GROUNDWATER PATHWAY ANALYSIS FOR LEAD-BASED PAINT (LBP) ARCHITECTURAL DEBRIS

BACKGROUND DOCUMENT

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June, 1998

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1.0 INTRODUCTION

A Monte Carlo modeling analysis was performed to assess the potential groundwater exposure due to lead associated with disposal of Lead Based Paint (LBP) debris managed in construction and demolition landfills (C & D) and municipal landfill waste management units. The modeling analysis was performed using the EPA's Composite Model for Leachate Migration with Transformation Products (EPACMTP; USEPA, 1995 a, b, c, and e). EPACMTP was selected for the analyses based on the following capabilities:

- Capability to perform a full, Monte Carlo-based, probabilistic exposure assessment.
- Utilization of a site-based, regional approach which incorporates EPA survey data on various types of waste management units which directly accounts for physical dependencies between various input parameters.
- Capability to accommodate finite source as well as continuous source conditions.
- Capability to accommodate chemical specific fate processes, including sorption, hydrolysis, and formation of (toxic) daughter products.
- Capability to incorporate geochemical effects on subsurface migration of lead through linkage with EPA's MINTEQA2 (Allison at al. 1991) metals speciation model.

EPACMTP has been designed for Monte Carlo groundwater exposure assessments. The model incorporates default probability distributions for the source, climatic and hydrogeologic parameters needed by the fate and transport model. In order to model the fate and transport of metals, including lead, a linkage has been developed between EPACMTP and the EPA's MINTEQA2, a geochemical speciation model. The subsurface mobility of metals is influenced by major geochemical variables (such as pH, amorphous iron, natural organic matter content and organic matter in the leachate) which control speciation sorption and solubility of metals species. The linkage with MINTEQA2 allows the impact of these geochemical conditions to be incorporated into the EPACMTP modeling analysis.

This document describes the application of EPACMTP to model the groundwater impact of lead based paint containing in architectural debris managed in C & D and municipal landfills. Section 2 describes the modeling approach and data sources used. Section 3 presents the results of the fate and transport modeling. Numerical stability of the Monte Carlo analysis output (receptor well concentrations) is also address in section 3.A summary of the analysis is presented in Section 4 and references are provided in Section 5.

2.0 MODELING APPROACH AND DATA SOURCES

2.1 Modeling Approach

The EPACMTP modeling approach for the LBP groundwater pathway analysis is summarized in Table 2.1. The modeling analysis was conducted in finite source, Monte Carlo mode for a C&D landfill waste management scenario. As an alternative, modeling analysis was also conducted for a municipal landfill waste management scenario. The groundwater fate and transport model was used to predict groundwater exposure concentration at a receptor located downgradient side of the landfill within a radius of one mile. The exposure concentration was taken to be the peak receptor well concentration occurring within 10,000 years following the initial release from the waste unit. The Monte Carlo fate and transport simulation provides a probability distribution of receptor well concentrations which can be used to determine the likelihood that a given exposure level will be reached (or exceeded).

Table 2.1 lists the methodology and data sources used to obtain values for the source-specific parameters, chemical-specific parameters, unsaturated and saturated zone parameters, and receptor well location parameters. All parameters are in principle described by probability distributions. The determination of the source related parameters for the LBP debris modeling analysis is discussed in section 2.2. Probability distributions for other model input parameters are presented in the EPACMTP background documents (USEPA, 1997a, 1996 a,b, c, and d). Key aspects of the modeling approach are discussed below.

EPACMTP simulates the subsurface fate and transport of individual contaminants in the leachate emanating from the base of a waste management unit. The model does not attempt to account for the processes occurring inside the waste unit itself, except for the following: The leachate generation rate was determined using the Hydrologic Evaluation of Landfills Program (HELP) model (Schroeder, 1994) based on a Subtitle D landfill with a 2-foot earthen cover with grass vegetation cover, but with no liner or leachate collection system present (USEPA, 1995-a). Given information on the total amount of waste placed in the landfill, EPACMTP will compute the time duration of the leachate pulse using straight-forward mass-balance principles. The model can handle either the case of a monofill, or of co-disposal of multiple wastes. In the latter case, the model accounts for the fact that the waste of concern occupies only a fraction of the landfill. The model however does not consider interactions among, or the combined effects of, multiple waste streams. In this case, the LBP debris was co-disposed with other C&D waste in C&D landfill scenario and with municipal waste in a municipal landfill scenario.

In the EPACMTP Monte Carlo modeling approach, the climatic and hydrogeological model input parameters are assigned values based on the geographical locations of landfill sites across the U.S. This approach preserves the interdependence between site location and the climatic and hydrogeological properties of the region where it is located. The waste management unit characteristics such as geographical location, unit area, depth and volume were obtained in a telephone survey of industrial Subtitle D waste management units conducted by USEPA in 1986. The information obtained in the survey along with climatic and hydrogeological properties of the waste management units are presented in a report (EPA 1996). In the LBP modeling analysis, the existing EPACMTP relations between climatic and hydrogeological regions for industrial Subtitle D landfills were used. The underlying assumption in using these relationships is that, in general, the overall distributions of C&D and municipal landfill sites are the same as the industrial Subtitle D landfills. In the modeling analysis, the receptor well was located anywhere down-gradient from the landfill, within one mile from the waste management unit.

Table 2.1 EPACMTP Modeling Options for LBP Analysis

Management Scenarios:	C&D landfill, alternatively municipal landfills
Modeling Scenario:	Monte Carlo; 10,000 iterations with constant concentration pulse source.
Exposure evaluation:	Down gradient groundwater receptor well; peak well concentration within 10,000 year exposure time limit.
Source Parameters:	
Waste Unit Area: Waste Stream Volume: Landfill Depth: Volume Fraction LBP debris Infiltration Rate:	From Report to Congress (EPA, 1985) Appendices C and D. Calculated Section 2.3 Calculated Section 2.3. Site-based, derived from water balance using HELP model (Schroeder, 1994).
Leaching Duration	Derived, continues until all constituent has leaches out.
Chemical Specific Parameters:	
Decay Rate:	No decay (metal)
Sorption:	MINTEQA2 sorption isotherm
Unsaturated Zone Parameters:	
Depth to groundwater: Soil Hydraulic Parameters: Fraction Organic Carbon: Bulk Density:	Site-based, from API/USGS hydrogeologic database (API, 1989; Heath,1984). ORD data based on national distribution of three soil types (sandy loam, silt loam, silty clay loam) ORD data based on national distribution of three soil types (sandy loam, silt loam, silty clay loam) ORD data based on national distribution of three soil types (sandy loam, silt loam, silty clay loam).

Table 2.1 EPACMTP Modeling Options for LBP Analysis (continued)

Saturated Zone Parameters:	
Recharge Rate: Saturated Thickness: Hydraulic Conductivity: Porosity: Bulk Density: Dispersivity: Groundwater Temperature: Fraction Organic Carbon: pH	Site-based, derived from regional precipitation/evaporation and soil type Site-based, from API/USGS (API, 1989; Heath, 1984). hydrogeologic database EPA Storet Database, Effective porosity derived from national distribution of aquifer particle diameter Derived from porosity Derived from distance to receptor well based on Gelhar's distribution of field scale dispersivity (EPA 1997a) Site-based, from USGS regional temperature map National distribution, from EPA STORET database National distribution, from EPA STORET database
Receptor Well Location:	
X-distance Y-distance Depth of Intake Point	Empirical distribution within 0-1 mile from waste unit Uniform within 1 mile downgradient radius Uniform throughout saturated thickness of aquifer

The (radial) distance between the receptor well and the down-gradient side of the landfill is given by an empirical probability distribution (Table 2.2), based upon reported distances between Subtitle D municipal landfills and the nearest downgradient domestic drinking water well. Table 2.2 indicates a median well distance of about one-quarter mile (427 m). The vertical position of the well intake point (z-direction) was taken to be uniform throughout the saturated thickness of the aquifer.

Fate and transport of metals such as lead in the subsurface is influenced by complex geochemical interactions. Deutsch (1997) provides a review of lead geochemistry and summarized some of these studies. EPA has also several papers discussing the issues involved in subsurface lead transport (USEPA, 1989, 1990, 1991 and 1992). To account for these processes, the OSW has developed and implemented a modeling approach which utilizes the MINTEQA2 metals speciation model in conjunction with the EPACMTP subsurface fate and transport model. The MINTEQA2 model has been applied to generate effective sorption isotherms reflecting variations in four geochemical master variables affecting metals fate and transport. These factors are: pH, leachate organic matter, natural organic matter in the soil or aquifer, and iron hydroxide content. Each of these parameters has a range of values, reflecting their nationwide probability distributions (USEPA, 1995 d). In addition, thirteen other geochemical constituents were identified as second type parameters which commonly occur in groundwater at concentrations great enough to warrant inclusion in the general background chemistry of the MINTEQA2 geochemical speciation modeling.

MINTEQA2 was used to generate a family of effective sorption isotherms (USEPA, 1995d), reflecting the variations and combinations of the four geochemical master variables. A sorption isotherm, by definition, is a plot of the sorbed phase concentration versus the solution phase concentration. MINTEQA2 generates the sorption isotherms in tabular form for various combinations of the four master variables and for both unsaturated and saturated zones. During each realization of the EPACMTP Monte Carlo simulation, one isotherm is selected at random from the set of isotherms for the metal being analyzed. In this manner, the variability in mobility of a metal which results from the nationwide variation in subsurface geochemical conditions, is incorporated into the Monte Carlo analysis.

The MINTEQA2 derived isotherms are generally nonlinear, i.e., the apparent partition coefficient between the dissolved phase and the soil or aquifer solid phase varies with concentration. Apparent partition coefficient is a ratio of sorbed phase concentration over dissolved phase concentration. To handle this condition (k_d), the unsaturated zone transport module of EPACMTP was modified to incorporate a method-of-characteristics based solution which handles even highly nonlinear isotherms very efficiently. The isotherms generated by MINTEQA2 exhibit near-linear behavior at low concentration values, and increasing nonlinearity at higher concentrations. Accurate modeling of the nonlinear sorption is therefore most important close to the leachate source, i.e., in the unsaturated zone underneath the waste unit. Because concentrations diminish with travel distance, and the contaminants are diluted in the ambient groundwater upon entry in the saturated zone, EPACMTP models nonlinear sorption only in the unsaturated zone. Once the leachate reaches the watertable and mixes with the ambient groundwater, concentrations will be in a much more dilute range, in which the isotherms generally are close to linear. Saturated zone transport is therefore modeled using a linear isotherm, with a k_d value selected based on the groundwater concentration underneath the waste unit, predicted by the model.

A sample plot of lead apparent partition coefficient distribution with dissolved phase concentration used in EPACMTP is shown in Figure 2.1. The figure depicts the dependence of the partition coefficient (k_d) upon the lead concentration. The figure illustrates a number of the key features of lead sorption behavior. The isotherm shows the high degree of non-linearity, with a greater than 3-orders-of-magnitude variation in k_d for concentrations greater than 10^{-6} mg/L. Secondly, the figure indicates the high values of k_d; at a concentration of 1 mg/L, the k_d still exceeds 1000 L/kg. This behavior is typical for lead which is a strongly sorbed metal, except under low pH conditions. These high k_d values result in long subsurface travel times and pronounced attenuation of concentration values, because much of the contaminant mass in the soil and groundwater is held on the solid phase.

Table 2.2 Frequency Distribution of Receptor Well Distance

Distance (m)	Cumulative Frequency
0.0	0.00
0.6	0.00
13.7	0.03
19.8	0.04
45.7	0.05
104	0.10
152	0.15
183	0.20
244	0.25
305	0.30
305	0.35
366	0.40
427	0.50
610	0.60
805	0.70
914	0.80
1160	0.85
1220	0.90
1370	0.95
1520	0.98
1610	1.00

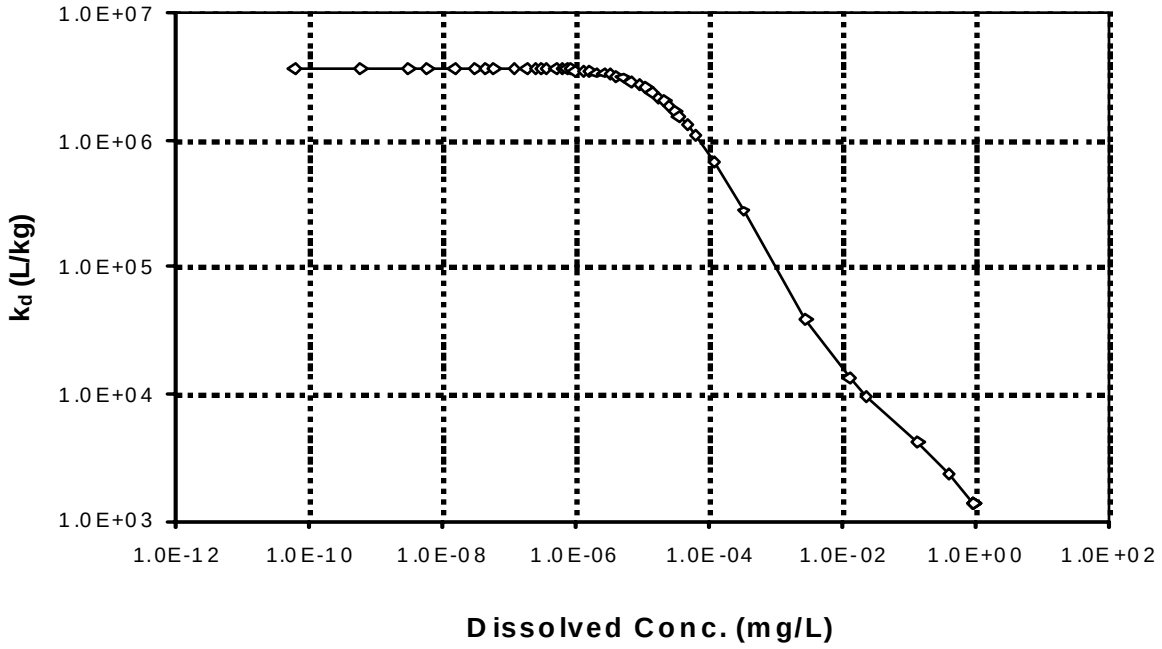

Figure 2.1 Sample lead apparent partition coefficient (K_d) distribution with dissolved phase concentration.

2.2 Modeling Scenarios for Lead Based Painted (LBP) Debris

The groundwater modeling analysis for LBP abatement debris plus painted demolition debris was performed considering two waste management scenarios; Construction and Demolition (C& D) landfills and Municipal Solid Waste Landfills (MSWLF), and two types of LBP debris; abatement and abatement plus painted demolition. A total 3 scenarios were modeled as summarized in Table 2.3.

Table 2.3 Modeling Scenarios for Lead Based Paint (LBP) Debris

LBP Debris Description	C&D Landfills	Municipal
LBP Abatement	**X**	**O**
LBP Abatement + Painted Demolition	**X**	**X**

X - Modeled
O - Not modeled

2.3 Source Related Parameters

EPACMTP requires specification of the waste source area, the leachate concentration emanating from the base of the waste source, and the duration of the leachate pulse. The first two parameters are entered directly as input parameters to the model. The leachate pulse duration, t_p , is calculated based upon the total amount of constituent in the landfill and the rate of leaching. The methodology is presented briefly below, followed by a discussion of how the various parameters were determined for the LBP analysis.

The duration of the leaching period is calculated in EPACMTP based upon elementary mass-balance considerations (USEPA, 1996 c). The following relationship is used:

$$A \cdot d \cdot F_{hw} \cdot P_w \cdot C_w = A \cdot I \cdot C_L \cdot t_p \qquad (1)$$

where

A = Waste unit area (m^2)
d = Waste unit depth (m)
F_{hw} = Volume fraction of waste in the unit
P_w = Waste density (kg/L)
C_w = Waste concentration (mg/kg)
C_L = Leachate concentration (mg/L)

I = Infiltration rate (m/y)
t_p = Leaching duration (y)

The left-hand side of (1) represents the total amount of constituent mass in the waste management unit; the right-hand side represents the mass that has been removed by leaching over the time period t_p. The formulation of (1) tacitly assumes that the leachate concentration, C_L, remains constant over the leaching period t_p, and that there are no mechanisms of removal other than leaching. EPACMTP alternatively can accommodate a gradual reduction in leachate concentration to represent source depletion (USEPA, 1996 b). However, this option was not used for the present analysis as the case of a constant leaching concentration is considered more appropriate for metals (USEPA, 1996 d). Equation (1) can be rewritten as

$$t_p = \frac{A \cdot d \cdot F_{hw} \cdot P_w \cdot C_w}{A \cdot I \cdot C_L} \qquad (2)$$

The waste source area, A, cancels in the above equation. This parameter is however still used elsewhere in the EPACMTP data input. The infiltration rate, I, is assigned values based on the nationwide OSW modeling approach for industrial landfills as described above. The remaining parameters in Eq. (2), which are needed, are the landfill depth, d, the fraction of the landfill volume occupied by LBP debris, F_{hw}, the density of the waste, the total concentration of lead in LBP debris, C_w, and the lead leachate concentration, C_L. The determination of these parameters is presented in the following sections.

Landfill Area

Data on the area of C&D landfills was obtained from the 1988 EPA Report (EPA 1988) to Congress on Solid Waste Disposal in the United States (EPA, 1988; Table 4-5). This table presents the data for Demolition Debris landfills. This report provides an overall breakdown of landfill acreage into categories of less than 10, 10 to 100, and greater than 100 acres. Lower and upper limits of 0.1 and 250 acres, respectively, were assigned to the waste unit area in the modeling analysis. The upper limit of 250 acres (approximately 10^6 m^2) was selected based on the fact that fewer than 1% of Industrial Subtitle D landfills exceed this size (USEPA, 1995-a). The resulting frequency distribution of landfill area used in the EPACMTP analysis is shown in Table 2.4.

Table 2.4 Frequency Distribution of C&D Landfill Area

Area		Cumulative Frequency
acres	m^2	
0.1	404.7	0.0
10.0	40,470	0.606
100.0	404,700	0.970
250.0	1,012,000	1.00

As an alternative, risk exposure associated with the disposal of lead based paint in municipal landfills was also evaluated. Data on the area of municipal landfills were obtained from a U.S. EPA report on national survey of solid waste (municipal) landfill facilities (USEPA, 1988). According to this report there are 6034 municipal landfills. The frequency distribution of the municipal landfill area is shown in Table 2.5.

Table 2.5 Frequency Distribution of Municipal Landfill Area

Area		Cumulative Frequency
acres	m^2	
0.99	4006.5	0.0
2.0	8094.0	0.1
5.0	20235.0	0.25
15.0	60705.0	0.5
48.0	194256.0	0.75
104.0	420888.0	0.90
2310.0	9348570.0	1.00

2.4 Landfill Depth and Waste Quantity

This section describes the derivation of the quantities of C&D waste and LBP debris disposed in landfills. Section 2.3.1 provides the estimated waste quantities based on EPA's estimate of the amount of LBP debris generated under lead abatement programs. Section 2.3.2 provides alternative and more conservative, i.e. higher waste amounts, estimates which include abatement and painted demolition debris.

2.4.1 Waste Quantities Estimated from Lead Abatement Programs

The annual loading of construction and demolition, including Removal and Renovation (R&R), waste in C&D landfills is estimated as 16.24×10^6 tons/year, comprised of 11.8×10^6 tons/yr of demolition waste, and 4.44×10^6 tons/yr of construction waste as shown in appendix C. The amount of LBP abatement debris generated under lead abatement programs is estimated as a total of 19.1×10^6 tons over the next 34 years. This estimate is derived as shown in appendix and summarized below:

1. Total LBP debris from abatement activities:

Public	-	1.4 million units x 1.55 tons/unit	= 2.17 million tons
Private	-	38.6 million units	
		6.4 million units of children x 2.60 tons/house (1 out of 6 homes have a child)	= 16.64 million tons
Child - occup.	-	family 118,000 x 1.55 tons/house	= 0.183 million tons
		daycare 6,900 x 15.99 tons/center	= 0.110 million tons
			19.1 million tons

2. Annual quantities of LBP abatement debris:

- pre-regulatory changes = 514,000 tons
- post-regulatory changes = 562,000 tons
(assumed 10% overall additional abatement)

Conclusion: at a constant waste generation rate of 562,000 tons per year, all abatements are likely to be completed in 34 years.

There are approximately 1800 C&D landfills in the US (ICF, 1995). Assuming a 50-year life of the landfill and the C&D waste disposal rate of $16.24\ 10^6$ tons/year (see above), the total amount of waste in all C&D landfills can be calculated as (50 yrs) x (16.24×10^6 tons/yr) = 8.12×10^8 tons. The dominant components of C&D waste mass are concrete and asphalt (ICF, 1995). For the present analysis, an average density of 1.5 short tons/yd^3 (= 1779 kg/m^3)[1] was used for C&D waste, thus, the waste mass of 8.12×10^8 tons thus corresponds to a waste volume of

$$8.12\ x\ 10^8\ tons\ x\ \frac{907\ kg/ton}{1779\ kg/m^3} = 4.14\ x\ 10^8\ m^3$$

From the available information on number of C&D landfills, the area distribution, and the total volume of C&D waste, an average depth of C&D landfills was calculated. The aggregate area of all C&D landfills was estimated from the data in Table 2.3, as follows: The landfill areas within each frequency interval in Table 2.3 were taken to be uniformly distributed, with an average area equal to the midpoint area of each interval. For example, within the first frequency interval, there are 0.606 x 1800 = 1091 landfills, with an average area of 0.5 x (404.7 + 40,470) = 20,437 m^2. The total area of landfills in this interval is calculated as 1091 x 20,437 = 22,297,149 m^2. The number of landfills and total area in each interval were calculated in this manner, and then added together to yield an estimated aggregate area of all C&D landfills of 2.06×10^8 m^2. Next, the average landfill depth is calculated as 2.01 m (ratio of waste volume 4.14×10^8 to area 2.06×10^8).

The volume fraction of C&D waste that is LBP abatement debris is estimated from the ratio of the total volume of C&D waste in a landfill to the LBP abatement waste volume, assuming one half of all C&D landfills receive LBP abatement waste (900 LBP landfills). The mass of LBP debris is converted to a waste volume assuming a density of 1.0 tons/yd^3 (= 1186 kg/m^3). LBP abatement debris consists primarily of wood materials (doors, windows, etc), and it is therefore appropriate to use a lower density than for bulk C&D waste. The total amount of LBP abatement debris produced is thus equal to:

[1] 1 short ton = 907.2 kg; 1 yd^3 = 0.765 m^3

$$19.1 \times 10^6 \ tons \ \times \ \frac{907 \ kg/ton}{1186 \ kg/m^3} = 1.46 \times 10^7 \ m^3$$

In the present analysis, it is assumed conservatively that the LBP abatement debris is distributed among one half of total of 1800 C&D landfills; the aggregate landfill volume is:

$$4.14 \ x \ 10^8 \ x \ \frac{900}{1800} = 2.07 \ x \ 10^8 m^3$$

The LBP abatement debris volume represents (1.46×10^7) ÷ (2.07×10^8) x 100 = 7.1% of the landfill volume. It was assumed for the present worst case analysis that both the landfill depth and the volume fraction of LBP abatement debris are the same in all C&D landfills that receive the waste. This assumption implies that the actual amount of LBP abatement debris in each landfill is proportional to the total volume (= area x depth) of that landfill.

2.4.2 Waste Quantities Including Painted Demolition Debris

Two alternative waste management scenarios were modeled. In the first scenario, the waste was assumed to be disposed in C&D landfills. Whereas, in the second senario the waste was assumed to be disposed in municipal landfills. The estimate of the LBP debris (including R&R debris) were also obtained based on two disposal scenarios: (1) before year 2010 and (2) after 2010; assuming that some of the LBP debris would be recycled and reused (Appendix C). The estimation of waste volumes and effective waste depths are discussed below.

2.4.2.1 LBP Debris Disposed in C&D Landfills

The waste quantities managed in Construction and Demolition (C&D) landfills are in Table 2.6.

Table 2.6 Quantities disposed of in C&D landfills (tons/yr)

	Before 2010	After 2010
C&D Debris	31 M	18 M
Painted demolition debris	0.83 M	0.83 M
Abatement debris	0.56 M	0.34 M
Total	**32.4 M**	**19.2 M**

Source: Appendix C

Using 1995 as the base year and assuming a 50-year life of a C&D landfill, as in Section 2.3.1, the total amount of landfilled waste was calculated as shown below:

$$(15 \text{ yrs} \times 32.4 \times 10^6 \text{ tons/yr}) + (35 \text{ yrs} \times 19.2 \times 10^6 \text{ tons/yr}) = 1.2 \times 10^9 \text{ tons}$$

Assuming an overall density of C&D waste of 1.5 tons/yd^3, this C&D waste quantity corresponds to a volume of

$$1.2 \times 10^9 \text{ tons} \times \frac{907 \text{ kg/ton}}{1779 \text{ kg/m}^3} = 6.04 \times 10^8 \text{ m}^3$$

Using the same aggregate C&D landfill area of 2.06 10^8 m^2 calculated in the previous section, the average C&D landfill depth is

$$\frac{6.04 \times 10^8}{2.06 \times 10^8} = 2.93 \text{ m}$$

Consistent with the previous section, it is assumed that LBP debris is generated for a total of 34 years. The total quantity of LBP debris disposed in landfills is therefore equal to

$$\left(15 \text{ yrs} \times (0.88 \times 10^6 + 0.53 \times 10^6 \text{ tons/yr})\right) + \left(19 \text{ yrs} \times (0.88 \times 10^6 + 0.34 \times 10^6 \text{ tons/yr})\right)$$
$$= 21.2 \times 10^6 + 23.2\ 10^6 = 44.3 \times 10^6 \text{ tons}$$

Using an LBP debris density of 1.0 tons/yd^3, the LBP debris quantity has a volume of

$$44.3 \times 10^6 \times \frac{907 \text{ kg/ton}}{1186 \text{ kg/m}^3} = 3.39 \times 10^7 \text{ m}^3$$

Assuming, as before, that the LBP debris is disposed in half of the total 1800 C&D landfills, the average volume fraction of LBP debris in the landfills that receive the waste is

$$\frac{3.39 \times 10^7 \text{ m}^3}{6.04 \times 10^8 \text{ m}^3} \times \frac{1800}{900} = 0.112$$

Compared to the waste amounts presented in Section 2.3.1, the above estimates indicate both a greater amount of C&D waste per landfill, as well as an increased proportion of LBP debris per landfill. The greater amount of LBP debris per landfill translates into a proportionally longer duration of the leachate pulse.

2.4.2.2 Waste Disposed in Municipal Landfills

From the available information on number of municipal landfills, the area distribution and waste volume, an effective waste depth of municipal landfills was calculated. The aggregate area of all municipal landfills was estimated from the data in Table 2.4, as follows: The landfill areas within each frequency interval in Table 2.4 were taken to be uniformly distributed, with an average area equal to the midpoint area of each interval. For example, within the first frequency interval, there are 0.1 x 6034 = 603 landfills, with an average area of 0.5 x (4006.5 + 8094) = 6050 m^2. The total area in this interval is calculated as 603 x 6050 = 3650729.9 m^2. The number of landfills and total area in each interval were calculated in this manner, and then added together to yield an estimated aggregate area of all municipal landfills of $3.5x10^9$ m^2. Next, the effective waste depth is calculated as 9.7E-3 m (ratio of waste volume 3.39×10^7 to area 2.06×10^8). The estimation of waste volume is discussed in the previous section. The waste fraction was set 1.0 to maintain mass balance.

The effective waste depth is not the true landfill depth. It represents the portion of the true landfill depth the LBP debris occupies. As discussed in section 2.2 , landfill depth is an input parameter in the calculation of the source duration. Equation 2 can be rewritten as:

$$t_p = \frac{M_w \cdot C_w}{A \cdot I \cdot C_L} \qquad (3)$$

where M_w, is the mass of the waste $(A \cdot d \cdot F_{hw} \cdot P_w)$, $A \cdot d$ is the volume of the landfill and F_{wh} is the volume of the L.P. debris over the volume of the landfill. Therefore, by using a waste fraction of 1.0 and an effective waste depth the mass could be conserved as show in equation (4).

$$M_w = A \cdot \frac{V_w}{A} \cdot 1 \cdot P_w \qquad (4)$$

The equation simplifies to volume of the L.P. debris (V_w) x the density of the waste (Pw).

2.5 Waste and Leachate Concentration

Information of the total and leaching concentrations of lead in LBP debris was obtained from reported sampling and analysis data of LBP debris, reported in the Background Document for the Lead Abatement Waste Study (SAIC, 1994). This report presents total lead concentrations, TCLP (Toxicity Characteristic Leaching Procedure) and SPLP (Synthetic Process Leaching Procedure) leaching concentrations in various types of LBP debris. The data are presented in Table 2.7. In LBP, lead exists in a variety of resin formulations that would tend to encapsulate it and render it relatively unreactive. It is often difficult to determine the geochemical environment of a landfill without actually knowing all the

wastes that are disposed at a site. The geochemistry of the landfill dictates the rate at which lead leaches out. Therefore, in order to model the most plausible and realistic scenario, the lead leachate concentrations reported by using SPLP and TCLP tests on actual waste samples were used as input in the case of C&D and municipal landfills, respectively. In general, TCLP values are greater than SPLP values. SPLP is more suitable in the case of nonmunicipal landfills because the environment in municipal landfills is conducive in the generation of organic acid leachate similar to the acetic acid leachate used in the TCLP. Since lead is more soluble in acidic leaching medium generated in the presence of organic wastes, TCLP values in general, tend to be greater than SPLP values. Both SPLP and TCLP estimate the actual lead that is released from the paint. The total lead content of the paint residue affects the estimate of the length of time leaching may continue. Once lead leaches into groundwater, the groundwater chemistry dictates its mobility. The variability of groundwater chemistry is simulated by the four master variables, as discussed in section 2.1.

In order to obtain a representative weighting for the different materials that compose LBP debris, the contributions of different substrates were converted to 'door equivalents'. It was assumed that the mass of one door is equivalent to 5 windows, 5 door casings, or 42 linear feet of soffit. To calculate composite percentages, all of the doors and 'door equivalents' were added, and the percentage contribution of each substrate calculated. The following weighting was obtained in this manner:

Doors:	44%
Soffits:	41%
Door casings:	9%
Windows:	5%
Shelves and misc.:	1%

Table 2.7 Total, TCLP and SPLP concentrations of Lead in LBP debris (from SAIC, 1994)

Sample	Substrate	Total (mg/kg)	TCLP (mg/L)	SPLP (mg/L)
EV-01	window	5.4	0.05	0.05
EV-02	window	34100	32.2	1.2
EV-03	window	3930	40.6	1.5
EV-04	window	12300	19.6	1.3
EV-05	door	58.4	0.25	0.05
EV-06	door	1370	0.22	0.05
EV-07	door	48.7	0.05	0.05
EV-08	door	2950	3.8	0.24
MIL-01	door	1900	7.6	0.4
MIL-02	door	2680	6.6	0.76
MIL-03	door	2250	7.4	0.92
MIL-04	door	2230	22	0.57
MIL-05	shelf	2720	34.6	2
MIL-06	shelf	235	0.85	0.05
MIL-07	window	4240	18.3	1.1
MIL-08	window	14700	19.5	1.4
MIL-09	window	1100	5.3	0.67
MIL-10	door casing	2250	59.1	13.9
MIL-11	door casing	7290	32.7	4.6
MIL-12	door casing	16100	17.3	0.77
MIL-13	door casing	12900	1.8	1.2
MIL-14	door casing	11100	1.8	1.4
MIL-15	window	2480	1.9	1.6
MIL-16	door casing	2600	0.21	0.18
MIL-17	door casing	4120	0.36	0.15
MIl-18	door casing	3840	2.6	4.1
MIl-19	soffit	7950	9.3	1.8
MIl-20	soffit	10800	15.2	1.4
MIl-21	soffit	9710	28.7	3
MIL-22	soffit	25100	20.9	1.8
BALT-01	int. trim	58000	143	7.2
BALT-02	baseboard	17600	65.1	8.3
BALT-03	baseboard	12300	36.7	3
BALT-04	window	51500	53.3	6.6
SF-01	ext. wood	2010	3	0.39
SF-02	ext. wood	10900	72.8	6.5
SF-03	int. wood	1430	0.25	0.05
SF-04	int. trim	423	3	0.14
AL-01	int. wood	13800	18.4	1.2
AL-02	tongue & gr.	5080	32.7	12
AL-03	plaster	4100	0.73	0.05
AL-04	fence	31.8	0.05	0.05

Table 2.7 Total, TCLP and SPLP concentrations of Lead in LBP debris (from SAIC, 1994) (continued)

Sample	Substrate	Total (mg/kg)	TCLP (mg/L)	SPLP (mg/L)
AL-05	door	16400	31.4	3.4
AL-06	door	4330	12.3	1.6
AL-07	int trim	6250	22.6	9.3
AL-08	door	5980	29.5	3.9
AL-09	door	3770	27	0.05

The actual leaching concentration emanating from C&D landfills is uncertain, due to a variety of factors, including variations in the composition of LBP debris and variations in the lead content of individual waste components. The present modeling analysis attempted to capture this variation by treating the leaching concentration, and the average total LBP debris lead concentration as random parameters. A frequency distribution of the leaching concentration was generated from the data in Table 2.7 in the following manner: It was assumed that the relative contributions of each of the substrate categories in table are the same in each case, but that the leaching concentration within each substrate category are variable. Using a bootstrap sampling procedure, i.e., random sampling with replacement, 2000 random values of LBP debris leaching concentration were generated. The 2000 values were taken to represent the nationwide frequency distribution of the lead leaching concentration. In each of the 2000 realizations, one random value of the SPLP leaching or TCLP concentrations presented in Table 2.7 was drawn for each substrate category (one value for doors, one value for soffits, etc.). A weighted average of these values was computed by using the weights assigned for each substrate category. The resulting frequency distribution of LBP debris leaching concentration is presented in Tables 2.8 and 2.9 and depicted in Figures 2.2 and 2.3.

Table 2.8 Cumulative Frequency Distribution of SPLP Leaching Concentration of LBP Debris

Leaching Concentration (mg/L)	Cumulative Probability
0.591	0.0
0.783	0.05
0.866	0.10
0.930	0.15
0.986	0.20
1.063	0.25
1.144	0.30
1.215	0.35
1.280	0.40
1.350	0.45
1.422	0.50
1.510	0.55
1.624	0.60
1.743	0.65
1.918	0.70
2.107	0.75
2.302	0.80
2.487	0.85
2.666	0.90
2.961	0.95
3.616	0.99
4.107	1.00

Table 2.9 Cumulative Frequency Distribution of TCLP Leaching Concentration of LBP Debris

Leaching Concentration (mg/L)	Cumulative Probability
3.90	0.0
6.81	0.05
8.00	0.10
9.09	0.15
10.07	0.20
10.93	0.25
11.87	0.30
12.60	0.35
13.46	0.40
14.34	0.45
15.33	0.50
16.21	0.55
17.06	0.60
18.01	0.65
18.96	0.70
20.14	0.75
21.22	0.80
22.50	0.85
24.27	0.90
26.21	0.95
29.40	0.99
33.25	1.00

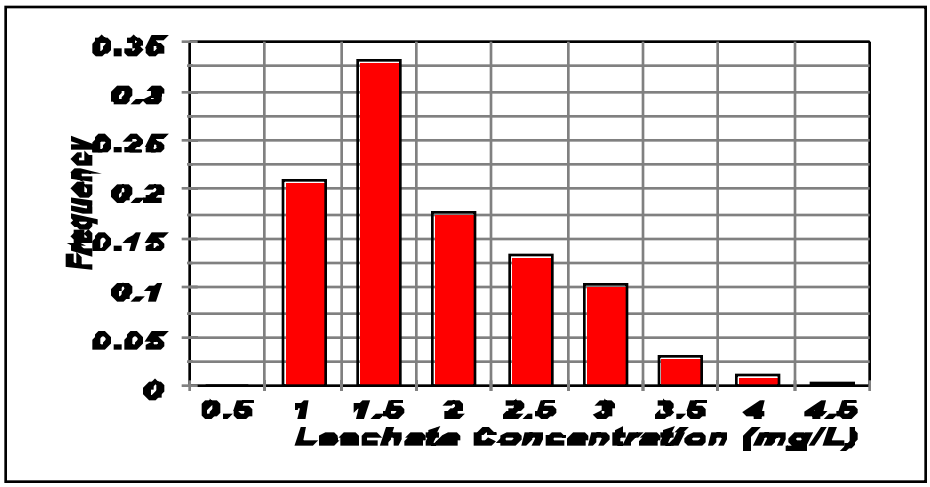

Figure 2.2 Frequency Distribution of LBP Debris SPLP Leaching Concentration.

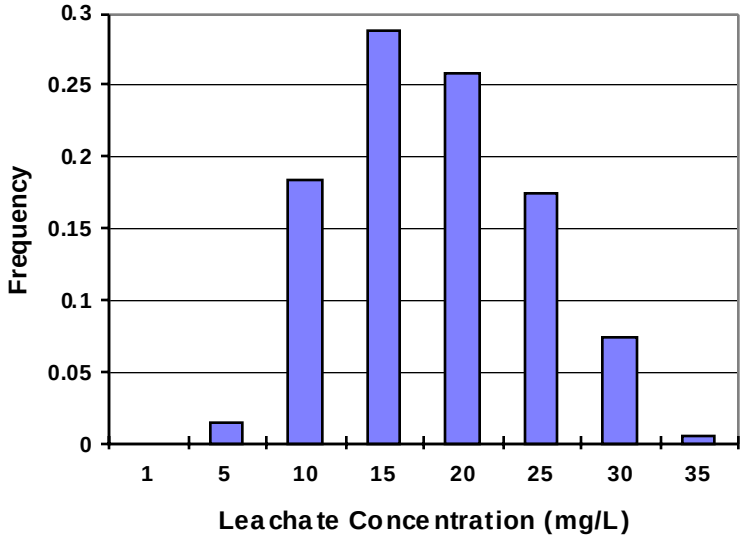

Figure 2.3 Frequency Distribution of LBP Debris TCLP Leaching Concentration.

For the modeling analysis of LBP debris disposed in C&D and municipal landfills, it was assumed that the total lead concentration in the LBP debris varies proportionally to the leaching concentration, with a constant ratio of waste-to-leaching concentration. This ratio was calculated from the ratio of the weighted average waste concentration in Table 2.5, to the weighted average leaching concentration. In the case of SPLP the ratio is 4997 and in the case of TCLP it is 509.

3.0 RESULTS

This section presents results of the groundwater modeling. Section 3.1 presents the results based on the waste quantity estimates for lead abatement debris as described in Section 2.3.1. The results of the analysis for combined LBP abatement debris and painted demolition debris are presented in Section 3.2.1. Section 3.2.2 addresses the risks associated with the combined lead abatement debris and painted demolition debris managed in municipal landfills. The municipal landfill scenario was investigated as an alternative way of managing wastes containing LBP debris. Section 3.3 addresses uncertainties in the fate and transport of lead and Section 3.4 discusses numerical stability of EPACMTP Monte Carlo concentration populations under the tail region of the concentration distribution. Results of the analysis are summarized in Section 4.0.

3.1 Results Based on LBP Abatement Debris in C&D Landfills

Results of the groundwater pathway analysis for lead using LBP abatement debris quantity estimates are summarized in Tables 3.1 - 3.5 and Figures 3.1 - 3.3. These results are based on a simulation run comprising 10,000 Monte Carlo realizations. Table 3.1 show the cumulative distribution of the groundwater exposure concentrations of lead obtained from the EPACMTP Monte Carlo analysis. Only the upper 20 percentile values are shown in this table. The results showed that the lead in groundwater exceeded the Federal Safe Drinking Water Action Level of 0.015 mg/L for tap water in about 4.5% of the cases. The results also indicated that concentrations at the receptor well were 1.00 E-10 or lower in 80% of the cases. These low concentrations are due to the fact that the peak concentration of lead did not reached the well within the allowed 10,000 year time frame in more than 70% of the Monte Carlo realizations. This is shown in Table 3.2 which presents the cumulative frequency distribution of the arrival times of the peak receptor well concentrations. Table 3.2 shows that the peak receptor well concentration was reached within 1,000 years in less than 1% of the cases and that the arrival time exceeds 10,000 years in about 70% of the Monte Carlo realizations. The shortest travel time in the 10,000 Monte Carlo runs was on the order of 112 years. This behavior was due to strong sorption of lead in the both unsaturated and saturated zones, which greatly retards the movement of the constituent and, thus, reduced the peak receptor well concentrations.

Figure 3.1 and Table 3.3 present the frequency distribution of peak receptor well concentration values within the 10,000 year exposure time horizon. The peak concentration did not exceed 0.005 mg/L (one-third of the 0.015 mg/L, the safe drinking water action level for tap water) in nearly 95% of the cases The action level (o.o15 mg/L) itself was exceeded in only 4.4% of the cases. The highest peak receptor well values encountered in the 10,000 realizations was 2.0 mg/L.

A more detailed analysis of those cases in which the 0.015 mg/L level was exceeded is presented in Figures 3.2 and 3.3 and Tables 3.4 and 3.5. Figure 3.2 and Table 3.4 show the frequency distribution of the 4.4% of the peak receptor well concentrations which exceed the action level for lead. Figure 3.3 and Table 3.5 present the arrival time distribution of the peak receptor well concentrations which exceed lead action level. It can be seen in Figure 3.3 and Table 3.5 that most of the exceedences occur after 5,000 or more years; the number of exceedences occurring within 2,000 years account for less than 0.2% of all cases. Under the assumption that there are 900 landfills receiving LBP debris, this corresponds to less than 2 cases nationwide in which the peak receptor well concentration of lead would exceed its action level within 2,000 years from present date.

Table 3.1 Cumulative Distribution of Peak Receptor Well Concentration of Lead in LBP Abatement Debris Managed in C & D Landfills.

Cumulative Probability %	Concentration (mg/L)
80	1.00E-10
85	4.10E-09
90	3.80E-06
94	1.50E-03
95	5.20E-03
95.63	1.50E-02
96	2.60E-02
97	8.20E-02
98	1.97E-01
99	4.30E-01
100	2.10E + 00

Table 3.2 Probability Distribution of Arrival Time(within 10,000 y) of the Peak Receptor Concentration of Lead in LBP Abatement Debris Managed in C & D landfills.

Cumulative Probability %	Travel Time (years)
0.01	112
1	1183
5	3678
10	5599
15	6419
20	7832
25	9285
29	9997

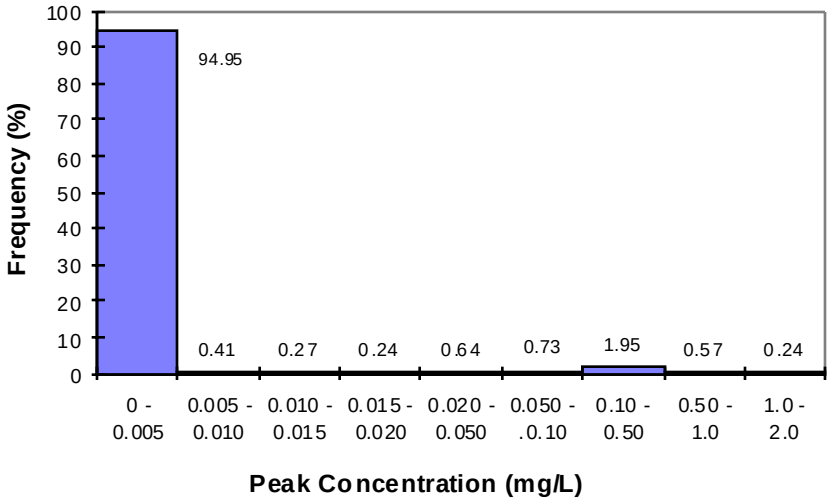

Figure 3.1 Frequency Distribution of Receptor Well Peak Concentration of Lead in LBP Abatement Debris Managed in C & D Landfills.

Table 3.3 Frequency Distribution of the Peak Receptor Well Concentration of Lead in LBP Abatement Debris Managed in C&D Landfills.

Peak Concentration (mg/L)	Frequency (% of total)
0 - 0.005	94.95
0.005 - 0.010	0.41
0.010 - 0.015	0.27
0.015 - 0.020	0.24
0.020 - 0.050	0.64
0.050 - 0.10	0.73
0.10 - 0.50	1.95
0.50 - 1.0	0.57
1.0 - 2.0	0.24

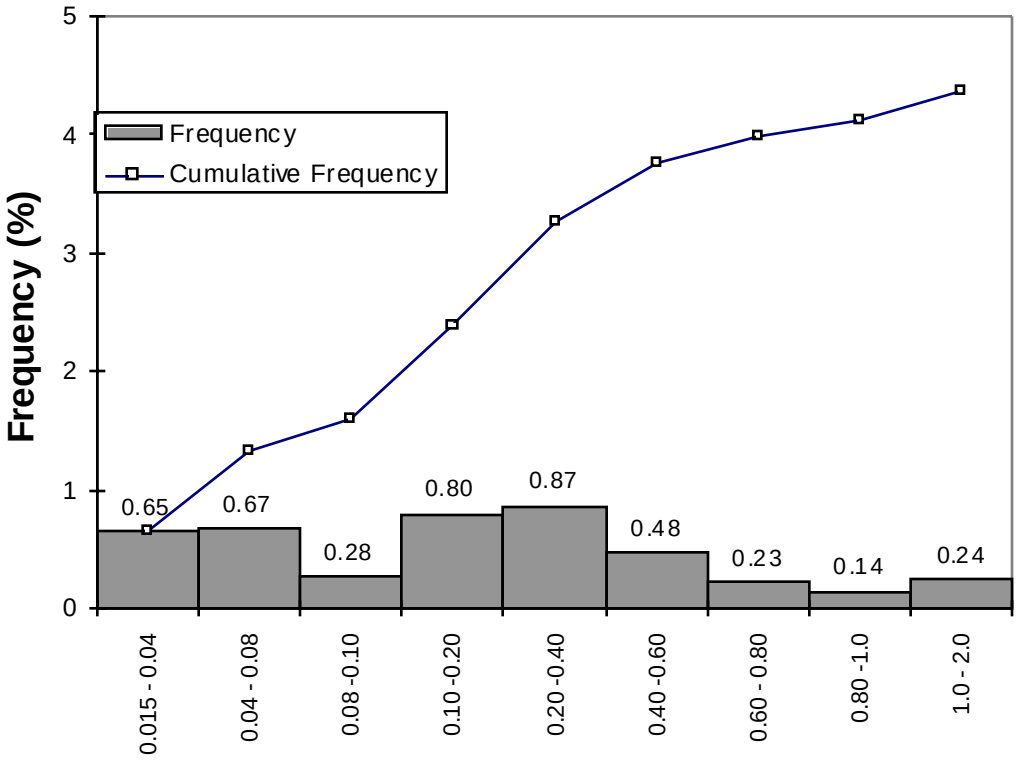

Figure 3.2 **Frequency Distribution of the Peak Receptor Well Concentrations which Exceed Lead Action Level (0.015 mg/L) for LBP Abatement Debris in C&D Landfills.**

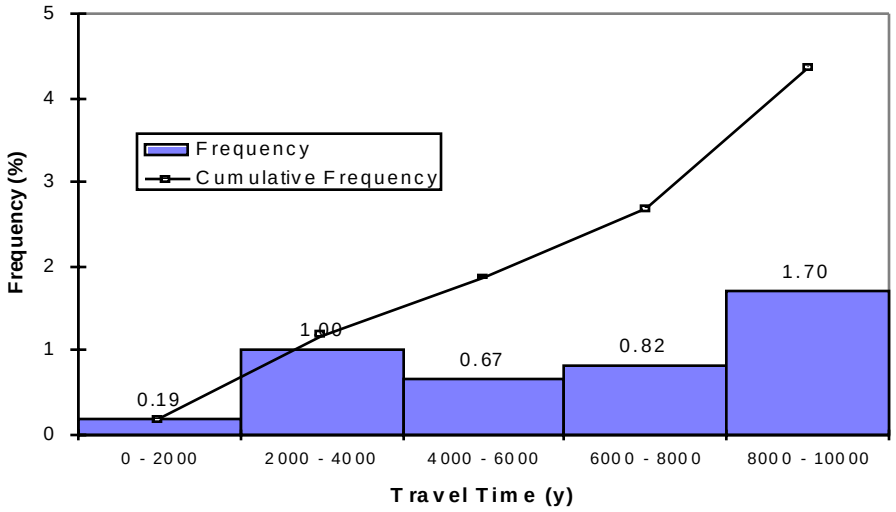

Figure 3.3 **Frequency Distribution of Arrival Time of the Peak Receptor Well Concentrations of Lead Exceeding the Action Level (0.015 mg/L) for LBP Abatement Debris in C& D Landfills.**

Table 3.4 Frequency Distribution of Peak Receptor Well Concentrations of Lead which Exceed Action Level (0.015 mg/L) for LBP Abatement Debris in C& D Landfills.

Peak Concentration (mg/L)	Frequency (% of total)	Cumulative Frequency (% of total)
0.015 - 0.04	0.650	0.650
0.04 - 0.08	0.670	1.320
0.08 - 0.10	0.280	1.600
0.10 - 0.20	0.800	2.400
0.20 - 0.40	0.870	3.270
0.40 - 0.60	0.480	3.750
0.60 - 0.80	0.230	3.980
0.80 - 1.0	0.140	4.120
1.0 - 2.0	0.240	4.360

Table 3.5 Frequency Distribution of Arrival Time of the Peak Well Concentrations of Lead Exceeding Action Level (0.015 mg/L) for LBP Abatement Debris in C&D Landfills.

Travel Time	Frequency (% of total)	Cumulative Frequency (% of total)
0 - 2000	0.18791	0.18791
2000 - 4000	1.00073	1.18864
4000 - 6000	0.66861	1.85725
6000 - 8000	0.81719	2.67444
8000 - 10000	1.7043	4.37

3.2 Results of the Analysis for Lead Abatement Debris Plus Painted Demolition Debris

The groundwater modeling analysis for LBP abatement debris plus painted demolition debris was performed considering two waste management scenarios: C& D landfills and municipal landfills. Ten thousand Monte Carlo realizations were performed for each scenario. The results for the two scenarios are discussed below. Tables 3.11 through 3.16 present results for the scenario in which lead abatement plus painted demolition debris were modeled in municipal landfills.

3.2.1 Results for C&D Landfills

Results of Lead based paint analysis for lead abatement debris plus painted demolition debris managed in C&D landfills are presented in Tables 3.6 through 3.10. Table 3.6 presents a tabulated frequency distribution of the peak receptor well concentration values which indicated that the safe drinking water action level for lead in tap water (0.015 mg/L) was at 95.15-th percentile of the cumulative peak well concentration distribution. This value is close, but slightly higher than the corresponding value for lead abatement wastes in Table 3.1. This difference is due to increased waste quantity (abatement debris plus demolition LBP debris) in the landfill.

Table 3.6 Probability Distribution of the peak Receptor Well Concentration of Lead for LBP Abatement Debris Plus Painted Demolition Debris in C&D Landfills.

Cumulative Probability %	Concentration (mg/L)
70	7.84E-14
75	7.69E-12
80	3.15E-10
85	1.10E-08
90	1.12E-05
95	1.25E-02
95.15	1.50E-02
96	4.89E-02
97	1.26E-01
98	2.75E-01
99	5.74E-01
100	2.21E + 00

Table 3.7 presents the probability distribution of arrival times for the peak receptor well concentrations. The data indicated that the arrival time of the peak concentrations at the receptor well was less than the 10,000 year cut-off time in approximately 33 percent of the cases. In comparison with Table 3.2, it can be seen that the distribution exhibits a slight shift towards larger time values. A detailed analysis of the cases in which the federal safe drinking water level (0.015 mg/L) was exceeded at the receptor well is presented in Tables 3.8 and 3.9. These tables correspond to Tables 3.4 and 3.5 in Section 3.1. Finally, Table 3.10 shows the frequency breakdown of the peak receptor well concentrations above the groundwater action level for lead. This table corresponds to Table 3.3 in Section 3.1. Appendix B presents the frequency distribution of key Monte Carlo input parameters for all cases where the peak receptor well concentration was above the action level for lead.

Table 3.7 Probability Distribution of Arrival Time (within 10,000 y) of the Peak Receptor Concentration of Lead in LBP Abatement Debris Plus Painted Demolition Debris in C&D landfills.

Cumulative Probability %	Travel Time (years)
0.1	57.73
1	1276.00
5	4421.00
10	5621.00
15	6160.00
20	7500.00
25	8762.00
30	9603.00
32.65	9999.00

Overall, the results of incorporating both abatement debris and painted demolition debris in the analysis were very close to the results of lead abatement debris presented in Section 3.1. Although the total amount of LBP debris in landfills was more than doubled (44.3 million tons vs 19.1 million tons) when wastes from abatement and demolition processes were combined, the effect on the peak exposure concentration was very small. This may be attributed to the fact that lead is strongly sorbed in the soil and, therefore, would take longer than the 10,000 year cut-off time used in this analysis to reach the receptor well. As shown in Tables 3.2 and 3.7 the percentage of cases in which the arrival time of the peak receptor concentration was less than 10,000 years was 29 and 33 percent respectively for LBP abatement debris and LBP abatement plus demolition debris scenarios. The increased waste quantity in the latter case may lead to a layered waste in the landfill. However, this will result primarily in an increased leachate pulse duration, but the modeled leachate concentration itself does not change over time. The data in Tables 3.1 and 3.6 indicated that the increased waste quantity had a small incremental effect on the distribution of the peak concentration at the receptor well.

Table 3.8 Frequency Distribution of the Arrival Time of the Peak Well Concentrations of Lead Exceeding Lead Action Level for LBP Abatement Debris plus Painted Demolition Debris in C&D Landfills.

Travel Time	Frequency (% of total)	Cumulative Frequency (% of total)
0 - 2000	0.06	0.06
2000 - 4000	0.45	0.51
4000 - 6000	0.85	1.36
6000 - 8000	0.78	2.14
8000 - 10000	2.74	4.88

Table 3.9 Frequency Distribution of the Peak Well Concentrations Exceeding Lead Action Level (0.015 mg/L) for LBP Abatement Debris Plus Painted Demolition Debris in C&D Landfills.

Peak Concentration (mg/L)	Frequency (% of total)	Cumulative Frequency (% of total)
0.015 - 0.04	0.690	0.690
0.04 - 0.08	0.680	1.370
0.08 - 0.10	0.200	1.570
0.10 - 0.20	0.880	2.450
0.20 - 0.40	0.900	3.350
0.40 - 0.60	0.580	3.930
0.60 - 0.80	0.280	4.220
0.80 - 1.0	0.270	4.490
1.0 - 3.0	0.400	4.880

Table 3.10 Frequency Distribution of the Peak Receptor Well Concentration of Lead in LBP Abatement Debris Plus Painted Demolition Debris in C&D Landfills.

Peak Concentration (mg/L)	Frequency (%)
0 - 0.005	94.4
0.005 - 0.10	0.47
0.010 - 0.015	0.26
0.015 - 0.020	0.17
0.020 - 0.050	0.73
0.050 - 0.10	0.67
0.10 - 0.50	2.07
0.50 - 1.0	0.83
1.0 - 3.0	0.37

3.2.2 Results for Municipal Landfills

The results of the analysis for LBP abatement debris and painted demolition debris managed in municipal landfills are presented in Tables 3.11 through 3.15. The municipal landfill scenario was examined as an alternative waste management option. Table 3.11 presents a tabulated frequency distribution of the peak receptor well concentrations for lead. It can be seen that federal safe drinking water level for lead (0.015 mg/L) is at the 98.27-th percentile of the distribution. The percentage of cases that exceed the action level for lead decreased from 4.85 in the case of C&D landfills to 1.73 for municipal landfills, despite the fact that the source leachate concentration values (TCLP Table 2.7) used for municipal landfills were higher than those (SPLP Table 2.6) used for C&D landfills. The decrease in the

peak receptor well concentration values is due to the larger number of municipal landfills (6034), compared to C&D landfills (1800), that are available for waste disposal. This means that the quantity of LBP debris per landfill is lower in the municipal landfill scenario.

Table 3.11 Probability Distribution of the peak Receptor Well Concentration of Lead in LBP Abatement Debris Plus Painted Demolition Debris in Municipal Landfills.

Cumulative Probability %	Concentration (mg/L)
90	6.40E-09
91	1.45E-08
93	9.13E-08
94	6.23E-07
95	2.09E-05
97	2.72E-03
98.27	**1.50E-02**
99	3.91E-02
100	7.27E + 00

Table 3.12 presents the probability distribution of arrival time for the peak receptor well concentration of lead. In comparison with Table 3.7 , it can be seen that the distribution exhibits a shift towards larger time values. The reason for this shift in time values and the associated lower receptor well concentrations are due to the lower waste quantity per municipal landfill. A detailed analysis of the cases in which the maximum receptor well concentration of lead exceeded the action level (0.015 mg/l) is presented in Tables 3.13 and 3.14. These tables correspond to Tables 3.8 and 3.9 in Section 3.2.1. Finally, Table 3.15 shows the frequency breakdown of the peak receptor well concentration. This table corresponds to Table 3.10 in Section 3.2.1.

Overall, the results of groundwater analysis for LBP abatement debris plus painted demolition debris managed in municipal landfills, as discussed above, indicated lower groundwater exposure to lead compared to the case in which the same waste is managed in C&D landfills.

Table 3.12 **Probability Distribution of Arrival Time (within 10,000 y) of the Peak Receptor Concentration for Lead Abatement Debris Plus Painted demolition debris in Municipal landfills.**

Cumulative Probability %	Travel Time (y)
0.1	73
1	804
2	1538
5	4024
10	5881
15	6783
20	9457
23.44	10000

Table 3.13 **Frequency Distribution of Arrival Time of the Peak Well Concentrations of Lead Exceeding the Action Level for LBP Abatement Debris plus Painted Demolition Debris in Municipal Landfills**.

Travel Time (y)	Frequency (% of total)	Cumulative Frequency (% of total)
0 - 2000	0.59	0.59
2000 - 4000	0.40	0.99
4000 - 6000	0.26	1.25
6000 - 8000	0.22	1.47
8000 - 10000	0.3	1.77

Table 3.14 Frequency Distribution of f the Peak Well Concentration of Lead Exceeding the Action Level for Lead Abatement Debris Plus Painted Demolition Debris in Municipal Landfills.

Peak Concentration (mg/L)	Frequency (% of total)	Cumulative Frequency (% of total)
0.015 - 0.04	0.77	0.77
0.04 - 0.08	0.44	1.21
0.08 - 0.10	0.11	1.32
0.10 - 0.20	0.26	1.58
0.20 - 0.40	0.1	1.68
0.40 - 0.60	0.05	1.73
0.60 - 0.80	0.02	1.75
0.80 - 1.0	0.01	1.76
1.0 - 3.0	0	1.76
> 3.0	0.01	1.77

Table 3.15 Frequency Distribution of the Peak Receptor Well Concentration of Lead in LBP Abatement Debris Plus Painted Demolition Debris in Municipal Landfills.

Peak Concentration (mg/L)	Frequency (%)
0 - 0.005	97.38
0.005 - 0.10	0.50
0.010 - 0.015	0.35
0.015 - 0.020	0.26
0.020 - 0.10	1.06
0.10 - 0.50	0.38
0.50 - 1.0	0.06
1.0 - 3.0	0
> 3.0	0.01

3.3 Uncertainties Associated with Fate and Transport of Lead

The simulations of lead transport were designed to be reasonably conservative through the use of inherently conservative assumptions. These conservative assumptions tend to compensate for some of the processes that may not be accurately accounted for in the Monte-Carlo simulations. Some of the uncertainties that may be related to subsurface lead transport are discussed below.

For example, the approach for the analysis assumed that all wells located within the down gradient half of a circle drawn within one mile radius around the landfill could be potentially exposed to landfill leachate. This may underestimate potential exposure, because landfill leachate plumes in porous soils (e.g., sand/gravel) where most wells would tend to draw water from generally do not disperse widely and would be expected to be long and narrow in shape. Moreover, the approach also assumed that all wells gather water from the uppermost layer of ground water below the ground surface, where leachate releases from the landfills would most likely be. This may overestimate potential exposure, because many private wells gather water from deeper layers of ground water which may not be exposed to the landfill leachate.

Fractured rocks are regarded as hydrogeologic media with high uncertainty in groundwater flow and solute transport properties. The site-based Monte-Carlo modeling approach used in this study utilizes actual hydrogeologic information from 12 distinct hydrogeologic environments (EPA 1995a). Fractured media are particularly common in some of the twelve hydrogeologic environments such as igneous and metamorphic rocks and dissolution of sedimentary rocks. Common fractures in igneous and metamorphic rocks include fractured granitic rocks, tuffs and fractured basaltic rocks. On the other hand, limestones are known to have cavities within their structures that significantly effect their hydrogologic properties. These two regional hydrogeologic environments are present in 130 waste management sites out of the total 790 industrial Subtitle D Landfill waste management facilities in EPA's database. This corresponds to approximately 25 percent of the facilities. Therefore, it is plausible that lead groundwater exposure risks associated with landfills located in areas where fractured hydrogeologic environments are dominant may be underestimated. For example, the presence of dissolution channels in limestone could provide a very rapid migration pathway for contaminants and thus lead to higher receptor well concentrations. That scenario is relatively unlikely, but is accounted for in the extreme values in the distributions of hydraulic and adsorption properties used in the Monte Carlo simulations.

There are some geologic features not explicitly included in the EPACMTP regional hydrogeologic environments. For example, macro pores in the vadose zone, and epikarstic zones at the top of bed rocks. Flow and transport in individual macro pores normally occurs in a relatively small spatial scale. The spatial dimensions of waste management units are often in the vicinity of tens of thousands of square meters. When all the chemical fluxes through all the macro pores and soil matrix underneath the waste management units are spatially averaged, the mean fluxes are approximately equal to the chemical fluxes determined using the infiltration rates over the respective waste management units. For epikarstic zones where the hydraulic conductivity may be greater than those in the subjacent zones, no provision is available in EPACMTP; however, fractures or solution channels in several epikarstic zones have been found to be sediment-filled and may not be more conductive than the lower zones.

Currently, sorption is allowed to occur in the fractured rocks under the assumption that they behave like equivalent porous media. Lead can be adsorbed to clay particles, calcium carbonate in limestone, and silicates in igneous rocks. The exposed adsorption sites in fractured media are fewer than

those in porous media, therefore it is possible that adsorption may be overestimated in fractured rocks. However, it must be pointed out here that another retardation mechanism is active in these rocks via matrix diffusion/uptake (intergranular and intra granular diffusion). Parameters relating to this type of diffusion have not been well quantified. Matrix diffusion retardation effects are not included in EPACMTP. Therefore, lead loss due to this phenomenon is conservatively excluded. Any overestimates of adsorption effect may be more than offset by underestimates of matrix diffusion retardation effects.

3.4 Numerical Stability of the Population in the Tail of the Peak Receptor Well Concentration Distribution

Numerical stability is the observed numerical changes in the characteristics (i.e., mean, variance, percentiles) of the Monte Carlo simulation output distribution as the number of simulations increases. Depending on the structure of the model and the distribution of input parameters, some outputs stabilize quickly, while other model outputs require more simulations before stabilizing. Ideally, Monte Carlo simulations should be repeated using several non-overlapping subsequence to check for stability and repeatability (USEPA, 1995a).

The stability of the Monte Carlo simulations used for the lead-based paint risk analysis was evaluated for both the municipal and construction and demolition landfill scenarios. To evaluate the stability of the tail of the output distributions (receptor well concentrations), a Monte Carlo simulation with 30,000 iterations was performed for each of the two scenarios. The stability of the tail was then evaluated with the following procedure:

1. 1000 sets of N simulations (N = 500 - 20,000) were selected randomly from the set of 30,000 simulations.
2. 90^{th}, 95^{th}, and 99^{th} percentile well concentrations were selected from each of the 1000 sets of N simulations.
3. Statistical calculations were performed on the 1000 values of 90^{th}, 95^{th} , and 99^{th} percentile well concentrations to generate mean and standard deviations for each "N simulations".
4. The mean 90^{th}, 95^{th}, and 99^{th} receptor well concentrations were plotted as a function of N simulations to assess the value of N at which the tail of the distribution became stable, i.e., the receptor well concentration no longer varies as a function of N (see Appendix B).
5. The mean receptor well concentration plots are shown in Appendix B. Plots are shown in Appendix B with both the 95% confidence interval of the mean and the 95% confidence interval of the population.

Figures B1 through B12 in Appendix B show that for the 90^{th} percentile well concentrations the output becomes very stable at about 10,000 simulations (Figures B.1 and B.7). As the well concentration percentile increases to 95, stability occurs at 12,000 simulations (Figures B.2 and B.8) and at the 99^{th} percentile, stability occurs at about 14,000 simulations (Figures B.3 and B.9). Figures B.4 - B.6 and Figures B.10 - B.12 demonstrate that the confidence interval of the mean is very narrow at greater than 4000 simulations.

A comparison of the mean 90^{th}, 95^{th}, and 99^{th} receptor well concentrations at high values of N, or the point of stability, are shown along with the standard deviation in Tables 3.16 (Municipal Landfill Scenario) and 3.17 (Construction and Demolition Landfill Scenario). These Tables indicate that although

the mean receptor well concentration seems to stabilize well before 15,000 simulations, the standard deviation continues to decrease past 20,000 simulations.

Table 3.16 Variation of Mean Well Concentration (mg/L) and Standard Deviation with Number of Iterations for Municipal Landfills.

No. of Simulations	90th Percentile		95th Percentile		99th Percentile	
	$\overline{C_w}$	σ	$\overline{C_w}$	σ	$\overline{C_w}$	σ
10,000	7.16e-09	1.85e-09	1.73e-05	1.24e-05	3.65e-02	6.54e-03
12,000	7.16e-09	1.73e-09	1.67e-05	1.06e-05	3.66e-02	5.89e-03
14,000	7.14e-09	1.55e-09	1.64e-05	9.39e-06	3.66e-02	5.7e-03
16,000	7.08e-09	1.44e-09	1.61e-05	8.56e-06	3.64e-02	5.21e-03
18,000	7.10e-09	1.37e-09	1.60e-05	7.57e-06	3.65e-02	4.89e-03
20,000	7.07e-09	1.25e-09	1.60e-05	7.28e-06	3.65e-02	4.79e-03

Table 3.17 Variation of Mean Well Concentration in (mg/L) and Standard Deviation of Well Concentrations for Construction and Demolition Landfills.

No. of Simulations	90th Percentile		95th Percentile		99th Percentile	
	$\overline{C_w}$	σ	$\overline{C_w}$	σ	$\overline{C_w}$	σ
10,000	6.51e-07	3.41e-07	2.42e-03	9.22e-04	4.41e-01	4.84e-02
12,000	6.41e-07	3.16e-07	2.39e-03	8.36e-04	4.40e-01	4.56e-02
14,000	6.26e-07	2.77e-07	2.37e-03	7.59e-04	4.41e-01	4.25e-02
16,000	6.24e-07	2.64e-07	2.36e-03	7.21e-04	4.41e-01	4.04e-02
18,000	6.14e-07	2.48e-07	2.34e-03	6.65e-04	4.41e-01	3.79e-02
20,000	6.12e-07	2.29e-07	2.35e-03	6.18e-04	4.42e-01	3.51e-02

4.0 SUMMARY

EPA modeled two disposal scenarios: LBP abatement debris in C&D landfills and LBP abatement debris plus painted demolition debris in C&D and municipal landfills. In the case of C&D landfills, EPA modeled a conservative disposal scenario where it was assumed that debris would be managed in 900 landfills (i.e., one-half of the existing 1,800 C&D landfills). Except for the total number of landfills (1,800), EPA did not have any site-specific data for commercial C&D landfills. EPA, therefore, assumed that the overall location distribution of C&D landfills and Subtitle D Municipal landfills in the United States are similar to the general distribution of the industrial Subtitle D landfills, and therefore, used 1986 Industrial D Landfill survey information to develop the hydrogeologic information, climatological data, and soil characteristics for use as input to the groundwater modeling analyses. A similar approach was used in the case of municipal landfills. Using the EPA's Composite Model for Leachate Migration with Transformation Products (EPACMTP) in conjunction with the MINTEQA2 metal speciation model, EPA modeled the lead leachate migration from the bottom of unlined C&D landfills and from the bottom of unlined Municipal Solid Waste Landfills (MSWLF) in the subsurface environment. The models and the modeling approach used here were also used in the HWIR proposal.

The groundwater modeling analysis used a Monte Carlo approach to determine the probability distribution of peak receptor well concentrations over a 10,000 year exposure period. The MINTEQA2 model was used to determine subsurface lead sorption isotherms under a range of environmental conditions, i.e., variation in pH and other factors controlling the subsurface mobility of lead. These data were used together with LBP debris volumes, hydrogeological, climatological and soil characteristics to generate 10,000 Monte Carlo realizations of the exposure concentration of lead in groundwater. The latter was defined as the peak concentration at a receptor placed anywhere down gradient from the landfill, within a one mile radial distance from the down gradient edge of the waste unit. In all of the scenarios, the source is conservatively assumed to leach at a constant concentration with time until all the waste in the landfill is depleted.

The modeling approach used for two disposal scenarios (scenario 1: LBP abatement debris disposal in C&D landfills and scenario 2: LBP abatement debris plus painted demolition debris disposal in both C&D and municipal landfills). The results for C&D landfills suggest the following:

1) The peak receptor well concentration would be between zero and 0.005 mg/l (one-third of the 0.015 mg/L safe drinking water action level for lead in tap water) in approximately 95 percent of receptor wells, located anywhere downstream of a C&D landfill, when LBP abatement debris was the only lead source. [The peak receptor well concentration would be below 0.005 mg/l in little more than 94 percent of the cases if LBP debris from abatement, demolition (including R&R activities) was collectively managed in C&D landfills.]

2) The modeling results show that only in less than 4.4 percent of the cases would the receptor well lead concentration exceed the regulatory action level (15 parts per billion) over a time horizon of 10,000 years and that most of these exceedences would occur 5,000 years after disposal of LBP debris from abatement activities. The receptor well concentration would exceed the regulatory action level over atime horizon of 10,000 years in only about 4.8 percent of cases if painted demolition debris (including R&R activity) were disposed along with LBP abatement debris in C&D landfills.

3) The modeling results also show that the federal safe drinking water action level for lead in the tap water (0.015 mg/L) was not exceeded at the receptor well during the first 500 years and between 500 and 1,000 years it may potentially exceed at only one site in 10,000 Monte Carlo simulated sites (i.e., frequency of 0.01%).

The results of the modeling analyses for municipal landfills suggests that the LBP debris can safely be disposed. The results indicate that it would take a few hundred years for lead from LBP debris at a few of the landfill sites to reach drinking water wells in excess of the acceptable drinking water level for lead. Specifically :

1) Over a time horizon of 10,000 years, the peak receptor well concentration would be between zero and 0.005 mg/L (one-third of the drinking water action level) in approximately 97.5 percent of the cases.

2) The concentration of lead in a receptor well would exceed the regulatory action level (0.015 mg/L) in 1.77 percent of cases over a 10,000 years time horizon.

3) The drinking water action level for lead would be exceeded in 0.11% of cases during the first 500 years and 0.20% cases during 500 to 1,000 years.

Thus, this modeling analysis suggests that the potential impact to groundwater resources from the disposal of LBP debris in MSWLFs will be negligible. The calculated risks are less than those calculated for the disposal of LBP debris in C&D landfills because the number of MSWLFs is much greater than the total number of C&D LFs, and MSWLFs tend to be much larger than C&D LFs. Therefore, the lead released from these MSWLFs tend to be less, although TCLP concentration were used as input for the model. (As discussed above, the TCLP lead concentrations are higher than the corresponding SPLP concentrations.)

Thus, the potential impact to groundwater resources from the disposal of lead-based paint debris in these landfills, even under the conservative assumptions, is observed in less than 95%of the Monte Carlo simulated sites. This is well below the 90 percentile protection level used by the Agency in the HWIR proposal (see 60 FR 66343, December 21, 1995). In landfills where groundwater contamination may become a cause of concern, the contamination would only occur over an extremely long period of time. This contrasts dramatically with the known impact of in-place lead contaminated architectural components. Additionally, any control measures such as groundwater monitoring, or leachate collection system would be of slight or no value because the groundwater would be monitored only during the active life to 30-years after closure of a landfill. Also, the use of a liner may not serve as an effective deterrent against the release of leachate, given the very long time frame associated with exposures of lead in nearby drinking water wells.

The results of the analysis performed to investigate the numerical stability of the population in tail of the peak receptor well concentration distribution indicated that the mean 90-th, 95-th and 99-th percentile values of the tail population became stable after performing 10,000 to 14,000 Monte Carlo iterations.

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