
β_1	log (PbF): Area-Weighted Arithmetic Mean (Wipe) Dust-Lead Loading from Any Floor (Carpeted or Uncarpeted)	0.066 (0.040)
β_2	log (PbW): Area-weighted Arithmetic Mean (Wipe) Dust-Lead Loading from Window Sills	0.087 (0.036)
β_3	log (PbS): Dripline Soil-Lead Concentration (fine soil fraction)	0.114 (0.035)
β_4	PbP: Indicator of Interior Paint/Pica Hazard	0.248 (0.100)
R^2	Coefficient of Determination	21.67%
σ^2_{Error}	Error	0.316

The Rochester multimedia model is used in the risk characterization (Chapter 5) to determine the probability that a child exposed to specific levels of lead in paint, dust and soil will have a blood-lead concentration at or above 10 µg/dL.

4.2.4 Measurement Error Adjustment

The fact that the Rochester multimedia model lead exposure variables for paint, dust and soil are subject to measurement error raises concerns about the need to account for this measurement error in the model building process. The term "measurement error" is used to describe uncertainty in the predictor variables attributable to sampling, spatial, laboratory and/or temporal variability. The presence of measurement error in predictor variables, if not accounted for in the statistical models, could result in biased predictions (Fuller, 1987). In addition, because different sampling methods were used in the Rochester study and the HUD National Survey, adjustments for those different sampling methods may be needed when applying the empirical model to the HUD National Survey data.

The first question to be asked when addressing measurement error is: Is an adjustment for measurement error necessary? The appropriateness of an adjustment for measurement error depends on the use of the statistical model. One primary differentiation in model use concerns whether the model is being used to characterize the relationship between observed blood-lead levels in children and "true" lead exposures, or whether the model is being used to predict blood-lead levels based on some source of measured levels of environmental lead. The primary use of the empirical model in the §403 rulemaking is for the latter case (prediction). Therefore, a classic errors-in-variables adjustment was not considered necessary (Carroll et al., 1995).

However, to predict the national distribution of childhood blood-lead concentrations (prior to and following implementation of §403 rules), the empirical model must be combined with environmental data observed in a nationally representative sample (the HUD National Survey). An empirical model unadjusted for the effects of differences in measurement error in the lead exposure predictor variables would be appropriate for prediction of the national distribution of blood-lead concentrations, if the following four assumptions were acceptable:

1. The sampling scheme for environmental lead implemented in the Rochester study (or other studies used for model building) is similar to the sampling scheme implemented in the HUD National Survey.

2. The sampling collection devices and instruments used to measure lead have similar properties with respect to measurement error between the Rochester study and the HUD National Survey.

3. The distribution of observed environmental lead levels is similar between the Rochester study and the HUD National Survey.

4. The characteristics of the true exposure relationship in the Rochester study is the same as in the U.S. as a whole.

Investigation of the data from the Rochester study and the HUD National Survey suggested that the first three assumptions were unacceptable. Therefore, an adjustment for the differences in measurement error between predictor variables used in the model building process and input variables from the HUD National Survey used in the prediction process is appropriate. Although this can be considered an adjustment for “measurement error,” the resulting model should not be interpreted as the "true" relationship between blood-lead and environmental lead exposure (measured without error). Rather, this adjustment accounts for the differences in variability of the measured data in the two studies to facilitate a better prediction of the national distribution of childhood blood-lead concentrations using the data from the HUD National Survey.

If the fourth assumption is not acceptable, it is questionable whether the Rochester study is an appropriate source of data for informed decisions concerning lead exposures nationwide. There is no evidence to suggest that the fourth assumption is unacceptable.

4.2.5 Specification of the Empirical Model

When using the empirical model to predict a national distribution of children’s blood-lead concentrations, differences in dust and soil variables between the Rochester study and the HUD National Survey are accounted for by first establishing a relationship between blood-lead and environmental variables, as measured by methods used in the Rochester study (the Rochester

Multimedia Model), and then adjusting this relationship to use environmental variables as measured in the HUD National Survey. The adjustment takes into account both systematic differences and differences in error structures between the two sets of data and involves fitting a classic errors in variables model (Carroll et al., 1995) as an intermediate step. The method provides an empirical model of the relationship between blood-lead concentration and floor and window sill dust-lead loadings and other covariates as observed in the HUD National Survey.

In addition, the intercept of the empirical model was adjusted so that the geometric mean of the predicted national distribution of children's blood-lead concentrations matches that observed in Phase 2 of NHANES III.

The final mathematical form of the empirical model is:

$$\ln(PbB) = \beta_0 + \beta_1 \cdot \ln(PbF_{BN}) + \beta_2 \cdot \ln(PbW_{BN}) + \beta_3 \cdot \ln(PbS) + \beta_4 \cdot PbP + e$$

where PbB represents the blood-lead concentration, PbF_{BN} and PbW_{BN} correspond to average dust-lead loadings from interior floors and window sills respectively (assuming the Blue Nozzle vacuum technique), PbS represents average soil-lead concentration for the yard, PbP represents paint/pica hazard, and e represents the residual error left unexplained by the model. These predictor variables were introduced in Section 4.2.1. Table 4-3 provides parameter estimates and associated standard errors for the model parameters. The standard errors provided in Table 4-3 were estimated using a bootstrap algorithm. The empirical model is not intended to be used to estimate the effect of a single medium on blood-lead levels. The model should only be used to predict a distribution of blood-lead levels when environmental lead levels for all media are known or estimated. Individual parameter estimates in Table 4-3 should not be interpreted in isolation. Specific details on the development of the empirical model are found in Appendix G.

Table 4-3. Parameter Estimates and Associated Standard Errors for the Empirical Model Used to Predict the National Distribution of Children's Blood-Lead Concentration Based on Data from the HUD National Survey

Variable	Parameter	Estimate (Standard Error)
Intercept	β_0	0.651 (0.154)
Floor Dust-Lead Loading (Blue Nozzle Vacuum)	β_1	0.032 (0.044)
Window Sill Dust-Lead Loading (Blue Nozzle Vacuum)	β_2	0.050 (0.031)
Soil-Lead Concentration (Yard Average)	β_3	0.094 (0.043)
Paint/Pica	β_4	0.256 (0.098)
Error	σ^2_{Error}	0.313

4.2.6 Estimating the National Distribution of Blood-Lead Using the Empirical Model

The empirical model is used to predict a national distribution of children's blood-lead concentrations both before and after interventions resulting from the §403 standards. Environmental conditions observed in the HUD National Survey are used as input to the empirical model for predicting blood-lead levels in children 1-2 years old. A population of children aged 1-2 years is both the representative population for this risk analysis and similar to the age group that was recruited in the Rochester Lead-in-Dust Study (thus the empirical model is representative of children in this age group).

The empirical model is used to estimate the average log-transformed childhood blood-lead concentration associated with each home in the HUD National Survey. Input variables were constructed from observed levels of lead in each residential unit, as described in Section 4.2.2, for prediction of the current national distribution of children's blood-lead concentrations. If the input values for the empirical model were missing for a home, then an imputed value is used in the risk analysis. The imputed values are summarized in Table 3-14 of Section 3.3.1.1, with more details provided in Appendix C1.

To predict a post-§403 national distribution of children's blood-lead concentrations, the following method was used to prepare soil- and dust-lead concentrations in the HUD National Survey Data for input into the empirical model:

1. Observed levels of lead in environmental variables in the HUD National Survey were compared to proposed §403 standards. Blue-nozzle vacuum floor and window sill dust-lead loadings were converted to wipe dust-lead loadings before comparison to the §403 standards.

2. §403 interventions were triggered in HUD National Survey residential units that had levels of lead in environmental variables that were above the proposed standard. If an intervention was triggered, assumed post-intervention lead levels in environmental variables were substituted for observed levels.

The geometric mean blood-lead concentrations predicted by the empirical model, an assumed geometric standard deviation of 1.6, and population weights adjusted to the 1997 population of children (aged 1-2 years) are used to predict the pre- and post-§403 national distributions of blood-lead concentrations.

4.3 UTILIZING DUST LEAD LOADINGS

The HUD National Survey is the only national survey of environmental lead levels and therefore was used for prediction of a national distribution of blood-lead concentrations. Dust lead measurements in the HUD National Survey were collected by the Blue Nozzle (BN) vacuum method. However, §403 standards for dust will be expressed as a measured lead loading collected by a dust wipe sample. As a result, the following conversions are necessary in the risk analysis methodology:

- ! Converting Blue Nozzle dust-lead loadings observed in the HUD National Survey to wipe-equivalent dust-lead loadings to determine the extent to which homes in the United States are impacted by example options for the §403 dust-lead standard.

- ! Converting post-intervention wipe dust-lead loadings to Blue Nozzle-equivalent dust-lead loadings for input into the empirical model.

In addition, conversion factors were used to convert dust-lead loadings under the BRM dust sampling method employed in the Baltimore R&M Study to wipe equivalents for production of prevalence tables in Chapter 3.

This section presents the equations for the above conversions. The conversion equations are presented for dust samples collected from floors and window sills, since §403 rules will include standards for those housing components. These equations were established using data from environmental field studies where dust samples were collected by both sampling techniques from adjoining ("side-by-side") sample areas. Detailed information concerning the development of all the conversion equations discussed in this section, including a discussion of the studies and sample sizes used to estimate the equations is available in USEPA, 1997.

The use of a measurement error adjustment for the conversion from Blue Nozzle vacuum to wipe-equivalent dust-lead loadings is described in USEPA, 1997, as well. This adjustment was required because the data employed to develop the conversion equations possessed different distributional characteristics than the data to which the conversion equations were applied to (HUD National Survey). Similar adjustments were not employed for the other conversions. For the wipe to Blue Nozzle vacuum dust-lead loading conversion, there was not enough information to determine whether an adjustment was appropriate, since the conversion is applied simply to the assumed post-intervention wipe lead loadings. In the case of the BRM to wipe conversions, the data sets were similar and no adjustment was needed.

4.3.1 Wipe Versus Blue Nozzle (BN) Vacuum Conversions

Three studies reported side-by-side wipe and BN vacuum dust-lead measurements:

1. CAPS Pilot Study (USEPA, 1995i)
2. National Center for Lead-Safe Housing (NCLSH)/Westat Study (Westat, 1995)
3. Baltimore Repair and Maintenance (R&M) Pilot Study (Battelle, 1992)

To obtain conversion factors from one collection method to another, log-linear regression models were fitted to dust-lead loading data for each study separately. Weighted averages of the parameter estimates from each model were used to obtain the following equations (written in the scale of the original data) for conversions between wipe and BN vacuum sampling methods. Confidence intervals and prediction intervals are provided in USEPA, 1997.

1. Equations used to predict a wipe dust-lead loading from a BN vacuum dust-lead loading:

   Uncarpeted Floors:

Homes built prior to 1940:	$Wipe_{load} = 5.66\,(BN_{load})^{0.809}$
Homes built 1940-1969:	$Wipe_{load} = 4.78\,(BN_{load})^{0.800}$
Homes built 1960-1979:	$Wipe_{load} = 4.03\,(BN_{load})^{0.707}$

   Window Sills:

   All homes: $Wipe_{load} = 2.95\,(BN_{load})^{1.18}$

   Note that differences among the three categories of houses determined by age prompted different conversion equations for dust-lead loading on uncarpeted floors, but different equations were not necessary for window sills. As an example of using these equations, a BN dust-lead loading of 100 µg/ft² on an uncarpeted floor in a house built prior to 1940 would be converted to a wipe dust-lead loading of 235 µg/ft². In developing these equations, the observed BN lead loadings ranged from 1.0 to 2,164 µg/ft² on floors, and 1.4 to 8,964 µg/ft² on window sills. Extrapolation is necessary for BN loadings outside this range.

2. Equations used to predict a BN vacuum dust-lead loading from a wipe dust-lead loading:

   Uncarpeted Floors:

   All units: $BN_{load} = 0.185\,(Wipe_{load})^{0.931}$

   Window Sills:

   All units: $BN_{load} = 0.955\,(Wipe_{load})^{0.583}$

   Thus, for example, a wipe lead loading of 100 µg/ft² on an uncarpeted floor would be converted to a BN lead loading of 13.5 µg/ft². In developing these equations, the observed wipe lead loadings ranged from 7.6 to 6,755 µg/ft² on floors, and from 3.0 to 425,000 µg/ft² on window sills.

4.3.2 Wipe Versus Baltimore Repair and Maintenance (BRM) Vacuum Conversions

In characterizing dust-lead loadings in the Baltimore R&M Study (Section 3.2.2.1) for this risk analysis, it was necessary to express the loadings relative to wipe collection techniques, rather than BRM vacuum techniques. Therefore, it was necessary to convert BRM dust-lead loadings to wipe-equivalent loadings to obtain the summary statistics provided in Chapter 3.

Four studies reported side-by-side wipe and BRM vacuum dust-lead measurements:

1. R&M Mini Study (Farfel, 1994)
2. Rochester Lead-In-Dust Study (USHUD, 1995a)
3. NCLSH 5-Method Comparison Study (Westat, 1995)
4. Milwaukee Low-Cost Intervention Study (USEPA, 1997)

These studies are described in USEPA, 1997.

An analogous approach to that presented in Section 4.3.1 was used to develop the following equations for predicting a wipe dust-lead loading from a BRM vacuum dust-lead loading:

Uncarpeted Floors: $\text{Wipe} = 8.34\ \text{BRM}^{0.371}$
Carpeted Floors: $\text{Wipe} = 3.01\ \text{BRM}^{0.227}$
Window Sills: $\text{Wipe} = 14.8\ \text{BRM}^{0.453}$

For instance, a BRM dust-lead loading of 100 $\mu g/ft^2$ on an uncarpeted floor would be converted to a wipe dust-lead loading of 46.0 $\mu g/ft^2$. In developing these equations, the observed BRM lead loadings ranged from 0.1 to 74,100 $\mu g/ft^2$ on uncarpeted floors, from 1.4 to 141,000 $\mu g/ft^2$ on carpeted floors, and from 0.3 to 4,170,000 $\mu g/ft^2$ on window sills.

Note that the floor dust-lead samples in the Baltimore R&M Study were collected as composite samples (i.e., sub-samples from different locations combined into a single sample), which eliminates the ability to distinguish uncarpeted floor samples from carpeted floor samples. However, it was possible to determine the number of uncarpeted and carpeted subsamples within each composite sample. Therefore, floor dust-lead loadings from the Baltimore R&M Study were converted to wipe-equivalent loadings as follows:

$$\text{Wipe} = p \bullet 8.34\ \text{BRM}^{0.371} + (1\text{-}p) \bullet 3.01\ \text{BRM}^{0.227},$$

where p represents the proportion of the composite sample obtained from uncarpeted floors, and BRM represents the dust-lead loading under BRM vacuum sampling techniques. For example, a BRM dust-lead loading of 100 $\mu g/ft^2$ in a composited floor-dust sample consisting of 3 uncarpeted and 2 carpeted subsamples would be converted to a floor wipe dust-lead loading of 31.1 $\mu g/ft^2$.

4.4 HEALTH OUTCOMES

This section presents the approach for determining the incidence of adverse health outcomes resulting from lead exposure in young children and characterizes the relationship of certain elevated blood-lead concentration thresholds and other health effect endpoints to blood-lead concentrations. These relationships are applied to predicted geometric mean blood-lead concentrations, as predicted by the IEUBK and empirical models, to relate environmental lead exposure to health effects. The risk characterization in Chapter 5 and risk management analysis in Chapter 6 apply these relationships to the predicted national distribution of blood-lead concentrations to estimate incidences of elevated blood lead concentrations and health effects in children aged 1-2 years. For example, the relationship between blood-lead concentration and IQ scores is used to estimate the average IQ point loss due to lead exposure and the percentage of children with IQ point decrements greater than or equal to one, two, or three IQ points.

4.4.1 Decrements in IQ Scores

The IQ point loss health effect represents the neurological loss that a child experiences due to low level lead exposure. The relationship between blood-lead concentration and IQ point decrements has received considerable study, as described in Section 2.3. Multiple attempts have been made to quantify the effect using meta-analysis to combine the varying estimates reported in the scientific literature (Schwartz, 1993; Pocock et al., 1994; Schwartz, 1994). Meta-analysis is an often used statistical technique that is used to combine the results of statistical summaries and inferences across multiple studies. Several estimates of the relationship between blood-lead concentration and IQ scores were identified, one of which is applied within this risk analysis. Two additional estimates are used in the sensitivity analysis (Section 5.4.2) to determine the extent to which the uncertainty in estimating this parameter affects the characterization of risk. The approach to developing the estimate used in the risk analysis is described below.

Schwartz (1994) conducted a random effects meta-analysis to quantify the relationship between blood-lead concentrations and IQ scores. The results from seven studies were employed to characterize the decrease in IQ score associated with increased blood-lead concentration. The three longitudinal and four cross-sectional studies included in the meta-analysis are summarized in Table 4-4. Each study estimated the effect that blood-lead concentration has on full-scale IQ score in primary school age children. Estimates of the IQ score decrease from three of the studies were not statistically different from zero. Additional details are provided in Appendix D2, Tables D2-1 and D2-2. A summary of the Schwartz (1994) article and a comparison of the results to those reported in similar papers (Schwartz, 1993; Pocock, et al, 1994) are also presented in Appendix D2.

The seven studies used linear or log-linear regression models to model the relationship between IQ scores and childhood blood-lead levels, along with other potentially important covariates. A log-linear regression model is a regression model fitted to the logarithm of the independent variables; in this application the independent variable is blood-lead concentration and the dependent variable is IQ score. Three of the studies included in the meta-analysis employed log-linear models, while the four remaining studies employed linear models. Schwartz conducted a threshold analysis, to determine whether there is a level, below which a relationship between blood-lead concentration and IQ score is not apparent. On the contrary, Schwartz concluded that the slope appears to be steeper at lower blood-lead concentrations. This conclusion is consistent with the log-linear form of the regression model. Despite this, the linear relationship is assumed in this risk analysis. The assumption of a linear model reduces the likelihood of overestimating the number of children with low blood-lead concentrations at risk, or who may benefit from actions taken in response to the §403 standards.

Table 4-4. Summary Information for Studies Included in the Schwartz (1994) Meta-Analysis.

Study	Number of Children	Blood-Lead Concentration (µg/dL) Range	Blood-Lead Concentration (µg/dL) Mean (SD)	Estimated Effect[1] on IQ Score (SE)	Other Study Information
Hawk, et al (1986)	75	6.2 - 47.4	20.9 (9.7)	2.55 (1.5)	Cross-sectional study of children age 3-7 in Lenoir and New Hanover counties, NC; Linear regression model
Hatzakis, et al (1987)	509	7.4 - 63.9	23.7 (9.2)	2.66 (0.7)	Cross-sectional study of primary school age children in a lead smelter community (Lavrion, Greece); Linear regression model
Fulton, et al (1987)	501	3.3 - 34.0	11.5[2]	2.56 (0.9)	Cross-sectional study of primary school age children in Edinburgh, Scotland; Log-linear regression model
Yule, et al (1981)	166	7.0 - 33.0	13.5 (4.1)	5.6 (3.2)	Cross-sectional study of primary school age children in London, England; Log-linear regression model
Bellinger, et al (1992)	147	0-25[3]	6.5 (4.9)	5.8 (2.1)	Longitudinal study in Boston, MA; Blood lead at age 2; IQ measured at school age; Linear regression model
Dietrich, et al (1993)	231	na	15.2 (11.3)	1.3 (0.9)	Longitudinal study in Cincinnati, OH; Integrated blood lead up to age 3; IQ measured at school age; Linear regression model
Baghurst, et al (1992)	494	<12.2 - >28.2	20 (na)	3.33 (1.5)	Longitudinal study in Port Pirie, Australia; Integrated blood lead up to age 3; IQ measured at school age; Log-linear regression model

[1] Effect represents average declines in IQ points associated with an increase in blood-lead concentration from 10 µg/dL to 20 µg/dL.
[2] Geometric Mean was reported for this study.
[3] The exact range was not reported. The 90th percentile was 12.5 µg/dL and all children were below 25 µg/dL.

Based on the modeled relationships reported for each study, Schwartz concluded that a doubling of blood-lead concentration from 10 µg/dL to 20 µg/dL results in a loss of 2.57 IQ (SE = 0.41) points, on average. Therefore, a 1 µg/dL increase in blood-lead concentration results in a loss of 0.257 IQ points, on average. (For an individual child, a greater or lesser IQ point loss may be observed.) This relationship is most applicable for blood-lead concentrations between 10 and 20 µg/dL, because of the modeling assumptions made in the studies that used log-linear models. However, the relationship is applied over a much broader range of blood-lead concentrations in the risk analysis. Similar effects were observed in the studies that employed linear and log-linear regression models, as shown in Table 4-4. In addition, the blood-lead concentrations ranged from <6.2 µg/dL to 63.9 µg/dL in the studies that employed linear regression models.

The relationship between environmental-lead levels and IQ point loss is presented in Figure 4-3, utilizing predicted geometric mean blood-lead concentrations from the IEUBK model. For each curve, the soil- or dust-lead concentrations were varied over a range of values, while all other IEUBK model parameters were held fixed as described in Section 4.1. Then the predicted geometric mean blood-lead concentration from the IEUBK model was used to estimate the average IQ point loss using the relationship established above. For example, approximately 2.3 IQ points are expected to be lost as a result of exposure to a soil or floor dust-lead concentration of 1,000 ppm.

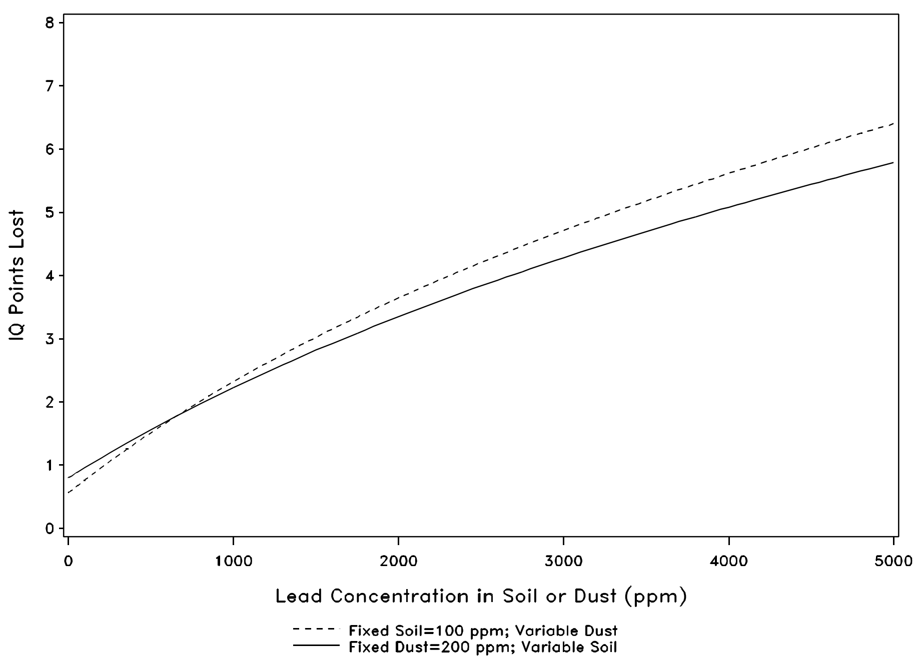

Figure 4-3. Estimated IQ Point Loss Due to Lead Exposure Plotted Against Concentration of Lead in Soil and Dust, Utilizing IEUBK Model Predictions to Relate Environmental Lead to Blood Lead

The relationship between IQ score decrement and blood-lead concentration is used in this risk analysis to estimate the average IQ point decrement for children exposed to various environmental-lead levels. Of interest is the extent to which average IQ score decrement is reduced upon promulgation of the §403 rule. Both the IEUBK and the empirical models are used to estimate the distribution of blood-lead concentrations of children exposed to a given set of environmental conditions observed in the HUD National Survey homes. The predicted blood-lead concentrations are multiplied by 0.257 to estimate the corresponding IQ point loss due to lead exposure for children exposed to these conditions. Then, the average IQ decrement and percentages of children with decrements of ≥ 1, ≥ 2, and ≥ 3 IQ points due to lead exposure are calculated.

4.4.2 Increased Incidence of IQ Scores Less Than 70

The increased incidence of IQ scores less than 70 resulting from lead exposure represents an increased likelihood of mental retardation resulting from lead exposure. An IQ of 70 is two standard deviations below the population mean IQ of 100 and can be used as an indicator of mental retardation. Children who are mildly mentally retarded require special education classes in school. Children who are severely mentally retarded may require life-long institutional care.

There are limited data available to estimate the increased likelihood of mental retardation resulting from lead exposure. Because of the lack of data, Wallsten and Whitfield (1986) used judgmental probability encoding methods to assess health risks due to lead exposure, particularly in the area of lower IQ scores. As part of this assessment, the increased percentage of children having IQ scores less than 70 was estimated for populations of children with elevated blood-lead levels. Judgmental probability encoding methods rely on expert judgement to estimate the effect of interest and are not applied when sufficient data are available to make the estimate.

In the Wallsten and Whitfield study, care was taken to select experts whose opinions spanned the range of respected opinion. The six experts who participated in the assessment of the relationship between IQ scores and blood-lead levels are listed in Table 4-5. These experts were asked to consider a hypothetical experiment in which a large number of children were randomly assigned at birth to either a control group, or one of six lead-exposure groups. Lead exposure was to remain fixed until the children reached age seven, at which time the Wechsler Intelligence Scale for Children – Revised (WISC-R) IQ test would be administered. Blood-lead levels were to be measured at age three. The lead exposure levels were such that at age three, members of each of the lead-exposure groups had blood-lead levels of 5, 15, 25, 35, 45, and 55 µg/dL. The experts were asked to estimate the mean and standard deviation of IQ scores in the control group. The experts also estimated the expected mean IQ differences between the control group and each exposure group. Each expert assumed that the IQ standard deviation in exposure groups was the same as that of the control group. This information was used to estimate the increased percentage, due to lead exposure, of children having IQ scores less than 70.

Table 4-5. Experts Who Participated in the Assessment of the Relationship Between IQ Scores and Blood-Lead Levels by Wallsten and Whitfield.

Expert	Affiliation
Kim Dietrich	University of Cincinnati
Claire Ernhart	Cleveland Metropolitan General Hospital
Herbert Needleman	University of Pittsburgh
Michael Rutter	Institute of Psychiatry, London, UK
Gerhard Winneke	University of Dusseldorf, Dusseldorf, West Germany
William Yule	Institute of Psychiatry, London, UK

If the expert thought it necessary, separate judgements were made according to socioeconomic status (SES). For this purpose, low SES was defined as children living in households with incomes at, or below, the fifteenth percentile; and high SES was defined as children living in households with incomes above the fifteenth percentile. Five of the six experts chose to make separate judgements based on socioeconomic status.

At blood-lead levels ranging from 2.5 to 27.5 µg/dL, the distribution of increased percentage of children having IQ scores less than 70 was reported by Wallsten and Whitfield, for each expert and SES (low and high). These distributions were combined by calculating the weighted average of the low SES median (15% of weight) and high SES median (85%). The increased percentage of children having IQ scores less than 70, due to lead exposure, was estimated from the weighted average of the medians as a piecewise linear function of blood-lead concentration. This function is reported in Table 4-6 and illustrated in Figure 4-4, over a range of blood-lead concentrations. For example, 0.6% = (-0.281 + 0.0432 x 20) of children with blood-lead concentrations of 20 µg/dL are expected to have IQ scores less than 70 due to lead exposure, above and beyond those whose IQ would naturally fall below that level. The relationship is extrapolated to include blood-lead concentrations below 2.5 and above 27.5 µg/dL.

Table 4-6. Piecewise Linear Function for Estimating the Judged Increased Percentage of Children Having IQ Scores Less Than 70 Due to Lead Exposure.

Range of Blood-Lead (PbB) Levels (µg/dL)	Function for Estimating Increased Percentage of Children Having IQ Scores less than 70 (IQ<70)
$0 < PbB \leq 5$	IQ<70 = 0.080 + 0.0036 PbB
$5 < PbB \leq 7.5$	IQ<70 = 0.022 + 0.0152 PbB
$7.5 < PbB \leq 10$	IQ<70 = -0.152 + 0.0384 PbB
$10 < PbB \leq 12.5$	IQ<70 = -0.084 + 0.0316 PbB
$12.5 < PbB \leq 15$	IQ<70 = 0.016 + 0.0236 PbB
$15 < PbB \leq 17.5$	IQ<70 = -0.260 + 0.0420 PbB
$17.5 < PbB \leq 20$	IQ<70 = -0.281 + 0.0432 PbB
$20 < PbB \leq 22.5$	IQ<70 = -0.145 + 0.0364 PbB
$22.5 < PbB \leq 25$	IQ<70 = -0.532 + 0.0536 PbB
$25 < PbB < \infty$	IQ<70 = -0.162 + 0.0388 PbB

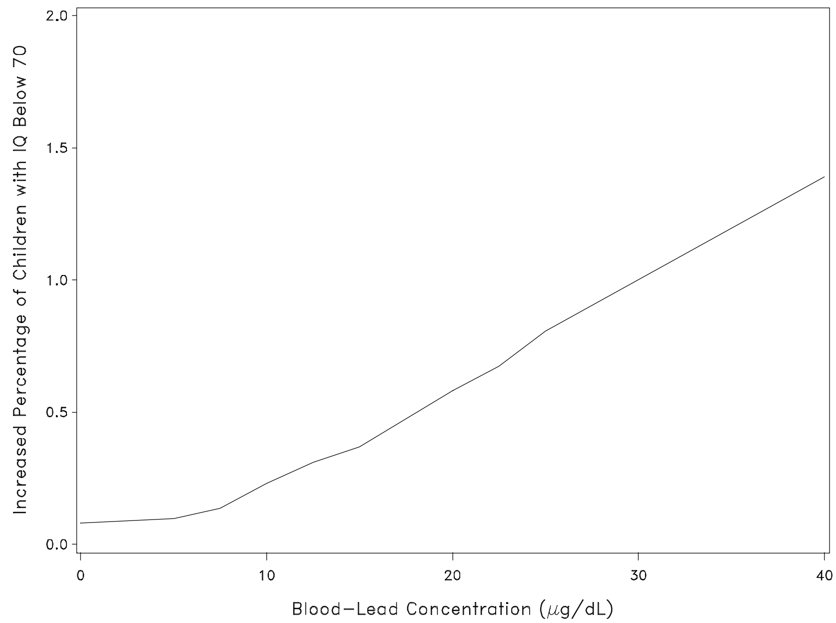

Figure 4-4. Judged Increase in Percentage of Children with IQ Below 70 Due to Lead Exposure, Plotted Against Blood-Lead Concentration

The relationship between environmental-lead levels and the increased percentage of children having IQ scores less than 70 is presented in Figure 4-5, utilizing IEUBK model predicted geometric mean blood-lead concentrations. For each curve, the soil- or dust-lead levels were varied over a range of values, while all other parameters were held fixed. The predicted geometric mean blood-lead concentration from the IEUBK model was used to estimate the increased percentage of children with IQ scores less than 70 due to lead exposure. For example, an additional 0.2% of children exposed to soil- or floor dust-lead concentrations of 1,000 ppm would be expected to have IQ scores less than 70 as a result of the exposure.

Figure 4-6 illustrates the relationships between geometric mean blood-lead concentration and the predicted percentage of children with a blood-lead concentration greater than or equal to 10 and 20 µg/dL, over a range of geometric mean blood-lead levels. This relationship was computed assuming a geometric standard deviation of 1.6 µg/dL and that blood-lead concentrations have a log-normal distribution. The same assumptions are applied in the risk characterization (Chapter 5). The relationships between lead concentrations in soil and dust and the incidence of blood-lead levels greater than or equal to 10 and 20 µg/dL are illustrated in Figure 4-7, utilizing geometric mean blood-lead concentrations as predicted by the IEUBK model. For each curve, the soil- or dust-lead concentrations were varied over a range of values, while all other model parameters were held fixed. The IEUBK model predicted geometric mean blood-lead concentration and the geometric standard deviation of 1.6 were used to calculate the percentage of children exposed to these environmental conditions who would have blood-lead concentrations greater than or equal to 10 and 20 µg/dL. As can be seen in Figure 4-7, approximately 40% of children exposed to soil or dust-lead concentrations of 1,000 ppm are

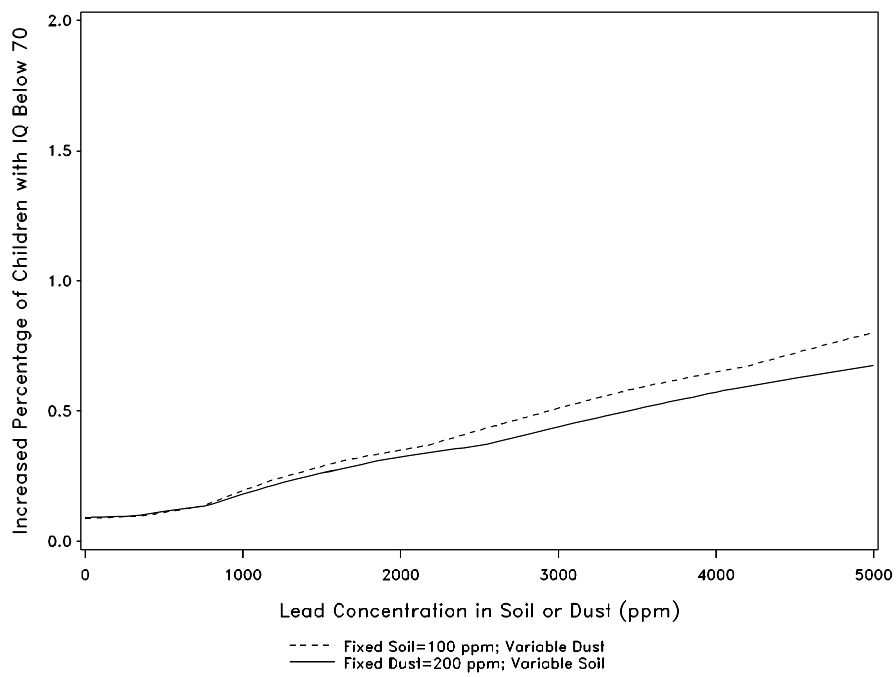

Figure 4-5. Increase in Percentage of Children with IQ Below 70 Due to Lead Exposure Plotted Against Concentration of Lead in Soil and Dust, Utilizing IEUBK Model Predictions to Relate Environmental Lead to Blood Lead.

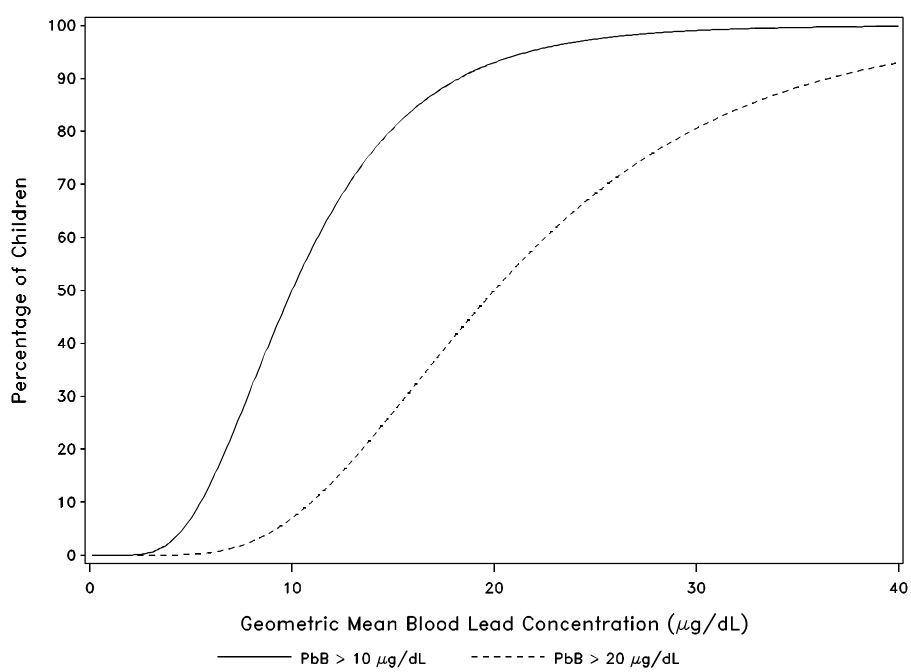

Figure 4-6. Percentage of Children with Blood-Lead Concentration ≥10 and 20 μg/dL Due to Lead Exposure Plotted Against Geometric Mean Blood-Lead Concentration, Assuming a GSD of 1.6.

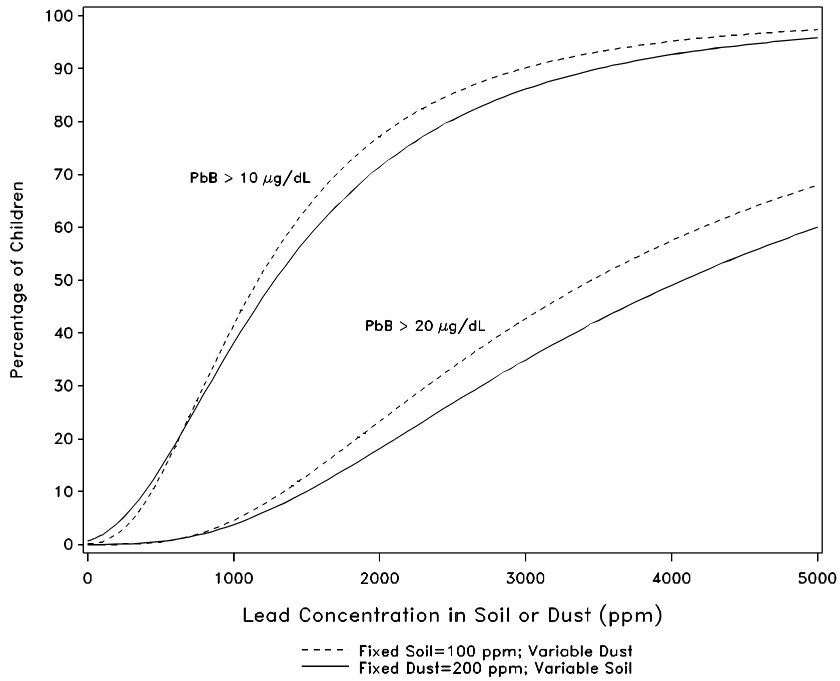

Figure 4-7. Percentage of Children with Blood-Lead Concentration ≥10 and 20 µg/dL Due to Lead Exposure Plotted Against Concentration of Lead in Soil and Dust, Utilizing IEUBK Model Predictions to Relate Environmental Lead to Blood Lead.

expected to have blood-lead concentrations of at least 10 µg/dL, whereas fewer than 5% of these children are likely to have blood-lead concentrations exceeding 20 µg/dL.

4.5 DOSE RESPONSE CHARACTERIZATION

This chapter summarized the approach taken to establish the relationship between exposures to lead in dust, soil, and paint and childhood blood-lead concentration and health effect endpoints. This relationship is used in the risk characterization (Chapter 5) to describe the risk to children aged 1-2 years under present environmental conditions. The relationship is also used in the risk management analysis (Chapter 6) to estimate the risk to children aged 1-2 years under candidate §403 standards. Establishing this relationship is very problematic, because only limited data exist relating specific health outcomes directly to environmental-lead levels.
Most environmental lead studies relate measures of residential lead exposure to measures of body lead burden, rather than directly to health effects. In addition, most studies of health effects of lead exposure relate specific health outcomes to measures of body lead burden, rather than directly to environmental-lead levels. Therefore, it is necessary to establish the relationship between environmental-lead levels and health outcomes for this risk analysis in two steps. First, blood-lead concentrations are estimated based on environmental-lead levels via quantitative models. Then, incidence of elevated blood-lead concentrations and health effect risks are estimated from those blood-lead concentrations.

Relationship Between Environmental Lead and Blood Lead

Two models are applied to relate environmental-lead levels to blood-lead concentrations, the IEUBK model and the empirical model. The IEUBK model takes user inputs for exposure and uptake through a biokinetic process of distributing the lead to key tissues to predict a distribution of blood-lead concentrations in children exposed to a specific combination of environmental conditions. The precursor to the biokinetic part of the IEUBK model was developed in 1985 as a tool for setting air lead standards and that version of the model was peer reviewed and found acceptable by EPA's Science Advisory Board. The most current version of the IEUBK model is used in this risk analysis. Although the IEUBK model was developed for point source applications, it is being used in this risk analysis to predict a national distribution of blood-lead concentrations.

The empirical model is a log-linear regression model, developed using data from the Rochester Lead-in-Dust Study to estimate the relationship between blood-lead concentrations in young children and observed lead levels in their primary residence. This model was developed specifically for this risk analysis and has not yet undergone peer review.

While the two models provide useful alternative views of the relationship between environmental and blood lead, neither model is optimal for this risk analysis. For example, the IEUBK model utilizes dust-lead concentrations, while the §403 standards for dust will be defined in terms of dust-lead loadings. Furthermore, the IEUBK model does not include a direct mechanism for the contribution of lead-based paint to childhood blood-lead levels (i.e., pica for paint). Thus estimated blood-lead concentrations are adjusted in homes with damaged lead-based paint to reflect this exposure pathway. This adjustment for paint pica has not undergone peer review. Although the empirical model was developed specifically for this risk analysis, different sampling methods were used in the Rochester study, upon which the empirical model is based, and the HUD National Survey, which is used for predicting the national distribution of blood-lead concentrations. Despite these shortcomings, the use of two different modeling approaches provides a more robust analysis for the risk analysis than either approach alone.

Relationship Between Blood Lead and Health Endpoints

Both the IEUBK model and the empirical model are used to predict the national distribution of blood-lead concentrations of young children. Incidence of elevated blood-lead concentration endpoints are calculated from the predicted national distributions, with no further modeling steps required.

While the existence of a relationship between decreased IQ scores and increased blood-lead concentrations is generally accepted in the scientific community, the quantification of the relationship is more problematic. Several estimates of the relationship between IQ scores and blood-lead concentrations were considered. The most central and most widely accepted of these is utilized in the risk assessment to calculate the average IQ decrement and the numbers of children with IQ decrement of ≥ 1, ≥ 2, and ≥ 3 points due to lead exposure. Two additional estimates are utilized in the sensitivity analysis (Section 5.4.2) to determine the extent to which the uncertainty in this parameter affects the risk characterization.

The relationship between blood-lead concentrations and the remaining health endpoint, the increased incidence of IQ scores below 70 due to lead exposure, was estimated using a piecewise linear function based on the distributions of IQ scores estimated by 6 experts who participated in a 1985 study that utilized judgmental probability encoding methods. Because of the lack of data to estimate this relationship, the use of expert judgement was unavoidable.

Impact on Risk Characterization

For the risk characterization, levels of lead in dust, soil, and paint from the HUD National Survey are provided as inputs to each of the models described above for the prediction of the national distribution of blood-lead concentrations for children aged 1-2 years. Although the HUD National Survey is the most comprehensive survey of residential-lead levels available, the application of these modeling tools to the HUD National Survey data is a limitation.

An important assumption in risk assessment is that the soil- and dust-lead concentrations represent a child's actual lead exposure. For site-specific risk assessments, children's activity patterns are used to guide the selection of sampling locations. It is uncertain whether the HUD National Survey data represent typical childhood lead exposure levels.

The empirical model was developed from environmental lead measures from a single study in one city and is being applied to nationally representative data. The empirical model predictor variables are similar, by design, to those available in the HUD National Survey. However, differences in sampling protocols exist between the studies, resulting in important differences in variables used to develop the empirical model and the HUD National Survey variables used for prediction. Most important is that dust samples were collected using different techniques in the two studies. In addition, dripline soil samples were used in developing the empirical model, while both dripline and remote area soil samples from the HUD National Survey are used for prediction. Information on interior paint condition and pica tendency was used in developing the empirical model, but both interior and exterior paint condition are used for prediction. Also, because no information on pica tendency was available for HUD National Survey homes, the proportion of children who exhibit pica for paint was estimated and this proportion was applied in all homes with damaged lead-based paint. To address these differences, the empirical model includes an adjustment for measurement error that takes into account both systematic differences and differences in error structures between the Rochester and HUD National Survey studies.

Impact on Risk Management Analysis

The IEUBK and empirical models are used in the risk management analysis to predict changes in blood-lead concentrations associated with reductions in environmental lead. In addition to the concerns described above, use of these tools in a post-§403 environment required developing relationships between different dust-lead measurements. These relationships are used only in the risk management analysis. Use of these relationships to convert lead levels from one sampling method to another is a weak link in the risk management analysis.

The following conversions are applied in the analysis of example options for risk management:

1. from pre-intervention blue nozzle (BN) vacuum lead loadings to pre-intervention wipe lead loadings, to determine whether an intervention is required;

2. from post-intervention wipe lead loadings to BN vacuum lead loadings, for input to the empirical model;

A great deal of uncertainty is associated with these conversions. (1) There was very little data upon which to base the conversions. The scarcity of data results in highly variable parameter estimates and makes it likely that individual data points may influence the analysis. The method for developing the conversion equations was designed to minimize the effect of influential observations and to account for differing variability across studies. (2) The range of data used to develop the conversions does not span the range of the HUD National Survey data. Thus, extrapolation is required to convert the lower portion of HUD National Survey BN lead loadings to wipe lead loadings. Fortunately, the affected homes are not expected to exceed any realistic set of standards. (3) There is considerable variability inherent in wipe lead loading measurements. The sensitivity analysis in Section 6.4 considers the effect of the uncertainty associated with these standards on the evaluation of risk management options.