Health Risk and Exposure Analyses in Support of the Review of the National Ambient Air Quality Standards for Lead

Update on Two Aspects of the Analytical Approach

Table of Contents

1. Introduction

The *Integrated Review Plan for the National Ambient Air Quality Standards for Lead. Volume 3: Planning Document for Quantitative Exposure/Risk Analyses* (IRP V3; U.S. EPA, 2023) described key considerations and initial plans for the health risk and exposure assessment (REA) for the current review of the national ambient air quality standards (NAAQS) for lead (Pb), as informed by the available evidence in this review.[1] The IRP V3 was released for public comment in May 2023, and was the subject of a consultation with the Clean Air Scientific Advisory Committee (CASAC) Lead Review Panel in June 2023.

Since the June 2023 consultation, staff in the Office of Air Quality Planning and Standards (OAQPS) of the U.S. Environmental Protection Agency (EPA) have been working to finalize the REA plan and associated methods, and to perform the assessment. In so doing, work has focused, in particular, on two areas: (a) the approach for the case studies and (b) the modeling approach for linking ambient air Pb to indoor residential dust Pb to estimate Pb concentrations in house dust arising from Pb in ambient air. Work in these areas has included expansion of both approaches, with accompanying preliminary evaluative work, particularly on the dust Pb modeling. This document describes our work in these areas.

With the Panel members' consultative advice we will move to implementation of the quantitative analyses. The complete REA will be summarized and its findings considered in the draft Policy Assessment document, with complete details presented in associated appendices to the draft PA. The draft PA, when completed, will be made available for public comment and will be the subject of review by the CASAC Lead Review Panel at a public meeting in 2025.

2. Overview of Analytical Approach

The REA analytical approach includes a focus on neurodevelopmental effects in young children and the use of the Integrated Exposure and Uptake Biokinetic (IEUBK, version 2) model to support exposure and blood Pb modeling (Figure 1). A key element of the assessment is the air-to-dust Pb modeling as this is a critical pathway for exposure of children to air-related Pb. Interindividual variability in blood Pb levels at the population level will be characterized through application of a geometric standard deviation (GSD) to the central tendency IEUBK output. The risk characterization will focus on Pb attributable IQ decrement and will utilize blood Pb concentration-IQ decrement response functions from epidemiologic studies of child study groups with blood Pb levels closest to those for children today exposed to air-related Pb.[2]

The uncertainty characterization planned for the assessment will utilize a largely qualitative approach adapted from the World Health Organization (WHO) approach for

[1] The IRP, Volume 3 (REA Planning Document), and other planning documents in this review, including the IRP, Volume 1 (Background Document) are available on the EPA webpage: https://www.epa.gov/naaqs/lead-pb-standards-planning-documents-current-review.

[2] As described in more detail on the IRP V3, studies by Canfield et al. (2003) and Bellinger and Needleman (2003) are the sources for these functions.

characterizing uncertainty in exposure assessment (WHO, 2008), accompanied by several quantitative sensitivity analyses.[3] The intention for this characterization is to identify and compare the relative impact that important sources of uncertainty may have on the exposure and risk estimates. Key aspects will include sensitivity analyses focused on key modeling elements and consideration of extent of extrapolation outside of associated supporting data (e.g., in estimating blood Pb levels with IEUBK and IQ decrements for blood Pb levels below those in studies providing the concentration-response functions).

The REA plan includes a case study approach, with the analytical approach applied to several case studies. These include a generalized local (hypothetical) case study (as described in the IRP V3), as well as two location-specific case studies: (1) a large urban area that includes a neighborhood with the highest air concentrations, influenced by a significant emissions source, and (2) the near-source neighborhood, alone.

The analytical approach is being updated from that used in the 2007 REA to support the 2008 Pb NAAQS review. Areas being updated include the approach for estimating ambient air Pb concentrations, the modeling approach linking ambient air Pb concentrations to indoor residential dust Pb concentrations, input parameters for the IEUBK modeling including the GSD used to model population-level exposure and the concentration-response functions used to estimate IQ decrements in young children. Figure 1 indicates in orange the areas updated since the 2007 assessment.

[3] This approach is routinely applied in health REAs for NAAQS reviews (e.g., U.S. EPA, 2020).

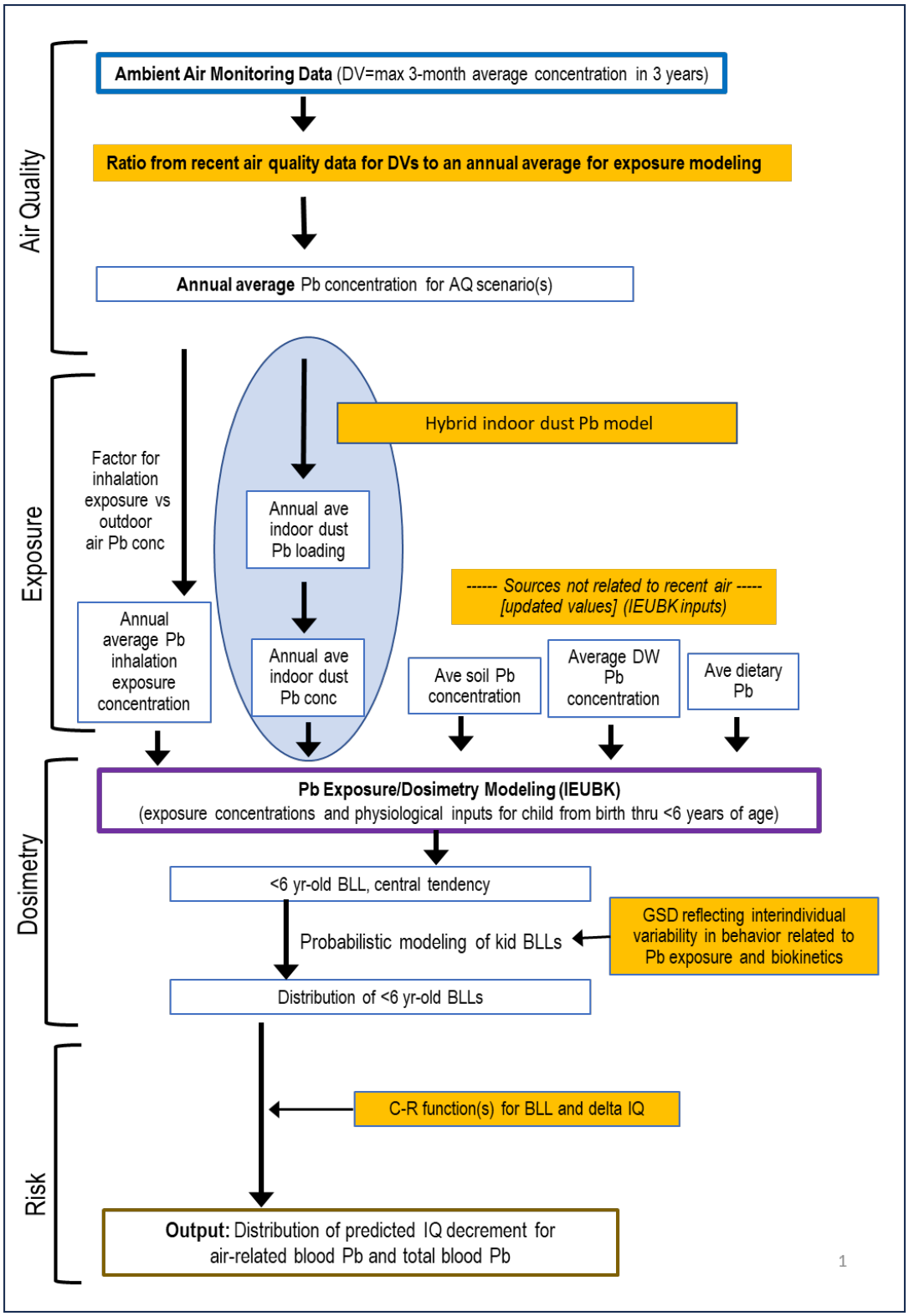

Figure 1. Analytical approach for the Pb NAAQS REA.

The subsequent two sections (sections 3 and 4, with associated subsections) focus on the two topics of case study design and Pb dust modeling. Full details will be presented in the REA. A brief summary of each is provided below:

- **Case study design**: In addition to the *generalized local case study*, we are including an assessment of exposure and risk for a larger urban residential population based on the Minneapolis-St. Paul metropolitan area. Inclusion of a location-specific urban area provides estimates of the distribution of exposure and risk estimates for a population of children in a real-world urban context. The Minneapolis-St. Paul urban area was selected because of its coverage by neighborhood-level monitors and because it includes a subarea impacted by a relatively large point source (Pb emissions from a secondary Pb smelter) that has the highest concentrations in the larger area. To more fully explore the nature of exposure and risk gradients across the study populations, we include two case studies based on Minneapolis-St. Paul including (a) *near-source neighborhood* focusing in on the residential area nearer to the secondary Pb smelter and (b) *full study area* of the larger urban expanse that includes both the neighborhood impacted by emissions from the secondary Pb smelter and the surrounding areas experiencing more typical (lower) urban area ambient air Pb concentrations.
- **Pb dust modeling**: The process of linking ambient air Pb to indoor residential dust Pb typically involves two steps: (a) modeling the loading of indoor dust Pb by ambient air Pb and (b) translating those indoor dust Pb loading estimates to equivalent dust Pb concentrations. Both empirical and mechanistic models can be used to complete each of these steps. In the 2007 risk assessment for the 2008 Pb NAAQS review, we developed a hybrid model, which combined a mechanistic approach for linking ambient air Pb to indoor dust loading with an empirical (regression) approach for translating dust Pb loading estimates to dust Pb concentrations. In this review, in addition to updating the hybrid model for use in this REA, including through updates to inputs to the mechanistic portion, we have also identified more recently available analyses relevant to the second step and developed a new empirical approach for the first step of relating ambient air Pb concentrations to indoor dust Pb loading. This empirical approach uses time trends over the past two decades (during which both ambient air Pb and indoor dust Pb have decreased) to derive a quantitative relationship directly between indoor dust Pb loading and ambient air Pb concentration. We intend to utilize the newly available methods in model evaluation and characterization of uncertainty in dust Pb estimates.

3. Case Studies – Spatial Definition with Assignment of Air Concentrations and Populations

In addition to the generalized local case study (described in IRP V3), the REA will also include two location-specific case studies based on the Minneapolis-St. Paul urban area. These location-specific case studies will provide estimates of the distribution of exposure and risk estimates in a real-world context. In all three case studies, the air quality scenarios to be assessed include air quality conditions just meeting the current Pb NAAQS and conditions just meeting potential alternate standards. The annual average air concentrations for these scenarios are inputs to the indoor dust Pb modeling approach, and the IEUBK blood Pb model.

An important detail in the design of a case study is the approach used to characterize ambient air Pb levels across that study area. Depending on the nature of potential sources of ambient air Pb, source-modeling and/or ambient monitoring data can be used to characterize ambient air Pb levels in the urban-residential context. Both approaches were used in characterizing ambient air Pb for the Minneapolis-St. Paul urban area.

In describing the approaches for each of the case studies below, we begin by describing what each is intended to represent (in terms of the nature of the ambient air-related exposure profile potentially impacting the child population involved). We then provide details regarding the method used to characterize ambient air Pb and relate it to children located within the study area. We also discuss the approach used to adjust ambient air Pb levels in simulating the attainment of alternative standard levels for a given study area.

3.1. Full Study Area

The Minneapolis-St. Paul urban area was selected as the basis for the two case studies based on how it addresses two important criteria: (a) the presence of an area with recent ambient air Pb concentrations near the current standard, and (b) a relatively large number of neighborhood-scale and near-source monitors to use in characterizing ambient air concentrations across the areas. For example, (a) this area includes a relatively large number of neighborhood-scale monitors allowing the characterization of ambient air concentrations across parts of the study area not impacted by point source emissions, and (b) it includes a subarea with air concentrations impacted by Pb emissions from a larger point source (secondary Pb smelter). It also includes a few much smaller Pb emissions sources contributing to much lower concentrations than those associated with the secondary Pb smelter.[4]

The outer perimeter of this study area was delineated by the outermost monitor locations (see Figure 2). This study area includes both larger residential areas without substantial point sources (covered by ambient neighborhood-scale monitoring data), and also a smaller residential

[4] The smaller Pb emissions sources include metals recycling facilities (active and closed) and airports. Although the ambient air Pb concentrations near these smaller sources are substantially lower than near the secondary Pb smelter, these sources were evaluated with regard to the potential for appreciable contributions to ambient air Pb concentrations in adjacent residential areas.

area in the vicinity of the secondary Pb smelter with estimated ambient air Pb concentrations higher than the surrounding urban area. The near-source neighborhood is both included within this full study area and also comprises its own study area (see section 3.2 below).

Ambient air Pb concentrations in parts of the full study area beyond the near-source neighborhood were characterized using the neighborhood monitor locations to partition the area into Thiessen polygons[5] (see Figure 3). While there is variation in ambient air Pb concentrations from polygon to polygon, these concentrations are substantially lower than those associated with the near-source neighborhood (surrounding the secondary Pb smelter). Consequently, it is anticipated that exposure and risk for children located within this case study will be substantially lower than those associated with the near-source neighborhood and the generalized local case studies. As such, the full study area will provide perspective on the range of exposures and risks associated with a larger urban area where most of the air Pb levels are expected to be substantially below the current NAAQS.

[5] Each polygon contains only a single monitor location and any location within each polygon is closer to its associated monitor location than to any monitor location.

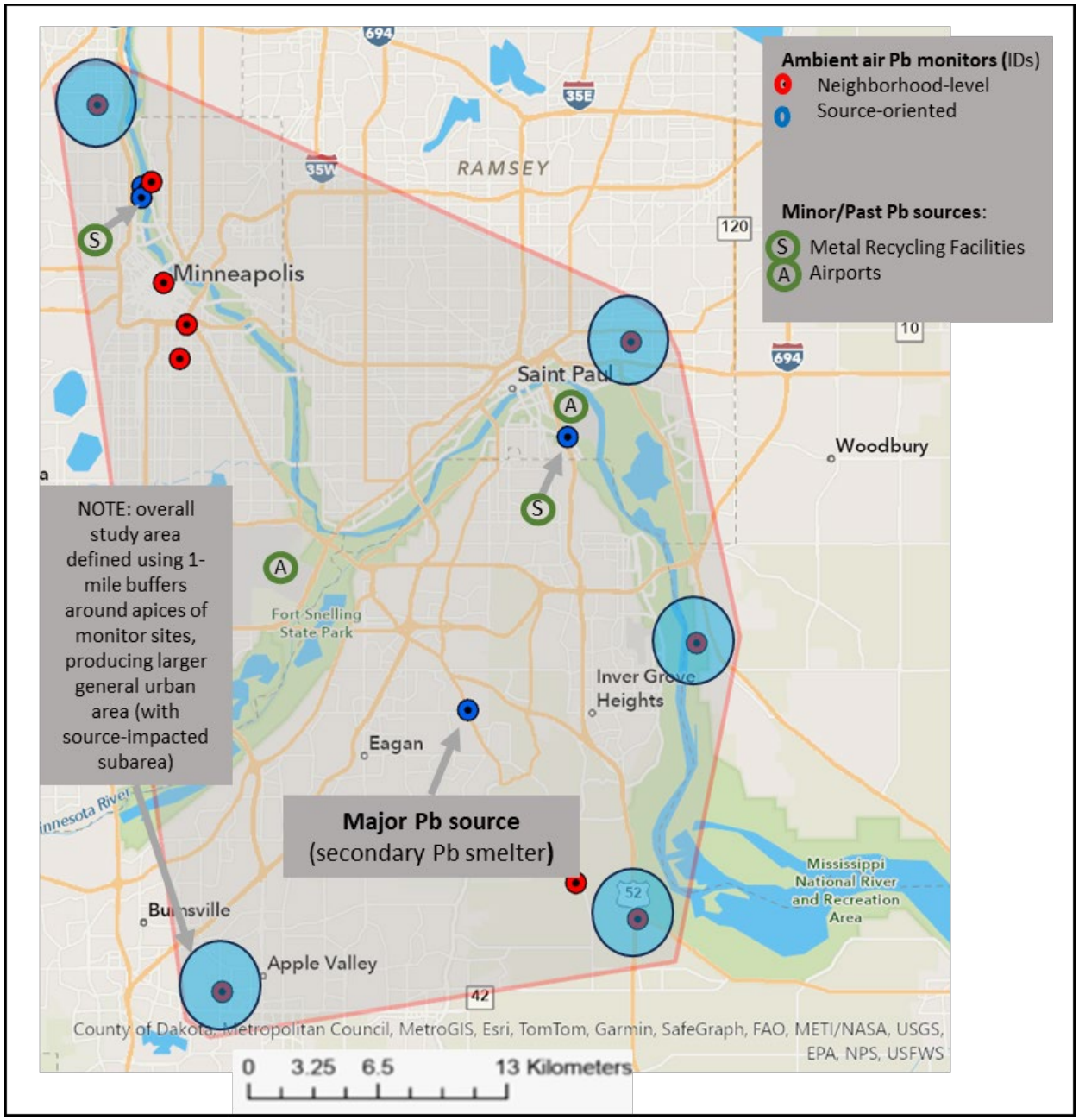

Figure 2. Full Minneapolis-St. Paul study area, with locations of monitors and Pb sources.

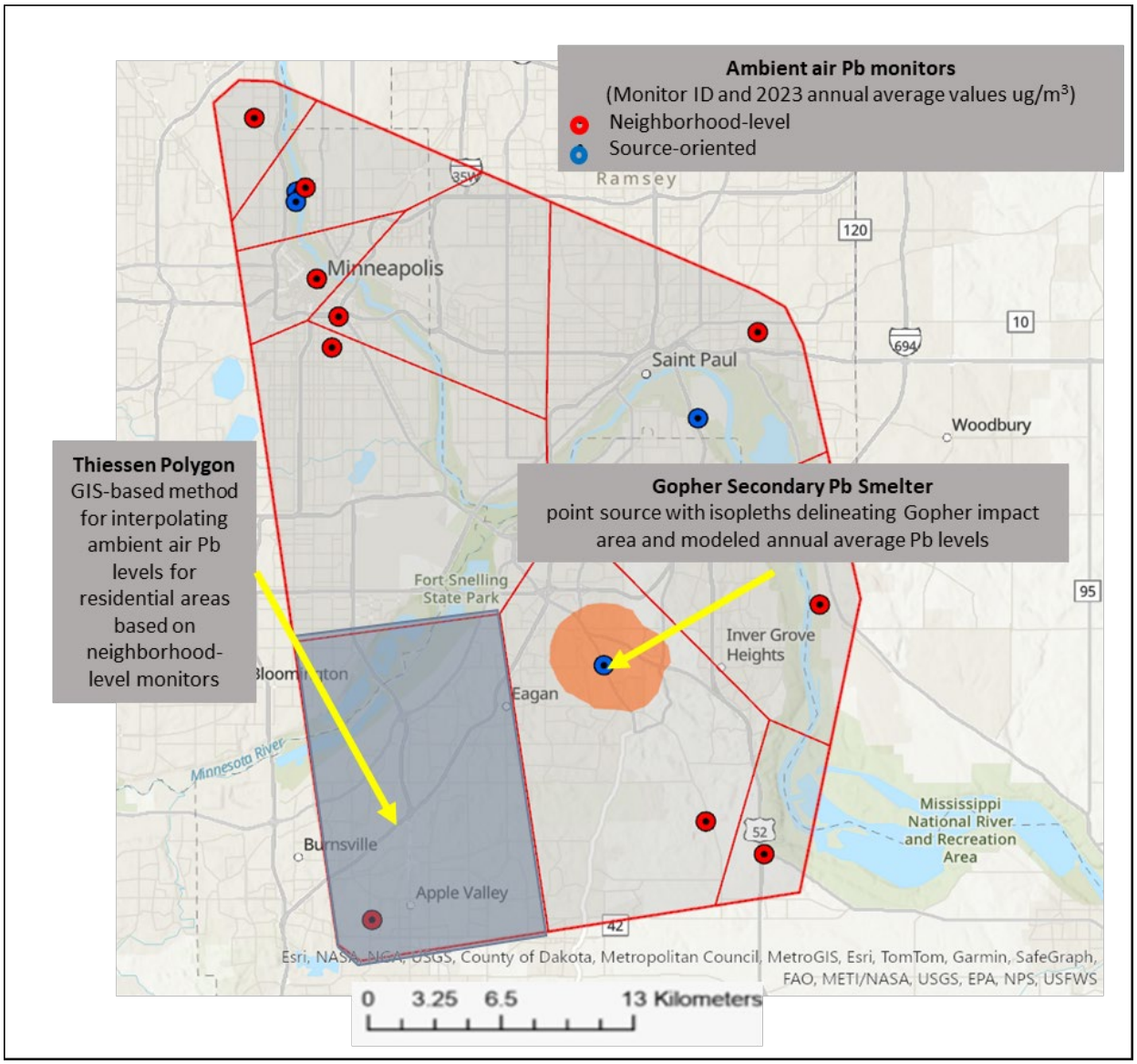

Figure 3. Full Minneapolis-St. Paul-based study area with Thiessen polygons and secondary Pb smelter impact area.

3.2. Near-Source Neighborhood

This near-source neighborhood case study is comprised of the small portion of the larger Minneapolis-St. Paul urban study area experiencing elevated ambient air Pb concentrations due to emissions from a secondary Pb smelter. The monitor at this source records the highest concentrations in the larger Minneapolis-St. Paul area. The design value for this monitor in recent years is nearly equal to the level of the current Pb NAAQS.[6]

This near-source neighborhood study area allows us to consider exposure and risk for a group of children exposed to elevated ambient air Pb associated with a specific source of Pb emissions. Ambient air Pb concentrations within this study area are characterized using both: (a) source-oriented monitoring data associated with the facility and (b) AERMOD[7] modeling results for the facility originally completed as part of EPA's national air toxics assessment work.[8] The modeling estimates for Pb concentrations were used to derive isopleths surrounding the facility which were compared with the source-oriented monitor data to evaluate reasonableness of the modeled results (there was close agreement). Using the AERMOD-based isopleths, the outer perimeter of this study area was delineated as the point where model-predicted concentrations just exceeded the concentrations of the surrounding Thiessen polygon. Within this near-source study area, the isopleths described the spatially refined gradient of ambient air Pb concentrations (see Figure 4). In this way, children residing in this near-source neighborhood were assigned varying ambient air concentrations that decreased with distance from the source consistent with the AERMOD results.

The near-source area is the focus for development of the different air quality scenarios. This is because this area is the portion of the full study area with the highest air Pb concentrations and because the outer boundary of this near-source study area specifies the extent of the source influence on air concentrations in the larger area. The procedure for simulating air quality just meeting the existing and potential alternative standards will utilize the AERMOD-estimated gradient in ambient air Pb concentrations across the isopleths to scale concentrations within the isopleths such that the inner-most ring has concentrations consistent with just meeting

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[6] A design value is a statistic that summarizes the ambient air concentrations for a given area in terms of the indicator, averaging time, and form of the standard. Design values can be compared to the level of the standard and are typically used to designate areas as meeting or not meeting the standard and assess progress towards meeting the NAAQS. Design values for Pb are the maximum 3-month average concentrations during a 3-year period based on validated monitoring data and calculated in accordance with 40 CFR 50, Appendix R (*https://www.ecfr.gov/current/title-40/chapter-I/subchapter-C/part-50/appendix-Appendix%20R%20to%20Part%2050*)

[7] The acronym, AERMOD, stands for AMS/EPA Regulatory Model.

[8] Annual average ambient air Pb modeling results for the secondary Pb smelter (Gopher facility) were generated as part of EPA's Air Toxics Screening Assessment for 2018. AirToxScreen is the successor to the previous National Air Toxics Assessment (NATA) and was developed as a screening tool for air toxics for state, local and Tribal air agencies. Gridded air modeling results for the Gopher facility (generated using AERMOD) were used to derive isopleths covering the near-source neighborhood study area. Details related to AirToxScreen modeling can be found at: https://www.epa.gov/AirToxScreen. Emissions profiles for this facility can be found here: https://www.epa.gov/AirToxScreen/2018-airtoxscreen-assessment-results.

the potential standard of interest.[9] From the air quality data for each air quality scenario of interest, annual average concentrations are derived given the annual time step inputs to the IEUBK blood Pb model (and to the dust Pb modeling approach described in section 4 below).

Because ambient air Pb concentrations in the portion of the full study area outside of the near-source neighborhood have consistently demonstrated ambient air Pb concentration patterns not influenced by the main source (secondary Pb smelter), and well below the current NAAQS for more than a decade, the ambient air Pb concentrations assigned to the Thiessen polygons outside the near-source neighborhood are to be held constant across the air quality scenarios.

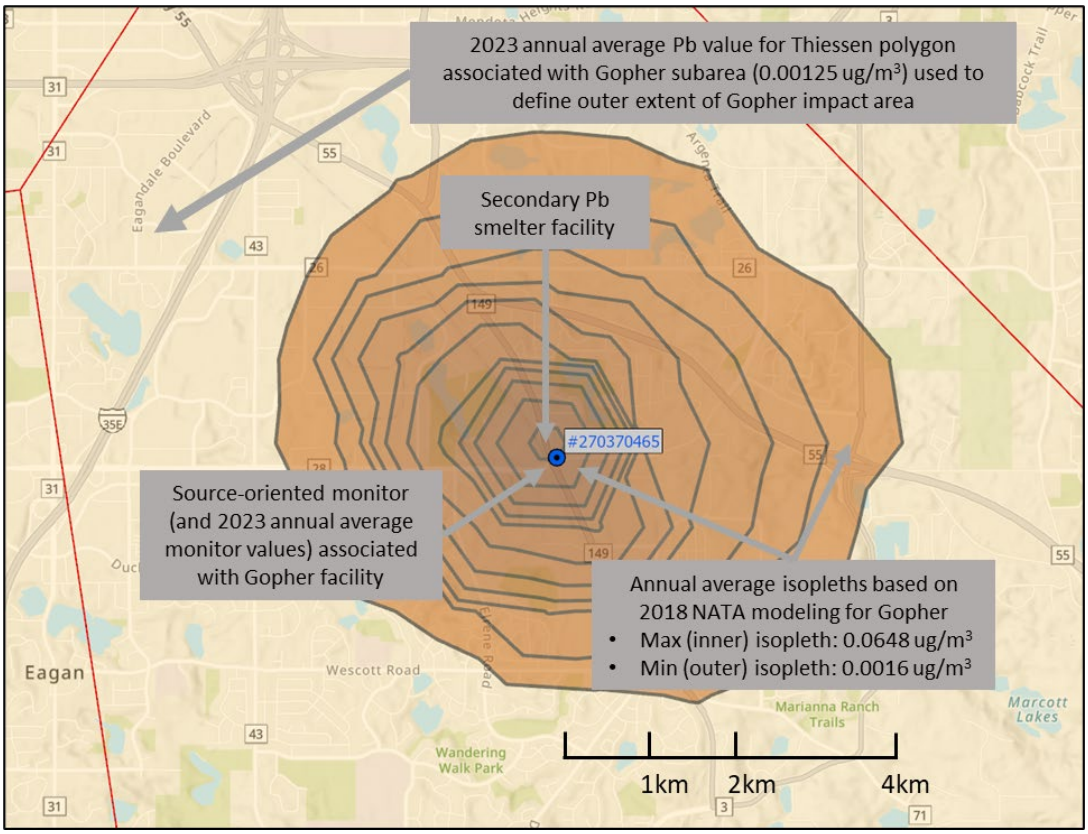

Figure 4. Near-source neighborhood study area surrounding secondary Pb smelter (indicated by darker color).

3.3. Generalized Local

The generalized local case study, described in the REA Planning Document (IRP V3), is a hypothetical neighborhood with a small group of children exposed to air Pb at the specific standard of interest. This means there is a uniform pattern of ambient air Pb concentrations across the hypothetical residential area. The uniform ambient air Pb concentrations associated with this study area are specified to be just equal to the particular standard level of interest.

[9] The design value for the regulatory monitor in this area was 0.14 µg/m³ in 2023 (see https://www.epa.gov/air-trends/air-quality-design-values).

4. Dust Pb Estimation Approaches

The incidental ingestion by young children of indoor residential dust loaded by ambient air Pb was an important exposure pathway in the 2007 REA and is expected to be an important policy-relevant exposure pathway in the REA for the current review. Consequently, in designing the REA, we focused on potential refinements and updates to the methods used in modeling this pathway. In this section, we begin by providing a conceptual overview of the exposure pathway, followed by a summary of the approaches used in modeling this pathway in the 2007 REA (those earlier methods formed the starting point for our work to update modeling for this pathway). After that, we present the update to the 2007 approach and also some additional methods based on development work over the past year.

Modeling of the relationship between outdoor ambient air Pb and indoor residential dust Pb involves the development of a dust Pb model (see Figure 5). Since the time step for inputs to the IEUBK model is one year, the dust Pb concentration input to the IEUBK model is an annual average dust Pb concentration. Accordingly, the approaches described here for estimating the concentration in dust Pb derived from ambient air utilize annual average air Pb concentrations as the inputs.

A key detail related to dust Pb modeling is the nature of the indoor dust Pb metrics involved, which include (a) dust Pb loading (i.e., micrograms [µg] of Pb per surface area of floor, typically square feet [ft^2]) and (b) dust Pb concentration (i.e., µg of Pb per gram [g] total dust on the floor). Measurement of indoor residential dust Pb through surveys can be done by either metric with the measurement method via either wipes or vacuuming. For loading studies, wipe-based measurements are the far more common method. Furthermore, mechanistic (compartmental) modeling of the relationship between ambient air Pb and indoor surface dust Pb typically involves an estimate of Pb loading (see below). However, blood Pb models, such as IEUBK, require the dust Pb input be in terms of dust concentration (mass Pb per mass dust) rather than surface loading; this is because the exposure modeling involves estimates of incidental ingestion of total dust (which contains Pb).

This need to translate from a dust Pb loading metric to dust Pb concentration necessitates that estimates of dust Pb loading, derived through either empirical or mechanistic modeling (see Figure 5), be converted into equivalent dust Pb concentrations prior to exposure and risk modeling. Further, the relationship between Pb loading in indoor dust (per surface area) and Pb concentration in total dust appears to be non-linear (U.S. EPA, 2007b, Appendix G, Attachment G-1) and continues to be challenging to characterize.

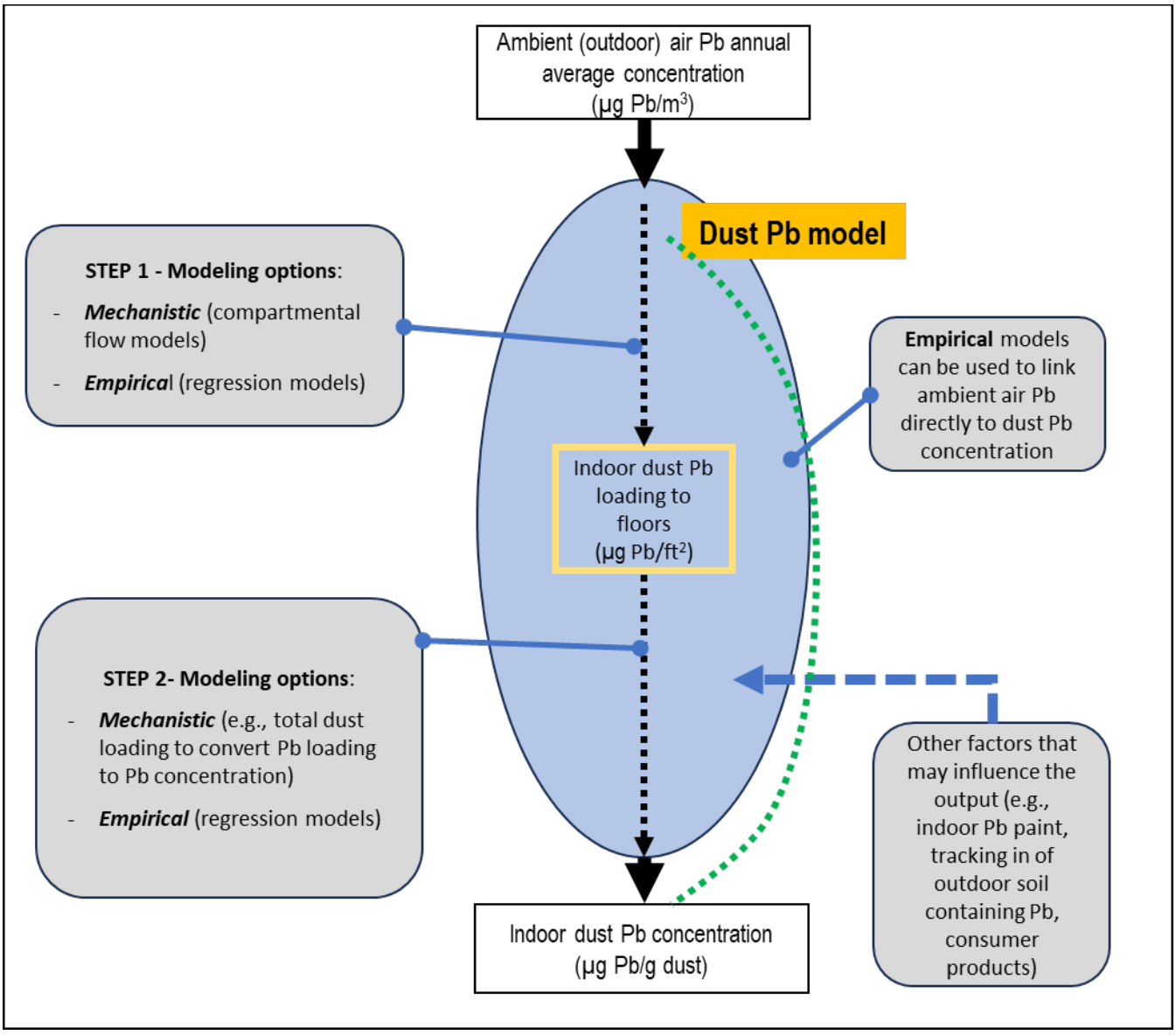

Figure 5. Conceptual analytical approach for dust Pb modeling.

In modeling the relationship between ambient air Pb and indoor dust Pb, empirical and/or mechanistic models have been employed as illustrated in Figure 5. Mechanistic models allow for the representation of key processes in the simulation of dust Pb loading and associated concentration (e.g., air exchange rates, penetration efficiency, cleaning frequency and efficiency, etc.), and for the resulting estimate to be specific to the outdoor ambient air Pb (and/or any other sources represented in the modeling). However, the complexity of these models, and the associated extensive parameterization, can contribute uncertainty to the estimates generated and make evaluation of the estimates' reasonableness challenging. By contrast, while empirical (regression) models have the advantage of utilizing real-world measurements (of ambient air Pb and indoor dust Pb) the potential for influence by factors other than those represented in the

regression models (or an imprecise representation) can also contribute significant uncertainty into the resulting estimates.

Additionally, while estimates of dust Pb in terms of concentration are the objective, studies analyzing data for dust Pb loading and dust Pb concentration (e.g., Bevington et al., 2021) indicate that this relationship is complex, potentially reflecting the relationship between Pb and total dust loading and challenges in the methods and protocols used to collect dust Pb loading and Pb concentration measurements as discussed later, as well as the sources contributing to Pb in indoor dust.

4.1. 2007 Approach – Focus on Hybrid Mechanistic-Regression Model

The 2007 REA relied on a Hybrid model that combined mechanistic and empirical approaches for modeling dust Pb (U.S. EPA, 2007a, section 3.1.4.1; U.S. EPA, 2007b, Appendix G, section G.3.2). This Hybrid 2-step approach combines a mechanistic (compartmental) model with an empirical model. In the first step, the mechanistic model links ambient air Pb concentrations (µg Pb/m^3 air) to dust Pb loading (µg Pb/ft^2 surface). In the second step, an empirical two-stage regression model is used to convert these dust Pb loading estimates to equivalent dust Pb concentrations (µg Pb/g dust). Each of these modeling elements is briefly summarized in the subsections below.

4.1.1. Step 1: Mechanistic Model

The first step of the Hybrid model is a mechanistic compartmental model that includes consideration of an array of processes that influence the buildup of Pb (transported from ambient [outdoor] air) in indoor dust. These processes include penetration, ventilation, indoor deposition, resuspension, filtration and cleaning activities. The model, once parameterized, was then solved for steady state, resulting in an analytical model of the form presented below (U.S. EPA, 2007b, Appendix G, section G.3.2.1). For our purposes, this model takes an annual average air Pb concentration (µg/m^3) and derives an annual average dust Pb loading estimate (µg/ft^2).

$$FLOOR\ LOADING = \frac{D \times AER \times P \times H}{CE \times CF \times (D + AER)} \times 0.09\ (m^2/ft^2) \times PbAIR$$

Where:
FLOOR LOADING = Pb loading on the floor (µg/ft^2)
D = deposition rate (h-1)
AER = air exchange rate (h-1)
P = penetration efficiency unitless)
H = ceiling height (meter [m])
CE = cleaning efficiency (unitless)
CF = cleaning frequency (cleanings/h)
PbAIR = concentration of Pb in the ambient air (µg/m^3)

Parameterization of the inputs is described in Appendix G, section G.3.2.2 of U.S. EPA (2007b). Based on the model conceptualization and inputs, the floor loadings generated with the

Hybrid model were assumed to be wipe-equivalent (and not vacuum equivalent). When inputs obtained from the literature were placed into the above model, a simplified function relating ambient air Pb ($\mu g/m^3$) to indoor dust Pb loading ($\mu g/ft^2$) was obtained:

$$\text{Dust Pb (wipe) loading } \mu g/ft^2 = 104.2 * \text{ air concentration } (\mu g/m^3)$$

$$\text{i.e., } PbWipe = 104.2\ (m^3/ft^2) \times PbAIR$$

4.1.2. Step 2: 2-Stage Empirical Model

To convert the estimates of dust Pb loading (wipe-equivalent) to dust Pb concentrations, the second step of the Hybrid model used a two-stage model comprised of two separate log-linear regression models. The first regression is used to convert the wipe-equivalent dust Pb loadings into vacuum-equivalent dust Pb loading. This conversion regression was developed by the EPA to support efforts to address potential lead-based contamination in the residential context and is based on three datasets (U.S. EPA, 1997). The second regression, which was developed as part of the 2007 REA supporting the 2008 Pb NAAQS review, converts the vacuum-equivalent dust Pb loadings to vacuum-equivalent concentrations (U.S. EPA, 2007b, Appendix G, Attachment G-1). This regression is based on dust Pb loading and concentration data collected as part of the HUD National Survey of Lead-Based Paint in Housing conducted between November 1989 and 1990 (U.S. EPA, 1995a). The regressions for both stages, and their underlying datasets are described below.

4.1.2.1. First-Stage – Wipe-Based Loading to Vacuum-Equivalent Loading

The regression model for the conversion of wipe-based Pb loading estimates into vacuum-equivalent Pb loading estimates was initially derived for purposes related to EPA's lead-based paint program (U.S. EPA, 1997). The final model was derived from three separate regression equations (U.S. EPA, 1997) for three different datasets: "CAP Pilot" (U.S. EPA, 1995b), "NCLSH/Westat" (Westat Inc., 1995) and "R&M Pilot" (U.S. EPA, 1996). The final equation and the three dataset-specific regression equations are presented in Table 1. A weighted average approach was used to obtain the final conversion equation (U.S. EPA, 1997, Appendix D). A linear model on a log scale was assumed to describe the relationship between wipe and vacuum measures within each study as well as across all studies.

Table 1. Regression equation for estimating vacuum-equivalent loading from wipe-equivalent loading, and underlying dataset-specific equations.

Combined Regression (for converting wipe-based loading to vacuum-based loading)	**$V=0.185*W^{0.931}$** Where, V=vacuum-based loading (µg/ft^2) W=wipe-based loading (µg/ft^2)		
	Dataset-Specific Regressions		
	CAP Pilot [A]	**NCLSH/Westat** [B]	**R&M Pilot** [C]
Dataset-Specific Log-Linear Regression	**$V=0.662*W^{0.708}$**	**$V=9.87*W^{0.034}$**	**$V=0.134*W^{0.993}$**
R^2 for log-linear regression model	**0.683**	**0.001**	**0.810**
N	**6**	**7**	**24**
Wipe dust Pb loading (µg/ft^2) geometric mean (range)	**51 (13.8-2498.5)**	**50.6 (33.5-91.4)**	**128 (6.3-13969)**
Vacuum dust Pb loading (µg/g) geometric mean (range)	**10.7 (1.9-149.1)**	**11.2 (5.4-20.9)**	**16.6 (1-2164)**

[A] Comprehensive Abatement Performance Pilot (U.S. EPA, 1995b)
[B] NCLSH Westat Blue Nozzle Study (Westat, 1995)
[C] Lead-Based Paint Abatement and Repair and Maintenance Study (U.S. EPA, 1996)
Source: U.S. EPA (1997), main document and Appendix D.

The final combined regression model was evaluated with regard to influential observations and residual analysis. Regarding the residual plots, variability was concluded to appear constant across predicted values (U.S. EPA, 1997). Confidence and prediction intervals were also derived for different magnitudes of wipe loading of interest at that time. For example, for the lowest wipe loading estimate considered (10 µg/ft^2), the predicted vacuum-equivalent loading estimate was 1.58 µg/ft^2 with a 95% confidence interval of 0.916 – 2.71 µg/ft^2.

In considering the final regression and the three underlying dataset-specific regression equations (and associated data), as summarized in Table 1, we note that the data from all three studies are generally above the Pb loading levels of interest in the current REA,[10] and the number of observations for each of the individual regressions are relatively small. However, the final conversion model parameters were derived as weighted averages of the parameters from the individual studies. For example, an estimate of the slope was obtained as the average of the slope estimates for the separate study-specific regressions, each weighted by the inverse of its estimated variance (as described in more detail in U.S. EPA [1997]). Thus, the NCLSH/Westat dataset, having the smallest range in data (for both variables), contributes little to the final regression relative to the other two datasets and, accordingly, the R^2 for the fit of the final

[10] For air quality meeting the current NAAQS, the 2007 REA Hybrid model produces a dust Pb loading (wipe-equivalent) estimate below 10 µg/ft^2 and a dust Pb concentration estimate below about 150 µg/g. The vast majority of dust Pb loading and concentration values in areas meeting the standard would be well below these.

regression would be a weighted combination of the R^2 values for the CAP Pilot and R&M Pilot and likely closer to the latter given its wider range and greater sample size. Taken together, these observations and those made in the evaluations at the time of the model's derivation (U.S. EPA, 1997) indicate a good model fit, although the data supporting the model derivation are for a range of dust Pb loading higher than the range of interest for the current REA.

4.1.2.2. Second-Stage Vacuum-Equivalent Loading to Vacuum-Equivalent Concentration

The dataset used as the basis for the second stage regression (for estimating vacuum-equivalent concentration from vacuum-equivalent loading) includes a wide range of loading and concentration values. The geometric mean of the vacuum-based loading measurements is 4.14 $\mu g/ft^2$ and the geometric mean of vacuum-based concentrations is 121.1 µg/g (U.S. EPA, 1995a). The regression model derived from this dataset has a model fit R^2 of 0.48 (EPA, 2007b, Appendix G). The geometric mean for these data reflect Pb levels in the range of interest for the current REA (e.g., below about 150 µg Pb/g dust) and the R^2 value indicates a reasonable fit for the regression model.

4.1.2.3. Overall Regression for Converting Wipe-based Loadings to Vacuum-Equivalent Concentrations

Regression models for each of the two stages described in the two prior sections were used in the 2007 REA and are further described in the associated documentation (EPA 2007a, Table 3-2). Aggregation of these two regressions results in the overall model for converting wipe-equivalent estimates of dust Pb loading to dust Pb concentrations (i.e., the 2-stage empirical model), which is summarized by the following formula:

$$PbDUST = EXP[4.92 + 0.52 \times \ln\left(0.185 \times (PbWIPE)^{0.931}\right)]$$

where:

PbDUST = indoor dust Pb concentration (µg/g)
PbWIPE = modeled (wipe-equivalent) loading of dust Pb ($\mu g/ft^2$)

4.1.3. Summary of Hybrid Model for Estimating Indoor Dust Concentration from Ambient Air Concentration

Combining the 2-stage empirical model, summarized in section 4.1.2.3 above, with the mechanistic model summarized in section 4.1.1 results in the overall Hybrid model used in the 2007 REA, which is summarized as follows:

$$PbDUST = EXP[4.92 + 0.52 \times \ln\left(0.185 \times (104.2 \times PbAIR)^{0.931}\right)]$$

where:

PbDUST = indoor dust Pb concentration ($\mu g/g^2$)
PbAIR = concentration of Pb in the ambient air ($\mu g/m^3$)

4.2. Current Approaches

The effort to update the dust Pb modeling approach for this REA involved considering both steps of the modeling approach: (a) the step for estimation of Pb loading and (b) the step for conversion of those dust Pb loading estimates to equivalent dust Pb concentrations. A complete dust Pb model requires an approach for addressing both steps – see Figure 5. We focused primarily on potential updates to the 2007 Hybrid model. We also reviewed more recent literature for studies and associated data that might provide new empirical methods for addressing either or both steps in the dust Pb modeling.

We begin by describing the 2024 update to 2007 method for linking ambient air Pb to indoor dust Pb loading (step 1 in section 4.2.1 below) and also information on another method for this step, and then describe methods for translating those loadings into equivalent dust Pb concentrations (step 2 in section 4.2.2). Both steps can be tackled using either mechanistic or empirical models. Section 4.2.3 describes the updated 2-step model for the REA, and additional approaches, each of which incorporates particular methods for addressing step 1 and step 2.

4.2.1. Step 1: Estimation of Pb Loading

Our effort to update methods for linking ambient air Pb to indoor dust Pb loading resulted both in an update to the Hybrid (mechanistic) approach used in the 2007 REA and the identification of an additional, new empirical method (the delta factor approach). Each of these is discussed in the following subsections.

4.2.1.1. Mechanistic (Updated)

As summarized in section 4.1.1 above, the Hybrid mechanistic model is an equation for predicting indoor dust Pb loading (wipe-equivalent) from ambient air Pb concentrations (under steady-state conditions). Our update to the model used in the 2007 REA involved extensive literature searches to identify new published literature that would support updates to the parameterization of the model. That literature search resulted in updates to a subset of the input parameters used in the model including: *air exchange rate, deposition rate (for particles indoors), cleaning efficiency* and *cleaning frequency* (details will be presented in REA). The combined net impact of these refinements to the model inputs on the final equation is shown below:

2007 REA parameterization:

Dust Pb (wipe) loading $\mu g/ft^2$ = 104.2 * air concentration ($\mu g/m^3$)

$$i.e.,\ PbWipe = 104.2\ (m^3/ft^2) \times PbAIR$$

Updated 2024 parameterization:

Dust Pb (wipe) loading $\mu g/ft^2$ = 75.0 * air concentration ($\mu g/m^3$)

$$i.e.,\ PbWipe = 75.0\ (m^3/ft^2) \times PbAIR$$

4.2.1.2. Empirical (Delta Factor)

The empirical approach relies on dust Pb survey data (measurements) from two American Healthy Homes Surveys (AHHS I and II) performed by the U.S. Housing and Urban Development. The AHHS I survey was completed in 2005-2006 and AHHS II was completed in 2017-2018 (U.S. HUD, 2011, 2021). These data, available only at the aggregated scale (e.g., central tendency by age class of housing stock),[11] were combined with ambient air Pb monitoring data (from the EPA-managed database of ambient air monitoring data, Air Quality System [AQS]) to characterize the temporal trend in the relationship between dust Pb loading measurements and ambient air Pb concentrations (both metrics have exhibited downward trend across the two time periods).

Specifically, we related median dust Pb loadings (for houses built after 1977) from the two survey periods to median ambient air Pb concentrations at non-source monitors for same time periods (2005-2006 and 2018-2019 for AHHS I and II, respectively). This yielded a *delta factor* describing the quantitative relationship of ambient air Pb concentration to dust Pb loading across the two time periods. We used the dust Pb loading for houses built after 1977 to focus on houses built after the primary period of leaded paint usage and minimize the influence of leaded paint on the result. Although there are other Pb sources to indoor dust (e.g., tobacco smoke, some imported consumer products), this approach also involves the presumption that differences in indoor dust Pb between the two time periods would have been primarily influenced by the changes in ambient air Pb over this period. Derivation of the delta factor is illustrated below in Figure 6.

[11] Ideally, we would have used these datasets to derive regression models based on home-level observations (i.e., dust Pb loading and ambient air Pb characterized at the house-level). However, we did not have access to address data for AHHS I and II datasets for privacy reasons and therefore, had to work at a more generalized central-tendency trend level across the two time periods.

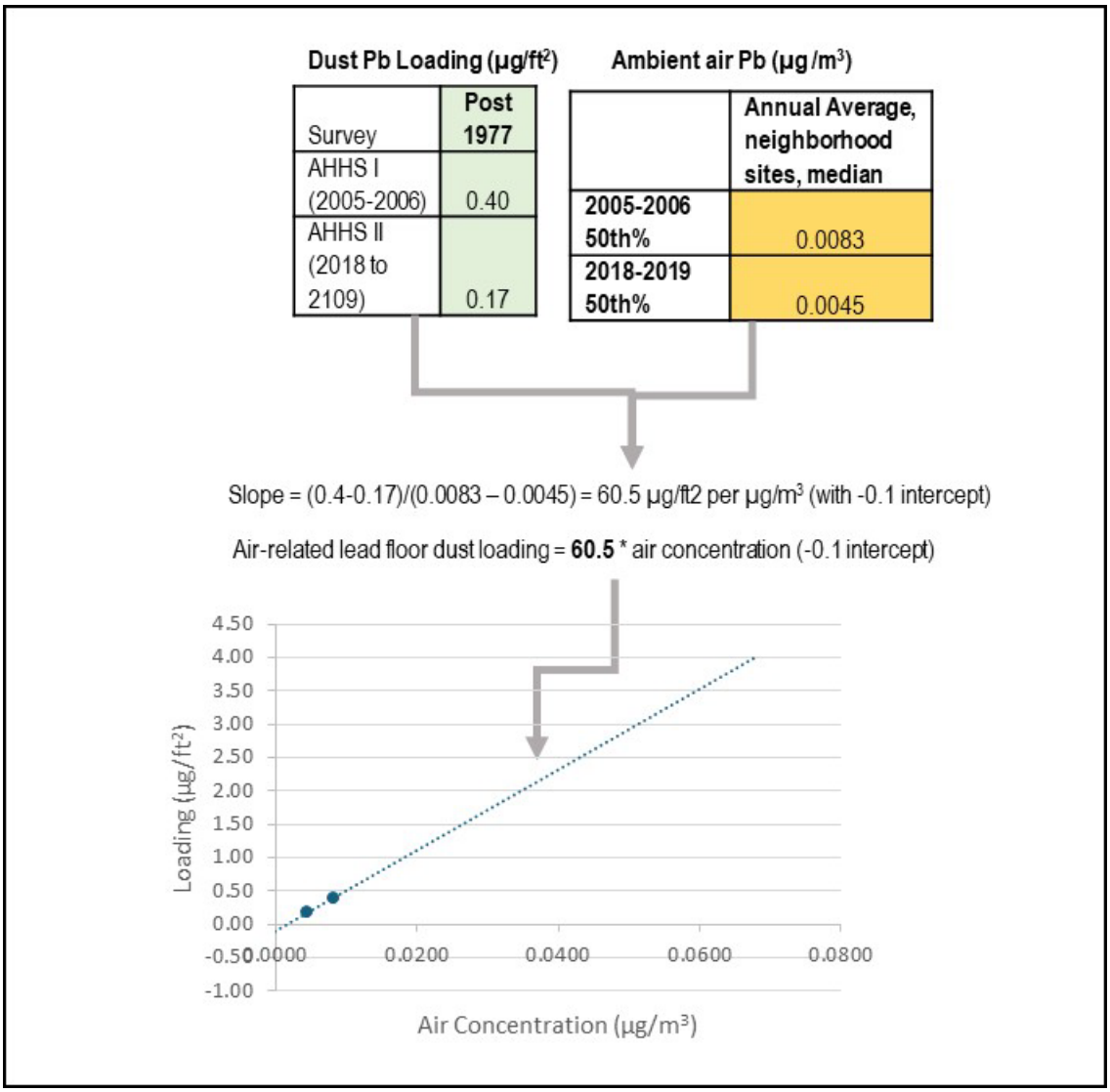

Figure 6. Empirical (delta factor) approach.

We recognize that the delta factor may be influenced by factors other than the trend in air concentrations over the period from 2005-2006 and 2018-2019 (e.g., changes in HVAC technology, changes in tracked-in Pb from soil, changes in dust derived from consumer products, and reductions in dust from leaded paint usage due to unleaded paint layers over leaded layers). In addition, the AQS monitoring data (though filtered to focus on neighborhood-level monitors) may not provide a representative central tendency estimate of ambient air Pb concentrations in air outdoor houses built after 1977. That is, the air Pb values may not be a good match for the AHHS I and II house measurements used in this statistical model. The reliance on a central-tendency based (two point) time trend rather than a more rigorous fitting of a regression curve using house-level observations is also a limitation and source of uncertainty. For example, the use of two median-based ratios (one for dust Pb and one for air concentration) prevents us from

assessing whether the shape of the temporal trend in the relationship of dust Pb loading and ambient air Pb over time is linear or non- linear.

Figure 7 compares dust Pb loading estimates based the updated Hybrid and Delta-factor models to each other and to the original 2007 Hybrid model over the range of ambient air concentrations represented by the two time period. This figure illustrates a moderate reduction in Pb loading estimates with the updated Hybrid mechanistic model, and provides a sense of the uncertainty associated with these estimates.

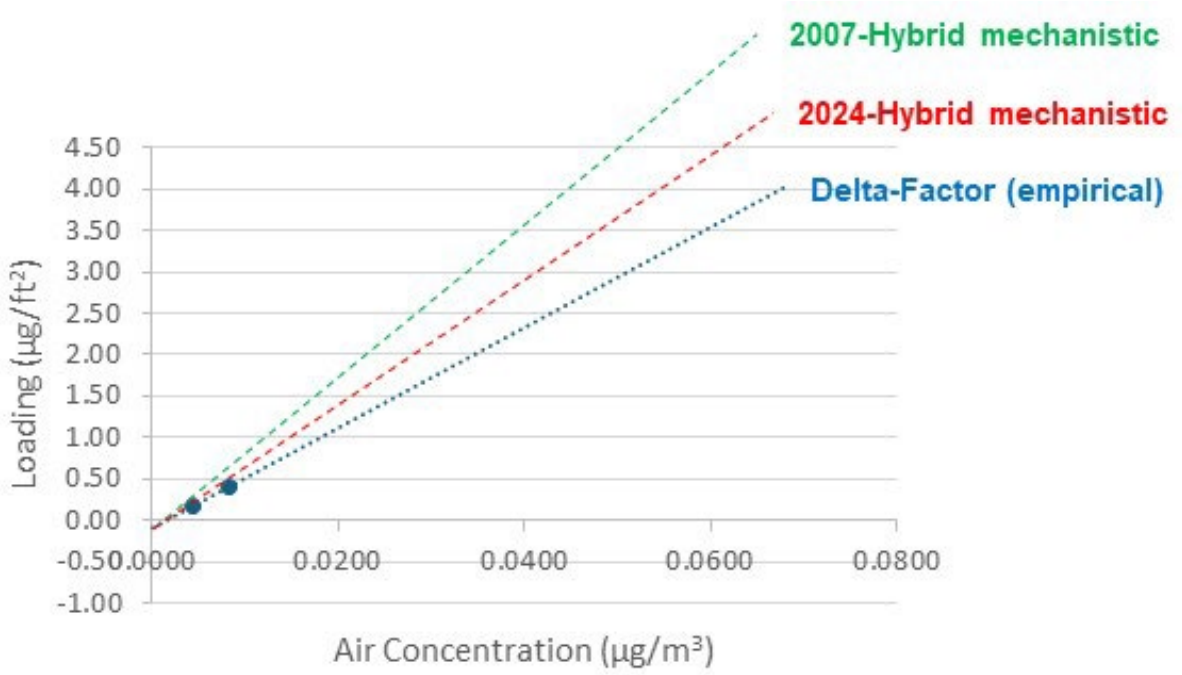

Figure 7. Indoor dust Pb loading estimates derived from annual average ambient air Pb concentrations using the 2007 and updated 2024 Hybrid-mechanistic models and using the Delta-factor empirical approach.

4.2.2. Step 2: Conversion of Dust Pb Loading into Concentration Estimate

As with the initial step of relating ambient air Pb to dust Pb loading, the second step of translating dust Pb loading estimates into equivalent dust Pb concentrations can be accomplished using either an empirical (regression) approach, or a mechanistic approach. In addition to our focus on the 2-step regression model used in the original 2007 Hybrid model, we also evaluated model 7 from Bevington et al. (2021).[12] Each of these options is discussed in greater detail below.

4.2.2.1. 2-Stage Regression

Although the datasets on which this regression approach is based are not recent, our review of the model derivation and consideration of the underlying study data finds that this approach (described in section 4.1.2) remains reasonable and appropriate for the current REA. Specifically: (a) while the datasets used in deriving the regression for converting dust Pb (wipe) loadings to equivalent vacuum loadings reflect Pb contamination levels likely higher than the current REA, the R^2 for this regression is relatively high; (b) the data used to derive the regression for converting vacuum-equivalent dust Pb loading into dust Pb concentration fall within the range of Pb contamination estimates expected for the current REA and the regression has a R^2 value indicative of a reasonable fit; and (c) inclusion of the conversion of wipe-equivalent dust Pb loading values into vacuum-equivalent values addresses an important potential source of uncertainty, thus increasing confidence in the overall approach. These observations indicate this to be a reasonable Step 2 approach. This model, as presented in section 4.1.2, is:

$$PbDUST = EXP[4.92 + 0.52 \times \ln\left(0.185 \times (PbWIPE)^{0.931}\right)]$$

where:

PbDUST = indoor dust Pb concentration (µg/g)
PbWIPE = modeled (wipe-equivalent) loading of dust Pb (µg/ft^2)

Dust Pb concentration estimates derived with this model are illustrated in Figure 8, along with estimates based on the Bevington et al. (2021) Model 7 (described in section 4.2.2.2 below).

4.2.2.2. Bevington et al. (2021) Model 7

The extensive literature search we conducted to investigate the availability of new approaches for converting dust Pb loadings to dust Pb concentrations (i.e., step 2) yielded a study by Bevington et al. (2021). This study uses five datasets in various combinations to develop 17

[12] The study by Bevington et al. (2021) develops 20 regression models for 20 different groupings of five datasets. The model #7 involves the datasets with generally lower loading and concentration values, which would be closer to those of interest in the REA than the other models developed from other combinations of datasets.

regression models. Three of the five datasets have loading measurements (or estimates)[13] in the range of loading values expected for the current REA. These are the HUD 1989-90 National Survey, the Canadian House Dust Study (CHDS) and the EPA's National Human Exposure Assessment Survey (NHEXAS). These datasets are listed in Table 2 below, with summary statistics for their dust Pb loading values.

Table 2. Summary of dust Pb loading datasets used in Bevington et al. (2021) regressions.

Study*	Sampling method	Dust Pb loading (µg/ft²)		
		Arithmetic Mean	Geometric Mean	Range
HUD National Survey (U.S. EPA, 1998)	wipe (converted)**	19.3	7.52	0.51 to 375
CHDS (Rasmussen et al., 2013)***	wipe	2.92	0.94	0.042 to 89.43
NHEXAS (Clayton et al., 1999)	wipe	64.1	6.5	0.2 to 9792

*Study information provided in Table 3 of Bevington et al. (2021).

**Loading data collected as vacuum in this survey and converted to wipe-equivalent values (Bevington et al, 2021, Supplemental Information, p. 5).

*** Measurements only collected in winter months.

The model based on these three datasets is Model 7 in Bevington et al. (2021). The model fit R^2 for Model 7 is 0.456. The three datasets used in fitting Model 7 together comprise a fairly large dataset, larger than (and inclusive of) the HUD 1989-90 National Survey dataset (used in the 2-stage regression). All three studies include a substantial portion of their observations well above the range of interest for the REA. The largest of the three datasets (the CHDS), while including the greatest number of Pb values within the range of interest, collected measurements only in the winter months, with potential implications on extent of representativeness across all months in a year (Rasmussen et al., 2013).

Additionally, while dust Pb loading in two of the three datasets are wipe-based measurements, the third study (HUD 1989-90 National Survey) is represented by dust Pb loading estimates that were converted to wipe-equivalent dust Pb loading values (from vacuum measurements) prior to use in the regression (Bevington et al, 2021, Supplemental Information, p. 5). The regression model (Bevington Model 7) is based on the pooled dataset with its mixture of wipe measurements and estimates derived through use of a separate empirical regression. This manipulation is a source of uncertainty to estimates based on the resultant model. The formula for this model (Model 7) is presented below and dust Pb concentration estimates derived using it are illustrated in Figure 8 along with estimates based on the Hybrid model's 2-stage regression.

Model 7: Dust Pb concentration = EXP(4.946+0.395*(LN(Pb-loading)-0.62))

[13] The loading measurements for the HUD 1989-90 survey are vacuum measurements. Wipe-based equivalents were estimated based on regressions from other datasets (Bevington et al, 2021, Supplemental Information, p. 5).

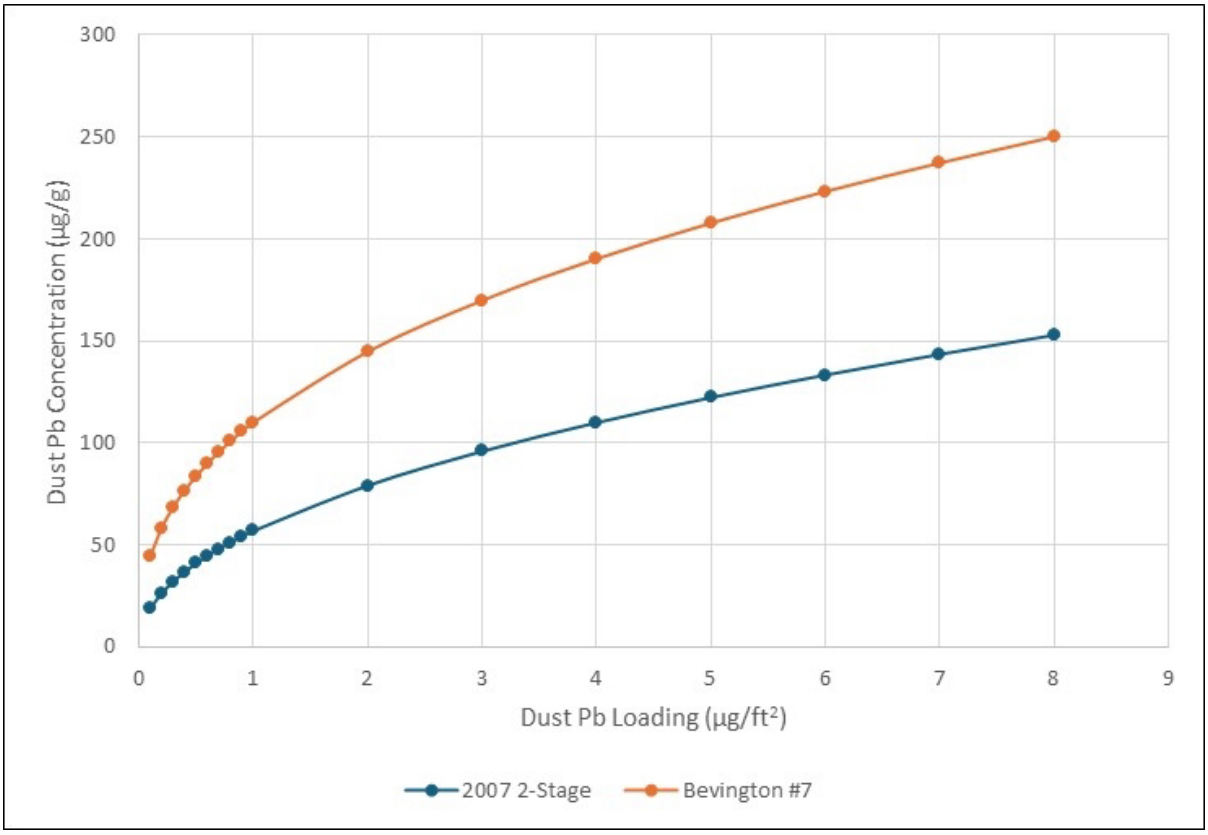

Figure 8. Dust Pb concentrations estimated from the 2007 2-stage regression and the Bevington et al. (2021) Model 7.

4.2.3. Current Model – From Air Pb Concentration to Dust Pb Loading and Concentration

In addition to the 2024 (updated) Hybrid (mechanistic-2stage) model, the various combinations of additional methods for steps 1 and 2 described above yield three other dust Pb modeling approaches. Each of these approaches is identified in Table 3 by a short-hand designation based on its options for step 1 and 2. Table 3 also recognizes the original 2007 Hybrid dust Pb model.

Table 3. Dust Pb modeling approaches developed for the current REA.

2-Step Dust Pb Approaches	Step 1 (estimating dust Pb loading from ambient air Pb concentration)		Step 2 (estimating dust Pb concentration from dust Pb loading)	
	Mechanistic (updated)	**Empirical Delta-Factor**	**2-Stage Regression**	**Bevington Model 7**
REA Model: 2024 (updated)Hybrid	X		X	
Additional methods				
Mechanistic - Model 7	X			X
Delta-factor - 2-Stage		X	X	
Delta-factor – Model 7		X		X
The 2007 Hybrid is the Mechanistic (2007 parameterization) combined with the 2-stage regression model.				

Dust Pb concentration estimates derived with the 2024 (updated) Hybrid model are presented in Figure 9, along with estimates based on the 2007 Hybrid and the three additional combinations of methods for each of the two steps. The updated Hybrid estimates fall within the range of estimates generated from the various additional methods. Key strengths and limitations, with their associated uncertainties, for the updated Hybrid model and for the various additional methods are summarized in Table 4.

Collectively, the available information provides an indication of the appreciable uncertainty associated with estimating indoor dust Pb concentrations from ambient air Pb concentrations, and does not indicate a clear benefit of alternatives to the Hybrid modeling approach. Thus, we find the 2024 (updated) Hybrid approach to remain the appropriate model for the REA, and also recognize the continuity it provides with the model for the 2007 REA in the 2008 Pb NAAQS review. We additionally find the additional methods to provide approaches for use in model evaluation, interpretation of dust Pb estimates, and characterization of uncertainty associated with the dust Pb estimation step of the REA.

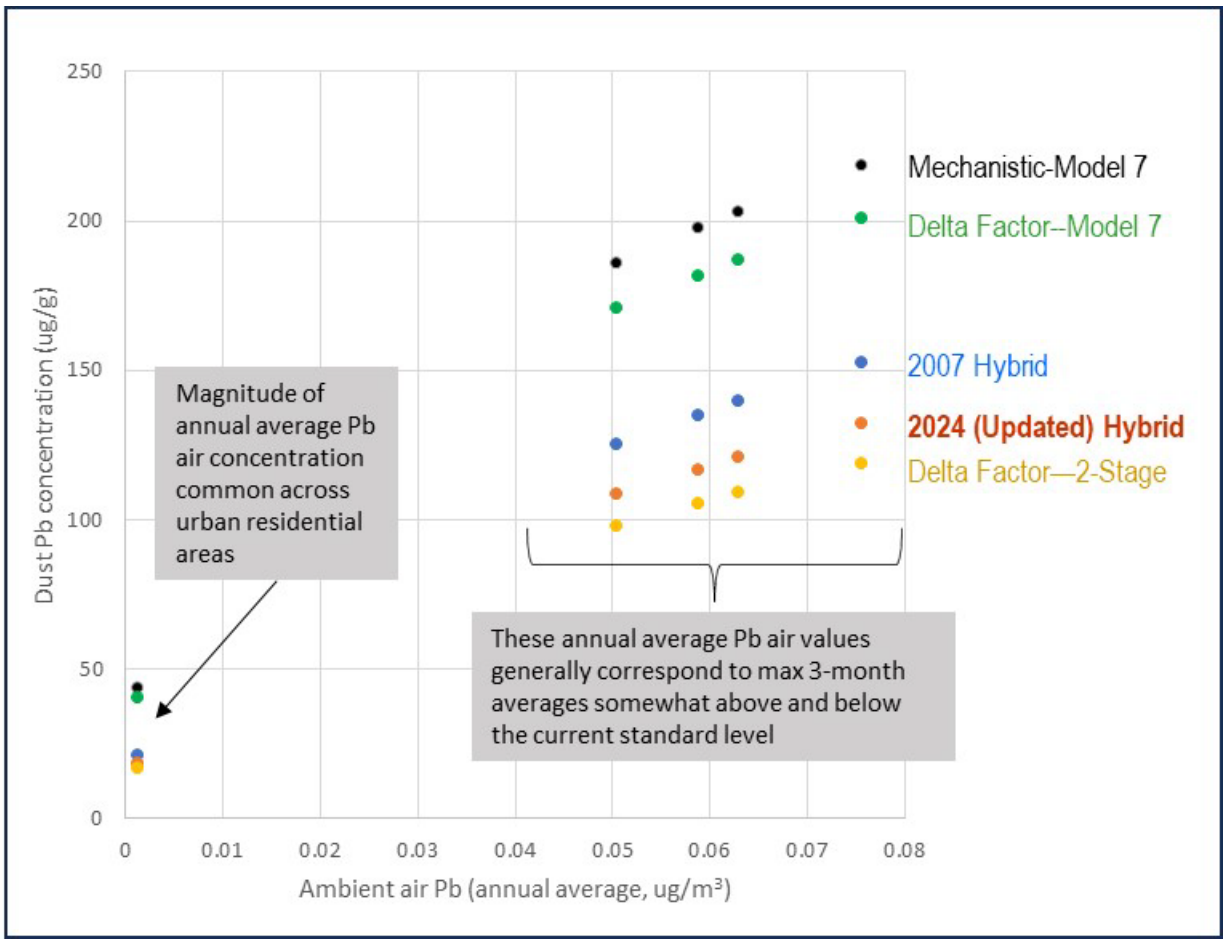

Figure 9. Dust Pb concentrations estimated by 2024 (updated) Hybrid, the 2007 Hybrid and other methods.

Table 4. Strengths and limitations of the 2024 (updated) Hybrid and additional methods.

	2024 (Update) Hybrid		Additional Methods	
	Step1: 2024 mechanistic	**Step 2: 2-stage regression**	**Step 1: Delta-Factor**	**Step 2: Bevington et al. (2021) Model 7**
Strengths	• Conceptual focus on dust Pb loading specific to ambient air as the Pb source • Parameters reflect influential processes in loading of ambient air Pb to indoor dust.	• Incorporates wipe-to-vacuum transformation for wipe data (1st regression), avoiding uncertainty contributed with a regression based on data of mixed sample types. • Extensive analyses supporting derivation of the first regression from three datasets. • Relatively higher R^2 values associated with both regressions, including 0.683-0.810 for the first and 0.48 for the second.	• Basis in real-world data. • Measured dust Pb concentrations in range of interest. • Focus on recent aged housing means likely little influence of leaded paint.	• Based on three datasets with combined n of1526 (Table 5, Bevington et al., 2021). • Reasonable fit, with R^2 =0.456 • Dust Pb loading values are inclusive of range of interest.
Limitations	• Overall complexity requires significant parameterization, making evaluation challenging.	• Relatively low n for wipe-to-vacuum transformation regression (Table 1) • Wipe-based loading values underlying regression for relating wipe to vacuum loading are somewhat above levels of interest.	• Derivation from two datapoints (a median for each time period) rather than a full dataset of individual house data. • Uncertainty associated with representativeness of air Pb concentration for each time period • Time-trend nature of slope means additional factors besides changes in ambient air could potentially impacting slope.	• Vacuum loading measurements in one of the three underlying datasets were adjusted to wipe-based estimates, contributing uncertainty to model. • Although inclusive of lower values of interest, most of the values in the 3 datasets are much higher than those of interest in the REA. • Largest dataset is limited to winter months.

* As described in section 4.1.2 above, while an R^2 value is not available for the combined regression model for the first stage (wipe loading-to-vacuum loading), R^2 values are available for the three regressions on which the final regression is based. Of the three (as summarized in Table 1 above), the regression based on the largest dataset and with the largest range of data values has the highest R^2 (0.81), which would dominate the fit of the final regression model, placing the R^2 for the final model likely between 0.683 and 0.810 (U.S. EPA, 1997, Appendix D).

4.3. Other Approaches Evaluated for Step 2 - Pb Concentration Estimation

Two other methods to converting estimates of dust Pb loading into equivalent dust Pb concentrations were evaluated. While initially promising, further exploration of each revealed significant limitations. These approaches are summarized in the subsections below.

4.3.1. AHHS II Regression

A dataset of house-level vacuum-based dust Pb measurements collected as part of the AHHS II survey (2017-2019) and dust Pb loading measurements were evaluated for use in deriving a regression relating dust Pb loadings to dust Pb concentrations. The relatively low dust Pb concentrations indicated a potential relevance for the current REA. However, limitations in data collection methods would be expected to contribute significant uncertainties to application of such a regression.

Unlike the HUD 1989-90 survey (U.S. EPA, 1995a), the AHHS II survey did not utilize a consistent rigorous procedure for collecting dust Pb loading and dust Pb concentration data. Instead, for the AHHS II survey, dust Pb concentration data were based on vacuum bags (and canisters) collected from residents during the survey process (i.e., did not reflect the rigorous protocol used in collecting wipe dust Pb loading samples during those visits). The more informal resident-based vacuum sample collection methodology is likely to contribute significant uncertainty to efforts to relate the two measurements across the survey dataset. And in fact, this was demonstrated by the low R^2 value (0.21) associated with a log-log model relating dust Pb loading (wipe based) on dust Pb concentration (vacuum based). Given the low R^2 value and the differences in protocols for collecting the wipe and vacuum data (as referenced earlier), we concluded that a regression developed from this survey dataset was too uncertain for considering use in the REA.

4.3.2. Dust Mass Approach

A last approach considered for converting dust Pb loading estimates into equivalent dust Pb concentrations, was the use of total dust loading. Dust Pb loading is the mass of Pb per surface area of floor. Conceptually, dividing dust Pb loading by the mass of total dust (i.e., total dust loading) for that same unit of surface area would yield mass of Pb per mass of total dust which is our analytical target.

Thus, we investigated the availability of studies that might support identification of a typical (central tendency) value of total dust loading in the residential setting. However, this value is not easily identified. Such a value can be back calculated from central-tendency measures of dust Pb loading and dust Pb concentrations (i.e., from the data in the HUD National Survey or from the more limited data associated with AHHS II). Use of a central tendency (single) dust Pb loading value derived in this way yielded unreasonably high dust Pb concentrations at higher dust Pb loadings, which may reflect a potential correlation between dust Pb loading and dust loading (e.g., higher dust Pb loadings are associated with higher total dust

loadings). Such a correlation may occur if the same processes producing elevated dust Pb are also producing higher total Pb (e.g., increased particulate intrusion into the house, reduced cleaning, greater aging and flaking of indoor paint, increased tracking in of outdoor soil).

If dust Pb loading and total dust loading are correlated, then as dust Pb loading increases, so too would total dust loading which can dampen the associated increase in dust Pb concentration. This will yield a concave non-linear curve for dust Pb concentration and dust Pb loading. However, replacement of the range of total dust loading with a single central-tendency total dust loading value produces a linear curve between dust Pb loading and dust Pb concentration which will result in a high-bias in estimates of dust Pb concentration at higher levels of dust Pb loading. Ideally a spline-function approach would be used to reflect different total dust loading values (for different ranges of dust Pb concentration) in creating the curve for translating dust Pb loading into dust Pb concentration. However, we did not have sufficiently refined total dust loading data to support this type of spline approach. Consequently, this method for translating dust Pb loading into equivalent dust Pb concentration was not concluded to be useful at this time.

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