3.0 Risk Assessment Overview

This section describes the conceptual framework for the paints listing risk assessment. Section 3.1 presents the conceptual framework for the human health risk assessment. This includes a description of waste streams and waste management practices, fate and transport modeling, exposure assessment, and calculation of protective waste and leachate concentrations. The framework for the probabilistic and deterministic analyses is described in Section 3.2. An overview of the ecological risk assessment is presented in Section 3.3. The ecological risk assessment was designed to evaluate whether waste concentrations determined to be protective of human heath are also protective of the environment.

3.1 Human Health Risk Assessment

The human health risk assessment for the paints listing determination is intended to evaluate nationwide risk to individuals who reside near waste management units (WMUs) used for paint waste disposal. The assumptions as to waste volumes, constituents, and waste management units used in this risk assessment were derived from information EPA gathered from paint facility site visits, EPA databases, and the RCRA 3007 Industry Survey conducted for this listing determination.

3.1.1 Waste Streams

Under RCRA Section 3001(e)(2), EPA was required to make hazardous listing determinations on certain wastes generated during the manufacture of paint in the United States. This determination was to be made within 15 months of enactment of the Hazardous Solid Waste Amendment. EPA, however, did not meet this deadline, and, in March 1989, the Environmental Defense Fund brought suit against the Agency for failure to complete the determination. A settlement agreement was reached that required EPA to finalize a listing determination on the original waste streams and one additional waste stream. The five waste streams EPA is required to examine for this listing determination are solvent cleaning wastes from tank and equipment cleaning operations, water and or caustic wastes from tank and equipment cleaning operations, wastewater treatment sludges, emission control dust, and off-specification products generated during the manufacture of paint.

For the purposes of this risk assessment, these waste streams were categorized into three major groups based on their physical characteristics:

- **Wastewater.** Wastewater includes solvent cleaning waste, wash water, and caustic wash.

- **Combined solid waste.** Combined solid wastes include sludge produced from all wastewater treatment processes, solid off-specification product, and emission control dust.

- **Emission control dust.** Emission control dust includes solids collected in emission control equipment during the handling of raw materials during production (e.g., pigments, resins).

Waste volume data were compiled from the 3007 survey for each waste stream for use in the risk assessment.

3.1.2 Waste Management Units

Three types of WMUs were selected for evaluation based on information gathered by EPA during site visits and from the 3007 survey (see Table 3-1). WMUs included

- Industrial landfills
- Treatment tanks
- Surface impoundments.

Because of the large number of paint manufacturing facilities located across the United States, national databases were used to capture the variation in WMU configurations. The data sources used for characterizing each type of WMU are described briefly below.

3.1.2.1 Landfills. Because most paint manufacturing facilities are located in urban areas, landfills used by a facility are most likely located off-site. Accordingly, landfills that accept waste from off-site sources were characterized using the Industrial D database (U.S. EPA, 1987). This database contains information on facilities that accept waste from on-site and off-site sources. For this assessment, 68 landfill facilities identified in the Industrial D database as accepting waste from off-site sources were selected. These units were used to characterize the distribution of landfills accepting paint wastes for disposal.

Table 3-1. Waste Management Scenarios Modeled

WMU	Waste Stream
Landfill	Emission control dust
Landfill	Combined solids
Treatment tank	Wastewater
Surface impoundment	Wastewater

3.1.2.2 <u>Treatment Tanks</u>. Tanks were identified using a subset of the treatment, storage and disposal facilities (TSDF) survey database (U.S. EPA, 1986) containing information on facilities that are used for treating waste in part or whole from off-site sources. This database contained information on 893 facilities or tanks at the time of this study. To reduce the number of facilities that were modeled, 200 individual tank units were sampled that represented the range of tank size and height in the TSDF database. These 200 tank units were used to characterize the distribution of treatment tanks accepting paint wastes for disposal.

As noted, only contaminant releases from off-site tanks were considered in this risk analysis. This decision was based on a bounding analysis conducted to determine if there was a need to evaluate an on-site tank scenario in the final risk assessment. The bounding analysis results were used to calculate target waste concentrations using a highly conservative modeling scenario. The results of the bounding analysis are presented in Appendix V.

3.1.2.3 <u>Surface Impoundments</u>. Surface impoundment data were contained in the Industrial D database. Unlike landfills, the Industrial D database did not contain any information on whether or not surface impoundments were accepting off-site wastes. It did not have many surface impoundments that were used only for backup during rain events or to contain unusual surges in process wastewater. Because these backup surface impoundments were not consistent with a WMU that would be accepting waste from other facilities, facilities were not considered that had only backup surface impoundments. The resulting database contained 1,903 facilities with surface impoundments. To reduce the number of facilities that were modeled, 200 individual surface impoundment units were sampled that represented the range of surface impoundment sizes in the Industrial D database. These 200 surface impoundment units were used to characterize the distribution of surface impoundments accepting paint wastes for disposal.

3.1.3 Constituents of Concern

Constituents of concern (COCs) associated with these waste streams were identified by EPA. The constituents selected were those already known to be contained in paint waste and for which all information necessary to model human health risk was available. Forty-three constituents of concern were selected–16 metal constituents and 27 organic constituents. Also present in paint mixtures are organometallic complexes. These complexes were not assessed directly due to the lack of toxicity benchmarks and physical and chemical parameters required to perform source and fate and transport modeling. A literature search was conducted to determine the availability of the required modeling parameters for the organometallic complexes. Results of this literature search indicating the lack of available information are presented in Appendix W.

Table 3-2 lists all COCs by Chemical Abstract Service (CAS) registry number, chemical name, and type (i.e., metal, organic).

3.1.4 Site Configuration and Environmental Setting

A single conceptual site layout was used to define the relationship between the WMU and the human or ecological receptor evaluated in this risk assessment. Forty-nine locations

Table 3-2. Constituents of Concern Evaluated in Paints Listing Risk Assessment

Constituent	CASRN	Type
Acrylamide	79-06-1	Organic
Acrylonitrile	107-13-1	Organic
Antimony	7440-36-0	Metal
Barium	7440-39-3	Metal
Benzene	71-43-2	Organic
Butyl alcohol, n-	71-36-3	Organic
Butylbenzylphthalate	85-68-7	Organic
Cadmium	7440-43-9	Metal
Chloroform	67-66-3	Organic
Chromium (III)	16065-83-1	Metal
Chromium (VI)	18540-29-9	Metal
Cobalt	7440-48-4	Metal
Copper	7440-50-8	Metal
Cresol, m-	108-39-4	Organic
Cresol, o-	95-48-7	Organic
Cresol, p-	106-44-5	Organic
Di(2-ethylhexyl)phthalate	117-81-7	Organic
Dibutylphthalate	84-74-2	Organic
Dichloromethane (methylene chloride)	75-09-2	Organic
Dimethylphenol, 2,4-	105-67-9	Organic
Divalent mercury	7439-97-6d	Metal
Ethylbenzene	100-41-4	Organic
Ethylene glycol	107-21-1	Organic
Formaldehyde	50-00-0	Organic
Lead	7439-92-1	Metal
Mercury (elemental)	7439-97-6e	Metal
Methanol	67-56-1	Organic
Methyl ethyl ketone (MEK)	78-93-3	Organic
Methyl isobutyl ketone (MIBK)	108-10-1	Organic
Methyl methacrylate	80-62-6	Organic
Nickel	7440-02-0	Metal

(continued)

Table 3-2. (continued)

Constituent	CASRN	Type
Nickel oxide	1313-99-1	Metal
Pentachlorophenol	87-86-5	Organic
Phenol	108-95-2	Organic
Selenium	7782-49-2	Metal
Silver	7440-22-4	Metal
Styrene	100-42-5	Organic
Tetrachloroethylene	127-18-4	Organic
Tin	7440-31-5	Metal
Toluene	108-88-3	Organic
Vinyl acetate	108-05-4	Organic
Xylene (mixed isomers)	1330-20-7	Organic
Zinc	7440-66-6	Metal

CASRN = Chemical Abstract Service Registry Number.

distributed around the continental United States were used to represent the distribution of paint manufacturing facilities. The same site layout was used to model each of the 49 locations. The locations provided the basis for determining site environmental characteristics.

3.1.4.1 <u>Conceptual Site Layout</u>. Figure 3-1 depicts the conceptual site layout. All receptors are located off-site near the WMU but beyond an intervening area called a buffer area. Beyond the buffer area is a residence, an agricultural field, and a waterbody. Depending on the release mechanisms for a specific WMU, off-site receptors can come into contact with COCs via the air, soil, above- and belowground produce, beef, dairy products, fish, and contaminated groundwater.

3.1.4.2 <u>Regional Environmental Setting</u>. There are over 600 paint manufacturing facilities in the continental United States located in over 40 states (U.S. EPA, 1999); therefore, environmental settings used in this risk assessment are generally representative of broad regions. The primary objective in characterizing a regional environmental setting was to represent the variation in environmental conditions that results from the geographic distribution of paint manufacturing facilities.

Because of the costs associated with the transport of waste material, it was assumed that paint wastes would be disposed of in a geographic distribution similar to the location of paint manufacturing facilities. Thus, the starting point for identifying the locations modeled was information on the geographic distribution of paint manufacturing facilities by state. Figure 3-2 shows the locations of paint manufacturing facilities based on 1997 TRI data and the states

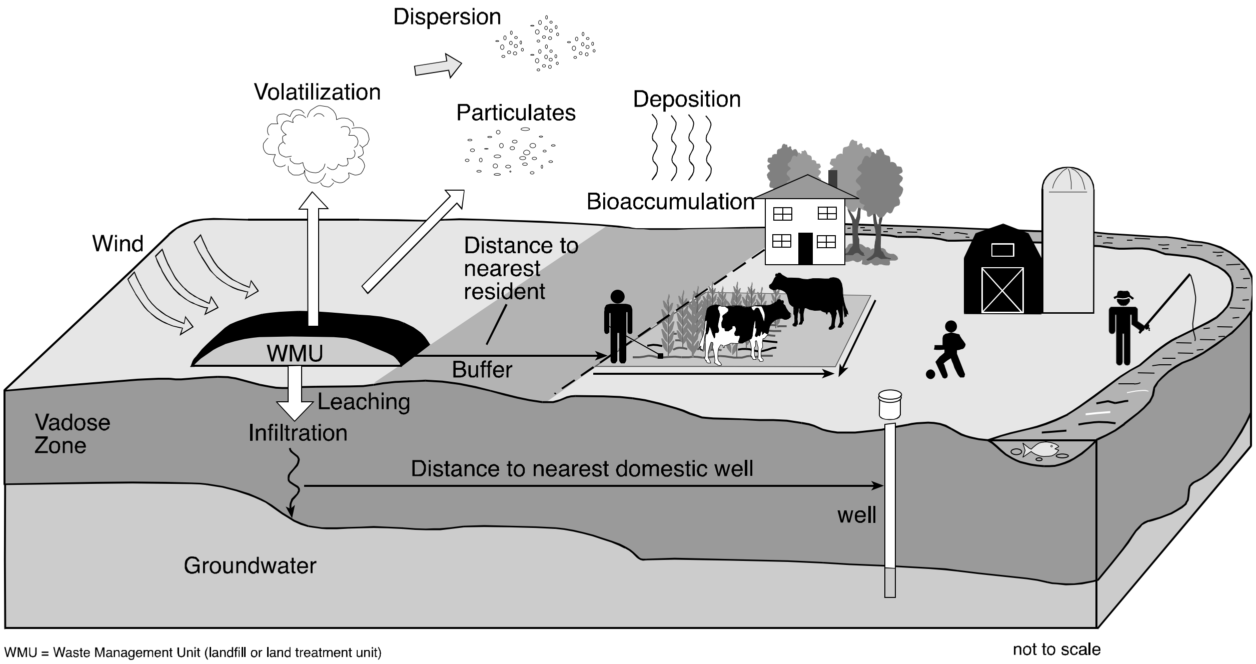

Figure 3-1. Conceptual site layout.

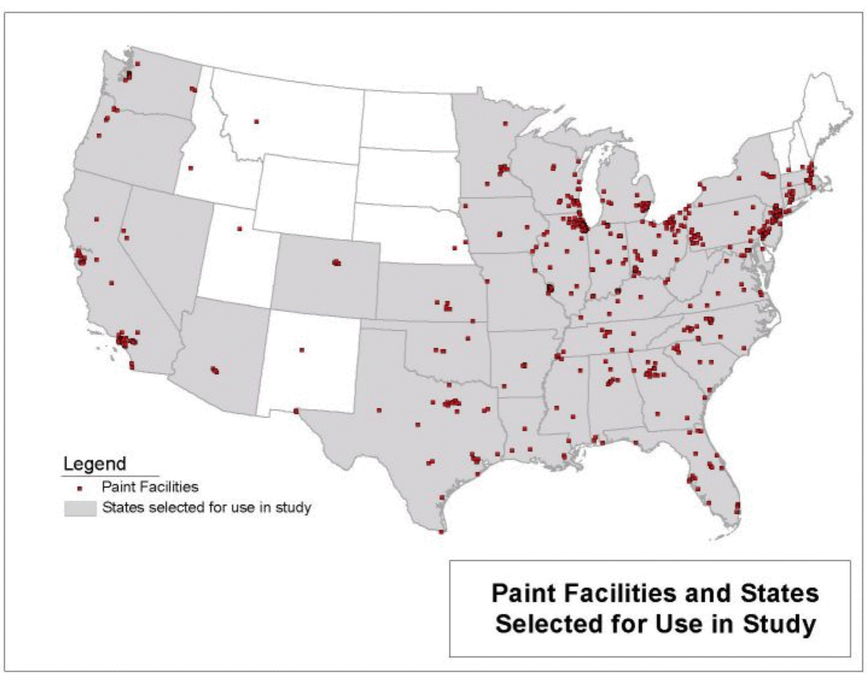

Figure 3-2. States included in the 1997 Census of Paint Manufacturing Facilities and Facility Locations from the 1997 TRI Survey.

selected for the analysis. States were chosen that were included in the 1997 Census of Paint Manufacturing Facilities (U.S. Department of Commerce, 1999) because census data were required to weight locations based on the amount of paint manufacturing in each state. Within each state, a meteorological station was identified so that the environmental setting could be characterized. If one meteorological station was judged to be inadequate to represent the range of climatic conditions surrounding paint manufacturing facilities (e.g., coastal vs. mountains), then additional meteorological stations were selected. In all, 49 meteorological stations were selected for modeling as shown in Figure 3-3.

Once the meteorological stations were identified, other environmental characteristics were determined based on the location of the meteorological station.

- Soils were characterized based on soil data within 20 miles of the meteorological station location.
- Aquifer types described in Newell et al. (1989) were selected based on aquifer types characteristic of the region in which the meteorological site is located.
- Agricultural field sizes were characterized based on median agricultural field size for counties within 20 miles of the meteorological station location.

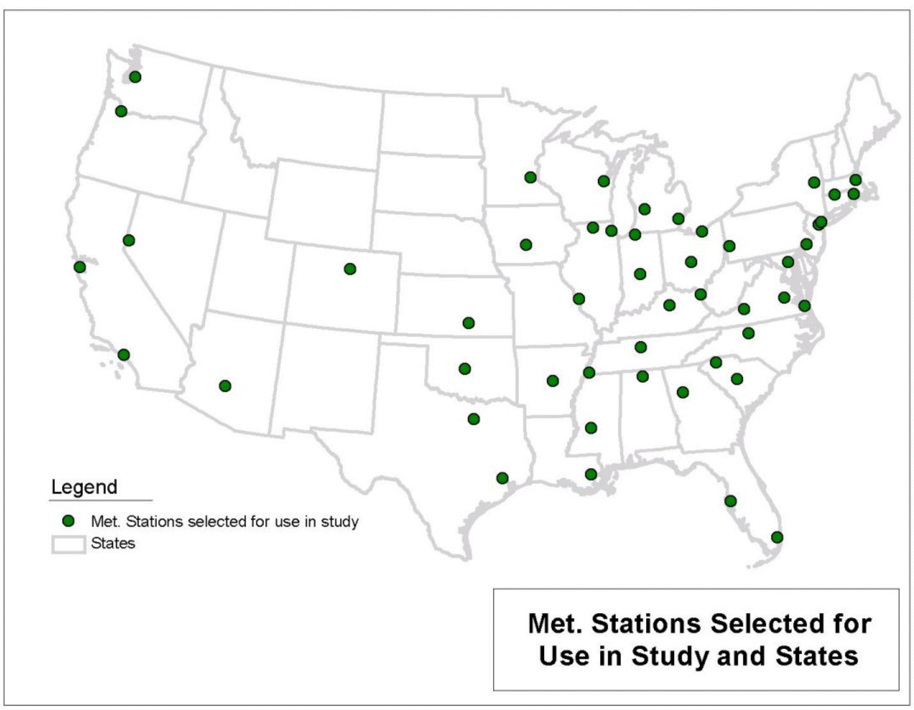

Figure 3-3. Meteorological stations included in the paints listing risk assessment.

3.1.5 Exposure Point Estimates

A series of models were used to estimate concentrations of COCs in the environment with which receptors may come into contact. A source partitioning model was used to estimate environmental releases of each COC from a WMU for each waste stream, as appropriate. These estimated environmental releases provided input to the fate and transport models to estimate media concentrations in air, soil, surface water, and groundwater. A farm food chain model was used to estimate COC concentrations in produce, beef, and dairy products. Aquatic bioconcentration factors were used to estimate concentrations in fish.

3.1.5.1 Source Partition Modeling. Each WMU evaluated has different release mechanisms that determine the media that can be impacted. Table 3-3 lists the primary release mechanisms that apply to each WMU.

Landfills. Wastes managed in off-site industrial landfills can release COCs as vapors or particles to the air via wind-blown erosion or as leachate to the groundwater. It was assumed that erosion and runoff from an operating industrial landfill are controlled; therefore, no overland transport of COCs was modeled.

Tanks. Wastes managed in tanks can release COCs into the atmosphere via volatilization. Because tanks contain liquid waste, no particulate emissions were considered from this WMU. Waste in the tanks was assumed not to leak so that no direct releases to the groundwater or soil would occur.

Surface Impoundments. Release mechanisms from surface impoundments included volatilizing to the air and leaching to the groundwater. Because surface impoundments contain liquid waste, no particulate emissions can occur from this WMU.

The source partition models require information on the WMU (e.g., surface area), waste stream (e.g., moisture content), and environmental setting (e.g., precipitation, temperature) to estimate environmental releases of COCs. The source partition model was also used to estimate infiltration rates for the landfill and surface impoundment. Because the concentration of COCs in the waste streams was to be the endpoint of this analysis, the source partition models were executed using a unit concentration (e.g., 1 mg/kg for solid wastes or 1 mg/L for liquid wastes).

Table 3-3. WMU and Primary Release Mechanisms

WMU	Volatilization	Wind-blown Particulates	Leaching
Landfill	✓	✓	✓
Tanks	✓		
Surface impoundments	✓		✓

Because off-site disposal was being evaluated, waste characteristics in the WMUs were based on general mixed waste characteristics. In addition, the source partition models were initially executed assuming that 100 percent of the waste being disposed of in the WMU was contaminated. In other words, the fraction of waste contaminated (fwmu) was set to 1. The source models use fwmu and the contaminant concentration in the waste stream to calculate the concentration of waste in the WMU based on simple dilution. By setting fwmu to 1, the source models were effectively used to evaluate mixed waste in the WMU, not the contaminated waste stream. Thus, target waste concentrations were first estimated that represent the concentration of waste in a WMU that would not result in health risk.

Estimations of the actual fwmus were calculated outside of the model using waste volumes from the 3007 Survey and the capacities of the WMUs. The actual fwmu was used in combination with the target waste concentration in the WMU to calculate the target waste concentration in the waste streams.

3.1.5.2 <u>Fate and Transport Modeling</u>. Fate and transport mechanisms are also depicted in Figure 3-1. As described above, a source partition model was used to determine the amount and nature of constituent release into the environment. Once in the environment, the released COCs could move through various compartments and into various environmental media. A multimedia approach was used to characterize the movement of COCs through the environment. The multimedia approach considered atmospheric concentrations, atmospheric deposition, soil concentrations, waterbody concentrations, groundwater concentrations, and indoor air concentrations.

Table 3-4 lists the environmental media that apply to each WMU. All of the WMUs under consideration had releases that could contaminate the air by primary mechanisms. Once in the air, COCs are transported via atmospheric processes and removed via both wet and dry deposition. These deposition mechanisms transport COCs to the soil and surface water compartments. Once in the soil, contaminants can also move to the surface water via erosion and runoff. Soil concentrations were calculated for both tilled and untilled soils. Tilled soils were used to represent soils under active cultivation, and untilled soils were used to represent areas such as the buffer area or pasture. Only landfills and surface impoundments have the potential to leach contaminants into the groundwater. COCs in groundwater can also be released into bathroom air when groundwater is used for showering.

Table 3-4. WMU and Impacted Environmental Media

WMU	Ambient Air	Soil	Surface Water	Sediment	Groundwater	Indoor Air (Shower)
Landfill	✓	✓	✓	✓	✓	✓
Tanks	✓	✓	✓	✓		
Surface impoundments	✓	✓	✓	✓	✓	✓

Although both aboveground and groundwater pathways were evaluated, they were treated separately in this analysis. This decision was based on differences in the time frame and receptor location. For most contaminants, it may take hundreds of years for a contaminated groundwater plume to impact a groundwater well, while aboveground contamination of air generally occurs simultaneously with the release. In addition, the aboveground receptor locations may not necessarily overlap (i.e., the aboveground receptors are randomly located around the WMU and may not coincide with the location of the groundwater plume).

3.1.5.3 Farm Food Chain Model. A farm food chain model was used to estimate the concentration of COCs in aboveground produce, belowground produce, beef, and dairy products. Table 3-5 presents the environmental media and vegetation considered in the farm food chain model. Aboveground produce is impacted via vapor transfer and deposition of COCs present in the air as well as uptake of COCs from tilled soil. Belowground produce is impacted only by uptake of COCs from tilled soil. The concentration of COCs was also estimated for forage, silage, and grain that is consumed by cattle. Although this risk assessment did not evaluate site-specific data (e.g., actual agricultural fields), agricultural field size was varied based on median agricultural field size data for each of the 49 locations modeled. Beef cattle and dairy cows also drink from contaminated surface waters. COCs that are ingested by these animals can contaminate both beef and dairy products.

3.1.5.4 Aquatic Food Chain Model. An aquatic food chain model was used to estimate the concentration of COCs in fish populations. Depending on chemical-specific parameters, COCs in surface water can contaminate fish via uptake and bioaccumulation to varying degrees. Trophic level 3 (T3) and 4 (T4) fish were considered in this analysis. Trophic level 3 fish are those that consume invertebrates and plankton. Trophic level 4 fish are those that consume other fish. Most of the fish that humans consume are T4 fish (e.g., salmon, trout, walleye, bass) and medium to large T3 fish (e.g., carp, smelt, perch, catfish, sucker bullhead, sauger).

3.1.6 Assessing Human Exposures

Human receptors may come into contact with COCs present in environmental media through a variety of pathways.

Table 3-5. Environmental Media and Vegetation Considered in the Farm Food Chain Model

	Aboveground Produce	Belowground Produce	Forage	Grain	Silage
Ambient air	✓		✓		✓
Soil	✓	✓	✓	✓	✓

3.1.6.1 <u>Human Receptors</u>. Seven human receptors were evaluated in this assessment:

- Adult resident
- Child resident
- Farmer
- Child farmer
- Fisher
- Adult resident (groundwater)
- Child resident (groundwater).

These receptors reflect the range of possible individual exposures for direct and indirect exposure pathways. Child exposures were evaluated based on a 1- to 6-year-old cohort. This cohort was selected because it is the most conservative for most exposure pathways and COCs evaluated in this risk assessment. Although both aboveground and groundwater pathways were evaluated, they were treated separately in this analysis. This decision was based on differences in the time frame and receptor location. For most contaminants, it may take hundreds of years for a contaminated groundwater plume to impact a groundwater well, while aboveground contamination of air generally occurs simultaneously with the release. In addition, the aboveground receptor locations may not necessarily overlap (i.e., the aboveground receptors are randomly located around the WMU and may not coincide with the location of the groundwater plume).

3.1.6.2 <u>Exposure Pathways</u>. Table 3-6 lists each receptor, along with the specific exposure pathways that apply to that receptor. Figures 3-4, 3-5, and 3-6 depict the environmental media and exposure pathways modeled in this assessment for residential, agricultural, and fisher scenarios. Exposure pathways are either direct, such as inhalation of ambient air, or indirect, such as the farm food chain pathways. The exposure pathways considered in this assessment were inhalation of ambient air, ingestion of soil, ingestion of aboveground produce, ingestion of root crops, ingestion of beef and dairy products, ingestion of fish, inhalation of indoor air via contaminated groundwater, and ingestion of drinking water. The groundwater pathways were considered separately from the aboveground pathways for the adult resident and the child resident because the time frame for groundwater exposure is often not consistent with that of other exposure pathways. Furthermore, aboveground receptors are randomly located and do not necessarily coincide with the location of the groundwater plume.

3.1.7 Toxicity Assessment and Risk Characterization

To characterize the risk from human exposures to a COC, toxicity information on each COC was developed for use with the exposure assessment results. For this risk assessment, the toxicity of a constituent was defined by a human health benchmark for each route of exposure (e.g., inhalation reference concentration, ingestion reference dose, cancer slope factor). Essentially, a benchmark is a quantitative value used to predict a chemical's possible toxicity and ability to induce a health effect at certain levels of exposure. These health benchmarks are derived from toxicity data based on animal studies or human epidemiological studies. Each benchmark represents a dose-response estimate that relates the likelihood and severity of adverse health effects to exposure and dose. Because individual chemicals cause

Table 3-6. Receptors and Exposure Pathways

Receptor	Inhalation of Ambient Air	Ingestion of Soil	Ingestion of Above- and Belowground Produce	Ingestion of Beef and Dairy Products	Ingestion of Fish	Inhalation of Indoor Air (Shower)	Ingestion of Drinking Water
Adult resident	✓	✓					
Child resident	✓	✓					
Farmer	✓	✓	✓	✓			
Child farmer	✓	✓	✓	✓			
Fisher	✓	✓			✓		
Adult resident[a]						✓	✓
Child resident[a]							✓

[a] Groundwater pathways were considered separately for the adult resident and the child resident because the time frame for ground water exposure is often not consistent with that of other exposure pathways. Furthermore, aboveground receptors are randomly located and do not necessary coincide with the location of the groundwater plume.

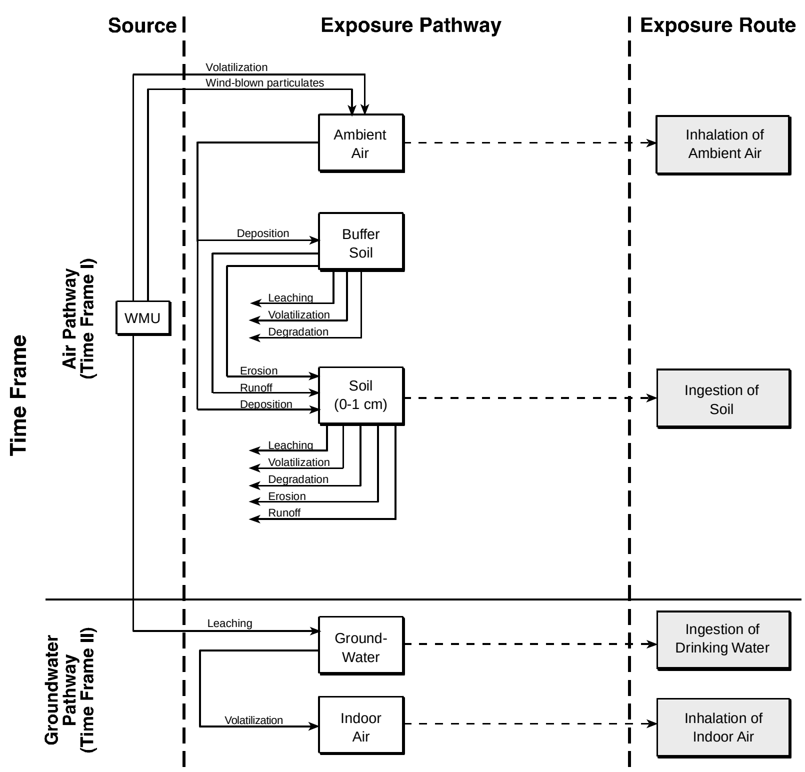

Figure 3-4. Residential scenario.

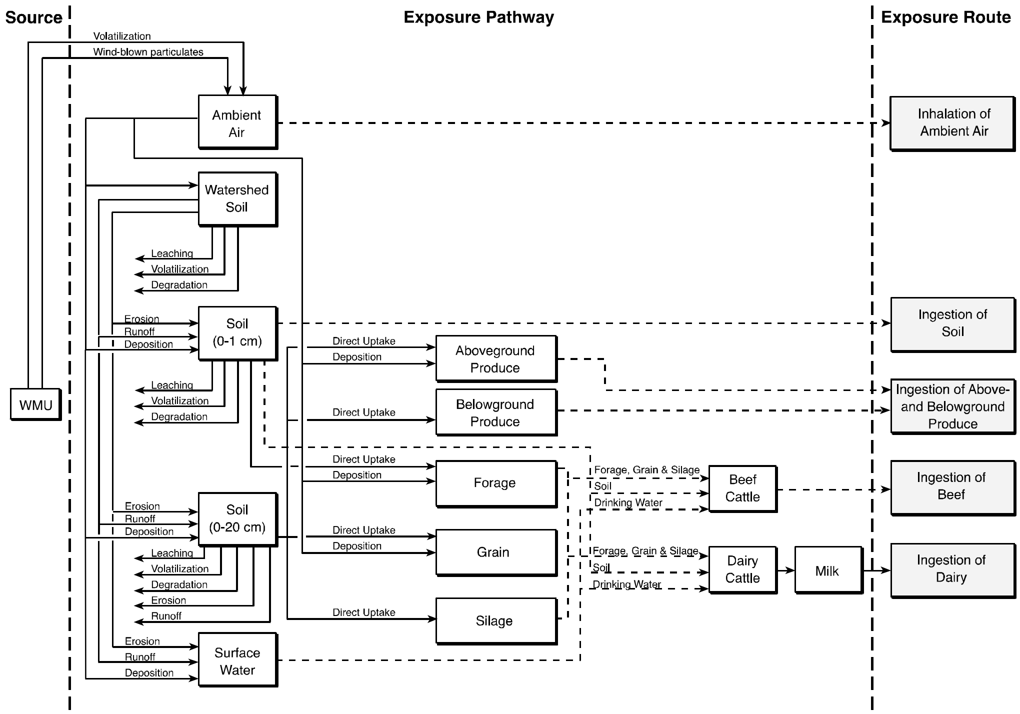

Figure 3-5. Agricultural scenario.

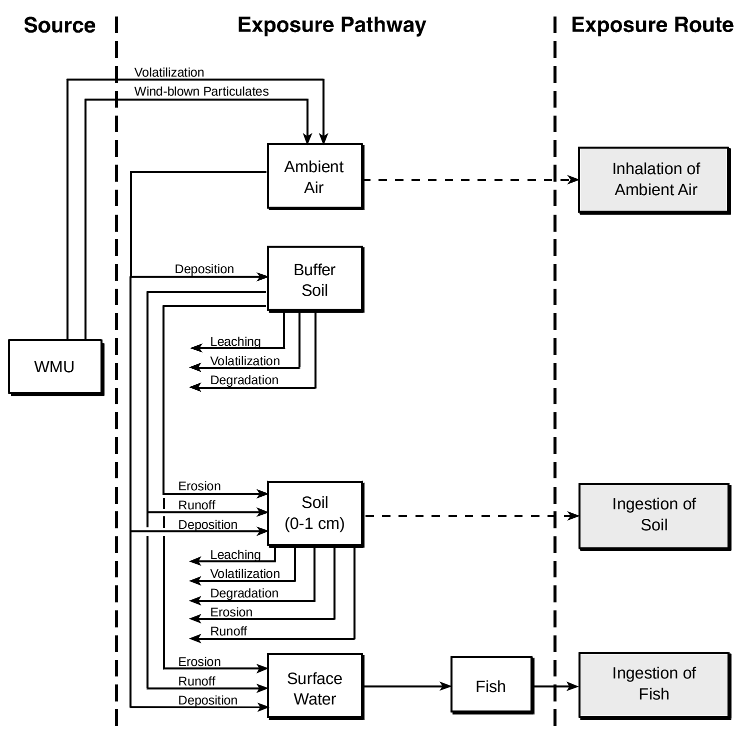

Figure 3-6. Fisher scenario.

different health effects at different doses, benchmarks are chemical specific. Human health benchmarks for chronic oral and inhalation exposures were needed for the risk characterization model. Table 3-7 summarizes the types of human health benchmarks used in this risk assessment.

Although it is not the endpoint of the paints listing risk assessment, risk characterization is necessary to establish the protective waste concentrations based on a target risk level. Several risk endpoints were used to characterize risk for the human receptors evaluated in this risk assessment. The term risk endpoint refers to the particular measure of human health hazard or risk (i.e., lifetime excess cancer risk). The risk endpoints used in this risk assessment are listed in Table 3-8.

A risk endpoint is a specific type of risk estimate (e.g., individual cancer risk estimate) that is used as the metric for a given risk category. The paints listing risk assessment evaluated specific categories of risk—cancer effects and noncancer effects. Each of the COCs evaluated in this risk analysis can be placed into one or both of these categories of risk depending on the health effect being considered (e.g., acrylonitrile was evaluated for cancer and noncancer effects).

3.1.7.1 <u>Assessment for Lead and Copper</u>. Neither lead nor copper have an oral reference dose to evaluate potential noncancer effects due to ingestion. Rather than explicitly evaluating oral noncancer effects using a reference dose, protective media concentrations were used to calculate target waste concentrations. Target waste concentrations calculated in this manner represent waste concentrations that can be disposed of without causing concentrations in the environment in excess of the standards. For the groundwater pathway, EPA's published drinking water action level was used for lead and copper. For lead only, a published soil screening level (SSL) was used based on EPA/OSWER's SSL. This level was developed using EPA's Integrated Exposure and Uptake Biokinetic (IEUBK) Model. This model considers lead exposure from various sources to estimate blood lead levels in children.

3.1.8 Calculating Protective Waste Concentrations

Based on calculated values for various risk endpoints, a waste concentration scaling factor was estimated. The scaling factor is the ratio of the value calculated for a given risk endpoint on the basis of unit exposure and unit risk values to target risk levels. Target risk levels have been set by EPA as 1×10^{-5} cancer risk and hazard quotient of 1.0. Cancer risks are a measure of the lifetime excess cancer risk for an individual and reflect the risk of cancer developing in an individual due to a lifetime of exposure to a particular constituent. A hazard quotient is a measure of human noncancer health hazard due to chronic exposure to a particular constituent and is measured as the ratio of the modeled dose for an individual to a reference dose that has been established by EPA or other government agency as a threshold below which human health effects are not likely to occur. Similarly, noncancer human health hazards can also be evaluated by comparing long-term air concentrations of a constituent to which an individual is exposed to a reference air concentration. In summary, protective waste concentrations were calculated based on total lifetime cancer risk, noncancer ingestion, and noncancer inhalation for all receptors for each disposal scenario for aboveground and groundwater as appropriate.

Table 3-7. Human Health Effects Evaluated for Paints Listing Risk Assessment

CASRN	Constituent	Reference Dose (mg/kg-d)	Reference Concentration (mg/m^3)	Oral CSF[a] $(mg/kg\text{-}d)^{-1}$	inh CSF[b] $(mg/kg\text{-}d)^{-1}$
79-06-1	Acrylamide	✓	✓	✓	✓
107-13-1	Acrylonitrile	✓	✓	✓	✓
7440-36-0	Antimony	✓	✓		
7440-39-3	Barium	✓	✓		
71-43-2	Benzene			✓	✓
71-36-3	Butyl alcohol, n-	✓			
85-68-7	Butylbenzylphthalate	✓			
7440-43-9	Cadmium	✓	✓		✓
67-66-3	Chloroform	✓	✓		
16065-83-1	Chromium (III)	✓			
18540-29-9	Chromium (VI)	✓	✓		✓
7440-48-4	Cobalt	✓	✓		
7440-50-8	Copper		✓		
108-39-4	Cresol, m-	✓			
95-48-7	Cresol, o-	✓			
106-44-5	Cresol, p-	✓			
117-81-7	Di(2-ethylhexyl)phthalate	✓	✓	✓	
84-74-2	Dibutylphthalate	✓			
75-09-2	Dichloromethane (methylene chloride)	✓	✓	✓	✓
105-67-9	Dimethylphenol, 2,4-	✓			
7439-97-6d	Divalent mercury	✓			
100-41-4	Ethylbenzene	✓	✓		
107-21-1	Ethylene glycol	✓	✓		
50-00-0	Formaldehyde	✓			✓
7439-92-1	Lead				
7439-97-6e	Mercury (elemental)		✓		
67-56-1	Methanol	✓	✓		
78-93-3	Methyl ethyl ketone (MEK)	✓	✓		
108-10-1	Methyl isobutyl ketone (MIBK)	✓	✓		
80-62-6	Methyl methacrylate	✓	✓		
22967-92-6	Methylmercury	✓			
7440-02-0	Nickel	✓	✓		
1313-99-1	Nickel oxide		✓		
87-86-5	Pentachlorophenol	✓	✓	✓	✓
108-95-2	Phenol	✓			
7782-49-2	Selenium	✓	✓		
7440-22-4	Silver	✓	✓		
100-42-5	Styrene	✓	✓		

(continued)

Table 3-7. (continued)

CASRN	Constituent	Reference Dose (mg/kg-d)	Reference Concentration (mg/m^3)	Oral CSF[a] $(mg/kg\text{-}d)^{-1}$	inh CSF[b] $(mg/kg\text{-}d)^{-1}$
127-18-4	Tetrachloroethylene	✓	✓	✓	✓
7440-31-5	Tin	✓			
108-88-3	Toluene	✓	✓		
108-05-4	Vinyl acetate	✓	✓		
1330-20-7	Xylene (mixed isomers)	✓	✓		
7440-66-6	Zinc	✓	✓		

[a] Oral cancer slope factor.
[b] Inhalation cancer slope factor.

Table 3-8. Risk Endpoints for Cancer and Noncancer Effects

Risk Category	Risk Endpoint	Definition
Cancer effects	Lifetime excess cancer risk - inhalation	Lifetime excess cancer risk resulting from inhalation exposure to a single chemical
	Lifetime excess cancer risk - ingestion	Lifetime excess cancer risk resulting from ingestion exposure to a single chemical
	Total lifetime excess cancer risk	Lifetime excess cancer risk resulting from multiple pathway exposures to a single chemical (inhalation and ingestion)
Noncancer effects	Ingestion hazard quotient	Ingestion pathway noncancer risk characterization from exposure to all ingestion pathway components for a single chemical
	Inhalation hazard quotient	Inhalation pathway noncancer risk characterization for a single chemical
Lead and copper	Ingestion hazard quotient based on drinking water action level	Ingestion pathway noncancer risk characterization based on groundwater concentration
Lead	Ingestion hazard quotient based on soil screening level	Ingestion pathway noncancer risk characterization based on soil concentration

Because the risk values calculated by the risk model are based on unit waste concentrations and because the models used in this risk assessment are linear, the scaling factor can be used to calculate a waste concentration that will result in a target risk level. The most restrictive (lowest) of the waste concentrations calculated in this manner was selected as the protective waste concentration. Protective waste concentrations were determined for aboveground and groundwater pathways separately because of the differences in the time frame and location of exposure. For the groundwater pathway, a protective leachate concentration was calculated as well.

If protective waste concentrations exceeded 1,000,000 ppm, the chemical was screened from the analysis because this concentration cannot physically exist. As an additional check, the protective waste concentrations were evaluated as to whether they were below the solubility limitations of the source models used. For tank and surface impoundments, chemical concentrations were compared with the aqueous solubility limit of the chemical. For landfills, the waste concentrations were evaluated using the source model.

3.2 Probabilistic and Deterministic Methods for Determining Exposure Point Concentrations

The primary methodology for this assessment was to estimate risk using a probabilistic (Monte Carlo) approach. A probabilistic analysis produces a distribution of risk or hazard for each receptor by varying parameter values over multiple iterations of the model. A deterministic analysis was also conducted resulting in point estimates of risk or hazard for each receptor based on a single execution of the models using a single value for each parameter in the analysis.

Section 3.2.1 provides an overview of the probabilistic analysis. The probabilistic analysis is also discussed in greater detail throughout the technical background document. Section 3.2.2 discusses the deterministic analysis in detail. The results of both analyses are presented in Appendix A.

3.2.1 Probabilistic Analysis

The probabilistic analysis was performed using a Monte Carlo simulation. In a Monte Carlo simulation, the models are run for a number of iterations, each producing a single result (e.g., a single estimate of cancer risk). For this assessment, 10,000 iterations were run in the Monte Carlo simulation. The output of the probabilistic analysis, therefore, is a distribution of 10,000 values. This distribution represents the distribution of possible outcomes, which reflects the underlying variability and uncertainty in the data used in the analysis. These results were then used to identify risk at various percentile levels (e.g., 90th percentile risk value).

3.2.1.1 Parameter Value Distributions. Many, but not all, model input parameters used in the Monte Carlo simulation were drawn from statistical distributions. Details on these parameter distributions are presented in the appendixes to this technical background document describing the models and data in detail:

- Appendix D, Chemical-Specific Parameters for Source Partitioning and Fate and Transport Models
- Appendix E, Waste Management Unit Parameters
- Appendix F, Variable Summary of Aboveground Fate and Transport Model
- Appendix G, Human Exposure Factors

In addition to parameter distributions, variability associated with location and WMU characteristics was explicitly considered in the setup of the source data used for the probabilistic analysis. First, locations were modeled based on the location of 49 selected meteorological stations, which were selected to generally represent the geographical distribution of paint manufacturing facilities. These locations were used to define a set of related environmental conditions (e.g., soil type, hydrogeologic environments) that characterize the environmental setting for the WMU. The 49 locations were then replicated to create a 10,000-record location file. The locations were replicated based on weights derived from amount of paint manufacturing per each state based on Census data (U.S. Department of Commerce, 1999). Location-dependent parameters are discussed in Section 4.2.

Second, WMUs were selected from their respective databases and replicated to produce three 10,000-record location WMU files: one for landfills, one for treatment tanks, and one for surface impoundments. These WMU files were used to define a set of WMU-specific parameters (e.g., surface area, depth). The WMUs were weighted based on the sampling procedure used to select WMUs from the underlying databases. The WMU-dependent parameters are discussed in Section 4.3.

Third, the location records and the WMU records were combined randomly to form three 10,000-record-location-WMU data sets. Each of these 10,000-record data sets is referred to as a source data file. There is one source data file each for landfills, treatment tanks, and surface impoundments. These source data files form the foundation of the Monte Carlo analysis. The source data files were then combined with waste stream data to define the waste management scenarios that were evaluated (see Section 3.1.2). The waste stream data define a set of parameter values associated with waste characteristics (e.g., bulk density).

3.2.2 Deterministic Analyses

Both central tendency and high-end deterministic risk assessments were conducted to quantify the risk or hazard. The central tendency assessment was used to describe risk or hazard for the "average" receptor in the population (the central tendency risk). For central tendency deterministic risk analyses, all parameter values were set at their central tendency or 50th percentile values.

The high-end assessment was used to describe the risk or hazard for individuals in small, but definable, high-end segments of the population (the high-end risk). Accordingly, the high-end deterministic risk analysis predicts the risks and hazards for those individuals exposed at the upper range of the distribution of exposures. EPA's *Guidance for Risk Characterization* (U.S. EPA, 1995) advises that "conceptually, high-end exposure means exposure above about the 90^{th} percentile of the population distribution, but not higher than the individual in the population who has the highest exposure," and recommends that ". . . the assessor should approach estimating high end by identifying the most sensitive variables and using high end values for a subset of these variables, leaving others at their central values."

For the paints high-end deterministic risk analyses, two parameters were set at their high-end values (generally 90^{th} percentile values), and all other parameters were set at their central tendency values. A sensitivity analysis was used to identify the two parameters that were set at high-end values. The sensitivity analysis was performed by alternately setting combinations of two parameters at high end to identify the parameters that most influence the analysis' outcome. The different results generated by the sensitivity analysis were compared and the two high-end parameters to which the analysis was "most sensitive" were selected for use in the high-end deterministic analysis (i.e., resulted in the lowest protective waste concentrations). The results of the sensitivity analysis are presented in Appendix C.

3.2.2.1 Selecting Central Tendency and High-End Parameters for Aboveground Pathways. Parameters considered for the aboveground high-end deterministic analyses were selected from previous listing risk assessments. Experience from previous risk assessments has shown that, although the particular parameters to which the models are most sensitive depends on constituents and pathways considered, modeled exposure and risk values are most sensitive to only a relatively small subset of parameters. In this assessment, a sensitivity analysis was conducted using the list of parameters identified in previous listing risk assessments:

- WMU surface area
- Distance to receptor
- Meteorological location
- K_d value for WMU
- K_d value for surface soil
- Exposure duration
- Waste volume.

Central tendency and high-end values for the parameters listed above are presented in Table 3-9. The derivation of these values is described briefly below.

Surface Area. Surface area is a sensitive parameter because it can have significant impact on air dispersion modeling and emission modeling results. For air dispersion modeling, larger surface areas tend to result in higher air concentrations and deposition values. For emission modeling, however, smaller areas can be associated with higher emissions for a given waste volume. Therefore, in this assessment, both the 10^{th} and 90^{th} percentile surface areas were identified for evaluation in the sensitivity analysis. The surface area values were selected based on an analysis of the WMU surface areas. WMUs that satisfied other criteria and had surface areas near 50^{th}, 10^{th}, and 90^{th} percentile surface areas were selected. The surface areas used in the deterministic analysis are presented in Table 3-9 as the central tendency (50^{th} percentile) and high-end values (10^{th} and 90^{th} percentiles).

Table 3-9. Parameters Varied in High-End Analysis for Aboveground Pathways

Input Parameter	Deterministic Analysis: Parameter Value — Central Tendency (CT) Value	Deterministic Analysis: Parameter Value — High-end (HE) Value	Data Source
Nongroundwater Modeling			
WMU surface area (m^2)			Based on analysis of the median, 90th, and 10th percentile surface areas of WMUs
Landfill	64,752	202,350 (lg) 4,150 (sm)	
Surface impoundments	4,047	74,690 (lg) 29 (sm)	
Tanks	11.09	173 (lg) 2.06 (sm)	
Distance to receptor (m)	300	75	Hazardous Waste Treatment, Storage, and Disposal Facilities - Organic Air Emission Standards for Process Vents and Leaks, Final Rule. (U.S. EPA, 1990)
Meteorological location	Indianapolis, IN	Hartford, CT	Based on analysis of ISCST3 outputs and on consideration of annual average wind speed and ambient air temperatures.
Distribution coefficient, K_d (metals only)	Metal-specific See Table 3-11	Metal-specific See Table 3-11	Based on distribution presented in Appendix H.
Exposure duration (years)			Based on data from the *Exposure Factors Handbook* (1997a) (Tables 15-164, 15-168, 15-176).
Adult resident	9	30	
Child resident	5	13	
Adult farmer	10	48.3	
Child farmer	10	48.3	
Fisher	9	30	
Waste volume (3007 Survey)			Median (CT) and 90th percentile values (HE) from Monte Carlo distribution developed from BRS. Data provided by EPA.
Dust (m^3/yr)	2.44	220.8	
Combined waste (m^3/yr)	1.42	163.8	
Aqueous waste (m^3/yr)	45.42	101.3	

Distance to Receptors. The distance from the source to the receptor was set at 300 meters for central tendency and 75 meters for high end. These same values were used in several previous waste listing determinations and were originally used in the risk assessment conducted for the Hazardous Waste Treatment, Storage, and Disposal Facilities - Organic Air Emission Standards for Process Vents and Leaks Final Rule (U.S. EPA, 1990). In this rule, 75 meters (250 feet) is based on the actual measured distance to the nearest resident for the worst-case facility and 300 meters (1,000 feet) is identified as the median distance in a distribution of distances to the nearest residence.

Meteorological Location. Central tendency and high-end meteorological locations were selected based primarily on an analysis of ISCST3 modeling results for 49 meteorological locations and secondarily on annual average windspeed and ambient temperatures. The analysis of the air dispersion modeling results considered both particulate and vapor outputs for a median-sized landfill. Five types of ISCST3 outputs were generated—air concentration of vapor, wet deposition of vapor, air concentration of particles, wet deposition of particles, and dry deposition of particles. Because dry deposition of vapors was calculated outside of ISCST3, selection of meteorological sites did not consider dry deposition of vapors. For each type of ISCST3 output, the average values estimated for the central tendency (300 meters) and high-end (75 meters) receptor distances were ranked. Based on these rankings, locations associated with 50th and 90th percentile ISCST3 outputs were identified. Indianapolis, Indiana, was selected to represent the central tendency conditions and Hartford, Connecticut, was identified as the high-end site.

As an additional check, the average annual windspeed and ambient air temperatures of the selected sites were examined to ensure that the selected sites were not associated with extreme values. Windspeed and temperature had to be considered because they are used as input to the emission model and can impact emission results (e.g., higher temperatures result in higher emissions). Table 3-10 summarizes these data for all three selected sites. As seen from this table, neither the central tendency nor high-end site is associated with extreme values.

K_d Value (metals only). Distribution coefficient, K_d, values for metals in this risk assessment are based on empirical data drawn from the scientific literature (see Appendix H). The K_d value for each metal was represented in the probabilistic analysis by either an empirical or log uniform distribution, depending on the quantity of data available. The K_d parameter was used in three different parts of the risk analysis, each treated independently of the other. Specifically, K_ds were selected for the WMU and the surficial soil. Because these values were modeled independently in the probabilistic analysis, they are treated independently for the sensitivity analysis.

The K_d values for these parameters were selected from the distributions used to determine K_d values for the probabilistic analysis. The 50th and 90th percentile values were selected for the central tendency and high end, respectively (see Table 3-11).

Exposure Duration. Exposure duration is an important parameter for determining cancer risk. Exposure durations were identified for each receptor based on data from the *Exposure Factors Handbook* (U.S. EPA, 1997a). The central tendency and high-end values (9 and 30 years, respectively) used for the exposure duration of adult residents and fishers were

Table 3-10. Average Ambient Temperatures for Central Tendency and High-end Sites

Meteorological Location	Average Annual Windspeed (m/s)	Average Ambient Temperature (°F)
Central tendency Indianapolis, IN	4.63	53
High-end Hartford, CT	4.12	50
Across all 49 stations	5.14 (56^{th} percentile)	56 (52^{th} percentile)

Table 3-11. Central Tendency and High-End Partition Coefficient Values Used in Deterministic Analyses

Chemical	CAS	CT	HE
Antimony	7440360	12.38	0.78
Barium	7440393	240.11	15.16
Cadmium	7440439	204	14
Chromium (III)	16065831	5977	442
Chromium (VI)	18540299	26.9	0.6
Cobalt	7440484	935	41
Copper	7440508	476	35.75
Divalent mercury	99991	4500	0.22
Lead	7439921	5310	20
Mercury	7439976	1000	1000
Nickel	7440020	440	18
Selenium	7782492	24.75	5.71
Silver	7440224	1200	26.8
Tin	7440315	4076.57	257.35
Zinc	7440666	2019.5	33.8

EPA-recommended values from the EFH (Table 15-176, U.S. EPA, 1997a). Exposure duration for the child resident was based on the data for 3-year-olds (the average child start age occurring between ages 1 and 6) (Table 15-168, U.S. EPA, 1997a). Exposure duration for the adult and child farmer was based on farm residence time data (Table 15-164, U.S. EPA, 1997a). Central tendency and high-end values were represented by the 50th and 90th percentile values, respectively.

Waste Volume. Waste volume is an important parameter because the relative amount of waste deposited in a WMU can have a significant impact on rate of constituent release to the environment. Paint waste volume distributions used in the 10,000 iterations of the probabilistic analysis were used to select central tendency and high-end values. The median waste volume was selected as the central tendency waste volume and the 90th percentile waste volume was selected as the high-end waste volume.

3.2.2.2 <u>Selecting Central Tendency and High-End Parameters for Groundwater Pathways</u>. Parameters considered for the groundwater high-end deterministic analysis were selected based on findings from previous listings risk assessments. Experience from previous risk assessments has shown that, although the particular parameters to which the models are most sensitive depends on constituents and pathways considered, modeled exposure and risk values are most sensitive to only a relatively small subset of parameters. In this assessment, a sensitivity analysis was conducted using the list of parameters identified in previous listings risk assessments. These parameters have been identified separately for landfills and surface impoundments:

Landfills. For landfills, only the following parameters were identified:

- Infiltration rate
- Depth.

Surface Impoundments. For surface impoundments, only the following parameter was identified:

- Water flux.

The remaining six parameters were evaluated for both surface impoundments and landfills:

- Distance to receptor well
- Depth to groundwater (vadose soil thickness)
- K_d value for WMU
- K_d value for vadose
- K_d value for aquifer
- Exposure duration
- Waste volume.

Central tendency and high-end values for the parameters listed above are presented in Table 3-12. The derivation of these values is described briefly below.

Table 3-12. Parameters Varied in High-end Analysis for Groundwater Parameters

Input Parameter	Deterministic Analysis: Parameter Value Central Tendency (CT) Value	 High-end (HE) Value	Data Source
Landfill Infiltration rate	Oklahoma City	Boston	Based on evaluation of Monte Carlo distribution, the locations associated with CT and HE infiltration for a CT landfill surface area were selected.
Depth of landfill Area (m^2) Depth (m)	 38,851 2.11	 40,470 7.57	Based on evaluation of Monte Carlo distribution (see Section 3.2.1.2).
Surface impoundments Water flux Area (m^2) Depth (m)	 1,012 1.40	 33,722 4.76	Based on evaluation of Monte Carlo distribution (see Section 3.2.1.2).
Distance to receptor well (m)	430	102	Median (CT) and 10th percentile value (HE) from the EPA survey of distances between municipal landfills and domestic drinking water wells (U.S. EPA, 1997b, citing U.S. EPA, 1993).
Depth to groundwater (m) (vadose zone thickness)	6.1	1.52	Based on evaluation of Monte Carlo distribution, which reflects application of EPACMTP data.
Distribution coefficient, K_d (metals only)	Metal-specific See Table 3-11	Metal-specific See Table 3-11	Based on distribution presented in Appendix H.
Exposure duration (years) Adult resident Child resident	 9 5	 30 13	*Exposure Factors Handbook* (U.S. EPA, 1997a)
Waste volume (3007 Survey) Dust (m^3/yr) Combined solids (m^3/yr) Aqueous waste (m^3/yr)	 2.44 1.42 45.42	 220.8 163.8 101.3	Median (CT) and 90th percentile (HE) values from Monte Carlo distribution developed from BRS data provided by EPA.

Landfill Infiltration Rate. For landfills, infiltration rates tend to be driven by the amount of precipitation in the area where the landfill is located; therefore, the meteorological location is a more important consideration than WMU dimensions. Accordingly, infiltration rate was addressed by first examining the distribution of infiltration rates calculated for the 10,000-iteration probabilistic analysis and identifying the 50th and 90th percentile values. Then meteorological locations most closely corresponding to the 50th percentile and 90th percentile infiltration rate were identified. The 50th and 90th percentile meteorological locations selected in this manner were used to model central tendency and high-end infiltration for landfills. All parameters that are location-dependent (e.g., soils) and that are correlated with location were also determined by the meteorological location chosen so that correlated parameters would remain correlated in the deterministic analysis.

Landfill Depth. Landfill depth is an important parameter for modeling landfill leachate releases. To select the value for this parameter, 50th and 90th percentile values were identified from the distribution of 10,000 landfill depth values in the probabilistic analysis. Landfill units with surface area close to the median landfill size strata were examined for depth values that were close to the 50th and 90th percentile depth values. Two landfill units were selected, one with a depth near the 50th percentile as the central tendency unit and one near the 90th percentile as the high-end unit. Specific landfill units were chosen so correlated WMU parameters would remain correlated in the deterministic analysis.

Surface Impoundment Water Flux. For surface impoundments, the amount of water moving through the bottom of the unit into the soil below the unit, called the water flux, is a factor to which the groundwater modeling is sensitive. Water flux tends to be driven by the surface impoundment dimensions; therefore, surface area and depth of liquid for a specific unit is a more important consideration than meteorological conditions. Accordingly, water flux was addressed by first multiplying the surface area by the infiltration rate (where infiltration rate is strongly affected by depth) to determine water flux. Then the distribution of water flux was calculated for each of the 10,000 iterations in the probabilistic analysis, and the 50th and 90th percentile values were identified. Two surface impoundment units associated with these water flux values were identified: one corresponding to the 50th percentile water flux and one corresponding to the 90th percentile water flux. The 50th and 90th percentile surface impoundment units selected in this manner were the central tendency and high-end surface impoundments used in this analysis. Specific surface impoundment units were chosen so correlated WMU parameters would remain correlated in the deterministic analysis.

Distance to Receptor Well. The distance to the receptor well is based on the EPA survey of distances between municipal landfills and domestic drinking water wells (U.S. EPA, 1997b, citing U.S. EPA, 1993). The central tendency (50th percentile closest well) distance is 430 meters and the high-end (10th percentile closest well) distance is 102 meters. It should be noted that the actual location of the receptor well is defined by three parameters: the longitudinal distance from the edge of the WMU to the well (i.e., X well), the depth from the surface to the well (i.e., Z well), and the distance from the center line of the plume (i.e., Y well). Because the major consideration in selecting values for this parameter is downgradient distance, the values for the other well location parameters, Y-well and Z-well, were not varied independently and were selected to be central tendency. Specifically, Y-well was set halfway between the plume centerline and the estimated edge of the plume. Z-well was selected as the 50th percentile value from the probabilistic analysis.

Depth to Groundwater. This parameter is also known as the vadose zone thickness. The central tendency and high-end values were selected from the distribution of 10,000 values used in the probabilistic analysis. The median value was selected as the central tendency value and the 90th percentile value was selected as the high-end value.

K_d Value (metals only). Distribution coefficient, K_d, values for metals in this risk assessment are based on empirical data drawn from the scientific literature (see Appendix H). The K_d value for each metal was represented in the probabilistic analysis by either an empirical or log uniform distribution, depending on the quantity of data available. The K_d parameter was used in three different parts of the risk analysis, each treated independently of the other. Specifically, K_ds were selected for the WMU and the surficial soil. Because these values were modeled independently in the probabilistic analysis; they are treated independently for the sensitivity analysis.

Exposure Duration. Exposure duration is an important parameter for determining cancer risk. Exposure durations were identified for each receptor based on recommendations in the *Exposure Factor Handbook* (U.S. EPA, 1997a). The central tendency and high-end values (9 and 30 years, respectively) used for the exposure duration of adult residents were EPA-recommended values from the EFH (Table 15-176, U.S. EPA, 1997a). Exposure duration for the child resident was based on the data for 3-year-olds (the average child start age occurring between ages 1 and 6) (Table 15-168, U.S. EPA, 1997a). Central tendency and high-end values were represented by the 50th and 90th percentile values, respectively.

Waste Volume. Waste volume is an important parameter because the relative amount of waste deposited in a WMU can have a significant impact on rate of constituent release to the environment. Paint waste volume distributions used in the 10,000 iterations of the probabilistic analysis were used to select central tendency and high-end values. The median waste volume was selected as the central tendency waste volume and the 90th percentile waste volume was selected as the high-end waste volume.

3.3 Ecological Risk Assessment

The ecological risk assessment was designed to evaluate whether modeled paint waste management practices are likely to cause adverse effects to the environment. This was accomplished by conducting a screening ecological risk assessment based on assumed waste concentrations. For the ecological risk assessment, waste concentrations were set at 750,000 ppm, The target waste concentration of 750,000 ppm was selected by EPA as a conservative value appropriate for a screening level assessment. Constituents for which the screening results indicate that target hazard quotients could exceed 1 were further assessed with a Tier 2 analysis. The approaches used in the screening level and Tier 2 ecological risk assessments are described in Section 3.3.3.

3.3.1 Ecological Receptors

Two general types of receptors were evaluated in the ecological assessment. For exposure through direct contact with contaminated media, the receptors were multispecies communities such as the soil invertebrate community or the terrestrial plant community. For indirect exposure through ingestion, the receptors were single species populations, such as white-tailed deer or raccoons, including representative trophic levels and feeding strategies. Evaluation of risk to receptor populations and communities included consideration of both aquatic and terrestrial habitats. Within each habitat, risk was evaluated at all trophic levels (i.e., position within the food chain) and for all feeding strategies (e.g., plant feeder, predator). Although actual WMU sites were not defined, it was assumed that WMUs occur in a variety of settings including terrestrial, wetland, and aquatic systems. Thus, the ecological receptors evaluated in this risk assessment include representative plants and animals from several different terrestrial, wetland, and aquatic habitats. In general, the receptors considered occur throughout most of the continental United States or throughout broad regions, such as east of the Mississippi River.

Relevant trophic levels and feeding strategies (i.e., herbivorous, omnivorous, and carnivorous diets) were established using simple food webs that describe dietary composition and predator-prey relationships in each of the three habitat types. Receptors representing each feeding strategy at each trophic level were selected. In addition, the receptors represent a cross section of general taxa at each trophic level. For example, invertebrates as well as vertebrates were included, and vertebrate receptors include amphibians, mammals, and birds.

The ecological assessment does not specifically address federally listed threatened or endangered species. Although the bald eagle, a federally protected species, is included in the assessment, actual occurrences of the bald eagle or its critical habitat in association with paint WMUs were not identified. Therefore, the inclusion of the bald eagle as a receptor does not constitute an endangered species assessment. Rather, the bald eagle was included in the assessment because, as an avian top predator, it fills a niche not otherwise well represented by other receptors.

3.3.2 Ecological Exposure Pathways

To determine the exposure pathways of concern, a conceptual model was developed based on the assessment of sources, release mechanisms, and constituents' toxicity and environmental behavior. The exposure pathways included in the assessment were

- Root uptake of COCs in soil or sediment by plants
- Biological uptake of COCs in surface water by aquatic animals (e.g., fish or aquatic invertebrates)
- Biological uptake of COCs in sediment by benthic invertebrates
- Biological uptake of COCs in soil by soil invertebrates

- Ingestion of COCs in surface water, soil, sediment, or food items (plants and animals) by terrestrial animals.

3.3.3 Ecological Toxicity Assessment and Risk Characterization

The screening ecological risk assessment compared modeled media concentrations with ecotoxicological benchmarks to derive hazard quotients. An HQ greater than 1 indicates that the media concentration for the modeled waste management scenario is greater than the applicable ecological benchmark and that there is a potential for adverse impacts to ecological receptors.

The ecotoxicological benchmarks were derived from toxicological data in the literature and in EPA databases. They are expressed as chemical stressor concentration limits (CSCLs) in soil, sediment, and surface water. CSCLs are media concentrations that are assumed to be protective for ecological receptors.

The media concentrations were modeled using the same methodologies as those used for the human health assessment, as described in Sections 4.0 and 5.0. However, EPA set the waste concentrations for the ecological assessment at 750,000 ppm. This waste concentration is considered appropriately conservative for the screening analysis. The exposure point concentrations for soil are the concentrations in the agricultural field, and the exposure point concentrations for sediment and surface water are the concentrations in the waterbody used to evaluate risk for the fisher. The HQs are calculated using the 50^{th} and 90^{th} percentile media concentrations for each waste stream in each WMU type.

For constituents with HQs of 1 or greater for both the 90^{th} and 50^{th} percentile media concentrations, Tier 2 methods were applied. In the Tier 2 assessment, constituent-specific waste concentrations were calculated. These Tier 2 concentrations are the waste concentrations that would result in a maximum HQ equal to 1 at the 50^{th} and 90^{th} percentile exposure levels.

3.4 References

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4.0 Source Characterization

This risk assessment provides a national characterization of waste management scenarios for wastes generated in the manufacture of paints. The sources in these scenarios are the waste management units in which paint waste could be disposed of. How these sources are characterized in terms of their physical dimensions, operating parameters, and location is fundamental to the construction of scenarios for modeling. The scenarios that underpin this assessment are based on an understanding of industry operations and waste management practices that has been derived from secondary data sources. Industry data on which to base the characterization of waste streams and waste management practices come from survey data (i.e., the 3007 Survey).

This analysis evaluates risk in both a deterministic and probabilistic manner. The deterministic analysis produces results that reflect central tendency and high-end estimates. The probabilistic analysis is based on a Monte Carlo simulation that produces a distribution of exposures and risks. The foundation for the Monte Carlo simulation is the source data that define the Monte Carlo iterations. Specifically for this analysis, 10,000 iterations were completed to define a distribution of WMU scenarios. Compiling the source data required characterizing the environmental setting in which waste management occurs and characterizing the waste management units (WMUs) in which paint waste streams are managed. This section discusses the compilation of the source data for the probabilistic analysis. The selection of parameters for the deterministic analysis is discussed comprehensively in Section 3.2.

Section 4.1 presents an overview of the source data development procedure. Section 4.2 summarizes development of the waste management scenarios evaluated in this risk assessment. Section 4.3 presents the methodologies used to characterize the environmental setting, including delineation of the site layout and environmental setting (e.g., meteorology, climate, soils, and aquifers). Section 4.4 describes how we characterized the WMUs, including capacities and surface areas.

4.1 Source Data Development Procedure

To capture the national variation in WMU practices for the Monte Carlo analysis, a database of 10,000 different waste management scenarios was created. These 10,000 scenarios provided the source data for the fate and transport modeling. Figure 4-1 provides an overview of the process used to compile the source data needed for source partition modeling and fate and transport modeling. These source data are organized into source data files. The source data files contain information on locations and WMUs used in the probabilistic analysis. As shown in Figure 4-1, completion of six tasks was required, some in parallel and some sequentially, to

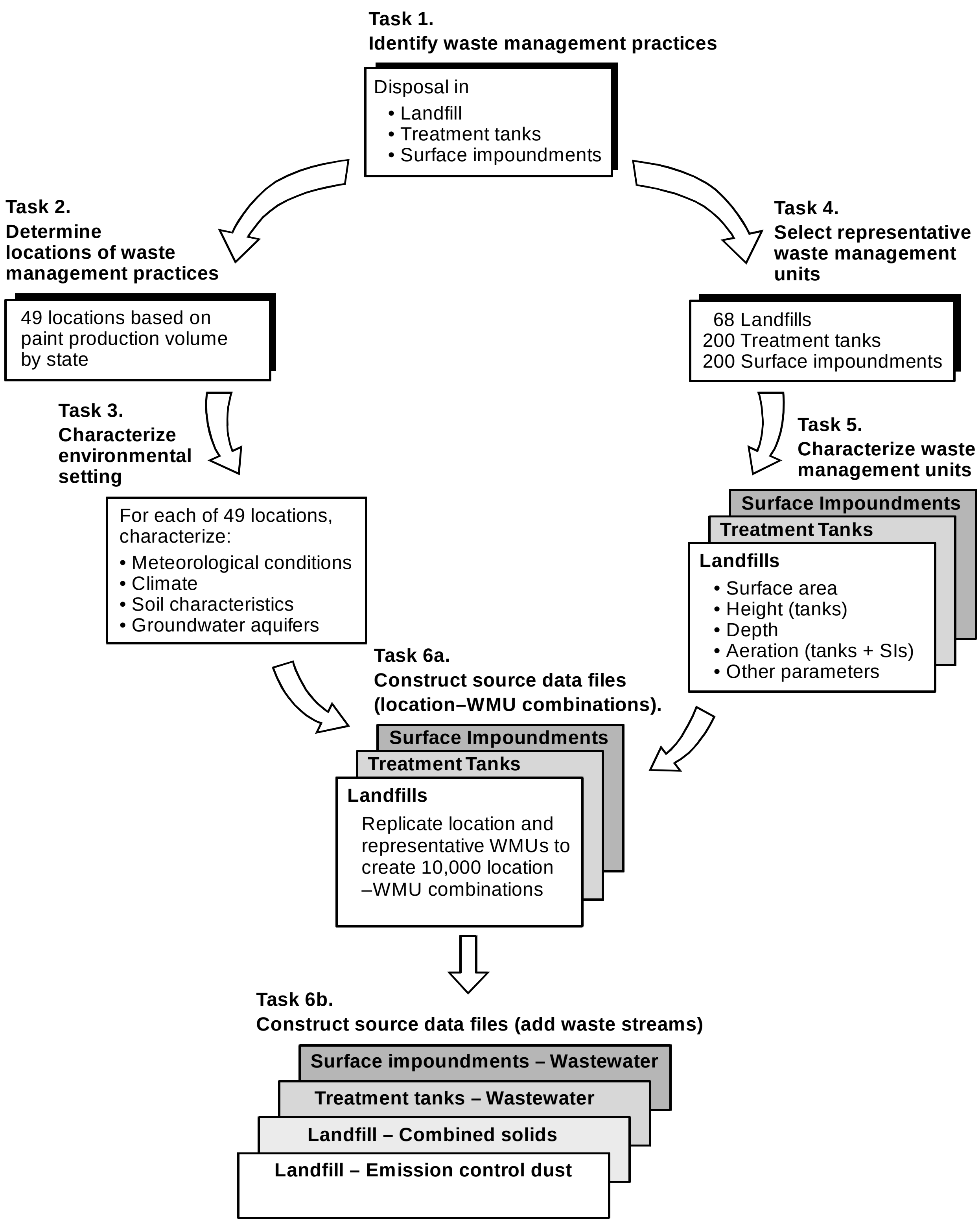

Figure 4-1. Process used to construct source data files used as basis for probabilistic analysis.

construct the source data files. The result was three source data files, one each for landfills, tanks, and surface impoundments.

Task 1. Identify Waste Management Practices

The first task in constructing a database of source characteristics was to identify the waste management practices to be evaluated in this risk assessment. Based on data in the open literature, national databases (e.g., Toxics Release Inventory), and industry site visits, EPA identified the waste management practices to be evaluated. Three WMUs were selected for inclusion in this risk assessment: landfills, treatment tanks, and surface impoundments.

Task 2. Determine Location of Waste Management Practices

The second task was to select locations to be modeled. Because specific paint waste disposal locations were not known, this analysis characterized environmental conditions based primarily on paint production volume by state. It is assumed that waste disposal locations are correlated with paint production locations. It was also assumed that nonhazardous waste from paint manufacturing facilities would be disposed of within reasonable transport distances of the facility. Therefore, locations for modeling were selected first for states according to the volume of paint manufactured and then by the general location of paint manufacturing facilities within the state. Because of the need for meteorological data for air dispersion modeling, locations selected were determined by the location of a meteorological monitoring station in proximity to manufacturing facilities. Forty-nine meteorological stations in 36 states were selected. The selection process is discussed in detail in Section 4.3.2.1.

Task 3. Characterize Environmental Setting

The environmental setting in which waste disposal occurs was characterized based on the location of the meteorological stations identified in Task 2. In Task 3 these locations were used to characterize meteorology, climate, soils, and aquifers. Meteorological data for a 5-year period were compiled and organized to provide data needed for the air dispersion modeling. Climate data were compiled to provide information used in source modeling and fate and transport modeling (e.g., annual precipitation, temperature). Soil characteristics within a 20-mile radius of the meteorological station location were developed for use in source modeling and fate and transport modeling. Aquifer types were also defined based on the location of the meteorological station. Both meteorological stations and aquifers were selected to capture the range of conditions found in the continental United States.

Task 4. Select Representative Waste Management Units

There are three types of WMUs evaluated in this assessment: landfills, treatment tanks, and surface impoundments. To determine the physical and operating characteristics used in air dispersion modeling and source partition modeling, representative WMUs were selected in this step. First, the databases from which individual WMUs could be selected and characterized were identified. These included the Industrial D database (Schroeder et al., 1987) for landfills and surface impoundments and the treatment, storage, and disposal facility (TSDF) database

(U.S. EPA, 1987) for treatment tanks. Then, representative WMUs were selected (68 for landfills) or sampled (200 each for treatment tanks and surface impoundments) from their respective databases. The landfills included all industrial landfills in the Industrial D database that were used for disposal of offsite wastes. Treatment tanks and surface impoundments were sampled from their respective databases using a stratified random sampling procedure. The statistical sampling procedures are further described in Section 4.4.

Task 5. Characterize Waste Management Units

The representative units selected in Task 4 were characterized in this task to develop the physical and operating parameter values that are used in source partition modeling and air dispersion modeling (e.g., surface area). The representative WMUs selected in Task 4 were divided into various strata (e.g., statistical groupings of data) based on surface area. Treatment tanks were stratified by both surface area and height. These strata define the physical dimensions of the representative WMUs (i.e., surface area and, for tanks, height) used in air dispersion modeling. Next, the operating characteristics needed for the source partition modeling were characterized (e.g., capacity of the WMU) for each of the representative 68 landfills, 200 treatment tanks, and 200 surface impoundments selected.

Task 6a. Construct Source Data Files (Location-WMU Combinations)

Constructing the 10,000-record source data files for use in the probabilistic analysis involved first combining the location data and WMU data. First, the location selected in Task 2 was replicated to produce 10,000 records based on weights assigned to each location. These weights were based on the amount of paint manufacturing activity in the various states. Second, the representative WMUs selected in Task 3 were replicated to produce 10,000 records for each WMU type. The replication of representative WMUs was based on weights assigned to the surface area or surface area-height strata. These weights were based on the sampling procedure and were designed to ensure that the 10,000 WMU records reflected the distribution of sizes in the original database. The 10,000 location records then were randomly combined with the 10,000 WMU records to produce the source data files. Three source data files were generated in this manner: one for landfills, one for treatment tanks, and one for surface impoundments. Each record in the source data files was identified by a model run identification number.

Task 6b. Construct Source Data Files (Add Waste Streams)

The three source data files were then combined with the waste stream data to define the waste management scenarios that were evaluated in the paints listing risk assessment. The landfill source data file was used to evaluate two waste streams: emission control dust and combined waste. The tank and surface impoundment source data files were used to evaluate wastewaters. Table 4-1 lists the source data file and waste streams for each scenario.

4.2 Waste Management Scenario Development

As discussed in the previous section, the first task in designing a risk assessment for paint manufacturing waste and waste management practices was to define the waste management

Table 4-1. Waste Management Scenarios Modeled

Source Data File	Waste Stream
Landfill	Emission control dust
Landfill	Sludge
Treatment tank	Wastewater
Surface impoundment	Wastewater

scenarios to be evaluated. A waste management scenario is made up of a waste stream disposed of in a type of waste management unit in a particular location. Section 3.1 describes in detail the waste management scenarios evaluated in this initial risk assessment. They are summarized here.

4.2.1 Characterization of Waste Streams

For the risk assessment, paint manufacturing waste streams were categorized into three major groups based on their physical characteristics:

- Aqueous waste
- Combined solids
- Emission control dust.

Each of the paint manufacturing waste streams was assumed to be disposed of in offsite disposal units. It was assumed that the paint wastes were mixed with other nonpaint wastes in the disposal unit. Thus, bulk waste parameters (e.g., bulk density, pH, and fraction organic carbon) required to estimate emissions using the source models were parameterized using generic industrial waste characteristics. Whenever possible, distributions were used to characterize the variability in waste parameters. These parameters, as well as the distributions used, are provided in Appendix L. In evaluating the landfill model, however, it was determined that the calculation of the particulate emission rate due to direct disposal of the waste on the landfill is reflective of the waste stream itself (i.e., the waste disposed of in the landfill was not immediately mixed in with other nonpaint wastes). The equation used to calculate the particulate emission rate due to direct disposal requires a value for moisture content (U.S. EPA, 1985, AP-42). Moisture content was parameterized based on industry data on sludge moisture content as provided by EPA (PPG Industries, Inc., 1999). A uniform distribution ranging from 0 to 15 percent was used for dust, and a uniform distribution ranging from 25 to 85 percent was used for combined solids.

The volume of disposed waste is required to calculate the concentration of a constituent disposed of in a WMU. Based on a survey of waste volumes produced by the paints industry (i.e., the 3007 survey), distributions of waste volume were provided by EPA for aqueous waste, emission control dust, and combined solids. For each waste type, a discrete distribution of waste volumes was provided along with corresponding weighting factors (Appendix S). Bulk density data were also required for emission control dust and combined solids. These data were also provided by EPA based on the results of the 3007 survey (Table 4-2). It should be noted that, for tanks and surface impoundments, the bulk density was assumed to be constant at 1 g/cm^3.

Using the waste volume data, the fraction of paint waste in the WMU (f_wmu) was calculated. The fraction was then used to calculate target waste concentrations in each waste stream. For landfills, f_wmu was defined as the ratio of the annual waste volume to the annual capacity of the WMU. If the waste volume was greater than the capacity of the landfill, the waste volume was set equal to the landfill capacity, thus resulting in an f_wmu equal to 1. For surface impoundments and tanks, f_wmu was calculated using the amount of waste processed annually in the WMU (i.e., the annual flow rate). As with landfills, if the annual amount of waste disposed of in the surface impoundment or tank was greater than the annual flow rate, the value was set equal to the flow rate, resulting in an f_wmu equal to 1.

4.2.2 Waste Management Units

Three types of WMUs were selected for evaluation based on information gathered by EPA. WMUs used for disposal of nonhazardous waste streams by the paint manufacturing industry include

- Industrial landfills
- Treatment tanks
- Surface impoundments.

Because of the large number of paint manufacturing facilities located across the United States, national databases were used to characterize WMUs accepting paint waste for disposal. The data sources used for characterizing each type of WMU are described in detail in Section 4.3.

Table 4-2. Distributions of Waste Stream Data

	Min	10%	50%	90%	Max
Annual Waste Stream Volumes from 3007 Survey (m^3/yr)					
Dust	0.15	0.38	2.44	220.8	297.7
Combined solids	0.02	0.15	1.42	163.8	1,615.4
Aqueous waste	0.57	1.14	45.42	101.3	394.5
Bulk Density (g/cm^3)					
Dust	0.22	0.36	1.40	2.70	3.59
Combined solids	0.061	0.36	1.32	2.40	3.59

4.2.3 Constituents

EPA selected 43 constituents of concern (COCs) for evaluation in this risk assessment—16 metal constituents and 27 organic constituents. The constituents of concern are listed in Table 3-2 along with their CAS numbers. Physical and chemical properties for each of the constituents evaluated in this risk assessment have been identified and are presented in Appendix D. Distribution coefficients (i.e., coefficients, k_d) for the metals are presented in Appendix H.

4.3 Site Characterization

The site characteristics used in this analysis were based on two conceptual site layouts and regional characterization of environmental parameters. The conceptual site layouts define the area in the immediate vicinity of the WMU. They also define the geographic relationship among important features such as the WMU boundary, agricultural field, resident location, and streams. There are two conceptual site layouts used in this analysis that are evaluated at each of the 49 locations selected for the analysis, all located within the continental United States. These 49 locations were selected to geographically represent the distribution of paint manufacturing facilities and, therefore, the geographic distribution of WMUs that could be used for disposal of paint wastes. These locations were used to capture national variability in meteorology, soils, climate, and aquifers.

4.3.1 Conceptual Site Layouts

This risk assessment was based on three site layouts that are conceptual rather than site-specific. The site layouts were designed to capture possible relationships between a WMU and individual receptors. Geographic features that are important for determining human and ecological exposures to chemicals released from the WMU (e.g., agricultural field, waterbody) were located relative to the WMU boundary.

The conceptual or general site layouts are shown in Figures 4-2 through 4-5. Shown in these figures are the WMU boundaries, the buffer area (i.e., an area between the WMU and the nearest human receptor), the agricultural field, the waterbodies, and the resident location. The site layouts were used to model three possible land use scenarios that may exist in areas surrounding the WMUs:

- Residential aboveground scenario (Figure 4-2)
- Residential groundwater scenario (Figure 4-3)
- Agricultural scenario (Figure 4-4)
- Fisher scenario (Figure 4-5).

The WMU size (D_r), the distance to the resident receptor (D_b), the size of the agricultural/field (S_{ag}), and the length of the waterbody were all varied as part of the Monte Carlo analysis.

4.3.1.1 <u>WMU Boundaries</u>. The WMU is represented as a circular source. The size of the source is determined by the surface area of the WMU. Section 4.4 describes the surface area

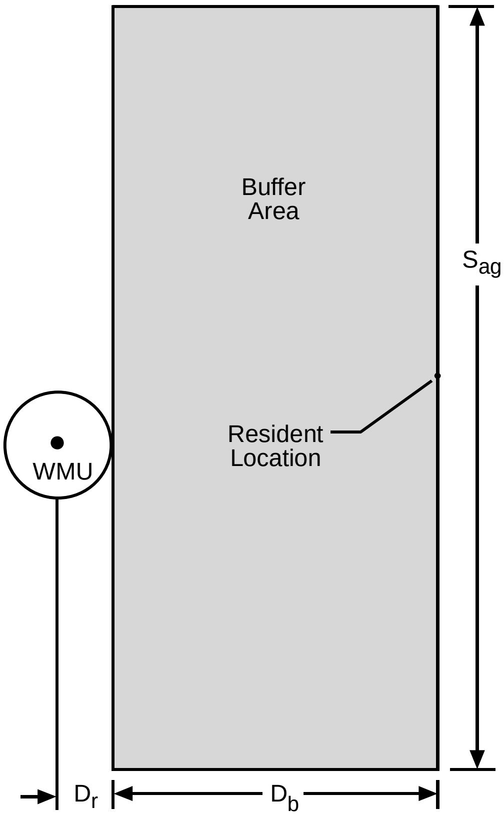

Figure 4-2. Conceptual site layout for residential aboveground scenario.

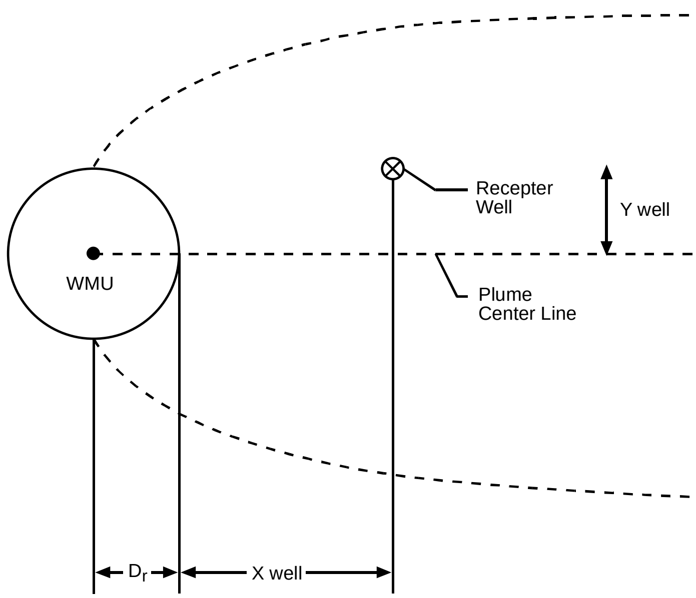

Figure 4-3. Conceptual site layout for residential groundwater scenario.

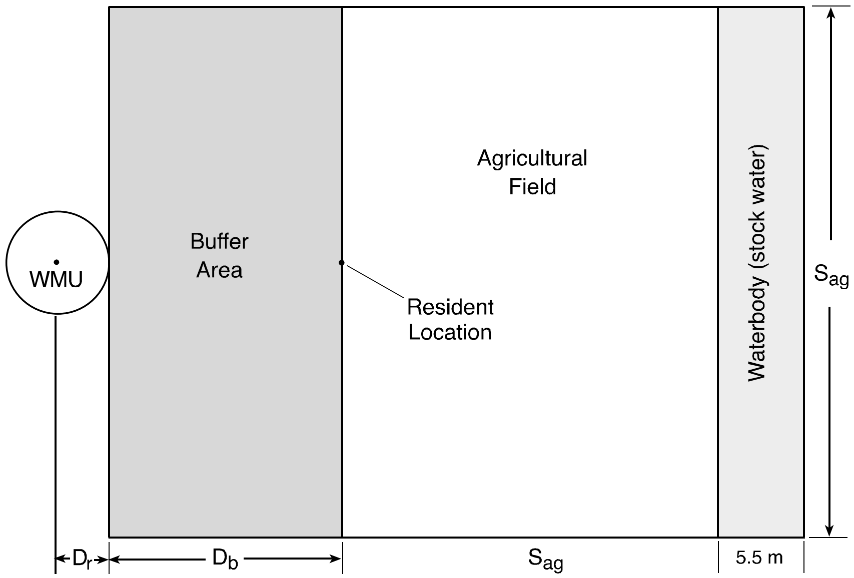

Figure 4-4. Conceptual site layout for agricultural scenario.

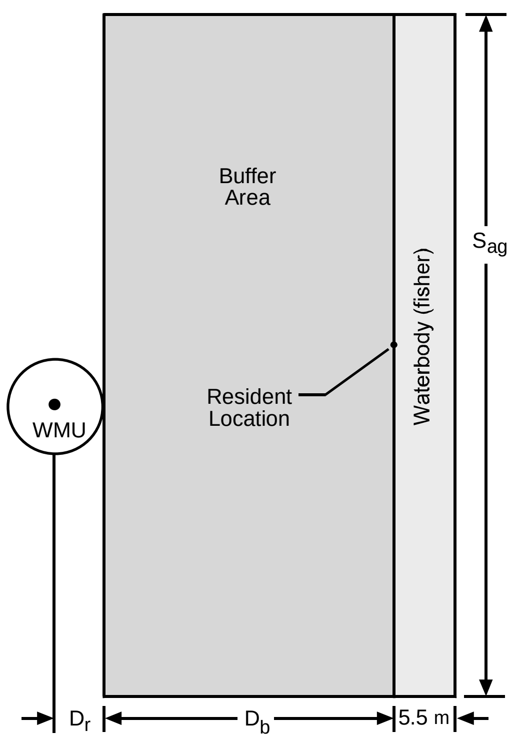

Figure 4-5. Conceptual site layout for fisher scenario.

characteristics of the WMU and the methods used to represent the different WMU surface areas by dividing the universe of WMUs into statistical strata that span the range of surface areas. This was done for each WMU type (landfills, tanks, and surface impoundments). The surface areas associated with each of the strata are inputs to the air dispersion model (see Section 5.2).

The WMU is assumed to be located on the property line of the facility to which it belongs. Adjacent to the WMU is a buffer area within which there is assumed to be no human activity that would present human risk. That is, there are no residences, agricultural activities, or fishing activities within the buffer. The buffer area lies between the WMU boundary and the resident location, agricultural field, or waterbody, depending on the scenario being modeled.

4.3.1.2 Residential Scenarios. There are two residential scenarios, one for the above-ground pathway and one for the groundwater pathway. Two separate fate and transport scenarios were established for the groundwater and air pathways. The decision to use separate scenarios for the two pathways was based on differences in time frame and location of exposure.

Residential Scenarios–Aboveground Pathway. The residential scenario was used to estimate risks to receptors (i.e., adult and child residents) living in the vicinity of the WMUs who obtained all food items from nonlocal sources (i.e., noncontaminated). Surveys conducted to support the *Hazardous Waste Treatment, Storage, and Disposal Facilities - Organic Air Emissions Standards for Process Vents and Equipment Leaks Final Rule* (55 FR 25454) have shown that the closest residence to a WMU boundary is approximately 75 m, and the median or central tendency distance from the WMU boundary is approximately 300 m. The 75-m distance is taken to be the 10^{th} percentile closest distance. Using these values, a normal distribution of resident locations was developed for the Monte Carlo analysis. It has a median value of 300 m, 75 m for the 10^{th} percentile closest distance and 525 m for the 90^{th} percentile farthest distance. The distance from the WMU boundary to the resident location was selected from this distribution for each iteration of the Monte Carlo analysis. Values selected were constrained to be between 50 and 550 m so as to avoid extreme values that would be inconsistent with the general scenario described by the site layout.

This site layout must also be oriented in terms of direction. In this assessment, the site layout is oriented along a randomly selected direction. That is, the centerline of the site layout is randomly varied from 1 to 360 degrees around the WMU. Thus, the resident location is determined by selecting a distance from the WMU boundary and the number of degrees varying from due north. Therefore, the resident location can be anywhere around the WMU between 50 and 550 m from the WMU boundary.

Residential Scenarios–Groundwater Scenario. Residential groundwater exposure is calculated based on residential use of well water. The receptor well is placed at a downgradient distance up to 1 mile, based on a nationwide distribution of nearest downgradient residential wells from Subtitle D municipal landfills (i.e., x-well) (U.S. EPA, 1988). This distribution is provided in Table 4-3. It is assumed that the same distribution holds for other types of waste units as well (i.e., industrial nonhazardous waste landfills and surface impoundments). The limits on the lateral direction from the plume centerline (i.e., y-well) and depth below the water table (i.e., z-well) of the well are discussed in Section 5.2.4.

Table 4-3. Distribution of Receptor Well Distance

Percentile	x-distance (m)
Minimum	0.6
10	104.0
20	183.0
30	305.0
40	366.0
50 (median)	427.0
60	610.0
70	805.0
80	914.0
90	1,220.0
Maximum	1,610.0

4.3.1.3 Agricultural Scenario. The agricultural scenario was used to estimate risks to receptors (i.e., adult and child farmers) living in the vicinity of the WMU who obtain a portion of their diet from food grown on land adjacent to the WMU. Receptors in the agricultural scenario also consume animal products from beef and dairy cattle raised on the agricultural field. The location of the residence in which the farmers live is determined in the same manner as for the residential scenario (see Section 4.3.1.2). The agricultural field is located near the WMU beginning where the buffer area ends (see Figure 4-4). The agricultural field is assumed to be a square with a surface area determined by averaging the median size of agricultural fields located in counties within a 20-mile radius of the locations modeled (see Section 4.3.2.1 for selection of locations). Therefore, the agricultural field size is different for each of the 49 locations modeled for this assessment and accounts for variation in agricultural field sizes within the continental United States.

The agricultural field sizes were taken from the Census of Agriculture. The Census of Agriculture (U.S. Department of Commerce, 1989, 1994) provides periodic and comprehensive statistics about agricultural operations, production, operators, and land use. It is conducted every 5 years for years ending in 2 and 7. Its coverage includes all operators of U.S. farms or ranches (Division A, SIC 01-02) that sold or normally would have sold at least $1,000 worth of agricultural products during the census year. In 1992, approximately 1.9 million operators produced $162 billion in crops and livestock. All operators report crop acreage and quantities harvested in addition to other information. Census of Agriculture data used for this analysis included county-level data on beef and dairy farms. Data for 1987 and 1992 were averaged. The agricultural field sizes used in this analysis are presented in Table 4-4.

Table 4-4. Agricultural Field Sizes by Location

Meteorological Station Identification Number	City	State	Farm Size (acres)	Farm Size (m^2)
14735	Albany	NY	143.7	581,400.6
13874	Atlanta	GA	48.0	194,381.3
93721	Baltimore	MD	45.0	182,237.7
14739	Boston	MA	43.8	177,267.3
94018	Boulder	CO	103.4	418,442.0
94846	Chicago	IL	85.2	344,876.1
14820	Cleveland	OH	42.8	173,110.3
13883	Columbia	SC	107.7	435,797.1
14821	Columbus	OH	115.4	467,054.2
14933	Des Moines	IA	155.3	628,496.9
94847	Detroit	MI	55.9	226,373.1
3927	Fort Worth	TX	51.2	207,167.8
14898	Green Bay	WI	149.6	605,509.9
13723	Greensboro	NC	61.0	247,058.0
3870	Greenville	SC	67.9	274,695.5
94860	Grand Rapids	MI	93.7	378,990.7
14740	Hartford	CT	53.6	217,089.6
12960	Houston	TX	55.2	223,483.3
3860	Huntington	WV	89.5	362,278.1
3856	Huntsville	AL	66.4	268,785.2
93819	Indianapolis	IN	86.5	350,008.9
3940	Jackson	MS	118.6	479,903.5
93820	Lexington	KY	81.0	327,795.0
13963	Little Rock	AR	134.6	544,533.2

(continued)

Table 4-4. (continued)

Meteorological Station Identification Number	City	State	Farm Size (acres)	Farm Size (m^2)
23174	Los Angeles	CA	8.3	33,426.6
13893	Memphis	TN	325.5	1,317,287.9
12839	Miami	FL	2.0	7,952.0
14922	Minneapolis	MN	76.4	309,227.7
13897	Nashville	TN	74.7	302,294.1
12916	New Orleans	LA	71.0	287,267.3
94728	New York	NY	14.1	56,955.0
14734	Newark	NJ	15.7	63,479.4
13737	Norfolk	VA	85.8	347,316.1
13967	Oklahoma City	OK	133.6	540,661.3
13739	Philadelphia	PA	43.0	174,024.5
23183	Phoenix	AZ	189.0	764,989.7
94823	Pittsburgh	PA	87.0	352,179.6
24229	Portland	OR	34.3	138,644.3
14765	Providence	RI	48.7	196,905.9
23185	Reno	NV	87.7	355,106.9
13740	Richmond	VA	112.6	455,580.7
13741	Roanoke	VA	105.2	425,538.1
94822	Rockford	IL	178.3	721,447.5
23234	San Francisco	CA	60.8	246,122.9
24233	Seattle	WA	20.9	84,580.1
14848	South Bend	IN	97.4	394,045.6
13994	St. Louis	MO	118.9	481,077.1
12842	Tampa	FL	24.0	97,207.9
3928	Wichita	KS	257.5	1,042,055.0

Adjacent to the agricultural field is a waterbody that is used as a drinking water source for livestock. The waterbody is assumed to be a rectangle 5.5 m wide and 0.21 m deep. These values are typical of a third-order stream (van der Leeden et al., 1990). The stream length is determined by the width of the agricultural field. Surface area of the stream is, therefore, determined by the fixed width (5.5 m) and the size of the agricultural field, which varies by meteorological station location as mentioned above.

4.3.1.4 <u>Fisher Scenario</u>. The fisher scenario was used to estimate risks to receptors (i.e., adult fishers) living in the vicinity of the WMUs who caught and consumed fish on a recreational basis from a waterbody located adjacent to the buffer. The location of the residence in which the fishers live is determined in the same manner as for the residential scenario (see Section 4.3.1.2). The waterbody is assumed to be a stream located downwind of the WMU, beginning where the buffer area ends (see Figure 4-5). The waterbody is assumed to be a rectangle 5.5 m wide and 0.21 m deep. These values are typical of a third-order stream (van der Leeden et al., 1990). The stream length is determined by the width of the agricultural field. Surface area of the stream is, therefore, determined by the fixed width (5.5 m) and the size of the agricultural field, which varies by meteorological station location (see Section 4.3.1.3).

4.3.2 Regional Environmental Setting

The purpose of the paints listing risk assessment was to develop national distributions of waste concentrations and leachate concentrations that would be protective of human health. The assessment was conducted using a fixed conceptual site model that could exist anywhere in the continental United States. Other parameters that would affect the results of this risk assessment are those that reflect regional environmental conditions (e.g., meteorology, soil characteristics, and groundwater hydrology), differences in WMU design, and differences in waste stream characteristics. The following sections describe the selection of parameter values used to describe the environmental setting used in this risk assessment.

The United States is characterized by differences among regions in climatic, soil, and groundwater regimes. Because specific paint waste disposal locations were not known, this analysis characterized environmental conditions based primarily on the paint production volume by state. In doing so, it was assumed that nonhazardous waste from paint manufacturing facilities would be disposed of within reasonable transport distances of the facility. The characterization of environmental setting, therefore, began with the selection of meteorological stations from those states within which paint manufacturing activity occurs. The meteorological stations provided much of the data needed for air dispersion modeling. Once these meteorological stations were selected, their locations were used to characterize climate, soils, and aquifers. These locations were then used to create the 10,000 location records needed for the source data files that support the probabilistic analysis.

4.3.2.1 <u>Meteorological Station Locations.</u> Selecting meteorological stations for use in this risk assessment consisted of three steps:

- Select states where paint waste is disposed of.

- Select meteorological stations to represent each state.
- Weight each meteorological station based on the amount of manufacturing in each state.

Select States Where Paint Waste Is Disposed of. The list of states to include in this study came from the 1997 Economic Census of Paint and Coating Manufacturing (U.S. Department of Commerce, 1999). The Census reported the dollar value of shipments made by paint manufacturing facilities by state. This information was used as a surrogate for the actual waste volumes, assuming that the dollar amount of paint produced would be an indicator of the amount of waste generated. In all, 36 states reported paint production volumes on a dollar value basis. The total value of shipments was $18,938,172,000. The value of shipments and percent distribution for each state were calculated and are presented in Table 4-5.

It should be noted that the 1997 Census included only states for which facility data can be reported. Data cannot be reported if the population of paint manufacturing facilities is so small that confidentiality could not be maintained if data were reported on a state level. Some states, of course, did not have any paint manufacturing facilities; for confidentiality reasons, others reported no production volume data even though, according to the 1997 Toxics Release Inventory, paint manufacturing facilities existed in these states. Because no paint production data were reported in the Census data, however, these states were not included in this analysis. For the most part, these states contained only one or two paint manufacturing facilities; thus, including them would have had no significant impact on the analysis.

Select Meteorological Stations to Represent Each State. To identify stations, locations of paint manufacturing facilities were obtained from the 1997 Toxics Release Inventory as shown in Figure 4-6. These locations were then compared to locations of meteorological stations across the contiguous United States.

In many states, the majority of paint manufacturing facilities were located in clusters in the immediate vicinity of a meteorological station. In these cases, the choice of which meteorological station to select was clear. In contrast, other states were characterized by facilities dispersed throughout the state, thus the choice of which meteorological station to select was not as clear. In these cases, meteorological stations were selected to ensure that the risk assessment contained a wide range of different meteorological and climate conditions. Meteorological stations located in regions of a state containing a relatively small number of facilities were not selected. The approach was also consistent with the goal of the analysis, which was to consider facilities on a national basis rather than attempting to evaluate every paint manufacturing facility in the country.

Also under consideration in the selection of a meteorological station was whether meteorological data were already processed for use in air dispersion models. An effort was made to select stations for which meteorological data had been processed previously. For example, if many paint facilities were located in a meteorological region that did not have any processed data, then adjacent stations in the same state that had similar conditions were considered. In some cases, nearby meteorological stations had significant differences in climatology (i.e.,

Table 4-5. Paint Manufacturing Activity by State Based on 1997 Census Data

State	Value of Shipments ($1,000)	Total Shipments (%)
Alabama	270,582	1.43
Arizona	62,161	0.33
Arkansas	168,165	0.89
California	1,772,812	9.36
Colorado	96,061	0.51
Connecticut	162,710	0.86
Florida	467,079	2.47
Georgia	706,400	3.73
Illinois	2,289,705	12.09
Indiana	419,762	2.22
Iowa	344,925	1.82
Kansas	55,100	0.29
Kentucky	642,322	3.39
Louisiana	102,654	0.54
Maryland	587,013	3.10
Massachusetts	283,806	1.50
Michigan	1,232,501	6.51
Minnesota	93,895	0.50
Mississippi	185,704	0.98
Missouri	585,263	3.09
Nevada	33,671	0.18
New Jersey	931,857	4.92
New York	246,808	1.30
North Carolina	582,872	3.08
Ohio	2,296,331	12.13
Oklahoma	88,988	0.47
Oregon	122,140	0.64
Pennsylvania	1,016,830	5.37
Rhode Island	26,367	0.14
South Carolina	42,739	0.23
Tennessee	276,333	1.46
Texas	1,327,809	7.01
Virginia	507,807	2.68
Washington	177,685	0.94
West Virginia	34,466	0.18
Wisconsin	696,849	3.68

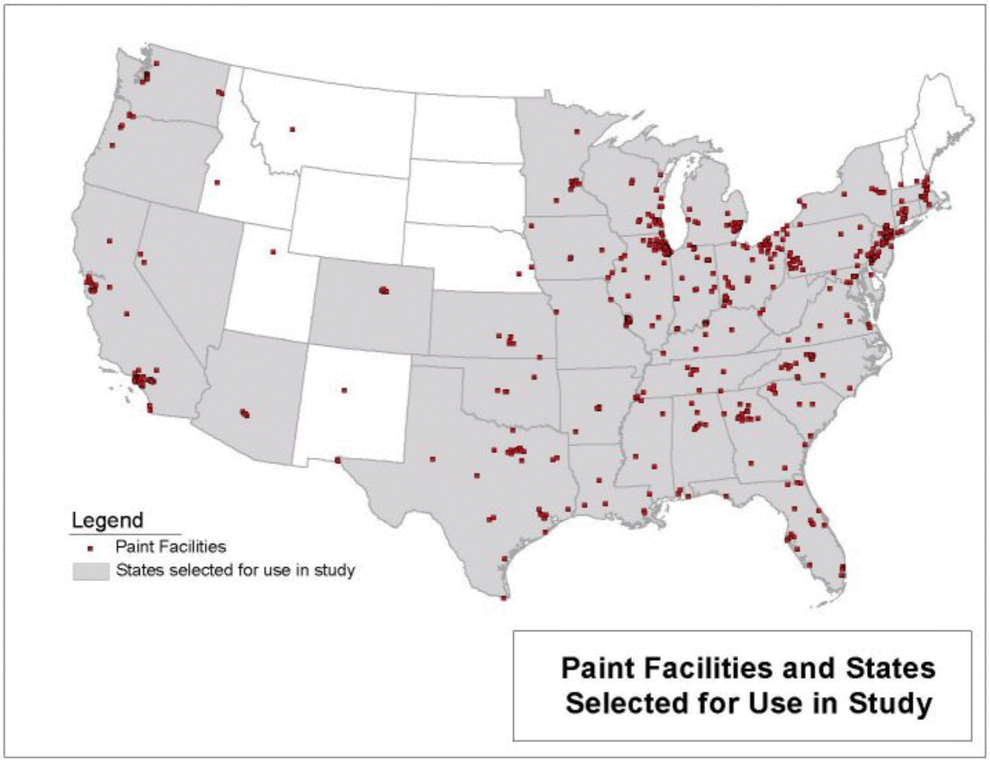

Figure 4-6. Locations of paint manufacturing facilities.

coastal versus inland climatology), in which case, the best meteorological station was selected and the new data were processed for use in air dispersion modeling.

In all, 49 meteorological stations were selected. The locations of these stations are provided in Figure 4-7. Table 4-6 provides the name of each meteorological station and a short description of the rationale used to select stations for each state.

Weight Each Meteorological Station Based on the Amount of Manufacturing in Each State. The percentage of total value of shipments provided in Table 4-5 was used to weight each meteorological location for the Monte Carlo analysis. If there was only one meteorological station selected within a state, then the percentage values reported in Table 4-6 were used as the weighting factor. If more than one meteorological station within a state was selected, then the percentage values reported in Table 4-6 were evenly divided among the meteorological stations and used as the weighting factor. For example, if a state has 10 percent of the national paint manufacturing activity, that state's meteorological stations would be represented in 10 percent of the 10,000 Monte Carlo iterations. If there were two meteorological stations in the state, each would be represented in 5 percent of the 10,000 Monte Carlo iterations. Therefore, within the 10,000 iterations of the probabilistic analysis, greater weight is given to meteorological stations representing states in which greater paint manufacturing activity occurred.

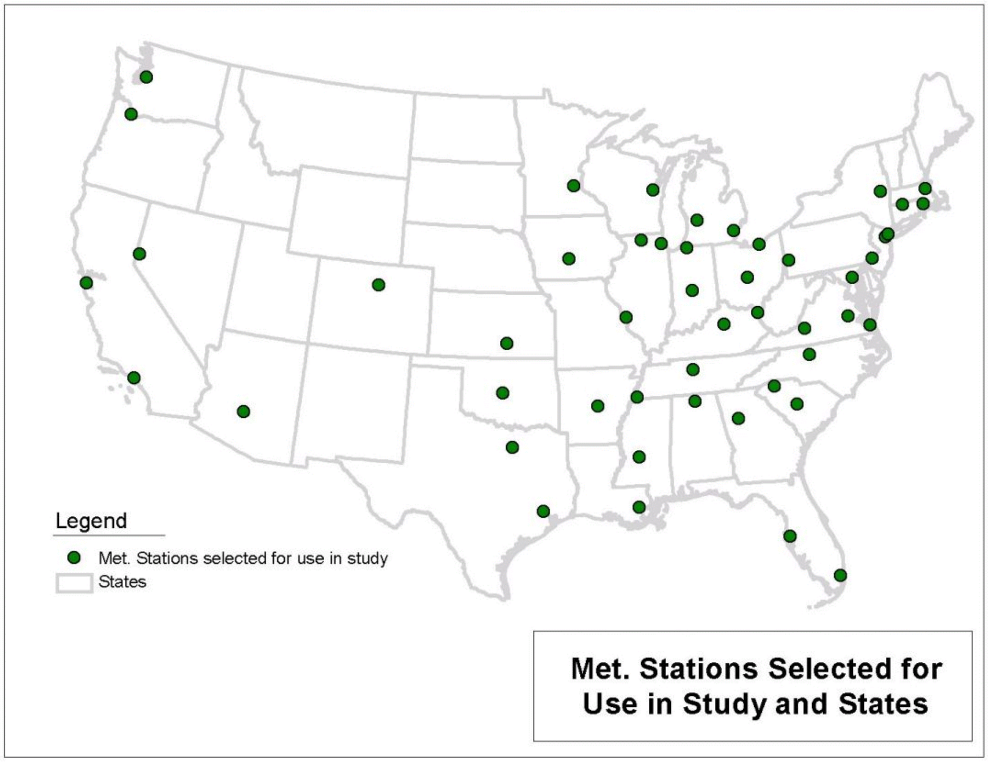

Figure 4-7. Locations of meteorological stations.

4.3.2.2 <u>Climate Data</u>. Meteorological stations selected for purposes of air dispersion modeling also provided climatic data that were necessary for source partition modeling or fate and transport modeling. For each of the 49 stations, the following data were compiled:

- Mean annual wind direction
- Mean annual windspeed
- Average temperature
- Average annual runoff
- Universal Soil Loss Equation (USLE) rainfall/erosivity factor.

4.3.2.3 <u>Soil Characterization</u>. The fate and transport models used in the paints risk assessment require surface soil properties to model erosion and overland transport and properties of the entire soil column to model leachate transport through the vadose zone to groundwater. As for meteorological and aquifer data, a regional approach was used to compile soil data for these modeling requirements. For this purpose, soils around the 49 meteorological stations were characterized. This regional characteristic of soil types captured variability in soils in a manner that is generally representative of paint manufacturing sites across the United States. A geographic information system (GIS) was used to compile soil texture and other soil data within a 20-mile radius around each meteorological station. Then database programs processed these data to create the input variables required by the models.

Table 4-6. Meteorological Stations for Each State

State	City	Percent Weight (%)	Meteorological Station ID	Rationale
Alabama	Huntsville	1.43	03856	The majority of the paint facilities in Alabama were located in the area containing the Huntsville meteorological station.
Arizona	Phoenix	0.33	23183	All paint manufacturing facilities in Arizona were located in a cluster around Phoenix.
Arkansas	Little Rock	0.89	13963	A small cluster of three facilities was located near Little Rock. One additional facility was located on the edge of the region associated with Little Rock that would be expected to have similar meteorological conditions.
California	Los Angeles San Francisco	4.68 4.68	23174 23234	Two clusters of paint manufacturing facilities in California represented the majority of facilities in the state. Data were available for both meteorological stations.
Colorado	Boulder	0.51	94018	One cluster of paint facilities was located in Colorado around Boulder.
Connecticut	Hartford	0.86	14740	Only one meteorological station was located in Connecticut, in Hartford, which coincided with the location of the majority of the paint facilities in the state.
Florida	Miami Tampa	1.23 1.23	12839 12842	Facilities were located throughout Florida on both the east and west coast. Thus a meteorological station was included from each coast, specifically, Miami and Tampa.
Georgia	Atlanta	3.73	13874	A large cluster of facilities was located near Atlanta. The majority of the facilities in the state were located in this region.
Illinois	Chicago Rockford	6.05 6.05	94846 94822	A very large cluster of facilities was located in the region around Chicago. Several facilities were also located near Rockford, which is directly adjacent to the area surrounding Chicago. It is possible that WMUs would be located in a region near an urban area. Thus, the meteorological region for Rockford was included since it is expected to have different meteorological conditions, being farther from Lake Michigan.

(continued)

Table 4-6. (continued)

State	City	Percent Weight (%)	Meteorological Station ID	Rationale
Indiana	South Bend Indianapolis	1.11 1.11	14848 93819	Several small clusters were present throughout Indiana. However, no one region contained a majority of facilities. Both Indianapolis and South Bend were chosen to represent the two major types of meteorology in the state.
Iowa	Des Moines	1.82	14933	Two of the three facilities in Iowa were located near Des Moines. The remaining facility was located farther west but was not expected to have dramatically different meteorological conditions.
Kansas	Wichita	0.29	03928	All but one of the paint manufacturing facilities in Kansas were located near Wichita.
Kentucky	Lexington	3.39	93820	The majority of the facilities in Kentucky were located in Louisville. However, processed data were not available for this station. Lexington had processed data available and was expected to have similar climatology.
Louisiana	New Orleans	0.54	12916	There was no single cluster of facilities in Louisiana. However, four out of the five facilities in the state were in the southern portions and were thought to be represented well with meteorological data from New Orleans.
Maryland	Baltimore	3.10	93721	Only one meteorological station was in Maryland, in Baltimore, which coincided with the location of the majority of the paint facilities in the state.
Massachusetts	Boston	1.50	14739	Only one meteorological station was in Massachusetts, in Boston, which also coincided with the location of the majority of the paint facilities in the state.
Michigan	Detroit Grand Rapids	3.25 3.25	94847 94860	One large cluster of facilities was located near Detroit. Several other facilities were located farther west and had different climatology. Thus, Grand Rapids was included in the analysis to account for this meteorology.
Minnesota	Minneapolis	0.50	14922	All but one of the facilities in Minnesota were located in the region represented by the Minneapolis meteorological station.

(continued)

Table 4-6. (continued)

State	City	Percent Weight (%)	Meteorological Station ID	Rationale
Mississippi	Jackson	0.98	03940	Only two paint facilities were located in Mississippi—near Jackson and in the region represented by Meridian. However, the latter facility was on the border for the Jackson region, which also was the source of the upper air data.
Missouri	St. Louis	3.09	13994	All but one of the facilities in Missouri were located in a cluster around St. Louis.
Nevada	Reno	0.18	23185	Nevada had only two paint facilities, both located in Reno. Processed data were not available for this meteorological station.
New Jersey	Newark	4.92	14734	One large cluster of facilities was located in New Jersey around the Newark area.
New York	Albany New York	0.65 0.65	14735 94728	The majority of facilities were located around the New York City area. Another smaller cluster of facilities was located near Albany, which has different climatology than New York.
North Carolina	Greensboro	3.08	13723	The majority of the facilities in North Carolina were located in the areas represented by Greensboro and Charlotte. Since these areas would have somewhat similar meteorology and data were available only for Greensboro, this area was chosen to represent the state.
Ohio	Cleveland Columbus	6.06 6.06	14820 14821	The largest cluster of paint facilities in Ohio was located in Cleveland. Several small clusters were present farther inland. Thus, the meteorological station in Columbus was chosen to represent this meteorology.
Oklahoma	Oklahoma City	0.47	13967	Four paint manufacturing facilities were located in Oklahoma, three of which were in Oklahoma City, which would have similar meteorology to the station in Tulsa.
Oregon	Portland	0.64	24229	Paint facilities in Oregon were located in three different meteorological regions with no one major cluster. However, all three regions would have similar meteorological conditions. Most of the facilities were located near Portland and processed meteorological data were available for that station.

(continued)

Table 4-6. (continued)

State	City	Percent Weight (%)	Meteorological Station ID	Rationale
Pennsylvania	Pittsburgh Philadelphia	2.68 2.68	94823 13739	The majority of facilities in Pennsylvania were located near Pittsburgh and Philadelphia. These areas have different meteorological conditions, therefore both were included.
Rhode Island	Providence	0.14	14765	Only one meteorological station was in Rhode Island, in Providence, which coincided with the location of the majority of the paint facilities in the state.
South Carolina	Columbia Greenville	0.11 0.11	13883 3870	A small cluster of facilities was located near Greenville. Two facilities were also located in Charlotte, which was expected to have somewhat different meteorology since it is much farther from the mountains.
Tennessee	Nashville Memphis	0.73 0.73	13897 13893	The majority of facilities in Tennessee were located in two clusters around Memphis and Nashville. Both meteorological stations were included since the conditions around them are expected to be different.
Texas	Houston Fort Worth	3.51 3.51	12960 03927	Two clusters of facilities were located in Texas, one in Houston and one in Fort Worth. These areas have different meteorological conditions since one is coastal and one is farther inland.
Virginia	Norfolk Roanoke Richmond	0.89 0.89 0.89	13737 13741 0.89	A couple of paint facilities were located in three different meteorological regions in Virginia, all of which would have unique climatological conditions. Thus, all three areas were included in the analysis using the data from Roanoke, Norfolk, and Richmond.
Washington	Seattle	0.94	24233	The majority of paint facilities in Washington were clustered around Seattle.
West Virginia	Huntington	0.18	03860	The only facilities in West Virginia were located in the Huntington region.
Wisconsin	Green Bay	3.68	14898	Most of the facilities in Wisconsin were located around Milwaukee. Several other facilities were located in the region represented by the Green Bay meteorological station. Since both areas are expected to have similar meteorological conditions and processed data were available only for Green Bay, it was used to represent the entire state.

Figure 4-8 depicts the soil data collection process, showing data sources, processing steps, and final variables prepared as model inputs. Soil properties are listed by data source and model in Appendix I.

Data Sources. The primary data source for soil properties is the State Soil Geographic (STATSGO) database. STATSGO is a repository of nationwide soil properties primarily compiled by the U.S. Department of Agriculture (USDA) from county soil survey data (USDA, 1994). STATSGO includes a 1:250,000-scale GIS coverage that delineates soil map units and an associated database containing soil data for each STATSGO map unit. (Map units are areas used to spatially represent soils in the database.)

In addition, two compilations of STATSGO data, each keyed to the STATSGO map unit GIS coverage, were used in the analysis as a convenient source of average soil properties:

- USSOILS. USSOILS (Schwarz and Alexander, 1995) averages STATSGO data over the entire soil column for each map unit.

- CONUS. CONUS (Miller and White, 1998) provides average STATSGO data by map unit and a set of 11 standardized soil layers.

Soil properties derived directly from STATSGO, CONUS, or USSOILS data include organic matter content, USLE K (erodibility) and S (slope) factors, and pH. A complete set of hydrological soil properties[1] was not available from STATSGO. To ensure consistent and realistic values, it was necessary to rely on established, nationwide relationships between hydrologic properties and soil texture or hydrologic soil group, both of which are available from STASTGO. Sources for these relationships include Carsel and Parrish (1988), Carsel et al. (1988), and Clapp and Hornberger (1978). These peer-reviewed references provide a consistent set of correlated hydrologic properties for each soil texture or hydrologic group.

Finally, two parameters—root zone depth and Soil Conservation Source (SCS) curve number (used for recharge calculations)—required site-based land use data as well as soil texture or hydrologic soil group. The land use data were obtained for each of the 49 locations from the Geographic Retrieval and Analysis System (GIRAS) land use database (U.S. EPA, 1994). GIRAS provides comprehensive land use data, in digital GIS format, for the conterminous United States. Land use/land cover information in GIRAS was mapped and coded using the Anderson classification system (Anderson et al., 1976), which is a hierarchical system of land use characterizations. This nationwide coverage is based on late-1970s to early-1980s satellite images and aerial photography. The relationships used to convert the land use and soil data were obtained from Dunne and Leopold (1978) for root zone depth and USDA (1986) for SCS curve number.

[1] Hydrological soil properties required for modeling include bulk density, saturated water content, residual water content, field moisture content, wilting point, saturated hydraulic conductivity, soil moisture coefficient *b*, and soil moisture retention parameters alpha and beta.

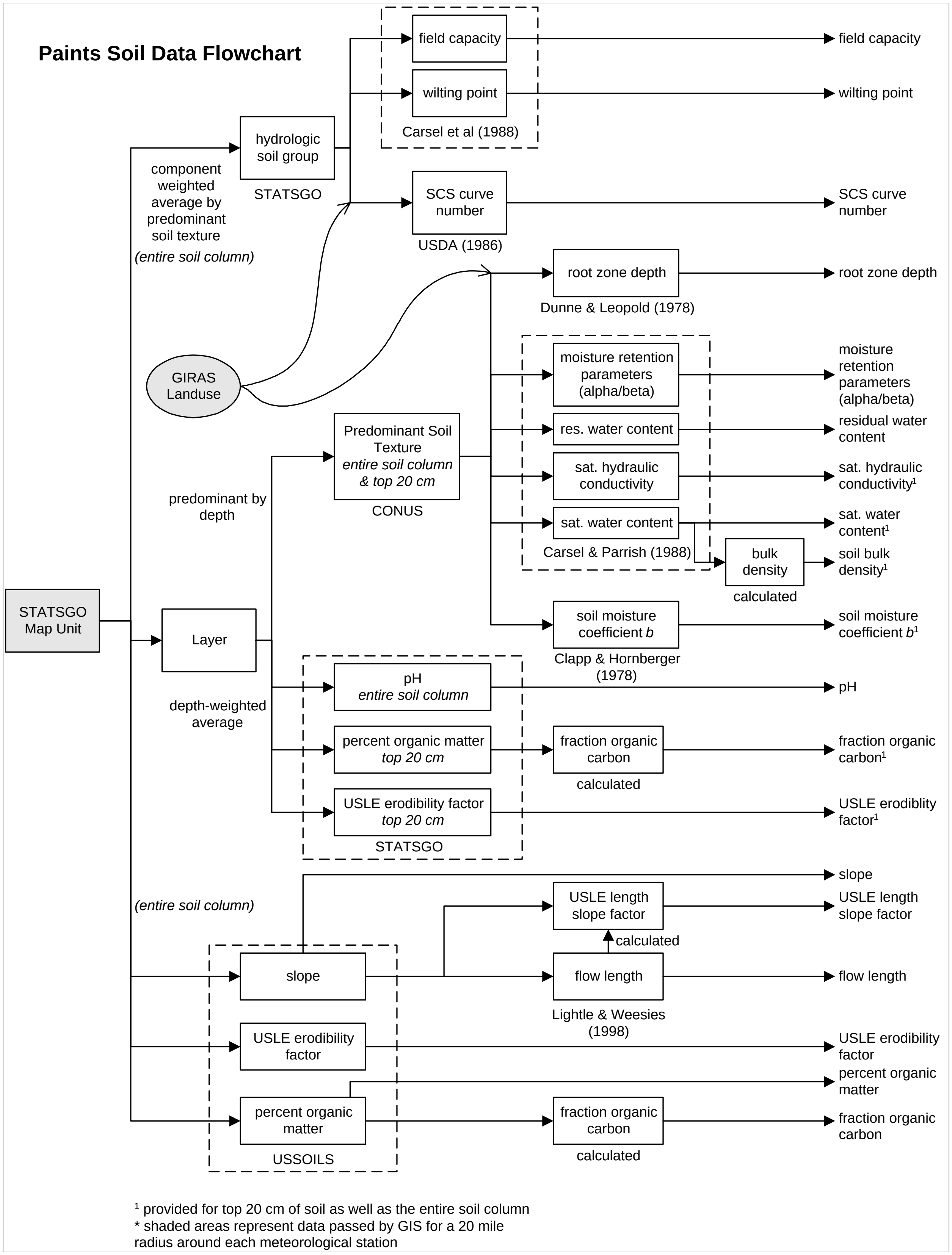

Figure 4-8. Soil data flowchart.

Methodology. The soil data collection methodology begins with GIS programs (in Arc Macro Language [AML]) that overlay a 20-mile radius around each meteorological location on the STATSGO map unit coverage to determine the STATSGO map units and their area within the radius. These data are then passed to data processing programs that derive predominant soil properties around each meteorological station, either through direct calculations or by applying established relationships in lookup tables. In deriving soil model inputs, the paints soil data processing effort bases all collected soil properties on the predominant soil type (texture and hydrologic group) for the STATSGO map units within a 20-mile radius of each meteorological station. Depending on modeling requirements, soil properties were derived for surface soils (top 20 cm), the entire soil column (to represent the vadose zone), or both, as shown in Figure 4-8. A detailed parameter-by-parameter description of how soil data were processed from the original data sources is provided in Appendix I.

4.3.2.4 <u>Hydrogeologic Environments and Aquifer Properties</u>. Locations evaluated in this risk assessment were established by the selection of meteorological stations (see Section 4.3.2.1) and define a regional framework for the collection of aquifer data. For aquifer properties (used by the source partition and groundwater models), it was necessary to designate hydrogeologic environments for each of the locations modeled so that correlated, national aquifer property data from the American Petroleum Institute (API) Hydrogeologic Database (HGDB; Newell et al., 1989; Newell et al., 1990) could be used in the analysis. The groundwater model, EPA's Composite Model with Transformation Product (EPACMTP) uses the HGDB data to specify probability distributions for each of four hydrogeologic parameters:

- Unsaturated zone thickness
- Aquifer thickness
- Hydraulic gradient
- Longitudinal hydraulic conductivity.

Average aquifer/vadose zone temperature was also required for the groundwater model. These were obtained from a map of groundwater temperatures for the continental United States in the *Water Encyclopedia* (van der Leeden et al., 1990). The remaining parameters were developed as described below.

The HGDB provides correlated data on these hydrogeologic parameters and an aquifer classification for approximately 400 hazardous waste sites nationwide, grouped according to 12 hydrogeologic environments described in Newell et al. (1990) and shown in Table 4-7. The empirical distributions of values for each of the four hydrogeologic parameters for each of the hydrogeologic environments are provided in *EPACMTP User's Guide* (U.S. EPA, 1997).[2]

[2] Note that EPACMTP also includes a 13th environment, with national average properties, for sites that cannot be easily classified into the 12 HGDB hydrogeologic environments. This general environment was not used in this paints analysis.

Table 4-7. Twelve Hydrogeologic Environments in EPACMTP

Code	Description
01	Metamorphic and igneous
02	Bedded sedimentary rock
03	Till over sedimentary rock
04	Sand and gravel
05	Alluvial basins valleys and fans
06	River valleys and flood plains with overbank deposits
07	River valleys and flood plains without overbank deposits
08	Outwash
09	Till and till over outwash
10	Unconsolidated and semiconsolidated shallow aquifers
11	Coastal beaches
12	Solution limestone

HGDB = Hydrogeologic Database.

Source: Newell et al. (1990).

To use the HGDB data, one or more of 12 HGDB hydrogeologic environments was assigned to each meteorological station location. In doing so, two concerns were important:

- Selecting HGDB hydrogeologic environments that were reasonably representative of the hydrogeologic region in which the meteorological station is located

- Including in the analysis as many of the 12 different HGDB hydrogeologic environments as necessary to ensure that the analysis reflected the variation of aquifer types for paint manufacturing facilities in the continental United States.

HGDB hydrogeologic environments were assigned to each of the 49 meteorological station locations using a U.S. Geological Survey (USGS) inventory of state groundwater resource maps (Heath, 1985) along with USGS GIS coverages of Heath hydrogeologic regions, productive aquifers, and surficial geology (Clawges and Price, 1999a-d). First, the GIS was used to overlay the 20-mile radius around each location on the Heath region coverage (Clawges and Price, 1999b) and assign a region(s) to each site. GIS coverages of productive aquifers (Clawges and Price, 1999c) and surficial geology (Clawges and Price, 1999d) were then used with state groundwater summary maps and descriptions (Heath, 1985) to determine the principal aquifer types present within the 20-mile radius. Hydrogeologic environments were then assigned by relating these aquifer types to the HGDB hydrogeologic environments using the crosswalk between Heath region, DRASTIC hydrogeologic setting, and HGDB environment provided in Appendix 1 of Newell et al. (1990) and DRASTIC setting descriptions from Aller et al. (1987).

HGDB hydrogeologic environment fractions (i.e., the portion of the region assigned to each of the 12 hydrogeological environments) were defined and used in the paints analysis as follows. If the 20-mile radius around a meteorological station contained only one HGDB environment, the fraction assigned was 1.0 and all groundwater model runs for this location were associated with that hydrological environment. If more than one HGDB environment was present, each environment was assigned an equal fraction based on the number of environments within the 20-mile radius.[3] These fractions were then used to generate the hydrogeologic environment for that location for each realization of the Monte Carlo groundwater modeling analysis. For example, if two hydrogeologic environments were present in the vicinity around one meteorological station, each would each be assigned a value of 0.5. When this site was chosen in the Monte Carlo analysis, half of the realizations were modeled with the first hydrogeologic environment and half were modeled with the second HGDB environment. Results of this process are presented in Appendix I.

The final step in the process was to construct a 10,000-record set of hydrogeologic environments and associated hydrogeologic parameters that match the meteorological station locations. Using the hydrogeologic environment fractions defined for each meteorological station location, summarized in Table 4-8, a hydrogeologic environment was assigned to each occurrence of that location in the 10,000-record location data set. For example, for the Des Moines, Iowa, meteorological station, there would be 182 occurrences in the location data set because there is only one meteorological station for Iowa and Iowa has 1.82 percent of the national production of paint waste. The fractions assigned to hydrogeologic environments for this location are 50, 25, and 25 percent for hydrogeologic environments 2, 6, and 7, respectively. Consequently, for this location, hydrogeologic environments 2, 6, and 7 would occur approximately 91, 45, and 45 times, respectively, depending on the random assignments made.

Once the hydrogeologic environments were assigned, a preprocessing run of EPACMTP was conducted to construct a set of randomly generated but correlated hydrogeologic parameter values for each occurrence of the hydrogeologic environments in the 10,000-record location data set. Missing values in the HGDB data set were filled using correlations, as described in U.S. EPA (1997). The unsaturated zone thickness generated from the preprocessing was also a parameter required in the surface impoundment source model. The output of the source modeling, along with the hydrogeologic parameter values used, was reported to an output file for the groundwater modeling analysis. Thus, the inputs to the surface impoundment and the groundwater model were synchronized to ensure consistency.

[3] The HGDB contains two environments for alluvial aquifers, with and without overbank deposits. Because the data sources used did not distinguish between these alluvial aquifer types, these were treated as a single alluvial environment when assigning weights (e.g., if alluvial was one of two environments at a site, a fraction of 0.25 would be assigned to both river valleys with overbank deposits and river valleys without overbank deposits).

Table 4-8. Summary of Aquifer Types

Location	Meteorological Station ID	HGDB Hydrological Environments										
		1	2	3	4	5	6	7	8	9	10	12
Albany, NY	14735			0.33			0.17	0.17	0.33			
Atlanta, GA	13874	1.00										
Baltimore, MD	93721	0.50									0.50	
Boston, MA	14739	0.50					0.25	0.25				
Boulder, CO	94018		1.00									
Chicago, IL	94846											1.00
Cleveland, OH	14820			1.00								
Columbia, SC	13883	0.50									0.50	
Columbus, OH	14821						0.17	0.17	0.33			0.33
Des Moines, IA	14933		0.50				0.25	0.25				
Detroit, MI	94847						0.17	0.17	0.33			0.33
Fort Worth, TX	03927		0.50				0.25	0.25				
Grand Rapids, MI	94860		0.33				0.17	0.17	0.33			
Green Bay, WI	14898		0.50	0.50								
Greensboro, NC	13723	1.00										
Greenville, SC	03870	1.00										
Hartford, CT	14740		0.50				0.25	0.25				
Houston, TX	12960										1.00	
Huntington, WV	03860		0.50				0.25	0.25				
Huntsville, AL	03856											1.00
Indianapolis, IN	93819			0.33			0.17	0.17		0.33		
Jackson, MS	03940						0.25	0.25			0.50	
Lexington, KY	93820											1.00
Little Rock, AR	13963						0.25	0.25			0.50	
Los Angeles, CA	23174					0.50	0.25	0.25				
Memphis, TN	13893						0.25	0.25			0.50	
Miami, FL	12839				0.50							0.50

(continued)

Table 4-8. (continued)

Location	Meteorological Station ID	HGDB Hydrological Environments										
		1	2	3	4	5	6	7	8	9	10	12
Minneapolis, MN	14922		0.50	0.50								
Nashville, TN	13897						0.25	0.25				0.50
New Orleans, LA	12916				0.50						0.50	
New York, NY	94728			0.50						0.50		
Newark, NJ	14734			0.50						0.50		
Norfolk, VA	13737										1.00	
Oklahoma City, OK	13967		0.50				0.25	0.25				
Philadelphia, PA	13739	0.50	0.50									
Phoenix, AZ	23183					1.00						
Pittsburgh, PA	94823		0.50				0.25	0.25				
Portland, OR	24229					0.50	0.25	0.25				
Providence, RI	14765								0.50	0.50		
Reno, NV	23185					0.50	0.25	0.25				
Richmond, VA	13740	0.50									0.50	
Roanoke, VA	13741	0.50										0.50
Rockford, IL	94822		0.33	0.33			0.17	0.17				
San Francisco, CA	23234					1.00						
Seattle, WA	24233					0.50			0.50			
South Bend, IN	14848						0.25	0.25	0.50			
St. Louis, MO	13994						0.25	0.25		0.50		
Tampa, FL	12842				0.50							0.50
Wichita, KS	03928					0.50	0.25	0.25				
Overall weight factor		0.12	0.13	0.08	0.03	0.09	0.11	0.11	0.06	0.05	0.11	0.12

4.3.2.5 <u>Estimation of Aquifer Recharge Rates</u>. A hydrology model with a daily time step was used to estimate annual average aquifer infiltration (recharge) rates. The hydrology model is an integral component of the HWIR99 3MRA land-based source modules and the watershed module as documented in *Modules for Nonwastewater Waste Management Units (Land Application Units, Wastepiles, and Landfills): Background and Implementation for the Multimedia, Multipathway, and Multireceptor Risk Assessment (3MRA) for HWIR99* (U.S. EPA, 1999). The hydrology model was applied in this risk assessment to estimate regional aquifer recharge rates by configuring the 3MRA land application unit module to represent a typical watershed at the site with hydrologic parameters (e.g., soil properties, average daily precipitation) provided for each location as discussed in previous sections.

The hydrology model can be thought of for purposes of this analysis as simply a unit soil column in the site's region that has soil properties typical of the region. The estimated infiltration through this unit soil column on average would be expected to be identical to infiltration through all watershed soils in the region of the site. An overview of the hydrology model is presented below.

The hydrology model is based on a daily soil moisture water balance performed for the root zone of the soil column. At the end of a given day, t, the soil moisture is updated as

$$SM_t = SM_{t-1} + P_t - RO_t - ET_t - IN_t \tag{4-1}$$

where

SM_t	=	soil moisture (cm) in root zone at end of day t
SM_{t-1}	=	soil moisture (cm) in root zone at end of previous day
P_t	=	total precipitation (cm) on day t
RO_t	=	storm runoff (cm) on day t
ET_t	=	evapotranspiration (cm) from root zone on day t
IN_t	=	infiltration (groundwater recharge) on day t (cm).

Precipitation is undifferentiated between rainfall and frozen precipitation; that is, frozen precipitation is treated as rainfall. Daily runoff is estimated by means of the widely used "curve number" method (USDA, 1986). Daily evapotranspiration (ET_t) is estimated as either the potential evapotranspiration (PET) if soil moisture is abundant or, if the PET exceeds the available soil moisture on that day, as a function of PET, the available soil moisture and the available soil moisture capacity. On wet days, runoff is estimated, evapotranspiration is then estimated, and, after these moisture removal mechanisms are accounted for, any soil moisture remaining above the soil's field capacity (amount that can be held by capillary action) is assumed to drain by gravity and become infiltration. If the daily infiltration thus estimated exceeds the soil's saturated hydraulic conductivity, a feedback loop exists to increase the runoff and/or evapotranspiration (if less than PET) until the infiltration does not exceed the saturated hydraulic conductivity. At the end of day t, the soil moisture (SM_t) is updated in accordance with Equation 4-1. On dry days, the same procedure is followed, except there is no runoff because there is no precipitation. At the end of the year, the infiltration is reported as an annual average of the daily values.

Groundwater recharge calculated as above was used in the fate and transport models for the air pathway to account for leaching losses from surficial soils. Groundwater recharge was also input to the groundwater model. For the groundwater model, however, a constraint was imposed that the groundwater recharge should not be less than the infiltration predicted by the source partition model. Therefore, groundwater recharge calculated as described above was used in the groundwater modeling unless source partition model infiltration rates were greater, in which case the groundwater recharge rate was set equal to the infiltration rate calculated by the source partition model.

4.4 Characterization of Waste Management Units

In this section, specific WMUs are characterized with respect to capacity and dimensions (e.g., area, depth). These dimensions and operating characteristics are important determinants of the modeled emission rates and dispersion factors used to estimate direct and indirect exposures. Source parameters that are specifically used to estimate emissions are discussed in Section 5.1, Source Partition Modeling of Constituent Releases.

4.4.1 Landfills

To model risks associated with disposal of paint wastes in landfills, detailed information about landfill characteristics was compiled. This section discusses the methods and data used to characterize landfills.

4.4.1.1 Selecting Representative Landfill Units. The primary source of data used to characterize landfills is the 1985 *Screening Survey of Industrial Subtitle D Establishments*, referred to as the Industrial D Screening Survey or Ind D (Schroeder et al., 1987). This survey was designed to collect information about nonhazardous (RCRA Subtitle D) waste management practices at industrial facilities across the United States. Data were gathered for the following land-based WMU types: landfills, wastepiles, land application units, and surface impoundments.

The Industrial D Screening Survey collected information on land-based Industrial D waste management operations for 17 industry groups[4] defined by EPA. Data from this survey have been used to represent Industrial D WMU characteristics in a variety of RCRA regulatory initiatives. Although the Industrial D data are more than 10 years old, they are the largest consistent set of data available on Industrial D WMU dimensions and characteristics. Information on the survey design, response rates, and overall data quality and completeness is provided in Schroeder et al. (1987), Clickner (1988), and Clickner and Craig (1988).

[4] Industry groups as follows: (1) organic chemicals; (2) primary iron and steel; (3) fertilizer and agricultural chemicals; (4) electric power generation; (5) plastic and resins; (6) inorganic chemicals; (7) stone, clay, glass, and concrete; (8) pulp and paper; (9) primary nonferrous metals; (10) food and kindred products; (11) water treatment; (12) petroleum refining; (13) rubber and miscellaneous products; (14) transportation equipment; (15) selected chemical and allied products; (16) textiles; and (17) leather and leather products.

There were 15,844 total sites in the Industrial D database. Of those, 2,850 reported that they managed waste in a landfill. Only 2,839 sites, however, reported surface area, which is a required parameter for the source and dispersion models. Another 96 sites lacked data because of confidential business information (CBI) claims, and 67 sites were outside of the contiguous United States (25 in Alaska and 37 in Hawaii). Sixty-eight landfills reported accepting wastes in all or part from off-site sources. These 68 landfills were selected for characterizing the landfills included in this assessment because the focus of this analysis was to model wastes that are predominantly managed in off-site disposal facilities.

Previous modeling efforts have uncovered issues associated with the internal consistency of the Industrial D data. For example, for certain facilities, the remaining capacity is greater than the total capacity of the unit. In other cases, depths calculated from site-specific data are unreasonably large or small. To address such problems, questionable data were culled and/or replaced using procedures developed for EPACMTP (described in U.S. EPA, 1997). These replacement values were generated using random realizations from the probability distribution of quantity and/or capacity conditioned on area. In addition, the existing Industrial D database contains some zero values for waste quantity and area that resulted from truncation of the third decimal place in the original database. When zero area or zero waste quantity was reported, a minimum bound of 0.005 acre (equal to 20.23 m^2) or 0.005 tonne (equal to 0.005 Mg) was used (U.S. EPA, 1997).

Appendix E presents raw data from the Industrial D Screening Survey (including replacement values) for the 68 Industrial D landfills addressed in this analysis. This information includes the types and numbers of WMUs at each site, the average area, the waste quantity, and the total capacity for each WMU.

4.4.1.2 <u>Determining Representative Landfill Surface Areas</u>. Conducting air dispersion modeling for each of the 68 landfills would have required an unacceptable amount of time. Therefore, the 68 landfills were assigned to 20 strata to characterize the range of sizes in the analysis. The median surface area for each stratum was used for air dispersion modeling. The Dalenius-Hodges procedure was used on the natural log of the area to assign the strata. Applying the natural log to the area values reduced the skewness of the distributions and allowed more strata to be assigned to the lower end of the distributions, where changes in surface area have the greatest effect on air modeling results. Thus, more data points are desired for the units with the smaller areas. A description of the Dalenius-Hodges procedure for determining strata is given in Appendix J.

Landfill units were modeled as ground-level area sources. Table 4-9 presents the source areas for each of the 20 strata that were used in the modeling analysis. These median surface areas were modeled as area sources for each of the 49 meteorological locations and the results used to represent all units selected from the strata. Dispersion modeling is discussed in detail in Section 5.2.

For the Monte Carlo analysis, a set of 10,000 records from the total of 68 landfills was needed. Each landfill was assigned an equal weight and replicated to produce the 10,000 records. Each record was identified by landfill identifier and bin number to which the particular landfill

Table 4-9. Median Surface Areas for Landfills Strata

Bin	Number of Landfills	Median Surface Area (m^2)
1	1	20.235
2	1	323.760
3	2	679.896
4	1	930.810
5	1	2023.500
6	3	5,058.750
7	5	8094.000
8	2	13,962.150
9	4	20,235.000
10	2	23,270.250
11	5	32,376.000
12	3	40,470.000
13	5	53,825.100
14	4	63,403.000
15	7	80,940.000
16	3	101,175.000
17	10	161,880.000
18	3	202,350.000
19	2	364,230.000
20	2	710,248.500
21	2	129,9087.000
Total	68	

was identified. The bin number was used to match the landfill to the dispersion modeling run for that bin's median surface area.

4.4.1.3 <u>Landfill Characteristics</u>. This section describes the approach used to develop characteristics for the landfill for use in source and air dispersion modeling. The data collected for landfill model inputs are extracted directly or calculated from Industrial D Screening Survey data (Schroeder et al., 1987). Other data used to supplement the Industrial D data are based on relationships taken from books, reports, and professional judgment.

Landfill Model Design. Landfill data collection assumed that only one type of landfill is used for disposal of waste (i.e., that there are no significant differences in the design of landfills depending on size or purpose). Other significant assumptions were that the landfill is excavated below ground surface, the unit receives waste for 30 years, the landfill is capped with soil cover to establish a vegetative cover after a cell is filled, and there is no liner. It was also assumed that there are controls in place to prevent overland transport of constituents to adjacent land areas by runoff or erosion.

Landfill Site-Specific Data. Unit-specific data for landfills were obtained from the Industrial D Screening Survey (Schroeder et al., 1987). These include total area, number of landfills at each site, total capacity, remaining capacity, and total 1985 annual waste quantity. When more than one landfill was present at a facility, average values were calculated by dividing the Industrial D data for each of the parameters by the number of landfills at each site. Appendix D shows raw data from the Industrial D Screening Survey for the 68 Industrial D landfills addressed in this analysis.

In accordance with previous EPA modeling efforts using the Industrial D Screening Survey, landfill capacities were screened from the Industrial D data when depth constraints were violated or capacity was missing. Depth was screened using the following procedures (U.S. EPA, 1997):

> The unit depth was calculated by dividing the unit capacity by the unit area and is constrained to be either greater than or equal to 2 feet, or less than or equal to 33 feet. The unit depth bounds were adopted from the previous toxicity characteristic rule effort.

Of the 824 landfills (reporting surface area) in the Industrial D Screening Survey, 232 had depth less than 2 ft or greater than 33 ft. In addition, 91 facilities were missing data on total capacity. Thus, landfill capacity was missing or screened for 323 landfills.

Landfill capacities to replace the 323 missing or removed values were estimated based on the correlation between surface area and capacity of the remaining landfills in the Industrial D data. The procedure used to replace values was similar to the following EPACMTP methodology (U.S. EPA, 1997):

> In cases where the unit depth or remaining capacity constraints were violated, the observed unit volume was replaced by generating a random realization from the volume probability distribution conditioned on area assuming that the unit area value was more likely to be correctly reported. The joint distribution was derived from the non-missing unit area/volume pairs that met the unit depth and remaining capacity constraints and was assumed to be lognormal. Missing values were generated from the joint area/volume probability if both the area and volume were missing, and from the corresponding conditional distribution if only one of the two values was missing. Final depth values were calculated by dividing the unit volume by the area.

One deviation in methodology was that EPACMTP uses remaining capacity as a screening tool for total capacity (remaining capacity must be less than total capacity). However, it was found that the remaining capacity was not a reliable data source to use for this purpose. In fact, for 30 facilities in the Industrial D database, it was impossible to calculate replacement values that would satisfy both the depth constraint and the remaining capacity constraint. Therefore, replacement values were calculated only in the cases where the depth constraint was violated or where a total capacity was not reported at all.

To calculate replacement capacity values, first a statistical regression of log (average total capacity) versus log (average surface area) was done on the facilities with known capacities. The regression yielded an equation for a best-fit line through the known values. This equation gave the capacity as a function of area, so the missing or screened capacities could be estimated based on the known areas. To provide a more probabilistic sampling of average capacities, and because the known capacities seemed to be in a limited range above and below the best-fit line, a positive or negative random number was generated within that range and added to the calculated log (average total capacity) to replace each missing capacity with a random value that was reasonable with respect to landfill area. This value was then used to calculate landfill depth as described above. Figure 4-9 shows the regression plots, including the replaced (random capacity) values, for landfills. The random numbers were generated to be between -0.6 and 0.6 based on the range of variability of the known capacity data (plotted on a log scale). The range was determined by plotting the values, drawing lines parallel to the regression best-fit line on either side, and then measuring the distance from the line to find a reasonable range. In addition, the depths calculated from the replaced capacities were also checked to make sure that they met the criterion of greater than 0.5 m and less than 10 m.

Cover Soil Properties. For purposes of this analysis, it was assumed that the soil used to cover the landfill was obtained from soil at or very near the facility and, in many cases, could be soil excavated to construct the landfill itself. Thus, soil properties for the vadose zone directly underlying the landfill were used for cover soil properties. The following cover soil parameters have been collected for use by the landfill model: fraction organic carbon, saturated hydraulic conductivity, saturated water content, and soil moisture coefficient b. Characterizing soils is discussed in Section 4.3.2.3.

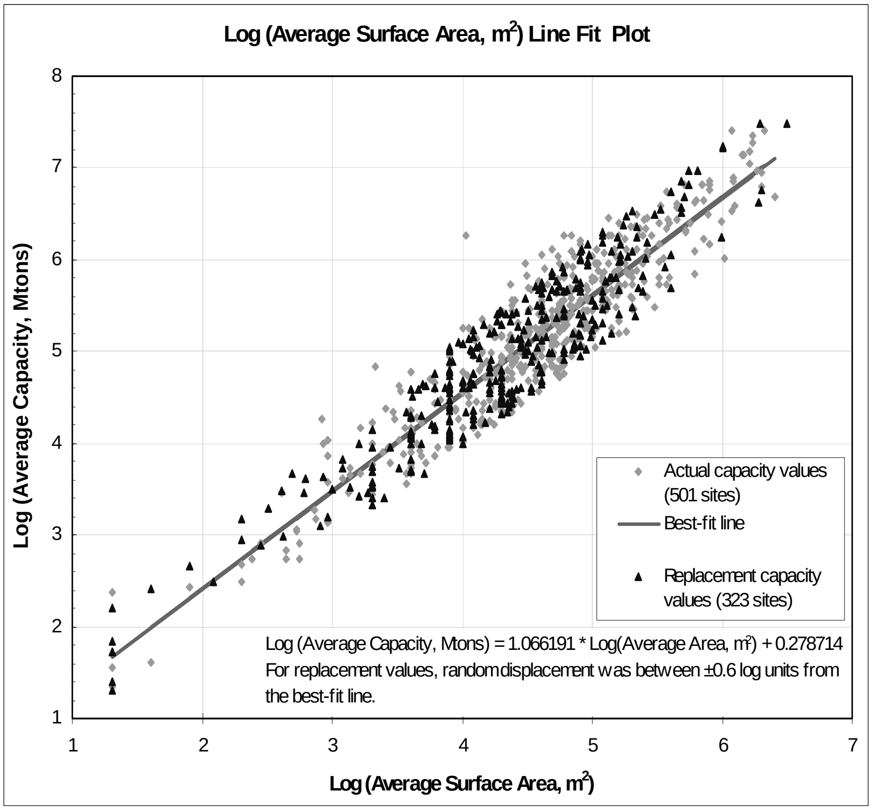

Figure 4-9. Correlation of total capacity to area for landfills.

4.4.2 Aerated and Nonaerated Treatment Tanks

To model risks associated with disposal of paint wastes in aerated and nonaerated treatment tanks, detailed information about tank characteristics was compiled. This section discusses the methods and data used to characterize aerated and nonaerated treatment tanks.

4.4.2.1 <u>Sampling Representative Treatment Tank Units</u>. The Industrial Subtitle D Survey (Schroeder et al., 1987) did not include tanks. Therefore, a tanks database was developed for this analysis that compiled flow rates and tank volumes. The primary source for these data was EPA's 1986 National Survey of Hazardous Waste Treatment, Storage, Disposal, and Recycling Facilities (TSDR) Database (U.S. EPA, 1987). This database is the result of a comprehensive survey of 2,626 hazardous waste TSDR facilities that requested information concerning 1986 waste management practices and quantities. Responses were received from 2,322 facilities. The TSDR survey included a specific questionnaire concerning tanks used at each facility. Responses to this questionnaire provided tank information for about 18,773 tanks at 1,700 facilities.

A subset of the TSDR survey responses was available for facilities that received any quantity of waste from off-site. This subset of data contained information on 8,510 tanks located at 710 facilities (approximately 45 percent of all of the tanks contained in the TSDR Survey).

This reduced data set was used to characterize tanks for this analysis. This data set matches the scope of the paints risk analysis well because management scenarios address only off-site disposal of waste (see Section 3.1). The subset data include a broad range of tank volumes ranging from less than 55 gallons to over 5 million gallons.

Table 4-10 shows the total number of tanks in the database, those culled from the database, and those remaining in the database that were used in this study. The totals add to somewhat less than 8,510 because the database included 1,270 units that, based on process codes, were not actually tanks. The remaining 7,240 were categorized as treatment or storage based on process codes, as described in Section 4.3.2.3; there were 472 tanks that specified both treatment and storage codes and that were therefore included in both treatment and storage datasets.

Several criteria were used in guiding the development of the tanks database. These criteria were applied to the TSDR survey data to determine which tanks should be excluded from the data set:

- **Flow rate.** Only those tanks reporting nonzero flow rates were included in the analysis.

- **Open versus covered tanks.** Only treatment tanks were considered in the analysis; closed or covered tanks were dropped because this study is concerned only with Industrial D scenarios and RCRA does not require covers for nonhazardous tanks.

- **Tank volume.** All tanks with a volume of 55 gallons or less were excluded from the analysis. These smaller-volume containers should be classified as drums and not tanks due to their size. Additionally, two very large tanks (approximately 30 million gallons), one aerated treatment and one nonaerated treatment, were reviewed because these tanks were many times larger than the next largest tanks and appeared to be nonrepresentative. The facility that owns both tanks was contacted and it was determined that both tanks have volumes of 3 million gallons, a value within the range represented by the other tanks in the database. Both values were corrected to 3 million gallons for this analysis.

- **Location**. Tanks located outside the continental United States were excluded from the database.

- **Storage Tanks.** All storage tanks were removed from the database.

4.4.2.2 <u>Determining Representative Tank Surface Areas and Heights</u>. To simplify the analysis, a sample of 200 tanks was drawn from the entire set of 893 tanks in the TSDR database. The sampling was conducted so as to preserve the range and distribution of tanks in the underlying database. The sample of 200 tanks was generated using the Dalenius-Hodges procedure (Cochran, 1963) based on the sample area of the units. The Dalenius-Hodges procedure is described in Appendix J.

Table 4-10. Summary of Tanks Removed from TSDR Survey Database

Description	Treatment Tanks
Number of sites reporting WMU type in TSDR database	2,346
Culled sites: zero size or flow	464
Culled sites: tank covered or cover not specified	979
Culled sites: size ≤ 55 gallons	6
Culled sites: outside contiguous United States	4
Total number of sites included in paints listing risk assessment data set	893

Because the sampling was conducted to preserve the range of tanks in the database, the 200 tanks selected were necessarily proportional to the number of tanks in a given size range in the total population of tanks. Therefore, a weighting factor was assigned to each tank in the sample of 200. The weighting factor was used in replicating the 200 tanks for the Monte Carlo analysis to produce 10,000 records. Thus, the 10,000 tanks selected for the Monte Carlo analysis mirrored the population of tanks in the TSDF database.

After the 200 tank units were selected, representative surface areas and heights needed to be defined for modeling. Because of resource constraints, modeling each of the 200 tanks was not reasonable. The 200-unit sample was divided into 31 area/height strata. The Dalenius-Hodges procedure was used on the natural log of the surface area to assign the strata. Applying the natural log to the surface area values reduced the skewness of the distributions and allowed more strata to be assigned to the lower end of the distributions, where changes in surface area have the greatest effect on air modeling results. This procedure produced 10 surface area bins. The tank units were also divided into four height categories to produce a total of 31 area-height bins that were used to perform the air dispersion modeling for tanks. Tables 4-11 and 4-12 present the median values for area and height that were used as input to the air dispersion modeling.

Table 4-11. Range of Heights in Tank Height Bins

Height Modeled for Bin (m)	Range of Heights (m)	
	Minimum	Maximum
0.5	0.035	1.42
2.0	1.500	2.43
3.5	2.500	4.49
5.5	4.500	6.27

Table 4-12. Median Surface Areas and Heights for Tanks Strata

Area Strata	Median Area (m^2)	Height for Bin (m)				Total in Area Strata
		0.5	2	3.5	5.5	
1	1.59	4	9			13
2	4.65	2	5	10		17
3	12.80	1		16		17
4	32.90	3		17		20
5	53.70	3	1	21		25
6	116.80	2	1	15	3	21
7	224.70	1	2	10	13	26
8	361.00	1		12	12	25
9	876.00	3	1	3	13	20
10	3,191.50	2	1	4	9	16
					Total	200

For the Monte Carlo analysis, a set of 10,000 records from the total of 200 tank units was produced as described above. Each record was identified by a tank identifier and bin number to which the particular tank was assigned. The bin number was used to match tank dispersion modeling results to the tanks' median areas and heights.

4.4.2.3 Aerated and Nonaerated Treatment Tank Characteristics.

Tank Classification. Industrial treatment tanks can be either quiescent or aerated/agitated. Examples of quiescent treatment tanks are clarifiers and filters (such as sand or mixed-media filters). In the absence of aeration, quiescent treatment tanks are still subject to small amounts of agitation during filling and emptying operations if they have above-surface intakes. Aeration or agitation in a wastewater treatment system transfers air to the liquid to improve mixing or increase biodegradation. The turbulence caused by aeration/agitation also enhances mass transfer to the air, thus increasing emissions. Therefore, for a given treatment volume, a facility with aerated tanks will have higher emissions than a facility with quiescent tanks.

To reflect emission characteristics associated with differences within the treatment tank category related to aeration intensity, three different tank categories were identified and modeled:

- High aerated treatment tanks

- Low aerated treatment tanks
- Nonaerated treatment tanks.

Sorting the tanks in the database into these three categories was done using the WMU code reported for each unit. Within the TSDR survey, the respondents were asked to provide a WMU code to describe the type of process for which each tank was used. The TSDR Survey used a broad range of WMU treatment codes (including codes for incinerators and belt filter presses). Classification of treatment tanks was based on those processes listed in Table 4-13.

The treatment tank WMU codes were evaluated further to determine the level of aeration used. High (HI) aeration was assigned to tanks reporting processes that actively mix the liquid surface for the purpose of aeration or that add diffused air. Low (LO) aeration was assigned to tanks reporting processes that are likely to require mixing devices due to the addition of chemicals or other purposes. No (NO) aeration was used for tanks that are purposefully operated to minimize mixing or agitation (e.g., a clarifier). The aeration level assignments for each WMU code are shown in Table 4-13. The high- versus low-aeration classification is based on the nature of the process description associated with the various process codes.

- Equalization, cyanide oxidation, general oxidation, chemical precipitation, and chromium reduction all involve adding and mixing a chemical into the wastewater followed by a quiescent period. Therefore, these tanks were classified as LO aeration because the chemical addition and mixing involve more agitation than a quiescent tank but involve no processes with intense agitation or forced air.

- Emulsion breaking included two different processes. Thermal heating simply involves heating and letting the wastewater stand, whereas chemical emulsion breaking involves chemical addition and mixing followed by a quiescent period. Therefore, thermal emulsion breaking was classified as NO and chemical emulsion breaking was classified as LO. The category “other emulsion breaking” was classified as LO because the other processes in the emulsion breaking category ranged from NO to LO, so this represented a conservative default classification in the absence of more specific process data.

- Filtration processes are quiet and generally covered; therefore, these were classified as NO aeration. Many of these, in fact, were eliminated from the database because covered tanks, as a class, were removed.

- Air flotation processes all involve high-energy forced air operations and are therefore, all classified as HI aeration.

- Oil skimming involves liquid phase separation, which requires quiescent conditions; therefore, NO aeration was assumed for these processes. Similarly, liquid phase separation processes were classified as NO aeration.

Table 4-13. TSDR Survey Wastewater Treatment Codes Used in Identifying Treatment Tanks

Process Code/Process		Aeration Level	Process Code/Process		Aeration Level
Equalization			**Filtration**		
1WT	Equalization	LO	34WT	Diatomaceous earth	NO
Cyanide oxidation			35WT	Sand	NO
2WT	Alkaline chlorination	LO	36WT	Multimedia	NO
3WT	Ozone	LO	37WT	Other filtration	NO
4WT	Electrochemical	LO	**Sludge dewatering**		
5WT	Other cyanide oxidation	LO	38WT	Gravity thickening	NO
General oxidation (including disinfection)			**Air flotation**		
6WT	Chlorination	LO	43WT	Dissolved air flotation	HI
7WT	Ozonation	LO	44WT	Partial aeration	HI
8WT	UV radiation	LO	45WT	Air dispersion	HI
9WT	Other general oxidation	LO	46WT	Other air flotation	HI
Chemical precipitation			**Oil skimming**		
10WT	Lime	LO	47WT	Gravity separation	NO
11WT	Sodium hydroxide	LO	48WT	Coalescing plate separation	NO
12WT	Soda ash	LO	49WT	Other oil skimming	NO
13WT	Sulfide	LO	**Other liquid phase separation**		
14WT	Other chemical precipitation	LO	50WT	Decanting	NO
Chromium reduction			51WT	Other liquid phase separation	NO
15WT	Sodium bisulfite	LO	**Biological treatment**		
16WT	Sulfur dioxide	LO	52WT	Activated sludge	HI
17WT	Ferrous sulfate	LO	54WT	Fixed film--rotating contactor	LO
18WT	Other chromium reduction	LO	57WT	Anaerobic	NO
19WT	Complexed metals treatment	LO	58WT	Other biological treatment	HI
Emulsion breaking			**Other wastewater treatment**		
20WT	Thermal	NO	60WT	Neutralization	LO
21WT	Chemical	LO	61WT	Nitrification	LO
22WT	Other emulsion breaking	LO	62WT	Denitrification	LO
Evaporation			63WT	Flocculation and/or coagulation	NO
31WT	Solar	NO	64WT	Settling (clarification)	NO
Fuel blending			66WT	Other wastewater treatment	LO
1FB	Fuel blending	LO	**Other processes**		
			1TR	Other treatment	LO

- Biological treatment processes are quite diverse and include HI aeration activated sludge processes and LO aeration film processes. The “other biological treatment” processes were classified as HI because the other processes in the biological treatment category ranged from NO to HI and HI represents a conservative default classification in the absence of more specific process data.

- Finally, the “other wastewater treatment” process in the “other wastewater treatment” category and the "other processes" category were classified as LO aeration as a default since no process information can be inferred from the description.

The numbers of tanks included in each classification are summarized in Table 4-14.

The tank database does appear to underrepresent highly aerated tanks. This may be due to the age of the survey data, reflecting that highly aerated biological processes were in less widespread use at that time than now. This underrepresentation introduces some uncertainty into the analysis, the result of which is that risks from aerated tanks may be underestimated.

Additional Tank Data Used for Imputation. In previous studies, additional data sources were identified to address tank-specific data gaps in the tanks database. These data included information collected in 1985 and 1986 during EPA site visits to aerated treatment systems in support of the development of RCRA air emission standards. In these studies, information on wastewater treatment systems at 54 facilities was collected based on site visits to these facilities conducted in 1985 and 1986. Data on the individual tanks (both aerated and nonaerated) were provided by the facilities during the site visits, including data on tank dimensions. The data on aerated tanks are summarized in RTI (1988) and Eichinger (1985). The data on nonaerated tanks were collected at the same site visits and are unpublished. Added to these data were five tanks from the TSDF background information document (U.S. EPA, 1991). This resulted in a supplemental database of 49 tanks (13 with high aeration, 9 with low aeration, and 27 with no aeration), presented in Table 4-15.

Table 4-14. Numbers of Tanks by Classification

Tank Classification	Number
Aerated treatment tanks	620
High aeration	29
Low aeration	591
Nonaerated treatment tanks	273
Total	893

Table 4-15. Summary of Tank Size Information Collected in EPA Site Visits for RCRA Air Emission Standards

Type of Unit	Aeration	Type of Aerator	Volume (m^3)	Area (m^2)	Depth (m)
Aerated trtmnt tank	HI	Mechanical	108	27	4.0
Aeration tank	HI	Mechanical	112	34	3.4
Bubbling pit	HI	Diffused	453	74	6.1
Aerated trtmnt tank	HI	Mechanical	1,600	430	3.7
Aeration tank	HI	Diffused	1,666	159	10.5
Aeration tank	HI	Mechanical	3,367	910	3.7
Aeration tank	HI	Diffused	3,785	730	5.2
Aeration tank	HI	Diffused	4,542	618	7.4
Aeration tank	HI	Mechanical	5,678	931	6.1
Aeration tank	HI	Diffused	5,764	1,051	5.5
Aux. Aer. tank	HI	Mechanical	21,804	4,459	4.9
Aeration tank	HI	Mechanical	26,546	5,806	4.6
Aeration tank	HI	Mechanical	41,261	11,241	3.7
Treatment tank	LO		30	13	2.4
Mixing tank	LO	Mechanical	68	9.3	7.3
Treatment tank	LO		76	26	2.7
Mixing tank	LO	Mechanical	112	34	3.4
No eq. basin	LO	Mechanical	191	84	2.3
So eq. basin	LO	Mechanical	240	109	2.2
Eq. basin	LO	Mechanical	681	200	3.4
Treatment tank	LO		800	65	12.0
Eq. basin	LO	Mechanical	41,261	11,241	3.7
Gravitator	NO		5.6	1.8	3.1
Prefilter	NO		132	39	3.4
Final filter	NO		154	42	3.7
Backwash clarifier	NO		207	71	2.9
Clarifier	NO		283	46	6.1
Biosludge holding tanks	NO		300	66	4.6
Biosludge holding tanks	NO		300	66	4.6
Primary clarifier	NO		641	263	2.4
Digestor	NO		819	117	7.0
Api separator	NO		836	457	1.8
Primary clarifier	NO		1,803	591	3.1
Biosludge thickener	NO		1,803	591	3.1
Clarifier/thickener	NO		2,504	410	6.1
Final clarifier	NO		2,513	687	3.7
Final clarifier	NO		2,513	687	3.7
Final clarifier	NO		2,513	687	3.7
Clarifier	NO		2,670	730	3.7
Clarifier	NO		2,670	730	3.7
Clarifier	NO		2,670	730	3.7
Clarifier	NO		3,065	804	3.8
Clarifier	NO		3,065	804	3.8
Clarifier	NO		3,065	804	3.8
Clarifier	NO		3,065	804	3.8
Ship’s ballast water	NO		3,394	1,271	2.7
Final clarifier	NO		3,918	1,430	2.7
Ship’s ballast water	NO		10,244	1,051	9.8
Solid waste disposal basin	NO		386,464	60,385	6.4

In addition to these data, several tank vendors were contacted to establish a reasonable high end for tank capacity and depth based on design principles. As a result, a reasonable maximum capacity for an open, partially, or completely aboveground tank was defined to be approximately 3 million gallons, and the depth of such a tank would not be expected to exceed 10 m (about 32 ft) (Kendall Smith, personal communication, AO Smith Industrial, March 16, 1999). These site visit tanks and hypothetical tanks were used only as a basis for imputing values and were not modeled in the analysis to maintain the integrity of the source database.

Estimation of Missing Data. The TSDR survey provided flow rate and tank volume data for use in characterizing tanks for this analysis. However, other key parameters, including depth, surface area, and height, also needed to be defined. In the absence of reported TSDR survey data, these parameters were calculated as described below. Other operating parameters (aeration parameters), which impact emission estimates but not dispersion, are discussed in Section 5.1.

The depth of the waste was imputed from the reported tank volume (or capacity). This was accomplished by developing a regression of log (depth) versus log (capacity) using data in the supplemental data set discussed below (49 tanks from the site visit/TSDF background information document (BID) database plus TSDF BID tanks). Because the site visit data did not include any very small tanks or many very large tanks, a cube-shaped 55-gallon tank and a 3-million-gallon/32-ft-deep tank (based on the vendor information) was included in the regression derivation. Regression lines were derived for aerated tanks (Equation 4-2) and nonaerated tanks (Equation 4-3), on the assumption that these might have different volume-to-depth relationships since aerated tanks may be shallower to facilitate aeration.

$$D = 10^{[0.1358 \times \log(V) + 0.2236]} \tag{4-2}$$

$$D = 10^{[0.1334 \times \log(V) + 0.1657]} \tag{4-3}$$

where

D = depth (m)
V = volume (m^3).

However, as can be seen in Figure 4-10, the two regressions were nearly identical. Therefore, a single regression was developed using all 49 tanks from the site visit/TSDF BID database plus the hypothetical tanks, as follows:

$$D = 10^{[0.1057 \times \log(V) + 0.2804]} \tag{4-4}$$

Comparisons of this regression with regressions done without one or both hypothetical tanks indicate that the hypothetical tanks do not unduly dominate the regression.

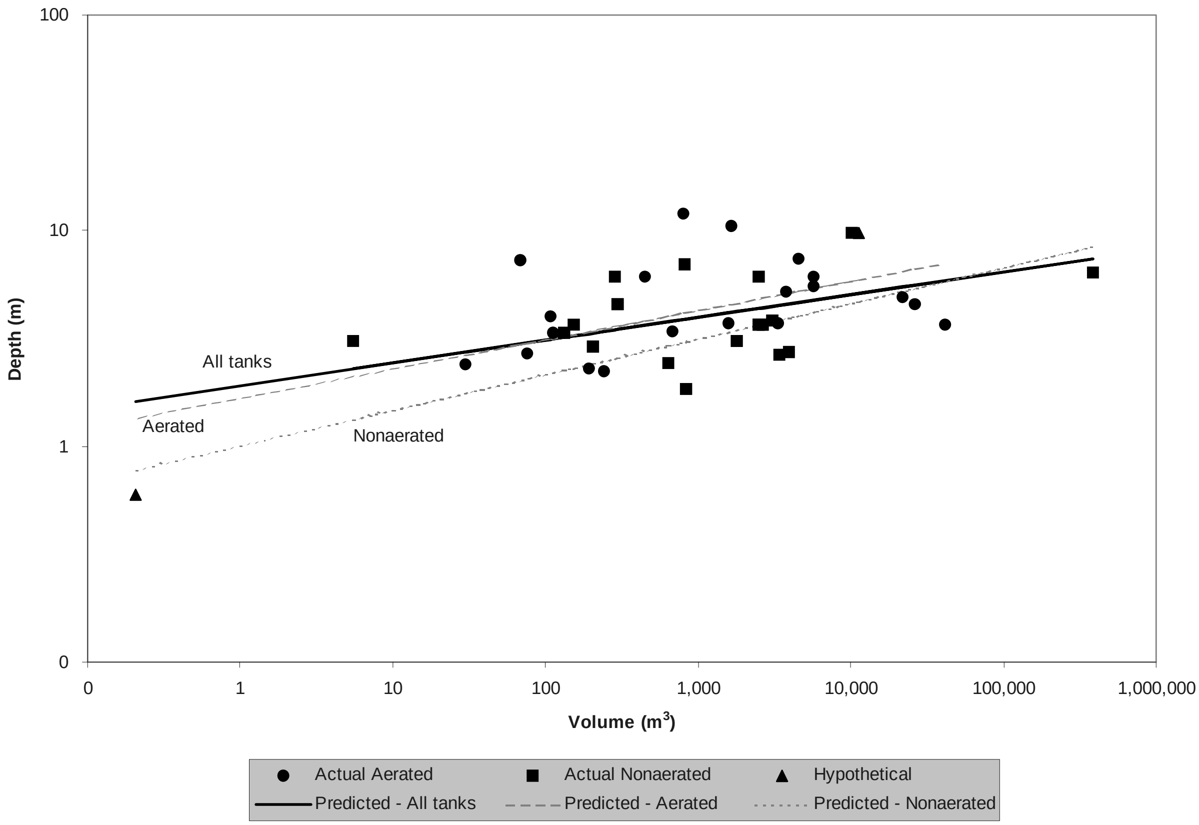

Figure 4-10. Comparison of tank depth regression lines.

This equation was then examined for the reasonableness of the depths predicted. Using Equation 4-4, 60-gallon tanks (the smallest tanks in the database) are approximately 1.9 m (6.2 ft) deep and about 39 cm (15.4 inches) in diameter. This seemed unrealistically tall and narrow; consequently, for very small tanks, a second equation was derived from the assumption of a cube-shaped tank:

$$D = V^{0.333} \tag{4-5}$$

For tanks of approximately 10 m^3, Equation 4-4 predicts approximately cube-shaped tanks; therefore, Equation 4-5 is used for tanks smaller than 10 m^3.

The largest tank in the TSDR database in the NO and LO aeration categories is 25,000 m^3, and the projected depth for this tank using Equation 4-6 is 5.6 m (18 ft), which is acceptable for mixing tanks. The largest tank in the TSDR database in the HI aeration category is 23,000 m^3, and the projected depth for this tank is 5.5 m (18 ft). In evaluating the predicted depth of HI aeration, the eight mechanically aerated tanks from the site visits were considered. The maximum depth from these data was 6.1 m (20 ft) and even this appeared to be an outlier compared to the other HI aeration, mechanically aerated tanks. Data for the other seven site visit tanks all have depths ranging from 3.35 m (11 ft) to 4.88 m (16 ft). The mid-range of the latter depths is approximately equivalent to a 1,000-m^3 tank as evaluated using Equation 4-4. Therefore, for HI aeration tanks greater than 1,000 m^3, a random depth was assigned using a uniform distribution with endpoints of 3.5 m and 4.8 m.

Table 4-16 summarizes the methods used to make an initial estimate of tank depth for each type of tank. While these methods were intended to represent the actual relationship between volume and depth as closely as possible, they imply a certain precision that is unrealistic. In fact, there will be variation in the dimensions of tanks of the same volume. To address that variation, a random variation was applied to these initial estimates using a normal distribution with a mean of 1 and 90 percent of the values between 0.8 and 1.2. The initial depth estimate was multiplied by this random factor to obtain a final depth estimate used in this analysis.

Surface area data were not provided in the TSDR Survey. In the absence of these data, surface area for each of the TSDR tanks was calculated by dividing tank volume by depth.

The height of the top of the tank above the ground is needed for dispersion modeling. Height is related to depth, but not necessarily equal to depth, as tanks may be partially in the ground. In the absence of height data being reported in the TSDR survey, height was imputed from depth using a two-step process:

1. A number was selected at random from 0 to 20 (uniform distribution).
2. If this number was less than the depth in meters + 0.5 meters, it was used as the height. If it was greater than the depth in meters + 0.5 meters, set height = depth + 0.5 m.

Table 4-16. Summary of Depth Imputation Techniques

Tank Type	Volume Range (m^3)	Imputation Technique
HI aeration	<10	Equation 4-5
	10-1,000	Equation 4-4
	>1,000	Uniform distribution from 3.5 to 4.8 m
LO aeration	<10	Equation 4-5
	≥10	Equation 4-4
NO aeration	<10	Equation 4-5
	≥10	Equation 4-4

None of the tank depths imputed were greater than 9.5 m; therefore, none of the heights above ground were more than 10 m (9.5 +0.5) using this method; 10 m above the ground is the realistic maximum height from a structural point of view, according to tank vendor contacts.

This approach establishes percentages of tanks of certain depths that will be all above-ground vs. partially or completely in ground:

- For 10-m tanks, about half are partly or all in ground (when the random selection is between 0 and 10; as the random number increases from 0 to 10, more and more of the tank depth is aboveground, until, at 10, all of it is), and about half are all aboveground (when the random pick is between 10 and 20).

- For 1-m tanks, about 5 percent are partly or all in ground (random numbers from 0 to 1) and 95 percent all aboveground (random numbers from 1 to 20).

- For 5-m tanks, 25 percent are partly or all in ground (random numbers from 0 to 5) and 75 percent all aboveground (random numbers from 5 to 20).

4.4.3 Surface Impoundments

A surface impoundment is an excavation or diked area typically used for the treatment, storage, or disposal of liquids or sludges containing free liquids. Liquids and solids typically separate in a surface impoundment by gravity settling. Liquids from surface impoundments are removed by draining, evaporation, or flow through an outlet structure. Accumulated solids are removed by dredging during impoundment operation or at the time of closure.

There are more than 180,000 surface impoundments in the United States (Hartley, 1992). Nearly 30,000 are used by industry, including chemical manufacturers, food processors, oil refineries, primary and fabricated metals, paper plants, and other commercial facilities. Based on their purpose, the three generic impoundment types are storage, disposal, and treatment.

To model risks associated with disposal of paint wastes in surface impoundments, detailed information about surface impoundment characteristics was compiled. This section discusses the methods and data used to sample and characterize surface impoundments.

4.4.3.1 <u>Sampling Representative Surface Impoundments Units</u>. The primary source of data used to characterize surface impoundments is the 1985 *Screening Survey of Industrial Subtitle D Establishments*, referred to as the Industrial D Screening Survey or Ind D (Schroeder et al., 1987). This survey was designed to collect information about nonhazardous (RCRA Subtitle D) waste management practices at industrial facilities across the United States for land-based WMU units: landfills, wastepiles, land application units, and surface impoundments.

Of the 15,844 total sites in the Industrial D database, 1,930 reported that they managed waste in a surface impoundment. Only 1,926 facilities, however, reported surface area, which is a required parameter for the source and dispersion models. Unlike landfills, survey data did not indicate whether or not waste was managed in off-site surface impoundments. However, respondents did indicate if the surface impoundments were used on a temporary basis (e.g., used for backup during rain events). Twenty-three facilities reported using surface impoundments only as backup storage units. These 23 facilities were excluded from the analysis. Thus, the total number of facilities considered was 1,903.

The data set was divided into six strata using the Dalenius-Hodges procedure (Cochran, 1963) on the surface areas of the units. The Dalenius-Hodges procedure sorts the data sets by area and uses the cumulative distribution of areas to determine the cutoffs for each stratum. Sorting the data set by the area before performing the systematic sample selection procedure preserved the range and distribution of the areas that appeared in the sample. The optimal selection of sample members when using the Dalenius-Hodges definition of strata is to select equal sizes from each stratum. Due to the skewed nature of the distribution of the surface areas in the data set, however, a very small percentage of the units were placed in the three strata containing the largest units. Therefore, all units in the these three strata were selected to be in the sample. A systematic sample selection procedure was used to select the remaining units needed to obtain a sample of 200 from each of the three strata with the smaller areas.

4.4.3.2 Determining Representative Surface Areas. To simplify the analysis, a sample of 200 surface impoundments was drawn from the entire set of 1,903 surface impoundments in the Industrial D database. The sampling was conducted so as to preserve the range and distribution of tanks in the underlying database. The sample of 200 surface impoundments was generated using the Dalenius-Hodges procedure (Cochran, 1963) based on the sample area of the units. A description of the Dalenius-Hodges procedure is given in Appendix J.

Because the sampling was conducted to preserve the range of surface impoundments in the database, the 200 surface impoundments selected were necessarily proportional to the number of surface impoundments in a given size range in the total population of surface impoundments. Therefore, a weighting factor was assigned to each surface impoundment in the sample of 200. The weighting factor was used in replicating the 200 surface impoundments for the Monte Carlo analysis to produce 10,000 records. Thus, the 10,000 surface impoundments selected for the Monte Carlo analysis mirrored the population of surface impoundments in the Industrial D database.

After the 200 surface impoundment units were selected, representative surface areas needed to be defined. The 200-unit sample was divided into 20 surface area strata, also called bins. The Dalenius-Hodges procedure was used on the natural log of the surface area to assign the strata. Applying the natural log to the surface area values reduced the skewness of the distributions and allowed more strata to be assigned to the lower end of the distributions, where changes in surface area have the greatest effect on air modeling results. Table 4-17 presents the median values for surface area that were used as input to the air dispersion modeling. A description of the Dalenius-Hodges procedure for determining strata is given in Appendix J.

For the Monte Carlo analysis, a set of 10,000 records from the total of 200 surface impoundments was produced as described above. Each record was identified by a surface impoundment identifier and bin number to which the particular surface impoundment was assigned. The bin number was used to match surface impoundment dispersion modeling results to the surface impoundment median surface areas.

Table 4-17. Median Surface Areas for Surface Impoundment Strata

Strata	Number of Surface Impoundments	Median Surface Area (m^2)
1	2	10.12
2	4	20.24
3	10	80.94
4	5	242.82
5	9	1,349.00
6	7	2,293.30
7	12	4,643.26
8	7	13,490.00
9	8	29,013.62
10	24	40,470.00
11	9	60,705.00
12	11	86,336.00
13	11	169,974.00
14	12	214,491.00
15	25	280,052.40
16	13	404,700.00
17	10	643,122.26
18	9	983,421.00
19	8	1,665,509.13
20	4	5,332,597.00
Total	200	

4.4.3.3 <u>Surface Impoundment Characteristics</u>. Unit-specific data for surface impoundments were obtained from the Industrial D Screening Survey (Schroeder et al., 1987). The data include total area, number of surface impoundments at each site, total capacity, and total 1985 annual waste quantity. Average values were calculated for use in the source model by dividing each of the parameters by the number of units at each site. Appendix D shows raw data from the Industrial D Screening Survey (including replacement values) for the 200 Industrial D units addressed in this analysis.

In accordance with previous EPA modeling efforts using the Industrial D Screening Survey, surface impoundment capacities were screened from the Industrial D data when either the capacity was missing or the depth constraint was violated. As with landfills, the unit depth was calculated by dividing the unit capacity by the unit area. The depth constraint was described by EPACMTP documentation as follows (U.S. EPA, 1997):

> The surface impoundment volume data were screened by constraining the calculated unit depth to be between 1 and 150 feet in order to eliminate unrealistic values.

Of the 1,926 Industrial D surface impoundment facilities that reported surface area, missing waste quantity values were replaced for 57 of these facilities. Replacement capacity values were calculated for 262 of the facilities with either missing or screened capacities. The procedures to replace waste quantity and capacity are conditioned on area, as described for landfills, and are consistent with EPACMTP methodology (U.S. EPA, 1997).

To calculate replacement values for the screened and missing annual waste quantities, first a statistical regression of log (average annual waste quantity) versus log (average surface area) was performed on the facilities with known quantities. The regression yielded an equation for a best fit line through the known values. This equation gave the waste quantity as a function of area, so the missing or screened waste quantities could be estimated based on the known areas. To provide a more probabilistic sampling of average waste quantities, and because the known quantities seemed to be in a limited range above and below the best-fit line (with some outliers), a positive or negative random number was generated within that range. This random number was then added to the calculated log (average waste quantity) to replace each missing waste quantity with a random value that was reasonable with respect to the surface impoundment area. Figure 4-11 shows the regression plot, including the replaced (random waste quantity) values, for surface impoundments.

To calculate replacement values for capacity, first a statistical regression of log (average total capacity) versus log (average surface area) was performed on the facilities with known capacities. The regression yielded an equation for a best-fit line through the known values. This equation gave the capacity as a function of area, so the missing or screened capacities could be estimated based on the known areas. To provide a more probabilistic sampling of average capacities, and because the known capacities seemed to be in a limited range above and below the best-fit line (with some outliers), a positive or negative random number was generated within that range and added to the calculated log (average total capacity) to replace each missing capacity with a random value that was reasonable with respect to the surface impoundment area.

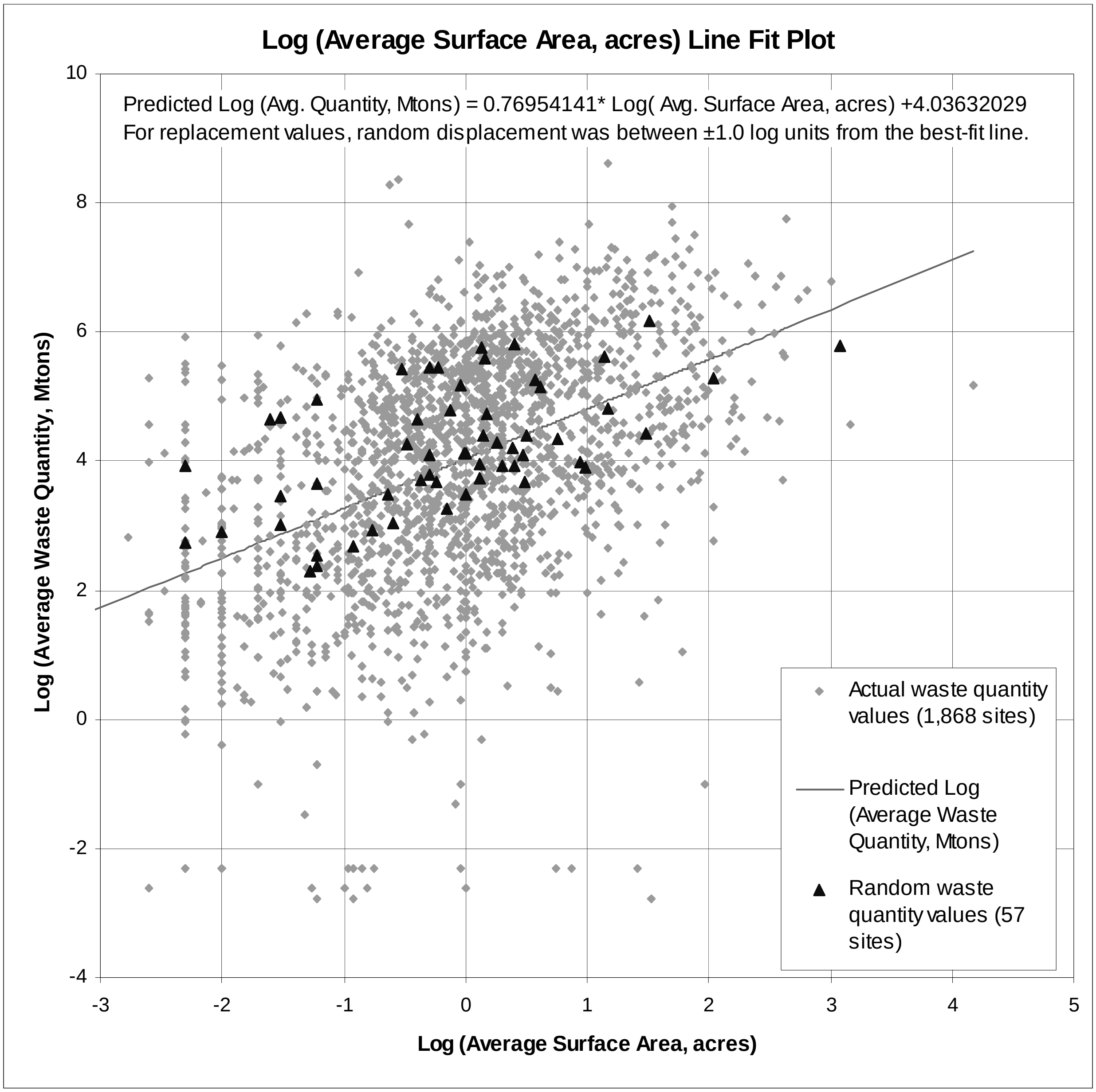

Figure 4-11. Correlation of waste quantity to area for surface impoundments.

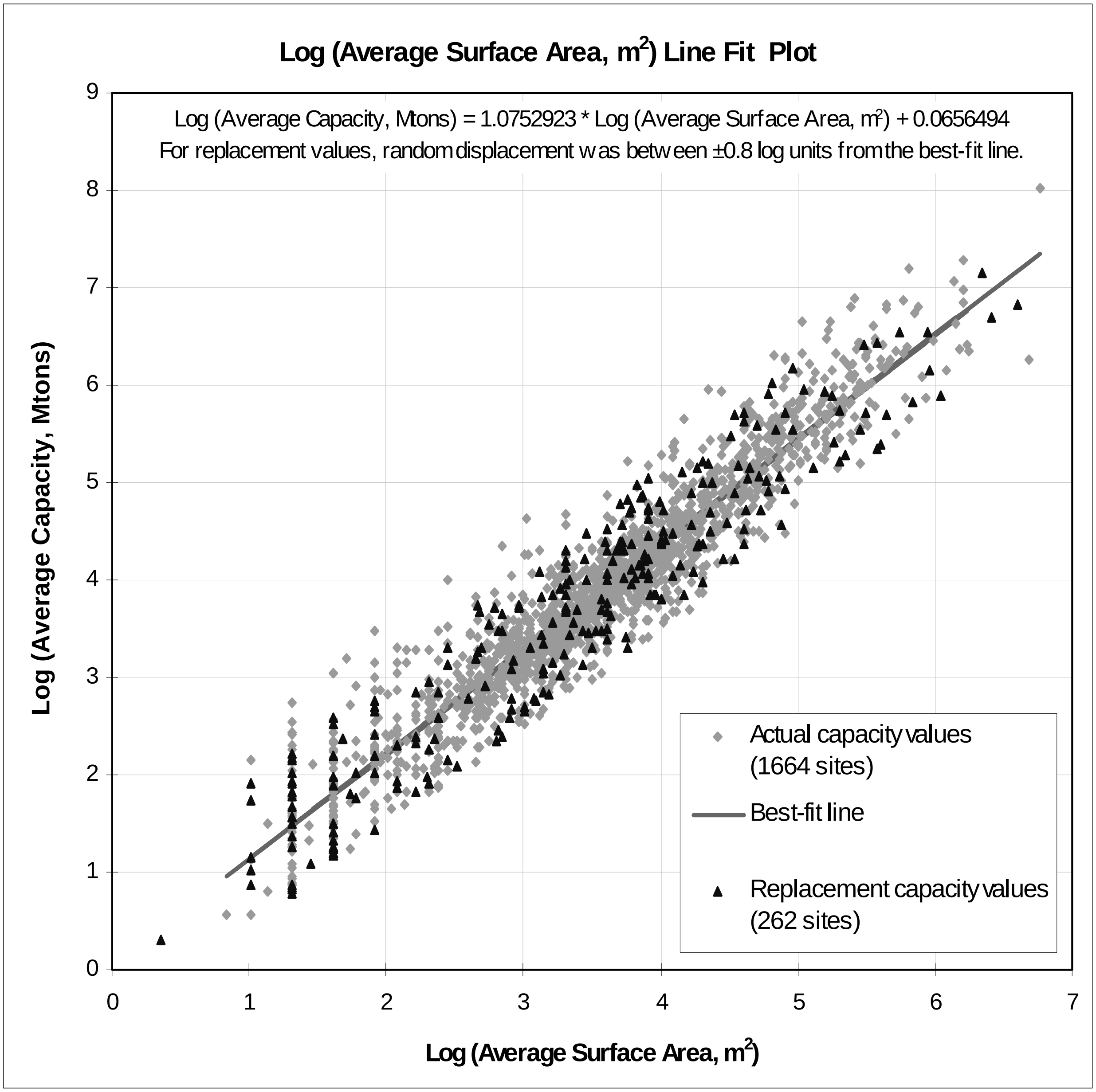

Figure 4-12. Correlation of total capacity to area for surface impoundments.

This value was then used to calculate depth as described above. Figure 4-12 shows the regression plots, including the replaced (random capacity) values, for surface impoundments.

Similar to tanks, surface impoundments will be operated with varying degree of aeration. Aeration characteristics were not a parameter reported in the Industrial D survey. In the absence of data, the distribution of aeration characteristics in tanks (i.e., HI, LO, or NO) was randomly applied to surface impoundments. Thus, each of the 200 surface impoundments was randomly assigned as having HI, LO, or NO aeration. These assignments were made so that the 10,000 iterations in the Monte Carlo analysis had approximately the same distributions of aeration characteristics for both surface impoundments and tanks. These aeration characteristics were used to determine various parameters required by the source model (i.e., number of impellers). The data used to define source model parameters are provided in Appendix E.

4.5 References

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