August 28, 2000

Risk Assessment for the Listing Determinations for Inorganic Chemical Manufacturing Wastes: Background Document

Prepared for

The Office of Solid Waste U.S. Environmental Protection Agency
401 M Street, SW (5307W)
Washington, DC 20460

EPA Contract Number 68-W-98-085

RTI Project Number 92U-7780.001.022

EPA Contract Number
68-W-98-085

RTI Project Number
92U-7780.001.022

Risk Assessment for the Listing Determinations for Inorganic Chemical Manufacturing Wastes: Background Document

August 28, 2000

Prepared for

The Office of Solid Waste U.S. Environmental Protection Agency
401 M Street, SW (5307W)
Washington, DC 20460

Prepared by

Center for Environmental Analysis
Research Triangle Institute
Research Triangle Park, NC 27709

Table of Contents

Section **Page**

Table of Contents (continued)

Section **Page**

Table of Contents (continued)

Section **Page**

Appendixes

Table of Contents (continued)

Figures

Tables

Number **Page**

Tables (continued)

Number **Page**

Tables (continued)

Number **Page**

Tables (continued)

Number **Page**

Tables (continued)

Number **Page**

Section **Page**

Errata

The errata Tables 3-9, 4-5, 4-8, and 5-10 correct typographical errors in the document. Errata Tables 6-9, 6-13, 6-22, and 6-30 correct the values for the hazard quotients calculated for manganese. The revised HQs reflect the inclusion of the modifying factor of 3 applied to the RfD for manganese when used to assess risk from the ingestion of manganese in drinking water. This modification was included in the risk analysis but the tables in the final document were not appropriately revised to reflect this change.

Table 3-9. Total Waste Stream Concentration and Leachate Concentration Data Used in Groundwater Modeling

Table 4-5. Parameters Used in Surface Water Screening for Onsite Landfills

Table 4-8. Results of Surface Water Screening Analysis for Surface Impoundments

Table 5-10. Percentiles of Generalized Adult ADD

Table 6-9. Risk Results Sludge Residues— Sodium Chlorate Sector Managed in Municipal Landfills

Table 6-13. Risk Results for Sulfate Process Secondary Gypsum—Titanium Dioxide Sector Managed in Onsite Industrial Landfill

Table 6-22. Comparison of Risk Results for Manganese in Ilmenite Process Wastewater Treatment Sludge for 100 Percent and 10 Percent Waste Quantities

Table 6-30. Comparison of Risk Results for Manganese in Ilmenite Wastewaters Managed in an Onsite Surface Impoundment for Z-Well Constrained with Z-Well Unconstrained

Table 3-9. Total Waste Stream Concentration and Leachate Concentration Data Used in Groundwater Modeling

Sector	Waste Stream	Constituent	TCLP (mg/L)	SPLP (mg/L)	SPLP filtrate (mg/L)	Total solid waste (mg/kg)	Total wastewater (mg/L)
HCN	Combined wastewaters-Du Pont Memphis (DM-1-HC-08)	Cyanide					0.638
		Acetonitrile					50
		Acrylonitrile					0.013
		Carbon tetrachloride					0.0015
		Chloroform					0.0083
		Dibromochloromethane					0.0013
		Methylene chloride					0.010
		Vinyl chloride					0.029
		Nitrite					11.5
		Copper					0.0063
		Iron					2.72
		Lead					0.0088
		Mercury					<0.0002

Table 4-5. Parameters Used in Surface Water Screening for Onsite Landfills

Waste Stream	Facility	Area (acres)	Infiltration Rate (m/yr)
Chloride/sulfate waste water treatment solids	Millennium HPP, Baltimore, MD	95	0.2609
Sulfate process, digestion sludge			
Sulfate process, gypsum			
Ilmenite wastewater treatment sludge titanium dioxide	Du Pont New Johnsonville, New Johnsonville, TN	27.5	0.4674

Table 4-8. Results of Surface Water Screening Analysis for Surface Impoundments

Waste Stream	Facility	COC (mg/L)	C_{an}	C_{River}	Minimum Screen Type	Minimum Screen Benchmark	Pass/Fail?
Hydrogen cyanide combined wastewaters	Du Pont Memphis Memphis, TN	Acetonitrile	50	5.5E-03	Shower	0.038	PASS
		Acrylonitrile	0.013	5.4E-07	HH-AWQC	0.000059	PASS
		Acrylamide	0.013	5.4E-07	HBL	0.00025	PASS
		Carbon tetrachloride	0.00150	6.3E-08	HH-AWQC	0.000025	PASS
		Chloroform	0.00830	3.5E-07	HH-AWQC	0.0057	PASS
		Dibromchloromethane	0.0013	5.4E-08	HH-AWQC	0.00041	PASS
		Hydrogen cyanide	0.638	7.0E-05	Shower	0.00058	PASS
		Methyl chloride	0.0300	1.3E-06	HBL	0.085	PASS
		Methylene chloride	0.010	4.2E-07	HH-AWQC	0.0047	PASS
		Nitrite	11.5	1.3E-03	HBL	1.6	PASS
		Vinyl chloride	0.0290	1.2E-06	HBL	0.0008	PASS
		Copper	0.00630	7.8E-07	Freshwater-AWQC	0.009	PASS
		Iron	2.72	3.4E-04	HH-AWQC	0.3	PASS
		Lead	0.00880	1.1E-06	Fresh water-AWQC	0.0025	PASS
		Mercury	0.0001	1.2E-08	Fresh water-AWQC	0.00077	PASS

Table 5-10. Percentiles of Generalized Adult ADD

Percentile	Adult ADD (L/kg-d)
1%	0.0054
5%	0.0075
10%	0.0091
25%	0.0125
50%	0.0175
75%	0.0245
90%	0.0336
95%	0.0404
99%	0.0575

Table 6-9. Risk Results Sludge Residues— Sodium Chlorate Sector Managed in Municipal Landfills

	Arsenic - Noncancer		Arsenic - Cancer		Manganese		Nickel		Zinc	
Percentile	Adult HQ	Child HQ	Adult Risk	Child Risk	Adult HQ	Child HQ	Adult HQ	Child HQ	Adult HQ	Child HQ
50th	0.0e+00	0.0e+00	0.0e+00	0.0e+00	1.8e-11	3.6e-11	0.0e+00	0.0e+00	0.0e+00	0.0e+00
75th	2.5e-07	5.3e-07	2.1e-11	1.5e-11	1.4e-05	2.8e-05	1.9e-10	3.9e-10	2.6e-14	5.4e-14
80th	1.1e-05	2.4e-05	9.1e-10	7.3e-10	3.4e-05	7.1e-05	8.2e-09	1.7e-08	1.6e-11	3.3e-11
85th	9.8e-05	2.0e-04	8.3e-09	6.1e-09	8.7E-05	1.8e-04	1.5e-07	3.2e-07	1.3e-09	2.7e-09
90th	5.7e-04	0.0011	5.0e-08	3.7e-08	2.2e-04	4.4e-04	1.5e-06	3.2e-06	2.0e-07	4.1e-07
95th	0.0033	0.0067	2.8e-07	2.3e-07	6.1E-04	0.0013	1.6e-05	3.1e-05	5.4e-06	1.1e-05
97.5th	0.010	0.021	9.0e-07	7.1e-07	0.0015	0.0030	5.6e-05	1.2e-04	3.4e-05	7.2e-05
99th	0.030	0.061	2.6e-06	2.1e-06	0.0035	0.0075	2.5e-04	5.1e-04	1.8e-04	4.1e-04

Table 6-13. Risk Results for Sulfate Process Secondary Gypsum—Titanium Dioxide Sector Managed in Onsite Industrial Landfill

	Antimony		Arsenic - Noncancer		Arsenic - Cancer		Manganese	
Percentile	**Adult HQ**	**Child HQ**	**Adult HQ**	**Child HQ**	**Adult Risk**	**Child Risk**	**Adult HQ**	**Child HQ**
50th	0.04	0.08	1.8e-10	3.9e-10	1.8e-14	1.4e-14	0.0014	0.0029
75th	0.15	0.32	3.5e-04	7.1e-04	2.8e-08	2.1e-08	0.026	0.054
80th	0.20	0.41	0.001	0.002	8.9e-08	7.0e-08	0.041	0.085
85th	0.25	0.52	0.003	0.006	2.6e-07	1.9e-07	0.064	0.13
90th	0.33	0.70	0.007	0.015	5.8e-07	4.5e-07	0.10	0.20
95th	0.47	0.99	0.015	0.031	1.4e-06	1.1e-06	0.15	0.32
97.5th	0.61	1.32	0.022	0.046	2.3e-06	1.7e-06	0.20	0.44
99th	0.83	1.79	0.032	0.068	3.9e-06	2.7e-06	0.28	0.61
Central Tendency	0.14	0.30	9.3e-06	2.0e-05	5.2e-10	6.8e-10	0.018	0.039
High End Full Distribution	0.33	0.71	0.01	0.02	6.1e-07	8.1e-07	0.077	0.16
High End Half Distribution	0.33	0.71	0.01	0.02	6.1e-07	8.1e-07	0.11	0.23

Table 6-22. Comparison of Risk Results for Manganese in Ilmenite Process Wastewater Treatment Sludge for 100 Percent and 10 Percent Waste Quantities

Percentile	100% Waste			10% Waste			Ratio		
	Groundwater Concentration (mg/L)	Adult HQ	Child HQ	Groundwater Concentration (mg/L)	Adult HQ	Child HQ	Groundwater Concentration (mg/L)	Adult HQ	Child HQ
50th	2.6e-04	1.0e-04	2.0e-04	2.1e-04	7.8e-05	1.6e-04	0.79	0.76	0.79
75th	0.20	0.072	0.15	0.13	0.048	0.10	0.65	0.66	0.67
80th	0.43	0.16	0.34	0.26	0.10	0.20	0.61	0.58	0.59
85th	0.97	0.37	0.76	0.55	0.22	0.45	0.56	0.60	0.59
90th	2.10	0.8	1.6	1.26	0.48	1.0	0.60	0.63	0.62
95th	4.17	1.6	3.3	2.75	1.0	2.2	0.66	0.66	0.66
97.5th	6.06	2.5	5.4	4.36	1.8	3.9	0.72	0.72	0.71
99th	8.84	4.1	8.6	6.68	3.1	6.3	0.76	0.74	0.74

Table 6-30. Comparison of Risk Results for Manganese in Ilmenite Wastewaters Managed in an Onsite Surface Impoundment for Z-Well Constrained with Z-Well Unconstrained

	Constrained Z-Well			Unconstrained Z-Well			Ratio of Unconstrained Z-Well to Constrained Z-Well		
Percentile	Groundwater Concentration (mg/L)	Adult HQ	Child HQ	Groundwater Concentration (mg/L)	Adult HQ	Child HQ	Groundwater Concentration (mg/L)	Adult HQ	Child HQ
50th	6.7e-06	2.4E-06	4.9E-06	1.1e-05	3.9e-06	7.8e-06	1.6e+00	1.6	1.6
75th	8.8e-05	3.2E-05	6.6E-05	1.3e-04	4.8e-05	9.9e-05	1.5e+00	1.5	1.5
80th	1.4e-04	5.4E-05	1.1E-04	2.1e-04	7.8e-05	1.6e-04	1.5e+00	1.5	1.5
85th	2.2e-04	8.6E-05	1.8E-04	3.3e-04	1.3e-04	2.7e-04	1.5e+00	1.5	1.5
90th	3.7e-04	1.5E-04	3.1E-04	5.9e-04	2.4e-04	5.1e-04	1.6e+00	1.6	1.6
95th	7.5e-04	3.2E-04	7.0E-04	1.4e-03	5.7e-04	0.0012	1.8e+00	1.8	1.7
97.5th	1.4e-03	6.0E-04	0.0013	2.6e-03	0.0012	0.0026	1.9e+00	2.0	2.0
99th	2.6e-03	0.0012	0.0025	5.6e-03	0.0026	0.0054	2.2e+00	2.2	2.2

1.0 Introduction

1.1 Background

The U.S. Environmental Protection Agency's (EPA's) Office of Solid Waste (OSW) is responding to a consent decree that established deadlines for EPA to propose and promulgate hazardous waste listing determinations for 14 production processes in the inorganics chemical manufacturing industry. Section 3001(e)(2) of the Resource Conservation and Recovery Act (RCRA), as amended, requires EPA to make listing determinations on whether to list specific inorganic chemical industry wastes as hazardous under Section 3001(b)(1) in accordance with the hazardous waste listing criteria. These criteria require EPA to list the waste if it is capable of posing a substantial present or potential hazard to human health or the environment when improperly treated, stored, transported, disposed of, or otherwise managed. Based on a settlement agreement with the Environmental Defense Fund (EDF), EPA must promulgate a final listing determination on or before October 31, 2001, with the listing determination to be proposed for public comment on or before August 30, 2000.

This document describes the risk assessment performed in support of EPA's listing determinations for wastes generated in the production of inorganic chemicals, as specified in the EDF/EPA settlement agreement. Based on an evaluation by EPA of current waste management practices in the inorganics chemical manufacturing sectors covered by the consent decree, EPA determined that wastes generated in five sectors required further evaluation, including quantitative analysis and assessment of risks to human health. The five sectors are

- Wastes from production of sodium phosphate from wet process phosphoric acid
- Sodium chlorate production wastes
- Inorganic hydrogen cyanide production wastes
- Titanium dioxide production wastes
- Antimony oxide production wastes.

The risk assessment described herein was restricted to characterizing the risks to human health through the groundwater pathway. Human exposures from groundwater that is used for drinking water and, where appropriate, showering were evaluated. The groundwater analysis was conducted in three phases: a Monte Carlo analysis, a sensitivity analysis, and a deterministic analysis. Statistical methods were used to analyze the results of the Monte Carlo analysis to identify the most sensitive parameters. This information was then used to define the parameter values for the deterministic analysis. In addition, a screening level analysis was conducted of potential impacts to surface waters from subsurface discharge of groundwater.

1.2 Purpose

The purpose of this risk assessment was to characterize human health cancer and noncancer risks from the management of specific wastes in each of the chemical manufacturing sectors for which EPA determined that a risk assessment was needed. Risks were characterized within the context of particular waste management scenarios, taking into consideration the various uncertainties underlying the analysis. This technical background document describes the methodology and assumptions for conducting the risk analysis, including fate and transport modeling and exposure modeling, specifies the parameter values and distributions used in the analysis, and presents the results.

1.3 Document Organization

This technical background document is organized as follows.

- Section 2.0, Analytical Framework, gives an overview of the technical approach used in the risk assessment.

- Section 3.0, Waste Characterization and Management, describes and documents the information on wastes used in the analysis. For each waste, the waste management scenarios, waste volumes, physical and chemical analyses, constituents of concern, and site locations are discussed.

- Section 4.0, Fate and Transport Modeling, describes the screening analyses used to identify wastes and constituents of concern (CoCs) for which groundwater modeling was conducted and the screening analyses conducted for surface waters. This section describes the groundwater modeling methodology, using EPA's Composite Model for Leachate Migration with Transformation Products (EPACMTP), and the site-specific data used in the modeling analyses. This section also discusses the modeling of infiltration and recharge rates for landfills and surface impoundments. Results of the fate and transport modeling are given in terms of both concentrations and dilution attenuation factors. The results of the probabilistic and deterministic groundwater modeling analyses are compared and contrasted.

- Section 5.0, Exposure Assessment, describes the selection of human receptors, the development of the average daily dose and the lifetime average daily dose distributions, and the underlying exposure factor distributions.

- Section 6.0, Human Health Risk Characterization, discusses all facets of the human health risk characterization, including methods, data inputs, results, and limitations and uncertainties.

- Section 7.0, References, lists all sources cited in this background document.

- Appendix A - Hydrogen Cyanide Production Sector

- Appendix B - Sodium Phosphate Production Sector
- Appendix C - Sodium Chlorate Production Sector
- Appendix D - Titanium Dioxide Production Sector
- Appendix E - Antimony Oxide Production Sector
- Appendix F - Development of Health-Based Levels for Household Water for Screening Volatile Constituents for Inhalation Risk
- Appendix G - Approach for Performing Sensitivity Analyses
- Appendix H - Surface Impoundment Infiltration Model
- Appendix I - Distribution Coefficients
- Appendix J - Human Health Benchmarks

2.0 Analytical Framework

This section gives an overview of the risk assessment technical approach, introducing the elements of the risk assessment and describing them in general. More detailed discussions of the methods, models, and data inputs used in individual components of the assessment are presented in Sections 3.0 through 6.0.

2.1 Establishing Risk Assessment Scenarios

The inorganics listing risk analysis used a single-exposure scenario as the basis for risk characterization for all waste stream/waste management scenario/constituent of concern (WS/WMS/CoC) combinations. This exposure scenario contains the following components:

- An offsite residence with a drinking water well located downgradient from the waste management unit
- Child and adult receptors who obtain all household water from the residential well
- Evaluation of the residential tap water exposure pathway.

The identification of exposure pathways, receptors, and CoCs is discussed in the following sections.

2.1.1 Identification of Exposure Pathways

The risk assessment focuses on chronic cancer and noncancer risk resulting from exposure to tap water drawn from residential wells downgradient from solid waste management units (SWMUs) where wastes from the manufacture of inorganic chemicals are managed. Groundwater was assumed to be contaminated from CoCs leaching from the SWMU into the underlying surficial aquifer and migrating downgradient to an offsite residential well. It was further assumed that the groundwater well was used as the sole source of tap water for the adults and children living at that residence. Tap water is assumed to be used as drinking water and for bathing or showering. In certain cases, the analysis also evaluated risks associated with subsurface discharges to surface waters that may or may not be used for drinking water. A number of inorganics chemical manufacturing facilities are located adjacent to navigable waters, which may intercept the flow of the surficial aquifer. For certain facilities, there is no possibility that residential wells could be placed between the SWMU and the surface waterbody; thus, for these facilities, only discharge to surface water was considered.

2.1.2 Identification of Receptors

Both child resident and adult resident receptor populations were modeled in the inorganics listing risk analysis. The adult resident was modeled using data for individuals between 20 and 64 years of age. For cancer risk, the child resident receptor was modeled as a 1- to 6-year-old because this cohort corresponds to the youngest cohort for which exposure duration variability data were available. In addition, the 1- to 6-year-old child cohort will generally experience a higher level of exposure relative to older child cohorts due to the elevated intake-to-body-weight ratio for the younger children. Thus, assessing risk to the 1- to 6-year-old child cohort covers older child cohorts without having to model them explicitly.

For noncancer risk, because exposure duration is not a factor, a single child cohort (i.e., 1- to 6-year-old cohort) was used. Tap water ingestion variability data were used for the 1- to 3-year and 4- to 6-year-old cohorts, and these age ranges were evaluated as a single cohort within the risk analysis. It should be noted that both the child cohorts used in this analysis exclude infant exposures in the first year of life.

2.1.3 Identifying Constituents of Concern

For this inorganics listing analysis, CoCs were defined as those chemical constituents that are present in leaching test extracts and that are not otherwise eliminated by initial screening-level analyses. Moreover, the pathways that were modeled in the fate and transport analysis were determined by additional screening analyses. This process is depicted in Figure 2-1, which shows the initial screening procedure followed by the fate and transport screening procedure.

All constituents present in leaching test extracts or wastewaters were considered in the initial screening analyses. These three analyses evaluated leachate concentrations against drinking water health benchmarks (HBLs), shower HBLs, and ambient water quality criteria, as indicated. If a constituent concentration in the leachate was found to be at or above drinking water HBLs or showering HBLs for volatile and semivolatile organic constituents (VOCs and SVOCs), it became a constituent to be evaluated in the fate and transport modeling. Leachate concentrations from municipal landfills were represented by the toxicity characteristic leachate procedure (TCLP), and leachate concentrations from Industrial D landfills were represented by the synthetic precipitation leaching procedure (SPLP) analytical results. Where the potential existed for subsurface releases to surface water, constituents with waste concentrations above ambient water quality criteria were also considered CoCs. The initial screening analyses are discussed further in Sections 3.0 and 4.0.

2.1.4 Exposure Pathway Screening

Constituents of concern defined by the initial screening analyses described above for each waste stream were additionally screened to determine whether they would be included in the risk modeling. These fate and transport screening analyses included

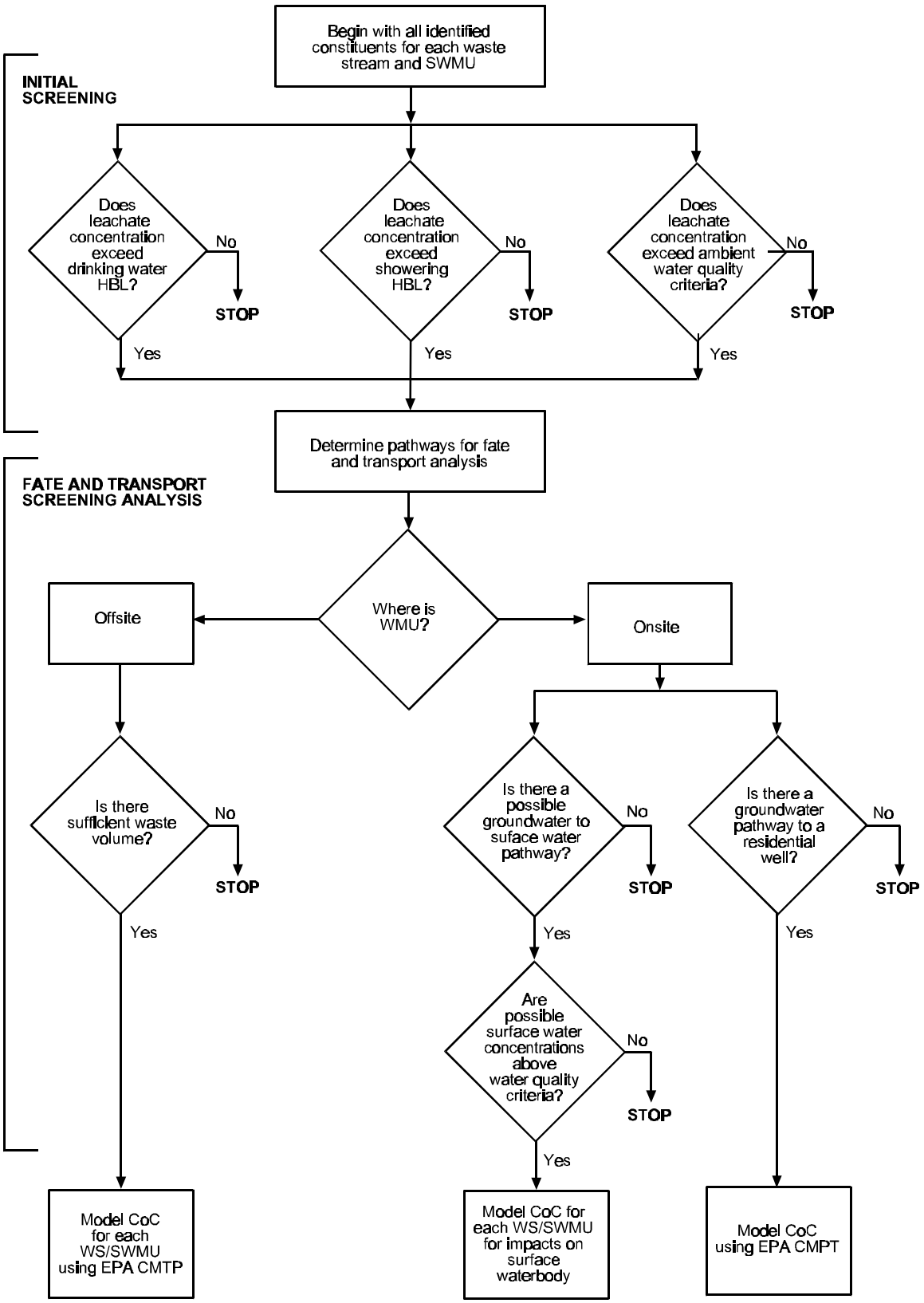

Figure 2-1. Procedure used for initial screening followed by fate and transport screening.

- For offsite SWMU where a groundwater pathway connecting the SWMU to a nearby residential well location was assumed, a de minimis waste volume screen was used to eliminate constituents from the groundwater modeling that were present in such minimal total quantities in the SWMU that they would pose no risk to human health under any leaching and aquifer conditions.

- For onsite SWMU where a groundwater to surface water pathway connects the SWMU to a nearby surface waterbody, a surface water screen was applied to constituents of waste streams managed in SWMUs. Constituents with concentrations estimated in the surface waterbody below ambient water quality criteria (AWQC) or HBL using maximum infiltration rates and low surface water dilution assumptions were not included in the groundwater modeling.

The de minimis screening analysis was applied only to small volume waste managed in offsite landfills. The surface water screening analysis was used only for wastes managed in onsite SWMUs where a surface body was identified and a groundwater to surface water pathway was apparent.

Constituents that passed either of these screening analyses were evaluated in the detailed groundwater fate and transport modeling conducted for this risk assessment. The fate and transport screening analyses and the detailed modeling are described in Section 4.0.

2.2 Waste Management Unit Characterization

EPA selected the types and locations of SWMUs for inclusion in the risk analysis based on responses to the 3007 Industry Questionnaire. Table 2-1 shows a matrix of the identified waste streams of concern and associated SWMU types. The four types of SWMUs included in the risk assessment are municipal landfills, on- and offsite industrial landfills, and surface impoundments.

2.2.1 Offsite Landfills—Municipal and Industrial D

Both offsite municipal landfills and industrial D landfills were modeled in this risk assessment. The parameters required for the risk assessment model were landfill area, waste volume, total waste concentration, and an estimated leachate concentration for each constituent. Landfill areas for both municipal and Industrial D landfills were based on the distribution of municipal landfill areas originally collected for use in the development of the toxicity characteristic regulation. The distribution of municipal landfill areas was used for both municipal and Industrial D landfills in this analysis because their distribution was judged to be representative of large commercial and municipal landfills that accept offsite industrial wastes. The distribution of areas for this subtype of industrial landfill was assumed to be more similar to the distribution of municipal landfill areas than to the distribution of areas for all Industrial D landfills. Total waste and leachate concentrations of constituents and annual waste volumes were based on information from EPA's waste stream sampling and analysis and the 3007 Questionnaire responses.

Table 2-1. Waste Stream/Waste Management Unit Combinations for Inorganic Chemical Manufacturing

Industry Segment	Waste Stream	Municipal Landfills	Offsite Industrial D Landfills	Onsite Industrial D Landfills	Surface Impoundment
Inorganic hydrogen cyanide production wastes	Commingled wastewater				✓
	Filter residues	✓	✓		
Sodium chlorate production wastes	Process sludge	✓			
	Filter wastes	✓	✓		
Sodium phosphate production wastes	Filter press cake		✓		
	Filter bags		✓		
Titanium oxide production wastes	Sulfate process digestion sludge			✓	
	Sulfate process: gypsum			✓	
	Sulfate process: digestion scrubber wastewater				✓
	Chloride and sulfate process: milling sand		✓		
	Off-spec titanium dioxide	✓			
	Chloride and sulfate process: mixed WWT solids			✓	
	Chloride and sulfate process: wastewaters				✓
	Ilmenite process: combined wastewaters				✓
	Ilmenite process: WWT solids		✓	✓	
Antimony oxide production wastes	Low antimony sludge			✓	

WWT = Wastewater treatment.

Offsite landfills were assumed to be located near the manufacturing facility generating each waste stream of concern. The general location of the current offsite SWMUs was assumed to represent the area within which any future waste management for the facility would be located. The soil, aquifer, and climate parameters used in the model were specific to the region where the current SWMU is located. The area within a 100-mile radius of each current SWMU was evaluated to determine the relative areal coverage of each of the three most representative soil types, the most applicable aquifer classifications for the area, and the most appropriate climate data to use to estimate infiltration rates and recharge for the area.

2.2.2 Onsite Industrial D Landfills

Onsite Industrial D landfills were modeled using site-specific data for landfill area, waste volume, and any soil and aquifer parameters available from EPA facility reports. In addition, soil and aquifer data were extracted from the STATSGO database (USDA, 1994a) for the area within the particular map unit where the facility is located and includes the nearest residences with drinking water wells of concern. A single aquifer type and one or more soil textures were identified in the vicinity of each SWMU. The concentrations of constituents in the leachate from onsite landfills were assumed to be the SPLP concentrations for that waste stream.

2.2.3 Onsite Surface Impoundments

Surface impoundments used in the inorganic chemical manufacturing industry are onsite SWMUs. For these SWMUs, the area, depth, and wastewater influent characteristics were obtained from EPA facility survey and sampling and analysis data. The soil and aquifer data for the immediate vicinity of the impoundment were characterized using site specific data sources and/or the STATSGO database (USDA, 1994a). Aquifer chacteristics and one or more soil textures were identified in the vicinity of each SWMU for use in the modeling. The modeling of surface impoundments was conducted in two stages. First, the infiltration rate from the surface impoundment was modeled using the infiltration algorithms from the surface impoundment source model developed for the HWIR analysis (U.S. EPA, 1999a. Second, the transport of the infiltrate through the unsaturated and saturated zones to the nearest downgradient drinking water well was modeled using EPA's Composite Model for Leachate Migration with Transformation Products (EPACMTP) (U.S. EPA, 1996a, b, 1997a). General descriptions of these models are provided in Section 2.4.2. More detailed descriptions are given in Section 4.3.3.

2.3 Waste Characterization

Waste streams considered for inclusion in this risk assessment were evaluated and characterized by EPA based on information from the 3007 Industry Questionnaire and the EPA waste stream sampling and analysis. Based on this information, EPA selected waste streams that warranted further assessment. Only the waste streams selected by EPA were addressed by the screening analysis and/or groundwater modeling described in this risk assessment background document. Sources of waste stream characteristic information required for estimating releases from SWMUs are presented in Table 2-2.

Table 2-2. Sources of Waste Stream Characteristics Required for Modeling in Each SWMU

Waste Stream Characteristic	Municipal Landfill (offsite)	Industrial D Landfill (offsite)	Industrial D Landfill (onsite)	Surface Impoundment
Annual waste quantity	3007 Survey	3007 Survey	3007 Survey	3007 Survey
Bulk density	EPA sampling data	EPA sampling data	EPA sampling data	NA
Waste concentration	EPA sampling data	EPA sampling data	EPA sampling data	EPA sampling data
Leachate concentration	EPA sampling data (TCLP)	EPA sampling data (SPLP)	EPA sampling data (SPLP)	EPA sampling data (SPLP filtrate)

NA = Not applicable.
SLP = Synthetic precipitation leaching procedures.
TCLP = Toxicity characteristics.

2.3.1 Use of 3007 Questionnaire Response Data

Waste management practices associated with waste streams were identified by EPA from the 3007 Industry Questionnaire. This information included types of SWMUs used to manage waste, SWMU locations, and waste volumes for each SWMU.

2.3.2 EPA Sampling and Analysis Data

EPA sampling and analysis data were used to characterize the wastes and leachate concentrations from the SWMUs used in the inorganic chemical industry. The waste streams were considered to be either wastewaters or nonwastewaters.

2.3.2.1 Wastewaters. Wastewaters managed in surface impoundments were modeled using leachate concentrations estimated from the measured concentrations in the SPLP filtrate or the total wastewater concentration. For wastewaters that contain less than 0.5 percent suspended solids, the measured concentrations in the total wastewater sample were used as the basis for the leachate concentration. For wastewaters containing greater than 0.5 percent solids, the SPLP was performed. The SPLP filtrate was analyzed separately from the extract and the filtrate analysis was used to represent the concentrations in the total wastewater available for leaching.

2.3.2.2 Nonwastewaters. The concentration of constituents in leachates from the two types of landfills was represented by results of two types of leaching analysis procedures. The leaching from municipal landfills was represented by TCLP results that reflect leaching in the slightly acidic environment expected in a municipal landfill.

SPLP results represent the leachate from a neutral pH environment. The results of the SPLP analysis were assumed to represent leachate from onsite and offsite Industrial D landfills and onsite monofills. The total waste concentrations were used to ensure that mass balance was maintained during modeling of all (municipal and industrial) landfills as finite sources.

2.4 Fate and Transport Modeling

The fate and transport modeling for this assessment considered releases to groundwater only. Groundwater was assumed to be contaminated from CoCs leaching from the SWMU into the underlying aquifer and migrating downgradient to offsite residential wells. Two models were used to characterize the fate and transport of CoCs. The infiltration rate from surface impoundments was modeled using a portion of the surface impoundment source model developed for the HWIR analysis (U.S. EPA, 1999a). The transport of the infiltrate through the unsaturated and saturated zones to the nearest downgradient drinking water well was modeled using EPACMTP (U.S. EPA, 1996a, b, 1997a). Environmental data collection methods are described below, followed by general descriptions of the surface impoundment infiltration rate model and the EPACMTP.

2.4.1 Environmental Data Collection

All of the soil parameters were dependent on soil texture classifications. The distribution of soil textures at each WMU site was considered regional for offsite WMUs. The distribution of soil types for each region was determined by identifying the soil texture classifications within a fixed radius of 100 miles around the current offsite facility managing the waste stream of interest. These data are contained in a STATSGO database according to map units. The predominant soil textures in each of the map units within the radius were identified, and the fraction of the total area within the radius covered by each soil type was determined. This fraction was used to set the distribution of soil textures used in the EPACMTP. The soil texture code was used to select values for the soil parameters required by the EPACMTP. These values were used to estimate transport in the unsaturated zone and to determine the infiltration for the WMU and the recharge rate for the surrounding region.

Climate data were used to estimate infiltration and recharge rate through the WMU and the surrounding area based on regional locations. The Hydrogeologic Evaluation of Landfill Performance (HELP) model was used by EPACMTP to develop infiltration and recharge rates for each soil texture and climate region combination. Infiltration and recharge depend on a combination of characteristics for soil and climate parameters. The climate parameters required by the model included long-term average precipitation, evapotranspiration, and surface runoff. National Oceanic and Atmospheric Administration (NOAA) data on precipitation and evaporation rates for 97 cities from the contiguous states were used as a source for regional climatic data in the HELP model. The climate center nearest the site and climate conditions of the WMU was selected for use in the modeling. WMU-specific infiltration and recharge rates could then be generated using the regional climate data. The infiltration rate was estimated assuming no landfill liner or leachate collection system. Therefore, infiltration for landfills was assumed to be the same as recharge.

Aquifer parameters were used in the EPACMTP model to estimate transport in the saturated zone downgradient of the WMU to the nearest residential well. The distribution of values used for the aquifer parameters was determined by assigning the WMU location to hydro geologic environments. The correlated aquifer parameters for the hydro geologic environments were obtained from the Hydro geologic Data Base (HGDB). However, for some onsite waste management units more locally specific aquifer characteristic data were obtained from the literature or site reports for use in the modeling.

2.4.2 Model Selection

The surface impoundments infiltration algorithms modeled the infiltration rate through the accumulated sediment at the bottom of the impoundment as the sediment layer changed over time. The model assumes that the surface impoundment has no engineered liner or leachate collection system.

The transport of CoCs to the nearest downgradient drinking water wells was modeled using EPACMTP. This model is used by EPA to make regulatory decisions for wastes managed in land disposal units (landfills, surface impoundments, waste piles, or land application units) for a number of EPA hazardous waste regulatory efforts. EPACMTP simulates flow and transport of contaminants in the unsaturated zone and aquifer beneath a waste disposal unit to yield the time history of the concentration arriving at a specified receptor well location. For use in a risk assessment, the receptor well concentration can be calculated as the peak or average concentration over a specified exposure time interval.

EPACMTP simulates steady-state flow in both the unsaturated zone and the saturated zone. For this analysis, the transient modeling option for finite source modeling scenarios was used. The output of EPACMTP is a prediction of the contaminant concentration arriving at a downgradient groundwater receptor well. EPACMTP can also perform Monte Carlo simulations to account for parametric uncertainty or variability. The flow and transport simulation modules of EPACMTP are linked to a Monte Carlo driver that permits a probabilistic evaluation of variability in model input parameters, as described by specified (joint) probability distributions.

2.5 Exposure Assessment

The exposure assessment estimated the dose to each receptor population by combining modeled CoC concentrations for tap water with relevant intake rates for the receptors being modeled. The inorganics listing risk assessment addresses chronic cancer and noncancer risk resulting from tap water ingestion and inhalation of volatile CoCs from tap water use (e.g., showering). The exposure assessment involved combining modeled drinking water well concentrations with applicable tap water ingestion rates to generate average daily dose (ADD) estimates for noncarcinogens and included exposure duration to generate lifetime averaged daily dose (LADDs) estimates for carcinogens. Elements of the exposure assessment are summarized in the following subsections.

2.5.1 Receptor Types

Child resident and adult resident receptor populations were modeled in the inorganics listing risk analysis. The child resident receptor was modeled as a 1- to 6-year-old. For evaluating carcinogen exposure in children, cohort aging was included in the analysis. Tap water ingestion variability data were available for the 1- to 3-year-old and 4- to 6-year-old cohorts, allowing these age ranges to be evaluated as a single cohort within a probabilistic analysis. It should be noted that the 1- to 6-year-old cohort used in this analysis excludes infant exposures in the first year of life.

The adult resident was modeled using data for individuals between 20 and 64 years of age. Cohort aging was not considered in modeling exposure for the adult resident, since this factor was expected to play a less significant role in determining overall exposure for the adult receptor relative to its importance in modeling child exposure.

2.5.2 Receptor Locations

The exposure assessment characterized residential exposure to CoCs that have migrated offsite in groundwater. The waste constituents dissolved in the leachate were transported downgradient to a groundwater receptor well. The exposure concentration was evaluated at the intake point of a hypothetical groundwater drinking water well located at a specified distance from the downgradient edge of the waste management unit. To be consistent with previous listing determinations, the distribution of distances to the nearest downgradient groundwater receptor well for all offsite SWMU types was based on survey information on the distance to nearest receptor for municipal landfills (U.S. EPA, 1997a, 1993). The data distributions for municipal landfills were used in this analysis for all offsite municipal and industrial landfills. For onsite SWMUs, wells were assumed to be located at site-specific locations no closer to the SWMU than the facility boundary and no farther away than 1 mile.

2.5.3 Receptor Activities

Pathways assessed for human exposure to contaminated residential well water included ingestion of drinking water and inhalation exposure to volatile constituents during daily showering. For the inhalation pathway, the risks estimated were primarily from exposures during daily showering.

2.6 Risk Characterization

2.6.1 Health Benchmarks

The inorganics listing risk assessment assessed chronic risk resulting from the inhalation and ingestion of CoCs contained in groundwater. Consequently, the toxicological benchmarks used were oral reference doses (RfDs), inhalation reference concentrations (RfCs), and cancer slope factors (CSFs). Cancer and noncancer effects were considered for each CoC for which benchmarks were available. Toxicological benchmarks were generally taken from the Integrated

Risk Information System (IRIS), although values from the Health Effects Assessment Summary Tables (HEAST) or other sources were also considered (e.g., NCEA toxicological issue papers).

2.6.2 Risk Descriptors

Human health risk characterization involved combining LADDs and ADDs with applicable toxicity factors (i.e., CSFs and RfDs/RfCs) to generate cancer risk and noncancer HQ estimates, respectively.

Cancer risk was characterized using lifetime excess cancer risk estimates to represent the excess probability of developing cancer over a lifetime as a result of exposure to the constituent of concern. Lifetime excess cancer risk estimates are the product of the LADD for a specific receptor/WS/WMS/CoC combination and the corresponding cancer slope factor.

Noncancer risk was characterized through the use of hazard quotients (HQs), which are generated by dividing an ADD by the corresponding RfD for ingestion. The ingestion hazard quotient uses the ADD as the exposure metric. An HQ establishes whether a particular individual has experienced exposure that places him or her either above or below a threshold of concern for a specific health effect. Therefore, unlike cancer risk estimates, HQs are not probability statements. The RfD and RfC represent "no-effects" levels that are presumed to be without appreciable risk from chronic exposures over a lifetime. They may be derived from human or animal studies and may include uncertainty factors to account for deficiencies in the available studies. The inhalation HQ is estimated by comparing the relevant air concentration (e.g., shower concentration) to the RfC. The result is the HQ. Since the RfC represents a protective environmental concentration and includes no dose estimate it is assumed to be protective of sensitive populations, including children, thus a single inhalation HQ is estimated for adults and children.

2.6.3 Risk Assessment Results

The results of the risk assessment conducted in support of the inorganic chemical industry listing decision included both probabilistic and deterministic estimates of groundwater concentrations at the residential well, the dilution and attenuation factors (DAFs) associated with these well concentrations, and risks or hazards associated with exposure to the residential use of the well water for drinking and/or showering. In this risk assessment the probabilistic analysis was conducted as the first step of the analytical procedure. The probabilistic analysis results are reported for the central tendency (50^{th} percentile) and several high end percentiles (75^{th}, 80^{th}, 85^{th}, 90^{th}, 95^{th}, 97.5^{th}, and 99^{th}). The DAFs are reported for the central tendency (50^{th} percentile) and several high end percentiles (10^{th}, 5^{th}, and 1^{st}). These results provide an evaluation of the distribution of the risk to the receptors, including those in the tail of the distribution.

A statistical analysis of the inputs and outputs of the probabilistic analysis was conducted as a second step in the risk assessment. This statistical sensitivity analysis determined the ranking of parameters for their contribution to increased risk from the central tendency risk estimates to high end risk estimates. The two parameters that ranked the highest were set to their high end values for the deterministic analysis.

A deterministic point estimate of central tendency and high end risk was made as the third and final step in the risk assessment. For the central tendency risk estimate all variable parameters in the groundwater analysis were set to median values used in the probabilistic analysis and the exposure assumptions were set at central tendency values as recommended in the EFH. For the high end estimate of risk the two parameters identified by the sensitivity analysis as contributing the most to increasing risk were set to their high end values. Parameters positively correlated with risk (e.g., WMU area) were set to their 90^{th} percentile value and parameters negatively correlated with risk (e.g., K_d) were set to their 10^{th} percentile value.

2.6.4 Sensitivity Analysis

A statistical regression analysis of the inputs and outputs of the probabilistic analysis was used to identify the contribution of each variable parameters to increased risk. This methodology provided insight into the interactions of parameters within the nonlinear groundwater model, EPACMTP. The sensitivity analysis included all direct inputs to the groundwater modeling and the risk equations and all intermediate inputs calculated within EPACMTP. This comprehensive evaluation of parameters provided insight into the analysis and highlighted the importance of parameters that had not previously been addressed, however, it did not address the importance of parameters that were constant in this analysis, for example, waste quantity. A sensitivity analysis was performed for every constituent of every waste stream and every waste management scenario. The results of the sensitivity analysis were used to identify the parameters to be set to high end for the deterministic analysis.

2.7 Uncertainty Analysis

EPA typically classifies the major areas of uncertainty in risk assessments as parameter uncertainty, scenario uncertainty, and model uncertainty. Parameter uncertainty is the "uncertainty regarding some parameter" of the analysis. Scenario uncertainty is "uncertainty regarding missing or incomplete information needed to fully define exposure and dose." Model uncertainty is "uncertainty regarding gaps in scientific theory required to make predictions on the basis of causal inferences" (U.S. EPA, 1992). This section identifies the primary sources for each of these types of uncertainty in the inorganic chemical manufacturing waste listing risk assessment.

2.7.1 Parameter Uncertainty

The sources of parameter uncertainty are measurement errors, sampling errors, variability, and use of generic or surrogate data (U.S. EPA, 1992). Many of the parameters that we used to quantify contaminant fate and transport and contaminant exposure and dose either were not measured or could not be measured precisely and/or accurately. Some of the most important and sensitive parameters in our analyses include those that describe waste composition; waste management practices; site characteristics (e.g., hydro geological, topographical, meteorological, and soils data); the physiologic and behavioral exposure characteristics of the receptors; the physical, chemical, and biochemical properties of the contaminants; and toxicological effects. The sensitivity analysis was used to identify the variable parameters having the greatest impact on risk. However, some parameters (e.g., waste quantity) were constant in this analysis and the uncertainty associated with constant parameters was not addressed.

2.7.2 Scenario Uncertainty

The sources of scenario uncertainty are descriptive errors, aggregation errors, errors in professional judgment, and incomplete analysis (U.S. EPA, 1992). Scenario uncertainty results from assumptions made concerning how receptors become exposed to contaminants and occurs because of the difficulty and general impracticality of making actual measurements of a receptor's exposure.

In certain cases, this risk analysis may have been incomplete, for example, scenario uncertainties in this analysis include:

- Landfills and surface impoundments are assumed not to have liners or leachate collection systems
- Evaluation of risks to infants (ages 0 to 1) is not considered
- Evaluation of the indoor exposure to household water uses besides showering is not considered
- Considered receptors are exposed to contaminated groundwater (i.e., wells exist downgradient from the source)

2.7.3 Model Uncertainty

The sources of model uncertainty are relationship errors and modeling errors (U.S. EPA, 1992). Models and their mathematical expressions are simplifications of reality that are used to approximate real-world conditions and processes and their relationships. Models do not include all parameters or equations necessary to express reality because of the inherent complexity of the natural environment and the lack of sufficient data to describe the natural environment. Consequently, models are based on numerous assumptions and simplifications and reflect an incomplete understanding of natural processes. The models selected for use in this risk assessment are described in Section 4.0. The selection was based on science, policy, and professional judgment. The groundwater model and the surface impoundment infiltration models were selected because they provided the information needed for this analysis and are, therefore useful for making listing determinations.

3.0 Waste Stream Characterization

The characterization of waste streams and their management units in this risk assessment was based on waste-stream-specific data. Onsite waste management practices were modeled using the descriptions provided by the facility or available through other site-specific sources such as RCRA Facility Investigation (RFI) or RCRA Facility Assessment (RFA) reports or state RCRA permits. Offsite management facilities were not modeled using facility-specific information and are, therefore, described generically. The location for these facilities was placed within 100 miles of the offsite facilities currently managing the wastes. The methods used to characterize each waste stream and its associated management practice are described in the following sections.

3.1 Sources of Waste Characterization Information

To characterize the wastes in this industry, EPA used two primary information sources and supplemented them with additional data sources. These two primary sources were the 3007 Industry Questionnaire and the EPA sampling and analysis data. The sources of waste characterization data are presented in Table 3-1.

Table 3-1. Sources of Waste Stream Characteristics Required for Modeling in Each SWMU

Waste Stream Characteristic	Solid Waste Management Unit			
	Municipal Landfill (offsite)	**Industrial D Landfill (offsite)**	**Industrial D Landfill (onsite)**	**Surface Impoundment**
Bulk density	EPA sampling data	EPA sampling data	EPA sampling data	NA
Waste concentration	EPA sampling data	EPA sampling data	EPA sampling data	EPA sampling data
Leachate concentration	EPA sampling data (TCLP)	EPA sampling data (SPLP)	EPA sampling data (SPLP)	EPA sampling data (SPLP filtrate)
Annual waste quantity	3007 Survey	3007 Survey	3007 Survey	3007 Survey

NA = Not applicable.
SPLP = Synthetic precipitation leaching procedure.
TCLP = Toxicity characteristic leaching procedure.

3.1.1 EPA Sampling and Analysis Data

EPA conducted sampling and analysis of all waste streams included in the risk analysis. The samples were analyzed for total constituent concentrations, specific gravity, moisture content, and pH. In addition, for all solid wastes and some wastewaters, two types of leaching tests were performed, the toxicity characteristic leaching procedure (TCLP) and the synthetic precipitation leaching procedure (SPLP). The data obtained from EPA's sampling and analysis were used to calculate the wet bulk density of the waste to represent the constituent concentration in the leachate and in the waste material.

3.1.1.1 Bulk Density of Waste Material. The wet bulk density of the waste was estimated using two measured waste parameters: specific gravity and the moisture content of the waste sample. Specific gravity represents the density of the solid portion of the waste stream only. To estimate bulk density from specific gravity, the percentages of solid and liquid waste are needed as well as the total porosity of the waste. The percentages of solids and liquids were available from the analytical data. Either the moisture content or percent solids was reported for all samples. The total porosity of the waste was assumed to be 45 percent, provided the pore volume represented by this porosity was sufficient to hold the measured moisture content of the waste. The bulk density was checked to ensure that the 45 percent pore volume was sufficient to accommodate the measured moisture content of the waste. If 45 percent porosity was not sufficient, the waste was assumed to be totally saturated. The minimum porosity needed to hold sufficient water to satisfy the measured water content of the waste was then estimated and the wet bulk density was recalculated assuming saturation of the waste. The wet bulk density for each waste stream was calculated using the sampling data. The results of these calculations are presented in Table 3-2.

3.1.1.2 Leachate Concentration. Two standard leaching procedure methods were used to estimate the concentration of constituents in the leachate from the waste management units modeled in this risk analysis. The TCLP method uses a dilute acetic acid solution as an extracting medium to represent the slightly acidic conditions expected in municipal landfills. The SPLP leaching procedure uses distilled water as the extracting medium to represent the neutral pH conditions expected in Subtitle D Industrial landfills. Thus, when modeling management in municipal landfills, TCLP extract concentrations were used to represent concentrations in landfill leachate, and, when modeling management in Industrial D landfills (either onsite or offsite), SPLP extract results were used to represent the constituent concentrations in the leachate from industrial landfills. In this analysis, when extract concentrations were below the detection limits of the analysis (but the constituent was known to be present in the waste), a value of half the detection limit was assumed as the concentration of the constituent. The constituent concentration data used in the analysis for each waste stream are presented in Appendixes A through E.

3.1.1.3 Waste Concentration. In addition to the concentration of constituents in the leachate, the total concentration of constituents in the waste is needed for the risk analysis of landfill scenarios. The total waste concentration was measured in the EPA waste sampling and analysis. Constituents not detected in the total waste analysis but assumed to be present in the waste stream were assumed to be present in concentrations of half the reported detection limit.

Table 3-2. Calculation of Bulk Density for Waste Streams from Measured Values of Specific Gravity and Percent Moisture or Solids

Sector	Waste	Waste ID	SG solids (kg/L)	Fraction Solids (kg/kg)	Fraction Liquid (kg/kg)	Total Porosity (L/L)	Dry Bulk Density (kg/L)	Wet Bulk Density (kg/L)
Hydrogen cyanide	Ammonia recycle filter waste	RH-1-HC-05	1.2	0.63	0.37	0.45	0.66	1.0
	Feed gas filters	NA	1.6	0.98	0.02	0.45	0.88	0.90
Sodium phosphate	Filter press cake	RCH-1-SP-01	NA	0.571	0.429	0.45	NA	2.0 (assumed)
	Filter press cake	RCH-1-SP-02	NA	0.283	0.717	0.45	NA	2.0 (assumed)
	Dust collector filter bags	RCH-1-SP-03	2	0.774	0.226	0.45	1.1	1.4
Sodium chlorate	Process sludge without Cr	HT-SN-01	2.4	0.607	0.393	0.61	0.94	1.5
	Process sludge without Cr	EC-SN-03	2.9	0.753	0.247	0.49	1.5	2.0
	Process sludge without Cr	EC-SN-01	2.9	0.95	0.05	0.45	1.6	1.7
	Process sludge without Cr	EC-SN-02	2.6	0.707	0.293	0.52	1.3	1.8
	Filter wastes without Cr	HT-FB-01	2.2	0.526	0.474	0.66	0.74	1.4
	Filter Wastes Without Cr	HT-FB-02	1.5	0.684	0.316	0.45	0.82	1.2
Titanium dioxide	Sulfate process digestion sludge	MI-SO-02	3	0.704	0.296	0.56	1.3	1.9
	Sulfate process secondary gypsum	MI-SO-03	2.7	0.534	0.466	0.70	0.80	1.5
	Chloride- sulfate process milling sand	KP-SO-05	2.5	0.858	0.142	0.45	1.4	1.6
	Off specification product	DPN-SO-02	3.4	1.00	0.00	0.45	1.9	1.9
	Chloride-sulfate WWT sludge	MI-SO-01	2.6	0.386	0.614	0.80	0.51	1.3
	Ilmenite process WWT sludge	DPE-SO-01	4.4	0.419	0.581	0.86	0.62	1.5
Antimony oxide	Low antimony slag	AC-1-AO-06	2.6	0.98	0.02	0.45	1.43	1.46
	Low antimony slag	AC-1-AO-01	2.7	0.98	0.02	0.45	1.48	1.51
	Feed gas filters		1.6	0.98	0.02	0.45	0.88	0.90

Total waste concentration was used with the annual waste quantity to determine total mass of each constituent in the waste. All constituents were assumed to be completely leached from the SWMU during the leaching period, using the finite source option in the groundwater model. Thus, the total waste concentration from the EPA sampling and analysis data was used to determine the fraction of constituent in the waste (Fw):

$$FW = \frac{Concentration \in the\ waste}{Concentration \in the\ leachate} \quad (3\text{-}1)$$

This fraction represents the rate at which the constituent is leached from the landfill. Leaching is assumed to continue until all constituent is removed from the landfill. Thus, by using the leachate concentration, leachate volume, and total mass of constituent in the landfill, the duration of leaching can be estimated.

3.1.2 Identifying Constituents of Concern

Constituents of concern in each waste stream were initially identified from the EPA sampling and analysis data. These CoCs were evaluated to determine if they required evaluation in this risk assessment and, if so, in which scenarios. Three initial screening analyses of CoC concentrations were conducted: a drinking water screen, a shower screen, and an ambient water quality criteria screen. Figure 3-1 highlights the process used in the initial screening analyses. If a constituent in a given waste stream is not screened out from further analysis, the more complete fate and transport modeling is conducted, as discussed in Section 4.0.

3.1.2.1 Drinking Water Screening. The drinking water screening methodology was designed to identify the potential for exposure to constituents through the ingestion of drinking water from a residential well. The human health benchmark levels are compared to the leachate concentrations (TCLP or SPLP concentrations), and those constituents with leachate concentrations above the HBLs are considered constituents of concern for this risk assessment. These screening levels are presented in Table 3-3.

3.1.2.2 Shower Screening. The shower screening methodology was designed to quantify potential for exposure to volatile constituents through the inhalation pathway during daily showering. The method calculates screening HBLs for comparison with leachate concentrations or wastewater concentrations for liquid wastes managed in surface impoundments. Additional detail, including governing equations, is provided in Appendix F.

Table 3-4 lists the household parameters used in the model, which were obtained from the *Exposure Factors Handbook* (U.S. EPA, 1997b) and McKone (1987) and were assumed constant in this analysis. For noncarcinogens, these factors and the physical and chemical properties of the volatile constituent determine the air concentration of each constituent. The air concentration is compared to the RfC to yield the hazard quotient. If the target HQ is set to 1.0, a water

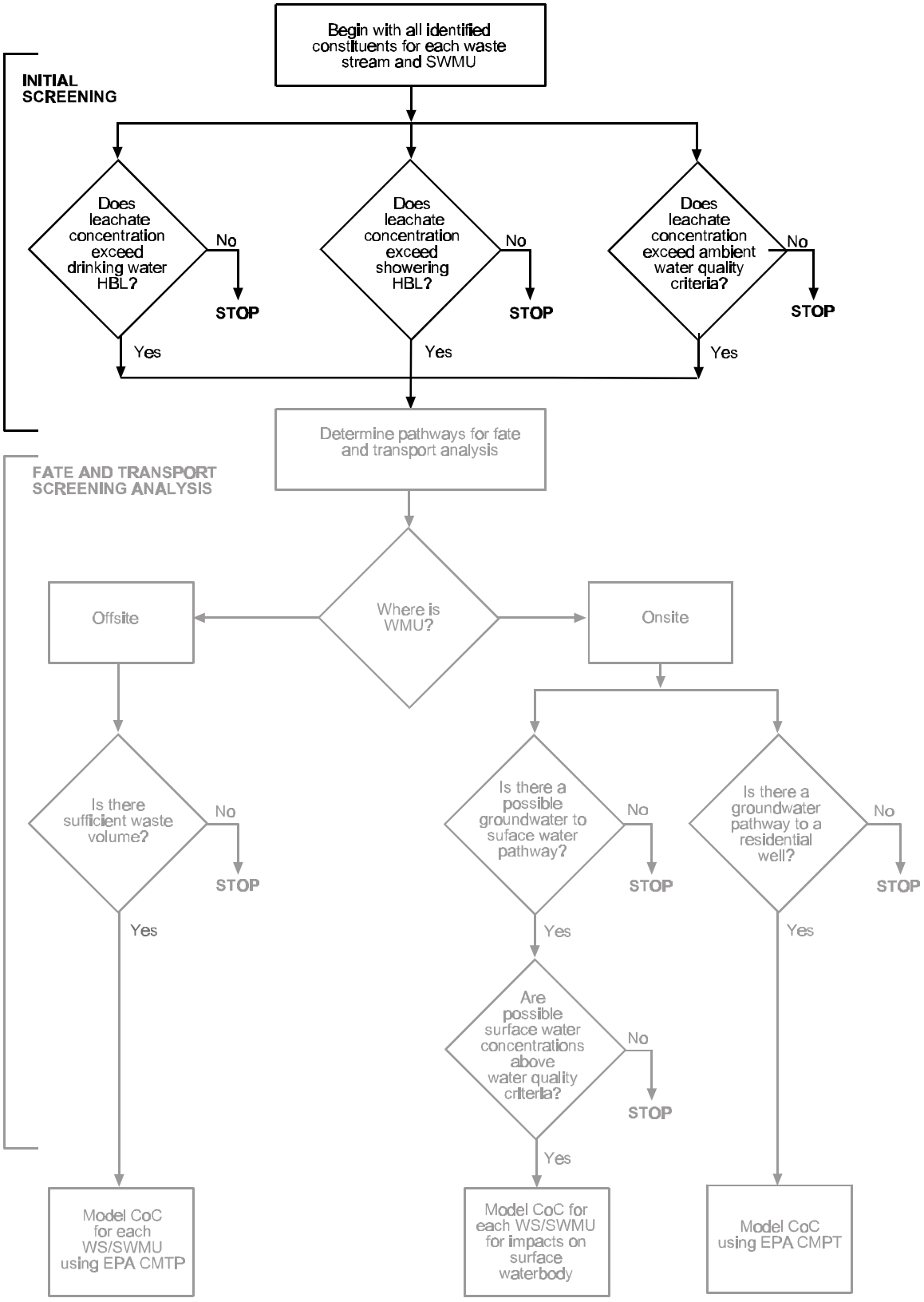

Figure 3-1. Procedure used for initial screening analysis.

Table 3-3. Health- and Ecology-Based Screening Levels

Substance	Drinking Water Ingestion or HBL[a,b]	AWQC				Waste Totals (mg/kg)
		Shower HBL	HH[c]	CCC[d]		
				Fresh Water	Salt Water	Soil Ingestion[a]
Acetonitrile	NA	0.036	NA	NA	NA	NA
Acrylamide	NA	20	NA	NA	NA	NA
Acrylonitrile	0.002	0.00045	0.000059	NA	NA	NA
Aluminum	16	NA	NA	0.087	NA	81,000
Antimony	0.0063	NA	0.014	NA	NA	32
Arsenic	0.00074 [e]	NA	0.000018 [f]	0.15	0.036	4.7
Barium	1.1	NA	1	NA	NA	5,600
Beryllium	0.031	NA	NA	NA	NA	160
Boron	1.4	NA	NA	NA	NA	7,200
Cadmium	0.0078	NA	NA	0.0022	0.0093	40
Carbon tetrachloride	NA	0.035	0.00025	NA	NA	NA
Chloroform	0.2	0.025	0.0057	NA	NA	NA
Chromium (III)	23	NA	NA	0.74	NA	120,000
Chromium (VI)	0.047	NA	NA	0.011	0.050	240
Cobalt	0.94[g]	NA	NA	NA	NA	4,800[g]
Copper	1.3[h]	NA	1.3	0.0090	0.0031	NA
Dibromochloromethane	0.01	0.035	0.00041	NA	NA	NA
Hydrogen cyanide	0.31	0.00052	0.7	0.0052	0.001	1,600
Iron	5 [i]	NA	0.3	1	NA	430,000
Lead	0.015[h]	NA	NA	0.0025	0.0081	400
Manganese	0.73	NA	0.05	NA	NA	3,800
Mercury (II)	0.0047	NA	0.000050	0.00077	0.00094	24
Methyl chloride	NA	0.035	NA	NA	NA	NA
Methylene chloride	1.3	0.13	0.0047	NA	NA	NA
Nickel	0.31	NA	0.61	0.052	0.0082	1,600
Nitrite	0.1	NA	NA	NA	NA	NA
Selenium	0.078	NA	0.17	0.0050	0.071	400
Silver	0.078	NA	NA	0.0034	0.0019	400
Thallium	0.0013	NA	0.0017	NA	NA	6
Vanadium	0.14	NA	NA	NA	NA	720
Vinyl chloride	0.0008	0.11	0.0020	NA	NA	NA
Zinc	4.7	NA	9.1	0.12	0.081	24,000

(continued)

Table 3.3 (continued)

NA = Not available.

[a] Health-based level (HBL) associated with a lifetime excess cancer risk of 1 in 100,000 or an HQ equal to 1.

[b] Except for arsenic, this value was calculated by assuming a 1- to 10-yr-old child having a drinking water intake rate of 64 mL/d (90th percentile value) or ~1.3 L/d. For arsenic, which is carcinogenic via ingestion, this value was calculated by assuming an adult age 20 and older having a drinking water intake of 21 mL/kg-d (mean value; ~1.4 L/d) and an exposure duration of 30 years (95th percentile value).

[c] National recommended water quality criteria for protection of human health (water + organism).

[d] National recommended water quality criterion continuous concentration (CCC) for protection of fresh water aquatic life.

[e] For comparison, background concentrations of arsenic in groundwater range from <0.001 to 0.01 mg/L.

[f] Based on a carcinogenic risk of 10-6.

[g] Based on a draft suggested guidance level for cobalt intake.

[h] Drinking water treatment action level, which triggers water systems into taking treatment steps if exceeded in more than 10 percent of tap water samples.

[i] Based on a provisional RfD for adults of 0.3 mg/kg-d derived from NHANES II (RDA for infants and children is higher than the RfD); assumes a drinking water intake for adults of 21 mL/kg-d (mean value; ~1.4 L/d) and a soil ingestion rate of 50 mg/d (mean value).

Table 3-4. Shower and Household Water Use Parameters

Parameter	Value	Units
Shower rate	5.5	L/min
Shower volume	2.00	m^3
Bathroom volume	10.0	m^3
Sh/B vent rate	100	L/min
Nozzle velocity	400	cm/s
Drop diameter	0.098	cm
Nozzle height	1.8	m
Time in shower	30	min

concentration that corresponds to this air concentration can be easily estimated. This water concentration is the HBL and is assumed to be protective of all adults and children. Table 3-5 presents the inhalation HBLs for noncarcinogens, the corresponding RfCs, and the physical and chemical properties used in the analysis.

The shower screening model also requires human exposure factors for carcinogenic constituents, including inhalation rate, body weight, and exposure duration for adults and children. Table 3-6 shows the central tendency and high-end values used for these exposure factors, which were obtained from the *Exposure Factors Handbook* (U.S. EPA, 1997b). These factors were varied when estimating inhalation HBLs for carcinogens by setting each to its high- end value

while holding all other factors in the analysis at central tendency. The lowest water concentration estimated for either adults or children was assumed to be the limiting HBL. Adult exposure duration was the limiting exposure factor for all constituents. Table 3-7 provides the limiting inhalation HBLs, the corresponding inhalation cancer slope factors, and the physical and chemical properties used to calculate the HBL for each carcinogen addressed in the analysis.

3.1.2.3 Ambient Water Quality Criteria Screening. The ambient water quality criteria (AWQC) screening methodology was designed to quantify potential for adverse effects to humans and aquatic organisms from exposure to constituents in groundwater discharged to surface water. AWQC criterion continuous concentration (CCC) values are presented for fresh water and salt water environments. The CCC is one of several components of the National Ambient Water Quality Criteria (NAWQC) for the protection of aquatic organisms. In general, it is equal to the lowest of the Final Chronic Value, the Final Plant Value, and the Final Residue Value. The latter three values are derived from chronic aquatic toxicity data or calculated from acute data when available chronic data are not adequate. The NAWQC methodologies include requirements for a minimally acceptable toxicological data set for calculating these values. A CCC is intended to be a good estimate of a threshold of unacceptable effects (as opposed to adverse effects). If maintained continuously, any concentration above the CCC is expected to

Table 3-5. HBLs, RfCs, and Physical and Chemical Properties for Constituents Evaluated for Noncancer Endpoints

Constituent	HBL (mg/L)	RfC (mg/m^3)	Henry's Law Constant ($atm\text{-}m^3/mol$)	Diffusion Coefficient in Water (cm^2/s)	Diffusion Coefficient in Air (cm^2/s)
Acetone	25	31	2.9E-05	1.1E-05	1.2E-01
Acetonitrile	0.036	0.06	3.5E-05	1.7E-05	1.7E-5
Acrylonitrile	0.00045	0.002	1.0E-4	1.3E-5	1.2E-1
Bromomethane	0.00038	0.005	1.4E-02	8.0E-06	8.0E-02
Carbon disulfide	0.053	0.7	1.3E-02	1.0E-05	1.0E-01
Chloroform	0.0025	0.081	3.7E-03	1.0E-05	1.0E-01
Hydrogen cyanide	0.00052	0.003	1.3E-04	1.8E-05	2.0E-01
Methacrylonitrile	0.011	0.0007	1.4E-06	8.0E-06	8.0E-02
Methyl ethyl ketone	2.2	1	1.1E-05	9.8E-06	8.1E-02
Methyl isobutyl ketone	0.019	0.08	1.2E-04	7.8E-06	7.5E-02
Methylene chloride	0.13	3.0	2.2E-3	1.2E-5	1.0E-1

HBL = Health-based level.
RfC = Reference concentration.

Table 3-6. Exposure Input Parameters for Inhalation of Carcinogens

Parameter	Adult	
	CT	High End
Event frequency (event/d)	1	1
Exposure frequency (d/yr)	350	350
Exposure duration(yr)	13	31
Body weight (kg)	70	89
Inhalation rate (m^3/d)	12.7	18.6

CT = Central tendency.

Table 3-7. HBLs, CSF_is, and Physical and Chemical Properties for Constituents Evaluated for Cancer as an Endpoint

Constituent	HBL (mg/L)	CSF_is $(mg/kg/d)^{-1}$	Henry's Law Constant (atm-m^3/mol)	Diffusion Coefficient in Air (cm^2/s)	Diffusion Coefficient in Water (cm^2/s)
Acrylamide	20	4.5	1.0E-9	1.1E-05	9.7E-02
Acrylonitrile	0.00045	0.24	1.0E-04	1.3E-05	1.2E-01
Benzene	0.071	0.029	5.4E-03	9.8E-06	8.8E-02
Bromodichloromethane	0.038	0.062	3.2E-03	8.0E-06	8.0E-02
Bromoform	0.86	0.0039	6.1E-04	8.0E-06	8.0E-02
Carbon tetrachloride	0.0035	0.053	3.0E-02	8.8E-06	7.8E-02
Dibromochloromethane	0.0035	0.084	7.8E-4	1.1E-5	2.0E-2
Chloroform	0.0025	0.081	3.7E-03	1.0E-05	1.0E-01
Methylene chloride	0.13	0.0016	2.2E-03	1.2E-05	1.0E-01
Methyl chloride	0.035	6.3E-3	8.8E-3	6.5E-6	1.3E-1
Vinyl chloride	0.011	0.015	2.7-02	1.2E-05	1.1E-01

HBL = Health-based level.
CSF_i = Cancer slope factor (inhalation).

cause unacceptable effects (Stephan et al., 1985). The freshwater criteria are used for all sites; however, in locations where salt water or brackish environments may be encountered, the salt water criteria are used if they are more conservative than the fresh water criteria. The AWQC levels are compared to the leachate concentrations and those constituents with leachate concentrations above the AWQCs are considered potential constituents of concern. These screening levels are also presented in Table 3-3.

3.1.3 SWMU Characterization

EPA distributed questionnaires to the inorganic chemical manufacturing facilities to obtain information about the wastes generated and their common waste management practices. These responses provided data for characterizing waste generation and waste management practices. The data obtained from the 3007 Questionnaires included annual waste generation rates, which were used to calculate the volume of waste managed in the WMU both annually and/or over the lifetime of the SWMU.

The annual waste quantity generated by each facility for each waste stream evaluated in the risk analysis was obtained from the 3007 Industry Questionnaire. The waste volumes were associated with the current management practice and that practice was assumed to be unchanged for the foreseeable future. The annual waste quantities, their current management practices, and the location of the current waste management unit are presented in Table 3-8.

All information on waste management practices for specific waste streams was obtained from the responses provided by manufacturing facilities in the 3007 Industry Questionnaire. Each waste stream characterization was matched with the waste management practice for the specific waste stream to be modeled. Types of data obtained from the Industry Questionnaire include

- Type of waste management unit
- Location of current waste management unit
- Current annual waste quantity.

The types of waste management units considered in this risk assessment are onsite and offsite Industrial D landfills, municipal landfills, and onsite surface impoundments. The types of waste management units and the waste stream combinations considered in this risk assessment are listed in Table 3-9.

3.1.3.1 Offsite Industrial D Landfills and Municipal Landfills. Offsite landfills were evaluated as management scenarios for waste streams in all sectors of the industry except the antimony oxide sector. As reported in the Industry Questionnaire, the offsite landfills used to manage the waste streams of interest in this risk analysis were both Industrial D and municipal landfills. Offsite landfill areas were characterized for this risk analysis with a single distribution of areas for both the Industrial D and municipal landfills. This empirical distribution of landfill areas was obtained from a survey conducted to characterize the population of municipal landfills

Table 3-8. Waste Quantities Modeled in the Inorganic Chemical Manufacturing Waste Listing Risk Assessment

Sector	Waste	Waste Management	Location	Annual Waste Quantity (MT/yr)
Hydrogen cyanide	Ammonia recycle filter waste	Industrial D LF	Anahuac, TX	21.5
		Municipal LF	Millington,TN	24
	Combined WW	Surface impoundment	Millington,TN	5,725,472
		Surface impoundment	Theodore, AL	748,300[a]
	Natural gas feed filters	Municipal LF	Millington,TN	0.2
Sodium phosphate	Filter press cake	Industrial D LF	Chicago, IL	108
	Dust collector filter bags	Industrial D LF	Chicago Il E. St. Louis, IL Augusta, GA	1.35 0.05 0.7
Sodium chlorate	Process sludge without Cr	Municipal LF	Elgin, SC Starkville, MS Ephrata, WA	135 130 89
	Filter wastes without Cr	Municipal LF	Blythe, GA Ephrata, WA	2.3 0.5
	Filter wastes without Cr	Industrial LF	Perdue Hill, AL	0.6
	Filter wastes with Cr	Industrial LF	Houston, MS	2.3
Titanium dioxide	Sulfate process digestion sludge	Onsite industrial LF	Baltimore, MD	24,494
	Sulfate process secondary gypsum	Onsite industrial LF	Baltimore, MD	51,710
	Chloride-sulfate process milling sand	Industrial D LF	Savannah, GA	200
	Off-spec product	Municipal LF	West Camden, TN Pass Christian, MS	295 268
	Chloride-sulfate WWT sludge	Onsite industrial LF	Baltimore, MD	93,121
	Sulfate process scrubber WW	Onsite SI	Baltimore, MD	1,702,333
	Commingled TiO_2 WW	Onsite SI	Baltimore, MD	2,961,801 outflow[b] 3,332,495 inflow
		Onsite SI	New Hamilton, MS	7,356,798
	Ilmenite WW	Onsite SI	Pass Christian, MS	11,178,200
		Onsite SI	New Johnsonville, TN	23,469,251
	Ilmenite process WWT sludge	Industrial LF	Edgemoor, DE	108,862
		Onsite LF	New Johnsonville, TN	121,000
Antimony oxide	Low antimony slag	Onsite industrial LF	Thompson Falls, MT	20

[a] After commingling HCN-process-only WW (20,800 MT/yr) with non-HCN process wastewaters.
[b] The outflow volume of 2,961,801 MT/yr was used in the analysis.

Table 3-9. Total Waste Stream Concentration and Leachate Concentration Data Used in Modeling Analyses

Sector	Waste Stream	Constituent	TCLP (mg/L)	SPLP (mg/L)	SPLP filtrate (mg/L)	Total solid waste (mg/kg)	Total wastewater (mg/L)
HCN	Ammonia recycle filters (RH-1-HC-05)	Antimony	0.55	0.59		81.5	
		Arsenic	0.045	0.039		5.8	
		Cadmium					
		(RH-1-HC-05)	<0.0050	<0.0050		<0.23	
		(RH-2-HC-05)	<0.05	0.0168		7.4	
		(DM-1-HC-04)	<0.05	<0.05		<5	
		(DM-2-HC-04)	0.087	0.0065		2.1	
		Nickel	0.5	0.61		1,460	
		Cyanide (total)	0.218	2.4		4	
	Feed gas filters	Barium	<2	0.069		168	
		Boron	7.4	<0.5		17,900	
		Lead	0.03	0.003		18.5	
		Nickel	0.4	<0.05		91	
		Zinc	13	<0.5		1,060	
	HCN process only wastewaters - Degussa (DG- 1-HC-07)	Acetonitrile					190[a]
	Combined wastewaters- Du Pont Memphis (DM-1-HC-08) (DM-2-HC-08)	Cyanide					0.638
		Acetonitrile					50
		Acrylonitrile					0.013
		Carbon tetrachloride					NA
		Carbon tetrachloride					0.0015

(continued)

[a] Waste concentration after commingling with non-HCN wastewaters is estimated to be 5.3 mg/L.

Table 3-9. (continued)

Sector	Waste Stream	Constituent	TCLP (mg/L)	SPLP (mg/L)	SPLP filtrate (mg/L)	Total solid waste (mg/kg)	Total wastewater (mg/L)
		Chloroform					0.001
		Chloroform					0.0083
		Dibromochloromethane					0.0013
		Methylene chloride					NA
		Methylene chloride					0.01
		Vinyl chloride					0.029
		Nitrite					11.5
		Copper					0.0063
		Iron					2.72
		Lead					0.0088
		Mercury					<0.0002
NaPO4	NaPO4 filter cake (RCH-1-SP-01) (RCH-1-SP-02)	Antimony	<0.5	0.0298		0.5	
			<0.5	<0.025		<0.5	
		Thallium	<2	0.0055		<2	
			<2	0.0079		<2	
	NaPO4 dust collector filter bag (RCH-1-SP-03)	Antimony	<0.5	0.309		48.8	
		Arsenic	<0.5	0.0064		<0.5	
NaC103	Process sludge without chromium (HT-SN-01) (EC-SN-03) (EC-SN-01) (EC-SN-02)	Arsenic	0.03	<0.05		14.3	
			<0.005	<0.05		<5	
			<0.005	<0.05		<5	
			<0.005	<0.05		<5	
		Lead	0.024	<0.03		14.8	
			<0.03	<0.03		139	
			0.12	0.001		19.3	
			0.05	0.002		34.9	

(continued)

Table 3-9. (continued)

Sector	Waste Stream	Constituent	TCLP (mg/L)	SPLP (mg/L)	SPLP filtrate (mg/L)	Total solid waste (mg/kg)	Total wastewater (mg/L)
		Manganese	0.08	<0.05		69.2	
			4.5	<0.05		238	
			0.5	<0.05		125	
			0.7	<0.05		51.9	
		Nickel	<0.2	<0.05		7.4	
			0.4	<0.05		12.1	
			<0.2	<0.05		<5	
			<0.2	<0.05		<5	
		Zinc	<2	<0.5		111	
			10.6	<0.5		279	
			<2	<0.5		<50	
			<2	<0.5		<50	
	Filter wastes without chromium (HT-FB-01) (HT-FB-02)	Arsenic	0.014	0.003		7.3	
			<0.005	<0.005		5.3	
		Antimony	0.018	<0.005		34.1	
			0.012	<0.005		<5	
		Boron	6.1	<0.05		<50	
			0.67	<0.5		<50	
		Cadmium	<0.05	<0.05		22.5	
			<0.05	<0.05		<5	
		Chromium VI	NA	<0.02		<0.8	
			NA	0.19		2.8	
		Lead	0.024	0.06		8.7	
			0.02	0.012		7.1	
	Filter wastes with chromium (KM-FB-01)	Arsenic	<0.5	0.005		<0.5	

(continued)

Table 3-9. (continued)

Sector	Waste Stream	Constituent	TCLP (mg/L)	SPLP (mg/L)	SPLP filtrate (mg/L)	Total solid waste (mg/kg)	Total wastewater (mg/L)
TiO2	Sulfate process digestion sludge (MI-SO-02)	Antimony	0.02	0.023		2.4	
		Vanadium	<0.05	0.42		17.7	
		Aluminum	<1	2		162	
		Copper	<0.25	0.37		67.5	
		Iron	<1	12		628	
		Lead	0.03	0.004		0.7	
		Manganese	0.29	0.36		25.2	
		Zinc	0.47	0.3		<5	
	Secondary gypsum	Antimony	0.11	0.055		3.2	
		Manganese	11.5	3.1		673	
		Arsenic	<0.5	<0.0035		0.8	
		Copper	<0.25	0.005		2.4	
		Nickel	0.14	0.009		10.5	
	Milling sand	Antimony	<0.5	0.024		<0.5	
	Off-spec product	Lead	0.06	0.002		0.6	
	Chloride-sulfate WWTS	Manganese	468	2.63		12,700	
		Thallium	<2	0.003		3	
		Aluminum	<1	0.24		8,740	
		Arsenic	<0.5	0		1.6	
	Ilmenite WWTS	Antimony	<0.021	0.02		0.9	
		Arsenic	<0.0035	0.001		2.2	
		Manganese	252	16.3		10,600	
		Thallium	0.28	0.012		3.7	

(continued)

Table 3-9. (continued)

Sector	Waste Stream	Constituent	TCLP (mg/L)	SPLP (mg/L)	SPLP filtrate (mg/L)	Total solid waste (mg/kg)	Total wastewater (mg/L)
	Ilmenite WWTS - New Johnsonville	Antimony	<0.5	0.021		0.7	
		Arsenic	<0.5	<0.0035		2.8	
		Iron	567	2.2		63,200	
		Manganese	47.4	1.5		2,890	
		Mercury	<0.002	<0.0002		0.2	
		Thallium	<2.0	<0.00225		7.2	
	Chloride-sulfate WW - millennium	Manganese			9.95		119
		Arsenic			<0.005		0.022
		Nickel			0.011		0.4
	Sulfate process scrubber WW- Millennium HPP	Aluminum					0.58
		Copper					0.006
		Manganese					0.58
		Mercury					0.0032
	Chloride-only WW Kerr McGee	Antimony			0.044		<0.05
		Arsenic			0.001		0.04
		Molybdenum			0.23		0.53
		Thallium			<0.005		0.086
		Manganese			0.46		25.9
	Ilmenite WW Delisle	Manganese					3.3
							3.34
		Thallium					<0.005
							0.013
		Vanadium					0.018
							0.63
		Aluminum					0.65
							3.1

(continued)

Table 3-9. (continued)

Sector	Waste Stream	Constituent	TCLP (mg/L)	SPLP (mg/L)	SPLP filtrate (mg/L)	Total solid waste (mg/kg)	Total wastewater (mg/L)
		Copper					0.03
							0.007
		Iron					1.44
							16.7
		Lead					<0.003
							0.005
		Nickel					0.013
							0.02
	Ilmenite WW New Johnsonville	Aluminum					3.1
		Iron					16.7
		Lead					0.005
		Manganese					3.34
		Thallium					0.013
		Vanadium					0.63
Antimony oxide	Low antimony slag (AC-1-AO-01) (AC-1-AO-06)	Antimony	55.8	114		11,500	
			110	211		127,000	
		Arsenic	2	2.93		301	
			3.1	3.81		478	
		Boron	9.8	9.27		<500	
			8.5	8.06		<2500	
		Selenium	0.6	0.55		<50	
			0.6	0.331		<250	
		Vanadium	1.3	1.14		<50	
			0.6	1		<250	

for establishing the Toxicity Characteristic Regulation. This distribution of areas is presented in Table 3-10. Offsite landfills were assumed to have no liners or leachate collection systems and to have an active lifetime of 30 years.

3.1.3.2 Onsite Industrial D Landfills. Onsite landfills were reported to be used to manage waste streams in the titanium dioxide sector and in the antimony oxide sector. Three waste streams were modeled as managed at a single onsite Industrial D Landfill. A site-specific landfill area and annual waste quantities were provided by the facility. Onsite landfills were assumed to have no liner or leachate collection system and to have a lifetime of 30 years. The site-specific data included some soil and aquifer parameters provided by the facility. Some location-specific data were obtained from other site-specific sources (i.e., RFI and/or RFA reports or permits). Some of these data were used in the modeling. In addition, general semi-site-specific data were identified and obtained from the State Soil Geographic (STATSGO) database for the map unit ID of the facility location. The concentrations of constituents in the leachate from onsite landfills were assumed to be represented by the SPLP extract concentrations for that waste stream. The methodology for obtaining appropriate location parameters for use in the groundwater modeling is described in Section 4.3.1.1. The areas of the onsite landfills are presented in Table 3-11.

Table 3-10. Empirical Distribution of Landfill Areas

Area Range (m^2)	Relative Probability
4,000 to 8,090	0.10
8,090 to 20,200	0.15
20,200 to 60,700	0.25
60,700 to 194,000	0.25
194,000 to 420,000	0.15
420,000 to 9,350,000	0.10
	1.00

Table 3-11. Sizes of Onsite Industrial Landfills

Location	Waste Streams	Areas (acres)
Millenium HPP, Baltimore, MD ILF	Chloride/sulfate waste water treatment solids Sulfate process, digestion sludge Sulfate process, gypsum	95
Du Pont, New Johnsonville, TN	Ilmenite process waste water treatment sludge	27.5
Thompson Falls, MT	Low antimony slag	0.2

3.1.3.3 Surface Impoundments. Surface impoundments were reported as onsite SWMUs by seven facilities in the hydrogen cyanide and titanium dioxide production sectors. For modeling onsite surface impoundments, site-specific values either were provided by the facility or were obtained from other site-specific sources (e.g., RFI and/or RFA reports or permits). The following data were needed for each impoundment to be modeled:

- Surface area of the impoundment (supplied by the facility)
- Depth of liquid level (assumed constant)
- Depth of sludge layer (assumed variable due to periodic dredging)
- Specific gravity of sludge particles (assumed to be equal to the specific gravity of associated sludge).

These data were available from a variety of sources for each facility managing waste streams of concern in surface impoundments. Thus, SWMU-specific data were used in the analysis for these parameters. The surface area for each unit was known and was entered as a constant value. The depth of the liquid layer was assumed constant and depth of sludge layer was assumed to vary over time for each waste management unit. No liner or leachate collection system was modeled and the impoundment was assumed to have an active life of 50 years. The concentration of constituents expected to leach from surface impoundments was assumed to be best represented by the SPLP filtrate results. These data were used when available. If these data were not available, the concentration in the wastewater itself was used as the constituent concentration in the leachate from the surface impoundment. The details of the surface impoundment modeling are presented in Section 4.3.2.2. The parameters used for onsite surface impoundments are provided in Table 3-12.

Table 3-12. Parameters for Onsite Surface Impoundments

Facility	Waste Stream	Area (m^2)	Liquid Depth (m)	Sludge Depth (m)	Sludge-Specific Gravity (g/cm^3)	Distance Under Impoundment to Aquifer (m)	Throughput (MT/yr)
Du Pont Memphis	Combined wastewaters from hydrogen cyanide production	11,200	1.08	0.3	1.2 & 2.5	1.0 - 4.7	5,725,472
Degussa Theodore, AL		1,737	1.8	0.152	2.5	1.0	748,300
Millennium HPP	Combined wastewaters from chloride sulfate process	450,000	1.8	0.2 - 1.0	2.6	2.0 - 3.0	2,961,801
Kerr McGee	Combined wastewaters from chloride sulfate process	148,645	2.44	0.2 - 1.22	1.3	1.0 - 6.7	7,356,798
Du Pont Delisle	Ilmenite process wastewaters	13,904	2.53	0.2 - 0.91	3.5 - 4.4	1.0 - 1.5	11,178,200
Millennium HPP Batch Attack Lagoon	Sulfate process digestion scrubber wastewater	50,000	0.91	0.2 - 6.1	3	1.0	1,702,333
New Johnsonville	Ilmenite process wastewaters	28,328	1.52	0.2 - 3.05	3.5	1.0	23,469,251

4.0 Modeling Exposure Concentrations

4.1 Fate and Transport Analysis

As discussed in Section 2, the risk analysis for the inorganic chemical manufacturing waste listing decision adopted a tiered approach to estimating exposure concentrations and risks (Figure 4-1). The first tier (described in Section 3) was an initial screen that assumed direct contact with wastes or waste leachates. In that initial screen, waste leachate constituent concentrations (TCLP or SPLP) were compared directly against health-based limits for drinking water and showering and against ambient water quality criteria (AWQC) for the protection of aquatic life. Waste/constituent combinations failing this screen were subjected to the fate and transport analysis described in this section.

As shown in Figure 4-1, the fate and transport analysis includes the following major steps:

- Determine pathways for fate and transport of constituents in the environment.
- Conduct fate and transport screening analyses, using simple conservative models for the surface water and groundwater pathways (Section 4.2).
- Conduct full-scale modeling for pathways/wastes/constituents that do not pass the fate and transport screen (Section 4.3).

Relevant exposure pathways were identified using the initial screening results. For waste managed offsite, infiltration of leachate into groundwater and subsequent transport to residential wells was assumed to be a pathway to human exposure, and this pathway was modeled using a regional approach. However, for waste managed onsite, information on site conditions was collected for one or more of the sites where the wastes were actually managed and this information was examined to determine whether exposure through the groundwater or surface water pathway was possible. If so, either or both pathways were modeled using a site-specific approach.

For example, for SWMUs close to a large waterbody, groundwater often discharges into the lake, river, or estuary, and downgradient drinking water wells are highly unlikely. In this case, only the surface water screen was conducted. For sites where downgradient wells are possible, a de minimis screen was applied for very low volume wastes to determine whether further analysis of fate and transport via the groundwater pathway was needed. Table 4-1 shows the exposure pathways analyzed for each waste that failed the initial screening and was managed in onsite landfills or surface impoundments.

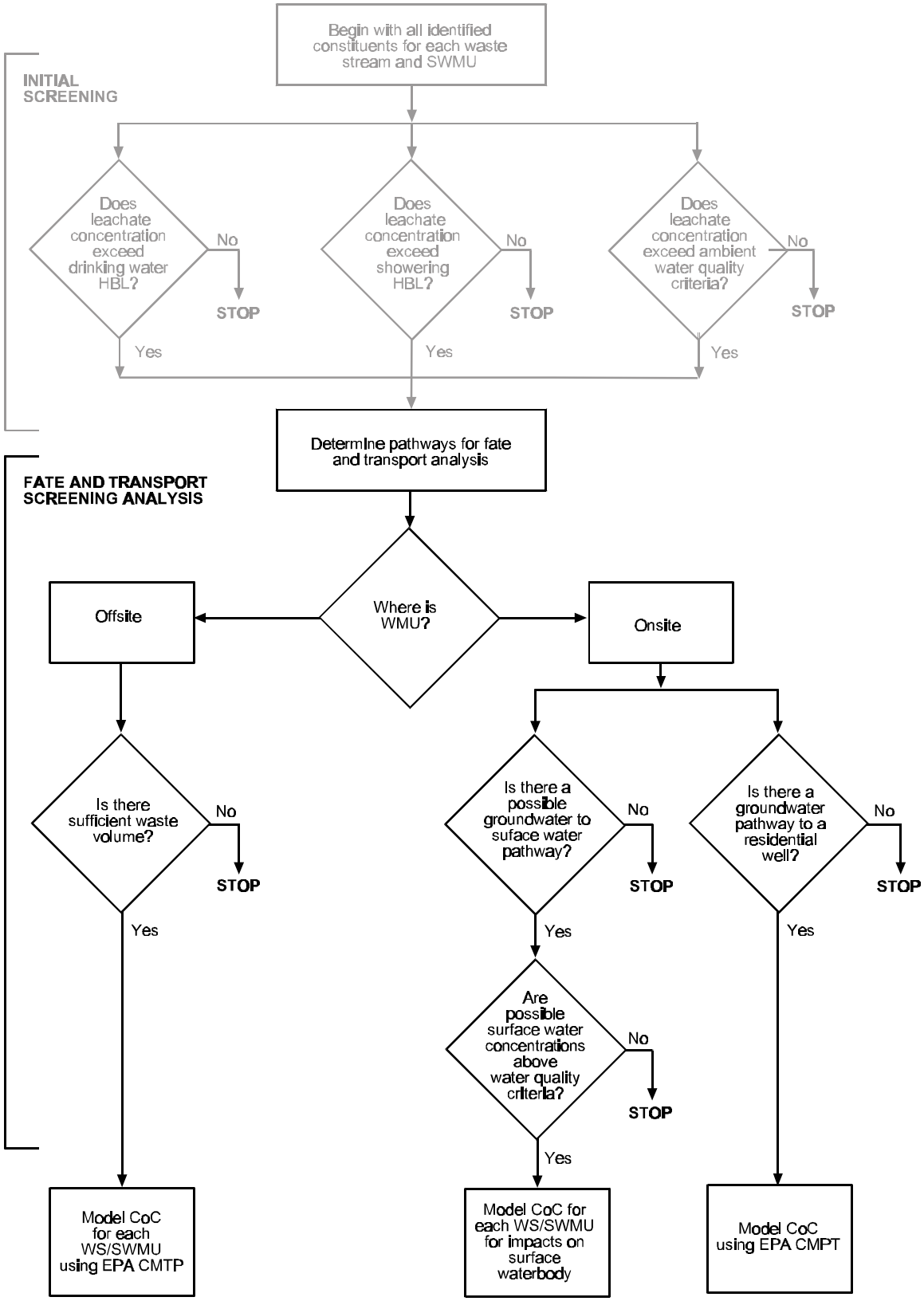

Figure 4-1. Fate and transport analysis for the inorganics chemical industry

Table 4-1. Exposure Pathways Addressed in Fate and Transport Analysis of Onsite Waste Management Units

Facility	Onsite Waste Management Unit Type	Exposure Pathways of Concern		Comments
		GW	SW	
Hydrogen Cyanide Sector				
Du Pont Memphis Plant, Millington, TN	Surface impoundment	No	Yes	Impoundment adjacent to Loosahatchie R. Canal; no downgradient drinking water wells
Degussa, Theodore, AL	Surface impoundment	Yes	No	No ambient water quality criteria exceeded in initial screening
Titanium Dioxide Sector				
Du Pont New Johnsonville Plant, New Johnsonville, TN	Landfill, surface impoundment	No	Yes	Impoundment discharge to Tennessee. River; no downgradient drinking water wells
Kerr-McGee Electrolytic Plant, Hamilton, MS	Surface impoundment	Yes	Yes	Impoundment near Tombigbee River
Millennium Hawkins Point Plant, Baltimore, MD	Landfill	Yes	Yes	Impoundment adjacent to Patapsco River; landfill at greater distance
	Surface impoundment	Yes	Yes	
Du Pont DeLisle Plant, Pass Christian, MS	Surface impoundment	Yes	Yes	Unit could discharge to St. Louis Bay
Antimony Oxide Sector				
U.S. Antimony Corp., Thompson Falls, MT	Landfill	Yes	No	No large waterbodies nearby

GW = Groundwater.
SW = Surface water.

Offsite waste management units (landfills) were assumed to impact only groundwater wells and, therefore, were only subject to the de minimis screen and possible full-scale modeling for the groundwater pathway.

This section describes the methodology and results, including assumptions and input data, used to model the fate, transport, and exposure concentrations for pathways identified to be of potential concern in the initial screen. Section 4.2 describes the screening analysis for surface

water as well as the de minimis screen conducted to identify waste streams that pose negligible risks through the groundwater pathway because of low waste volumes. Section 4.3 describes the full-scale modeling of the groundwater pathway using EPA's Composite Model for Leachate Migration with Transformation Products (EPACMTP).

4.2 Screening Analyses

Two fate and transport screening analyses were applied to the waste streams and constituents that did not pass the initial screen described in Section 3.

- A **de minimis screen,** applied where the groundwater pathway was of potential concern to identify wastes for which constituent concentrations and annual waste volumes are so low maximum risks remain below a level of concern.

- A **surface water screen**, applied to wastes managed in SWMUs that are assumed to discharge through groundwater to waterbodies to identify constituents that may adversely impact surface water quality.

Depending on the results of these screening models, waste/constituent combinations would be subjected to full-scale (less conservative) modeling (i.e., the waste failed the screen) or would be eliminated from further analysis or concern (i.e., the waste passed the screen).

Each of these screening methods was designed to be conservative so that wastes that passed each screen could be safely determined to pose no increased risk. The rationale and methodology behind each of these screening methods is explained in the following sections.

4.2.1 De Minimis Screen

Waste streams managed in offsite landfills that did not pass the initial screen were subjected to a second de minimis screen to identify wastes that are produced in such small volumes that they do not pose significant risk to human health through the groundwater pathway. The de minimis screen uses total waste concentrations, assumes the waste is disposed of in a landfill, and is based on a simple mass balance hydrologic model using conservative assumptions about site conditions and exposure. The screen assumes that the total annual mass of constituent in the waste stream is dissolved in the minimal annual volume of infiltration passing through the landfill. The resulting concentration is compared to the drinking water HBL. These assumptions are conservative because they maximize the drinking water concentration to which the receptors are exposed.

4.2.1.1 Assumptions. The de minimis methodology adopts the following conservative assumptions:

- All constituent mass placed in the landfill leaches from the waste annually.

- Soil conditions producing the lowest infiltration rates are present beneath the landfill.

- Landfill size is small (10th percentile of municipal landfills; 8,090 m^3 or 2 acres) (U.S. EPA, 1988).

- Receptors drink the leachate directly from the landfill.

The method assumes that the annual waste produced is placed in a single landfill and is evenly spread over the landfill area in the course of a year and is the only source of constituent entering the landfill. The offsite landfill is assumed to be a 10^{th} percentile from the distribution of municipal landfill areas (U.S. EPA, 1988) to minimize the volume of infiltrate used to dissolve the waste constituent, thus maximizing the concentration. Other assumptions include a daily per capita consumption of 83 gal/d, use of half the detection limit for waste concentrations below detection, and use of 30 percent Survey data for waste volumes and EPA sampling and analysis data for constituent concentrations.

4.2.1.2 Methodology. The method calculates leachate concentrations by first calculating the total mass of constituent in the waste:

$$M_T = C_T \times M_w \quad (4\text{-}1)$$

where

M_T = total constituent mass placed in landfill annually (mg)
C_T = total constituent concentration in waste (mg/kg)
M_w = annual waste production (kg).

Annual waste production was obtained from the waste stream data described in Section 3.0.

The leachate concentration is then estimated by dividing the total constituent mass by the amount of leachate produced from the landfill in a year:

$$C_L = M_T / (I \times A \times 1{,}000\ L/m^3) \quad (4\text{-}2)$$

where

C_L = leachate concentration (mg/L)
I = annual infiltration rate (m/yr)
A = landfill area (m^2).

The final step in the process was to construct a 10,000-record set of hydrogeologic environments and associated hydrogeologic parameters for each offsite landfill modeled. Using the hydrogeologic environment fractions defined for each 100-mile radius area, a hydrogeologic environment was assigned to each occurrence of that location in the 10,000-record location data set. For example, for the Du Pont Delisle Plant, the fractions assigned to hydrogeologic environments are 50, 25, and 25 percent for hydrogeologic environments 10, 6, and 7, respectively. Consequently, for this location, hydrogeologic environments 10, 6, and 7 would

occur approximately 5,000, 2,500, and 2,500 times, respectively, depending on the random assignments. Each waste production facility subject to the de minimis analysis was assigned to a HELP climate center, and the infiltration rate for the most limited soil texture (silty clay loam soil) for that climate center was used.

4.2.1.3 Screening Results. For the de minimis screen, the leachate concentration (C_L) was compared directly against the HBL (i.e., receptors were assumed to drink the leachate). When the leachate concentration was less than or equal to the HBL, the waste was screened from additional analysis (i.e., was not of concern from the human risk perspective). When the leachate concentration exceeded the HBL, the full-scale groundwater modeling exercise described in Section 4.3 was conducted.

At certain sites with very low infiltration rates, the amount of leachate produced annually by an 8,090-m^3 landfill is less than the amount of water consumed by a family of four. For these cases, if the site did not pass the first-stage de minimis screening, the total annual constituent mass (M_T) was divided by the family water consumption rate (83 gal/d × 4) to get the exposure concentration. If this concentration was equal to or lower than the HBL, the waste was screened from further risk analysis. If it exceeded the HBL, the waste was subjected to a full fate and transport analysis for the groundwater exposure pathway. The results of the de minimis screening analysis are presented in Table 4-2.

4.2.2 Surface Water Screen

The surface water screening analysis was conducted to quantify the potential for exposure to the constituent of concern in surface waterbodies by both human and aquatic life as a result of the infiltration of the constituent into soils beneath the SWMU and the subsequent transport in aquifers and discharge into the surface waterbodies.

4.2.2.1 Assumptions. To simplify the surface water screening methodology and to ensure conservative results, it was assumed that

- The SWMU is located adjacent to a surface waterbody—such as a river, stream, or lake
- The liquid in the SWMU leaks through the base of the unit and the underlying vadose zone to the aquifer
- Constituent concentrations are not decreased during subsurface transport by the processes of diffusion, dispersion, dilution, biodegradation, chemical hydrolysis, or sorption (the groundwater DAF is 1.0)
- All of the seepage from the aquifer discharges into the river immediately and is fully mixed with the river water
- The river is initially uncontaminated.

Table 4-2. Summary of De Minimis Screening Analysis Results

Sector	Waste Stream	Constituent	Waste Concentration (mg/kg)	Waste Quantity (kg/yr)	Total Mass (mg/yr)	Minimal Infiltration Rate (m/yr)	Landfill area (m^2)	Minimal Volume (L)	Estimated Leachate Concentration (mg/L)	HBL (mg/L)	DAF to Screen
HCN	Feed gas filters	Barium	168	200	33600	0.2824	8090	2,284,616.00	0.015	1	0.015
		Boron	17900	200	3580000	0.2824	8090	2,284,616.00	1.567	1.4	**1.12**
		Lead	18.5	200	3700	0.2824	8090	2,284,616.00	0.002	0.015	0.11
		Nickel	91	200	18200	0.2824	8090	2,284,616.00	0.008	0.31	0.026
		Zinc	1060	200	212000	0.2824	8090	2,284,616.00	0.093	5	0.019
Sodium phosphate	Dust collector filter bags	Antimony	48.8	1350	65880	0.06	8090	485,000.00	0.14	0.006	**22**
		Arsenic	0.25	1350	338	0.06	8090	485,000.00	0.0007	0.0007	**1**
Sodium chlorate	Filter waste without chromium	Antimony	34.1	500	17050	0.0003	8090	439,600.00	0.04	0.006	**6.46**
		Arsenic	7.3	500	3650	0.0003	8090	439,600.00	0.01	0.0007	**11.9**
		Boron	25	500	12500	0.0003	8090	439,600.00	0.03	1	0.03
		Cadmium	22.5	500	11250	0.0003	8090	439,600.00	0.03	0.008	**3.20**
		Chromium (VI)	2.8	500	1400	0.0003	8090	439,600.00	0.0032	0.05	0.06
		Lead	8.7	500	4350	0.0003	8090	439,600.00	0.01	0.015	0.66
	Filter waste with chromium	Arsenic	0.25	2300	575	0.2824	8090	2,284,616.00	0.000254	0.0007	0.3

DAF = Dilution attenuation factor.

The result of this screening calculation is an estimate of the final concentration of the constituent of concern in the river after the leachate from the SWMU has mixed with the water in the river.

4.2.2.2 Human Health Screen and Aquatic Life Screen. There are two sets of results from the surface water screen: fail/pass for human health and fail/pass for aquatic life. A given constituent passes the screen when the final river concentration is less than the appropriate toxicity benchmark for human health and for aquatic life. Conversely, a constituent fails when the river concentration is equal to or exceeds either the human health or aquatic life toxicity benchmark.

In the human health screen, the primary benchmark is the human health (HH) level associated with the ambient water quality criteria, hereafter referred to as AWQC-HH. If the final river concentration exceeds the AWQC-HH level, an additional screen against the health-based level is performed. The HBL is the concentration in drinking water that corresponds to a lifetime cancer risk of 1 in 100,000 or a hazard quotient of 1.0. In the aquatic life screen, the benchmark is the AWQC continuous concentration criterion (CCC) or AWQC-CCC. The fresh water AWQC-CCC is used for waste sites not located near salt waterbodies. The minimum of the fresh water and salt water AWQC-CCC is used for waste sites adjacent to brackish or salt water surface waterbodies. Table 4-3 lists the HBL, AWQC-HH, and AWQC-CCC levels for the constituents of concern in the surface water screening conducted in this study.

4.2.2.3 Screening Procedure. The first step of the analysis was to determine the infiltration rate from the waste management unit. This procedure is different for landfills and surface impoundments.

For landfill scenarios, the infiltration rate was obtained by using the HELP model. The closest climate center and the most conservative soil type were chosen for each SWMU location. Since more than one soil type can be present, the most conservative soil type was chosen from the soils expected to be encountered at the SWMU site (based on available GIS data) as the soil with the highest infiltration rate. For sites with all three soil types present (silty clay loam, silt loam, and sandy loam), the most conservative soil type is sandy loam. However, for a few of the surface impoundment sites, this soil type was not present. In these cases, silt loam or silty clay loam was used as the soil type in the surface water screening analysis. Based on the chosen soil type and climate center, the landfill infiltration rate was then estimated using the HELP model.

For surface impoundment scenarios, infiltration rate was calculated using the surface impoundment (SI) source model, as described in Section 4.3.2.2. For this screening analysis, a sludge layer thickness of 20 cm was used as input to the source model. Soil parameter values and liquid depth of the impoundment were chosen to be consistent with those used for the groundwater modeling. As was done for the infiltration rate calculated for use in the groundwater modeling, the infiltration rate calculated for use in the screening procedure was capped so as to prevent groundwater mounding from reaching the base of the impoundment and to limit the infiltration rate to be equal to or less than 99 percent of the impoundment inflow rate. None of the infiltration limitations were encountered while estimating infiltration from surface impoundments for surface water screening.

Table 4-3. Health- and Ecology-Based Screening Levels

Substance	Drinking Water Ingestion or HBL[a,b]	Shower HBL	AWQC		
			HH[c]	CCC[d]	
				Fresh Water	Salt Water
Acetonitrile	NA	0.036	NA	NA	NA
Acrylamide	NA	20	NA	NA	NA
Acrylonitrile	0.002	0.00045	0.000059	NA	NA
Aluminum	16	NA	NA	0.087	NA
Antimony	0.0063	NA	0.014	NA	NA
Arsenic	0.00074 [e]	NA	0.000018 [f]	0.15	0.036
Carbon tetrachloride	0.0085	0.035	0.00025	NA	NA
Chloroform	0.2	0.025	0.0057	NA	NA
Copper	1.3[g]	NA	1.3	0.0090	0.0031
Dibromochloromethane	0.01	0.035	0.00041	NA	NA
Hydrogen cyanide	0.31	0.00052	0.7	0.0052	0.001
Iron	5 [h]	NA	0.3	1	NA
Lead	0.015[g]	NA	NA	0.0025	0.0081
Manganese	0.73	NA	0.05	NA	NA
Mercury (II)	0.0047	NA	0.000050	0.00077	0.00094
Methyl chloride	0.085	0.035	NA	NA	NA
Methylene chloride	0.15	0.13	0.0047	NA	NA
Nickel	0.31	NA	0.61	0.052	0.0082
Nitrite	1.6	NA	1	NA	NA
Thallium	0.0013	NA	0.0017	NA	NA
Vanadium	0.14	NA	NA	NA	NA
Vinyl chloride	0.0008	0.11	0.0020	NA	NA
Zinc	4.7	NA	9.1	0.12	0.081

NA = Not available.

[a] Health-based level (HBL) associated with a lifetime excess cancer risk of 1 in 100,000 or an HQ equal to 1.

[b] Except for arsenic, this value was calculated by assuming a 1- to 10-yr-old child having a drinking water intake rate of 64 mL/d (90th percentile value) or ~1.3 L/d. For arsenic, which is carcinogenic via ingestion, this value was calculated by assuming an adult age 20 and older having a drinking water intake of 21 mL/kg-d (mean value; ~1.4 L/d) and an exposure duration of 30 years (95th percentile value).

[c] National recommended water quality criteria for protection of human health (water + organism).

[d] National recommended water quality criterion continuous concentration (CCC) for protection of fresh water aquatic life.

[e] For comparison, background concentrations of arsenic in groundwater range from<0.001 to 0.01 mg/L.

[f] Based on a carcinogenic risk of 10^{-6}.

[g] Drinking water treatment action level, which triggers water systems into taking treatment steps if exceeded in more than 10 percent of tap water samples.

[h] Based on a provisional RfD for adults of 0.3 mg/kg-d derived from NHANES II (RDA for infants and children is higher than the RfD); assumes a drinking water intake for adults of 21 mL/kg-d (mean value; ~1.4 L/d) and a soil ingestion rate of 50 mg/d (mean value).

Table 4-4 lists the infiltration rates used for surface water screening by site and SWMU type.

After the appropriate infiltration rate I was obtained, an areal leakage rate Q_i from beneath the waste management unit was calculated as follows:

$$Q_i = A\ I \tag{4-3}$$

where

A = area of the waste management unit (m^2)
I = infiltration rate (m/yr).

Waste management unit areas are also shown in Table 4-4 and Table 4-5.

The next step was to calculate a river dilution factor (RD) to account for the mixing of the seepage volume with the river water. RD is defined as

$$RD = \frac{Q_{River}}{Q_i} \tag{4-4}$$

where

Q_{river} = river flow rate (m^3/yr).

The choice of river flow rate depends on the receptor (human health or aquatic life), type of risk the constituent of concern poses (cancer or noncancer), and the location of the site relative to salt water or brackish waterbodies. For all aquatic life screenings that use the freshwater AWQC-CCC as the benchmark, Q_{river} is defined as the lowest 7-day average flow in a 10-year period (7Q10). For human health screening purposes, the flow rate Q_{river} is defined as follows:

- For carcinogens (e.g., arsenic), Q_{river} is defined as the harmonic mean flow rate.
- For noncarcinogens, Q_{river} is defined as the lowest 30-day average in a 5-year period (30Q5).

The tidal flushing rate of 203 m^3/s in the Patapsco River was used for the Millennium HPP facility near Baltimore, Maryland, because of the facility's proximity to the Chesapeake Bay. This flushing rate was used for both human health and aquatic life screening calculations. Table 4-6 shows the river flow rates that were used for the remaining inorganic facilities in the surface water screening.

Assuming that leachate migrates through the subsurface and into the river with no decrease in concentration and that the leachate is instantaneously and fully mixed with clean river

Table 4-4. Parameters Used in Surface Water Screening for Onsite Surface Impoundments

Facility	Waste Stream	Area (m^2)	Liquid Depth (m)	Sludge Depth (m)	Sludge-Specific Gravity (g/cm^3)	Distance Under Impoundment to Aquifer (m)	Throughput (MT/yr)	Infiltration Rate (m/yr)
Du Pont Memphis[a]	Combined Wastewaters from Hydrogen Cyanide Production	11,200	1.08	0.3	1.2 & 2.5	1.0 - 4.7	5,725,472	0.71
Millinneium HPP[b]	Combined Wastewaters from Chloride Sulfate Process	450,000	1.8	0.2 - 1.0	2.6	2.0 - 3.0	2,961,801	0.85
Kerr McGee[b]	Combined Wastewaters from Chloride Sulfate Process	148,645	2.44	0.2 - 1.22	1.3	1.0 - 6.7	7,356,798	0.70
Du Pont Delisle[b]	Ilmenite Process Wastewaters	13,904	2.53	0.2 - 0.91	3.5 - 4.4	1.0 - 1.5	11,178,200	1.34
Millennium HPP Batch Attack Lagoon[b]	Sulfate Process Digestion Scrubber Wastewater	50,000	0.91	0.2 - 6.1	3	1.0	1,702,333	2.00
New Johnsonville[b]	Ilmenite Process Wastewaters	28,328	1.52	0.2 - 3.05	3.5	1.0	23,469,251	1.13

[a] Inorganic Hydrogen Cyanide Listing Background Document for the Inorganic Chemical Listing Determination, August, 2000 (U.S. EPA, 2000b)
[b] Antimony Oxide Listing Background Document for the Inorganic Chemical Listing Determination, August, 2000 (U.S. EPA, 2000a)

Table 4-5. Parameters Used in Surface Water Screening for Onsite Landfills

Waste Stream	Facility	Area (acres)	Infiltration Rate (m/yr)
Chloride/sulfate waste water treatment solids	Millennium HPP, Baltimore, MD	95	0.2609
Sulfate process, digestion sludge			
Sulfate process, gypsum			
Ilmenite wastewater treatment sludge titanium dioxide	Du Pont New Johnsonville, New Johnsonville, TN	27.5	0.4674
Low antimony slag	Thompson Falls, MT	0.2	0.0069

water, the resulting final river concentration is related to the appropriate analytical concentration in the leachate through the following equation:

$$C_{river} = C_{An} / \mathrm{RD} \qquad (4\text{-}5)$$

where

C_{river} = final river concentration (mg/l^3)
C_{an} = analytical concentration in the leachate (mg/l^3).

For the industrial landfill the SPLP concentration is used as C_{an}. For surface impoundment scenarios the SPLP filtrate valve was used if available, otherwise the total waste water concentration was used.

The final river concentration was then compared with the AWQC-HH concentration for the human health screening and the AWQC-CCC for the aquatic life screening. Specifically, if C_{river} was less than the appropriate toxicity benchmark for a given constituent, then that constituent passed the surface water screening and no further analysis was conducted. However, if C_{river} equaled or exceeded the benchmark, then that constituent failed the screening. In this case, a full fate and transport modeling analysis for the groundwater pathway would be conducted for the waste.

4.2.3 Screening Results

Table 4-7 summarizes the results of the surface water screening analysis for landfills, and Table 4-8 summarizes the results of the surface water screening analysis for surface impoundments. For every industrial sector, the table shows that all wastes and constituents were screened out and no constituents were subject to a full groundwater to surface water fate and transport analysis.

Table 4-6. River Flow Statistics Collected for Use in Surface Water Screen

Facility	River	HUC	Flow Statistics (ft^3/s)				Comments	Period	Source
			30Q5	7Q10	Mean	Harmonic Mean			
Du Pont New Johnsonville Plant	Tennessee R	06040005	19,500	10,900	57,120	35,944	Savannah, TN, below dam at Pickwick (regulated flow)	1946-1992 (7Q10); 1946-1998 (mean)	USGS, Nashville, TN
Kerr McGee Electrolytic Plant	Tombigbee R	03160101	73	66	2,060	446	"Legal" 7Q10; regulated by Tenn-Tom; low flow preregulated values; time-sampling error = 12%	pre-1975 data	USGS, Pearl, MS
Du Pont Delisle Plant	Jourdan R	03170009	*127.63*	732.23	140	393	"svtnflow" and "mnflow" data, RF1 segment 22	Compiled early 1980s	Reach File 1 (RF1) database
Du Pont Delisle Plant	Wolf R	03170009	44	40	645	195	"Legal" 7Q10; near Landon, MS	1971-1999	USGS, Pearl, MS
Du Pont Memphis Plant	Loosahatchie R Canal	08010209	80.7	71.5	392	212	River mile 30.4 (near Arlington, TN)	1970-1992 (7Q10); 1970-1998 (mean)	USGS, Nashville, TN
Millennium Hawkins Point Plant									

Notes: Estimated 30Q5 = 1.1 x 7Q10, except for Tennessee River, which is 30Q5 = 1.4 x 7Q10 (U.S. EPA, 1991).
Estimated harmonic mean = 1.194 x (mean)0.473 x (7Q10)0.552 (U.S. EPA, 1991).
USGS data obtained via phone contact on 4/12/2000.
RF1 data available on BASINS CD-ROM (U.S. EPA Office of Water).

Table 4-7. Results of Surface Water Screening Analysis for Landfills

Waste Stream	Facility	COC (mg/L)	C_{an}	C_{River}	Minimum Screen Type	Minimum Screen Benchmark	Pass/Fail?
Chloride/sulfate wastewater treatment solids	Millenium HPP, Baltimore, MD ILF	Aluminum	0.24	3.75E-06	Fresh water-AWQC	0.087	PASS
		Arsenic	0.00005	7.82E-10	HH	0.000018	PASS
		Manganese	2.63	4.11E-05	HH	0.05	PASS
		Thallium	0.003	4.69E-08	HBL	0.0013	PASS
Sulfate process, digestion sludge	Millenium HPP, Baltimore, MD ILF	Aluminum	2.0000	3.13E-05	Fresh water-AWQC	0.087	PASS
		Antimony	0.0230	3.59E-07	HBL	0.0063	PASS
		Copper	0.3700	5.78E-06	Salt water-AWQC	0.0031	PASS
		Iron	12.0000	0.000188	HH	0.3	PASS
		Lead	0.0040	6.25E-08	Fresh water-AWQC	0.0025	PASS
		Manganese	0.3600	5.63E-06	HH	0.05	PASS
		Zinc	0.3000	4.69E-06	Salt water-AWQC	0.081	PASS
Sulfate process, gypsum	Millenium HPP, Baltimore, MD ILF	Antimony	0.0550	8.6E-07	HBL	0.0063	PASS
		Arsenic	0.0035	5.47E-08	HH	0.000018	PASS
		Copper	0.0050	7.82E-08	Salt water-AWQC	0.0031	PASS
		Manganese	3.1000	4.85E-05	HH	0.05	PASS
		Nickel	0.0090	1.41E-07	Salt water-AWQC	0.0082	PASS
Ilmenite wastewater treatment sludge titanium dioxide	Du Pont New Johnsonville, New Johnsonville, TN ILF	Antimony	0.021	8.29E-08	HBL	0.0063	PASS
		Arsenic	0.0035	7.5E-09	HH	0.000018	PASS
		Iron	2.2	8.69E-06	HH	0.3	PASS
		Manganese	1.5	5.92E-06	HH	0.05	PASS
		Mercury	0.0002	7.9E-10	HH	0.00005	PASS
		Thallium	0.00225	8.89E-09	HBL	0.0013	PASS

Table 4-8. Results of Surface Water Screening Analysis for Surface Impoundments

Waste Stream	Facility	COC (mg/L)	C_{an}	C_{River}	Minimum Screen Type	Minimum Screen Benchmark	Pass/Fail?
Hydrogen cyanide combined wastewaters	Du Pont Memphis Memphis, TN	Acetonitrile	50	5.5E-3	Shower	0.038	PASS
		Acrylonitrile	0.013	5.5E-7	HBL	0.000059	PASS
		Acrylamide	0.013	5.5E-7	HBL	0.00025	PASS
		Carbon tetrachloride	0.00150	6.3E-8	Shower	0.000025	PASS
		Chloroform	0.00830	3.5E-7	Shower	0.0057	PASS
		Dibromochloromethane	0.0013	5.5E-8	HBL	0.00041	PASS
		Hydrogen cyanide	0.638	7.0E-5	Shower	0.00058	PASS
		Methylchloride	0.0300	1.3E-6	Shower	0.085	PASS
		Methylene chloride	0.010	4.2E-7	Shower	0.0047	PASS
		Nitrite	11.5	1.3E-3	HBL	1.6	PASS
		Vinyl chloride	0.0290	1.2E-6	HBL	0.0008	PASS
		Copper	0.00630	6.9E-7	Salt water-AWQC	0.0031	PASS
		Iron	2.72	3.0E-4	Fresh water-AWQC	0.3	PASS
		Lead	0.00880	9.7E-7	Fresh water-AWQC	0.0025	PASS
		Mercury	0.0001	1.1E-8	Fresh water-AWQC	0.0008	PASS

(continued)

Table 4-8. (continued)

Waste Stream	Facility	COC (mg/L)	C_{an}	C_{River}	Minimum Screen Type	Minimum Screen Benchmark	Pass/Fail?
Chloride/sulfate wastewaters	Millenium HPP, Baltimore, MD ILF	Arsenic	0.0050	2.99E-07	HH	0.000018	PASS
		Manganese	9.95	5.95E-04	HH	0.05	PASS
		Nickel	0.011	6.58E-07	Salt water-AWQC	0.0082	PASS
	Kerr McGee Hamilton, MS	Antimony	0.044	7.0E-05	HBL	0.0063	PASS
		Arsenic	0.001	2.6E-07	HH	0.000018	PASS
		Manganese	0.46	7.3E-04	HH	0.05	PASS
		Molybdenum	0.23	3.7E-04	HBL	0.078	PASS
		Thallium	0.005	8.0E-06	HBL	0.0013	PASS
Sulfate process, digestion scrubber water	Millenium HPP, Baltimore, MD	Aluminum	0.58	9.04E-06	Fresh water-AWQC	0.087	PASS
		Copper	0.006	9.35E-08	Salt water-AWQC	0.0031	PASS
		Manganese	0.58	9.04E-06	HH	0.05	PASS
		Mercury	0.0032	4.99E-08	Fresh water-AWQC	0.0008	PASS
Ilmenite process, wastewaters	Du Pont Delisle Delisle, MS	Aluminum	3.1	3.76E-04	Fresh water-AWQC	0.087	PASS
		Copper	0.030	3.64E-06	Salt water-AWQC	0.0031	PASS
		Iron	16.7	2.03E-06	Fresh water-AWQC	0.3	PASC
		Lead	0.005	6.06E-07	Fresh water-AWQC	0.0025	PASS
		Manganese	3.34	4.05E-06	HH	0.05	PASS
		Nickel	0.02	2.43E-06	Salt water-AWQC	0.0082	PASS
		Thallium	0.013	1.58E-06	HBL	0.0013	PASS
		Vanadium	0.63	7.6E-05	HBL	0.14	PASS

(continued)

Table 4-8. (continued)

Waste Stream	Facility	COC (mg/L)	C_{an}	C_{River}	Minimum Screen Type	Minimum Screen Benchmark	Pass/Fail?
	Du Pont New Johnsonville, New Johnsonville, TN	Aluminum	3.1	5.7E-06	Fresh water-AWQC	0.087	PASS
		Iron	16.7	3.1E-05	Fresh water-AWQC	0.3	PASS
		Lead	0.005	9.2E-04	Fresh water-AWQC	0.05	PASS
		Manganese	3.34	6.1E-06	HH	0.00005	PASS
		Thallium	0.013	2.4E-08	HBL	0.0013	PASS
		Vanadium	0.63	1.2E-06	HBL	0.14	PASS

4.3 Groundwater Modeling

Groundwater fate and transport modeling was conducted for constituents of waste streams that did not pass the screening analyses described in Section 4.2. The modeling was conducted for solid wastes managed in onsite or offsite landfills and wastewaters managed in onsite surface impoundments and was directed toward estimating groundwater concentrations in residential drinking water wells downgradient from the waste management units. SWMU characteristics and constituent concentrations were obtained from data on current wastes and management practices described in Section 3.0 of this report.

The analysis used EPACMTP, a state-of-the-science vadose zone and groundwater fate and transport model designed specifically for regulatory applications. The model can be applied in both a probabilistic (Monte Carlo) and deterministic mode and was specifically modified for this application. Both Monte Carlo and deterministic model runs were conducted in this analysis. The Monte Carlo results were statistically analyzed to identify sensitive parameters for the high-end deterministic runs. Environmental modeling data were collected using both regional site-based and site-specific approaches, depending on whether wastes were managed offsite (regional data) or onsite (site-specific). Distributions were used to characterize potential site-to-site variability and within-site uncertainty in model input parameters.

Section 4.3.1 describes the modeling methodology, including a description of EPACMTP, code modifications for this analysis, and the Monte Carlo and deterministic modeling approaches. SWMU characteristics and waste constituent concentrations are described in Section 3.0. Modeling techniques to estimate infiltration rates for landfills and surface impoundments, as well as recharge outside of SWMUs, are described in Section 4.3.2. Environmental data, including soil properties, aquifer characteristics, and receptor well locations were collected as described in Section 4.3.3, and chemical properties, notably soil-water partition coefficients for metals and hydrolysis rate for cyanide, are described in Section 4.3.4. Section 4.3.5 describes the Monte Carlo input source data file. Model results are compared and discussed in Section 4.3.6.

4.3.1 Modeling Methodology

Only releases to groundwater were considered in this risk assessment. The EPACMTP groundwater model was used to estimate the fate and transport of constituents through the subsurface environment, as described here.

4.3.1.1 Description of EPACMTP. The groundwater pathway modeling conducted for two phases of the analysis (Monte Carlo analysis and deterministic analysis) was performed to determine the residential groundwater well exposure concentrations resulting from the release of waste constituents from the waste management unit. Liquid that percolates through the waste unit generates leachate, which can infiltrate from the bottom of the SWMU into the subsurface. For landfills, this liquid is in the form of precipitation; for surface impoundments, the liquid is the wastewater managed in the impoundments. The waste constituents dissolved in the leachate are then transported via aqueous phase migration through the vadose zone (unsaturated zone that lies below the bottom of the SWMU and above the water table) to the underlying aquifer (or

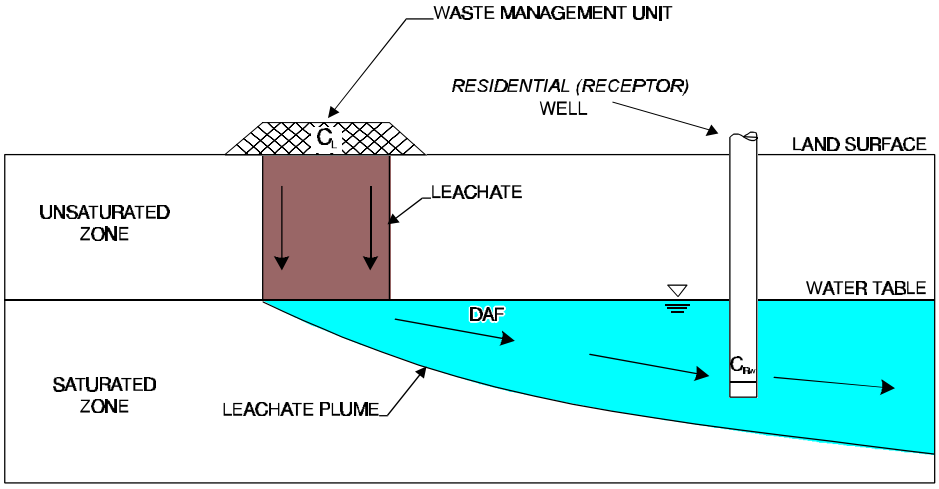

Figure 4-2. Schematic diagram of groundwater modeling scenario.

saturated zone) and then downgradient to a groundwater receptor well. The exposure concentration is evaluated at the intake point of a hypothetical groundwater drinking water well located at a specified distance from the downgradient edge of the waste management unit. This well is referred to hereafter as the "receptor well." This conceptual model of the groundwater fate and transport of contaminant releases from SWMUs is illustrated in Figure 4-2.

The conceptual procedure described here is quantitatively evaluated with a groundwater model developed by EPA, EPACMTP (U.S. EPA, 1996a, b, 1997a). EPACMTP is used by EPA to make regulatory decisions for wastes managed in land disposal units (landfills, surface impoundments, wastepiles, or land application units) for a number of EPA hazardous waste regulatory determinations. EPACMTP simulates flow and transport of contaminants in the unsaturated zone and aquifer beneath a waste disposal unit to predict the maximum concentration arriving at a specified receptor well location. For use in risk assessments, the receptor well concentration can be reported as the peak concentration or as the highest average concentration over an appropriate exposure time interval.

Fate and transport processes accounted for in the model are advection, hydrodynamic dispersion, linear and nonlinear sorption at equilibrium, and chemical hydrolysis. The composite model consists of two coupled modules: (1) a one-dimensional (1-D) module that simulates infiltration and dissolved contaminant transport through the unsaturated zone; and (2) a saturated zone flow and transport module that can be run in either a fully 3-D or quasi-3-D mode. Quasi-3-D mode simplifies the fully 3-D flow and transport solutions to one of two 2-D conditions. For conditions where the saturated zone is thin and the contaminant mass flux into the saturated zone is large, fully mixed conditions are assumed and an areal (x-y) planar approximation is implemented. For conditions in which flow in the horizontal transverse (y) direction is of minor significance, such as when infiltration through the SWMU area is relatively low compared to the groundwater flow rate, a vertical 2-D cross-sectional solution is employed where a numeric solution is achieved in the x-z plane and an analytical solution is used to expand this in the

transverse (y) direction. EPACMTP uses an automatic criterion for determining which of these quasi-3-D scenarios to apply based on the combination of aquifer parameters input by the user. The principal benefit of this quasi-3-D approach is that it provides substantial savings in computational effort, making large-scale Monte Carlo simulations feasible. It is for this reason that the quasi-3-D approach was used for all of the Monte Carlo runs in the Inorganic Listing Determination analysis. Fully 3-D solutions were used for the deterministic runs.

It is assumed that the soil and aquifer are uniform porous media and that flow and transport are described by the flow equation and the advection-dispersion equation, respectively. The flow equation is based on Darcy's law, which states that the flow per unit area of groundwater through porous media is the product of hydraulic conductivity and hydraulic gradient. The advection-dispersion equation describes solute transport by flowing groundwater (advection) and hydrodynamic dispersion resulting from mechanical mixing and molecular diffusion.

Flow and Transport Equations Used in EPACMTP. The groundwater flow simulation is based on the following simplifying assumptions:

- The aquifer is homogeneous
- Groundwater flow is steady-state
- Flow is isothermal and governed by Darcy's law
- The fluid is slightly compressible and homogeneous and
- The principal directions of the hydraulic conductivity tensor are aligned with the Cartesian coordinate system.

The governing equation for steady-state flow in three dimensions is

$$k_r K_x \frac{\partial^2 H}{\partial x^2} + k_r K_y \frac{\partial^2 H}{\partial y^2} + k_r K_z \frac{\partial^2 H}{\partial z^2} = 0 \qquad (4\text{-}6)$$

where

H	=	hydraulic head (m)
k_r	=	relative permeability (dimensionless)
K_x, K_y, and K_z	=	hydraulic conductivities (m/yr) in the longitudinal (x), horizontal transverse (y), and vertical (z) directions, respectively.

Further details about these parameters may be found in Freeze and Cherry (1979). Equation (4-6) is solved subject to the boundary conditions given in U.S. EPA (1996a).

Flow in the vadose zone is modeled as steady-state, one-dimensional, and vertical from underneath the source SWMU toward the water table. The lower boundary of the vadose zone is the water table. The flow in the vadose zone is predominantly gravity-driven; therefore, the vertical flow component accounts for most of the fluid flux between the source and the water table. The flow rate is determined by the long-term average infiltration rate through the SWMU.

For the saturated zone, relative permeability k_r is equal to unity. Flow in the saturated zone is based on the assumption that the contribution of recharge from the unsaturated zone is small relative to the regional flow in the aquifer, and the saturated aquifer thickness is large relative to the head difference that establishes the regional gradient. The implication is that the saturated zone can be modeled as having a uniform thickness, with mounding underneath the waste source represented by an increased head distribution along the top boundary.

The governing equation for transport in three dimensions is

$$\frac{\partial}{\partial x_i}\left(D_{ij}\frac{\partial C_l}{\partial x_j}\right) - V_i\frac{\partial C_l}{\partial x_i} = \theta R_l\frac{\partial C_l}{\partial t} + \theta\ Q_l\lambda_l C_l + \sum_{m=1}^{M}\xi_{lm}Q_m\lambda_m C_m \qquad (4\text{-}7)$$

where a subscript coordinate and Einstein summation convention are used to simplify the notation, that is,

x_1, x_2, and x_3	=	x, y, and z coordinate directions, respectively
t	=	time
C_l	=	concentration of the l-th component species in the n_c member decay chain,
		λ_l and
R_l	=	first-order decay coefficient and retardation coefficient, both for species l,
Q_l and Q_m	=	correction factors to account for sorbed phase decay of species l and
		parent m, respectively,
θ	=	water content.

For computation of the longitudinal, horizontal transverse, and vertical dispersion coefficients (D_{xx}, D_{yy}, and D_{zz}), the conventional dispersion tensor for isotropic porous media is modified to allow the use of different horizontal transverse and vertical dispersivities (U.S. EPA 1996a). The dispersion coefficients are given by

$$
\begin{aligned}
D_{xx} &= \alpha_L \frac{V_x^2}{|V|} + \alpha_T \frac{V_y^2}{|V|} + \alpha_V \frac{V_z^2}{|V|} + \theta D^* \\
D_{yy} &= \alpha_L \frac{V_y^2}{|V|} + \alpha_T \frac{V_x^2}{|V|} + \alpha_V \frac{V_z^2}{|V|} + \theta D^* \\
D_{zz} &= \alpha_L \frac{V_z^2}{|V|} + \alpha_V \frac{V_y^2}{|V|} + \alpha_V \frac{V_x^2}{|V|} + \theta D^* \\
D_{xy} &= D_{yx} = (\alpha_L - \alpha_T) \frac{V_x V_y}{|V|} \\
D_{xz} &= D_{zx} = (\alpha_L - \alpha_V) \frac{V_x V_z}{|V|} \\
D_{yz} &= D_{zy} = (\alpha_L - \alpha_V) \frac{V_y V_z}{|V|}
\end{aligned}
\qquad (4\text{-}8)
$$

where α_L, α_T, and α_V are the longitudinal, horizontal transverse, and vertical dispersivity (m), respectively, and D^* is the effective molecular diffusion coefficient (m^2/yr).

The water content, θ, and Darcy velocity V_i, are defined below:

$$\theta = \varphi S_w \qquad (4\text{-}9)$$

$$
\begin{aligned}
V_x &= - k_r K_x \frac{\partial H}{\partial x} \\
V_y &= k_r K_y \frac{\partial H}{\partial y} \\
V_z &= k_r K_z \frac{\partial H}{\partial z}
\end{aligned}
\qquad (4\text{-}10)
$$

where

ϕ = effective porosity
S_W = degree of water saturation.

In the saturated zone, $S_W = 1$. Equation (4-7) is solved separately for the vadose and saturated zones. Details of boundary conditions and solution methods are given in U.S. EPA (1996a).

The retardation factor for each of the member species is given by

$$R = 1 + \frac{P_b \, ds}{\theta \, dC} \tag{4-11}$$

where

ρ_b = bulk density (g/m^3)
S = adsorbed concentration (g/g).

The subscript *l* has been dropped for convenience. Assuming the adsorption isotherm follows the equilibrium Freundlich equation

$$S = k_1 C^{\eta} , \tag{4-12}$$

the retardation coefficient can be written as

$$R = 1 + \frac{\rho_b}{\theta} k_1 \eta C^{\eta - 1} . \tag{4-13}$$

The coefficient Q is given by

$$Q = 1 + \frac{\rho_b}{\theta} k_1 \eta C^{\eta - 1}. \tag{4-14}$$

Note that, in general, the retardation factor is a nonlinear function of concentration. The Freundlich isotherm becomes linear when the exponent $\eta = 1$. The Freundlich coefficient, k_1 in this case, is the same as the familiar solid-liquid phase partition coefficient, K_D. When sorption is linear, the coefficients R and Q also become identical. For all the inorganic chemicals reported herein, $\eta = 1$, $\lambda_l = 0$, and $n_c = 1$.

EPACMTP does not account for heterogeneity, preferential pathways such as fractures and macropores, or colloidal transport, which may affect migration of strongly sorbing constituents such as metals.

EPACMTP simulates steady-state flow in both the unsaturated zone and the saturated zone; contaminant transport can be either steady-state or transient. The steady-state modeling option is used for continuous source modeling scenarios; the transient modeling option is used for finite source modeling scenarios. The output from EPACMTP is a prediction of the contaminant

concentration arriving at a downgradient groundwater receptor well. This can be either a steady-state concentration value, corresponding to a continuous source scenario, or a time-dependent concentration, corresponding to a finite source scenario. In the latter case, the model can calculate the peak concentration arriving at the well or a time-averaged concentration corresponding to a specified exposure duration, e.g., a 9-year average residence time. For cases where a noncarcinogenic contaminant was modeled, the finite source option and a 9-year averaging time for groundwater concentration were used in the risk assessment. Where carcinogens were modeled (such as arsenic), the averaging time varied depending on the exposure duration used in the exposure assessment. Thus, the groundwater averaging time in the modeling was chosen to correspond to the exposure duration used in the subsequent exposure and risk assessment. For instance, for the Monte Carlo analysis for noncarcinogens, the highest 9-year average groundwater concentration that occurs during the 10,000-year modeling period was reported as the model output, and the risk assessment used 9 years as the exposure duration when making the risk calculations. For carcinogens, the groundwater averaging time and exposure duration are assumed to follow a prespecified probability distribution instead of being input as constant values. For each given realization, however, the groundwater averaging time and exposure duration are identical.

For the probabilistic analysis, 10,000 realizations were conducted for each modeling scenario, with the inputs specified as constant values, derived values, or statistical or empirical distribution of values. Each realization comprises a complete and distinct set of model input parameters and the flow and transport solution derived from those inputs. The input parameters for each realization are chosen by EPACMTP from the user-specified values or distributions based on a sequence of randomly generated numbers as specified in Section 4.3.1.3. For the deterministic analysis, one realization is conducted for each central tendency or high-end modeling scenario and all inputs are specified as constant or derived values.

The Monte Carlo groundwater pathway analysis was performed with 10,000 realizations based on the results of a previous bootstrap analysis to maintain consistency with previous listing projects, such as the Petroleum Refining and Lead Lased Paint Analyses. Bootstrap analysis is a technique of replicated resampling (usually by a computer) of an original data set for estimating standard errors, biases, confidence intervals, or other measures of statistical accuracy. It can automatically produce accuracy estimates in almost any situation without requiring subjective statistical assumptions about the original distribution.

In this case, the bootstrap analysis upon which this decision was based was documented in *EPACMTP Sensitivity Analyses* (U.S. EPA, 1996d). This report presents a bootstrap analysis conducted in response to public comments regarding the number of realizations used for the 1995 proposed Hazardous Waste Identification Rule (HWIR). In using a Monte Carlo modeling approach, a higher number of realizations usually leads to a more convergent and more accurate result. However, it is not generally possible to determine beforehand how many realizations are needed to achieve a specified degree of convergence since the value can be highly dependent on parameter distributions. Therefore, EPA conducted a bootstrap analysis for the EPACMTP model to evaluate how convergence improves with increasing numbers of realizations. The analysis was based on a continuous source, landfill disposal scenario in which the 90th percentile dilution-attenuation factor (DAF) was 10. The bootstrap analysis results suggested that, with

10,000 realizations, the expected value of the 90th percentile DAF was 10 with a 95 percent confidence interval of 10 ± 0.7. Decreasing the number of realizations to 5,000 increased the confidence interval to 10 ± 1.0. Because the parameter distributions used in the analyses for HWIR and this Listing Determination were similar, the HWIR-related bootstrap analysis results were considered applicable to the Monte Carlo analysis for this Listing Determination.

A decision was made to use 10,000 realizations for the probabilistic groundwater pathway analysis for this Listing Determination to balance the desire for optimal convergence against other inherent sources of uncertainty associated with data sources and conservative assumptions embodied in EPA's exposure and risk analysis. The fluctuation range of ± 0.7 was considered relatively minor when considering potential effects due to uncertainties associated with data and assumptions. Additionally, conservative values used in the analysis (such as constituents with the lowest sorption coefficients and degradation rates as surrogates for the group of constituents) balanced any potential lack of conservatism resulting from limiting the analysis to 10,000 realizations.

Source Terms and Release Mechanisms. The release of contaminants into the subsurface constitutes the source term for the groundwater fate and transport model. Because the modeled subsurface fate and transport processes are the same for each waste management scenario, the conceptual differences between different waste management scenarios are reflected solely in how the model source term is characterized. The contaminant source term for the EPACMTP fate and transport model is defined in terms of four primary parameters: (1) area of the waste unit, (2) leachate flux rate emanating from the waste unit (infiltration rate), (3) constituent-specific leachate concentration, and (4) duration of the constituent leaching. Leachate flux rate and leaching duration are determined as a function of both the design and operational characteristics of the waste management unit and the waste stream characteristics (waste quantities and waste constituent concentrations).

Incorporation of Finite Source Methodology In EPACMTP. The finite source methodology required the following modifications to the original infinite source, steady-state transport scenario:

- A transient source term was used with time-varying leachate concentration, C_L.
- Transient transport was simulated with dynamic linking between the unsaturated zone and saturated zone modules.
- A time-averaged receptor well concentration, $\bar{C}_{RW}$ was calculated.
- To satisfy regulatory criteria, C_w^0 versus C_L^0 (initial waste and leachate concentrations, respectively) was determined.

Transient Source Term. This section describes the transient source term in EPACMTP for the case of a landfill waste management scenario.

The fate and transport model requires that the leachate concentration be input as a function of time, $C_L(t)$. The leachate concentration $C_L(t)$ used in the model directly represents the concentration of the leachate released from the base of the waste management unit as a boundary condition for the fate and transport model. EPACMTP accounts for the time variation as a constant concentration pulse condition or as an exponentially decreasing leachate concentration. EPACMTP does not attempt to account explicitly for the multitude of physical and biochemical processes inside the waste unit that may control the release of waste constituents.

Figure 4-3 presents a conceptual view of leaching processes in a waste unit and the impact of a number of processes on the leachate concentration as a function of time (U.S. EPA, 1990a). This figure illustrates the complex interactions that may control contaminant leaching. A simplified approach is incorporated into EPACMTP because leaching cannot be quantified accurately as a function of varying chemical and waste matrix properties (U.S. EPA, 1990a). The intended use of EPACMTP is for generic application to a wide range of site conditions and chemical constituents.

In the simplest and most conservative case, the leachate concentration remains constant until all of the contaminant mass has leached out of the disposal unit. This case is referred to as the nondepleting source scenario. The boundary condition for the fate and transport model then becomes a constant concentration pulse described by the following relationship:

$$A_w \, d \, F_h \, C_w^0 \, \rho_{hw} = I \, A_w \, C_L^0 \, t_p \tag{4-15}$$

from which the pulse duration t_p derived to be

$$t_p = \frac{d \, F_h \, \rho_{hw}}{I} \frac{C_w^0}{C_L^0}. \tag{4-16}$$

Use of equation 4-16 implies that there is no degradation of the constituent inside the waste unit and no losses by other mechanicisms such as volatilization other than leaching. In a Monte Carlo simulation, the source parameters, waste unit depth (d) volume fraction of the unit containing the waste (F_h), the density of the waste (ρ_{hw}), and the infiltration rate through the waste site, (I), are treated as random variables with a specified probability distribution. The initial waste concentration, C_w^0, and initial leachate concentration, C_{LL}^0, are waste-specific parameters with a fixed value of C_w^0/C_L^0 assigned for each Monte Carlo simulation run. The pulse duration is, therefore, completely defined.

The second source scenario handled by EPACMTP is that in which the leachate concentration decreases exponentially with time as a result of depletion of the source. Consider the general mass balance expression for the waste unit source term:

$$A_w \, d \, F_h \, \rho_{hw} \, (C_w^0 - C_w(t)) = A_w \, I \int_0^t C_L(t) \, dt \tag{4-17}$$

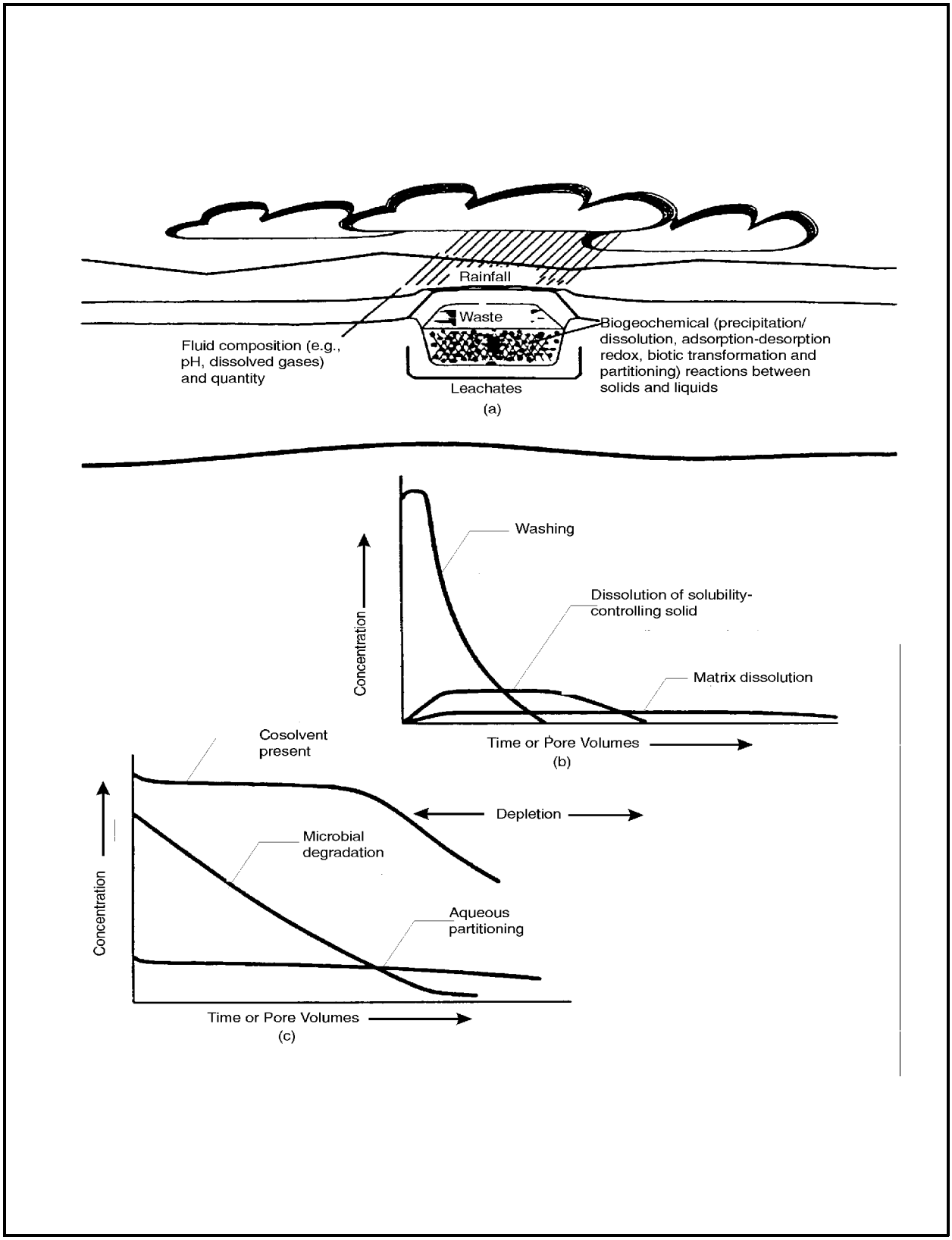

Figure 4-3. **Conceptual view of leaching in a waste unit (U.S. EPA, 1990): (a) generation of leachates, (b) potential leaching stages for inorganic contaminants, (c) generation of organic leachates.**

where $C_w(t)$ is the waste concentration at time t, and the remaining terms are as defined previously. Equation 4-17 can be written alternatively as

$$A_w\, d\, F_h\, \rho_{hw} \frac{dC_w}{dt} = -A_w\, I\, C_L(t). \tag{4-18}$$

To solve Equation 4-18, assume that the total waste concentration, C_w, consists of a contribution from the solid phase of the waste, C_s, and a contribution from the liquid phase of the waste. The concentration in the liquid phase is taken to be the same as the leachate concentration, C_L, and is related to the solid phase concentration through a linear equilibrium partitioning process:

$$C_w = C_s + \frac{\theta_w}{\rho_{hw}} C_L \tag{4-19}$$

$$= kC_L + \frac{\theta_w}{\rho_{hw}} C_L \tag{4-20}$$

or

$$\frac{C_w}{C_L} = k + \frac{\theta_w}{\rho_{hw}} \tag{4-21}$$

where k is the partition coefficient and θ_w is the volumetric water content of the waste. The factor θ_w/ρ_{hw} converts the liquid phase concentration to a mass basis. bBecause of the unit conversion involved, the numerical value of C_w/C_L can be less than 1.0. Using Equation 4-21 in 4-18 and rearranging yields

$$\frac{dC_L}{dt} = \frac{-I}{d\, F_h\, \rho_{hw} \left(k + \theta_w/\rho_{hw}\right)} C_L \tag{4-22a}$$

which can be rewritten in the form:

$$\frac{dC_L}{dt} = \frac{-I}{dF_h\rho_{hw}\ (C_w^0/C_L^0)} C_L \tag{4-22b}$$

where C_w^0 is the initial total waste concentration, and C_L^0 is the initial leachate concentration. Integration of Equation 4-22b gives

$$C_L(t) = C_L^0\ e^{-\lambda' t} \tag{4-23}$$

in which

$$\lambda' = \frac{I}{d\, F_h\, \rho_{hw} (C_W^0 / C_L^0)} \qquad (4\text{-}24)$$

and

$$C_L(t=0) = C_L^0 \ . \qquad (4\text{-}25)$$

The case of linear equilibrium partitioning between the solid and liquid phase of the waste leads to an exponential decrease in the source leachate concentration with an apparent first-order rate coefficient given by Equation 4-24. It is assumed that the constituent does not degrade inside the waste unit, nor is it removed by processes (e.g., volatilization) other than leaching.

For the EPACMTP modeling conducted for this Listing Determination, the nondepleting source option of a constant concentration pulse release (Equation 4-16) is used for surface impoundment scenarios. The depleting source option (Equation 4-23) is used for landfill scenarios. For the landfill scenario, the initial leachate concentration and the waste concentration are given by analytical data. For the surface impoundment scenario, the wastewater concentration is also provided by analytical data. For the depleting source scenario, the value of the aqueous phase partition coefficient, *k*, does not need to be known; it is implicitly given by the ratio of the initial waste concentration to initial leaching concentration.

For this analysis, two landfill scenarios were modeled: a municipal and an industrial Subtitle D (nonhazardous) landfill. Both of these types of landfills were assumed to have a final earthen cover but no liner or leachate collection system. The leachate flux through the landfill was the result of infiltration of ambient precipitation through the landfill cover. Infiltration rates used for landfills in the analysis were determined using the HELP model (U.S. EPA, 1997a) as described in Section 4.3.2.1. For this assessment, it was assumed that the landfill had a 30-year operational life, the average active lifetime of municipal Subtitle D landfills (U.S. EPA, 1988) and that leaching was the only loss mechanism. For the landfill scenarios, the depleting source option was used. In this case, linear equilibrium partitioning between the solid and liquid phases of the waste, and depletion of the source lead to an exponential decrease in the leachate concentration with time. The initial leachate concentrations used in the modeling analysis were sampled Toxicity Characteristic Leaching Procedure (TCLP) concentrations for the municipal landfill disposal scenario and sampled Synthetic Precipitation Leaching Procedure (SPLP) concentrations for the industrial landfill disposal scenario. To ensure that the infiltration rate through the landfill was not unrealistically high, the maximum rate was set at the saturated hydraulic conductivity of the vadose zone. This constraint does not always guarantee that the groundwater mound created by infiltration from the landfill does not rise above the bottom elevation of the landfill. The issue of how potential groundwater mounding in the landfill scenario may affect modeled downgradient groundwater concentrations has not been fully investigated. The issue of groundwater mounding beneath waste management units may be of concern because groundwater mounding could violate the assumption of a uniformly thick aquifer and could lead to an unnaturally increased hydraulic gradient and accelerated downgradient transport. In EPACMTP, a check is implemented to

minimize groundwater mounding by limiting the maximum infiltration rate to the saturated conductivity in the vadose zone.

For the surface impoundment scenario, the modeled unit was assumed to be an unlined impoundment. Although the infiltration rate can be computed as a derived parameter within the EPACMTP model, this option requires site-specific data on the thickness and conductivity of the sludge at the base of the impoundment. Because these data are rarely available, the infiltration rate calculation algorithm in the surface impoundment source model developed for the HWIR99 project was used to develop a simplified program that estimates infiltration rates for the surface impoundment scenario. This algorithm has two advantages over the methodology used in EPACMTP; specifically, the algorithm ensures that the infiltration rate is capped so that: (1) if a groundwater mound is created, it does not rise to the bottom elevation of the surface impoundment; and (2) if the saturated hydraulic conductivity of the soil is high, the infiltration rate does not exceed 99 percent of the impoundment influent flow rate. The surface impoundments modeled in this analysis were for wastewater treatment rather than for disposal impoundments and the infiltration rate is expected to be only a small fraction of the wastewater throughput. This surface impoundment source model is described more fully in Section 4.3.2.2.

Location and Time of Exposure. The selected receptors for the groundwater pathway were hypothetical adult and child residents who obtained drinking water from a groundwater well. The exposure point was a drinking water well located downgradient of a waste management unit containing inorganic chemical manufacturing wastes. The location of the receptor well is described more fully in Section 4.3.3.2 for offsite waste management facilities and Section 4.3.3.3 for onsite waste management facilities.

4.3.1.2 Description of Required Code Modifications. For the Inorganics Listing Determination, modifications were made to EPACMTP (version 1.2.1) to facilitate the groundwater analysis.

In addition to the main input data file, an extra input file may be specified in EPACMTP version 1.2.1, also referred to as the source data file. The source data file contains values of parameters whose distribution types are set to “88” in the main input data file. The source data file permits output from SWMU source models (e.g., the surface impoundment source model) to be used as input to EPACMTP and provides the means to correlate parameters, such as leachate concentration, infiltration rate, and soil and aquifer type, to facility location. The data output routines of the surface impoundment source model were modified to generate an output file that provides not only the infiltration rate output for each iteration, but also the corresponding input values of modeling parameters that are common to both the source model and EPACMTP.

EPACMTP version 1.2.1 was limited in its capabilities in the following areas, necessitating several modifications to perform the required analysis for the Inorganics Listing Determination risk assessment:

- **#** Records in the source data file could only be sampled randomly with replacement; that is, the records could be chosen more than once if the number of Monte Carlo realizations exceeded the number of records in the source data file; EPACMTP could

not sequentially read each record in the source data file to keep it coordinated with the source model on an iteration-by-iteration basis. This capability was needed to use the same sequence of values for the groundwater averaging time in the modeling and the exposure duration in the risk assessment.

- In a Monte Carlo analysis, the soil parameter values (and infiltration and recharge rates, which are determined in part by soil type) could not be specified in the source data file because the soil type was automatically varied among the three soil types.

The following code changes were made to EPACMTP version 1.2.1 to overcome the limitations listed above:

- Records in the source data file can now be sampled sequentially or randomly with replacement.

- Soil types, soil parameter values, and infiltration and recharge rates can now be specified in the source data file.

4.3.1.3 Monte Carlo Analysis. Application of the EPACMTP model requires input values for the source-specific, chemical-specific, unsaturated zone-specific, and saturated zone-specific model parameters. Each of these input parameters can be represented by a probability distribution reflecting the range of variation that may be encountered at the modeled waste site(s). The fate and transport simulation modules in EPACMTP are linked to a Monte Carlo module to allow quantitative estimation of the uncertainty in the downgradient receptor well concentration due to uncertainty and variability in the model input parameters.

Following is a brief description of the general Monte Carlo methodology used in EPACMTP. Additional information about Monte Carlo modeling using EPACMTP can be found in EPACMTP documents (U.S. EPA, 1996a, 1996b, and 1997a).

The Monte Carlo option in EPACMTP is based on the module incorporated in EPA's Composite Model for Landfills (EPACML) (U.S. EPA, 1990). This module has been enhanced in three ways: (1) to account more directly for dependencies between various model parameters by using data from actual waste sites across the United States; (2) to include a site-based methodology to directly associate the appropriate regional climatic and hydrogeologic conditions to the location of a waste site, and (3) to account for statistical correlations between two or more model parameters (e.g., hydraulic conductivity and gradient) when missing parameter values are generated.

To run EPACMTP in Monte Carlo mode, a probability distribution must be provided for each input parameter (except constant or derived parameters). The Monte Carlo methodology is then performed as follows:

1. Read main input data file.

2. For those parameters whose probability distribution types are "88," read the corresponding parameter values for all the Monte Carlo records from the source data file and store them in a variable array, indexed by the Monte Carlo record number.

3. Set Monte Carlo run number irun = 1.

4. For those parameters whose probability distribution types are other than "88" and "-1" (derived type), generate a realization of the parameters given the distribution types and the allowable lower and upper bounds.

5. For those parameters whose probability distribution types are "88," extract from the saved array the corresponding values for the Monte Carlo record number = irun (sequential sampling).

6. For those parameters whose probability distribution types are "-1" (to be derived), derive the parameter values.

7. Perform the contaminant fate and transport simulation. The result is given in terms of the predicted contaminant concentrations in a downgradient receptor well.

8. Increment irun and repeat steps 4 through 7 as many times as desired.

9. Statistically analyze model output to yield the cumulative probability distribution of the resulting groundwater concentrations. This distribution of values is then used as input to the probabilistic risk assessment.

The EPACMTP input parameters considered in the groundwater Monte Carlo modeling are presented in Table 4-9. For the surface impoundment source model, the depth of the sludge layer was varied as described in Section 3.1.3.3; the ponding depth was set to a constant value based on facility information; and the hydraulic conductivity of the sediment layer at the base of impoundment and the underlying unsaturated zone were derived as described in Section 4.3.2.2 and Appendix I. This model was used to generate the infiltration rate input to EPACMTP for the surface impoundment scenario.

Modified Regional Site-Based Methodology. The regional site-based approach offers several advantages over a strictly nationwide methodology. This methodology relies on data compiled at actual waste sites around the country, which can be linked to databases of climatic and hydrogeologic parameters through the use of climate and hydrogeologic indices. Thus, the regional site-based approach attempts to approximate the ideal situation where a complete set of the required site-specific values is available for each Monte Carlo realization without requiring the extensive sampling that would be required to actually gather these data.

The specific methodology for data gathering employed for this risk assessment can be summarized as follows:

Table 4-9. EPACMTP Input Parameters for Monte Carlo Modeling

Waste Management Scenario/Parameter	Input Data Source
Landfill scenario	
WMU area (m^2)	
Municipal landfill	EPA Municipal Landfill Survey
Onsite industrial landfill	Site-specific
Offsite industrial landfill	EPA Municipal Landfill Survey
Initial leachate concentration (mg/L)	TCLP or SPLP concentration from site specific analytical data
Regional recharge rate (m/yr)	Location-specific (HELP modeled rates)
Infiltration rate (m/yr)	Location-specific (HELP modeled rates)
Pulse duration (yr)	Derived from waste and leachate concentrations, WMU parameter values, and infiltration rate
Surface impoundment scenario	
WMU area (m^2)	Site-specific
Leachate concentration	SPLP filtrate or total wastewater concentration(s)
Regional recharge rate (m/yr)	Location-specific
Infiltration rate (m/yr)	Derived using SI source model
Pulse duration (yr)	50 years
Chemical-specific parameters	
Organics	
Hydrolysis rate (yr^{-1})	Constituent-specific (Kollig, 1993)
K_{OC} (L/kg)	Constituent-specific (Kollig, 1993)
Inorganics	
K_d (L/kg)	Empirical or statistical distribution of values from the scientific literature
Both organics and inorganics	
Exposure duration (yr)	9 for noncarcinogens; distribution of values for carcinogens
Unsaturated zone parameters	
Sat. hydraulic cond (cm/h)	Distribution based on soil type (Carsel and Parrish, 1988)
Hydraulic parameter, α (cm^{-1})	Distribution based on soil type (Carsel and Parrish, 1988)
Hydraulic parameter, β	Distribution based on soil type (Carsel and Parrish, 1988)
Residual water content	Distribution based on soil type (Carsel and Parrish, 1988)
Saturated water content	Distribution based on soil type (Carsel and Parrish, 1988)
Depth to groundwater (m)	Distribution based on HG region [a] (Newell et al., 1989)
Organic matter content (%)	Distribution based on soil type (Carsel et al., 1988)
Bulk density (g/cm^3)	Distribution based on soil type (Carsel et al., 1988)

(continued)

Table 4-9. (continued)

Waste Management Scenario/Parameter	Input Data Source
Saturated zone parameters	
Particle diameter (cm)	National distribution (U.S. EPA, 1997a)
Saturated thickness (m)	Distribution based on HG region [a] (Newell et al., 1989)
Hydraulic conductivity (m/yr)	Distribution based on HG region [a] (Newell et al., 1989)
Hydraulic gradient (m/m)	Distribution based on HG region [a] (Newell et al., 1989)
Longitudinal dispersivity (α_L)	Derived from distance to well (Gelhar et al., 1992; U.S.EPA, 1997a)
Transverse dispersivity (α_T)	Derived from distance to well (Gelhar et al., 1992; U.S.EPA, 1997a)
Vertical dispersivity (α_V)	Location-specific
Groundwater temperature (°C)	Value based on soil type
Groundwater pH	National distribution (U.S. EPA, 1997a)
Fraction organic carbon	
Receptor well location	
X-well distance (m)	National distribution [a]
Y-well location (m)	Random inside plume
Z-well depth (m)	Random, 0-10 m below water table

[a] HG is the HydroGeologic database for modeling (Newall et al., 1989; U.S. EPA, 1997a). Distribution may also be based on site-specific data for onsite SWMUs.

- For sites where adequate site-specific data on soil and aquifer parameters are not available: (1) the site's geographic location is correlated with available Geographic Information System (GIS) data and aquifer maps to classify the underlying aquifer as 1 of 13 types and to classify the soil as 1 of 3 types; (2) the site's geographic location is used to place the site within 1 of 97 climatic regions in the continental United States; and (3) the hydrogeologic and climatic indices are then used to define the site-specific distributions of hydrogeologic and climatic parameter values, respectively.

- For sites where adequate site-specific data on soil and aquifer parameters are available: (1) site-specific data are used to define the soil type(s) and values (or distribution of values) for aquifer parameters; and (2) the site's geographic location is used to place the site within 1 of 97 climatic regions in the continental United States,and this climatic index and the soil type(s) present at the site are then used to define the site-specific infiltration and recharge rates.

Once the soil type is defined for a facility, the values for each of the soil parameters are randomly chosen from a distribution of values appropriate for that soil type (Carsel et al., 1988). These distributions are specified within the EPACMTP code, as described in U.S. EPA (1997a).

Data sources for the modified regional site-based methodology that were used to conduct this analysis include: (1) the infiltration and recharge analysis performed for 97 U.S. climatic centers using the HELP model (U.S. EPA, 1997a); (2) the USGS inventory of the groundwater resources of each state (USGS, 1985); and (3) the HydroGeologic DataBase for Modeling (HGDB) (Newell et al., 1989; U.S. EPA, 1997a), developed from a survey of hydrogeologic parameters for actual hazardous waste sites in the United States.

For this listing determination, facility-specific values for SWMU location and waste, soil, and aquifer characteristics were used to the extent possible for all on-site SWMUs. Where site-specific data were not available for on-site SWMUs and for all off-site SWMUs, the following parameters were obtained from the HGDB database (Newell et al., 1989; U.S. EPA, 1997a):

- Depth to groundwater (m)
- Aquifer thickness (m)
- Hydraulic conductivity (m/yr)
- Hydraulic gradient (m/m).

For all onsite and offsite landfills modeled for this listing determination, except for the U.S. Antimony onsite landfill, facility-specific values for the landfill infiltration rate were obtained by determining the soil type(s) present at the site from GIS data sources and then finding the nearest HELP climate center. The infiltration rate for each soil type was then read from the database of HELP-modeled infiltration rates, as described in Section 4.3.2.1. The ambient recharge rate was then set equal to the chosen landfill infiltration rate for a given soil type. For the onsite landfill at U.S. Antimony, precipitation data from the operating permit were used to calculate the infiltration rate. For surface impoundments, the infiltration rate was calculated using the surface impoundment source model, as described in Section 4.3.2.2; the ambient recharge rate was set equal to the HELP model recharge rate for the nearest climate center.

For onsite facilities without adequate site-specific data and for all offsite SWMUs, the USGS inventory of state groundwater resource maps (USGS, 1985) and available GIS data were used to identify the predominant hydrogeologic environment (or aquifer type) underlying each SWMU to be modeled. Once the aquifer type was determined, the HGDB was then used to specify the probability distribution for each of the groundwater parameters. The HGDB provides data on depth to groundwater, aquifer thickness, hydraulic gradient, hydraulic conductivity, and hydrogeologic classification for approximately 400 hazardous waste sites nationwide. These site-specific data were then regrouped according to hydrogeologic classification, and 13 aquifer types were classified (12 specific environments and one category called “other”). Each aquifer type consists of a distribution of values for each of the four aquifer parameters.

For this analysis, each site to be modeled was located on the appropriate state groundwater map from USGS (1985), and available GIS data were compiled and evaluated. Then the primary aquifer type for that location was classified according to the 13 aquifer types. If more than one aquifer type was present within a 100-mile radius of the facility, each aquifer type chosen was assigned a percentage roughly corresponding to the areal extent of that aquifer type. The aquifer types and the parameter values for each are provided in the *EPACMTP User's Guide* (U.S. EPA, 1997a).

4.3.1.4 Sensitivity Analysis. A sensitivity analysis was performed on the Monte Carlo results to identify the most sensitive parameters to vary during the high-end deterministic model runs. A statistical regression analysis of the inputs and outputs of the probabilistic analysis was used to identify the contribution of each variable parameter to increased risk. This methodology provided insight into the interactions of parameters within the nonlinear groundwater model, EPACMTP. The sensitivity analysis included all direct inputs to the groundwater modeling and the risk equations and all intermediate inputs calculated within EPACMTP. This comprehensive evaluation of parameters provided insight into the analysis and highlighted the importance of parameters that had not previously been addressed; however, it did not address the importance of parameters that were constant in this analysis, for example, waste quantity. A sensitivity analysis was performed for every constituent of every waste stream and every waste management scenario. The results of the sensitivity analysis were used to identify the parameters to be set to high end for the deterministic analysis. The statistical methodology for this analysis is described in detail in Appendix G. The results of all of the sensitivity analyses performed are shown in Appendices A through E.

4.3.1.5 Deterministic Analysis. A deterministic analysis was also performed to make point estimates of a central tendency risk and a two-parameter high-end risk.

To estimate central tendency risk, all parameters in the analysis (groundwater inputs and inputs to the risk equation) were set at central tendency values; then the groundwater model (EPACMTP) was run in Monte Carlo mode with the number of realizations being set to 1. The modeling results were then used to calculate the resulting risk. For parameters whose values were not derived (e.g., from an empirical distribution, statistical distribution, or as a constant value), the central tendency values were specified as the median values from their respective probability distributions. Those parameters derived in the Monte Carlo analysis were also derived in the deterministic analysis. An example of such a derived variable is the aquifer seepage velocity, which was calculated using aquifer hydraulic conductivity along the flow direction and aquifer hydraulic gradient. In this example, the hydraulic conductivity and gradient were set to their respective median values, and the seepage velocity was derived from the hydraulic conductivity and gradient.

The goal of the two-parameter high-end analysis was to determine the groundwater concentration to be used to estimate the high-end risk. In the two-parameter high-end analysis, all input parameters were set to their central tendency values with the exception of the two most sensitive parameters identified in the sensitivity analysis, which were set to their high-end values. The high-end value is specified as the 90^{th} percentile if the input is positively correlated with the resulting groundwater concentration or the 10^{th} percentile value if it is negatively correlated with concentration. That is, since a higher infiltration rate generally leads to a higher groundwater concentration, the high-end value for infiltration rate would be the 90^{th} percentile value from the input distribution. However, for metals, a lower value for the partition coefficient K_d would generally result in a higher groundwater concentration, so the high-end value for K_d would be the 10^{th} percentile value from the input distribution.

The central tendency and high-end parameter values that were used in this analysis are presented in Section 4.3.6.

4.3.2 Modeling of Infiltration and Recharge Rates

EPACMTP requires inputs for both infiltration and recharge rates. Infiltration is defined as water percolating through an SWMU (in this case, landfill or surface impoundment) to the underlying soil, while recharge is water percolating through the soil to the aquifer outside of the SWMU. For recharge and infiltration through landfills, EPACMTP uses estimates from the HELP model, a hydrologic model for conducting water balance analysis of landfills, cover systems, and soil systems (U.S. EPA, 1994a, b). In the context of EPACMTP, HELP has been run for three soil textures (sandy loam, silt loam, silty clay loam) and 97 climatic centers across the country to represent nationwide variability in soil properties, cover characteristics, and climatic data (e.g., precipitation and evapotranspiration) that affect recharge and infiltration rates. For this risk assessment, landfill infiltration and recharge rates were selected from this set of data to represent site conditions of each inorganic chemical waste disposal site. Section 4.3.2.1 describes this selection process and the HELP modeling in greater detail.

The HELP results are not appropriate for surface impoundments because infiltration from an impoundment is mainly driven by the wastewater hydraulic head rather than precipitation. To estimate surface impoundment infiltration rates, infiltration algorithms were extracted from the surface impoundment model used in the HWIR 3MRA model (U.S. EPA, 1999a). As described in Section 4.3.2.2, this model uses data on impoundment design and operation in conjunction with underlying soil properties and aquifer characteristics to estimate surface impoundment infiltration rates.

4.3.2.1 Hydrologic Evaluation of Landfill Performance (HELP) Model. For the landfill scenario, the leachate flux results from infiltration of ambient precipitation through the landfill cover. Infiltration rates used for all landfills other than the onsite landfill at U.S. Antimony, were taken from the results of an existing HELP modeling analysis (U.S. EPA, 1997a). The HELP model calculates the net infiltration rate using a water balance approach, which includes precipitation, evapotranspiration, and surface runoff, among other factors. For the landfill at US Antimony, the infiltration rate was calculated from precipitation data included in the facility operating permit. For all landfills, the recharge rate was assumed to equal the infiltration rate.

The landfill scenario simulated with the HELP model was a representative Subtitle D landfill with a 2-foot earthen cover, using climatic data from 97 climatic stations located throughout the United States (U.S. EPA, 1996a,b). The modeling included separate runs for sandy loam, silt loam, and silty clay loam soils. These three soils were used in EPACMTP to represent the three main categories of soil texture (coarse, medium, and fine soil textures) based on the Soil Conservation Service (SCS) soil mapping database and the U.S. Department of Agriculture's definitions of soil texture. Using NOAA data on precipitation and evaporation rates, 97 cities from the contiguous 48 states were selected as climatic centers.

The WMU-specific infiltration and recharge rates were then generated for each soil type at each climate center using the HELP model (U.S. EPA, 1997). The result of this modeling effort was a database of infiltration and recharge rates for each of three soil types at each of 97 climate centers. (Four types of SWMUs were included in the original HELP modeling analysis, but only

the landfill infiltration rates were used for this listing determination.) Table 4-10 presents the landfill infiltration and recharge rates used in this analysis. Because the modeled landfill was assumed to be unlined, the landfill infiltration and recharge rates for a given facility were assumed to differ only if the soil type of the final landfill cover was different from the naturally occurring soil type in the vicinity of the landfill. Results from previous sensitivity analyses showed that recharge rate was not a sensitive input parameter, so the recharge rate was set equal to the infiltration rate for the soil type and landfill scenario.

4.3.2.2 SI Infiltration Model. Surface impoundments are onsite SWMUs in the inorganics chemical manufacturing industry. Modeling these surface impoundments requires calculating an infiltration rate from the impoundment. For this analysis, infiltration was estimated using a portion of the SI source model developed for the HWIR analysis (U.S. EPA, 1999a). Appendix H describes in detail the algorithms, assumptions, and inputs used in this model, which calculates the infiltration rate through the accumulated sediment at the bottom of the impoundment. These algorithms allow the sediment layer to change over time and also limit infiltration through natural processes such as clogging of the native soil materials underlying the impoundment or mounding due to flow-limiting aquifer characteristics. No lines or leachate collection system is assumed to exist beneath the unit. The modeled processes limit infiltration as follows:

- **Effective hydraulic conductivity of consolidatable sediment layer.** As sediment accumulates at the base of the impoundment, the weight of the liquid and upper sediments tends to compress (or consolidate) the lower sediments. This consolidated sediment acts as a filter cake, and its hydraulic conductivity may be much lower than the nonconsolidated sediment.

- **Effective hydraulic conductivity of clogged native material.** As liquids infiltrate soils underlying the impoundment, suspended particulate matter accumulates in the soil pore spaces, reducing hydraulic conductivity and lowering infiltration rates.

- **Limitations on maximum infiltration rate from mounding.** If the calculated infiltration rate exceeds the rate at which the saturated zone can transport the groundwater, the groundwater level will rise into the unsaturated zone and the assumption of zero pressure head at the base of the unsaturated zone is violated. This groundwater "mounding" will reduce the effective infiltration rate so that the maximum infiltration rate is estimated as the rate that does not cause the groundwater mound to rise to the bottom elevation of the SI unit.

- **Limitations on maximum infiltration rate by SI influent rate.** Under certain conditions of high soil-saturated hydraulic conductivity and long residence time in the SI, it is possible that the modeled infiltration rate may exceed the SI influent flow rate. To avoid mass balance violations in these cases, the modeled infiltration rate is set equal to 99 percent of the influent flow rate. This limitation was never invoked in this analysis.

Table 4-10. Landfill Infiltration Rates

Sector	Waste Stream	Facility Name and LF Location	Landfill Infiltration Rate (m/yr)		
			Sandy Loam	Silt Loam	Silt Clay Loam
HCN	Filter residues	Rohm and Haas Anahuac, TX	0.4641	0.3647	0.2817
		Du Pont Memphis Millington, TN	0.4336	0.3531	0.2824
	Feed gas filters	Du Pont Memphis Millington, TN	0.4336	0.3531	0.2824
$NaClO_3$	Sludge w/o Cr	Huron Tech Elgin, SC	0.3287	0.2609	0.2123
		Eka Starkville, MS	0.4336	0.3531	0.2824
		Eka Ephrata, WA	0.0023	NA	0.0003
	Filter wastes w/o Cr	Huron Tech Elgin, SC	0.3287	0.2609	0.2123
		Eka Ephrata, WA	0.0023	NA	0.0003
		442 Corporation Purdue Hill, AL	0.3993	0.3416	0.2822
$NaPO_4$	Filter press cakes	Rhodia-Chicago Heights Chicago Heights, IL	0.1138	0.0798	0.0620
	Filter bags	Chicago, IL	0.1138	0.0798	0.0620
		East St. Louis, IL	0.1676	0.1435	0.0704
		Augusta, GA	0.3993	0.3416	0.2822
Sb_2O_3	Slag	US Antimony Thompson Falls, MT	0.0132	0.0069	NA
TiO_2	Sulfate digestion sludge	Millennium HPP Baltimore, MD	0.2609	0.2007	0.1641
	Sulfate secondary gypsum	Millennium HPP Baltimore, MD	0.2609	0.2007	0.1641
	Cl/SO_4 milling sand	Kemira Savannah, GA	0.3287	0.2609	0.2123
	Off-spec TiO^2	Du Pont New Johnsonville New Johnsonville, TN	0.5395	0.4674	0.3769
		Du Pont Delisle Pass Christian, MS	0.7445	0.5893	0.4503
	Cl/SO_4 waste water treatment solids	Millennium HPP Baltimore, MD	0.2609	0.2007	0.1641
	Ilmenite waste water treatment solids	Du Pont Edgemoor Edgemoor, DE	0.2609	0.2007	0.1641
		Du Pont New Johnsonville New Johnsonville, TN	0.5395	0.4674	0.3769

The SI infiltration model simulates these processes using SI input data and various assumptions described in Appendix I. If either of the limitations above is triggered, a flag is set in the model output. The data used to describe a surface impoundment in the evaluation of a WS/WMU combination of the hydrogen cyanide (HCN) sector are presented in Table 4-11 as an example of the data required to estimate infiltration from an onsite surface impoundment. The distribution of infiltration rates for all the surface impoundments evaluated for the inorganic chemical industry are presented in Table 4-12.

Table 4-11. Surface Impoundment Parameters for Commingled Wastewaters, HCN Sector

Parameter	Units	Range of Values
Soil hydraulic conductivity (Ksat)	(m/d)	0.00018 to 7.07
Hydraulic parameter alpha_s	(1/m)	0.2316 to 21.05
Hydraulic parameter beta_s		1.09 to 2.555
Depth of liquid (d_liq1)	(m)	1.829
Depth of sediment (d_sed2)	(m)	0.1524 to 2.5
Particle density (rho_part)	(g/cm^3)	2.5
Depth of vadose zone (d_s)	(m)	1.68 to 2.50
Aquifer thickness (AquThick)	(m)	13.01 to 61
Aquifer hydraulic conductivity (AquSATK)	(m/d)	24.67 to 68.5
Area of surface impoundment (A_tot)	(m^2)	1737
Daily throughput	(m^3/d)	2050.137
Distance to nearest waterbody (R_infinite)	(m)	982

4.3.3 Environmental Data

Environmental data required by EPACMTP include soil properties, aquifer characteristics, and receptor well distance, direction, and depth. The sources for these data are described in Section 4.3.3.1. Data collection methodologies differed for onsite and offsite waste management scenarios.

For wastes managed in offsite municipal or industrial landfills, the analysis assumed that that future waste management would occur within a 100-mile radius around the general location of the current offsite landfills. Soil and aquifer data were collected within these 100-mile radius areas using a GIS, a spatial data management system that can collect mapped environmental data

Table 4-12. Distribution of Infiltration Rates for Surface Impoundments Evaluated in the Inorganic Chemical Manufacturing Waste Listing Determination Risk Assessment

Percentile	Millennium HPP (m/yr)	Kerr McGee (m/yr)	Du Pont Delisle (m/yr)	Degussa Theodore, AL (m/yr)	Du Pont Memphis (m/yr)	Millennium HPP Batch Attack Lagoon (m/yr)	Du Pont New Johnsonville (m/yr)
50th	0.56	0.40	0.92	1.13	0.47	1.02	0.66
75th	0.69	0.53	1.11	1.18	0.63	1.54	0.87
80th	0.73	0.57	1.17	1.20	0.64	1.68	0.95
85th	0.78	0.62	1.24	1.22	0.67	1.83	1.03
90th	0.85	0.70	1.34	1.26	0.71	2.00	1.13
95th	0.97	0.82	1.47	1.31	0.78	2.18	1.25
99th	1.14	1.04	1.64	1.44	0.98	2.38	1.41

from existing nationwide datasets. The GIS was used to determine location-dependent parameters (e.g., soil types and hydrogeologic environments) within the 100-mile radius area of interest. To represent uncertainty in offsite landfill locations, the frequency with which parameters were observed within the area of interest was used as the frequency distribution to develop a series of 10,000 soil and aquifer types for use in the Monte Carlo analysis. Section 4.3.3.2 describes the data collection methodology for offsite landfills.

Onsite Industrial D landfills and surface impoundments were modeled using site-specific data on soils, aquifers, and receptor locations where available. Locations of the SWMUs were obtained from facility documents and confirmed against EPA EnviroFacts locations. EnviroFacts is EPA's Web-based environmental data clearinghouse, which includes a compilation of locations for EPA regulated facilities. Accurate locations enabled GIS data sources to be used to supplement site-specific data that were inadequate for modeling purposes. In this case, parameters were varied during Monte Carlo runs to represent uncertainty in the site-specific data values for each location. Section 4.3.3.3 describes the data collection methodology for onsite landfills and surface impoundments.

4.3.3.1 Data Sources. The primary data source for soil properties is the 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, 1994a). 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 spatial database.) Within each map unit, STATSGO contains data for component soils, smaller soil units that are not mapped in the STATSGO GIS coverage. In addition, CONUS, a compiled subset of STATSGO data keyed to the STATSGO map unit GIS coverage (Miller and White, 1998) also was used in the analysis as a source of predominant soil texture by map unit.

Soil texture and pH were derived directly from STATSGO or CONUS. A complete set of hydrologic soil properties required by EPACMTP 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 contained in EPACMTP. As shown in Table 4-13, sources for these relationships include Carsel and Parrish (1988) and Carsel et al. (1988). These peer-reviewed references provide a consistent set of correlated hydrologic properties for each soil texture or hydrologic group. Implementation of these relationships within EPACMTP is described in U.S. EPA (1997a).

Table 4-13. Soil Parameters Derived from Soil Texture by EPACMTP (U.S. EPA, 1997a)

Soil Parameter	Units	Data Source
Saturated hydraulic conductivity	cm/h	Carsel and Parrish (1988)
Alpha (moisture retention)	1/cm	
Beta (moisture retention)	Unitless	
Residual water content	Unitless	
Saturated water content	Unitless	
Percent organic matter	Unitless	Carsel et al. (1988)
Bulk density	g/cm^3	

Regional aquifer data were obtained from the sources shown in Table 4-14. The primary source for aquifer data is the American Petroleum Institute (API) HGDB (Newell et al., 1989; Newell et al., 1990). EPACMTP uses the HGDB data to specify probability distributions for each of four hydrogeologic parameters shown in Table 4-14. 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 4-15. 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* (EPA, 1997a).[1]

To use the HGDB data in this analysis, HGDB hydrogeologic environments were assigned for the onsite or offsite data collection efforts using a USGS inventory of state groundwater resource maps (USGS, 1985); USGS GIS coverages of Heath hydrogeologic regions, productive aquifers, and surficial geology (Clawges and Price, 1999a-d); and hydrogeologic setting descriptions from Aller et al. (1987).

Average aquifer/vadose zone temperature was obtained from a map of groundwater temperatures for the continental United States in the *Water Encyclopedia* (Van der Leeden et al., 1990).

[1] 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 analysis.

Table 4-14. Aquifer Parameters Derived by EPACMTP (U.S. EPA, 1997a and 1997b)

Parameter	Units	Data Source
Saturated zone thickness	m	API Hydrogeologic Database (Newell et al., 1989 and 1990); based on hydrogeologic environment
Hydraulic conductivity	m/yr	
Hydraulic gradient	unitless	
Depth to groundwater	m	
Particle diameter	cm	Shea (1974)

Table 4-15. HGDB Hydrogeologic Environments in EPACMTP

Code	Description
01	Metamorphic & 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)

Finally, facility-specific documents and reports available through the USGS literature were used to collect or verify soil properties, aquifer data, and receptor well locations for the site-specific modeling of onsite waste management operations. Table 4-16 provides these references for each facility modeled with onsite SWMUs.

4.3.3.2 Data Collection Methodology for Offsite SWMUs. Soil properties and aquifer characteristics data were collected regionally for offsite SWMUs. Distributions of model input variables were developed to represent the variability of the soil and aquifer data within a 100-mile-radius area of each offsite SWMU. This radius area represents the uncertainty associated in the future locations of offsite waste management facilities.

Table 4-16. Site-Specific Documents Reviewed for Onsite Landfills and Surface Impoundments

Facility	Report
Kerr-McGee Chemical, LLC Hamilton, MS	RCRA Confirmatory Sampling Report, Kerr-McGee Chemical LLC, Hamilton, MS. Prepared by Roy F. Weston, Inc., for Kerr-McGee. July 1998.
	RCRA Facility Assessment of Kerr-McGee Chemical Corporation, Hamilton, MS. Prepared by A.T. Kearney, Inc., for EPA Region IV. June 1995.
Millennium Hawkins Point Plant Baltimore, MD	RCRA Operation & Maintenance Inspection of SCM Chemicals Hawkins Point Plant Batch Attack Lagoon, Baltimore, MD. Prepared by Maryland Department of the Environment for U.S. EPA Region III. October 1994.
Du Pont DeLisle Plant Pass Christian, MS	RCRA Facility Investigation Report. Prepared by Du Pont Corporate Remediation Group for Du Pont DeLisle Plant. December 1999.
Degussa Corporation Theodore, AL	Preliminary Investigation Soil/Groundwater Assessment Report for the Sodium Cyanide Plant, Theodore, AL. Degussa Corporation March 1997.
Du Pont Memphis Plant Millington, TN	RCRA Facility Investigation Work Plan Memphis Plant. E.I. du Pont Nemours & Co., Inc. Memphis, TN. Du Pont Environmental. July 1992.
U.S. Antimony Corp. Thompson Falls, MT	United States Antimony Corp. Stibnite Hill Mine Project. Operating Permit 00045. 6th Review. January 1999.

Soil Properties. With the exception of soil pH, all soil parameters required by EPACMTP are generated within EPACMTP using the nationwide relationships with soil texture shown in Table 4-17. Data collection for offsite landfills, therefore, focused on determining the frequency of soil textures within the 100-mile-radius area of interest around each offsite landfill location. A GIS was used to determine the STATSGO soil map units and their areas within each area of interest. As shown in Figure 4-3, this information was then passed to a data processing system that determined the predominant soil texture for each map unit (from the CONUS database) and then computed the fraction of the 100-mile-radius area covered by each of the 12 standard USDA soil textures contained in STATSGO.

Because EPACMTP uses three soil texture classifications to represent the 12 USDA textures, it was necessary to assign each of the USDA textures to an EPACMTP texture. Table 4-17 shows these assignments along with the mean hydrological soil properties that were considered in making the assignments. The database used these assignments to calculate, from the USDA texture fractions, the fraction of the area of interest covered by each EPACMTP soil texture. These fractions were then applied to set the frequency distributions used to develop 10,000 soil EPACMTP texture codes for each offsite landfill modeled in this analysis. For each Monte Carlo model run, a soil texture was passed to EPACMTP, which used the texture to assign the hydrologic parameters shown in Figure 4-4.

Table 4-17. Soil Texture Crosswalk for EPACMTP MegaTextures

EPACMTP MegaTexture	USDA Texture	ρ_b (kg/L)	W_{CS}	W_{CR}	β	K_{sat} (m/yr)
Sandy loam (SNL)	Sand	1.5	0.43	0.045	4.05	29.7
	Loamy sand	1.6	0.41	0.057	4.38	14.59
	Sandy loam	1.6	0.41	0.065	4.90	4.42
Silt loam (SIL)	Sandy clay loam	1.6	0.39	0.10	7.12	1.31
	Loam	1.5	0.43	0.078	5.39	1.04
	Silt loam	1.5	0.45	0.067	5.30	0.45
	Clay loam	1.6	0.41	0.095	8.52	0.26
	Silt	1.4	0.46	0.034	na	0.25
Silty clay loam (SCL)	Clay	1.6	0.38	0.068	11.4	0.20
	Sandy clay	1.6	0.38	0.10	10.4	0.12
	Silty clay loam	1.5	0.43	0.089	7.75	0.07
	Silty clay	1.7	0.36	0.070	10.4	0.02

Bulk density (ρ_b) calculated from saturated water content (W_{CS}).
W_{CS}, hydraulic conductivity (K_{sat}), residual water content (W_{CR}) from Carsel and Parrish (1988).
Soil moisture coefficient (β) from Clapp and Hornberger (1978); no values for silt.

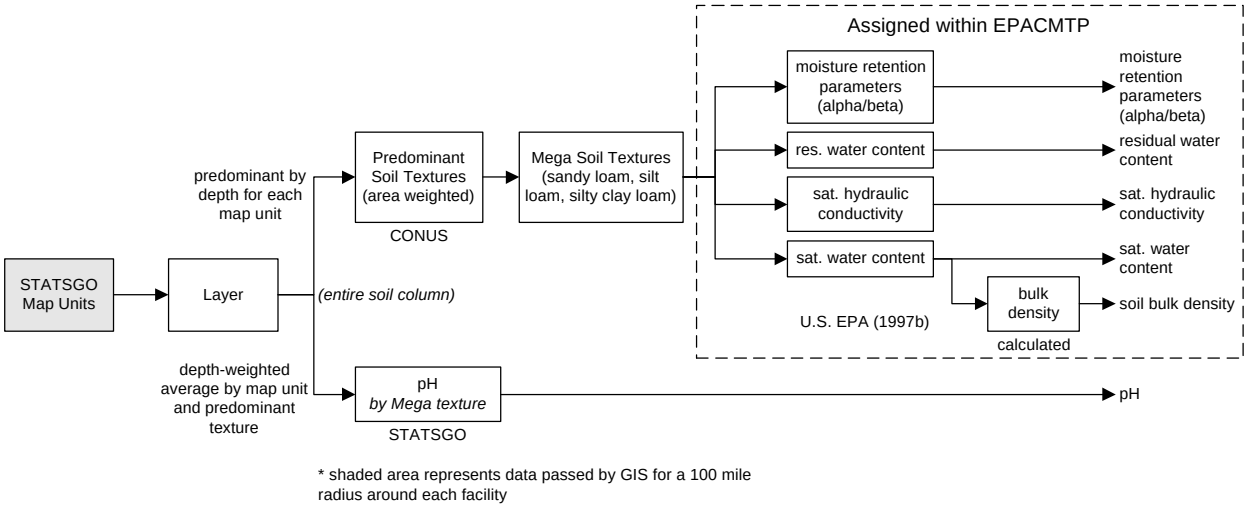

Figure 4-4. Data collection process for soil properties for offsite landfills.

The area within the radius covered by each soil texture was used to set the distribution of soil textures used in the EPACMTP. Because the EPACMTP uses three soil textures, it was necessary to roll up the 12 STATSGO textures to the closest EPACMTP soil texture. Table 4-17 shows the crosswalk between USDA texture and megatexture used to calculate megatexture fractions for each 100-mile-radius area of interest. This crosswalk was developed considering mean hydrologic property information for each USDA texture shown in Table 4-17.

Final soil texture fractions for each offsite location are shown in Table 4-18, along with soil pH and infiltration rates for each texture. EPACMTP uses these soil texture codes to select values for the parameters listed in Table 4-13 from a distribution of values appropriate for each soil texture within EPACMTP. These soil texture fraction also were used to estimate transport in the unsaturated zone and to determine the landfill infiltration rates and recharge rates for each site.

Although not assigned by soil texture within EPACMTP, soil pH was calculated in a similar fashion for each of the three EPACMTP soil textures.

Aquifer Characteristics. For offsite landfills, aquifer characteristics were assigned by EPACMTP based on hydrogeologic environment as described in U.S. EPA (1997a). Hydrogeologic environments were assigned for the 100-mile radius around each offsite facility using both GIS data sources and the non-GIS data sources listed in Table 4-16. First, the GIS was used to overlay the 100-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 (USGS, 1985) to determine the principal aquifer types present within the 100-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). This cross-walk table inables the geological data collected by the GIS to be related to the HGDB environment. The table shows, by Heath region, the HGDB environment for each DRASTIC setting

HGDB hydrogeologic environment fractions (i.e., the portion of the region assigned to each of the 12 hydrogeologic environments) were defined and used for offsite landfills as follows. If the 100-mile radius area contained only one HGDB environment, the fraction assigned was 1.0 and all groundwater model runs for this location were associated with that hydrologic environment. If more than one HGDB environment was present, each environment was assigned an equal fraction based on the number of environments within the 100-mile radius, except for the two alluvial (river valley) hydrogeologic environments (with and without overbank deposits). Because these subenvironments could not be distinguished within the general alluvial valley aquifer types provided in the GIS coverages, each was given half of the weight of the alluvial valley aquifer. For example, if the sand and gravel and alluvial hydrogeologic environment were both present in the area of interest around a facility, the assigned hydrogeologic environment fractions would be 0.5 sand and gravel, 0.25 river valleys with overbank deposits, and 0.25 river valleys without overbank deposits. Unequal fractions were also assigned for southeastern sites near the Coastal Plain/Piedmont Boundary (Augusta, GA, Blythe, GA, Elgin, SC). The fractions

Table 4-18. Soil Data and Infiltration Rates: Offsite Landfills

Facility / City	HELP Climate Center	EPACMTP Soil Texture	%	LF Infiltration/ Recharge (m/yr)	Soil pH
Hydrogen cyanide sector					
Anahuac, TX	96 Lake Charles, LA	Silty clay loam (SCL)	56	0.2817	6.6
		Silt loam (SLT)	32	0.3647	5.7
		Sandy loam (SNL)	12	0.4641	5.7
Millington, TN	90 Little Rock, AK	Silty clay loam (SCL)	38	0.2824	6.3
		Silt loam (SLT)	58	0.3531	5.4
		Sandy loam (SNL)	4	0.4336	5.5
Sodium chlorate sector					
Blythe, GA	95 Atlanta, GA	Silty clay loam (SCL)	24	0.2822	5.3
		Silt loam (SLT)	40	0.3416	5.1
		Sandy loam (SNL)	36	0.3993	5.2
Elgin, SC	93 Charleston, SC	Silty clay loam (SCL)	43	0.2123	5.2
		Silt loam (SLT)	26	0.2609	5.0
		Sandy loam (SNL)	31	0.3287	5.3
Ephrata, WA	9 Yakima, WA	Silty clay loam (SCL)	3	0.0003	7.3
		Silt loam (SLT)	56	0.0000	7.3
		Sandy loam (SNL)	41	0.0023	6.9
Perdue Hill, AL	95 Atlanta, GA	Silty clay loam (SCL)	7	0.2822	6.0
		Silt loam (SLT)	59	0.3416	5.0
		Sandy loam (SNL)	34	0.3993	5.0
Starkville, MS	90 Little Rock, AK	Silty clay loam (SCL)	32	0.2824	5.7
		Silt loam (SLT)	62	0.3531	5.0
		Sandy loam (SNL)	5	0.4336	5.0

(continued)

Table 4-18. (continued)

Facility / City		HELP Climate Center	EPACMTP Soil Texture	%	LF Infiltration/ Recharge (m/yr)	Soil pH
Sodium phosphate sector						
	Augusta, GA	95 Atlanta, GA	Silty clay loam (SCL)	27	0.2822	5.3
			Silt loam (SLT)	37	0.3416	5.1
			Sandy loam (SNL)	37	0.3993	5.3
	Chicago, IL	42 Chicago, IL	Silty clay loam (SCL)	41	0.0620	6.9
			Silt loam (SLT)	36	0.0798	6.8
			Sandy loam (SNL)	23	0.1138	6.5
	East St. Louis, IL	54 East St. Louis, IL	Silty clay loam (SCL)	57	0.0704	5.9
			Silt loam (SLT)	42	0.1435	5.7
			Sandy loam (SNL)	1	0.1676	6.3
Antimony oxide sector						
U.S. Antimony	Thompson Falls, MT	8 Pullman, WA	Silty clay loam (SCL)	0	NA	NA
			Silt loam (SLT)	5	0.0069	7.7
			Sandy loam (SNL)	95	0.0132	6.3
Titanium dioxide sector						
Du Pont DeLisle	Pass Christian, MS	92 New Orleans, LA	Silty clay loam (SCL)	26	0.4503	6.5
			Silt loam (SLT)	49	0.5893	5.1
			Sandy loam (SNL)	25	0.7445	5.0
Du Pont Edgemoor	Edgemoor, DE	71 Philadelphia, PA	Silty clay loam (SCL)	7	0.1641	5.5
			Silt loam (SLT)	58	0.2007	5.2
			Sandy loam (SNL)	35	0.2609	4.8
Du Pont Johnsonville	West Camden, TN	89 Nashville, TN	Silty clay loam (SCL)	40	0.3769	5.5
			Silt loam (SLT)	55	0.4674	5.3
			Sandy loam (SNL)	5	0.5395	5.2

(continued)

Table 4-18. (continued)

Facility / City	HELP Climate Center	EPACMTP Soil Texture	%	LF Infiltration/ Recharge (m/yr)	Soil pH
Kemira Savannah, GA	93 Charleston, SC	Silty clay loam (SCL)	11	0.2123	5.3
		Silt loam (SLT)	49	0.2609	5.0
		Sandy loam (SNL)	40	0.3287	5.4

in these cases represent the approximate fraction of the area of interest in the Piedmont or Coastal Plain provinces. Aquifer parameters resulting from this process are presented in Table 4-19 for the offsite landfills modeled in this analysis.

These fractions were then used as a frequency distribution to generate a distribution of 10,000 codes for 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 equal areas in the vicinity near one facility location, each would 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. Certain parameters of temperature, pH, and foc were not used in this analysis. These additional parameters are used when modeling organic constituents that hydrolyze in the environment and the rate of this degradation process is altered by these parameters. However, metals do not degrade and are not affected by these parameters. The pH parameter is known to affect the K_d of metals; however, in the groundwater modeling in this risk assessment empirical, K_ds are used and thus pH is not a factor.

The final step in the process was to construct a 10,000-record set of hydrogeologic environments and associated hydrogeologic parameters for each offsite landfill modeled. Using the hydrogeologic environment fractions defined for each 100-mile radius area, a hydrogeologic environment was assigned to each occurrence of that location in the 10,000-record location data set. For example, for the Du Pont Delisle Plant, the fractions assigned to hydrogeologic environments are 50, 25, and 25 percent for hydrogeologic environments 10, 6, and 7, respectively. Consequently, for this location, hydrogeologic environments 10, 6, and 7 would occur approximately 5,000, 2,500, and 2,500 times, respectively, depending on the random assignments.

Receptor Well Location—Offsite Facilities. To predict the groundwater concentration at a downgradient receptor well with EPACMTP, the well location must be specified in all three dimensions for each realization: x-direction (longitudinal), y-direction (lateral), and z-direction (vertical). The distance from the downgradient edge of the SWMU to the well, measured along the plume centerline (the longitudinal distance), is called X-WELL. The distance from the plume centerline to the well, measured perpendicular to X-WELL, (the lateral distance) is called Y-WELL. The depth of the well intake point below the water table (the vertical distance) is called Z-WELL.

Table 4-19. Aquifer Assignments: Offsite Landfills

Facility	Inputs				Background Information				
	GW Temp ºC	Hydrogeologic Environment		Fraction	Hydrogeologic Region	Productive Aquifers	Surficial Geology	DRASTIC Setting	
Hydrogen Cyanide Sector									
Millington, TN	17	4	Sand and gravel	0.50	Atlantic and Gulf Coastal Plain	Unconsolidated sand and gravel (Tertiary sand)	Deeply weathered loess [es]; Loam - texture variable [rl]	10Ab	Unconsolidated & semiconsolidated shallow surficial aquifer
		6	River valleys and floodplains with overbank deposits	0.25	Alluvial Valleys	Unconsolidated watercourse	Floodplain and alluvium gravel terraces [al]	10Ba	River alluvium with overbank deposits
		7	River valleys and floodplains without overbank deposits	0.25	Alluvial Valleys	Unconsolidated watercourse	Floodplain and alluvium gravel terraces [al]	10Bb	River alluvium without overbank deposits
Anahuac, TX	24	4	Sand and gravel	0.50	Atlantic and Gulf Coastal Plain	Unconsolidated sand and gravel (Gulf Coast)	Sandy coastal ground with organic layer [gp]; Loam, texture variable [rl]	10Ab	Unconsolidated & semiconsolidated shallow surficial aquifer
		6	River valleys and floodplains with overbank deposits	0.25	Alluvial Valleys	Unconsolidated watercourse	Floodplain and alluvium gravel terraces [al]	10Ba	River alluvium with overbank deposits
		7	River valleys and floodplains without overbank deposits	0.25	Alluvial Valleys	Unconsolidated watercourse	Floodplain and alluvium gravel terraces [al]	10Bb	River alluvium without overbank deposits

(continued)

Table 4-19. (continued)

Facility	Inputs				Background Information				
	GW Temp °C	Hydrogeologic Environment		Fraction	Hydrogeologic Region	Productive Aquifers	Surficial Geology	DRASTIC Setting	
Sodium Phosphate Sector									
Chicago, IL	12	3	Till over sedimentary rock	0.33	Glaciated Central	Consolidated sandstone/carbonate	Ice-laid deposits (till), mostly sand and silt [ts]; Gravel, sand and clay deposited by glacial streams [w]; Wisconsinian loess [wl]	7Aa	Glacial till over bedded sedimentary rocks
		8	Outwash	0.33	Glaciated Central	Unconsolidated sand and gravel		7Ba	Outwash
		12	Solution Limestone	0.33	Glaciated Central	Shallow dolomite and limestone		7Ac	Glacial till over solution limestone
E St. Louis, IL	14	4	Sand and gravel	0.25	Glaciated Central	No aquifer mapped	Pre-Wisconsinian drift [pW]; Wisconsinian loess [wl]	7B; 7C	Outwash; Moraine
		6	River valleys and floodplains with overbank deposits	0.125	Alluvial Valleys	Unconsolidated watercourse	Floodplain and alluvium gravel terraces [al]	7Ea; 6Fa	River alluvium with overbank deposits
		7	River valleys and floodplains without overbank deposits	0.125	Alluvial Valleys	Unconsolidated watercourse	Floodplain and alluvium gravel terraces [al]	7Eb; 6Fb	River alluvium without overbank deposits
		9	Till and till over outwash	0.25	Glaciated Central	No aquifer mapped	Pre-Wisconsinian drift [pW]; Wisconsinian loess [wl]	7A	Glacial till

(continued)

Table 4-19. (continued)

Facility	Inputs				Background Information				
	GW Temp °C	Hydrogeologic Environment		Fraction	Hydrogeologic Region	Productive Aquifers	Surficial Geology	DRASTIC Setting	
		12	Solution limestone	0.25	Nonglaciated central	Consolidated carbonate (Pennsylvania and Mississippian limestone)	Deeply weathered loess [es]; Red clay, kaolinitic [rls]; Residuum with abundant quartz [rgr]; Sandy or silty residuum, includes loess [rsi]	6E	Solution limestone
Augusta, GA	19	1	Metamorphic and igneous	0.40	Piedmont and Blue Ridge	No aquifer mapped	Residuum with abundant quartz [rgr]; Micaceous residuum [rsh]; Sandy residuum [rs]	8D	Regolith
		10	Unconsolidated and semiconsolidated shallow aquifer	0.60	Atlantic and Gulf Coastal Plain	Unconsolidated sand and gravel; Consolidated sandstone/carbonate under sand and gravel	Loam-texture variable [rl]; Sandy coastal ground with organic layer [gp]	10Ab	Unconsolidated & semiconsolidated shallow surficial aquifer

(continued)

Table 4-19. (continued)

Facility	Inputs				Background Information				
	GW Temp °C	Hydrogeologic Environment		Fraction	Hydrogeologic Region	Productive Aquifers	Surficial Geology	DRASTIC Setting	
Sodium Chlorate Sector									
Elgin, SC	18	1	Metamorphic and igneous	0.70	Piedmont and Blue Ridge	No aquifer mapped	Residuum with abundant quartz [rgr]; Micaceous residuum [rsh]; Sandy residuum [rs]	8D	Regolith
		10	Unconsolidated and semiconsolidated shallow surficial aquifer	0.30	Atlantic and Gulf Coastal Plain	Unconsolidated sand and gravel; Consolidated sandstone/carbonate under sand and gravel	Loam-texture variable [rl]; Sandy coastal ground with organic layer [gp]	10Ab	Unconsolidated & semiconsolidated shallow surficial aquifer
Starkville, MS	19	6	River valleys and floodplains with overbank deposits	0.25	Alluvial Valleys	Unconsolidated watercourse	Floodplain and alluvium gravel terraces [al]	10Ba	River alluvium with overbank deposits
		7	River valleys and floodplains without overbank deposits	0.25	Alluvial Valleys	Unconsolidated watercourse	Floodplain and alluvium gravel terraces [al]	10Bb	River alluvium without overbank deposits
		10	Unconsolidated and semiconsolidated shallow surficial aquifer	0.50	Atlantic and Gulf Coastal Plain	Unconsolidated sand and gravel	Loam - texture variable [rl]; Sandy residuum [rs]; Clay residuum [rc]	10Ab	Unconsolidated & semiconsolidated shallow surficial aquifer

(continued)

Table 4-19. (continued)

Facility	Inputs				Background Information				
	GW Temp °C	Hydrogeologic Environment		Fraction	Hydrogeologic Region	Productive Aquifers	Surficial Geology	DRASTIC Setting	
Ephrata, WA	14	7	River valleys and floodplains without overbank deposits	0.50	Alluvial Valleys	Unconsolidated watercourse	Floodplain and alluvium gravel terraces [al]	3C	River alluvium
		2	Bedded sedimentary rock	0.50	Columbia Lava Plateau	Consolidated volcanic; Consolidated volcanic under sand and gravel.	Basalt [b]; Wisconsinan loess [wl]; Sand sheets [s]	3C, 3D	Lava flows
Blythe, GA	18	1	Metamorphic and igneous	0.40	Piedmont and Blue Ridge	No aquifer mapped	Residuum with abundant quartz [rgr]; Micaceous residuum [rsh]; Sandy residuum [rs]	8D	Regolith
		10	Unconsolidated and semiconsolidated shallow surficial aquifer	0.60	Atlantic and Gulf Coastal Plain	Unconsolidated sand and gravel; Consolidated sandstone/carbonate under sand and gravel	Loam-texture variable [rl]; Sandy coastal ground with organic layer [gp]	10Ab	Unconsolidated & semiconsolidated shallow surficial aquifer

(continued)

Table 4-19. (continued)

Facility	Inputs				Background Information				
	GW Temp ºC	Hydrogeologic Environment		Fraction	Hydrogeologic Region	Productive Aquifers	Surficial Geology	DRASTIC Setting	
Perdue Hill, AL	20	6	River valleys and floodplains with overbank deposits	0.17	Alluvial Valleys	Unconsolidated watercourse	Floodplain and alluvium gravel terraces [al]	10Ba	River alluvium with overbank deposits
		7	River valleys and floodplains without overbank deposits	0.17	Alluvial Valleys	Unconsolidated watercourse	Floodplain and alluvium gravel terraces [al]	10Bb	River alluvium without overbank deposits
		10	Unconsolidated and semiconsolidated shallow surficial aquifer	0.33	Atlantic and Gulf Coastal Plain	Unconsolidated sand and gravel	Loam-texture variable [rl]; Backshore deposits [bm]	10Ab	Unconsolidated & semiconsolidated shallow surficial aquifer
		12	Solution limestone	0.33	Southeast Coastal Plain	Consolidated carbonate	Loam-texture variable [rl]; Backshore deposits [bm]	11A	Solution limestone and shallow surficial aquifers

(continued)

Table 4-19. (continued)

Facility	Inputs				Background Information				
	GW Temp °C	Hydrogeologic Environment		Fraction	Hydrogeologic Region	Productive Aquifers	Surficial Geology	DRASTIC Setting	
Titanium Dioxide Sector									
Du Pont Edgemoor, DE	13	10	Unconsolidated and semiconsolidated shallow surficial aquifer	0.33	Atlantic and Gulf Coastal Plain	Unconsolidated sand and gravel	Loam - texture variable [rl]; Sandy residuum [rs]	10Ab	Unconsolidated & semiconsolidated shallow surficial aquifer
		2	Bedded sedimentary rock	0.33	Nonglaciated Central	Consolidated sandstone & limestone	Sandy/stony colluvium [co/ss,sh]; Triassic residuum [rtr]	6Da&b; 6H	Alternating sandstone, limestone, and shale - thin or deep regolith; Triassic basins
		1	Metamorphic and igneous	0.33	Piedmont and Blue Ridge	No aquifer mapped	Residuum with abundant quartz [rgr]; Micaceous residuum [rsh]; Sandy residuum [rs]	8D	Regolith
Du Pont West Camden, TN	17	12	Solution limestone	0.33	Nonglaciated Central	Consolidated carbonate & sandstone	Sandy/silty residuum [rsi]; Red clay; Cherty red clay [rlc]	6E	Solution limestone

(continued)

Table 4-19. (continued)

Facility	Inputs				Background Information				
	GW Temp °C	Hydrogeologic Environment		Fraction	Hydrogeologic Region	Productive Aquifers	Surficial Geology	DRASTIC Setting	
		10	Unconsolidated and semiconsolidated shallow surficial aquifer	0.33	Atlantic and Gulf Coastal Plain	Unconsolidated sand and gravel	Loam, texture variable [rl]; Deeply weathered loess [es]	10Ab	Unconsolidated & semiconsolidated shallow surficial aquifer
		6	River valleys and floodplains with overbank deposits	0.17	Alluvial Valleys	Unconsolidated watercourse	Floodplain and alluvium gravel terraces [al]	10Ba	River alluvium with overbank deposits
		7	River valleys and floodplains without overbank deposits	0.17	Alluvial Valleys	Unconsolidated watercourse	Floodplain and alluvium gravel terraces [al]	10Bb	River alluvium without overbank deposits
Kemira Savannah, GA	20	10	Unconsolidated and semiconsolidated shallow surficial aquifer	0.50	Atlantic and Gulf Coastal Plain	Consolidated sandstone/ carbonate & same under sand and gravel	Sea islands [si]; Sandy coastal ground [gp]; Loam, texture variable [rl]	10Ab	Unconsolidated & semiconsolidated shallow surficial aquifer
		4	Sand and gravel	0.50	Southeast Coastal Plain	Consolidated carbonate & same under sand and gravel	Sea islands [si]; Sandy coastal ground [gp]; Loam, texture variable [rl]	11B	Coastal deposits
Pass Christian, MS	21	10	Unconsolidated and semiconsolidated shallow surficial aquifer	1.00	Atlantic and Gulf Coastal Plain	Unconsolidated sand and gravel	Loam, texture variable [rl]; Deeply weathered loess [es]	10Ab	Unconsolidated & semiconsolidated shallow surficial aquifer

For the Monte Carlo analysis, Z-WELL is varied uniformly throughout the aquifer thickness or throughout the upper 10 m of the aquifer thickness, whichever is less. That is, the well depth is never allowed to exceed 10 m below the water table. This limitation for Z-well was chosen primarily for two reasons: (1) to be consistent with a residential well scenario (these wells are generally shallow because of the increased costs of drilling a deeper well) and (2) to produce a conservative estimate of risk (because the infiltration rate is generally lower than the groundwater seepage velocity, groundwater plumes generally tend to be relatively shallow).

For the deterministic analysis, Z-WELL was specified as 5 m or half the aquifer thickness, whichever is less. Although groundwater plumes generally tend to be relatively shallow in thick aquifers, the actual midpoint of the plume was determined by the climatic and hydrogeologic setting. While this limitation on Z-WELL does not guarantee that the well depth will correspond to the middle of the plume (and thus to the highest groundwater concentration), it is generally expected that such a limitation on Z-WELL will give a conservative (or high) estimate of groundwater well concentrations.

In most hydrogeologic settings, it is unlikely that the top of the groundwater plume will be more than 10 m below the water table. However, as the plume moves downgradient, it can be pushed deeper by the downward flow of ambient recharge if this recharge rate is similar to or higher than the groundwater seepage velocity. In such cases, this deepening effect becomes more pronounced as the plume moves farther downgradient from the SWMU.

To verify that this limitation on Z-WELL did not bias the results toward lower groundwater concentrations for this analysis, a test was performed for a waste stream in the titanium dioxide sector: the sulfate digestion sludge that is disposed of in the onsite landfill at the Millennium Hawkins Point Plant. This waste stream was chosen because, for this facility, site-specific values of 2,500 to 5,000 ft (762 to 1,524 m) were used for the distribution of receptor well distances. These values essentially comprise the upper half of the default distribution of 2 to 5,282 feet (0.6 to 1,610 m) (U.S. EPA, 1997a, 1993). The EPACMTP model was run twice for this facility and waste stream; the two Monte Carlo runs were identical except that, in one case, the Z-WELL was constrained to lie within the upper 10 m of the aquifer if the aquifer was greater than 10 m deep (aquifer thickness was one of the parameters varied in the Monte Carlo modeling runs), and, in the other case, the Z-WELL was allowed to vary between the top and the bottom of the aquifer. The results indicated that the 50^{th} and 90^{th} percentile DAFs from these two runs were identical, and the 10^{th} percentile DAFs differed by less than 1 percent.

There are two options in EPACMTP for specifying the longitudinal and lateral position of the well: (1) specifying X-WELL and Y-WELL directly as either constant values or statistical or empirical distributions of values; or (2) deriving X-WELL and Y-WELL as a function of the radial distance to the well *R* and the angle off of the plume centerline θ.

For the Monte Carlo analysis, the well position was specified using R and θ. To be consistent with previous listing determinations, the radial distance to the nearest downgradient groundwater receptor well for all offsite SWMUs was based on information for municipal landfills (U.S. EPA, 1997a, 1993). That is, the data from this distribution were used for both municipal and offsite industrial landfills. The distribution of downgradient distances from municipal landfills

to the nearest drinking water well and the associated cumulative distributions are presented in Table 4-20. The angle off of the plume centerline θ was specified as a uniform distribution from 0° to 90°. In addition, an EPACMTP control parameter called LYCHECK was set to true, which constrains the well to lie within the approximate areal extent of the plume. This setting, together with the specified values for *R* and θ, ensure that the distribution of well locations that were chosen represent a uniform distribution between the plume centerline and the estimated lateral extent of the plume for any given value of X-WELL.

For the deterministic analysis, the well position was specified using X-WELL and Y-WELL. The central tendency values for these parameters was derived from the input distributions. Specifically, the central tendency value for the longitudinal distance to the well was specified as the median value from the input distribution. The central tendency value for the lateral position of the well was then calculated using the SWMU area and the given value for X-WELL so that the lateral position of the well was half way between the plume centerline and the estimated lateral edge of the plume. The high-end value for X-WELL was specified as the 10^{th} percentile value from the input distribution, and the high-end value of Y-WELL was specified as 0.0 (on the plume centerline).

4.3.3.3 Data Collection for Onsite SWMUs. Data collection for onsite landfills and surface impoundments differed from that for offsite landfills in that the goal was to capture site-specific conditions as closely as possible from the available sources. Given this objective, site-specific documents were the preferred source of data, with data gaps filled using the more comprehensive nationwide data sources. In this context, distributions were developed and applied in the Monte Carlo analysis to represent uncertainty in site-specific conditions.

Soil Properties. Although the site-specific reports for the onsite SWMUs did contain some information on soils, in no instance did the reports provide sufficiently detailed and consistent information on all of the soil parameters in Table 4-13. Using the GIS, SWMU locations were assigned to a single STATSGO soil map unit to each site. In all cases, soil texture information for each assigned STATSGO map unit was consistent with the information provided in the site-specific reports.

Figure 4-5 illustrates the soil data collection methodology for onsite landfills and surface impoundments. In this case, data on component soil textures within the assigned map unit were used to develop the megatexture percentages in a similar manner as described for STATSGO map units for the offsite facilities. The component soil textures in the selected map unit for the site were determined along with the fraction of the total area within the map unit covered by each soil texture. This fraction was used to set the frequency distribution for developing a distribution of 10,000 soil texture codes used in the EPACMTP. The EPACMTP soil texture fractions were developed from the STATSGO soil textures, using the same STATSGO soil texture to EPACMTP soil texture crosswalk illustrated in Table 4-17.

Final soil texture fractions for each onsite location are shown in Table 4-21, along with soil pH and infiltration rates for each soil texture. As in the offsite analysis, EPACMTP uses these soil texture codes to select values for the parameters listed in Table 4-13 from a national

Table 4-20. Empirical Distribution of Distance to Closest Residential Well

Distance to Nearest Well (m)	Cumulative Probability
0.0	0.0
0.6	0.0
13.7	0.03
19.8	0.04
45.7	0.05
104	0.1
152	0.15
183	0.2
244	0.25
305	0.35
366	0.4
427	0.5
610	0.6
805	0.7
914	0.8
1160	0.85
1220	0.9
1370	0.95
1520	0.98
1610	1.0

Source: U.S. EPA, 1993. *Parameter Values for EPA's Composite Model for Landfills (EPACML) Used in Developing Nationwide Regulations: Toxicity Characteristic Rule*. Office of Solid Waste, Washington, DC.

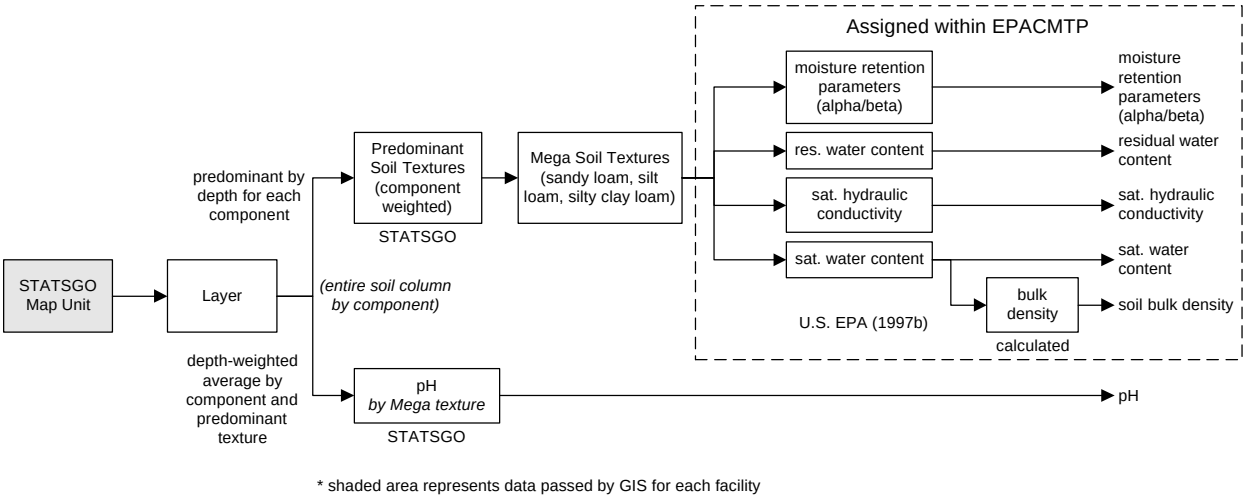

Figure 4-5. Data collection process for soil properties for onsite landfills and surface impoundments.

Table 4-21. Soil Data and Infiltration Rates: Onsite Landfills and Surface Impoundments

Facility	Unit Type	Climate Center	STATSGO Soil Map Unit	EPACMTP Soil Texture	Percent	pH	Landfill Infiltration/ Recharge Rate (m/yr)
Degussa Corp.	SI	92 New Orleans, LA	AL224	Sandy loam (SNL)	80	4.80	0.7445
				Silt loam (SLT)	20	5.01	0.5893
Du Pont Delisle	SI	92 New Orleans, LA	MS183	Sandy loam (SNL)	52	4.82	0.7445
				Silt loam (SLT)	48	4.63	0.5893
Du Pont Memphis	SI	90 Little Rock, AK	TN010	Silt loam (SLT)	92	5.17	0.3531
				Silty clay loam (SCL)	8	5.03	0.2824
Du Pont New Johnsonville	SI	89 Nashville, TN	TN073	Silt loam (SLT)	52	5.19	0.4674
				Silty clay loam (SCL)	48	5.24	0.3769
Kerr-McGee	SI	90 Little Rock, AK	MS137	Sandy loam (SNL)	34	4.95	0.4336
				Silt loam (SLT)	66	4.96	0.3531
Millennium HPP	LF & SI	71 Philadelphia, PA	MD007	Sandy loam (SNL)	36	4.83	0.2609
				Silt loam (SLT)	25	4.33	0.2007
				Silty clay loam (SCL)	39	4.49	0.1641
U.S. Antimony	LF	8 Pullman, WA	MT647	Sandy loam (SNL)	95	6.32	0.0132
				Silt loam (SLT)	5	7.71	0.0069

Soil data from analysis of STATSGO database and GIS coverages.
Infiltration/recharge rates are HELP-derived regional values by soil texture (U.S. EPA, 1997a).

distribution of values appropriate for each soil texture within EPACMTP. For landfills the soil textures were also used to select regional landfill infiltration rates and recharge rates from the EPACMTP model database. These infiltration and recharge rates were developed using the HELP model regional climate data and soil properties.

For the deterministic analysis, none of the soil parameters were determined to be one of the two most sensitive parameters, so central tendency values were used for the soil parameters except for soil dispersivity, for which the median values from the soil-specific distributions were used in the Monte Carlo analysis. The central tendency value for soil dispersivity was derived from the central tendency unsaturated zone thickness. For organic compounds, the central tendency value for soil K_d was derived from k_{oc} and f_{oc}. For metals, the central tendency value for soil K_d was the median value from the metal-specific K_d distribution used in the Monte Carlo analysis. The high-end soil K_d value was set equal to the 10th percentile value from this distribution.

Aquifer Characteristics. Aquifer properties were estimated using site-specific data for six facilities: the Kerr-McGee plant at New Hamilton, Mississippi; U.S. Antimony near Thompson Falls, Montana; the Du Pont Memphis Plant in Millington, Tennessee; the Millennium Hawkins Point Plant near Baltimore, Maryland; the Du Pont Delisle Facility in Pass Christian, Mississippi; and the Degussa Facility in Theodore, Alabama. Data for these six on-site locations are provided in Table 4-22. For Kerr-McGee, a RCRA compliance sampling report contained aquifer hydraulic conductivity measurements from a 60-h pump test, a laboratory permeability test, and a slug test, which showed very good agreement, along with a value for hydraulic gradient and a range for aquifer thickness. Depth-to-water measurements, available from 72 monitoring wells, were used to develop an empirical distribution for vadose zone thickness.

For U.S. Antimony, borehole logs contained in a permit showed a very consistent subsurface lithology dominated by gravel in an alluvial mountain valley hydrogeologic setting. A lognormal distribution of hydraulic conductivities for this material was developed based on information on gravel aquifers in Freeze and Cherry (1979) and Fetter (1994), along with a range for the appropriate DRASTIC hydrogeologic setting (1Ba, Alluvial Mountain Valleys - East) in Aller et al. (1987). Hydraulic gradient was estimated from a water table map provided in the permit. However, these measurements were taken during March when infiltration and gradient would be expected to be extremely high. To represent annual average conditions, the range of gradients from these measurements, centered around the median value from HGDB environment 1Ba (Alluvial mountain valleys - east), was used as a uniform distribution in the model.

Aquifer thickness at U.S. Antimony was determined based on the borehole logs and water table data. The permit information indicated a seasonally variable water table at the site, ranging from 11 to 60 ft. Because an annual average depth to water is needed by the model, an average of these extremes was used. To account for uncertainty in whether the measured values were typical and whether the average of the point estimates in the permit was a true average over the entire year, a correlated range of aquifer and vadose zone thicknesses was developed to represent year-to-year uncertainty in these variables.

Table 4-22. Site-Specific Aquifer Data: Onsite Landfills and Surface Impoundments

Variable	Distribution Type	Min	Central Tendency	Max	Units	Comments
Degussa Corporation						
Unsaturated thickness	Constant		1.0		m	Value chosen to be consistent with minimum value allowed in SI source module
Aquifer thickness	Uniform	13	36.8	61	m	O'Neil and Mettee, 1982
Longitudinal hydraulic conductivity (K)	Uniform	9,000	17,000	25,000	m/yr	O'Neil and Mettee, 1982
Hydraulic gradient	Uniform	0.0022	0.0047	0.0072	m/m	1998 Risk-Based CA Plan
Radial distance to well (R)	Uniform	945	1,260	1,610	m	U.S. EPA, 2000b
Groundwater temperature	Constant		22		°C	
Du Pont Memphis Plant						
Longitudinal hydraulic conductivity	Constant		11,000		m/y	1992 RFI Work Plan
Hydraulic gradient	NA	NA	NA	NA	NA	Not needed because only screening analysis performed.
Aquifer thickness	Constant		11		m	Based on hydraulic conductivity and transmissivity data from 24-h pump test
Unsaturated thickness	NA	NA	NA	NA	NA	Not needed because only screening analysis performed.
Groundwater temperature	NA	NA	NA	NA	NA	Not needed because only screening analysis performed.
Du Pont New Johnsonville						
Unsaturated thickness	NA	NA	NA	NA	NA	Not needed because only screening analysis performed.
Aquifer thickness	Empirical	1.00	7.62	36.6	m	HG Region 6
Longitudinal hydraulic conductivity (K)	Empirical	3.15	1,890	107,000	m/yr	HG Region 6
Hydraulic gradient	NA	NA	NA	NA	NA	Not needed because only screening analysis performed.

(continued)

Table 4-22. (continued)

Variable	Distribution Type	Min	Central Tendency	Max	Units	Comments
Groundwater temperature	NA		NA		NA	Not needed because only screening analysis performed.
Du Pont DeLisle Plant						
Unsaturated thickness	Triangular	1.0	1.18	1.5	m	Dec. 1999 RFI
Aquifer thickness	Uniform	61.0	76.0	91.0	m	Dec. 1999 RFI
Longitudinal hydraulic conductivity (K)	Uniform	5.5	2,550	5,140	m/yr	July 12,1999 document
Hydraulic gradient	Uniform	0.0020	0.0040	0.0060	m/m	July 12,1999 document
Radial distance to well	Uniform	610	1,020	1,524	m	U.S. EPA, 2000f
Groundwater temperature	Constant		21		°C	
Kerr-McGee New Hamilton Plant						
Unsaturated thickness	Empirical	0.53	2.84	6.69	m	1998 RCRA Sampling Report
Aquifer thickness	Uniform	2	6.48	11	m	1998 RCRA Sampling Report
Longitudinal hydraulic conductivity	Constant		5,480		m/yr	1998 RCRA Sampling Report
Hydraulic gradient	Constant		0.0060			1998 RCRA Sampling Report
Radial distance to well	Uniform	610	1,010	1,524	m	U.S. EPA, 2000f
Groundwater temperature			18		°C	
Millennium Hawkins Point Plant						
Unsaturated thickness	Uniform	2.0	2.5	3	m	1994 RCRA O & M Inspection
Aquifer thickness	Empirical	20	40	60	m	Professional judgment used to interpret data presented in Mack and Achmad, 1986 and Achmad, 1991
Longitudinal hydraulic conductivity	Empirical	113	4,320	22,700	m/yr	Values appropriate for beach sand
Hydraulic gradient	Triangular	0.0014	0.0020	0.0026	m/m	Chapelle, 1985; Achmad, 1991
Radial distance to well	Uniform	762	1,100	1,524	m	U.S. EPA, 2000f
Groundwater temperature	Constant		13		°C	

(continued)

Variable	Distribution Type	Min	Central Tendency	Max	Units	Comments
U.S. Antimony Stibnite Hill Mine						
Unsaturated thickness	Uniform	9.4	10.9	12.4	m	1990 Operating Permit
Saturated thickness	Uniform	15	16.5	18	m	1990 Operating Permit
Longitudinal hydraulic conductivity	Lognormal	3,200	30,100	290,000	m/yr	Freeze and Cherry, 1979: Fetter, 1994; HG Region 5
Hydraulic gradient	Triangular	0.00075	0.0050	0.0093	m/m	*1990 Operating Permit* range derived from HG Region 5
Radial distance to well	Uniform	335	899	1,610	m	U.S. EPA, 2000a
Groundwater temperature	Constant		10		°C	

At the Du Pont Memphis plant, the RCRA RFI Work Plan provides the results of a 24-h pumping test on the surficial alluvial aquifer that underlies the surface impoundments of interest. Test results were given in terms of both hydraulic conductivity and transmissivity, which enabled a site-specific estimate for aquifer thickness to be calculated. A range for depth-to-groundwater (DSOIL) also was given for the surficial aquifer. No site-specific estimate for hydraulic gradient was available for the site. Based on the available descriptions of the site hydrogeology, a range of 0.002 to 0.005, the 40^{th} and 60^{th} percentile values for HGDB environment 6, was selected as a reasonable range for the site (DuPont, 1992).

Because regional data resulted in unrealistic infiltration rates for the surface impoundment scenario at the Millennium Hawkins Point Plant, site-specific data for aquifer thickness, hydraulic conductivity, and gradient were acquired. These site-specific data were gathered from available aquifer assessments and groundwater modeling reports. Aquifer thickness values were obtained from cross-section analyses (Achmad, 1991; Mack and Achmad, 1986), well logs, and geologic descriptions (Chapelle, 1985). Gradient values were taken from potentiometric maps provided by Chapelle (1985). A range of thickness values was employed to accommodate the uncertainty due to conflicting stratigraphic interpretations among the sources. Hydraulic conductivity was not cited directly in any of the sources, so it was derived from transmissivity values when provided in the source(s). In cases where only specific capacity was available, a conversion from specific capacity to transmissivity was made using the interrelationship described by Domenico and Schwartz, (1990, citing Theis et al., 1963). Additionally, transmissivity values were corrected for partial-penetration effects using the Kozeny factor, which accounts for the ratio of well screen length to aquifer thickness (Powers, 1981).

For the surface impoundment facility at Degussa, Theodore, Alabama, the underlying aquifer is the Citronelle aquifer, generally consisting of gravelly sand to fine sand with some discontinuous lenses of sandy clay. The depth to the water table was chosen to be 1.0 m for the groundwater modeling, which is consistent with minimum value allowed in the surface

impoundment module. The minimum and maximum hydraulic gradients at this site were obtained from contour maps (Degussa, 1998), and a uniform distribution was assumed. Aquifer transmissivity and aquifer thickness values were obtained from regional hydrogeologic reports (Gillett et al., 1995; O'Neil and Mettee, 1982). The transmissivity values were then converted to conductivity values by dividing by the appropriate aquifer thickness. Uniform distributions were assumed for both the conductivity and aquifer thickness.

For the surface impoundment at the Du Pont Delisle site, both depth to the water table and aquifer thickness were obtained from data presented in a recent RCRA report for the facility (Du Pont, 1999a). A triangular distribution was specified for depth to water table, and a uniform distribution was assumed for aquifer thickness. Aquifer conductivity and hydraulic gradients were obtained from data presented in another recent facility report (Du Pont, 1999b) and both were assumed to follow respective uniform distributions.

Except for water table depth, site-specific information was not adequate to assign a consistent set of correlated aquifer properties at the Du Pont facility in New Johnsonville, Tennessee. The HGDB data for hydrogeologic environment 6 were used for values of aquifer thickness, conductivity, and gradient for the surface impoundment at Du Pont New Johnsonville. Depth-to-water measurements available for this site from facility reports indicated a very shallow unsaturated zone with the water table located from zero to 1 meter below the base of the impoundment. In this case, the surface impoundment source model sets the depth to the water table to a minimum value of 1.0 m. So, for consistency, the value of 1.0 m was also used in the groundwater modeling.

As with the offsite facilities, aquifer temperature was obtained for the onsite locations from the national map in van der Leeden et al. (1990). Aquifer pH was assumed to be the same as soil pH for all sites.

For all onsite facilities with the exception of U.S. Antimony, the aquifer particle diameter was drawn from an empirical distribution provided by EPA (U.S. EPA, 1997a). Aquifer effective porosity and bulk density were then derived from the particle diameter using Equations 7.3 and 7.4 of U.S. EPA (1997a). For U.S. Antimony, more site-specific values for the effective porosity and bulk density values were available; as a result, these two parameters were each set to a constant value. For this facility, the aquifer particle diameter was not needed.

Aquifer longitudinal dispersivity was calculated by assuming that the longitudinal dispersivity follows the distribution given in Gelhar et al. (1992) for a reference well distance of 152.4 m and by scaling it appropriately for other well distances as described in U.S. EPA (1997a). The transverse dispersivity was set equal to 1/8 of the longitudinal dispersivity, and the vertical dispersivity 1/160 of the longitudinal dispersivity value. Values for aquifer fraction organic carbon were generated from a Johnson SB distribution (U.S. EPA, 1997a).

For the deterministic analysis, the selection of central tendency and high-end values for most aquifer parameters follows the procedures described in Section 4.3.1.5, with the following exceptions:

- The effective porosity and bulk density were specified as constant values and their central tendency values were obtained from the corresponding values generated in the Monte Carlo run (note that no high-end values were needed since effective porosity and bulk density were not the most sensitive parameters for any constituents in any of the waste streams).

- The central tendency value for aquifer longitudinal dispersivity was calculated by assuming that the longitudinal dispersivity equaled 4.35 m (which is the 50th percentile value from the Gelhar et al. (1992) distribution) for a well distance of 152.4 m and by scaling this value to the actual well distance.

Central tendency values for the transverse and vertical dispersivities were set equal to 1/8 and 1/160 of the central tendency longitudinal dispersivity, respectively.

Receptor Well Location—Onsite SWMUs. For the Monte Carlo and deterministic analyses, the Z-WELL values used when modeling onsite SWMUs were identical to the values used when modeling offsite SWMUs; the method for determining these values is described in Section 4.3.3.2.

For the Monte Carlo analysis, the well position was specified using radial distance *R* and angle off of the plume centerline θ. A uniform distribution for *R* was used for onsite landfills and surface impoundments. The minimum value in the distribution was determined based on the distance from the SWMU to either the nearest facility property boundary or the nearest residence in the downgradient direction. The maximum value in the distribution was either 1,524 m (5,000 ft) or 1,610 m (5,280 ft or 1 mile). For all onsite facilities, a uniform distribution of well distances was assumed. Table 4-23 presents the data used for the radial distance to the receptor well for the onsite SWMUs. As was done in the case of offsite SWMUs, θ was specified as a uniform distribution from 0° to 90° and LYCHECK was set to "True." The latter condition ensures that the well is always within the lateral plume boundary. Table 4-24 presents the aquifer assignments used for onsite landfills.

For the deterministic analysis, the distance to the well was specified using X-WELL and Y-WELL instead of the radial coordinate system used in Monte Carlo modeling. The central tendency and high-end values of X-WELL were specified as the median value and the 10th percentile value from the corresponding to the uniform distribution (Table 4-23) for radial distance *R* used in the Monte Carlo modeling, respectively. The median value of Y-WELL was defined as half the distance from the plume centerline to the plume boundary. The distance from plume centerline to the boundary is given by Equation 2.4.4b in the EPACMTP User's Guide (U.S. EPA, 1997a). The high-end value of Y-WELL was taken to be on the plume centerline. The central tendency value of Z-WELL was taken to be the median value of all the Z-WELL values generated in the corresponding Monte Carlo runs. No high-end value for Z-WELL was needed because Z-WELL was not a very sensitive parameter.

Table 4-23. Distance to Receptor Well for Onsite SWMUs

Facility Name and Location	SWMU type	Radial Distance to Well (m)		Data Source
		Min	Max	
Degussa Theodore, Alabama	SI	945	1,610	EPA data package U.S. EPA, 2000b
US Antimony Thompson Falls, Montana	LF	335	1,610	EPA data package U.S. EPA, 2000a
Du Pont Delisle Pass Christian, Mississippi	SI	610	1,524	EPA data package U.S. EPA, 2000f
Kerr-McGee New Hamilton, Mississippi	SI	610	1,524	EPA data package U.S. EPA, 2000f
Millennium HPP Baltimore, Maryland	SI/LF	762	1,524	EPA data package U.S. EPA, 2000f

4.3.4 Chemical Properties

Chemical properties used in the analysis include hydrolysis rate constants and the organic carbon partition coefficient K_{OC} for the organic constituents (cyanide and acetonitrile) and soil-water partition coefficients for metals. These were collected from measured literature values as available.

4.3.4.1 Cyanide Hydrolysis and K_d. For this analysis, subsurface chemical hydrolysis of cyanide was accounted for in the modeling, and, because of a lack of reliable data, cyanide biodegradation was assumed to be zero. Free cyanide exists as HCN at neutral to acid pH (e.g., the pK_a of HCN is 9.3), and, for the groundwater modeling, it was assumed that during subsurface transport, cyanide would neither volatilize nor form metallic cyanide complexes. Site-specific hydrolysis rates for cyanide were calculated from chemical-specific hydrolysis rate constants and soil and aquifer temperature and pH values as explained in Section 4.3.1.1. Although the model can also account for the formation and subsequent fate and transport of toxic daughter products, the daughter products of cyanide are harmless (CO_2 and NH_3). The chemical hydrolysis rate constants for cyanide were obtained from Kollig (1993); the neutral rate constant is 29 yr^{-1} and the acid and base rate constants are both zero.

The groundwater pathway analysis accounts for equilibrium sorption of waste constituents onto the soil and aquifer materials using a partition coefficient K_d. Cyanide was assumed not to adsorb to subsurface materials (K_{OC} = 0 and therefore K_d = 0); details of how K_d is calculated for organic constituents are given in Section 4.3.1.1. Because HCN is a non-ionic species, it can be assumed to behave similarly to a conservative anionic tracer with respect to sorption (Kollig, 1993).

Table 4-24. Aquifer Assignments: Onsite Landfills and Surface Impoundments

Facility	GW Temp (°C)	Inputs		Background Information					
		HGDB Hydrogeologic Environment		Heath Hydrogeologic Region	Productive Aquifers	Surficial Geology	DRASTIC Setting		Site-Specific Data
Degussa Corporation	22	10	Unconsolidated and semiconsolidated shallow aquifers	Atlantic and Gulf Coastal Plain	Semiconsoli-dated sand aquifers	Backshore deposits [bm]	10Ab	Unconsolidated & semi-consolidated shallow surficial aquifer	Pliocene/ Miocene (Citronelle?) formation; coastal lowlands aquifer system
Du Pont Memphis	17	6	River valleys and floodplains with overbank deposits	Atlantic and Gulf Coastal Plain	Unconsolidated sand and gravel	Floodplain and alluvial gravel terraces [al]	10Ba	River alluvium with overbank deposits	Quaternary alluvium and terrace deposits (based on soils and surficial geology)
Du Pont New Johnsonville	14	6	River valleys and floodplains with overbank deposits	Nonglaciated Central	Unconsolidated sand and gravel	Floodplain and alluvial gravel terraces [al]	6Fa	River alluvium with overbank deposits	Quaternary alluvium and terrace deposits (based on soils and surficial geology)

(continued)

Table 4-24. (continued)

Facility	GW Temp (°C)	Inputs		Background Information					
		HGDB Hydrogeologic Environment		Heath Hydrogeologic Region	Productive Aquifers	Surficial Geology	DRASTIC Setting		Site-Specific Data
Du Pont Delisle, MS	21	10	Unconsolidated and semiconsolidated shallow aquifers	Atlantic and Gulf Coastal Plain	Unconsolidated sand and gravel	Loam, texture variable (sand to clay) [rl]	10Ab	Unconsolidated & semi-consolidated shallow surficial aquifer	Thick permeable silty sand unit (>200' thick)
Kerr-McGee Hamilton, MS	18	7	River valleys and floodplains without overbank deposits	Atlantic and Gulf Coastal Plain	Unconsolidated watercourse	Floodplain and alluvial gravel terraces [al]	10Bb	River alluvium without overbank deposits	Quaternary alluvium and terrace deposits
Millennium HPP, Baltimore, MD	13	10	Unconsolidated and semiconsolidated shallow aquifers	Atlantic and Gulf Coastal Plain	Unconsolidated sand and gravel	Backshore deposits [bm]	10Ab	Unconsolidated and semiconsolidated shallow surficial aquifer	Variable tertiary alluvial deposits
U.S. Antimony, MT	10	5	Alluvial basins, valleys, and fans	Western Mountain Ranges	No aquifer mapped	Stony colluvium on metamorphic rocks [co/m]	1Ba	Alluvial mountain valleys - east	“Clean" gravel or "fine" gravel aquifer

Note: HGDB data and assignments used only for DeLisle and Millennium HPP. See Table 4-18 for site-specific aquifer data used in analysis.

4.3.4.2 Acetonitrile Hydrolysis and K_d. For this analysis, subsurface chemical hydrolysis of acetonitrile (methyl cyanide) was accounted for in the modeling, and, because of a lack of reliable data, acetonitrile biodegradation was assumed to be zero.

Although the model can also account for the formation and subsequent fate and transport of toxic daughter products, there is no toxicity data in IRIS or HEAST for two of the daughter products of acetonitrile (acetamide and acetic acid). The reference dose for the third daughter product, ammonia, is very high (0.97 mg/kg-d), indicating a low toxicity. For these reasons, the fate and transport of these daughter products was not modeled. The chemical hydrolysis rate constants for acetonitrile were obtained from Kollig (1993); the base rate constant is 45 yr^{-1} and the acid and neutral rate constants are both zero.

The groundwater pathway analysis accounts for equilibrium sorption of waste constituents onto the soil and aquifer materials using a partition coefficient K_d. The K_{OC} value for acetonitrile was also taken from Kollig (1993) and is 0.193 mL/g, indicating only a slight degree of sorption to organic matter in the aquifer materials. Therefore, acetonitrile can be assumed to behave essentially the same as a conservative species with respect to sorption (Kollig, 1993).

4.3.4.3 Metal Partition Coefficients (K_d values). The metals-modeling methodology in EPACMTP incorporates two options to specify the K_d for a given metal: distributions of values or sorption isotherms. For this analysis, the K_d for metals was defined based on a comprehensive review of literature K_d values. Based on this review, K_d was defined as an empirical distribution when sufficient data are available or a log uniform distribution of values when fewer data are available from the scientific literature. The second option is the automated use of adsorption isotherms, which are expressions of the equilibrium relationship between the aqueous concentration and the sorbed concentration of a metal (or other constituent) at constant temperature. This second option was not used for this analysis because of current modeling limitations for generating metal sorption isotherms.

Methodology. The comprehensive literature review focused on identifying and compiling experimentally derived K_d values for soil and aquifer materials from published literature. Collected values were compiled along with geochemical and measurement parameters most likely to influence the K_d. Details of the literature search and data collection strategy are provided in Appendix I.

A set of criteria were defined for identifying K_d values from the literature. The criteria included:

- Natural soil or aquifer media as opposed to pure mineral phases or treated soils
- Aqueous solutions (extractants) with low ionic strength (0.1 M), low humic material concentrations (< 5 mg/L), and dilute metal concentrations
- Absence of organic chelates (e.g., EDTA)
- pH values in the range of 4 to 10.

Batch leach tests were considered to represent systems closer to equilibrium and were preferred over column tests (when both were available for the same study and soil). The effort also included field studies of soil porewater or measured retardation factors (Rf). If multiple K_d values were reported for the same soil type within a single reference, only one K_d was selected to avoid biasing the data in favor of any one researcher. The value selected was that most closely approximating natural conditions (i.e., unadjusted values on untreated samples using natural extractants).

Distributions based on the collected measured values were used to represent K_d as a distributed variable in the Monte Carlo runs. Two approaches were used to generate these distributions, depending on the availability of data: (1) a rank-order percentile approach was used to formulate empirical probability distributions from available measurements for metals with six or more literature K_d values, or (2) a log uniform distribution, based around the median of available measurements, was used when measurements were not likely to capture natural variability (five or fewer samples). The range of this distribution (three log units or three orders of magnitude) was based on the observation that, for the empirical distributions, the average range of measured values was about three log units.

Results. The empirical K_d distributions used in the Monte Carlo analysis are represented as percentiles in Table 4-25; the actual values used for each metal are provided in Appendix I. The log uniform distributions (applied for antimony, molybdenum, thallium, and vanadium) are shown in Table 4-26.

Uncertainty. K_d is metal-specific as well as system-specific. Depending on the metal and the system conditions, K_d varies over several orders of magnitude, and, to accurately capture the variability in K_d, measurements, an empirical distribution should cover these conditions. The collection of soil/porewater systems chosen for study by various researchers and reported in the literature was not specifically selected to represent the national population of such systems and therefore may not accurately represent the true national distribution of K_ds.

In addition to this sampling uncertainty, there are other potentially significant uncertainties that add to the variability in individual K_d measurements.

- **Detection limits.** Depending on experimental method, metal detection limits may limit the observed maximum K_d value, leading to artificially low estimates..
- **Measurement method.** Experimental methods impact K_d, and some variability in K_d values is due to different measurement approaches. For example, batch tests are more likely to be at equilibrium than field methods, but can produce lower concentrations due to a larger liquid-to-solid ratio.
- **Extractant.** Extraction fluid varies and can impact results (although this effort did limit values to those measured using solutions representing natural systems).
- **Redox conditions.** Uncontrolled or unknown redox conditions can be significant, especially for redox-sensitive metals (e.g., Cr, As, Se) and may not reflect natural systems.

Table 4-25. Empirical Distributions Used to Represent Soil/Water Partition Coefficients (K_d Values) for Metals

Metals	No. of K_d values used	Distribution of K_d										
		0	0.1	0.2	0.3	0.4	0.5	0.6	0.7	0.8	0.9	1
As	35	0.0025	2.38	17.86	42	117	225	1947	2863	3733	7162	31079
B	34	0.033	0.15	0.409	0.86	0.97	1.16	1.35	1.51	2.16	3.54	4.06
Cd	102	1	14	26.3	62.8	133.3	201.5	461.8	670	1000	2200	30080
Mn	12	32	28.8	33	91	120	144	280	424	963	3872	7200
Ni	40	1.5	18	49.5	235	341	441	1263	1790	2750	4510	8751
Pb	39	9	20	100	1159	3428	5310	16973	30000	42250	60000	75401
Se	14	1.085	6.5	10.5	15	18.9	24.4	27.5	30.6	34.5	45	57.4
Zn	40	1.35	34	44	234	1,284	2,020	2,760	4,609	5,574	8,866	41,000

Table 4-26. Log Uniform Distributions Used to Represent Soil/Water Partition Coefficients (K_d Values) for Metals

Metal	Count	Distribution Type	Minimum K_d	Maximum K_d
Mo	5	Log uniform	0.68	682
Sb	2	Log uniform	0.39	393
Tl[a]	0	Log uniform	1	1000
V	2	Log uniform	5.0	5012

[a] Based on sorption study that measured K_d as a function of pH (Loux et al., 1990).

Concentration dependence. Single measured values do not capture the nonlinear dependence of K_d on metal concentration. The literature collection effort attempted to collect only values for dilute solutions, but some degree of nonlinearity may be reflected in the literature values depending on experimental conditions.

The last uncertainty has implications to the basic assumptions of the modeling effort. The use of literature K_d values assumes that the sorption capacity of the soils is adequate to accommodate the modeled metals concentration so that there are always available sorption sites on soil and aquifer particles. Although of potential concern for high concentration or high metal loadings, this assumption is not expected to result in large errors because of compensating effects (i.e., desorption of contaminant from the sorption media once the concentration has passed) and because, for many or most metals in this analysis, concentrations in environmental media tend to be within the linear concentration range.

Finally, the magnitude of the uncertainty in K_d values used in this analysis has a significant metal-dependent component because not all metals were equally represented in the scientific literature. Several metals (e.g., Cd, As, Pb) have been more widely studied, while others, notably Tl and Sb, have very little available data. Assuming that a larger number of measurements covers a broader range of conditions, uncertainty in a particular metal's K_d values should decrease with an increasing number of samples used to create the emprirical data set.

4.3.5 Monte Carlo Input Source Data File for EPACMTP

To define the data inputs to the EPACMTP model to represent regional and site-specific scenarios, the model was revised to accept data from a source data file containing 10,000 rows of specified values (see Section 4.3.1.2). These 10,000 rows of input values were generated prior to modeling in order to maintain appropriate data correlations. This section provides an example of how this data input file was developed. The method described here was adapted for use for each waste stream and waste management unit evaluated in the risk analysis. Table 4-27 shows the parameters in the source data file and how these parameters are related.

Table 4-27. Parameters in the Source Data File and Linkages

Parameters in Source Data File	Linkage
Leachate concentration	Waste sample
Waste bulk density	Waste sample
Waste volume	Waste stream and location
FS ratio	Waste sample
WMUs area	WMUs
Soil texture	Location
Groundwater temperature	Location
Recharge rate	Location and soil texture
Infiltration rate	Location and soil texture
Groundwater pH	Location and soil texture
Unsaturated zone thickness	Aquifer type
Saturated zone thickness	Aquifer type
Aquifer hydraulic conductivity	Aquifer type
Aquifer hydraulic gradient	Aquifer type
Groundwater averaging time	Exposure duration

FS = Finite source ratio.

4.3.5.1 Waste Sample Parameters. The waste stream data used for this analysis were obtained from the EPA sampling and analysis (see Section 3.0). Each set of sample data (leachate concentration, waste concentration, and bulk density) is considered a suite of data and data are, therefore, linked throughout the analysis. Each sample is assumed to have an equal probability for occurrence in the analysis.

Example: For the ammonia recycle filter wastes in the HCN sector managed in an offsite industrial landfill, only one data sample was assumed applicable. The single data set of SPLP results was used for the initial leachate concentration, total waste concentration, and bulk density, which were assumed constant for all 10,000 iterations.

4.3.5.2 Waste Management Unit Parameters. The type of waste management used for each waste stream is determined from the EPA final data package. For offsite landfills (industrial or municipal), a single distribution of landfill areas is assumed. Table 4-28 provides the distribution of landfill areas and the respective probabilities that a particular landfill will fall within that landfill area range. These data were used to compile 10,000 randomly selected landfill areas, which were generated for use as input data for the source data file.

Example: For the ammonia recycle filter wastes in the HCN sector managed in an offsite industrial landfill, the distribution of landfill areas is presented in Table 4-28.

4.3.5.3 Location and/or Soil Texture Parameters. Three major soil textures and their relative percentages were identified by the methods detailed in the previous sections for each location where the waste is managed. If a single location was reported to manage a waste, these percentiles were used to generate a distribution of 10,000 soil texture codes as inputs in the source data file. If multiple sites were reported to manage a waste, the locations were assumed equally likely to occur and the locations were chosen randomly first and then soil codes were chosen according to the distribution of soil textures for the chosen location. These 10,000 soil texture codes are inputs to the source data file and are used to link the soil the soil texture distributions for all locations managing a waste with the soil parameters within the EPACMTP model (saturated hydraulic conductivity, moisture retention parameters α and β, residual water content, saturated water content, percent organic matter, and soil bulk density). In addition, the infiltration and recharge rates associated with both soil texture and climate region are linked to the distribution of 10,000 soil textures and locations. Groundwater temperature is linked only to location and is also entered in the source data file according to the location code, but is identical

Table 4-28. Distribution of Landfill Areas

WMU Area (m^2)	Percentage
4000 - 8090	0.0997
8090 - 20200	0.15
20200 - 60700	0.25
60700 - 194000	0.25
194000 - 420000	0.15
420000 - 9350000	0.1

Note: For very high volume wastes (i.e., in the titanium dioxide sectors),very small landfills have insufficient capacity to hold the waste generated over the 30-year lifetime of the landfill and, thus, are not used in the distribution. The maximum depth of landfills is truncated at 68 m.

for all soil textures. Therefore, all combinations of data associated with a location and/or soil texture are internally consistent in the source data set and are thus maintained as internally consistent in the analysis. For onsite SWMUs, the distribution of distances to receptor wells is specific and the distribution is linked to the location.

Example: For the ammonia recycle filter wastes in the HCN sector managed in an offsite industrial landfill in Anahuac, TX, the distribution of soil textures is presented in Table 4-29.

4.3.5.4 Location and/or Aquifer Parameters. The major aquifer types and their percentages in the region of the SWMU were identified by the previously described methods. This distribution of aquifer types was used to generate a distribution of 10,000 aquifer type codes. These codes are keyed to the aquifer parameters in the HGDB database used by the model. These parameters are

- Hydraulic conductivity of the aquifer
- Hydraulic gradient
- Aquifer thickness
- Vadose zone thickness.

Thus, all aquifer parameters are internally consistent in the model and reflect conditions within the region or at the site of the SWMU managing the waste stream of concern.

Example: For the ammonia recycle filter wastes in the HCN sector managed in an offsite industrial landfill, the distribution of aquifer types is presented in Table 4-30.

Table 4-29. Distribution of Soil Textures for Ammonia Recycle Filter Waste Stream Managed in an Industrial Landfill Anahuac, TX

Soil Texture	Soil Texture Code	Percentage
Silty loam	1	0.32
Sandy loam	2	0.12
Silty clay loam	3	0.55

Table 4-30. Aquifer Types Common in Region of SWMU Managing Ammonia Recycle Filter Waste Stream in an Industrial Landfill, Anahuac, TX

Aquifer Type	Aquifer Type Code	Percentage
Sand and gravel	4	0.5
River valleys/flood plains w/ overbank deposits	6	0.25
River valleys/flood plains w/o overbank deposits	7	0.25

4.3.5.5 Groundwater Averaging Time/ Exposure Duration. For noncarcinogens, the groundwater averaging time was set to a constant value of 9 years in the source data file. However, for carcinogens, the groundwater averaging time was assumed to be the same as the exposure duration used in the risk analysis. Thus, the distribution of exposure duration distribution used in this analysis (as discussed in Section 5.2.1) was used as the distribution of groundwater averaging times. In this way, the values for the groundwater averaging time could be exactly matched to the exposure duration value used in the risk calculation for each iteration of the analysis. The exposure duration distribution data were entered into Crystal Ball and 10,000 randomly selected values were generated for each of the adult and child receptors. These values were included in the source data file as the groundwater averaging time to be used by the EPACMTP model. During the calculation of risk for the adult and child receptors, the identical distribution of 10,000 exposure durations was used, thus maintaining internal consistency in the duration of exposure throughout the risk analysis. These durations are presented in Table 4-31.

4.3.5.6 Summary. The source data file described here maintains internal consistency throughout the analysis and ensures that impossible combinations of data do not occur through the misapplication of site-specific, regional, and national data distributions in the probabilistic analysis. In this way, data that are known to be specific to a site or sample are used only in iterations that apply to that site or sample and no inappropriate combinations occur during Monte Carlo sampling.

Table 4-31. Empirical Distribution of Exposure Durations/ Groundwater Averaging Times used in Risk Analysis (years)

Child Exposure Duration	Adult Exposure Duration	Percentage
3	6	0.25
5	11	0.25
8	18	0.25
13	27	0.15
17	35	0.05
22	49	0.05

4.3.6 Results

The probabilistic results of the groundwater modeling are presented in Tables 4-32 through 4-66. Table 4-67 presents the parameters used in the deterministic analysis for all wastewaters. Tables 4-68 and 4-69 present the deterministic results of the groundwater modeling. These tables present the modeled distributions of residential well concentrations and the corresponding DAFs for each constituent for each waste stream managed in landfills.

Table 4-32. Groundwater Concentrations for Ammonia Recycle Filters Managed in Industrial Landfills

Percentile	Antimony (mg/L)	Arsenic - Non-cancer (mg/L)	Arsenic - Cancer (mg/L)	Cadmium (mg/L)	Nickel (mg/L)	Cyanide (mg/L)
50th	4.91E-05	*5.34E-16*	5.29E-16	9.73E-10	2.34E-12	0.00E+00
75th	3.30E-04	*2.08E-07*	2.08E-07	1.38E-07	1.08E-05	0.00E+00
80th	5.17E-04	*6.63E-07*	6.63E-07	3.46E-07	3.92E-05	0.00E+00
85th	8.72E-04	*2.10E-06*	2.08E-06	8.83E-07	1.20E-04	0.00E+00
90th	1.65E-03	*6.88E-06*	6.88E-06	2.67E-06	4.09E-04	0.00E+00
95th	4.02E-03	*2.82E-05*	2.80E-05	9.76E-06	1.74E-03	3.90E-16
97.5th	8.51E-03	*8.90E-05*	8.87E-05	2.63E-05	4.97E-03	1.16E-12
99th	1.77E-02	*2.72E-04*	2.62E-04	6.38E-05	1.30E-02	3.54E-10

Table 4-33. DAF—Ammonia Filtration Residues Hydrogen Cyanide Sector Managed in Industrial D Landfills

	DAFs				
Percentile	Antimony	Arsenic	Cadmium	Nickel	Cyanide
50th	1.20E+04	7.31E+13	5.05E+06	2.61E+11	1.00E+30
10th	3.57E+02	5.67E+03	4.12E+03	1.49E+03	1.00E+30
5th	1.47E+02	1.40E+03	1.38E+03	3.50E+02	6.16E+15
1st	3.34E+01	1.43E+02	2.38E+02	4.70E+01	6.79E+09

Table 4-34. Groundwater Concentrations for Ammonia Recycle Filters Managed in Municipal Landfills

Percentile	Antimony (mg/L)	Arsenic - Noncancer (mg/L)	Arsenic - Cancer (mg/L)	Cadmium (mg/L)	Nickel (mg/L)	Cyanide (mg/L)
50th	5.42E-05	5.83E-16	5.83E-16	5.03E-09	2.92E-12	0.00E+00
75th	3.61E-04	2.32E-07	2.32E-07	4.45E-07	1.20E-05	0.00E+00
80th	5.68E-04	7.42E-07	7.42E-07	8.23E-07	4.31E-05	0.00E+00
85th	9.60E-04	2.35E-06	2.34E-06	1.65E-06	1.33E-04	0.00E+00
90th	1.81E-03	7.69E-06	7.68E-06	3.43E-06	4.39E-04	0.00E+00
95th	4.34E-03	3.12E-05	3.12E-05	9.89E-06	1.90E-03	4.44E-17
97.5th	8.99E-03	9.92E-05	9.92E-05	2.27E-05	5.37E-03	2.28E-13
99th	1.88E-02	3.14E-04	3.04E-04	5.60E-05	1.30E-02	5.89E-11

Table 4-35. DAF—Ammonia Filtration Residues Hydrogen Cyanide Sector Managed in Municipal Landfills

	DAFs				
Percentile	**Antimony**	**Arsenic—cancer**	**Cadmium**	**Nickel**	**Cyanide**
50th	1.01E+04	7.72E+13	9.13E+06	1.71E+11	1.00E+30
10th	3.04E+02	5.85E+03	1.25E+04	1.14E+03	1.00E+30
5th	1.27E+02	1.44E+03	4.26E+03	2.63E+02	4.91E+15
1st	2.93E+01	1.43E+02	6.63E+02	3.84E+01	3.70E+09

Table 4-36. Groundwater Concentrations—Boron in Feed Gas Filter-Hydrogen Cyanide Sector Managed in Municipal Landfills

Percentile	**Groundwater Concentration (mg/L)**
50th	0.002
75th	0.010
80th	0.013
85th	0.020
90th	0.033
95th	0.064
97.5th	0.119
99th	0.232

Table 4-37. DAF—Feed Gas Filter-Hydrogen Cyanide Sector Managed in Municipal Landfills

Percentile	**DAF**
	Boron
50th	3690
10th	223
5th	115
1st	32

Table 4-38. Groundwater Concentrations—Acetonitrile in Combined Wastewaters Managed in Onsite Surface Impoundment

Percentile	Groundwater Concentration (mg/L)
50th	1.96E-03
75th	5.60E-03
80th	7.15E-03
85th	9.23E-03
90th	1.26E-02
95th	1.84E-02
97.5th	2.71E-02
99th	4.01E-02

Table 4-39. DAFs for Combined Wastewaters Managed in Onsite Surface Impoundment

Percentile	DAF for Acetonitrile
50th	2,708
10th	420
5th	288
1st	132

Table 4-40. Groundwater Concentrations Filter Press Cakes

Percentile	Antimony (mg/L)	Thallium (mg/L)
50th	3.85E-07	1.63E-08
75th	5.16E-06	9.07E-07
80th	8.92E-06	1.69E-06
85th	1.59E-05	3.19E-06
90th	3.20E-05	6.96E-06
95th	7.64E-05	1.63E-05
97.5th	1.55E-04	3.20E-05
99th	3.20E-04	6.77E-05

Table 4-41. DAFs Filter Cakes—Sodium Phosphate Sector Managed in Industrial D Landfills

Percentile	DAFs	
	Antimony	Thallium
50th	49,769	413,046
10th	634	958
5th	250	394
1st	62	92

Table 4-42. Groundwater Concentrations Dust Collector Bags

Percentile	Antimony (mg/L)
50th	2.98E-07
75th	3.56E-06
80th	6.03E-06
85th	1.12E-05
90th	2.39E-05
95th	6.50E-05
97.5th	1.58E-04
99th	4.39E-04

Table 4-43. DAFs Filter Bags—Sodium Phosphate Sector Managed in Industrial D Landfills

Percentile	DAFs
	Antimony
50th	1,035,697
10th	12,952
5th	4751
1st	704

Table 4-44. Groundwater Concentrations Sludge Residues—Sodium Chlorate Sector Managed in Municipal Landfills

Percentile	Arsenic (mg/L)	Lead (mg/L	Manganese (mg/L)	Nickel (mg/L)	Zinc (mg/L)
50th	0.00E+00	0.00E+00	4.27E-11	0.00E+00	0.00E+00
75th	4.31E-09	2.15E-12	3.70E-05	2.14E-10	4.03E-13
80th	2.14E-07	2.44E-10	9.51E-05	9.32E-09	2.71E-10
85th	1.70E-06	5.28E-08	2.27E-04	1.73E-07	2.43E-08
90th	9.44E-06	4.49E-06	5.59E-04	1.70E-06	3.61E-06
95th	5.66E-05	7.95E-05	1.58E-03	1.75E-05	1.01E-04
97.5th	1.47E-04	2.91E-04	3.61E-03	6.32E-05	6.36E-04
99th	3.98E-04	9.98E-04	8.43E-03	3.20E-04	2.70E-03

Table 4-45. DAFs Sludge Residues—Sodium Chlorate Sector Managed in Municipal Landfills

	DAFs				
Percentile	Arsenic	Manganese	Nickel	Lead	Zinc
50th	>1E+30	1.16E+9	>1E+30	>1E+30	>1E+30
25th	2.04E+5	1.15E+4	1.41E+8	2.26E+10	3.18E+13
15th	1897	1956.94	2.79E+5	3.16E+5	2.03E+8
10th	437	797	34648	7685	6.43E+5
5th	110	254	4690	369	10868
1st	18	56	610	22	650

Table 4-46. Groundwater Concentrations Filter Residues—Sodium Chlorate Sector Managed in Municipal Landfills

Percentile	Antimony	Arsenic	Cadmium
	Groundwater Conc. (mg/L)	Groundwater Conc. (mg/L)	Groundwater Conc. (mg/L)
50th	0.00E+00	0.00E+00	0.00E+00
75th	6.84E-09	4.31E-12	4.20E-09
80th	3.40E-08	1.04E-10	3.81E-08
85th	1.04E-07	6.93E-09	1.88E-07
90th	3.89E-07	9.60E-08	7.85E-07
95th	1.83E-06	8.64E-07	3.47E-06
97.5th	5.57E-06	3.60E-06	9.89E-06
99th	1.63E-05	1.43E-05	2.56E-05

Table 4-47. DAFs and Risk Results Filter Residues—Sodium Chlorate Sector Managed in Municipal Landfills

Percentile	DAFs		
	Arsenic	Antimony	Cadmium
50th	>1E+30	>1E+30	>1E+30
25th	6.44E+8	2.16E+6	9.08E+6
15th	4.25E+5	1.62E+5	1.73E+5
10th	5.14E+4	3.53E+4	3.69E+4
5th	6511	7630	9584
1st	373	901	1005

Table 4-48. Groundwater Modeling Results for Arsenic in Filter Wastes in Sodium Chlorate Sector Managed in Industrial D Landfill, Perdue Hill, AL

Groundwater		DAF	
Percentile	**Concentration (mg/L)**	**Percentile**	**DAF**
50th	0.00E+00	50th	1E+30
75th	1.99E-09		
80th	1.31E-08		
85th	5.79E-08		
90th	2.11E-07	10th	12905
95th	8.55E-07	5th	3217
97.5th	2.49E-06		
99th	7.12E-06	1st	401

Table 4-49. Groundwater Modeling Results for Sulfate Process Digestion Sludge Managed in Onsite Industrial D Landfill, Baltimore, MD

	Antimony	Vanadium
Percentile	**Groundwater Concentration (mg/L)**	**Groundwater Concentration (mg/L)**
50th	3.37E-04	2.05E-05
75th	1.38E-03	1.31E-03
80th	1.72E-03	2.44E-03
85th	2.15E-03	4.33E-03
90th	2.73E-03	7.98E-03
95th	3.53E-03	1.50E-02
97.5th	4.19E-03	2.28E-02
99th	4.79E-03	3.28E-02
Central Tendency	1.11E-03	2.45E-04
High End Full Distribution	1.41E-03	1.33E-02
High End Half Distribution	NA	NA

Table 4-50. DAFs for Sulfate Process Digestion Sludge Managed in Onsite Industrial D Landfill, Baltimore, MD

Percentile	DAF for Antimony	DAF for Vanadium
50th	68	20,493
10th	8	53
5th	7	28
1st	5	13
Central Tendency	21	1,714
High End Full Distribution	16	31
High End Half Distribution	NA	NA

Table 4-51. Groundwater Modeling Results for Sulfate Process Secondary Gypsum Managed in Onsite Industrial D Landfill, Baltimore, MD

	Antimony	Arsenic	Manganese
Percentile	Groundwater Concentration (mg/L)	Groundwater Concentration (mg/L)	Groundwater Concentration (mg/L)
50th	8.92E-04	3.63E-12	4.01E-03
75th	3.57E-03	6.03E-06	7.05E-02
80th	4.42E-03	1.86E-05	1.13E-01
85th	5.52E-03	5.19E-05	1.70E-01
90th	6.93E-03	1.25E-04	2.49E-01
95th	8.85E-03	2.45E-04	3.54E-01
97.5th	1.05E-02	3.26E-04	4.32E-01
99th	1.20E-02	4.03E-04	5.26E-01
Central Tendency	2.84E-03	1.40E-07	4.29E-02
High End Full Distribution	6.70E-03	1.67E-04	1.80E-01
High End Half Distribution	6.70E-03	1.67E-04	2.51E-01

Table 4-52. DAFs for Sulfate Process Secondary Gypsum Managed in Onsite Industrial D Landfill, Baltimore, MD

Percentile	DAF for Antimony	DAF for Arsenic	DAF for Manganese
50th	62	481,826,441	774
10th	8	14	12
5th	6	7	9
1st	5	4	6
Central Tendency	19	12,491	72
High End Full Distribution	8	10	17
High End Half Distribution	8	10	12

Table 4-53. Groundwater Modeling Results for Milling Sand Managed in Offsite Industrial D Landfill

Groundwater		DAF	
Percentile	Concentration (mg/L)	Percentile	DAF
50th	1.87E-06	50th	12851
75th	1.26E-05		
80th	1.94E-05		
85th	3.32E-05		
90th	6.42E-05	10th	374
95th	1.59E-04	5th	151
97.5th	3.32E-04		
99th	7.04E-04	1st	24
Central Tendency	6.31E-06	Central Tendency	3803
High End Full Distribution	6.39E-06	High End Full Distribution	3754
High End Half Distribution	NA	High End Half Distribution	NA

Table 4-54. Groundwater Modeling Results for Lead in Off-Specification Product Managed in Offsite Municipal Landfill

Percentile	Groundwater Concentration (mg/L)	Percentile	DAF
50th	0.00E+00	50th	1E+30
75th	1.50E-20		
80th	6.52E-16		
85th	1.53E-12		
90th	2.46E-08	10th	2.44E+6
95th	1.12E-06	5th	53,701
97.5th	7.21E-06		
99th	3.29E-05	1st	1,824
Central Tendency	0.00E+00	Central Tendency	1E+30
High End Full Distribution	1.71E-05	High End Full Distribution	3,511
High End Half Distribution	NA	High End Half Distribution	NA

Table 4-55. Groundwater Modeling Results for Chloride Sulfate WWT Sludge Managed in Onsite Industrial D Landfill

	Manganese	Thallium
Percentile	Groundwater Concentration (mg/L)	Groundwater Concentration (mg/L)
50th	4.84E-03	4.89E-05
75th	1.06E-01	3.10E-04
80th	1.71E-01	4.00E-04
85th	2.59E-01	4.89E-04
90th	3.67E-01	5.84E-04
95th	5.10E-01	6.98E-04
97.5th	6.16E-01	7.90E-04
99th	7.06E-01	8.95E-04

Table 4-56. DAFs for Chloride Sulfate WWT Sludge Managed in Onsite Industrial D Landfill

Percentile	DAF for Manganese	DAF for Thallium
50th	543	61
10th	7	5
5th	5	4
1st	4	3

Table 4-57. Groundwater Modeling Results for Ilmenite Process WWT Sludge Managed in Offsite Industrial D Landfill

	Arsenic as a Carcinogen	Antimony	Manganese	Thallium
Percentile	Groundwater Concentration (mg/L)	Groundwater Concentration (mg/L)	Groundwater Concentration (mg/L)	Groundwater Concentration (mg/L)
50th	1.05E-19	3.03E-04	2.64E-04	7.14E-05
75th	4.56E-07	1.83E-03	1.98E-01	8.49E-04
80th	3.77E-06	2.54E-03	4.27E-01	1.35E-03
85th	1.56E-05	3.40E-03	9.66E-01	2.03E-03
90th	6.30E-05	4.72E-03	2.10E+00	3.07E-03
95th	2.35E-04	7.01E-03	4.17E+00	4.66E-03
97.5th	4.00E-04	9.33E-03	6.06E+00	6.09E-03
99th	5.67E-04	1.22E-02	8.84E+00	7.90E-03
Central Tendency	4.10E-10	5.84E-04	3.05E-02	NA
High End Full Distribution	3.41E-05	5.81E-04	5.31E-01	NA
High End Half Distribution	7.36E-05	4.07E-03	2.41E+00	NA

Table 4-58. DAFs for Ilmenite Process WWT Sludge Managed in Offsite Industrial D Landfill

Percentile	DAF for Arsenic	DAF for Antimony	DAF for Manganese	DAF for Thallium
50th	9.50E+15	66	61,801	168
10th	16	4	8	4
5th	4	3	4	3
1st	2	2	2	2
Central Tendency	2,441,406	34	535	35
High End Full Distribution	29	34	31	10
High End Half Distribution	14	5	7	5

Table 4-59. Groundwater Modeling Results for Chloride Sulfate Process WW Managed in Onsite Surface Impoundment—Kerr McGee

	Arsenic as a Carcinogen	Antimony	Molybdenum	Thallium
Percentile	Groundwater Concentration (mg/L)	Groundwater Concentration (mg/L)	Groundwater Concentration (mg/L)	Groundwater Concentration (mg/L)
50th	6.85E-13	1.43E-04	4.40E-04	3.19E-06
75th	2.64E-07	6.71E-04	2.13E-03	1.60E-05
80th	7.87E-07	9.73E-04	3.09E-03	2.33E-05
85th	1.90E-06	1.53E-03	4.92E-03	3.73E-05
90th	4.51E-06	2.55E-03	8.45E-03	6.55E-05
95th	1.34E-05	4.92E-03	1.76E-02	1.39E-04
97.5th	3.41E-05	7.34E-03	2.89E-02	2.42E-04
99th	8.72E-05	1.04E-02	4.60E-02	4.11E-04
Central Tendency	8.8E-7	6.9E-04	2.1E-03	1.6E-05
High End Half Distribution	6.3E-05	5.1E-03	1.7E-02	1.9E-04

Table 4-60. DAFs for Chloride Sulfate Process WW Managed in Onsite Surface Impoundment—Kerr McGee

Percentile	DAF for Arsenic	DAF for Antimony	DAF for Molybdenum	DAF for Thallium
50th	1.46E+09	307	523	784
10th	222	17	27	38
5th	75	9	13	18
1st	11	4	5	6
Central Tendency	1140	64	110	161
High End Half Distribution	16	9	14	13

Table 4-61. Groundwater Modeling Results for Chloride Sulfate Process WW Managed in Onsite Surface Impoundment—Millennium HPP

	Arsenic as a Carcinogen	Manganese
Percentile	Groundwater Concentration (mg/L)	Groundwater Concentration (mg/L)
50th	1.34E-15	1.74E-04
75th	2.37E-07	5.82E-03
80th	1.01E-06	9.38E-03
85th	3.26E-06	1.51E-02
90th	8.58E-06	2.41E-02
95th	3.14E-05	4.05E-02
97.5th	8.87E-05	5.64E-02
99th	2.49E-04	8.28E-02
Central Tendency	4.0E-09	2.4E-03
High End Half Distribution	3.7E-05	1.4E-02

Table 4-62. DAFs for Chloride Sulfate Process WW Managed in Onsite Surface Impoundment—Millennium HPP

Percentile	DAF for Arsenic	DAF for Manganese
50th	1.87E+12	57,332
10th	291	413
5th	80	246
1st	10	120
Central Tendency	6.2E+05	4181
High End Half Distribution	68	706

Table 4-63. Groundwater Modeling Results for Ilmenite Process WW Managed in an Onsite Surface Impoundment—Du Pont Delisle

	Manganese	Thallium	Vanadium
Percentile	Groundwater Concentration (mg/L)	Groundwater Concentration (mg/L)	Groundwater Concentration (mg/L)
50th	6.73E-06	1.82E-07	1.82E-07
75th	8.77E-05	1.60E-06	1.60E-06
80th	1.40E-04	2.49E-06	2.49E-06
85th	2.24E-04	4.19E-06	4.19E-06
90th	3.73E-04	7.93E-06	7.93E-06
95th	7.52E-04	1.84E-05	1.84E-05
97.5th	1.41E-03	3.61E-05	3.61E-05
99th	2.57E-03	7.95E-05	7.95E-05
Central Tendency	2.3E-06	1.0E-08	1.6E-09
High End Half Distribution	4.9E-05	2.4E-07	1.0E-05

Table 4-64. DAFs for Ilmenite Process WW Managed in Onsite Surface Impoundment—Du Pont Delisle

Percentile	DAF for Manganese	DAF for Thallium	DAF for Vanadium
50th	492,772	30,919	1,309,069
10th	8,859	865	4,466
5th	4,403	410	1,965
1st	1,300	111	509
Central Tendency	1,434,783	240,385	11,464,968
High End Half Distribution	67,073	10,373	60,577

Table 4-65. Groundwater Modeling Results for Low Antimony Slag Managed in an Onsite Landfill—Thompson Falls, MT

Percentile	Antimony Groundwater Concentration (mg/L)	Arsenic Groundwater Concentration (mg/L)	Boron Groundwater Concentration (mg/L)	Selenium Groundwater Concentration (mg/L)	Vanadium Groundwater Concentration (mg/L)
50th	3.04E-03	0.00E+00	2.90E-04	8.86E-06	1.77E-08
75th	1.39E-02	4.03E-06	9.57E-04	3.10E-05	2.44E-06
80th	1.93E-02	1.15E-05	1.28E-03	4.11E-05	4.66E-06
85th	2.85E-02	2.74E-05	1.76E-03	5.61E-05	8.75E-06
90th	4.60E-02	6.94E-05	2.63E-03	8.69E-05	1.86E-05
95th	9.43E-02	2.02E-04	4.87E-03	1.68E-04	5.13E-05
97.5th	1.67E-01	4.47E-04	8.41E-03	2.85E-04	1.07E-04
99th	3.38E-01	1.02E-03	1.59E-02	5.64E-04	2.25E-04

Table 4-66. Groundwater DAFs for Low Antimony Slag Managed in an Onsite Landfill—Thompson Falls, MT

Percentile	DAF for Antimony	DAF for Arsenic	DAF for Boron	DAF for Selenium	DAF for Vanadium
50th	50,108	1.00E+30	29,707	49,104	5.96E+07
10th	3,811	48,729	3,216	4,335	56,395
5th	1,960	16,314	1,724	2,203	20,520
1st	581	3,451	520	651	4,631

Table 4-67. Parameter Values Used in the Deterministic Analysis

Waste Stream	Constituent	CT or HE	Aquifer Thickness (m)	Aquifer Sat K (m/yr)	Hydraulic Gradient (m/m)	Aquifer K_d (cm^3/g)	WMU Area (m^2)	Infiltration Rate (m/yr)	Saturated Hydraulic Conductivity (m/yr)	Unsaturated Zone Thickness (m)	Unsaturated Zone K_d (cm^3/g)	X- Well (m)	Y-Well (m)	Z-Well (m)	Highest Average Receptor Well Conc. (mg/L)	Initial Leachate Conc. (mg/L)	DAF[1]
NaClO3 Sludge	Arsenic	CT	9.14	595	0.0100	225.0	61000	0.261	11.61	4.57	225.00	427.0	117.9	3.32	3.88E-07	0.0025	6,448
	Arsenic	HE	9.14	595	0.0100	225.0	61000	0.353	11.61	4.57	2.60	427.0	117.9	3.32	2.09E-06	0.0025	1,198
TiO2 Sulfate Process Gypsum Millenium HPP	Arsenic	CT	7.62	284	0.0050	225.0	384000	0.201	11.37	1.50	225.0	1140.0	270.0	2.53	1.40E-07	0.00175	12,491
	Arsenic	HE	7.62	284	0.0050	2.6	384000	0.201	11.37	1.50	2.6	1140.0	270.0	2.53	1.67E-04	0.00175	10
	Manganese	CT	7.62	284	0.0050	113.0	384000	0.201	11.37	1.50	113.0	1140.0	270.0	2.53	4.29E-02	3.10000	72
	Manganese	HE	7.62	284	0.0050	34.6	384000	0.201	11.37	1.50	113.0	838.0	254.0	2.53	2.51E-01	3.10000	12
	Antimony	CT	7.62	284	0.0050	12.4	384000	0.201	11.37	1.50	12.4	1140.0	270.0	2.53	2.84E-03	0.05500	19
	Antimony	HE	7.62	284	0.0050	0.8	384000	0.201	11.37	1.50	12.4	1140.0	0.0	2.53	6.70E-03	0.05500	8
Off-Spec TiO2 DuPont New Johnsonville	Lead	CT	9.14	1550	0.0050	5310.0	61600	0.467	7.57	3.96	5310.0	427.0	118.0	3.09	0	0.06000	1.E+30
	Lead	HE	9.14	1550	0.0050	20.0	61600	0.467	7.57	3.96	20.0	427.0	118.0	3.09	1.71E-05	0.06000	3,511
Cl&SO4 Milling Sand Kemira	Antimony	CT	7.62	374	0.0050	12.4	61200	0.261	26.40	3.81	12.4	427.0	118.0	2.68	6.31E-06	0.02400	3,803
	Antimony	HE	7.62	374	0.0050	12.4	61200	0.261	26.40	1.52	0.8	427.0	118.0	2.68	6.39E-06	0.02400	3,754
Surfate Digestion Sludge Millenium HPP	Antimony	CT	7.62	284	0.0050	12.4	384000	0.201	11.37	1.50	12.4	1140.0	270.0	2.53	1.11E-03	0.02300	21
	Antimony	HE	7.62	284	0.0050	12.4	384000	0.201	11.37	1.50	0.8	1140.0	0.0	2.53	1.41E-03	0.02300	16
	Vanadium	CT	7.62	284	0.0050	158.0	384000	0.201	11.37	1.50	158.3	1140.0	270.0	2.53	2.45E-04	0.42000	1,714
	Vanadium	HE	7.62	284	0.0050	10.0	384000	0.201	11.37	1.50	10.0	1140.0	270.0	2.53	1.33E-02	0.42000	31
Ilmenite WWT Solids DuPont Edgemoor	Antimony	CT	10.70	189	0.0100	12.4	60000	0.201	22.61	5.79	12.4	427.0	117.0	3.39	5.84E-04	0.02000	34
	Antimony	HE	10.70	189	0.0100	12.4	60000	0.201	22.61	5.79	12.38	104.0	0.0	3.39	4.07E-03	0.02000	5
	Arsenic	CT	10.70	189	0.0100	225.0	60000	0.201	22.61	5.79	225.0	427.0	117.0	3.39	4.10E-10	0.00100	2.44e+06
	Arsenic	HE	10.70	189	0.0100	2.6	60000	0.201	22.61	5.79	225.0	104.0	93.5	3.39	7.36E-05	0.00100	14
	Manganese	CT	10.70	189	0.0100	113.0	60000	0.201	22.61	5.79	113.0	427.0	117.0	3.39	3.05E-02	16.3000	535
	Manganese	HE	10.70	189	0.0100	34.6	415000	0.201	22.61	5.79	113.0	427.0	237.0	3.39	2.41E+00	16.3000	7
	Thallium	CT	10.70	189	0.0100	na	60000	0.201	22.61	5.79	na	427.0	117.0	3.39	3.46E-04	0.0120	35
	Thallium	HE	10.70	189	0.0100	na	60000	0.201	22.61	5.79	na	104.0	93.5	3.39	2.42E-03	0.0120	5

(continued)

Table 4-67. (continued)

Waste Stream	Constituent	CT or HE	Aquifer Thickness (m)	Aquifer Sat K (m/yr)	Hydraulic Gradient (m/m)	Aquifer K_d (cm^3/g)	WMU Area (m^2)	Infiltration Rate (m/yr)	Saturated Hydraulic Conductivity (m/yr)	Unsaturated Zone Thickness (m)	Unsaturated Zone K_d (cm^3/g)	X- Well (m)	Y-Well (m)	Z-Well (m)	Highest Average Receptor Well Conc. (mg/L)	Initial Leachate Conc. (mg/L)	DAF[1]
Chloride sulfate WW Kerr McGee SI Scenario	Arsenic	CT	6.48	5480	0.0060	225.0	149000	0.401	22.30	2.84	225	1070	198.0	2.7	8.77E-07	0.0010	1,100
	Arsenic	HE	6.48	5480	0.0060	2.6	149000	0.401	22.30	2.84	3	1070	198.0	2.7	6.28E-05	0.0010	16
	Antimony	CT	6.48	5480	0.0060	12.4	149000	0.401	22.30	2.84	12	1070	198.0	2.7	6.89E-04	0.0440	64
	Antimony	HE	6.48	5480	0.0060	0.8	149000	0.401	22.30	2.84	12	1070	0.0	2.7	5.11E-03	0.0440	9
	Thallium (logUKd)	CT	6.48	5480	0.0060	31.6	149000	0.401	22.30	2.84	32	1070	198.0	2.7	1.55E-05	0.0025	160
	Thallium (logUKd)	HE	6.48	5480	0.0060	2.0	149000	0.401	22.30	2.84	2	1070	198.0	2.7	1.86E-04	0.0025	13
	Mo	CT	6.48	5480	0.0060	21.5	149000	0.401	22.30	2.84	22	1070	198.0	2.7	2.09E-03	0.2300	110
	Mo	HE	6.48	5480	0.0060	1.4	149000	0.401	22.30	2.84	22	1070	0.0	2.7	1.65E-02	0.2300	14
Chloride sulfate WW MHPP SI Scenario	Arsenic	CT	40.00	4320	0.0020	225.0	450000	0.563	11.40	2.49	225	1140	285.0	5.0	4.02E-09	0.0025	620,000
	Arsenic	HE	40.00	4320	0.0020	2.6	450000	0.563	11.40	2.49	3	1140	285.0	5.0	3.69E-05	0.0025	68
	Manganese	CT	40.00	4320	0.0020	113.0	450000	0.563	11.40	2.49	113	1140	285.0	5.0	2.38E-03	9.9500	4,200
	Manganese	HE	40.00	4320	0.0020	34.6	450000	0.563	11.40	2.49	113	1140	0.0	5.0	1.41E-02	9.9500	710
Illmenite WW Delisle SI Scenario	Manganese	CT	76.00	2550	0.0040	113.0	13900	0.923	52.10	1.19	113	1070	118.0	5.0	2.30E-06	3.3000	1,400,000
	Manganese	HE	76.00	2550	0.0040	34.6	13900	0.923	52.10	1.19	113	1070	0.0	5.0	4.92E-05	3.3000	67,000
	Thallium	CT	76.00	2550	0.0040	31.6	13900	0.923	52.10	1.19	32	1070	118.0	5.0	1.04E-08	0.0025	240,000
	Thallium	HE	76.00	2550	0.0040	2.0	13900	0.923	52.10	1.19	32	1070	0.0	5.0	2.41E-07	0.0025	10,000
	Vanadium	CT	76.00	2550	0.0040	158.0	13900	0.923	52.10	1.19	158	1070	118.0	5.0	1.57E-09	0.0180	1.10e+07
	Vanadium	HE	76.00	2550	0.0040	10.0	13900	0.923	52.10	1.19	158	1070	118.0	5.0	1.04E-05	0.6300	60,000

(continued)

Table 4-67. (continued)

Waste Stream	Constituent	CT or HE	Aquifer Thickness (m)	Aquifer Sat K (m/yr)	Hydraulic Gradient (m/m)	Aquifer K_d (cm^3/g)	WMU Area (m^2)	Infiltration Rate (m/yr)	Saturated Hydraulic Conductivity (m/yr)	Unsaturated Zone Thickness (m)	Unsaturated Zone K_d (cm^3/g)	X- Well (m)	Y-Well (m)	Z-Well (m)	Highest Average Receptor Well Conc. (mg/L)	Initial Leachate Conc. (mg/L)	DAF[1]
Antimony oxide low antimony slag	Arsenic	CT	16.50	30,100	0.0050	225.00	809	0.3150	185.00	10.90	225.00	972.0	86.0	4.95	1.58E-06	2.93	1.85e+06
	Arsenic	HE	16.50	30,100	0.0050	2.60	809	0.3150	185.00	10.90	2.60	972.0	86.0	4.95	2.47E-04	2.93	11,900
	Boron	CT	16.50	30,100	0.0050	1.17	809	0.3150	185.00	10.90	1.17	972.0	86.0	4.95	7.13E-04	8.06	11,300
	Boron	HE	16.50	11,400	0.0050	1.17	809	0.3150	185.00	10.90	1.17	972.0	0.0	4.95	6.14E-03	8.06	1,310
	Antimony	CT	16.50	30,100	0.0050	12.40	809	0.3150	185.00	10.90	12.38	972.0	86.0	4.95	6.60E-03	114.00	17,300
	Antimony	HE	16.50	30,100	0.0050	12.40	809	0.3150	185.00	10.90	12.38	972.0	0.0	4.95	1.16E-01	211.00	1,830
	Selenium	CT	16.50	30,100	0.0050	24.80	809	0.3150	185.00	10.90	24.75	972.0	86.0	4.95	2.89E-05	0.33	11,400
	Selenium	HE	16.50	11,400	0.0050	24.80	809	0.3150	185.00	10.90	24.75	972.0	0.0	4.95	2.37E-04	0.33	1,400
	Vanadium	CT	16.50	30,100	0.0050	158.00	809	0.3150	185.00	10.90	158.30	972.0	86.0	4.95	9.59E-06	1.00	104,000
	Vanadium	HE	16.50	30,100	0.0050	9.98	809	0.3150	185.00	10.90	158.30	972.0	0.0	4.95	4.36E-05	1.00	22,900

Table 4-68. Deterministic Groundwater Modeling Results with High-End Parameters, Nonwastewaters Managed in Landfills—Titanium Dioxide Sector

Waste Stream	Constituent of Concern	Central Tendency (CT) or High-End (HE) Case	K_d	Longitudinal Distance to Well (m)	Distance from Plume Centerline to Well (m)	Depth of Unsaturated Zone (m)	Highest Average Receptor Well Concentration (mg/L)	Groundwater DAF (using Avg. Conc.)
Chloride-sulfate process wastewater treatment sludge Millennium HPP	Manganese	CT	113	1140.0	270.0	1.5	0.049	54
		HE (sat Kd & Y-Well)	**34.6**	1140.0	0	1.5	0.37	7
	Thallium	CT	500	1140.0	270.0	1.5	0.00027	11
		HE (Y-Well & Intake Rate)	500	1140.0	0	1.5	0.00039	8
Sulfate process secondary gypsum Millennium HPP	Arsenic	CT	225	1140.0	270.0	1.5	1.40E-07	12,491
		HE (unsat & sat Kd)	**2.6**	1140.0	270.0	1.5	1.67E-04	10
	Manganese	CT	113	1140.0	270.0	1.5	4.29E-02	72
		HE (sat Kd & X-Well)	34.6	838.0	270.0	1.5	2.51E-01	12
	Antimony	CT	12.4	1140.0	270.0	1.5	6.70E-03	8
		HE (sat Kd and Y-Well)	0.8	1140.0	0.0	1.5	6.70E-03	8
Off-spec titanium dioxide Du Pont New Johnsonville	Lead	CT	5310	427.0	118.0	3.96	0.00E+00	1.E+30
		HE (unsat & sat Kd)	20	427.0	118.0	3.96	1.71E-05	3,511
Cl&SO4 milling sand Kemira	Antimony	CT	12.4	427.0	118.0	3.81	6.31E-06	3,803
		HE (unsat K_d &DSOIL)	**0.8**	427.0	118.0	1.52	6.39E-06	3,754

(continued)

Table 4-68. (continued)

Waste Stream	Constituent of Concern	Central Tendency (CT) or High-End (HE) Case	K_d	Longitudinal Distance to Well (m)	Distance from Plume Centerline to Well (m)	Depth of Unsaturated Zone (m)	Highest Average Receptor Well Concentration (mg/L)	Groundwater DAF (using Avg. Conc.)
Sulfate digestion sludge Millenium HPP	Antimony	CT	12.4	1140.0	270.0	1.5	1.11E-03	21
		HE (unsat K_d & Y-Well)	0.8	1140.0	0.0	1.5	1.41E-03	16
	Vanadium	CT	158	1140.0	270.0	1.5	2.45E-04	1,714
		HE (unsat & sat K_d)	**10**	1140.0	270.0	1.5	1.33E-02	31
Ilmenite WWT solids Du Pont Edgemoor	Antimony	CT	12.4	427.0	117.0	5.79	5.84E-04	34
		HE (X & Y-Well)	12.4	**104.0**	0.0	5.79	4.07E-03	5
	Arsenic	CT	225	427.0	117.0	5.79	4.10E-10	2,441,406
		HE (sat K_d & X-Well)	2.6	**104.0**	93.5	5.79	7.36E-05	14
	Manganese	CT	113	427.0	117.0	5.79	3.05E-02	535
		HE (sat K_d & Area)	34.6	427.0	237.0	5.79	2.41E+00	7
	Thallium	CT	500	427.0	117.0	5.79	0.00038	31
		HE (X-Well & Intake Rate)	500	104.0	93.5	5.79	0.0018	7

Bold indicates high-end parameters.

Table 4-69. Deterministic Groundwater Modeling Results with High-End Parameters, Wastewaters Managed in Surface Impoundments—Titanium Dioxide Sector

Waste Stream	Constituent of Concern	Central Tendency (CT) or High-End (HE) Case	K_d	Longitudinal Distance to Well (m)	Distance from Plume Centerline to Well (m)	Depth of Unsaturated Zone (m)	Highest Average Receptor Well Concentration (mg/L)	Groundwater DAF (using Avg. Conc.)
Chloride-sulfate process wastewaters Millennium HPP	Arsenic	CT	225	1140.0	285	5	4.0E-9	6.2E+5
		HE (sat Kd & unsat Kd)	**2.6**	1140.0	285	5	3.7E-5	68
	Manganese	CT	113	1140.0	285	5	0.0024	4181
		HE (sat Kd & Y-Well)	34.6	1140.0	0	5	0.014	706
Chloride-sulfate process wastewaters Kerr McGee	Arsenic	CT	225	1070	198	2.69	8.8E-07	11401
		HE (sat Kd & Y-Well)	**2.6**	1070	0	2.69	6.3E-05	16
	Antimony	CT	12.4	1070	198	2.69	6.9E-4	64
		HE (sat Kd & Y-Well)	0.8	1070	0	2.69	5.1E-03	9
	Molybdenum	CT	21.5	1070	198	2.69	2.1E-03	110
		HE (sat Kd & Y-Well)	1.36	1070	0	2.69	1.7E-02	14
	Thallium	CT	31.6	1070	198	2.69	1.6E-05	8161
		HE (sat K_d & unsat K_d)	2	1070	198	2.69	1.9E-4	13

Bold indicates high end parameters

(continued)

Table 4-69. (continued)

Waste Stream	Constituent of Concern	Central Tendency (CT) or High-End (HE) Case	K_d	Longitudinal Distance to Well (m)	Distance from Plume Centerline to Well (m)	Depth of Unsaturated Zone (m)	Highest Average Receptor Well Concentra-tion (mg/L)	Groundwater DAF (using Avg. Conc.)
Ilmenite process wastewaters Du Pont Delisle	Manganese	CT	113	1070	118	5	5.84E-04	34
		HE (sat K_d & Y-Well)	34.6	1070	0.0	5	4.07E-03	5
	Thallium	CT	225	1070	118	5	4.10E-10	2,441,406
		HE (sat K_d & Y-Well)	2.6	**1070**	0	5	7.36E-05	14
	Vanadium	CT	113	1070	118	5	3.05E-02	535
		HE (sat K_d & leachate concentration)	34.6	1070	118	5	2.41E+00	7

Bold indicates high-end parameters.

4.3.7 Shower Model

Pathways for human exposure to contaminated residential well water are not limited to ingestion as drinking water. There is the potential for exposure to volatile constituents through the inhalation pathway during household water use, especially daily showering. A single constituent, acetonitrile, in the combined wastewaters waste stream in the hydrogen cyanide sector managed in an onsite surface impoundment at the Degussa facility in Theodore, AL, was evaluated for risk through this pathway. The concentration of acetonitrile in the leachate from this surface impoundment was assumed to be 5.3 mg/L based on the measured wastewater concentration of 190 mg/L and dilution of the combined HCN wastewaters by the total wastewater throughput to the surface impoundment. This surface impoundment was reported to be covered with a flexible membrane cover to prevent volatilization of constituents. In addition, the facility reported no biological treatment occurred in this impoundment so no treatment was modeled. Thus, the concentration of acetonitrile measured at the exit from the impoundment was assumed representative of the leachate concentration.

The risks estimated by this model are primarily from exposures during daily showering; however, additional exposures during additional time in the bathroom were considered. The shower parameters used in the model are presented in Table 4-70. Many of these factors were assumed constant in this analysis. Thus, for noncarcinogens, these factors and the physical and chemical properties of the volatile constituent determine the air concentration of each constituent. The air concentration was compared to the reference concentration to yield the hazard quotient. This HQ was assumed to be applicable to adults and children. Table 4-71 presents the RfCs and the physical and chemical properties used in the shower analysis.

4.3.7.1 <u>Methodology: Description of Groundwater, Noningestion Exposure Model for Inhalation</u>. The model used in this analysis is based on the equations presented in McKone (1987). The model estimates the change in the shower air concentration based on the mass of constituent lost by the water (fraction emitted or emission rate) and the air exchange rate between the various model compartments (shower, the rest of the bathroom, and the rest of the house) following the same basic model construct described by Little (1992). The resulting differential equations were solved using finite difference numerical integration.

The basis for estimating the concentration of constituents in the indoor air is based on the mass transfer of constituent from water to shower air.

This equation estimates the overall mass transfer coefficient from tap water to air from showering:

$$K_{ol} = \beta \times \left(\frac{2.5}{D_1^{2/3}} + \frac{1}{D_a^{2/3} H} \right)^{-1} \qquad (4\text{-}26)$$

where

K_{ol} = overall mass transfer coefficient (cm/s)
β = proportionality constant $(cm/s)^{-1/3}$
D_l = diffusion coefficient in water (cm^2/s)
D_a = diffusion coefficient in air (cm^2/s)
H' = dimensionless Henry's law constant ($=41*H_{LC}$).

Table 4-70. Shower Model Parameters

Parameter	Value
Shower rate	5.5 L/min
Shower time	10.0 min
Shower volume	2.00 m^3
Bathroom volume	10.0 m^3
Sh/B vent rate	100 L/min
Nozzle velocity	400 cm/s
Drop diameter	0.098 cm
Nozzle height	1.8 m
Time in shower	15.0 min

Table 4-71. RfC and Physical and Chemical Properties for Acetonitrile

Constituent	RfC (mg/m^3)	Henry's Law Constant ($atm\text{-}m^3/mol$)	Diffusion Coefficient in Air	Diffusion Coefficient in Water
Acetonitrile	0.06	2.9E-05	1.7E-05	1.3E-01

The constituent emission rate is estimated from the change in the shower water concentration as the water falls, which is calculated using the overall mass transfer coefficient as follows:

$$\delta c/\delta t = -K_{ol}(A/V)(c - y_s/H') \quad (4\text{-}27)$$

where

c	=	liquid phase (droplet) constituent concentration (µg/cm^3 or mg/L)
t	=	time (s)
A	=	total surface area for mass transfer (cm^2)
V	=	total volume of water within the shower compartment (cm^3)
y_s	=	gas phase constituent concentration in the shower (µg/cm^3 or mg/L)
H	=	dimensionless Henry's law constant.

Consequently, in addition to the overall mass transfer coefficient, the emission rate of a contaminant within the shower is dependent on the surface-area-to-volume ratio of the shower water (within the shower) and the concentration driving force between the water and the shower air.

The shower emissions can be modeled based on falling droplets as a means of estimating the surface-area-to-volume ratio for mass transfer and the residence time of the water in the shower compartment. Equation 4-27 can then be integrated assuming the compound concentration in the gas phase is constant over the time frame of the droplet fall. The time required for a droplet to fall equals the nozzle height divided by the water droplet velocity. The ratio of the surface area to volume for the droplet is calculated as $6/d_p$ (i.e., by assuming a spherical shape). By assuming the drops fall at terminal velocity, the surface-area-to-volume ratio and the residence time can be determined based solely on droplet size. A droplet size of approximately 1 mm (0.1 cm) was selected. The terminal velocity for the selected droplet size was approximately 400 cm/s. The fraction of constituent emitted from a water droplet at any given time can then be calculated by integrating Equation 4-27 and rearranging as follows:

$$f_{em} = 1 - C_{out}/c_{in} = (1 - f_{sat})(1 - e^{-N}) \quad (4\text{-}28)$$

where

f_{em}	=	fraction of constituent emitted from the droplet (dimensionless)
c_{out}	=	droplet constituent concentration at shower floor/drain (mg/L)
c_{in}	=	droplet constituent concentration entering the shower (mg/L)
f_{sat}	=	$y_s/(H\ c_{in})$ = fraction of gas phase saturation (dimensionless)
N	=	dimensionless overall mass transfer coefficient = $K_{ol}\,(6/d_p)\,(h/v_t)$
d_p	=	droplet diameter = 0.1 (cm)
h	=	nozzle height (cm)
v_t	=	terminal velocity of droplet = 400 (cm/s).

The gas phase constituent concentration in the shower was then calculated for each time step for the duration of the shower. The air exchange rate between the shower and the bathroom was included in the estimation of the gas phase concentration of the constituents in the shower.

$$y_{s,t+1} = y_{s,t} + [Q_{gs} \times (y_{b,t} - y_{s,t}) \times (t_{t+1} - t_t) + E_{s,t}]/V_s \quad (4\text{-}29)$$

where

$y_{s,t+1}$ = gas phase constituent concentration in the shower at the end of time step (mg/L)

$y_{s,t}$ = gas phase constituent concentration in the shower at the beginning of time step (mg/L)

Q_{gs} = volumetric gas exchange rate between shower and bathroom (L/min)

$y_{b,t}$ = gas phase constituent concentration in the bathroom at the beginning of time step (mg/L)

$(t_{t+1}-t_t)$ = calculation time step

$E_{s,t}$ = mass of constituent emitted from shower between time t and time t+1 (mg)

V_s = volume of shower stall (L).

The shower model also provides direct estimates of the bathroom and whole house exposure. The risk from inhalation exposures in the remainder of the house is generally several orders of magnitude less than the risk from inhalation exposures in the bathroom and during showering.

4.3.7.2 Exposure Factors. Where available, the exposure parameters used in this analysis are values cited in the *Exposure Factors Handbook* (U.S. EPA, 1997b). The remaining exposure factors required for this analysis were obtained from McKone (1987). The original articles were obtained to verify the values used in the analysis. Parameter values are presented in Table 4-72.

The equation used to estimate a hazard quotient from inhalation is expressed as

$$HQ = \frac{C_{air}}{RfC} \quad (4\text{-}29)$$

where

HQ = hazard quotient (unitless)
C_{air} = average concentration of constituent in air (mg/m^3)
RfC = reference concentration (mg/m^3).

Table 4-72. Exposure Input Parameters for Inhalation

	Adult		Child	
Parameter	**CT**	**High End**	**CT**	**High End**
Event frequency (event/d)	1	1	1	1
Exposure frequency (d/yr)	350	350	350	350
Exposure duration(yr)	9	30	7.3	8
Body weight (kg)	70	70	16	21

CT = Central tendency.

5.0 Exposure Assessment

The purpose of exposure assessment is to estimate the dose to each receptor by combining modeled constituent of concern concentrations for key media with relevant intake rates for the individuals being modeled. The inorganic chemical manufacturing waste listing risk assessment focused on chronic cancer and noncancer risk resulting from tap water exposures. Consequently, for this analysis, exposure assessment involved combining modeled residential well concentrations with adult and child tap water ingestion rates and exposure durations to generate both average daily dose estimates for noncarcinogens and lifetime averaged daily dose estimates for carcinogens.

For all WS/SWMU/CoC combinations evaluated in this analysis, groundwater was assumed to be contaminated from CoCs leaching from the SWMU, through the vadose zone, into the underlying aquifer, and migrating downgradient to the offsite residential well location. It was further assumed that the groundwater well was used as the sole source of tap water for the adults and children living in that residence.

The risk was estimated for each individual chemical in each waste stream. However, the risk associated with aggregate exposures to multiple chemicals could occur if multiple CoCs were present in a waste stream, reached the residential well within the same time frame, and had the same toxicological endpoint. The time it takes for CoCs to reach a residential well is affected by a number of chemical-specific properties, notably the partition coefficient (K_d); thus, the decision to aggregate cancer or noncancer risks or hazards for different CoCs was based on the potential for temporal overlap of the concentration plateaus for those CoCs. Toxicological endpoints vary from constituent to constituent, so the decision to aggregate cancer or noncancer risk or hazards for different CoCs was also based on the potential for additive effects at the same target organ.

5.1 Human Receptors

Both child and adult residents were modeled in the inorganics chemical manufacturing waste listing risk analysis. For cancer risk, the child resident was modeled as a 1- to 6-yr-old, with a variable starting age (for exposure) and cohort aging when applicable (i.e., in assessing lifetime average exposures where the duration of exposure must be considered). The 1- to 6-yr-old cohort was selected as the initial cohort because this age group corresponds to the youngest cohort for which exposure duration variability data were available. In addition, the 1- to 6-yr-old child cohort generally experiences a higher exposure level relative to older children because of the high intake-to-body-weight ratio for the 1- to 6-yr-old. Thus, the estimates made for the younger child were also protective for older children.

For noncancer risk, the younger child (1- to 6-yr-old age range) was modeled. Note: The use of the 1- to 6-yr-old child cohort in this analysis excluded exposures in the first year of life.

The adult resident was modeled as an individual between 20 and 64 years of age. Cohort aging was not considered in modeling exposure for the adult resident, because it was expected to play a less significant role in determining overall exposure for the adult receptor compared to the child receptor. For noncancer risks, all individuals within all age ranges were considered.

5.2 Exposure Parameter Variability Distributions Used in Probabilistic Analysis

The probabilistic analysis requires exposure parameter variability distributions for exposure duration and tap water ingestion rates. Specifically, exposure duration variability distributions were required for the 1- to 6-yr-old child and 20- to 64-yr-old adult resident cohorts.

5.2.1 Exposure Duration

Exposure duration variability was characterized using discrete distributions based on percentile data obtained from the 1997 *Exposure Factors Handbook* (EFH, U.S. EPA, 1997). The decision to use discrete exposure duration variability distributions in characterizing variability in **exposure duration** rather than developing continuous distributions is based on the following considerations:

- The goal of avoiding uncertainty associated with fitting statistical models to percentile data. No recommendation is provided in the EFH for fitting a distribution to exposure duration data, even though percentile data characterizing interindividual variability are presented in the 1997 EFH for this exposure parameter. Consequently, uncertainty would be associated with fitting statistical models to the exposure duration variability data, because statistical parameters (e.g., geometric means and geometric standard deviations) based on the underlying study data were not available. By using the percentile data "as is" to develop discrete variability distributions, uncertainty associated with fitting statistical models was avoided.

- The goal of ensuring that the high end of the distribution was well represented. The percentile data for exposure duration include 90^{th}, 95^{th} and 99^{th} percentile values, which increases confidence that the upper end of the exposure parameter variability distribution was reflected with reasonable accuracy in the exposure assessment.

However, uncertainty was introduced into the analysis through the use of discrete variability distributions for characterizing exposure duration. Specifically, discrete variability distributions have inherent clustering of data that can produce exposure estimates that are themselves clustered and do no fully reflect the distribution of exposure parameters across the

population.[1] In addition, although a 99th percentile value was available for developing the discrete variability distributions for exposure duration variability, this value may not fully represent an upper bound for interindividual variability. It is possible that individuals could have exposure durations that are 25 to 50 percent higher than the 99th percentile values available for this exposure parameter. However, these individuals would probably represent upper-bound members of the risk distribution (i.e., individuals at the 99th percentile of the risk distribution or higher). Given the emphasis in this analysis on capturing high-end risk (i.e., 90th to 95th percentile risk estimates), uncertainty introduced by not fully reflecting exposure levels beyond the 99th percentile should not impact the validity of the analysis.

Discrete distributions for exposure duration for both the child and adult resident receptors were developed using data from Table 15-168 of the 1997 EFH. These data include the following percentile estimates: 25th, 50th, 75th, 90th, 95th, and 99th. Data for the 3-yr-old cohort from Table 15-168 in the *Exposure Factors Handbook* were used as the basis for the exposure duration variability distribution for the child resident receptor, and data from the 42-yr-old cohort were used as the basis for the adult resident distribution. In both cases, because cohorts in Table 15-168 did not exactly match the cohorts being modeled in the inorganics analysis, the decision was made to select the cohort age from Table 15-168 that fell nearest the median of the cohorts being modeled in this analysis. The discrete variability distributions for exposure duration derived for this analysis are presented in Table 5-1.

5.2.2 Tap Water Intake Rates

Tap water intake rate variability distributions were required for five cohorts (i.e., 1 to 3, 4 to 6, 7 to 10, 11 to 19 and 19 to 64 year age ranges). For childhood exposures, all individuals start their exposure periods while they are in the youngest cohorts (1 to 3 or 4 to 6 years); however, for exposure to carcinogens, the exposure duration may model the child aging into older age groups, including an adult age group for very long exposure durations. Thus, it is necessary to have distributions of intake rates for all child and adult age groups. Tap water ingestion rate data standardized for body weight (i.e., with units of mL/kg-d) were used in this analysis. Because intake data that were standardized for body weight were used, body weight was not an independent variable in the analysis.

The statistical parameters used to derive the five lognormal distributions for tap water ingestion rates are presented in Table 5-1. A critical issue in using continuous variability distributions in probabilistic risk analysis is the truncation of these distributions to avoid inclusion of exposure parameter estimates that are unreasonable (truncation is typically not an issue with discrete distributions since the upper-bound values in these distributions are generally defined as the highest percentile value for which data are available from the underlying study). In selecting the truncation strategy to develop continuous distributions, care must be taken to avoid the inclusion of unrealistic values, while still allowing for consideration of individuals who

[1]Because there is greater interindividual variability associated with tap water ingestion rates (which are represented in this analysis using continuous variability distributions) than with exposure durations, the degree of clustering in exposure estimates was minimized.

Table 5-1. Variability Distributions for Exposure Parameters Used in Probabilistic Risk Analysis

Receptor Population/ Cohort Age Group	Percentile Values and Statistical Parameters Used to Define Discrete and Continuous Variability Distributions	References/Comments
Exposure duration input parameters (yr)		
Child resident	*discrete variability distribution:* 25th% - 3 50th% - 5 75th% - 8 90th% - 13 95th% - 17 99th% - 22	1997 EFH Table 15-168
Adult resident	*discrete variability distribution:* 25th% - 6 50th% - 11 75th% - 18 90th% - 27 95th% - 35 99th% - 49	1997 EFH Table 15-168
Tap water ingestion rates (mL/kg-d)		
1- to 3-yr-old cohort	*lognormal distribution:* mean: 46.8 STD: 28.1 truncation value (3 GSDs): 211.35	1997 EFH Table 3-7 1997 EFH Table 3-7 derived
4- to 6-yr-old cohort	*lognormal distribution:* mean: 37.9 STD: 21.8 truncation value (3 GSDs): 164.26	1997 EFH Table 3-7 1997 EFH Table 3-7 derived
7- to 10-yr-old cohort	*lognormal distribution:* mean: 26.9 STD: 15.3 truncation value (3 GSDs): 114.87	1997 EFH Table 3-7 1997 EFH Table 3-7 derived
11- to 19-yr-old cohort	*lognormal distribution:* mean: 18.2 STD: 10.8 truncation value (3 GSDs): 81.03	1997 EFH Table 3-7 1997 EFH Table 3-7 derived
20- to 64-yr-old cohort	*lognormal distribution:* mean: 19.9 STD: 10.8 truncation value (3 GSDs): 80.00	1997 EFH Table 3-7 1997 EFH Table 3-7 derived

could experience intake rates beyond the 99th percentile (i.e., high-end exposure). A number of different strategies have been used in previous analyses to truncate exposure parameter variability distributions, including (1) setting the upper bound between 2 and 3 GSDs, and (2) setting the upper bound at twice the 99th percentile. For this analysis, exposure parameter variability distributions for tap water ingestion rates were truncated at 3 GSDs. This approach produced upper-bound tap water ingestion rates that fell between the 99th percentile and twice the 99th percentile, which represents a reasonable approximation of high-end behavior without including unreasonably high intake rates and is better than using an empirical distribution of intake rates that does not consider the possibility of any exposures above the 99th percentile. The truncation values for each of the tap water ingestion rate variability distributions are also included in Table 5-1. Tables 5-2 and 5-3 present the intake rate data from the lognormal distributions developed for this risk assessment and compares them with the empirical data from Table 3-7 for the EFH.

5.2.3 Shower Model Parameters

Percentile data for time spent taking a shower (T_shower) and time spent in the bathroom not in the shower (T_bathroom) are provided in Tables 15-21 and 15-23, respectively, of the EFH (U.S. EPA, 1997c). These data are presented in Table 5-4. Percentile data were used to fit parametric models (gamma, lognormal, and Weibull) using maximum likelihood estimation. Measures of goodness of fit were used to select the most appropriate model for each age variable. The parametric shapes selected as the most appropriate for these data are presented in Table 5-4. The parameter estimates (scale, shape, and location) for each model are provided in Table 5-5 for use with Crystal Ball Monte Carlo software or software requiring similar statistics.

Table 5-2. Comparison of Lognormal Distribution with Empirical Data for Percentiles of Tap Water Intake Rates for Adults

Percentile	Lognormal Distribution (based on Table 3-7) mL/kg-d	Empirical Data Total Tap Water Intake (Table 3-7) mL/kg-d	Recommended Drinking Water Intake Rates (Table 3-30) mL/kg-d
1%	5.40	2.2	
5%	7.50	5.9	
10%	9.10	8.0	
25%	12.50	12.4	
50%	17.50	18.2	19
75%	24.50	25.3	
90%	33.60	33.7	34
95%	40.40	40.0	
99%	57.50	54.8	

Table 5-3. Comparison of Percentiles of Tap Water Intake Rates Between Lognormal Distribution and Empirical Data for Empirical Data for Child Age Groups (mL/kg-d)

	Lognormal Distribution (based on Table 3-7)	Empirical Data for Total Tap Water Intake (Table 3-7)	Lognormal Distribution (based on Table 3-7)	Lognormal Distribution (based on Table 3-7)	Lognormal Distribution (based on Table 3-7	Empirical Data for Total Tap Water (Table 3-7)	Lognormal Distribution (based on Table 3-7	Empirical Data for Total Tap Water (Table 3-7)	Recommended Drinking Water Intake Rates (Table 3-30)
Percentiles	**1-to 3-yr-old**		**4-to 6-yr-old**		**7-to 10-yr old**		**11- to 19-yr old**		**1- to 10-yr old**
1%	11.1	2.7	9.6	3.4	6.8	2.2	4.4	1.2	
5%	15.8	11.8	13.7	10.3	9.8	7.4	6.3	4.3	
10%	19.6	17.8	16.5	14.9	11.8	10.3	7.8	6.5	
25%	27.4	27.2	22.9	21.9	16.2	16.0	10.8	10.6	
50%	39.6	41.4	32.7	33.3	23.4	24.0	15.7	16.3	31
75%	75.7	60.4	47.1	48.7	33.5	35.5	22.7	23.6	
90%	81.1	82.1	65.6	69.3	45.6	47.3	31.7	32.3	64
95%	99.4	101.6	78.6	81.1	55.2	55.2	39.1	38.9	79.4
99%	144.1	140.6	112.7	103.4	78.2	70.5	55.9	52.6	

Table 5-4. Percentile Data Used to Fit Parametric Models for Duration in Shower and Bathroom

EFH Data–Shower Parameters (minutes)													Distributions		
Parameter	Age Cohort	N	P02	P05	P10	P25	P50	P75	P90	P95	P98	P99	Distribution	Pop-Estd Mean	Pop-Estd SDev
T_shower	All ages	3,547	4		5	10	15	20	30	35	50	60	gamma	16.7	9.91
T_bathroom	All ages	3,533			1	3	5	10	20	30	40	50	Weibull	8.5	8.84

N = number of samples; P02-P99 = percentiles; Pop-Estd = population-estimated; SDev = standard deviation.

Table 5-5. Fit of Parametric Models for Duration in Shower and Bathroom

Parameter	Distribution	GAMMA SCALE ALPHA	GAMMA SHAPE BETA	WEI SCALE ALPHA	WEI SHAPE BETA	Location	Min	Max	Units	Source of min/max
T_shower	gamma	5.89	2.83			0	1	60	min	Professional judgment
T_bathroom	Weibull			8.36	0.96	0	1	180	min	Professional judgment

5.3 Central Tendency and High-End Exposure Parameters Used in Deterministic Analysis

The full set of central tendency and high-end exposure parameters used in the analysis are presented in Table 5-6.

Exposure duration values for the adult resident were obtained from 1997 EFH Table 15-176, which presents recommended values for population mobility. Because the 1997 EFH does not provide recommended values for population mobility for the child cohort that was modeled in this analysis (i.e., the 1- to 6-yr-old child resident), both central tendency and high-end exposure duration values for the child resident were obtained from 1997 EFH Table 15-168. Specifically, the mean and 90th percentile residential occupancy period for the 3-yr-old age group (the midpoint of the 1- to 6-yr-old child resident cohort) were selected.

The adult resident tap water ingestion rates were obtained from 1997 EFH Table 3-30, which presents recommended drinking water intake rates. Specifically, the mean and 90th percentile intake rates normalized for body weight from Table 3-30 were selected. However, Table 3-30 does not present recommended drinking water intake rates for the 1- to 6-yr-old child cohort modeled in the deterministic analysis, so these values were obtained from 1997 EFH

Table 5-6. Central Tendency and High-End Exposure Parameter Values Used in Deterministic Risk Analysis

Receptor Population/ Cohort Age Group	Central Tendency Values (Mean)	High-End Value (90^{th} Percentile)	References/Comments
Exposure duration input parameters (yr)			
Child resident	6.5	13	1997 EFH Table 15-168
Adult resident	9	30	1997 EFH Table 15-176
Tap water ingestion rates (mL/kg-d)			
1- to 6-yr-old cohort	42.1	75.3	1997 EFH Table 3-7
7- to 10-yr-old cohort	26.9	47.3	1997 EFH Table 3-7
11- to 19-yr-old cohort	18.2	32.3	1997 EFH Table 3-7
19- to 65-yr-old cohort	19.9	33.7	1997 EFH Table 3-7

Table 3-7.[2] Specifically, for the 1- to 6-yr-old child cohort, the mean tap water intake rates were generated by taking a weighted sum of the mean intake rates for the 1- to 3- and 4- to 6-yr-old child cohorts. A similar procedure was used to generate a high-end tap water intake rate for the 1- to 6-yr-old child cohort, except that 90^{th} percentile tap water intake rates were used. Central tendency and high-end tap water intake rates for older child cohorts used in generating the LADD estimate for the child cohort were identified directly from Table 3-7; there was no need to use weighted averaging techniques to derive these values (i.e., cohort-specific tap water intake rates did not have to be averaged).

5.4 Childhood Exposures

The probabilistic analysis produced 10,000 iterations of risk results. Each iteration included exposure parameters as well as modeled residential well concentrations. Therefore, a data set including 10,000 exposure parameter values was needed for both the adult and child resident.

[2]Note that, unlike the probabilistic analysis, which uses separate tap water ingestion rates for the 1- to 3- and 4- to 6-yr-old cohorts in modeling exposure for the 1- to 6-yr-old age group, the deterministic analysis uses a single tap water ingestion rate for the 1- to 6-yr-old cohort. The rationale for this is that the probabilistic analysis uses a random start age within the 1- to 6-yr-old age range, the deterministic analysis uses a fixed starting age of 3 years, and, consequently, the deterministic analysis requires a tap water ingestion rate that is reflective of the 1- to 6-yr-old cohort as a whole rather than each of the subgroups.

5.4.1 Average Daily Dose (Noncancer Endpoints)

The ADD estimates for the child resident receptor were generated by combining a daily intake rate that reflected variability in tap water ingestion rates with a residential well concentration. This produced a distribution of 10,000 ADD estimates. The ADD distribution was used, in turn, to generate a distribution of 10,000 noncancer HQs for that WS/SWMU/CoC combination for the child resident receptor.

The daily intake rate for the child resident was generated using a two-step procedure for determining tap water ingestion rate variability for the 1- to 6-yr-old cohort. The procedure involved: (1) random selection of either the 1- to 3- or 4- to 6-yr-old cohort for the child being modeled and (2) random sampling of a tap water ingestion rate from the tap water ingestion rate distribution for that age. This approach generated a daily intake rate for the child resident that reflected the age-specific differences in tap water ingestion rates that occurs within the 1- to 6-yr-old cohort.

Cohort aging was not considered in characterizing noncancer risk for the child resident because emphasis was placed on capturing the highest chronic exposure level within this age group, which was expected to occur in children in the youngest cohort due to their higher intake rate to body weight ratio. The exposure parameter variability distributions for tap water ingestion for both the 1- to 3- and 4- to 6-yr-old cohorts were normalized for body weight (intakes are expressed as L/kg-d), which eliminated the need to account for the correlation between body weight and tap water ingestion rate.

Once the daily intake rate data set was generated, it was combined with the residential well concentration data set to generate a discrete distribution of ADD estimates. The following equation was used to generate each ADD estimate for the child resident receptor:

$$ADD_{child} = IR \times C_{drinking\ water} \times \frac{1\ L}{1000\ mL} \qquad (5\text{-}1)$$

Parameter	Definition (units)
ADD_{child}	Modeled Average Daily Dose for the child resident receptor (mg/kg-d)
IR	Tap water ingestion rate sampled from the 1- to 6-yr-old cohort variability distribution for tap water ingestion normalized for body weight (mL/kg-d)
$C_{drinking\ water}$	Modeled maximum 9-yr average annual drinking water well CoC concentration (mg/L)

The generalized distribution of the child ADD without the residential well concentration component is the same as the child intake distribution converted to L/kg-day. The ADD distribution percentiles are presented in Table 5-7.

Table 5-7. Percentiles for Child ADD (L/kg-d)

	Lognormal Distribution (based on Table 3-11)	Total Tap Water Intake (Table 3-7)			Recommended Drinking Water Intake Rates (Table 3-30)
Percentiles	1- to 6-yr-old	1- to 3-yr-old	4- to 6-yr-old	1- to 6-yr-old (average of 1- to 3-yr-old and 4- to 6-yr-old)	1- to 10-yr-old
1%	0.0101	0.0027	0.0034	0.0031	
5%	0.0144	0.0118	0.0103	0.0111	
10%	0.0178	0.0178	0.0149	0.0164	
25%	0.0249	0.0272	0.0219	0.0246	
50%	0.0359	0.0414	0.0333	0.0374	0.031
75%	0.0525	0.0604	0.0487	0.0546	
90%	0.0731	0.0821	0.0693	0.0757	0.064
95%	0.0893	0.1016	0.0811	0.0914	0.0794
99%	0.1296	0.1406	0.1034	0.1220	

5.4.2 Lifetime Averaged Daily Dose (Cancer Endpoints)

The LADD estimates for the child resident were generated by combining the lifetime averaged daily intake rate data set with the residential well concentration data set for a given WS/SWMU/CoC combination. Because the probabilistic analysis conducted for the inorganic chemical manufacturing waste listing risk assessment included variability in exposure duration, longer exposure durations could be sampled than the age range of the initial cohort. Consequently, cohort aging was considered in generating lifetime averaged daily intake rates for the child resident receptor. Two older child age ranges (6 to 10 years old and 11 to 19 years old) and an adult were used in modeling cohort aging. Separate variability distributions for tap water ingestion rates were developed for each of these age ranges. In addition, to provide increased refinement in capturing variability in tap water ingestion rates within the 1- to 6-yr-old child cohort, separate tap water ingestion rate variability distributions were developed for the 1- to 3- and 4- to 6-yr-old age groups. (Note that the 1- to 3- and 4- to 6-yr-old cohort variability distributions are the same as the distribution described for generating noncancer ADD estimates for the child cohort.)

The procedure used to generate lifetime averaged daily intake rates is as follows:

1. Sample a random start age between 1 and 6 years. Based on the randomly sampled start age for exposure, that individual will either fall into the 1- to 3- or 4- to 6-yr-old cohort for purposes of selecting the initial tap water ingestion rate used in generating the LADD estimate.

2. Sample an exposure duration value from the exposure duration variability distribution established for the 1- to 6-yr-old.

3. Determine whether, based on this exposure duration value, the modeled individual will age out of the initial cohort and, if so, how many subsequent age groups will be included in cohort aging for that individual.

4. Sample tap water ingestion rates for each age range included in the modeled individual's exposure. Note: A single sampled intake rate was used to represent intake for all the years that individual spends within a given age range.

5. Generate a time-weighted average tap water ingestion rate for that individual based on the number of years spent in each age group and the sampled intake rates.

6. Divide that time-weighted average tap water ingestion rate by the averaging time used in cancer risk assessment. Note: An exposure frequency of 350 d/yr is also assumed in generating lifetime averaged daily intake rates.

It was assumed that drinking water patterns for a given individual were not likely to change significantly as that individual aged (i.e., their percentile rank relative to others in their cohort regarding tap water ingestion remained the same as the child aged through subsequent age ranges). Although it was reasonable to assume that there would be some degree of correlation in the tap water ingestion rates experienced by individuals as they age, the assumption of a perfect correlation introduced uncertainty into the analysis. However, if a no-correlation approach was adopted, then risk could potentially be underestimated since the number of individuals modeled with relatively high tap water ingestion rates over their entire childhood could be lower, resulting in reduced high-end LADD estimates.

The procedure described above was used to generate the lifetime averaged daily intake rates for the child resident. These lifetime averaged daily intake rates were combined with residential well concentration data sets for individual WS/SWMU/CoC combinations to generate distributions for LADD. The groundwater averaging time used to estimate the residential well concentrations was matched for every iteration with the exposure duration used to estimate the LADD. The equation used to generate each LADD estimate for the child is

$$LADD_{child} = \frac{C_{drinking\ water} \times EF \times \frac{1\ L}{1{,}000\ mL} \times \sum_{1}^{i\ cohorts} (IR_{cohort\ i}\ ED_{cohort\ i})}{AT} \qquad (5\text{-}2)$$

Parameter	Definition (units)
$LADD_{child}$	Modeled Lifetime Average Daily Dose for the child resident receptor (mg/kg-d)
$C_{drinking\ water}$	Modeled drinking water well CoC concentration derived using an averaging time that corresponds to the exposure duration sampled for this LADD estimate (mg/L)
EF	Exposure frequency (d/yr)
i cohorts	Number of cohorts that the modeled individual ages through; this value is dependent on the exposure duration value and random start age sampled (unitless)
$IR_{cohort\ i}$	Tap water ingestion rate sampled from *cohort i* variability distribution for tap water ingestion normalized for body weight (mL/kg-d)
$ED_{cohort\ i}$	The portion of the overall exposure duration value sampled for this modeled individual that the individual spends within cohort i (yr)
AT	Averaging time used to generate a lifetime average intake rate (d).

Note: LADD estimates are generated using an exposure frequency of 350 d/yr and an averaging time of 25,500 days.

The generalized distribution of the child LADD without the residential well concentration component is presented in Table 5-8.

Table 5-8. Generalized Distribution of Child LADD
(Cohort Aging Included)

Percentiles	LADD (L/kg-d)
1%	0.000502
5%	0.000787
10%	0.001004
25%	0.001517
50%	0.002507
75%	0.004152
90%	0.006395
95%	0.008251
99%	0.013391

5.4.3 Deterministic Analysis

5.4.3.1 Average Daily Dose (Noncancer Endpoints). Central tendency tap water ingestion rates identified for the child resident were combined with maximum modeled 9-year average drinking water well concentrations to produce central tendency ADD estimates for each WS/SWMU/CoC combination. High-end ADD estimates were generated using two input parameters set to high-end values (these could be either fate/transport-related parameters or they

could include the tap water ingestion rate along with a fate/transport parameter). Cohort aging was not considered in generating the ADD estimate for the child. The ADD algorithm for the child resident receptor is presented in Equation 5-1. The tap water intake rates in Table 5-2 are the same as generalized ADDs except that the units are converted from mL/kg-d to L/kg-d. The corresponding central tendency and high-end values are, therefore, 0.0421 and 0.0753 L/kg-d, respectively. The central tendency value is slightly above the 50th percentile value (0.0359 L/kg-d) from the lognormal distribution and the high-end value is slightly above the 90th percentile value (0.0731) .

5.4.3.2 Lifetime Average Daily Dose (Cancer Endpoints). Both central tendency and high-end deterministic LADD estimates for the child resident assume that exposure begins at 3 years of age. This value corresponds to the midpoint of the 1- to 6-yr-old age range used to define the child resident.

The central tendency LADD estimate for the child resident was produced by first generating a lifetime averaged daily intake rate for tap water ingestion. The lifetime averaged daily intake rate was estimated by assuming the central tendency exposure duration of 6.5 years and the central tendency tap water ingestion rates established for the first two child cohorts (i.e., 1 to 6 and 7 to 10 years), since the central tendency exposure duration (6.5 years) combined with the starting age (3 years) will result in the central tendency child aging to 9.5 years of age. The lifetime averaged daily intake rate was then combined with a drinking water well concentration for a specific WS/SWMU/CoC combination to generate an LADD for that WS/SWMU/CoC combination.

High-end LADD estimates for the child resident were generated similarly, except that two input parameters were set to high end (these parameters may or may not be exposure parameters). Because it was assumed that exposure begins at 3 years of age in modeling the child resident LADD estimates, if exposure duration were selected as a high-end parameter, then cohort aging would result in the child aging into the 11- to 19-yr-old cohort. However, exposure duration was never found to be a high-end parameter.

The issue of correlation between intake rates from different cohorts considered during cohort aging was not an issue for the deterministic analysis since stochastic sampling of tap water ingestion rates for different cohorts was not used. When tap water ingestion was identified as a sensitive parameter, then high-end values were used for all cohorts considered during cohort aging—a procedure that essentially assumed complete correlation between tap water ingestion rates for individuals across age groups.

Depending on the parameters identified as high-end in the sensitivity analysis, one of four different combinations of tap water ingestion rates and exposure durations was used to generate the high-end deterministic LADD estimate for the child resident for each WS/SWMU combination. In the interest of clearly identifying the exposure parameters that were used in the deterministic analysis, the **annualized** tap water ingestion rates that would result from each of these four combinations are presented in Table 5-9 (i.e., the time-weighted average tap water ingestion rate that reflects the amount of time that the modeled child would spend in each cohort).

Table 5-9. Matrix of Annualized Tap Water Ingestion Rates Resulting from Four Possible Combinations of Central Tendency and High-End Tap Water Ingestion Rates and Exposure Durations

	Central Tendency Tap Water Ingestion Rate (mL/kg-d)	High-End Tap Water Ingestion Rate (mL/kg-d)
Central tendency exposure duration (yr)	ED: 6.5 yr Tap water IR: 42.1(1-6 yr); 26.9 (7-10) ***Tap water IR: 36.3***	ED: 6.5 yrs Tap water IR: 74.3 (1-6 yr); 47.3 (7-10) ***Tapwater IR: 63.9***
High-end exposure duration (yr)	ED: 13 yr Tap water IR: 42.1(1-6 yr); 26.9 (7-10); 18.2 (11-19 yr) ***Tap water IR: 28.2***	ED: 13 yrs Tap water IR: 74.3 (1-6 yr); 47.3 (7-10); 32.3 (11-19 yr) ***Tap water IR: 49.8***

The LADD algorithm for the child resident receptor is presented in Equation 5-2.

5.5 Adult Exposures

5.5.1 Average Daily Dose (Noncancer Endpoints)

As with the child resident receptor, ADD estimates for the adult resident receptor were generated by combining 10,000 averaged daily intake rates that reflect variability in drinking ingestion rates with 10,000 residential well concentrations for a specific WS/SWMU/CoC combination. This produced a distribution of 10,000 ADD estimates. The ADD distribution was, in turn, used to generate a distribution of 10,000 noncancer HQs for that WS/SWMU/CoC combination for the adult resident.

The daily intake rate for the adult resident was generated by sampling from the tap water ingestion rate variability distribution established for the adult cohort. The intake rate was assumed constant throughout the adult exposure duration. The tap water ingestion for the adult cohort was normalized for body weight (i.e., intake units were L/kg-d); consequently, variability in body weight was not a separate input in the analysis.

The 10,000 daily intake rate was combined with the 10,000 residential well concentrations for each WS/SWMU/CoC combination to yield 10,000 ADD estimates. The equation used to generate each ADD estimate for the adult resident is

$$ADD_{adult} = IR \times C_{drinking\ water} \times \frac{1\ L}{1{,}000\ mL} \qquad (5\text{-}3)$$

Parameter	Definition (units)
ADD_{adult}	Modeled Average Daily Dose for the adult resident receptor (mg/kg-d)
IR	Tap water ingestion rate sampled from the adult cohort variability distribution for tap water ingestion normalized for body weight (mL/kg-d)
$C_{drinking\ water}$	Modeled maximum 9-year modeled average annual drinking water well concentration (mg/L)

The generalized distribution of the adult ADD without the residential well concentration component is the same as the adult intake distribution converted to L/kg-d. The ADD distribution percentiles are presented in Table 5-10.

Table 5-10. Percentiles of Generalized Adult ADD

Percentile	Adult ADD (L/kg-d)
1%	0.054
5%	0.0075
10%	0.0091
25%	0.0125
50%	0.0175
75%	0.0245
90%	0.0336
95%	0.0404
99%	0.0575

5.5.2 Lifetime Average Daily Dose (Cancer Endpoints)

The LADD estimates for the adult resident were generated by combining 10,000 lifetime averaged daily intake rates for the adult resident with 10,000 drinking water well concentrations for a given WS/SWMU/CoC. The groundwater averaging time used to estimate the residential well concentration was matched with the exposure duration for each iteration of the risk estimate. For the adult resident, an exposure duration and a single tap water ingestion rate were sampled. An averaging time of 70 years was also used in this calculation. The equation used to generate each LADD estimate for the adult resident is

$$LADD_{adult} = \frac{C_{drinking\ water} \times IR_{adult\ cohort} \times ED_{adult\ cohort} \times EF \times \frac{1\ L}{1{,}000\ mL}}{AT} \qquad (5\text{-}4)$$

Parameter	Definition (units)
$LADD_{adult}$	Modeled Lifetime Average Daily Dose for the adult resident receptor (mg/kg-d)
$C_{drinking\ water}$	Modeled drinking water well CoC concentration derived using an averaging time that corresponds to the exposure duration sampled for this LADD estimate (mg/L)
IR_{adult}	Tap water ingestion rate sampled from the adult variability distribution for tap water ingestion normalized for body weight (mL/kg-d)
ED_{adult}	Exposure duration value sampled for this modeled adult resident (yr)
EF	Exposure frequency (d/yr)
AT	Average lifetime used to generate a lifetime average intake rate (d).

Note: LADD estimates are generated using an exposure frequency of 350 d/yr and an average lifetime of 25,500 days (i.e., 365 d × 70 yr).

The generalized distribution of the adult LADD without the residential well concentration component is presented in Table 5-11.

Table 5-11. Percentiles of Generalized Adult LADD

Percentile	Adult LADD (L/kg-d)
1%	0.000573
5%	0.00089
10%	0.00116
25%	0.00187
50%	0.00335
75%	0.00587
90%	0.00953
95%	0.0125
99%	0.0201

5.5.3 Adult Deterministic Exposures

5.5.3.1 Average Daily Dose (Noncancer Endpoints). Central tendency tap water ingestion rates identified for the adult resident were combined with maximum modeled 9-year average residential well concentrations to produce central tendency ADD estimates for each WS/SWMU/CoC combination. High-end ADD estimates were generated using two input parameters set to high-end values (these could be either fate/transport-related parameters or they could include the tap water ingestion rate). Cohort aging was not considered in generating the

ADD estimate for the adult. The ADD algorithm for the adult resident receptor is presented in Equation 5-3. The tap water intake rates in Table 5-6 are identical to the generalized ADDs except that the unit for the ADD is L/kg-d. The adult central tendency and high-end values (i.e., 0.0199 and 0.0337 L/kg-d, respectively, compare well with the 50th and 90th percentile values (0.0175 and 0.0336 L/kg-d) from the generalized ADDs in Table 5-10.

5.5.3.2 Lifetime Average Daily Dose (Cancer Endpoints). The central tendency LADD estimate for the adult resident was generated by combining the central tendency tap water ingestion rate for the adult resident with the central tendency adult exposure duration, an averaging time, and a residential well concentration for a specific WS/SWMU/CoC combination. Note that the residential well concentration was based on an averaging time that matches the exposure duration used in the calculation. High-end LADD estimates for the adult resident were generated in a similar fashion except that two of the input parameters were set to high-end values (these may or may not be exposure parameters). As with the probabilistic analysis, cohort aging was not considered in generating LADD estimates for the adult resident receptor. The LADD algorithm for the adult resident receptor is presented in Equation 5-4. The adult exposure factors used in the deterministic analysis are presented in Table 5-12.

Table 5-12. Matrix Resulting from Four Possible Combinations of Central Tendency and High-End Tap Water Ingestion Rates and Exposure Durations for Adult Receptors

	Central Tendency Tap Water Ingestion Rate (mL/kg-d)	High-End Tap Water Ingestion Rate (mL/kg-d)
Central tendency exposure duration (yr)	ED: 9 yr Tap water IR: 19.9	ED: 9 yr Tap water IR: 33.7
High-end exposure duration (yr)	ED: 30 yr Tap water IR: 19.9	ED: 30 yr Tap water IR: 33.7

5.6 Lead Screening Analysis

This section describes the lead screening methodology. The lead screening methodology used IEUBK as the basis for generating incremental blood lead (PbB) levels; background PbB levels were characterized using data obtained from Phase 2 of NHANES III (CDC, 1997). The screening methodology focused on characterizing PbB levels for the 1- to 5-yr-old child. This age range was selected because it best matches both the 1997 EFH data on tap water ingestion used in the rest of the risk analysis as well as the CDC data on background PbB levels.

The screening analysis included a high-end deterministic calculation of incremental increase in PbB levels due to exposure to lead concentration in the residential well, background PbB, and total PbB levels. The screening was completed for each of the WS/WMS combinations that were modeled for lead.

The 50th percentile tap water ingestion rate for the child resident (1 to 6 years of age) was used with the 90th percentile maximum 9-yr average groundwater concentration (for lead)

identified for the WS/WMS being modeled in order to generate a single lead dose rate for the child resident. This value was entered into the IEUBK model and the model was run for 5 years of exposure. Note that, because this incremental PbB distribution uses the GSD of 1.6 specified in the IEUBK model, this distribution reflects not only interindividual variability in pharmacokinetics related to lead, but also interindividual variability in behavior, including intake rates. Next, the 90th percentile PbB level was identified from this lognormal distribution. This PbB level represents a high-end incremental PbB level for that particular WS/SWMU combination.

Next, a lognormal distribution characterizing national background exposure for 1- to 5-yr-old children based on data from Phase 2 of the CDC's NHANES III study was used to obtain a high-end background PbB level. Note that this lognormal distribution is not defined per se in the CDC report, but rather was extrapolated using percentile PbB data provided in that report (this extrapolation was based on the geometric mean (GM) PbB level and data on the percentage of the children exceeding 10 µg/dL). The 90th percentile PbB level obtained from this lognormal background distribution represented high-end background lead exposure among 1- to 5-yr-olds across the nation.[3]

The final step in the screening analysis was to sum the high-end incremental PbB levels described above with the high-end background PbB level to generate a single high-end total PbB level for each WS/SWMU combination being considered for lead. This high-end total PbB level was then compared to the action level established for lead of 10 µg/dL.[4] This approach for generating the total PbB level assumed that incremental and background lead exposure are correlated, which is a conservative assumption and may not necessarily hold for all locations.

5.6.1 Uncertainties in Lead Screening Analysis

There are several sources of uncertainty associated with this lead screening approach that warrant discussion:

- **The use of the GSD of 1.6 to represent interindividual variability in tap water ingestion rates:** The assumption that the GSD of 1.6 used in IEUBK to characterize both pharmacokinetic and behavioral variability provided for variability in tap water ingestion rates is used. However, as described in the

[3] The CDC's NHANES III study presents a median PbB level for the 1- to 5-year-old age group of 2.7 µg/dL and estimates that 4.4 percent of this population has PbB levels greater than or equal to 10 µg/dL. When these data points are used to estimate the GSD for a fitted lognormal distribution using the procedure described above, they produce a GSD of 2.04 ug/dL. This GSD of 2.04 µg/dL was then combined with the geometric mean of 2.7 µg/dL to produce the lognormal distribution characterizing background PbB levels for the 1- to 5-year-old age group.

[4] The Centers for Disease Control and Prevention (CDC) has set an "intervention level" for childhood lead poisoning at 10 µg/dL. This level was reduced in 1991 from the previous threshold level of 25 µg/dL based on scientific evidence that adverse health effects can occur at levels as low as 10 µg/dL (HUD, 1995). However, the CDC does not recommend environmental or medical intervention at 10 µg/dL. They recommend medical evaluation at or above 20 µg/dL or if blood lead levels of 15 to 19 µg/dL persist. Various counseling, monitoring, and community-wide prevention activities were recommended for levels between 10 and 19 µg/dL (HUD, 1995).

IEUBK guidance manual, the data sets used to derive the GSD of 1.6 likely focused on pathways traditionally associated with lead exposure (i.e., incidental soil ingestion and ingestion of paint chips) and not necessarily on exposure to lead in drinking water. Consequently, there is uncertainty associated with using this GSD to provide coverage for interindividual variability in tap water ingestion rates. The degree to which variability in tap water ingestion rates is misrepresented using the GSD of 1.6 cannot be readily quantified without having detailed behavioral data (e.g., ingestion rate estimates as well as PbB levels) for the individuals surveyed in the studies used to develop the GSD of 1.6 cited in the IEUBK guidance manual.

Differences in cohort tap water ingestion rate data and the age group modeled in the IEUBK analysis: Additional uncertainty is introduced into the analysis by using tap water ingestion rates for the 1- to 6-yr-old age group in modeling lead risk for the 1- to 5-yr-old child. This difference between cohorts is unavoidable because (1) incremental PbB levels need to be generated for the 1- to 5-yr-old child in order to match the CDC data on background exposure and (2) the 1997 EFH does not provide tap water ingestion rates specifically for the 1- to 5-yr-old (data on the 1- to 6-yr-old cohort is the nearest match). The degree of uncertainty introduced with this cohort disconnect is not considered significant enough to affect conclusions drawn from the lead screening analysis.

The assumption of linearity in projecting total PbB levels: The screening analysis assumed linearity in the relationship between incremental lead exposure and background PbB levels. Specifically, since incremental PbB levels were generated using IEUBK and then these incremental values were added to background PbB levels characterized outside of the IEUBK model (i.e., based on data obtained from the CDC report as described above), there was an explicit assumption of linearity in modeled total PbB levels. However, as lead dose estimates increase, the model behaves in a nonlinear fashion, (i.e., as the lead dose rate increases, modeled PbB levels also increase, but at a diminishing rate, reflecting saturation kinetics associated with lead uptake). Consequently, by modeling background lead exposure outside of the IEUBK model and summing these background estimates with incremental PbB levels derived using the IEUBK model (i.e., assuming linearity in modeling total PbB levels), the total PbB levels that are generated may be conservative to some extent. It is important to note, however, that the degree of conservatism introduced through the assumption of linearity is likely to be minimal since incremental exposure (modeled using IEUBK) is significantly lower than background exposure.

6.0 Human Health Risk Characterization

The inorganic chemical manufacturing listing determination risk assessment was conducted to characterize chronic human health risk resulting from exposure to chemicals of concern (CoCs) through use of residential well water. For purposes of this listing analysis, CoCs were defined as those chemical constituents present in the waste or in leaching test extracts at levels above drinking water HBLs (and showering HBLs for VOCs and SVOCs). Where the potential existed for subsurface releases to surface water, chemical constituents that were above ambient water quality criteria in the waste were also potential CoCs.

As discussed in Section 4.2, screening level analyses were performed for chemical constituents that exceeded HBLs in surface water. However, all surface water CoCs screened out as a result of these analyses, indicating a low potential for risk to human health or aquatic life. Therefore, no additional risk analyses were performed on these wastes. Also, low-volume wastes that are landfilled were subjected to a de minimis screen. This screen was used to identify CoCs that were unlikely to pose a human health risk from groundwater exposures due to their small amounts. Consequently, no additional risk analyses were performed for these CoCs.

Key Attributes of the Inorganics Listing Risk Assessment

Chemicals of concern: CoCs defined as chemicals present in waste or in leaching test extracts at levels above HBLs.

Residential drinking water ingestion: Risk characterization for all waste stream/waste management scenario combinations based on a single exposure scenario involving offsite residential exposure to CoCs that have migrated in groundwater. Both child and adult resident receptor populations are modeled.

Probabilistic and deterministic risk analysis: Central tendency and high-end risk estimates generated using both a probabilistic and deterministic risk framework.

The inorganic chemical manufacturing waste listing risk assessment was designed to characterize both central tendency and high-end cancer and noncancer risk associated with residential exposure to CoCs that migrated offsite in groundwater. The groundwater modeling generated residential well concentrations that were waste- and waste-management-scenario-specific. The exposure assessment used an exposure scenario that included both child and adult residents of a household obtaining tap water from a residential well located downgradient from the SWMU.

This risk assessment included both probabilistic and deterministic components that generated risk estimates for each waste stream/solid waste management unit/constituent of concern (WS/SWMU/CoC) combination. The probabilistic risk analysis uses distributions that reflected variability in modeled CoC residential well water concentrations, exposure duration,

and tap water ingestion rates to generate pathway-specific risk distributions. The deterministic risk analysis generated point estimates of central tendency and high-end risk. The central tendency deterministic risk was calculated by setting all input parameters to the median values of input distributions used for the Monte Carlo analysis. A statistically based sensitivity analysis of the probabilistic risk analysis inputs and results was used to identify the two most sensitive input parameters, which were set to high-end values for the high-end deterministic risk estimate. A separate sensitivity analysis was required and conducted for each WS/SWMU/CoC combination.

6.1 Human Health Benchmarks

6.1.1 Noncancer Risks

EPA uses RfDs to evaluate noncancer effects for ingestion exposures and defines RfD as "an estimate (with uncertainty spanning perhaps an order of magnitude or greater) of a daily exposure level for the human population, including sensitive subpopulations, that is likely to be without an appreciable risk of deleterious effects during a lifetime" (U.S. EPA, 1989). RfDs are expressed in milligrams of chemical intake per kilogram body weight per day (mg/kg-d).

RfDs are the primary benchmarks used to evaluate noncarcinogenic hazards posed by environmental exposures to chemicals and are based on the "threshold" approach, which is based on the theory that there is a "safe" exposure level (a threshold) that must be exceeded before a toxic effect occurs. RfDs do not provide true dose-response information in that they are **estimates** of an exposure level or concentration that is believed to be below the threshold level or no observed adverse effects level (NOAEL). It also is important to understand that all RfDs are not necessarily equivalent expressions of toxicity. The degree of uncertainty and confidence levels in the various RfDs varies a great deal and is based on different toxic effects. RfDs that have been verified by an intra-Agency workgroup are listed in IRIS.

RfDs are derived from the highest NOAEL for the most sensitive effect identified in human epidemiological studies or from subchronic or chronic studies in laboratory animals. If a NOAEL was not identified in any of the available studies, the lowest observed adverse effect level (LOAEL) was used. If the studies reported dose levels as parts per million (ppm) in the diet or water, the dose levels were converted to mg/kg-d based on the consumption level and body weights of the test subjects. It is generally assumed that dose levels expressed on a mg/kg-d basis are equivalent in humans and animals; therefore, dose adjustments were not necessary unless chemical-specific pharmacokinetic data indicated that a dose adjustment was appropriate.

EPA uses RfCs to evaluate noncancer effects for inhalation exposures. The RfC is an air concentration that is considered protective of all individuals, including sensitive subpopulations. The RfC has no dose component.

Once an appropriate NOAEL or LOAEL was identified, the characteristics and the quality of the data were examined and the NOAEL or LOAEL was divided by uncertainty factors and modifying factors to derive the RfD. Uncertainty factors are applied to address limitations of the available toxicological data and are necessary to ensure the RfD is protective for individuals in the general population. Factors of 10 are most commonly used as uncertainty factors (Table 6-1).

Table 6-1. Standard Uncertainty and Modifying Factors

Description	Rationale	Values
Interspecies variation	Extrapolation from animal data to humans	3 to 10
Intraspecies variation	Accounts for sensitive individuals (e.g., children, elderly, asthmatics)	1 to 10
Subchronic to chronic	A subchronic study was used to derive a chronic RfD or RfC	3 to 10
LOAEL to NOAEL	A LOAEL was used instead of a NOAEL	1 to 10
Incomplete database	Lack of data for critical endpoints (e.g., reproductive and developmental)	1 to 10
Modifying factor	Accounts for additional uncertainties per professional judgment	1 to 10

An uncertainty factor of 3 may be used if appropriate pharmacokinetic data (or a model) are available. The default value for the modifying factor is 1. All uncertainty factors and modifying factors are multiplied together to derive the total uncertainty factor, with 3,000 being the maximum recommended value (U.S. EPA, 1994c). The use of uncertainty factors is based on long-standing scientific practice.

6.1.2 Cancer Risks

Measures of carcinogenic potency, the CSFs and URFs, may be derived from a number of statistically and/or biologically based models. Traditionally, the linearized multistage model has been the default model for extrapolating cancer slope factors for low doses; however, other models also have been used. Although several models may provide a good fit to the experimental data, the slope factors at low doses may be different by up to several orders of magnitude depending on which model is used. EPA's proposed cancer risk guidelines propose significant changes to the default methodology (U.S. EPA, 1996c). Although the new methodology has been used to develop some benchmarks listed in IRIS (e.g., for PCBs), all of the cancer benchmarks used in this report are based on the linearized multistage model.

CSFs and URFs are used to evaluate cancer risks for ingestion and inhalation exposures, respectively. Unlike RfDs and RfCs, CSFs and URFs do not represent "safe" exposure levels; rather, they are derived mathematically as the 95 percent upper confidence limit of the slope of the linear portion of the dose-response curve. That is, they relate levels of exposure with a probability of effect or risk. The CSF is expressed in units of $(mg/kg\text{-}d)^{-1}$ and the URF is expressed in units of $(\mu g/m^3)^{-1}$. For this risk assessment, URFs were converted into inhalation CSFs to calculate risk.

6.2 Risk Descriptors

Human health risk characterization involves combining lifetime average daily dose (LADDs) and average daily dose (ADDs) with applicable toxicity factors (i.e., cancer slope factors and RfDs/RfCs) to generate cancer risk and noncancer HQ estimates, respectively. The methodology used to generate cancer and noncancer risk estimates is described below.

6.2.1 Lifetime Excess Cancer Risk

Cancer risk is characterized using lifetime excess cancer risk estimates to represent the excess probability of developing cancer over a lifetime as a result of exposure to the constituent of interest. Lifetime excess cancer risk estimates are the product of the LADD for a specific receptor/WS/SWMU/CoC combination and the corresponding cancer slope factor, as shown in Equation 6-1:

$$Lifetime\ excess\ cancer\ risk = LADD \times CSF \quad (6\text{-}1)$$

Parameter	Definition (units)
LADD	Lifetime average daily dose (mg/kg BW/d)
CSF	Cancer slope factor $(mg/kg\ BW/d)^{-1}$

The cancer slope factor is derived from either human or animal data and is taken as the upper bound on the slope of the dose-response curve in the low-dose region, generally assumed to be linear, expressed as a lifetime excess cancer risk per unit exposure. The same slope factor was used for estimating cancer risks for both the child and adult resident receptors. However, individuals exposed to carcinogens in the first few years of life may be at increased risk of developing cancer. Therefore, significant uncertainties and unknowns exist regarding the estimation of lifetime cancer risks in children.

6.2.2 Ingestion Hazard Quotient

Noncancer risk is characterized through the use of hazard quotients, which are generated by dividing an ADD by the corresponding reference dose. The ingestion hazard quotient uses the ADD as the exposure metric. An HQ establishes whether a particular individual has experienced exposure that places him or her either above or below a threshold of concern for a specific health effect. Therefore, unlike cancer risk estimates, HQs are not probability statements. The reference dose represents a “no-effects” level that is presumed to be without appreciable risk from chronic exposures over a lifetime. The RfD may be derived from human or animal studies and may include uncertainty factors to account for deficiencies in the available studies. Equation 6-2 shows the derivation of the ingestion hazard quotient:

$$HQ_{ing} = \frac{ADD}{RfD} \quad (6\text{-}2)$$

Parameter	Definition (units)
ADD	Average daily dose (mg/kg-d)
RfD	Reference dose (mg/kg-d)

6.2.3 Inhalation Hazard Quotient

Noncancer inhalation risk is characterized through the use of hazard quotients, which are generated by dividing an air concentration by the corresponding reference concentration. An HQ establishes whether a particular individual has experienced exposure that places him or her either above or below a threshold of concern for a specific health effect. Therefore, unlike cancer risk estimates, HQs are not probability statements. The reference concentration represents a "no-effects" level that is presumed to be without appreciable risk from chronic exposures over a lifetime. The RfC may be derived from human or animal studies and may include uncertainty factors to account for deficiencies in the available studies. Equation 6-3 shows the derivation of the inhalation hazard quotient:

$$HQ_{inh} = \frac{CONC_{AIR}}{RfC} \quad (6\text{-}3)$$

Parameter	Definition (units)
$CONC_{AIR}$	Air concentration (mg/m^3)
RfD	Reference concentration (mg/m^3)

6.3 Risk Results Generated Using Probabilistic and Deterministic Risk Analyses

The inorganics listing risk assessment included a probabilistic component and a deterministic component, both of which were aimed at characterizing central tendency and high-end residential risk for each WS/SWMU/CoC combination considered in the inorganics listing analysis. The methodologies described above for generating cancer risk and noncancer HQ estimates based on LADDs and ADDs were used to generate risk results for both the probabilistic and deterministic analyses.

6.3.1 Probabilistic Analysis Results

The probabilistic analysis generated up to four distributions of 10,000 risk estimates for each WS/SWMU/CoC combination. These results included cancer risk distributions for arsenic in the adult and child residents and noncancer HQ distributions for arsenic and all other CoCs in the adult and child residents. Arsenic was the only carcinogen that did not otherwise screen out. Once each of the distributions of 10,000 risk or HQ values was generated, specified percentiles (e.g., 50^{th}, 80^{th}, 90^{th}, 95^{th}, and 99^{th}) from the distribution were selected and reported as the probabilistic risk analysis results. These results reflected the range of variability for the total analysis, including variability in the source parameters, the environmental setting, the location of the receptors, and the interindividual variability in exposure parameters.

To learn more about the importance of each of the variable parameters to the overall risk analysis, a sensitivity analysis was conducted. A new sensitivity analysis protocol was implemented for this listing that is based on the response surface regression approach.

6.3.2 Sensitivity Analysis

Sensitivity analyses were conducted for previous listings by evaluating how much change in risk occurred as a result of varying an individual input parameter from a median or mean value to a 90^{th} percentile or high-end value. When the risk depends on the aggregate impact of a number of input parameters, however, such an approach may not necessarily identify the most important input variables. This may occur for several reasons:

- The ranges chosen for the various input parameters may not be defined consistently.
- Various input parameters may interact with one another (i.e., the effect of input X_1 on an outcome Y depends on the level of other inputs X_2, X_3, etc., so that the observed effect of X_1 depends on what values were chosen for the other variables as well).
- Nonlinear effects may obscure the effect of the input parameter (e.g., if only low and high levels of an input variable are examined, but the relationship between the risk and the input variable is of a quadratic nature, then the importance of the input parameter may be overlooked).

To address such issues, statistical regression methods were used to perform the sensitivity analyses. Although regression methods have distinct advantages over previous approaches, certain limitations remain. Regression methods are not capable of determining the sensitivity of model results to input parameters that are not varied in the analysis (e.g., assumptions) or are not otherwise included within the scope of the analysis (e.g., model-derived parameters). If, for some reason, the most important parameters are not varied or their variability is improperly characterized, the sensitivity analysis may not identify them as being the most important parameters.

The sensitivity analysis was conducted on a data set generated during groundwater pathway modeling. For example, a set of input parameters (X_1, X_2, ..., X_p) was used in the modeling simulation. Second, the risk equation was added to the analysis.

The different situations can be distinguished:

- The result of interest is the groundwater concentration of some contaminant; in this case, the Xs are all associated with site and environmental conditions.

- The result of interest is the risk associated with human exposures to the groundwater concentration of the contaminant; in this case, additional Xs associated with the intake rates, contact durations, etc., are also involved.

The regression approach uses the various combinations of X values that were used during the simulation and the resulting groundwater concentration and risk values as input data to a regression model. Functions of the results variables (denoted as Ys) were treated as dependent variables; for example, Y denoted the logarithm of the groundwater contaminant concentration or of the risk. Functions of the Xs were treated as independent variables. The goals of the approach were (1) to determine a fairly simple polynomial approximation to the simulation results that expressed the Ys as functions of the Xs, (2) to optimize this "response surface" and assess the importance of the various Xs by performing statistical tests on the model parameters, and (3) to rank the Xs based on their relative contribution (in terms of risk) to the final response surface regression model.

These goals were realized using a second-order regression model. Such a model takes the following form:

$$\hat{Y} = \hat{\beta}_0 + \sum_{k=1}^{p} \hat{\beta}_k x_k + \sum_{k=1}^{p} \hat{\beta}_{kk} x_k^2 + \sum_{k=1}^{p-1} \sum_{j=k+1}^{p} \hat{\beta}_{kj} x_k x_j \qquad (6\text{-}4)$$

where the βs are the least squares regression estimates of the model parameters.

The statistical significance of the parameters associated with the first-order, squared, and cross product terms were tested and all nonsignificant terms were removed from the model. The parameters in this reduced model were then reestimated and the process of testing was repeated. This was done to capture the most important independent variables (Xs) that influence the dependent variables (Ys). Details on the response surface regression approach are found in Appendix G.

Once the final regression model was developed, the input parameters (Xs) were ranked based on percentage of risk accounted for by that parameter. The percent risk was calculated using the following equation:

$$Percent\ Risk = \frac{[FMSS - RMSS]}{[FMSS + ERSS]} \quad (6\text{-}5)$$

where

FMSS = model sum of squares for the final model

RMSS = model sum of squares for a model in which all terms involving x_u are removed (i.e., a reduced model)

ERSS = model error sum of squares.

The two parameters responsible for the largest percentage of the risk are the two parameters set to high-end values in the deterministic analysis.

At the outset of this risk analysis, the entire distribution of 10,000 iterations was used in the sensitivity analysis. However, as the analysis proceeded, it was noted that, when the two-high-end parameters determined by the sensitivity analysis were set to high end, little variation from the central tendency risk value was observed. At this point, the methodology of the sensitivity analysis was reexamined, and it was decided that the analysis should focus on the 50th percentile risk and above, given that the primary purpose of the sensitivity analysis was to determine what parameters were most important in predicting high-end risk, rather than the entire range of risk, across all percentiles. Thus, for the waste streams in the titanium dioxide sector and all waste streams evaluated after the titanium dioxide wastes, the sensitivity analysis was performed using the 5,000 iterations resulting in the highest risk. Thus, the parameters having the greatest effect on the higher range of risk were identified by the sensitivity analysis. In cases where the analysis was performed on the full 10,000 iterations and the top 5,000 iterations, the results of the sensitivity analysis were frequently different, but not in all cases.

Sensitivity analyses were performed for all combinations of WS/SWMU/CoCs for the inorganic chemical manufacturing risk analysis. Of these analyses,14 were performed on the entire 10,000 iterations only. The remaining analyses were performed either only on the top half of the distribution or on the top half and the whole distribution. The results of the sensitivity analysis were variable from waste stream to waste stream and from constituent to constituent. However, several parameters reappeared frequently as one of the two most influential parameters.

The most common parameters identified as risk drivers were the K_d in the aquifer and the K_d in the unsaturated zone. In the small-volume waste streams where the analysis was performed on the full 10,000 iterations, the second most common high-end parameter was the unsaturated

zone thickness. Only one case showed infiltration rate as a high-end variable.[1] In the higher volume waste streams common in the titanium dioxide sector, the most common high-end parameters are K_ds in the saturated zone and the unsaturated zone and the distance from the plume center line to the well (y-well). The only exceptions to these parameters are two occurrences each of the longitudinal distance to the well (X-well) and the consumption of drinking water and one occurrence of waste management unit size. For the wastewaters managed in surface impoundments, the most frequently occurring high-end parameters are again K_ds in the saturated zone and the unsaturated zone and the distance from the plume center line (y-well).[2] The sensitivity analysis for each WS/SWMU/CoC, including the F-test results, is documented in Appendixes A through D. The two high-end parameters identified using the sensitivity analysis and used for the deterministic analysis are presented in Table 6-2.

Table 6-2. High-End Parameters Identified by Sensitivity Analysis

Waste Stream	Constituent of Concern	High-End Parameters
Chloride-sulfate process wastewater treatment sludge Millennium HPP	Manganese	Saturated Zone Kd & Y-Well)
	Thallium	Y-Well & Drinking Water Intake Rate)
Sulfate process secondary gypsum Millennium HPP	Arsenic	Unsaturated zone K_d & Saturated Zone K_d
	Manganese	Saturated Zone K_d & X-Well
	Antimony	Saturated Zone K_d & Y-Well)
Off-spec titanium dioxide Du Pont New Johnsonville	Lead	Unsaturated zone K_d & Saturated Zone K_d
Chloride and sulfate process milling sand Kemira	Antimony	Unsaturated zone K_d & Depth to groundwater
Sulfate digestion sludge Millennium HPP	Antimony	Unsaturated zone K_d & Y-Well)
	Vanadium	Unsaturated zone K_d & Saturated Zone K_d

(continued)

[1] This case was for arsenic in sodium chlorate sludge and occurred because this waste stream is managed in areas with variable climates and one of the sites is very dry; thus, in this case, infiltration rate is a factor associated with greater risk.

[2] In addition, in a single case, leachate concentration appeared as one of the two high-end parameters.

Table 6-2. (continued)

Waste Stream	Constituent of Concern	High-End Parameters
Ilmenite WWT solids Du Pont Edgemoor	Antimony	X-Well & Y-Well)
	Arsenic	Saturated Zone K_d & X-Well
	Manganese	Saturated Zone K_d & SWMU Area
	Thallium	X-Well Drinking Water Intake Rate
Ilmenite process wastewaters Du Pont Delisle	Manganese	Saturated Zone K_d & Y-Well)
	Thallium	Saturated Zone K_d & Y-Well)
	Vanadium	Saturated Zone K_d & leachate concentration
Chloride-sulfate process wastewaters Millennium HPP	Arsenic	Saturated Zone K_d & Unsaturated zone K_d
	Manganese	Saturated Zone K_d & Y-Well
Chloride-sulfate process wastewaters Kerr McGee	Arsenic	Saturated Zone K_d & Y-Well)
	Antimony	Saturated Zone K_d & Y-Well
	Molybdenum	Saturated Zone K_d & Y-Well
	Thallium	Saturated Zone K_d & Unsaturated zone K_d
Antimony oxide low antimony slag	Arsenic	Unsaturated zone K_d & saturated zone K_d
	Antimony	Initial Concentration& Y-Well
	Boron	Hydraulic conductivity &Y-Well
	Selenium	Hydraulic conductivity &Y-Well)
	Vanadium	Saturated zone K_d & Y-Well

6.3.3 Deterministic Analysis Results

The deterministic analysis generated point estimates of central tendency and high-end risk based on setting the two most sensitive parameters, as determined by the sensitivity analysis, to their high-end values. The deterministic analysis generated risk estimates for each WS/WMU/ CoC combination, including (1) central tendency and high-end cancer risk estimates for the child

resident, (2) central tendency and high-end noncancer HQ estimates for the child resident, (3) central tendency and high-end cancer risk estimates for the adult resident, and (4) central tendency and high-end noncancer HQ estimates for the adult resident.

Risk results for all of the CoCs provided by industry sector/waste stream/solid waste management unit, including both the Monte Carlo and the deterministic results, for the adult and child are presented in Tables 6-3 through 6-20. For the deterministic analysis, the two high-end parameters used in the analysis are presented.

6.3.4 Level of Concern

The nominal level of concern for EPA hazardous waste listings is an excess lifetime cancer risk estimate equal to $1x10^{-5}$ or a noncancer HQ of 1. The nominal level of concern is applied to risk for each waste stream or waste management unit. For the inorganic chemical manufacturing industry, arsenic was the only carcinogen included in the risk analysis. Thus, additivity of cancer risk was not considered in this analysis. The level of concern for cancer was based only on the risk estimated for arsenic, and arsenic did not reach a level of concern in any waste stream. Some of the contaminants evaluated for noncancer health effects produced HQs that were near to or greater than 1. However, the additivity of noncancer risks was not a factor for the CoCs included in this risk analysis.

6.4 Key Findings

The key findings of the risk assessment are as follows:

- Low-volume nonwastewater wastes in the hydrogen cyanide, sodium phosphate, and sodium chlorate manufacturing sectors of the inorganic chemical manufacturing industry show no risk in excess of 1E-05 for arsenic or no HQ in excess of 1 for other constituents at the 90^{th} or 95^{th} percentile in any waste stream reported managed in municipal or offsite industrial landfill. However, antimony in ammonia recycle filters, when modeled as managed in either the industrial D or municipal landfill, exceeds an HQ of 1 at the 99^{th} percentile for the child receptor.

- Combined wastewaters managed in onsite surface impoundments in the hydrogen cyanide sector showed an HQ greater than 1 for acetonitrile for the inhalation pathway during showering at the 99^{th} percentile. No exceedances of ambient water quality criteria were found for any CoC in combined wastewaters in the hydrogen cyanide sector.

- Wastewaters in the titanium dioxide sector managed in onsite surface impoundments showed no risk in excess of 1E-05 for arsenic or no HQ in excess of 1 for other constituents in any waste stream at the 90^{th} or 95^{th} percentile in any waste stream and no exceedances of ambient water quality criteria. However, antimony in the chloride process wastewaters exceeded an HQ of 1 at the 99^{th} percentile.

Table 6-3. Risk Results for Ammonia Recycle Filter Waste from Hydrogen Cyanide Sector Managed in Industrial D Landfills

	Antimony		Arsenic - Noncancer		Arsenic - Cancer		Cadmium		Nickel		Cyanide	
Percentile	Adult HQ	Child HQ	Adult HQ	Child HQ	Adult Risk	Child Risk	Adult HQ	Child HQ	Adult HQ	Child HQ	Adult HQ	Child HQ
50th	0.0022	0.0044	3.3e-14	6.9e-14	2.8e-18	2.1e-18	3.0e-08	6.1e-08	1.8e-12	3.7e-12	0	0
75th	0.015	0.032	1.2e-05	2.5e-05	1.0e-09	7.4e-10	5.0e-06	1.0e-05	9.6e-06	1.9e-05	0	0
80th	0.024	0.050	4.0e-05	7.9e-05	3.4e-09	2.5e-09	1.2e-05	2.6e-05	3.5e-05	7.3e-05	0	0
85th	0.041	0.085	1.3e-04	2.6e-04	1.1e-08	8.1e-09	3.2e-05	6.6e-05	1.1e-04	2.3e-04	0	0
90th	0.079	0.16	4.2e-04	8.4e-04	3.5e-08	2.8e-08	9.3e-05	1.9e-04	3.7e-04	7.7e-04	0	0
95th	0.19	0.39	0.0018	0.0037	1.6e-07	1.2e-07	3.6e-04	7.5e-04	0.0016	0.0034	3.6e-16	7.4e-16
97.5th	0.40	0.84	0.0054	0.011	4.8e-07	3.6e-07	9.3e-04	0.0020	0.0045	0.010	1.0e-12	2.1e-12
99th	0.83	1.8	0.016	0.034	1.6e-06	1.1e-06	0.0024	0.0055	0.011	0.025	3.7e-10	7.5e-10

Table 6-4. Risk Results for Ammonia Recycle Filter Waste from Hydrogen Cyanide Sector Managed in Municipal Landfills

	Antimony		Arsenic - Noncancer		Arsenic - Cancer		Cadmium		Nickel		Cyanide	
Percentile	Adult HQ	Child HQ	Adult HQ	Child HQ	Adult Risk	Child Risk	Adult HQ	Child HQ	Adult HQ	Child HQ	Adult HQ	Child HQ
50th	0.0024	0.0049	3.6e-14	7.6e-14	3.1e-18	2.2e-18	1.7e-07	3.5e-07	2.2e-12	4.7e-12	0	0
75th	0.017	0.035	1.3e-05	2.8e-05	1.1e-09	8.2e-10	1.6e-05	3.2e-05	1.1e-05	2.1e-05	0	0
80th	0.026	0.055	4.5e-05	8.8e-05	3.8e-09	2.9e-09	3.0e-05	6.1e-05	3.9e-05	8.0e-05	0	0
85th	0.045	0.094	1.4e-04	2.9e-04	1.2e-08	9.2e-09	5.8e-05	1.2e-04	1.2e-04	2.5e-04	0	0
90th	0.087	0.18	4.6e-04	9.4e-04	3.9e-08	3.1e-08	1.3e-04	2.7e-04	4.0e-04	8.5e-04	0	0
95th	0.20	0.42	0.0020	0.0041	1.8e-07	1.3e-07	3.6e-04	7.6e-04	0.0017	0.0037	3.8e-17	8.3e-17
97.5th	0.41	0.89	0.0061	0.013	5.4e-07	4.1e-07	8.5e-04	0.0017	0.0048	0.010	1.7e-13	3.5e-13
99th	0.88	1.9	0.018	0.038	1.8e-06	1.3e-06	0.0021	0.0046	0.012	0.025	5.2e-11	1.0e-10

Table 6-5. Risk Results for Feed Gas Filters from Hydrogen Cyanide Sector Managed in Municipal Landfills

Percentile	Boron	
	Adult HQ	Child HQ
50th	3.9e-04	0.0010
75th	0.0020	0.0040
80th	0.0030	0.0060
85th	0.0040	0.0080
90th	0.0070	0.014
95th	0.014	0.031
97.5th	0.026	0.054
99th	0.051	0.11

Table 6-6. Risk Results for Acetonitrile from Combined Wastewaters from Hydrogen Cyanide Sector Managed in Onsite Surface Impoundment, Theodore, AL

Percentile	HQ_{Inh}
50th	0.050
75th	0.14
80th	0.18
85th	0.23
90th	0.32
95th	0.47
97.5th	0.67
99th	1.0

Table 6-7. Risk Results for Filter Cake Waste in Sodium Phosphate Sector Managed in Industrial D Landfills

	Antimony		Thallium	
Percentile	**Adult HQ**	**Child HQ**	**Adult HQ**	**Child HQ**
50th	1.7e-05	3.4e-05	3.4e-06	6.9e-06
75th	2.2e-04	4.6e-04	2.0e-04	4.1e-04
80th	4.0e-04	8.4e-04	3.7e-04	7.7e-04
85th	7.5e-04	0.0015	7.3e-04	0.0015
90th	0.0014	0.0031	0.0016	0.0034
95th	0.0035	0.0075	0.0038	0.0080
97.5th	0.0073	0.016	0.0082	0.017
99th	0.015	0.032	0.018	0.039

Table 6-8. Risk Results for Acetonitrile from Filter Bag Waste in Sodium Phosphate Sector Managed in Industrial D Landfills

	Antimony	
Percentile	**Adult HQ**	**Child HQ**
50th	1.3e-05	2.7e-05
75th	1.6e-04	3.3e-04
80th	2.7e-04	5.7e-04
85th	5.2e-04	0.0011
90th	0.0011	0.0023
95th	0.0030	0.0061
97.5th	0.0077	0.016
99th	0.020	0.043

Table 6-9. Risk Results Sludge Residues— Sodium Chlorate Sector Managed in Municipal Landfills

	Arsenic - Noncancer		Arsenic - Cancer		Manganese		Nickel		Zinc	
Percentile	Adult HQ	Child HQ	Adult Risk	Child Risk	Adult HQ	Child HQ	Adult HQ	Child HQ	Adult HQ	Child HQ
50th	0	0	0	0	6.1e-12	1.2e-11	0	0	0	0
75th	2.5e-07	5.3e-07	2.1e-11	1.5e-11	4.5e-06	9.4e-06	1.9e-10	3.9e-10	2.6e-14	5.4e-14
80th	1.1e-05	2.4e-05	9.1e-10	7.3e-10	1.1e-05	2.4e-05	8.2e-09	1.7e-08	1.6e-11	3.3e-11
85th	9.8e-05	2.0e-04	8.3e-09	6.1e-09	2.9e-05	6.1e-05	1.5e-07	3.2e-07	1.3e-09	2.7e-09
90th	5.7e-04	0.0011	5.0e-08	3.7e-08	7.2e-05	1.5e-04	1.5e-06	3.2e-06	2.0e-07	4.1e-07
95th	0.0033	0.0067	2.8e-07	2.3e-07	2.0e-04	4.3e-04	1.6e-05	3.1e-05	5.4e-06	1.1e-05
97.5th	0.0098	0.021	9.0e-07	7.1e-07	4.9e-04	0.0010	5.6e-05	1.2e-04	3.4e-05	7.2e-05
99th	0.030	0.061	2.6e-06	2.1e-06	0.0012	0.0025	2.5e-04	5.1e-04	1.8e-04	4.1e-04

Table 6-10. Probabilistic Risk Results for Sodium Chlorate Production Filter Wastes Without Chromium Managed in an Industrial D Landfill Arsenic - Cancer

Percentile (mg/L)	Adult Risk	Child Risk
75th	9.1e-12	6.5e-12
80th	6.6e-11	4.7e-11
85th	2.7e-10	2.1e-10
90th	1.0e-09	7.7e-10
95th	5.0e-09	3.7e-09
97.5th	1.3e-08	1.0e-08
99th	4.5e-08	3.4e-08

Table 6-11. Risk Results for Filter Residues— Sodium Chlorate Sector Managed in Municipal Landfills

	Antimony		Arsenic - Noncancer		Arsenic - Cancer		Cadmium	
Percentile	Adult HQ	Child HQ	Adult HQ	Child HQ	Adult HQ	Child HQ	Adult HQ	Child HQ
50th	0	0	0	0	0	0	0	0
75th	3.1e-07	6.4e-07	2.3e-10	5.0e-10	2.0e-14	1.6e-14	1.3e-07	2.8e-07
80th	1.4e-06	3.0e-06	6.6e-09	1.3e-08	5.8e-13	4.4e-13	1.4e-06	2.8e-06
85th	4.8e-06	1.0e-05	4.1e-07	8.7e-07	3.6e-11	2.8e-11	6.2e-06	1.3e-05
90th	1.7e-05	3.7e-05	5.9e-06	1.2e-05	5.5e-10	3.9e-10	2.7e-05	5.8e-05
95th	8.6e-05	1.8e-04	5.4e-05	1.1e-04	4.7e-09	3.7e-09	1.4e-04	2.7e-04
97.5th	2.8e-04	5.7e-04	2.0e-04	4.4e-04	2.0e-08	1.5e-08	3.7e-04	7.7e-04
99th	7.8e-04	0.0016	9.6e-04	0.0019	7.6e-08	6.7e-08	0.0012	0.0026

Table 6-12. Risk Results for Sulfate Process Digestion Sludge—Titanium Dioxide Sector Managed in Onsite Industrial Landfill

Percentile	Antimony		Vanadium	
	Adult HQ	Child HQ	Adult HQ	Child HQ
50th	0.014	0.029	3.7e-05	7.9e-05
75th	0.059	0.12	0.0027	0.0055
80th	0.076	0.16	0.0049	0.010
85th	0.10	0.20	0.0087	0.018
90th	0.13	0.27	0.016	0.033
95th	0.18	0.39	0.032	0.066
97.5th	0.24	0.52	0.052	0.11
99th	0.33	0.71	0.081	0.17
Central Tendency	0.055	0.12	0.0005	0.0012
High End Full Distribution	0.070	0.15	0.030	0.062
High End Half Distribution	0.070	0.15	0.030	0.062

Table 6-13. Risk Results for Sulfate Process Secondary Gypsum—Titanium Dioxide Sector Managed in Onsite Industrial Landfill

	Antimony		Arsenic - Noncancer		Arsenic - Cancer		Manganese	
Percentile	Adult HQ	Child HQ	Adult HQ	Child HQ	Adult Risk	Child Risk	Adult HQ	Child HQ
50th	0.038	0.077	1.8e-10	3.9e-10	1.8e-14	1.4e-14	4.7e-04	9.8e-04
75th	0.15	0.32	3.5e-04	7.1e-04	2.8e-08	2.1e-08	0.0086	0.018
80th	0.20	0.41	0.0011	0.0023	8.9e-08	7.0e-08	0.014	0.028
85th	0.25	0.52	0.0030	0.0063	2.6e-07	1.9e-07	0.022	0.044
90th	0.33	0.70	0.0073	0.015	5.8e-07	4.5e-07	0.032	0.067
95th	0.47	0.99	0.015	0.031	1.4e-06	1.1e-06	0.050	0.11
97.5th	0.61	1.3	0.022	0.046	2.3e-06	1.7e-06	0.068	0.15
99th	0.83	1.8	0.032	0.068	3.9e-06	2.7e-06	0.095	0.20
Central Tendency	0.14	0.30	9.3e-06	2.0e-05	5.2e-10	6.8e-10	0.006	0.013
High End Full Distribution	0.33	0.71	0.011	0.023	6.1e-07	8.1e-07	0.026	0.054
High End Half Distribution	0.33	0.71	0.011	0.023	6.1e-07	8.1e-07	0.036	0.076

Table 6-14. Risk Results for Chloride Sulfate Process Milling Sand—Titanium Dioxide Sector Managed in Industrial Landfill

	Antimony	
Percentile	**Adult HQ**	**Child HQ**
50th	8.4e-05	1.7e-04
75th	5.6e-04	0.0011
80th	9.00e-04	0.0019
85th	0.0016	0.0032
90th	0.0030	0.0062
95th	0.0080	0.017
97.5th	0.016	0.035
99th	0.036	0.078
Central Tendency	3.1e-04	6.6e-04
High End Full Distribution	3.2e-04	6.7e-04
High End Half Distribution	3.2e-04	6.7e-04

Table 6-15. Risk Results for Chloride Sulfate Process Wastewater Treatment Sludge—Titanium Dioxide Sector Managed in Industrial Landfill

	Manganese		Thallium	
Percentile	**Adult HQ**	**Child HQ**	**Adult HQ**	**Child HQ**
50th	0.0020	0.0040	0.011	0.023
75th	0.040	0.080	0.065	0.13
80th	0.060	0.13	0.084	0.17
85th	0.090	0.19	0.11	0.22
90th	0.14	0.28	0.14	0.29
95th	0.22	0.46	0.19	0.41
97.5th	0.29	0.63	0.24	0.52
99th	0.39	0.84	0.33	0.72
Central Tendency	0.030	0.060	0.068	0.14
High End	0.15	0.33	0.16	0.37

Table 6-16. Risk Results for Ilmenite Process Wastewater Treatment Sludge—Titanium Dioxide Sector Managed in Industrial Landfill

	Arsenic - Cancer		Antimony		Manganese		Thallium	
Percentile	**Adult Risk**	**Child Risk**	**Adult HQ**	**Child HQ**	**Adult HQ**	**Child HQ**	**Adult HQ**	**Child HQ**
50th	5.0e-22	4.4e-22	0.013	0.026	1.0e-04	2.0e-04	0.015	0.031
75th	2.2e-09	1.6e-09	0.078	0.16	0.070	0.15	0.18	0.38
80th	1.7e-08	1.4e-08	0.11	0.23	0.16	0.34	0.30	0.61
85th	7.5e-08	6.2e-08	0.15	0.32	0.37	0.76	0.44	0.91
90th	3.2e-07	2.3e-07	0.22	0.46	0.77	1.6	0.69	1.4
95th	1.1e-06	8.6e-07	0.35	0.75	1.6	3.3	1.1	2.4
97.5th	2.4e-06	1.7e-06	0.51	1.1	2.5	5.4	1.6	3.4
99th	4.5e-06	3.1e-06	0.74	1.6	4.1	8.6	2.4	5.2
Central Tendency	1.5e-12	2.0e-12	0.029	0.062	0.010	0.030	0.086	0.18
High End Full Distribution	1.3e-07	1.7e-07	0.029	0.061	0.23	0.48	0.31	0.66
High End Half Distribution	2.7e-07	3.6e-07	0.20	0.43	1.0	2.2	0.60	1.3

Table 6-17. Risk Results for Chloride Sulfate Wastewaters—Titanium Dioxide Sector Managed in Surface Impoundment Kerr McGee

	Arsenic - Noncancer		Arsenic - Cancer		Antimony		Molybdenum		Thallium	
Percentile	Adult HQ	Child HQ	Adult Risk	Child Risk	Adult HQ	Child HQ	Adult HQ	Child HQ	Adult HQ	Child HQ
50th	4.5e-11	8.8e-11	3.5E-15	2.9E-15	0.010	0.010	0.0020	0.0030	0.0010	0.0010
75th	1.6e-05	3.2e-05	1.2E-09	9.7E-10	0.030	0.060	0.0080	0.016	0.0040	0.0070
80th	4.5e-05	9.1e-05	3.6E-09	3.0E-09	0.040	0.090	0.011	0.024	0.0050	0.011
85th	1.1e-04	2.3e-04	9.3E-09	7.1E-09	0.070	0.14	0.020	0.040	0.0080	0.018
90th	2.7e-04	5.6e-04	2.5E-08	1.8E-08	0.12	0.24	0.030	0.070	0.015	0.031
95th	8.5E-04	0.0018	8.5E-08	6.1E-08	0.23	0.47	0.060	0.14	0.032	0.068
97.5th	0.0021	0.0044	2.0E-07	1.5E-07	0.36	0.76	0.11	0.23	0.056	0.123
99th	0.0054	0.011	4.8E-07	3.4E-07	0.57	1.3	0.18	0.42	0.098	0.218
Central Tendency	5.8E-05	1.2E-04	3.2E-09	4.3E-09	0.034	0.073	0.0083	0.018	0.0039	0.0082
High End	0.0042	0.0090	2.3E-07	3.0E-07	0.25	0.54	0.066	0.14	0.046	0.098

Table 6-18. Risk Results for Chloride Sulfate Wastewaters—Titanium Dioxide Sector Managed in Surface Impoundment Millennium HPP

	Arsenic - Noncancer		Arsenic - Cancer		Manganese	
Percentile	Adult HQ	Child HQ	Adult Risk	Child Risk	Adult HQ	Child HQ
50th	7.3e-14	1.5e-13	7.5E-18	4.8E-18	6.3E-5	1.3E-4
75th	1.3e-05	2.6e-05	1.1E-09	7.7E-10	0.0022	0.0045
80th	5.7e-05	1.1e-04	4.8E-09	3.6E-09	0.0035	0.0073
85th	1.9e-04	3.8e-04	1.5E-08	1.2E-08	0.0058	0.012
90th	5.4E-04	0.0011	4.6E-08	3.5E-08	0.0093	0.019
95th	0.0020	0.0043	1.9E-07	1.4E-07	0.017	0.036
97.5th	0.0054	0.011	5.5E-07	4.0E-07	0.028	0.059
99th	0.016	0.037	1.6E-06	1.2E-06	0.043	0.089
Central Tendency	2.7E-07	5.6E-07	1.5E-11	2.0E-11	0.0010	0.0022
High End	0.0025	0.0052	1.4E-07	1.8E-07	0.0060	0.013

**Table 6-19. Risk Results for Ilmenite Process Wastewaters—
Titanium Dioxide Sector Managed in Surface Impoundment
Du Pont De Lisle**

Percentile	Manganese		Thallium		Vanadium	
	Adult HQ	Child HQ	Adult HQ	Child HQ	Adult HQ	Child HQ
50th	2.4E-6	4.9E-6	3.9E-05	8.0E-05	1.2E-07	2.4E-07
75th	3.2E-5	6.6E-5	3.5E-04	7.3E-04	7.4E-06	1.5E-05
80th	5.4E-5	1.1E-4	5.8E-04	0.0012	1.5E-05	3.2E-05
85th	8.6E-5	1.8E-4	9.4E-04	0.0020	3.5E-05	7.2E-05
90th	1.5E-4	3.1E-4	0.0018	0.0038	8.6E-05	1.8E-04
95th	3.3E-4	7.0E-4	0.0045	0.0092	2.9E-04	6.0E-04
97.5th	6.0E-4	0.0013	0.0091	0.019	6.7E-04	0.0014
99th	0.0012	0.0025	0.020	0.044	0.0017	0.0037
Central Tendency	9.9E-7	2.07E-6	2.6E-6	5.5E-06	3.5E-09	7.3E-09
High End	2.1.E-5	4.5E-5	6.0E-05	1.3E-04	2.3E-05	4.9E-05

Table 6-20. Risk Results for Low Antimony Slag Managed in Onsite Landfill - Antimony Oxide Sector

	Antimony		Arsenic - Noncancer		Arsenic - Cancer		Boron		Selenium		Vanadium	
Percentile	**Adult HQ**	**Child HQ**	**Adult HQ**	**Child HQ**	**Adult Risk**	**Child Risk**	**Adult HQ**	**Child HQ**	**Adult HQ**	**Child HQ**	**Adult HQ**	**Child HQ**
50th	0.13	0.27	0	0	0	0	5.5e-05	1.1e-04	3.0e-05	6.3e-05	3.4e-08	6.8e-08
75th	0.62	**1.3**	2.4e-04	4.9e-04	1.9e-08	1.5e-08	2.0e-04	4.1e-04	1.1e-04	2.4e-04	4.7e-06	9.8e-06
80th	0.90	**1.8**	6.5e-04	1.33E-03	5.4e-08	4.2e-08	2.6e-04	5.5e-04	1.5e-04	3.2e-04	9.0e-06	1.9e-05
85th	**1.4**	**2.8**	0.0017	0.0034	1.4e-07	1.1e-07	3.7e-04	7.7e-04	2.2e-04	4.6e-04	1.8e-05	3.6e-05
90th	**2.2**	**4.6**	0.0040	0.0082	3.8e-07	2.7e-07	5.6e-04	0.0012	3.4e-04	7.0e-04	3.7e-05	7.6e-05
95th	**4.5**	**9.4**	0.013	0.027	1.2e-06	8.9e-07	0.0011	0.0023	6.65E-04	0.0014	1.1e-04	2.3e-04
97.5th	**8.5**	**18**	0.029	0.061	2.9e-06	2.1e-06	0.0019	0.0039	0.0012	0.0025	2.3e-04	5.0e-04
99th	**17**	**37**	0.066	0.14	7.1e-06	4.4e-06	0.0033	0.0074	0.0022	0.0048	5.0e-04	0.0011
Central Tendency	0.33	0.70	1.0e-04	2.0e-04	5.8e-09	7.7e-09	2.0e-04	3.0e-04	1.0e-04	2.0e-04	2.0e-05	4.0e-05
High End	**5.8**	**12**	0.020	0.030	9.1e-07	1.2e-06	0.0010	0.0030	9.0e-04	0.0020	1.0e-04	2.0e-04

Bold indicates a risk above a level of concern.

- High-volume nonwastewater wastes in the titanium dioxide sector showed no risk from arsenic in excess of 1E-05; however, antimony, manganese, and thallium showed HQs near or above a level of concern in the secondary gypsum waste stream and the ilmenite process wastewater treatment sludge. The secondary gypsum waste managed in an onsite landfill showed no HQ in excess of 1 at the 90th or 95th percentile. However, antimony exceeded an HQ of 1 at the 99th percentile for the child receptor. For the ilmenite process wastewater treatment sludge managed in an offsite Industrial D landfill, manganese and thallium exceeded an HQ of 1 at the 90th percentile; in addition, antimony exceeded an HQ of 1 at the 99th percentile. No exceedances of ambient water quality criteria were found for any CoC in nonwastewater wastes managed in onsite landfills in the titanium dioxide sector.

- Low antimony slag showed HQs above a level of concern for antimony; HQs exceeded 1 at the 90th percentile and increased to values in excess of 10 at the 99th percentile. However, arsenic showed no risk in excess of 1E-05, even though concentrations of arsenic in the leachate were 4,000 to 5,000 times the HBL.

The low-volume nonwastewater wastes in the hydrogen cyanide, sodium phosphate, and sodium chlorate manufacturing sectors of the inorganic chemical manufacturing industry showed no excess risk or HQ at the 90th or 95th percentile. This was attributed in part to the very low mass loadings of constituents in the landfill. The mass of constituent was so low that, in many cases, the peak concentration failed to reach the residential well within the 10,000-year time frame of the groundwater modeling. Much of the constituent was sorbed to soil particles before reaching the aquifer, leaving very low concentrations to move slowly through the aquifer to the residential well. Very low concentrations are predicted to reach residential wells downgradient from SWMUs managing most of these wastes.

One exception to this was the ammonia recycle filters in the hydrogen cyanide sector. For this waste, antimony did exceed an HQ of 1 for the child receptor at the 99th percentile. Antimony has a lower Kd than most other CoCs and, due to the higher mass loading for this waste, higher groundwater concentrations are predicted to reach residential wells. However, given the small magnitude of the exceedance (an HQ of less than 2) and the relatively low probability of occurrence (1 percent or less), risks from this waste are expected to be low.

Combined wastewaters in the hydrogen cyanide sector managed in onsite surface impoundments show no cancer risk in excess of 1E-05 for arsenic or no HQ in excess of 1 for other constituents at the 90th or 95th percentile. However, acetonitrile did exceed an HQ of 1 at the 99th percentile. Given the very small magnitude of the exceedance (an HQ only slightly in excess of 1) and the relatively low probability of occurrence (less than 1 percent), risks from this waste are expected to be low. One source of uncertainty for these wastewaters is that only inhalation exposures during showering could be assessed for acetonitrile. However, oral exposures to acetonitrile will occur concurrently with inhalation exposures if groundwater is also used for drinking water. This exposure pathway could not be assessed because the data needed to derive an oral RfD for acetonitrile were not available. Nevertheless, drinking water exposures are likely to increase risks from acetonitrile beyond what they would otherwise be.

Wastewaters in the titanium dioxide sector managed in onsite surface impoundments show no cancer risk in excess of 1E-05 for arsenic or no HQ in excess of 1 for other constituents at the 90th or 95th percentile. These waste streams exhibit relatively small mass loadings to the underlying aquifer as compared to nonwastewater wastes that are landfilled. Wastewaters are managed in surface impoundments for only limited periods of time during treatment or storage (prior to additional treatment or discharge to publicly owned treatment works [POTWs] or discharge to surface water under National Pollutant Discharge Elimination System [NPDES] permits). Antimony in the chloride process-only wastewaters in the titanium dioxide sector did exceed an HQ of 1 for the child receptor at the 99th percentile. However, given the small magnitude of the exceedance (an HQ of less than 2) and the relatively low probability of occurrence (1 percent or less), risks from this waste are expected to be low.

High-volume nonwastewater wastes in the titanium dioxide sector show no cancer risk from arsenic in excess of 1E-05; however, antimony, manganese, and thallium show HQs near or above a level of concern in waste streams managed in onsite and offsite industrial D landfills. These higher volume wastes (ilmenite process sludge and secondary gypsum) increase the mass loading of constituent in the landfill so that more constituent is available to reach the groundwater aquifer and be transported to residential wells downgradient from the SWMU. However, even these relatively high waste volumes do not contain enough antimony to generate antimony HQs above 1 except at the 99th percentile.

Low antimony slag showed an HQ for antimony well above a level of concern; however, arsenic did not. Arsenic concentrations in the leachate exceeded the HBL by 4,000 to 5,000 times; however, this waste did not show cancer risks in excess of 1E-05. There are several potential reasons for this. First, the site where the waste is landfilled is in a mountainous location with high seasonal rainfall and a very porous sand and gravel aquifer. These conditions are associated with a high infiltration rate, high hydraulic conductivity, high hydraulic gradient, and a very thick aquifer seasonally. These conditions can result in very high dilution of constituents reaching a downgradient residential well. Second, the empirical distribution of K_d values obtained from the literature for arsenic is dominated by K_d values from a single study. All the values from this study are at the high end of the distribution. (This uncertainty is discussed more fully in the uncertainty section for K_ds.) This study may disproportionately skew the distribution of groundwater concentrations to lower values and, therefore, contribute to the relatively low cancer risks for this waste.

The chemical contaminants of concern in this risk assessment that exhibit the highest risk—antimony, manganese, and thallium and their associated salts, along with acetonitrile—are associated with their own particular types of health effects. The hazard evaluation performed on these chemicals involved review of noncancer and cancer effects data. RfDs and RfCs (exposure levels that are likely to be without appreciable risk to the general population including sensitive individuals) were used as the basis for assessing noncancer risks. For these chemicals, currently accepted health risk assessment methodologies—no-effect and lowest-effect levels and associated dose-response benchmarks—were used. The only carcinogen evaluated in this risk assessment was arsenic. Epidemiological studies of human populations exposed to drinking water high in arsenic have shown increased incidence of multiple internal cancers (including cancers of the liver, kidneys, lung, and bladder) and an increased incidence of skin cancer. A

more detailed discussion of the toxicity of arsenic and other inorganic chemicals included in this risk analysis is found in Appendix J. A discussion of the human health benchmarks for acetonitrile, antimony, manganese, and thallium follows.

6.4.1 Acetonitrile

Acetonitrile (also called methyl cyanide) is highly water soluble, does not bind well to soil, and does not hydrolyze significantly. Therefore, acetonitrile can move through the soil readily and enter groundwater (U.S. EPA, 1985).

In humans, data are limited to case reports of acute acetonitrile exposure with little information on exposure level. In occupational settings, case reports have reported nausea, respiratory distress, and impaired motor activity following acute exposures. Human health effects associated with breathing or otherwise consuming smaller amounts of acetonitrile over long periods of time are not known.

Laboratory studies show that repeated exposure to high levels of acetonitrile can adversely affect the blood as well as the nervous system, the lungs, and the liver. Abnormal histopathology was largely limited to the lungs and included congestion, hemorrhage, and edema (U.S. EPA, 1999b). The lethal effects of acetonitrile are thought to be associated with metabolism of ACN to form cyanide, leading to respiratory paralysis and inhibition of the central nervous system. Evidence from animal studies also shows that high levels (1,800 ppm) of acetonitrile can adversely affect reproductive success (i.e., increases in nonlive implants and early resorptions). Reproductive effects have not been found in the more recent studies.

The principal types of oral exposure studies are the reproductive/developmental and lethality studies. A study involving gavage doses reported no effects on fertility, nor were fetal anomalies reported (U.S. EPA, 2000c). Slight decreases in fetal body weight were seen in all exposure groups, although not dose related (U.S. EPA, 2000c). Acetonitrile caused thin stomach walls in the cardiac region of rabbits that died from gavage exposure during gestation days 6-18 (U.S. EPA, 2000c, citing Argus Research Labs, 1984), although other reproductive and developmental effects were not seen. EPA has no oral reference dose for acetonitrile due to the lack of suitable studies for determining an RfD.

The RfC of 0.06 mg/m^3 for acetonitrile is based on a NOAEL of 60 mg/m^3, an uncertainty factor of 100, and a modifying factor of 10 (U.S. EPA, 2000c). The NTP subchronic (13-week) study in the mouse, supported by the results of a follow-on chronic study, was the principal study and involved B6C3F1 mice in 10 mice per sex per group exposed whole-body to acetonitrile concentrations of 0, 100, 200, 400, 800, or 1,600 ppm (2,686 mg/m^3) for 6 h/d, 5 d/wk. All animals in the 1,600-ppm group died within 4 weeks. Although final body weights were significantly reduced at 400 ppm in males, these results were not considered toxicologically significant. Liver changes included absolute and relative weight increases and hepatocellular vacuolation. None of these effects were considered biologically significant. Incidences of forestomach squamous epithelial hyperplasia were significantly increased in the 800-ppm males and the ≥200-ppm females. Hyperkeratosis and inflammatory cell infiltrate (hyperplasia-

associated effects) were also found. In the highest exposed female group, significant increases were found for focal ulcers in the forestomach. No lung effects were reported.

Mortality of 1 in 10 female mice exposed to 400 ppm acetonitrile was the critical effect. There was uncertainty associated with the mice forestomach hyperplasia and in combination with the other liver effect uncertainty, and, because of a lack of forestomach hyperplasia in the complementary rat study, there was no unambiguous NOAEL. Although mortality is not usually selected as the critical effect, it appears to be the most appropriate effect for acetonitrile. A steep exposure-response curve is consistent with other cyanide-containing chemicals.

The RfC was derived in accordance with EPA's regional deposited gas ratio (RDGR) method. Acetonitrile is considered a category 2 gas because it is highly soluble in water, is metabolized to cyanide in the liver, and does not react directly with respiratory tissues. The RDGR for these gases is 1, and, when the exposure level is adjusted for duration, the NOAEL of 200 ppm becomes 60 mg/m^3.

No uncertainty factor was applied for use of a subchronic study, because lethality did not occur at lower levels in the longer-term mouse or rat studies. Given known metabolism of cyanide-containing compounds, increased exposure is not expected to increase mortality. The database insufficiency UF was based on the lack of data on reproductive endpoints in combination with the understanding that acetonitrile does not accumulate in the body, the developmental effects observed seem to be marginal, and these effects occur at concentrations lethal to dams. The modifying factor of 10 represents the likelihood that exposure may be the result of grooming of contaminated fur. Contributions to forestomach lesions are less likely due to exposure from direct inhalation.

Evidence suggests that acetonitrile is not a carcinogen. In an NTP study (U.S. EPA, 1999b, citing NTP, 1996) rats and mice showed no significant evidence of cancer or mutagenicity. Although there were positive trends seen in the incidence of adenoma, carcinoma, or a combination of the two in livers of rats (male), no significant dose-related trend was found.

6.4.2 Antimony

The reference dose for antimony is 4.0E-04 mg/kg-d based on a LOAEL of 0.35 mg/kg-d, an uncertainty factor of 1,000, and a modifying factor of 1 (U.S. EPA, 2000c). The RfD was based on a study using male and female rats exposed to 0 or 5 ppm (0.35 mg/kg-d) potassium antimony tartrate in water (U.S. EPA, 2000c, citing Schroeder et al., 1970). The critical effects identified for this study are decreased longevity and blood glucose levels and altered cholesterol levels (U.S. EPA, 2000c). Because only one level of antimony was administered, a NOAEL could not be established in the study. An uncertainty factor of 1,000 was applied based on a tenfold factor for extrapolation from animals to humans, a tenfold factor to protect sensitive individuals, and an additional tenfold factor for use of a LOAEL (U.S. EPA, 2000c). For antimony, EPA assigned a low confidence rating for this RfD, because only one species and one dose level were used, a NOAEL was not determined, and gross pathology and histopathology were not well described.

In a similar study (U.S. EPA, 2000c, citing Kanisawa and Schroeder, 1969), groups of CD-1 mice (54/sex) were given potassium antimony tartrate in drinking water at 0 or 5 mg/L (5 ppm) for 540 days (18 months). Lifespans were significantly reduced in both males and females, but the degree of antimony toxicity was less severe in mice than rats. (U.S. EPA, 2000c, citing Bradley and Fredrick, 1941, and Browning, 1969) reported disturbances in glucose and cholesterol metabolism in rats ingesting 5 mg/L antimony, but no signs of injury to the heart were observed in rats receiving doses up to 100 mg/kg-d.

Cardiovascular and gastrointestinal health effects appear to be the primary concern for oral exposure to antimony following acute exposures. In addition, myocardial effects have been observed in occupational studies following inhalation exposures. Reproductive effects (spontaneous abortion and premature delivery) have also been observed in female workers.

Myocardial effects are among the best-characterized human health effects associated with antimony exposure. Studies by (U.S. EPA, 2000c, citing Brieger et al., 1954) suggest an inhalation NOEL for myocardial damage to be approximately 0.5 mg/m^3. This exposure is approximately equivalent to an oral dose of 0.003 mg/kg body weight/day. Parallel studies in rats and rabbits resulted in observation of EKG alterations following exposure to 3.1 to 5.6 mg/m^3. There are, however, no adequate data on oral exposure to antimony that permit a reasonable estimate of a no effects level regarding heart damage. One study (U.S. EPA, 2000c, citing Belyaeva, 1967) indicated that women workers exposed in an antimony plant experienced a greater incidence of spontaneous abortions than did a control group of nonexposed working women. A high rate of premature deliveries among women workers in antimony smelting and processing was also observed (U.S. EPA, 2000c, citing Aiello, 1955).

EPA has not undertaken a complete evaluation and determination of evidence for human carcinogenic potential for antimony.

6.4.3 Manganese

Manganese is an essential element in humans, with an estimated safe and adequate daily dietary intake of 2 to 5 mg/d for adults and adolescents. No cases of manganese deficiency have been observed in the general population.

The RfD for manganese is 0.14 mg/kg-d, based on a NOAEL of 0.14 mg/kg-d and an uncertainty factor of 1 for ingestion in the diet (U.S. EPA, 2000c). For ingestion in drinking water or in soil ingestion, a modifying factor of 3 is applied. The RfD, therefore, is 0.047 mg/kg-d for manganese in drinking water in this risk assessment. The RfD was based on many studies of daily consumption of manganese in the human population (U.S. EPA, 2000c, citing Freeland-Graves et al., 1987, NRC, 1989, and WHO, 1973). As reported in U.S. EPA (U.S. EPA, 2000c, citing NRC, 1989), the National Research Council determined an estimated safe and adequate daily intake (ESADDI) of manganese to be 2 to 5 g/d for adults. The NRC considered 10 mg/d to be safe for occasional intake. The World Health Organization (U.S. EPA, 2000c, citing WHO, 1973) reviewed several studies of adult diets and reported the average daily consumption of manganese to range from 2.0 to 8.8 mg/d and concluded that 2 to 3 mg/d is adequate for adults and 8 to 9 mg/d is “perfectly safe.” From all this, EPA concluded that an

appropriate reference dose for manganese for dietary exposures is 0.14 mg/kg-d (10 mg/d) (U.S. EPA, 2000c).

An uncertainty factor of 1 reflects information taken from normal long-term diets with no adverse health effects and the fact that manganese is an essential element in the diet. A modifying factor of 1 was applied for assessing exposure to manganese from food; however, a modifying factor of 3 is recommended when assessing exposure from drinking water or soil. There is some increased uptake of manganese from water in fasting individuals. An epidemiological study by Kondakis et al. (U.S. EPA, 2000c, citing Kondakis et al., 1989) of manganese in well water used for drinking water in Greece raises the possibility of adverse health effects associated with lifetime consumption of drinking water containing about 2 mg/L of manganese. Much higher concentrations of manganese in infant formula and possibly increased manganese absorption, reduced excretion, and increased passage between blood and brain by neonates are all of concern (U.S. EPA, 2000c). According to EPA, these considerations, in addition to the likelihood that any adverse neurological effects of manganese are likely to be irreversible and not manifested for many years after exposure, warrant caution until more definitive data are available.

Manganese has been shown to be a neurotoxin with oral exposure in both humans and animals, although the evidence is limited. Drinking water was a source of manganese exposure in six Japanese families and caused manganism-like symptoms. Contamination was reported at 14 mg/L and several people were affected severely (U.S. EPA, 2000c, citing Kawamura et al., 1941). More recent examples, involving aboriginal populations in Australia and Israelis with Parkinsonism, support the evidence for the connection between manganese and neurological effects (ATSDR, 1997).

The epidemiologic study of manganese in drinking water (U.S. EPA, 2000c, citing Kondakis et al., 1989) involved three areas in northwest Greece. Manganese concentrations in natural well water were 3.6 to 14.6 µg/L in area A, 81.6 to 252.6 µg/L in area B, and 1,600 to 2,300 µg/L in area C. The total population of the three areas studied ranged from 3,200 to 4,350 people. The study included only individuals over the age of 50 drawn from a random sample of 10% of all households (n=62, 49 and 77 for areas A, B and C, respectively). The authors reported that "all areas were similar with respect to social and dietary characteristics," but few details were reported. Although the amount of manganese in the diet was not reported, the authors indicated that food intakes are expected to be comparable for all three areas. The individuals chosen were submitted to a neurologic examination, the score of which represents a composite of the presence and severity of 33 symptoms (e.g., weakness/fatigue, gait disturbances, tremors, dystonia). Whole blood and hair manganese concentrations also were determined. The mean concentration of manganese in hair was 3.51, 4.49, and 10.99 µg/g dry weight for areas A, B and C, respectively ($p<0.0001$ for area C versus A). The difference in mean neurologic scores for area C versus A was significantly increased (Mann-Whitney $z=3.16$, $p=0.002$ for both sexes combined). In a subsequent logistic regression analysis, the authors reported that there is a significant difference in neurologic scores between areas A and C even when both age and sex are taken into account.

The individuals examined in the Kondakis study (U.S. EPA, 2000c, citing Kondakis et al., 1989) also had exposure to manganese in their diet. This was originally estimated to be 10 to 15 mg/d but was subsequently lowered to 5 to 6 mg/d. Because of the uncertainty in the amount of manganese in the diet and the amount of water consumed, it was not possible to estimate the total oral intake of manganese in this study. This study, nevertheless, raises significant concerns about possible adverse neurological effects at doses not far from the range of essentiality. Because of this concern, EPA recommends that a modifying factor of 3 be applied when assessing risk from manganese in drinking water or soil.

EPA has assigned a ranking of medium confidence to the RfD because many studies have reported similar findings with regard to the normal dietary intake of manganese in humans and because there is no single study used to derive the RfD for manganese; however, no quantitative information is available to indicate toxic levels of manganese in the diet of humans.

No studies are available regarding carcinogenic effects in humans or animals from inhalation exposure to manganese nor are studies available regarding cancer in humans from oral exposure to manganese. Several oral animal studies reported negative results: one study reported an increased incidence of thyroid gland follicular cell adenomas and hyperplasia, and one study noted an increased incidence of pancreatic tumors, all from exposure to manganese sulfate (ATSDR, 1997). EPA has Classified manganese as Group D, Not Classifiable as to Human Carcinogenicity (U.S. EPA, 2000c).

6.4.4 Thallium

Studies on workers exposed to high levels of thallium by inhalation indicate that it may affect the central nervous system, with effects such as paresthesia, numbness of toes and fingers, “burning feet,” and muscle cramps. Ingestion of thallium has been associated with hair loss in humans. Hair loss was reported to be temporary, and no skin changes were reported. Peripheral neuropathy was reported in cases of thallium poisoning in China. In animal studies, hair loss, nervous system effects, and abnormalities in testicular morphology have been reported from thallium exposure (ATSDR, 1992).

The RfD for thallium is 8.0 E-05 mg/kg-d based on a NOAEL of 0.25 mg/kg-d, an uncertainty factor of 3,000, and a modifying factor of 1. The RfDs were based on a 90-d study in which rats were treated by gavage with thallium sulfate at doses of 0, 0.01, 0.05, or 0.25 mg/kg-d (0, 0.004, 0.02, or 0.10 mg thallium/kg-d) (U.S. EPA, 1986) (U.S. EPA, 2000c, citing U.S. EPA, 1986). No differences between the control groups and the exposed groups were observed in body weights, body weight gain, food consumption, or organ weights. Moderate dose-related changes were observed in some blood chemistry parameters (increased SGOT, LDH, and sodium and decreased blood sugar). Increased levels of the circulating enzymes (SGOT and LDH) may be indicative of liver or other disease. The only grossly observed finding was alopecia; however, microscopic evaluation did not reveal any histopathologic alterations. The highest dose was selected as a NOAEL (U.S. EPA, 2000c). An uncertainty factor of 3,000 was applied based on a tenfold factor to extrapolate from subchronic to chronic data, a tenfold factor for extrapolating from animals to humans, a tenfold factor for sensitive human subpopulations, and a threefold factor to account for lack of reproductive and chronic toxicity data (U.S. EPA, 2000c).

For thallium, EPA has assigned a ranking of low confidence to the RfD (U.S. EPA, 2000c) because of uncertainties in the results and because supporting studies show adverse health effects at doses slightly higher than the NOAEL and because the information base provides only one subchronic study and some anecdotal human data.

Human case studies suggest that the nervous system is susceptible to thallium toxicity from oral exposure at high doses. At lower levels, the human evidence is not available, but animal studies with intermediate exposure (240 days) showed changed motor sensory capacities. Thallium may be a greater developmental and reproductive toxin from oral exposure (ATSDR, 1992).

EPA classifies thallium as Group D, Not Classifiable as to Human Carcinogenicity, based on the lack of carcinogenicity data in animals and humans.

There is the potential that exposure to manganese and thallium in combination could result in greater harm than that for either metal individually. One endpoint of possible concern is central nervous system (CNS) effects and respiratory irritation, inflammation, and disease. Responses of the human body to manganese and thallium involve the CNS, but the specific responses appear to involve somewhat different manifestations, i.e., problems of coordination for manganese and more peripheral nerve changes for thallium exposure. These effects have been identified through inhalation pathway. Although the potential for additive or potentiated responses exists, drawing conclusions as to their cumulative risk is highly problematic because of the lack of focused research and methodologies for these inorganic chemicals.

6.5 Uncertainty Analysis

EPA typically classifies the major areas of uncertainty in risk assessments as parameter uncertainty, scenario uncertainty, and model uncertainty. Parameter uncertainty is the "uncertainty regarding some parameter" of the analysis. Scenario uncertainty is "uncertainty regarding missing or incomplete information needed to fully define exposure and dose." Model uncertainty is "uncertainty regarding gaps in scientific theory required to make predictions on the basis of causal inferences" (U.S. EPA, 1992). This section identifies the primary sources of each of these types of uncertainty in the inorganic chemical manufacturing waste listing risk assessment and qualitatively describes how each may influence the results of the risk assessment.

6.5.1 Parameter Uncertainty

The sources of parameter uncertainty are measurement errors, sampling errors, variability, and use of generic or surrogate data (U.S. EPA, 1992). Many of the parameters used to quantify contaminant fate and transport and contaminant exposure and dose either were not measured or could not be measured precisely and/or accurately. Some of the most important and sensitive parameters in this analysis were those that describe waste composition; waste management practices; site characteristics (e.g., hydrogeological, topographical, meteorological, and soils data); the physiologic and behavioral exposure characteristics of the receptors; the physical, chemical, and biochemical properties of the contaminants; and toxicological effects.

The primary sources of parameter uncertainty in the inorganic chemical manufacturing risk assessment include the following:

- The risk analyses were based on a limited set of waste sample (concentration) data. The objective of sampling is to characterize the waste produced by a particular industrial process. If the available samples are not truly representative of the waste, risk could be underestimated or overestimated. In addition, leaching of chemical contaminants from landfills was characterized using the TCLP leach test for municipal landfills and SPLP leach test for Industrial D landfills. However, leaching behavior may differ from that indicated in the leach tests depending on waste- and site-specific conditions.

- EPA obtained site-specific information regarding the design and operation of onsite waste management units used by the inorganic chemical manufacturing industry from the 3007 Survey; however, it was necessary to make assumptions concerning waste management in offsite landfills. A number of the facilities reported using offsite nonhazardous landfills to dispose of sludges and filter wastes in several sectors. Modeled landfills were either municipal landfills or industrial D landfills as reported used in the 3007 Survey. However, it was assumed that the distribution of the surface areas of all offsite landfills was represented by the distribution of municipal landfill areas.

- Regional databases were used to obtain the parameter values for the soil and aquifer conditions in the vicinity of all offsite SWMUs modeled in this risk assessment. For onsite SWMUs, when site-specific data were available, uncertainty was reduced but not eliminated. Use of these databases and limited site-specific data may result in either overestimates or underestimates of risk.

- Empirical data were used to characterize partitioning of chemical contaminants between the aqueous phase and soil and aquifer materials. The K_d values used in this risk analysis are based on values found in the literature. The values for all constituents are assumed to range over at least 3 orders of magnitude. For values with five or fewer literature values available for establishing a distribution of K_d values, a lognormal distribution is assumed centered on the mean value of the available log K_ds and extending for 1.5 log units on each side of the log mean. This uncertainty could result in either an underestimation or an overestimation of risk.

- Sources of uncertainty in toxicological benchmarks include one or more of the following: extrapolation from laboratory animal data to humans, variability of response within the human population, extrapolation of responses at high experimental doses under controlled conditions to low doses under highly variable environmental conditions, and adequacy of the database (number of studies available, toxic endpoints evaluated, exposure routes evaluated, sample sizes, length of study, etc.). Toxicological benchmarks are designed to be conservative

(that is, overestimate risk) because of the uncertainties and challenges associated with condensing toxicity data into a single quantitative expression.

- # CSFs can vary by several orders of magnitude depending on the extrapolation model used. A limited understanding of cancer biology in laboratory animals and humans adds to the uncertainty of identifying true human carcinogens. The primary carcinogenic CoC in this risk analysis is arsenic. Arsenic is a known human carcinogen.

6.5.2 Scenario Uncertainty

The sources of scenario uncertainty are descriptive errors, aggregation errors, errors in professional judgment, and incomplete analysis (U.S. EPA, 1992). Scenario uncertainty results from assumptions made concerning how receptors become exposed to contaminants and occurs because of the difficulty and general impracticality of making actual measurements of a receptor's exposure.

EPA assumed that waste management units (both onsite and offsite) were not lined and had no leachate collection systems. The effectiveness of liners and leachate collection systems in preventing chemical contaminants from leaching into groundwater over long time periods is uncertain.

For offsite waste management units (e.g., municipal landfills and industrial D landfills), EPA obtained data from a survey of municipal landfills to develop a distribution of distances to the nearest downgradient residential well and used that distribution to characterize the distance to receptor wells for both municipal landfills and offsite industrial D landfills. For onsite waste management units, EPA used the information that was available on the site to determine the general direction of groundwater flow and the closest possible distance to a residential well. Receptor wells were assumed to occur over a uniform range from this distance out to 1 mile from the waste management unit and within the lateral extent of the groundwater plume. The direction of groundwater flow and the location of residential drinking water wells are important sources of uncertainty in the risk analysis.

This analysis did not include all exposure scenarios. Two scenarios not included in this analysis were

- # Evaluation of risks to infants (age 0 to 1)
- # Evaluation of indoor exposure to household water uses besides showering.

As discussed previously in this document, evaluation of these additional scenarios or pathways would likely increase the estimates of risk, but the increases would likely be small and not impact the major findings of the risk assessment.

Exposure modeling relies on default assumptions concerning population activity patterns, mobility, and other factors. These default assumptions may be a source of aggregation error because it was assumed that the populations that reside near the inorganic chemical waste

management units were homogeneous and representative of the national population. The *Exposure Factors Handbook* (U.S. EPA, 1997b) is the source of the most current data concerning exposure factors for the national population and was used for modeling exposure in this risk assessment. To the extent that actual exposure could vary from these assumptions, risks could be underestimated or overestimated.

6.5.3 Model Uncertainty

The sources of model uncertainty are relationship errors and modeling errors (U.S. EPA, 1992). Models and their mathematical expressions are simplifications of reality that are used to approximate real-world conditions and processes and their relationships. Models do not include all parameters or equations necessary to express reality because of the inherent complexity of the natural environment and the lack of sufficient data to fully describe it. Consequently, models are based on various assumptions and simplifications and reflect an incomplete understanding of natural processes. The models selected for use in this risk assessment are described in Section 4 of this document. The selection was based on science, policy, and professional judgment. The groundwater model and the surface impoundment infiltration models were selected because they provided the information needed for this analysis and because they are thought to be state-of-the-science.

Even though the models used in the risk analyses are used widely and have been accepted for numerous applications, they each retain significant sources of uncertainty. These include the following:

- EPACMTP (used to model groundwater fate and transport) does not model colloidal transport nor does it model possible geochemical interactions among different contaminants in the leachate and the subsurface environment. The EPACMTP modeling incorporates the following assumptions: (1) transverse dispersion is negligible in the unsaturated zone, potentially resulting in an overestimation of risks; (2) receptors use the uppermost aquifer rather than a deeper aquifer as a domestic source of drinking water, which overestimates risks where the uppermost aquifer is not used; and (3) hydrogeologic conditions that influence contaminant fate and transport are uniform spatially (i.e., no heterogeneity or fractured flow) as well as uniform temporally (i.e., over the 10,000-year time frame modeled), potentially resulting in underestimation or overestimation of receptor well concentrations.

- The infiltration rates used in this analysis were developed using the HELP model and rely on regionalized climatic data and generalized soils data. These are not site-specific data but are intended to represent the range of conditions expected in the area. The surface impoundment infiltration model accounts for uncertainty using the probabilistic mode to produce a distribution of infiltration rates reflecting the variability of the input parameters. The variable parameters include sludge depth, underlying soil properties, depth to the water table, and aquifer parameters.

Evaluated as a whole, the sources of model uncertainty in our analysis could result in either an overestimation or underestimation of risk.

6.6 Uncertainties of Key Findings

Uncertainty and variability are associated with all risk analyses. This risk analysis used many waste-stream-specific and site-specific data in an effort to reduce the range of uncertainty and variability. The probabilistic analysis was intended to take the remaining parameter variability into account to the extent possible. The results of the probabilistic analysis provide a distribution of risks that reflect that variability and uncertainty. The statistically based sensitivity analysis aided in understanding the interactions of these parameters and was used to help identify the parameters having the greatest influence on the modeled groundwater concentrations and associated risks.

The waste streams managed at a given location were characterized by a single waste volume reported by the facility generating the waste. This waste volume was assumed constant over the lifetime of the waste management unit. The waste concentration and the leachate concentrations were also characterized by a single sample from the waste and were assumed constant over the life of the waste management unit. Parameters such as these were not included in the sensitivity analysis because they did not vary in this risk analysis. However, the sensitivity analysis was used to determine which of the variables in the analysis were associated with the greatest change in risk. K_d in the aquifer and the unsaturated zone are the most common sensitive parameters.

Partition coefficients reported in the literature and presented in the K_d database are subject to a variety of uncertainties. Many previous studies have demonstrated that, in a variety of soils and for a variety of metals, partition coefficients vary with pH and with the concentration of sorbing phases in the soil matrix (e.g., weight percent organic matter content, weight percent hydrous ferric oxides, and corresponding oxides of aluminum and manganese) (Janssen et al., 1997; Hassan and Garrison, 1996; Bangash et al., 1992; Anderson and Christensen, 1988).

It is well known that dissolved ligands present in soil porewater (e.g., dissolved organic matter, anthropogenic organic acids) may complex with metals, reducing their propensity for sorption in proportion to the concentration of the ligands (Christensen et al., 1996). Within the population of soils, the natural variability in soil pH and in the composition of soil and its associated porewater results in variation in K_d over orders of magnitude, even for a single metal. For this reason, any comprehensive compilation of K_d values selected from the literature should present values that define a distribution. In fact, for a particular metal, K_d depends on these and other characteristics of the soil/porewater system, and, in a regional risk assessment, it is desirable to include the regional population of soil/porewater systems to obtain a frequency distribution of K_d specific to that region.

Apart from uncertainties in representing the expected variation in K_d that arise from variation in soil/aquifer properties, there are significant uncertainties associated with individual K_d values. Sources of uncertainty in individual literature K_d values include the following:

- Detection limits in measuring metal concentrations may result in limiting the observed maximum K_d value.

- Equilibrium conditions may not have existed in the experiment for measuring media concentrations. Most batch experiments are carried out over a time span of 1 or 2 days. Equilibrium may or may not have been attained, and unaccounted for nonequilibrium processes may have occurred.

- Some variability in collected K_d values may reflect variability in the different methods of measurement (e.g., batch experiments, measurements from natural soil and associated porewater, calculation from tracer/retardation studies).

- Some variability in collected K_d values may reflect variability in extractants used in batch tests. Some researchers used soil porewater or groundwater as the extractant. Others used distilled water or a solution of electrolyte. The modeling in which these K_d values are to be used may implicitly prescribe an extractant that is dissimilar to any actually used in literature studies (e.g., landfill leachate).

- Some uncertainty in the reported K_d values is associated with uncontrolled or unknown redox conditions during the course of experimental measurements, especially for redox-sensitive metals (e.g., Cr, As, Se).

- Some uncertainty in the K_d values is due to neglecting the impact of total system concentration of metal on the magnitude of K_d. Numerous studies have documented the dependence of K_d on total metal concentration—K_d tends to decrease as the total metal concentration increases. No attempt has been made in this compilation of literature values to investigate or represent the dependence of K_d on metal concentration. Instances in cited references of the use of Freundlich isotherms to represent such a dependence have been treated by computing the K_d appropriate for a dissolved metal concentration of 1 ppm. The K_d values compiled here are likely to be more representative of those in systems with low metal concentration than those in systems with high metal concentration.

The goal of the literature collection effort was to develop partition coefficient distributions that represent the national or regional populations. Unfortunately, the collection of soil/porewater systems chosen for study by various researchers and reported in the literature are unlikely to be representative of the national or regional population of soil/porewater systems, and collections of K_d values obtained from the literature are unlikely to be representative of K_d for a particular metal under all conditions. Furthermore, the degree to which the soil systems reported in the literature adequately represent the population of soils varies greatly among the different metals for which K_d values have been obtained.

Depending upon the number of measured K_d values compiled from the scientific literature, two different approaches were used to develop partition coefficient distributions: loguniform and empirical treatments. For the loguniform treatment, the average of the collected log K_d values was assumed to define a central tendency value, and the minimum and maximum

were established as 1.5 log units below and above the average value. For the empirical treatment, the K_d values were simply assumed to represent the true frequency distribution.

For any particular metal, the degree to which these two methods of establishing the frequency distribution of K_d is indicative of the true frequency distribution of K_d is unknown. It may be assumed that, as the number of values contained in a distribution increases, a closer approximation to the national distribution is achieved. Other contributing factors include the number of individual studies that were used to compile K_d values and the number of soil types. The greater the variability, the more likely that a better approximation is achieved. Generally speaking, it may be appropriate to assume that the frequency distributions for K_d represented by the empirical method are more representative of the true national frequency distributions because they include more sampled K_d values and greater variabilities.

The following is a brief summary of the distributions used in this risk assessment on a metal-by-metal basis.

- ***Acetonitrile.*** For organic constituents like acetonitrile, the K_d is assumed to be the product of the organic carbon partition coefficient (Koc) and the fraction organic carbon (foc) of the soil.

- ***Arsenic.*** Arsenate (As V) and arsenite (As III) are the primary forms of arsenic in nature. Arsenate is the predominant form in well-oxidized environments, whereas arsenite occurs predominantly under reduced conditions. However, due to relatively slow redox transformations, both species are often present in either redox environment.

  One of the objectives of the literature search was to provide unique K_d values for all relevant oxidation states. Although a total of 35 K_d values were compiled for arsenic from the literature, only 3 of the 10 references provided sufficient information to determine oxidation state. Although limited information was available concerning oxidation state of arsenic in the environment, the prevalent species of arsenic present in the waste stream were unknown. Furthermore, site-specific geochemical redox conditions were not characterized. Hence, arsenic was treated independent of oxidation state.

  Five of the arsenic K_d values characterize aquifer conditions consisting primarily of silty sand sediment with pH approximately equal to 7. The remaining 30 K_d values approximate soil conditions having pH values that range from 5.3 to 11. Particulate organic carbon content ranged from 0.34 to 2.8 weight percent, and clay content was reported as 6.5 weight percent for one group of 20 soil samples collected throughout the Netherlands.

  Although the K_d values approximating aquifer conditions are generally less than those characterizing soil conditions, the aquifer K_d values vary over 5 orders of magnitude, whereas the soil K_d values vary only over 3 orders of magnitude. The highest and most closely matched K_d values were associated with the 20

Netherlands soil samples. This reflects the similarity in experimental conditions among the Netherlands samples. The K_d values from the Netherlands soil samples reflect the release of arsenic from aged contaminated soils. In this risk assessment, K_d is used to estimate the fraction of constituent in the leachate that is sorbed to soil and aquifer particles from the dissolved phase. This study of Netherlands soil samples may, therefore, skew the distribution of K_ds to the high end of the range of the reported literature values. This could have an effect on the modeled well water concentrations by lowering the arsenic concentrations in the dissolved phase.

- ***Manganese.*** The most stable and dominant oxidation state of manganese in the environment is Mn II. Manganese is sorbed by many components in soils, including clay, organic matter, and iron oxides. Twelve K_d values for manganese were compiled from four references. Three of the K_d values approximate soil conditions, and nine approximate aquifer conditions. The aquifer samples range from sandy till to heavy clay. There was no correlation between pH and K_d for the dataset. Data are insufficient to determine if possible correlations between K_d and other soil and aquifer parameters existed.

- ***Antimony.*** Antimony is characterized by four oxidation states (-III, 0, +III, and +V). In oxidizing environments, $Sb(OH)_6^-$ is the dominant species for pH values greater than 3. The anionic character of antimony suggests that it would not be highly sorbed under alkaline or oxidizing conditions. However, as the pH decreases to weakly acidic conditions, adsorption reactions may increase in importance. Only two measured K_d values were found in the scientific literature for antimony and a loguniform distribution was developed.

- ***Thallium.*** Distribution coefficients describing the behavior of thallium were not available in the scientific literature for either soil or aquifer conditions. A loguniform distribution would have been assumed for metals characterized by five or fewer literature K_d values. Given the absence of a single measured K_d value, a different approach was taken. This approach relied on a study conducted at the EPA Laboratory in Athens, Georgia (Loux et al., 1990). Distribution coefficients were measured for aquifer/groundwater samples collected from six states (Wisconsin, Oregon, Florida, Texas, Utah, and New Jersey). The samples were subjected to acid-base additions so that K_d could be measured as a function of pH. pH values ranging from 2 to 11 were used in the study. The resultant K_d ranged from 0 to approximately 3 L/kg for thallium. Because this was the only instance of measured K_d values found for thallium, this range was used to define a loguniform distribution.

6.6.1 Well Location

The second parameter that appears with equal frequency is the location of the residential well (y-well) on the center line of the downgradient contaminant plume. In this risk assessment, the location of the residential well is constrained within the lateral extent of the downgradient

plume. Y-well is a sensitive parameter for all the constituents of greatest concern in this risk assessment and is the only parameter that is one of the two most sensitive parameters for all constituents in both onsite and offsite landfills. The location of the well within the lateral extent of the plume is a more sensitive parameter than the longitudinal distance of the residential well from the source. The longitudinal distance from the source to the well (x-well) appears as one of the two most sensitive parameters only twice and, in both cases, it is for constituents in the ilmenite process wastewater treatment solids waste stream, which is managed in an offsite Industrial D landfill. For offsite facilities, the x-well location of residential wells is characterized by the distribution of closest residential drinking water wells developed from a survey of municipal landfills. The wells in this distribution are assumed to be closer to the landfill than is assumed for the onsite landfills in this risk assessment, where the facility property boundary or nearest residence is assumed to be the closest x-well location possible for a residential well. The well location is constrained within the downgradient plume in both the x and y directions.

6.6.2 Drinking Water Intake Rate

The intake of drinking water also appears twice as one of the two most sensitive parameters. In both cases, it is for the constituent thallium. The uncertainty associated with this parameter is characterized by the lognormal distribution developed from the data in the EFH on the variability of drinking water consumption.

6.6.3 Surface Area of SWMU

The area of the waste management unit appears as one of the two most sensitive parameters in the case of manganese in ilmenite process wastewater treatment solids managed in an offsite Industrial D landfill. The area of the SWMU was varied only when wastes are managed in offsite management units where the size of the unit is not known. The distribution of municipal landfill sizes developed from a survey of municipal landfills was used to characterize the distribution of landfill areas for commercial Industrial D landfills in this risk assessment. The capacity of the modeled landfill was checked to be sure that the landfill could hold the volume of waste assumed sent to the facility over a period of 30 years, thus reducing the range of SWMU areas to the high-end of the distribution for very high-volume waste streams, such as the ilmenite process wastewater treatment sludge.

6.6.4 Thickness of Vadose Zone

The thickness of vadose zone appears once as one of the two most sensitive parameters for antimony in sulfate process milling sand managed in an offsite Industrial D landfill. This is a low-volume waste with a low mass loading of the CoC antimony in the landfill. The antimony is significantly sorbed to the soil in spite of its relatively low K_d, and, thus, when the unsaturated zone is thicker, the mass of constituent reaching the water table is much less than when the unsaturated zone is thinner. The variability of the thickness of the unsaturated zone in this analysis was high because this waste is managed in an offsite Industrial D landfill that was assumed to be located anywhere within a 100-mile radius of the SWMU currently receiving the waste. This variability contributes to the uncertainty concerning the depth of the unsaturated zone used for modeling this waste.

Three additional analyses have been performed to investigate additional uncertainty issues.

6.6.5 Waste Volume for Ilmenite Sludge

There is significant uncertainty about the volume of waste in the ilmenite process sludge waste stream that may be considered exempted from hazardous waste regulation under the Bevill exemption. If the total volume of waste is considered to be nonexempt, then the risk results for this waste stream indicate the potential for these wastes to exceed the nominal risk level of concern. However, because of the uncertainty concerning the volume of nonexempt waste subject to the listing determination, an analysis was performed using a waste quantity that is 10 percent of the total quantity of waste generated. This scenario corresponds to one in which 90 percent of the annual waste volume is assumed to be Bevill-exempt and is disposed of in a separate SWMU located elsewhere. This analysis showed that the concentrations of chemical contaminants at a residential well are reduced; however, the magnitude of the reduction was not as great as the reduction in the volume of waste as generated. The hazard quotients for the waste as generated and the hazard quotients for the 10 percent waste volume and the ratio of the two are presented in Tables 6-21, 6-22, and 6-23. The groundwater DAFs for the waste as generated and the DAFs for the 10 percent waste volume and the ratio of the two are presented in Tables 6-24, 6-25, and 6-26. The results show that the high-end DAFs are lower by about a factor of 2.

6.6.6 Aquifer Parameters for Chloride and Sulfate Process Onsite Landfill

The aquifer parameters for the onsite landfill at the Millennium HPP facility were characterized using regional data from the HGDB database. The following waste streams were modeled as managed in this landfill:

- Sulfate process: digestion sludge
- Sulfate process: secondary gypsum
- Combined chloride sulfate process wastewater treatment sludge.

However, when the onsite surface impoundment at this same facility was modeled, aquifer data specific to that locale were used to describe the underlying aquifer. This was done to address the constraints imposed by the surface impoundment infiltration model. The surface impoundment infiltration model restricts infiltration to the aquifer to prevent mounding of the groundwater above the bottom of the surface impoundment. Using the regional data for the aquifer limited surface impoundment infiltration to unrealistically low levels. The landfill infiltration model does not limit infiltration in this way. However, because the landfill and surface impoundment are proximate to one another, there is no reason why the aquifer parameters should differ. Table 6-27 compares the regional aquifer data originally used to model the landfill with the site-specific aquifer data used to model the onsite surface impoundment.

To investigate the effect of using the site-specific aquifer data to model the onsite landfill, antimony in secondary gypsum waste was modeled using the same site-specific aquifer parameters used for the onsite surface impoundment. The risk results obtained using the site-specific aquifer data were compared to the results obtained using the regional aquifer data from

Table 6-21. Comparison of Risk Results for Antimony in Ilmenite Process Wastewater Treatment Sludge for 100 Percent and 10 Percent Waste Quantities

Percentile	100% Waste			10% Waste			Ratio		
	Groundwater Concentration (mg/L)	Adult HQ	Child HQ	Groundwater Concentration (mg/L)	Adult HQ	Child HQ	Groundwater Concentration (mg/L)	Adult HQ	Child HQ
50th	3.03E-04	0.013	0.03	1.08E-04	0.005	0.010	0.35	0.37	0.37
75th	1.83E-03	0.08	0.16	6.91E-04	0.03	0.06	0.38	0.39	0.39
80th	2.54E-03	0.11	0.23	9.93E-04	0.05	0.09	0.39	0.41	0.41
85th	3.40E-03	0.15	0.32	1.44E-03	0.07	0.14	0.42	0.44	0.44
90th	4.72E-03	0.22	0.46	2.17E-03	0.10	0.22	0.46	0.46	0.47
95th	7.01E-03	0.35	0.75	3.76E-03	0.18	0.38	0.54	0.51	0.50
97.5th	9.33E-03	0.51	**1.1**	5.36E-03	0.27	0.57	0.57	0.53	0.53
99th	1.22E-02	0.74	**1.5**	7.69E-03	0.44	0.93	0.63	0.60	0.60

Bold indicates a risk above a level of concern.

Table 6-22. Comparison of Risk Results for Manganese in Ilmenite Process Wastewater Treatment Sludge for 100 Percent and 10 Percent Waste Quantities

Percentile	100% Waste			10% Waste			Ratio		
	Groundwater Concentration (mg/L)	Adult HQ	Child HQ	Groundwater Concentration (mg/L)	Adult HQ	Child HQ	Groundwater Concentration (mg/L)	Adult HQ	Child HQ
50th	2.64E-04	3.4e-05	6.7e-05	2.1e-04	2.6e-05	5.3e-05	0.79	0.76	0.79
75th	0.20	0.02	0.05	0.13	0.02	0.03	0.65	0.66	0.67
80th	0.43	0.05	0.11	0.26	0.03	0.07	0.61	0.58	0.59
85th	0.97	0.12	0.25	0.55	0.07	0.15	0.56	0.60	0.59
90th	2.10	0.26	0.53	1.26	0.16	0.33	0.60	0.63	0.62
95th	4.17	0.52	1.1	2.75	0.35	0.72	0.66	0.66	0.66
97.5th	6.06	0.83	1.8	4.36	0.59	1.3	0.72	0.72	0.71
99th	8.84	1.4	2.9	6.68	1.0	2.1	0.76	0.74	0.74

Table 6-23. Comparison of Risk Results for Thallium in Ilmenite Process Wastewater Treatment Sludge for 100 Percent and 10 Percent Waste Quantities

	100% Waste			10% Waste			Ratio		
Percentile	Groundwater Concentration (mg/L)	Adult HQ	Child HQ	Groundwater Concentration (mg/L)	Adult HQ	Child HQ	Groundwater Concentration (mg/L)	Adult HQ	Child HQ
50th	7.14E-05	0.02	0.03	4.57E-05	0.01	0.02	0.64	0.63	0.63
75th	8.49E-04	0.18	0.38	5.13E-04	0.11	0.23	0.60	0.59	0.59
80th	1.35E-03	0.30	0.61	7.87E-04	0.17	0.35	0.58	0.57	0.57
85th	2.03E-03	0.44	0.91	1.21E-03	0.27	0.57	0.60	0.62	0.62
90th	3.07E-03	0.69	**1.4**	1.90E-03	0.43	0.91	0.62	0.62	0.63
95th	4.66E-03	**1.1**	**2.4**	3.22E-03	0.78	**1.6**	0.69	0.69	0.66
97.5th	6.09E-03	**1.6**	**3.4**	4.40E-03	**1.1**	**2.5**	0.72	0.70	0.72
99th	7.90E-03	**2.4**	**5.2**	6.08E-03	**1.7**	**3.7**	0.77	0.73	0.72

Bold indicates a risk above a level of concern.

Table 6-24. Comparison of DAFs for Antimony in Ilmenite Process Wastewater Treatment Sludge for 100 Percent and 10 Percent Waste Quantities

Percentile	100% Waste	10% Waste	Ratio
50th	66	186	2.8
10th	4	9	2.2
5th	3	5	1.9
1st	2	3	1.6

Table 6-25. Comparison of DAFs for Manganese in Ilmenite Process Wastewater Treatment Sludge for 100 Percent and 10 Percent Waste Quantities

Percentile	100% Waste	10% Waste	Ratio
50th	61,801	77,749	1.3
10th	8	13	1.7
5th	4	6	1.5
1st	2	2	1.3

Table 6-26. Comparison of DAFs for Thallium in Ilmenite Process Wastewater Treatment Sludge for 100 Percent and 10 Percent Waste Quantities

Percentile	100% Waste	10% Waste	Ratio
50th	168	262	1.6
10th	4	6	1.6
5th	3	4	1.4
1st	2	2	1.3

Table 6-27. Comparison of Regional Aquifer Data to Site-Specific Aquifer Data for Millennium HPP Facility

Parameter	Regional Data (Hydrogeologic Environment 10)	Site-Specific Data Collected for Surface Impoundment Modeling
Hydraulic conductivity (m/yr)	3-19,600	113-22,700
Hydraulic gradient (m/m)	0.000001 - 0.1	0.0014-0.0026
Aquifer thickness (m)	1-55	20-60
Vadose zone thickness (m)	0-3	0-3 m

the HGDB database. Antimony in secondary gypsum was chosen as the constituent/waste stream combination to be modeled because it is the only combination managed in this onsite landfill that approached a nominal risk level of concern when modeled using the regional aquifer data. The effect of using the site-specific aquifer data was to increase the modeled DAFs and reduce the hazard quotients. Therefore, the site-specific aquifer data can be presumed to lower the modeled groundwater concentrations and HQs for the other wastes and chemical constituents. The results of the comparison of the groundwater modeling and risk results for antimony in secondary gypsum modeled in an onsite landfill at the Millennium HPP facility are presented in Tables 6-28 and 6-29.

6.6.7 Z Well Constraint for Thick Aquifer

For this risk analysis, the depth of a residential well was constrained to lie within the top 10 m of the aquifer. This was based on the presumption that residential wells will not be drilled deeper than necessary to sustain adequate flows for residential usage. Also, any groundwater contamination from nearby SWMUs is generally expected to be highest in the upper portion of the aquifer. However, in the area near the Du Pont Delisle facility, residential wells are known to penetrate much deeper than 10 m into the very thick surficial aquifer present in that area. To investigate the sensitivity of the model results at this site to the depth of the well, a receptor well was located in the aquifer constrained only by the depth of the aquifer. These results were compared with the risks results obtained for the Z-well constrained within the upper 10 m of the aquifer. These risk results and DAFs are compared in Tables 6-30 and 6-31 for manganese in ilmenite wastewaters managed in an onsite surface impoundment in Delisle, MS; in Tables 6-32 and Table 6-33 for Thallium, and in Table 6-34 and 6-35 for Vanadium. The results show lower DAFs and higher hazard quotients for the unconstrained Z-well than for the Z-well constrained to the upper 10 meters of the aquifer; however, the differences were less than a factor of 2.

6.6.8 Shape of Aquifer Hydraulic Conductivity Distribution

Another area of uncertainty that was investigated is the distribution of hydraulic conductivity values used for the antimony oxide waste stream managed in an onsite landfill at the U.S. Antimony facility. No site-specific data were available to adequately characterize the aquifer in this location; therefore, a large body of data known to be applicable to gravel aquifers was used to describe the hydraulic conductivity. Since only the range of data were known, it was necessary to specify the shape of the distribution. Therefore, a triangular distribution was specified using the range of logs for the conductivity with the midpoint at the center of the log distribution. Since there is some evidence that the distribution of hydraulic conductivities is lognormal, a second analysis was performed using a lognormal distribution with the extremes and the center of the distribution the same as the "log triangular" distribution. This comparison is presented in Tables 6-36 and 6-37. The results show that the groundwater concentrations and hazard quotients are higher by about a factor of 2 for the lognormal distribution. This is due to the fact that the lognormal distribution is weighted toward somewhat lower hydraulic conductivity values than the logtriangular distribution. These distributions are presented graphically in Figure 6-1.

Table 6-28. Comparison of Risk Results for Antimony in Secondary Gypsum Managed in an Onsite Landfill Modeled Using Regional and Site-Specific Aquifer Data

	Regional Aquifer Data			Site-Specific Aquifer Data			Ratio		
Percentile	Groundwater Concentration (mg/L)	Adult HQ	Child HQ	Groundwater Concentration (mg/L)	Adult HQ	Child HQ	Groundwater Concentration (mg/L)	Adult HQ	Child HQ
50th	8.92E-04	0.04	0.08	3.49E-04	0.02	0.03	0.39	0.40	0.40
75th	3.57E-03	0.15	0.32	2.04E-03	0.09	0.18	0.57	0.57	0.57
80th	4.42E-03	0.20	0.41	2.74E-03	0.12	0.25	0.62	0.61	0.61
85th	5.52E-03	0.25	0.52	3.66E-03	0.16	0.34	0.66	0.65	0.65
90th	6.93E-03	0.33	0.70	5.00E-03	0.23	0.49	0.72	0.71	0.70
95th	8.85E-03	0.47	0.99	6.97E-03	0.35	0.75	0.79	0.76	0.75
97.5th	1.05E-02	0.61	**1.32**	8.67E-03	0.48	**1.04**	0.83	0.79	0.79
99th	1.20E-02	0.83	**1.79**	1.05E-02	0.66	**1.43**	0.88	0.79	0.80

Bold indicates a risk above a level of concern.

Table 6-29. Comparison of DAFs for Antimony in Secondary Gypsum Managed in an Onsite Landfill Modeled Using Regional and Site-Specific Aquifer Data

Percentile	Regional Aquifer Data	Site-specific Aquifer Data
50th	62	158
10th	8	11
5th	6	8
1st	5	5

Table 6-30. Comparison of Risk Results for Manganese in Ilmenite Wastewaters Managed in an Onsite Surface Impoundment for Z-Well Constrained with Z-Well Unconstrained

Percentile	Constrained Z-Well			Unconstrained Z-Well			Ratio of Unconstrained Z-Well to Constrained Z-Well		
	Groundwater Concentration (mg/L)	Adult HQ	Child HQ	Groundwater Concentration (mg/L)	Adult HQ	Child HQ	Groundwater Concentration (mg/L)	Adult HQ	Child HQ
50th	6.7e-06	8.1e-07	1.6e-06	1.1e-05	1.3e-06	2.6e-06	1.6	1.6	1.6
75th	8.8e-05	1.1e-05	2.2e-05	1.3e-04	1.6e-05	3.3e-05	1.5	1.5	1.5
80th	1.4e-04	1.8e-05	3.7e-05	2.1e-04	2.6e-05	5.4e-05	1.5	1.5	1.5
85th	2.2e-04	2.9e-05	6.0e-05	3.3e-04	4.4e-05	9.1e-05	1.5	1.5	1.5
90th	3.7e-04	5.0e-05	1.0e-04	5.9e-04	8.0e-05	1.7e-04	1.6	1.6	1.6
95th	7.5e-04	1.1e-04	2.3e-04	1.4e-03	1.9e-04	4.0e-04	1.8	1.8	1.7
97.5th	1.4e-03	2.0e-04	4.3e-04	2.6e-03	4.1e-04	8.5e-04	1.9	2.0	2.0
99th	2.6e-03	4.1e-04	8.5e-04	5.6e-03	8.7e-04	1.8e-03	2.2	2.2	2.2

Table 6-31. Comparison of DAFs for Manganese in Ilmenite Wastewaters Managed in an Onsite Surface Impoundment for Z-Well Constrained with Z-Well Unconstrained

Percentile	Constrained	Unconstrained	Ratio
50th	492,772	308,268	0.626
10th	8,859	5,602	0.632
5th	4,403	2,466	0.560
1st	1,300	597	0.459

Table 6-32. Comparison of Risk Results for Thallium in Ilmenite Wastewaters Managed in an Onsite Surface Impoundment for Z-Well Constrained with Z-Well Unconstrained

	Constrained Z-Well			Unconstrained Z-Well			Ratio of Unconstrained Z-Well to Constrained Z-Well		
Percentile	Groundwater Concentration (mg/L)	Adult HQ	Child HQ	Groundwater Concentration (mg/L)	Adult HQ	Child HQ	Groundwater Concentration (mg/L)	Adult HQ	Child HQ
50th	1.82E-07	3.9E-05	8.0E-05	2.88E-07	6.2E-05	1.3E-04	1.58	1.57	1.57
75th	1.60E-06	3.5E-04	7.3E-04	2.45E-06	5.4E-04	0.0011	1.53	1.55	1.58
80th	2.49E-06	5.8E-04	0.0012	3.89E-06	8.6E-04	0.0018	1.56	1.48	1.50
85th	4.19E-06	9.4E-04	0.0020	6.42E-06	0.0015	0.0031	1.53	1.60	1.54
90th	7.93E-06	0.0018	0.0038	1.20E-05	0.0029	0.0059	1.51	1.56	1.57
95th	1.84E-05	0.0045	0.0092	2.95E-05	0.0073	0.015	1.60	1.63	1.67
97.5th	3.61E-05	0.0091	0.019	6.44E-05	0.016	0.036	1.78	1.80	1.88
99th	7.95E-05	0.020	0.044	1.54E-04	0.041	0.082	1.94	1.99	1.89

Table 6-33. Comparison of DAFs for Thallium in Ilmenite Wastewaters Managed in an Onsite Surface Impoundment for Z-Well Constrained with Z-Well Unconstrained

Percentile	Constrained Z-Well	Unconstrained Z-Well	Ratio of Unconstrained Z-Well to Constrained Z-Well
50th	30,919	18,859	0.610
10th	865	555	0.642
5th	410	241	0.587
1st	111	56	0.500

Table 6-34. Comparison of Risk Results for Vanadium in Ilmenite Wastewaters Managed in an Onsite Surface Impoundment for Z-Well Constrained with Z-Well Unconstrained

Percentile	Constrained Z-Well			Unconstrained Z-Well			Ratio of Unconstrained Z-Well to Constrained Z-Well		
	Groundwater Concentration (mg/L)	Adult HQ	Child HQ	Groundwater Concentration (mg/L)	Adult HQ	Child HQ	Groundwater Concentration (mg/L)	Adult HQ	Child HQ
50th	6.2e-08	1.2e-07	2.4e-07	1.0e-07	1.9e-07	3.9e-07	1.60	1.57	1.60
75th	3.6e-06	7.4e-06	1.5e-05	5.9e-06	1.2e-05	2.4e-05	1.63	1.57	1.56
80th	7.6e-06	1.5e-05	3.2e-05	1.3e-05	2.5e-05	5.3e-05	1.66	1.68	1.67
85th	1.7e-05	3.5e-05	7.2e-05	2.8e-05	5.4e-05	1.1e-04	1.64	1.54	1.52
90th	4.4e-05	8.5e-05	1.8e-04	6.7e-05	1.4e-04	2.8e-04	1.55	1.63	1.60
95th	1.4e-04	2.9e-04	6.0e-04	2.1e-04	4.4e-04	9.4e-04	1.53	1.54	1.57
97.5th	3.2e-04	6.7e-04	0.0014	5.4e-04	0.0011	0.0024	1.69	1.72	1.67
99th	7.4e-04	0.0017	0.0037	1.3e-03	0.0030	0.0064	1.74	1.77	1.73

Table 6-35. Comparison of DAFs for Vanadium in Ilmenite Wastewaters Managed in an Onsite Surface Impoundment for Z-Well Constrained with Z-Well Unconstrained

Percentile	Constrained Z-Well	Unconstrained Z-Well	Ratio of Unconstrained Z-Well to Constrained Z-Well
50th	1,309,069	746,123	0.570
10th	4,466	2,940	0.658
5th	1,965	1,190	0.606
1st	509	257	0.504

Table 6-36. Comparison of Risk Results for Antimony in Low Antimony Slag Managed in Onsite Landfill - Antimony Oxide Sector—Triangular Distribution with Lognormal Distribution of Aquifer Hydraulic Conductivity

	Triangular Distribution			Lognormal Distribution			Ratio		
Percentile	Groundwater Concentration	Adult HQ	Child HQ	Groundwater Concentration	Adult HQ	Child HQ	Groundwater Concentration	Adult HQ	Child HQ
50th	1.35E-03	0.06	0.12	3.04E-03	0.13	0.27	0.44	0.46	0.44
75th	6.08E-03	0.28	0.57	1.39E-02	0.62	**1.3**	0.44	0.45	0.45
80th	8.63E-03	0.39	0.82	1.93E-02	0.90	**1.8**	0.45	0.43	0.45
85th	1.32E-02	0.60	**1.3**	2.85E-02	**1.4**	**2.8**	0.46	0.44	0.45
90th	2.18E-02	**1.0**	**2.1**	4.60E-02	**2.2**	**4.6**	0.47	0.46	0.46
95th	4.26E-02	**2.1**	**4.4**	9.43E-02	**4.5**	**9.4**	0.45	0.47	0.47
97.5th	7.69E-02	**3.9**	**8.2**	1.67E-01	**8.5**	**18**	0.46	0.46	0.45
99th	1.52E-01	**7.7**	**17**	3.38E-01	**17**	**37**	0.45	0.44	0.45

Bold indicates a risk above a level of concern.

Table 6-37. Comparison of Risk Results for Arsenic as a Carcinogen in Low Antimony Slag Managed in Onsite Landfill - Antimony Oxide Sector—Triangular Distribution with Lognormal Distribution of Aquifer Hydraulic Conductivity

Percentile	Triangular Distribution			Lognormal Distribution			Ratio		
	Groundwater Concentration	Adult Risk	Child Risk	Groundwater Concentration	Adult Risk	Child Risk	Groundwater Concentration	Adult Risk	Child Risk
50th	1.87E-16	6.3e-19	6.3e-19	0	0	0	NA	NA	NA
75th	3.20E-06	1.5e-08	1.2e-08	4.03E-06	1.9e-08	1.5e-08	0.79	0.80	0.77
80th	6.95E-06	3.5e-08	2.7e-08	1.15E-05	5.4e-08	4.2e-08	0.60	0.65	0.64
85th	1.55E-05	7.7e-08	5.9e-08	2.74E-05	1.4e-07	1.1e-07	0.57	0.54	0.54
90th	3.64E-05	2.0e-07	1.5e-07	6.94E-05	3.8e-07	2.7e-07	0.52	0.52	0.56
95th	9.58E-05	5.9e-07	4.4e-07	2.02E-04	1.2e-06	8.9e-07	0.47	0.51	0.50
97.5th	2.23E-04	1.4e-06	1.0e-06	4.47E-04	2.9e-06	2.1e-06	0.50	0.48	0.49
99th	4.81E-04	3.5e-06	2.0e-06	1.02E-03	7.1e-06	4.4e-06	0.47	0.49	0.46

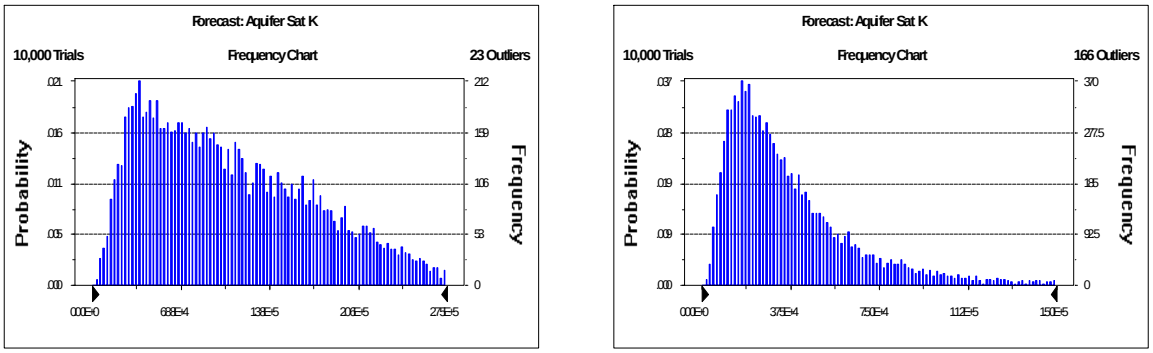

Figure 6-1. Triangular and lognormal distributions of aquifer hydraulic conductivity.

7.0 References

Achmad, G. 1991. *Simulated Hydrologic Effects of the Development of the Patapsco Aquifer System in Glen Burnie, Anne Arundel County, Maryland*. Report of Investigations No. 54. Prepared by U.S. Department of the Interior, U.S. Geological Survey, and the Anne Arundel County Department of Utilities.

Aller, L., T. Bennett, J.H. Lehr, R.J. Petty, and G. Hackett (Linda Aller, Truman Bennett, Jay H. Lehr, Rebecca J. Petty, Glen Hackett). 1987. *DRASTIC: A Standardized System for Evaluating Ground Water Pollution Potential Using Hydrogeologic Settings*. EPA-600/2-87-035. Prepared by U.S. Environmental Protection Agency, Robert S. Kerr Environmental Research Laboratory, Office of Research and Development, Ada, OK, 188p. April.

Anderson, P.R., and T.H.and Christensen. 1988. Distribution coefficients of Cd, Co, Ni, and Zn in Soils. *Journal of Soil Science*, 39:15-22.

ATSDR (Agency for Toxic Substances and Disease Registry). 1992. *Toxicological Profile for Thallium* (Update). Public Health Service, U.S. Department of Health and Human Services, Atlanta, GA.

ATSDR (Agency for Toxic Substances and Disease Registry). 1997. *Toxicological Profile for Manganese* (Draft Update). Public Health Service, U.S. Department of Health and Human Services, Atlanta, GA. September.

Bangash, M.A., and J. Hanif. 1992. Sorption behavior of cobalt on illitic soil. *Waste Management*, 12:29-38.

Carsel, R.F., and R.S. Parrish. 1988. Developing joint probability distributions of soil water retention characteristics. *Water Resources Research*, 24(5):755-769. May.

Carsel, R.F., R.S. Parrish, R.L. Jones, J.L. Hansen, and R.L. Lamb. 1988. Characterizing the uncertainty of pesticide leaching in agricultural soils. *Journal of Contaminant Hydrology*, 2:111-124.

CDC (Centers for Disease Control). 1991. *Preventing Lead Poisoning in Young Children*. Public Health Service, U.S. Department of Health and Human Services, pp. 1-5. October.

Chapelle, F.H. 1985. *Hydrogeology, Digital Solute-Transport Simulation, and Geochemistry of the Lower Cretaceous Aquifer System Near Baltimore, Maryland*. Report of Investigation No. 43. Prepared by U.S. Geological Survey, Department of the Interior.

Christensen, J.B., D.L. Jensen, and T.H. Christensen. 1996. Effect of dissolved organic carbon on the mobility of cadmium, nickel, and zinc in leachate polluted groundwater. *Water Research,* 30:3037-3049.

Clapp, R.B., and G.M. Hornberger. 1978. Empirical equations for some soil hydraulic properties. *Water Resources Research,* 14:601-604.

Clawges, R.M. and C.V. Price (Rick M. Clawges, Curtis V. Price). 1999a. Digital Data Sets Describing Principal Aquifers, Surficial Geology, and Ground-Water Regions of the Conterminous United States. Version 99-77. Rapid City, South Dakota.

Clawges, R.M. and C.V. Price (Rick M. Clawges, Curtis V. Price). 1999b. Ground-water Regions of the Conterminous United States with Unconsolidated Watercourses. Version 99-77. Rapid City, South Dakota.

Clawges, R.M. and C.V. Price (Rick M. Clawges, Curtis V. Price). 1999c. Surficial geology of the conterminous United States. Version 99-77. Rapid City, South Dakota.

Clawges, R.M. and C.V. Price (Rick M. Clawges, Curtis V. Price). 1999d. Productive aquifers of the conterminous United States. Version 99-77. Rapid City, South Dakota.

Degussa Corporation. 1998. *Risk-Based Corrective Action Plan for the Sodium Cyanide Production Unit at Degussa Corporation Alabama Facility*. Theodore, AL. March 19.

Domenico, P.A., and F.W. Schwartz (Patrick A. Domenico, Franklin W. Schwartz). 1990. *Physical and Chemical Hydrogeology*. John Wiley & Sons,

Du Pont Delisle Plant. 1999. *RCRA Facility Investigation Report*. EPA ID MSD 096 046 792. Pass Christian, MS. December 7.

Fetter, C.W. 1994. *Applied Hydrogeology*. 3rd edition. Volume 4 Prentice-Hall, Inc., Englewood Cliffs, New Jersey. 98p.

Gelhar, L.W., C. Welty, and K.R. Rehfeldt (Lynn W. Gelhar, Claire Welty, Kenneth R. Rehfeldt). 1992. A critical review of data on field-scale dispersion in aquifers. *Water Resources Research,* 28(7):1955-1974. July.

Hassan, S.M., and A.W. Garrison. 1996. Distribution of chromium species between soil and porewater. *Chemical Speciation and Bioavailability,* 8:85-103.

Kerr-McGee Chemical LLC 1998. *RCRA Confirmatory Sampling Report Kerr-McGee Chemical LLC, Hamilton, Mississippi*. Hamilton, MS. July.

Kollig, H.P. 1993. *Environmental Fate Constants for Organic Chemicals Under Consideration for EPA's Hazardous Waste Identification Projects*. EPA/600/R-93/132. Prepared for U.S. Environmental Protection Agency, Office of Research and Development, Environmental Research Laboratory., Athens, GA. August.

Little, J.C. (John C. Little). 1992. Applying the two-resistance theory to contaminant volatilization in showers. *Environmental Science & Technology*, 26(No. 7):1341-1349.

Loux, N.T., C.R. Chafin, and M.H. Sayed 1990. *Statistics of Aquifer Material Properties and Empirical pH-dependent Partitioning Relationships for As(III), As(V), Ba(II), Be(II), Cd(II), Cr(VI), Cu(II), Hg(II), Ni(II), Pb(II),Sb(V), Se(IV), Se(VI), Tl(I) and ZN(II)*. Prepared for Office of Solid Waste. Prepared by U.S. Environmental Protection Agency, Environmental Research Laboratory, Athens, GA, pp. 1-16. March.

McKone, T.E. 1987. Human exposure to volatile organic compounds in household tap water: the indoor inhalation pathway. *Environmental Science and Technology*, 21(12):1194-1201.

Miller, D.A., and R.A. White. 1998. *A Conterminous United States Multilayer Soil Characteristics Dataset for Regional Climate and Hydrology Modeling*. Website at http://www.essc.psu.edu/soil_info/index.cgi?soil_data&index.html.

Newell, C.J., L.P. Hopkins, and P.B. Bedient (Charles J. Newell, Loren P. Hopkins, Philip B. Bedient). 1990. A hydrogeologic database for ground-water modeling. *Ground Water*, 28(No. 5):703-714.

O'Neil, P.E. and M.F. Mette 1982. *Alabama Coastal Region Ecological Characterization: 2. A Synthesis of Environmental Data. Ecological Survey of Alabama*. Information Series 61 - 62. Prepared by Geological Survey of Alabama, Environmental Division, University, AL.

Shea, J.H. 1974. Deficiencies of clastic particles of certain sizes. *Journal of Sedimentary Petrology*, 44(4):985-1003. December.

Stephan, C.E., D.I. Mount, D.J. Hansen, J.H. Gentile, G.A. Chapman, and W.A. Brungs. 1985. *Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses*. PB85-227049. Prepared by Environmental Research Laboratory, Office of Research and Development, U.S. Environmental Protection Agency, Duluth, MN. January.

U.S.EPA (Environmental Protection Agency). 1986. *Industrial Subtitle D Facility Study. List of Questions in Telephone Survey* . Prepared by U.S. Environmental Protection Agency. October 20.

U.S.EPA (Environmental Protection Agency). 1988. *National Survey of Solid Waste (Municipal) Landfill Facilities* (Draft). EPA/530-SW88-034. Prepared by U.S. Environmental Protection Agency, Office of Solid Waste and Emergency Response, Washington, DC. September.

U.S.EPA (Environmental Protection Agency). 1989. *Risk Assessment Guidance for Superfund. Volume I: Human Health Evaluation Manual (Part A)* (Interim Final). EPA/540/1-89/002. Prepared by U.S. Environmental Protection Agency, Office of Emergency and Remedial Response, Washington, DC. December.

U.S.EPA (Environmental Protection Agency). 1990. *Background Document for EPA's Composite Model For Landfills (EPACML)*. Prepared by U.S. Environmental Protection Agency, Office of Solid Waste, Washington, D.C., 1-B-5. February.

U.S.EPA (Environmental Protection Agency). 1991. *Technical Support Document For Water Quality-based Toxics Control*. EPA/505/2-90-001. Prepared by U.S. Environmental Protection Agency, Office of Water, Washington, D.C.. March.

U.S.EPA (Environmental Protection Agency). 1992. *Guidelines for Exposure Assessment*. EPA/600Z-92/001. Office of Research and Development, Risk Assessment Forum, Washington, DC. May.

U.S.EPA (Environmental Protection Agency). 1993. *Parameter Values for Developing Nationwide Regulations with the EPA's Composite Model for Landfills (EPACML)*. Prepared for HydroGeologic, Inc and Office of Solid Waste. Prepared by U.S. Environmental Protection Agency, Office of Solid Waste, Washington, D.C., 1-G-12. July.

U.S.EPA (Environmental Protection Agency). 1994a. *The Hydrologic Evaluation of Landfill Performance (HELP) Model. User's Guide for Version 3*. EPA/600/R-94/168a. Office of Research and Development, Washington, DC. September.

U.S.EPA (Environmental Protection Agency). 1994b. *The Hydrologic Evaluation of Landfill Performance (HELP) Model. Engineering Documentation for Version 3*. EPA/600/R-94/168b. Office of Research and Development, Washington, DC. September.

U.S.EPA (Environmental Protection Agency). 1994d. *RCRA Operation & Maintenance Inspection of SCM Chemicals. Hawkins Point Plant Batch Attack Lagoon, Baltimore, Maryland*. Maryland Department of the Environment Waste Management Administration Hazardous Waste Program, Region III, Philadelphia, PA. October.

U.S.EPA (Environmental Protection Agency). 1994c. *Guidance Manual for the Integrated Exposure Uptake Biokinetic Model for Lead in Children*. EPA/540/R-93/081. Prepared for Office of Emergency and Remedial Response. Prepared by U.S. Environmental Protection Agency, Technical Review Workgroup for Lead, Research Triangle Park, NC. February.

U.S.EPA (Environmental Protection Agency). 1996a. *EPA's Composite Model for Leachate Migration with Transformation Products, EPACMPT. Background Document for Finite Source Methodology for Chemicals with Transformation Products*. R09-96.588. Prepared by U.S. Environmental Protection Agency, Office of Solid Waste, Washington, D.C.. October 8.

U.S.EPA (Environmental Protection Agency). 1996b. *EPA's Composite Model for Leachate Migration with Transformation Products; EPACMTP. Background Document*. Office of Solid Waste, Washington, DC. September.

U.S.EPA (Environmental Protection Agency). 1996d. *EPACMTP Sensitivity Analysis*. Prepared by U.S. Environmental Protection Agency, Office of Solid Waste, Washington, D.C., pp. 1-17. March.

U.S.EPA (Environmental Protection Agency). 1996c. *Proposed Guidelines for Carcinogenic Risk Assessment*. EPA/600/P-92/003C. Prepared by U.S. Environmental Protection Agency, Office of Research and Development, Washington, D.C. April.

U.S.EPA (Environmental Protection Agency). 1997a. *EPA's Composite Model for Leachate Migration with Transformation Products. EPACMTP: User's Guide*. Office of Solid Waste, Washington, DC.

U.S. EPA (Environmental Protection Agency). 1997b. *Exposure Factors Handbook. Volume I: General Factors: Volume II: Food Ingestion Factors; Volume III: Activity Factors*. EPA/600/P-95/002Fa. Prepared by U.S. Environmental Protection Agency, Office of Research and Development, Washington, DC. August.

U.S.EPA (Environmental Protection Agency). 1999b. *Toxicological Review of Acetonitrile in Support of Summary Information on the Integrated Risk Information System (IRIS)*. National Center for Environmental Assessment, Office of Research and Development, Washington, DC. January.

U.S.EPA (Environmental Protection Agency). 1999a. *Source Modules for Tanks and Surface Impoundments. Background and Implementation for the Multimedia, Multipathway, and Multireceptor Risk Assessment (3MRA) for HWIR99* (Draft Report). Office of Solid Waste, Washington, DC. October.

U.S.EPA (Environmental Protection Agency). 2000a. *Antimony Oxide Listing Background Document for the Inorganic Chemical Listing Determination*. Prepared by U.S. Environmental Protection Agency, Office of Solid Waste. August.

U.S.EPA (Environmental Protection Agency). 2000e. *Sodium Phospate Listing Background Document for the Inorganic Chemical Listing*. Prepared by U.S. Environmental Protection Agency, Office of Solid Waste. August.

U.S.EPA (Environmental Protection Agency). 2000f. *Titanium Dioxide Listing Background Document for the Inorganic Chemical Listing Determination*. Prepared by U.S. Environmental Protection Agency, Office of Solid Waste. August.

U.S.EPA (Environmental Protection Agency). 2000d. *Sodium Chlorate Listing Background Document for the Inorganic Chemical Listing Determination*. Prepared by U.S. Environmental Protection Agency, Office of Solid Waste. August.

U.S.EPA (Environmental Protection Agency). 2000c. *Integrated Risk Information System (IRIS)*. National Center for Environmental Assessment, Office of Research and Development, Washington, DC. August.

U.S.EPA (Environmental Protection Agency). 2000b. *Inorganic Hydrogen Cyanide Listing Background Document for the Inorganic Chemical Listing Determination*. Prepared by U.S. Environmental Protection Agency, Office of Solid Waste. August.

U.S.HUD (Housing and Urban Development). 1995. *Guidelines for the Evaluation and Control of Lead-based Paint Hazards in Housing*. Prepared by U.S. Department of Housing and Urban Development, Department of Housing and Urban Development. June.

United States Antimony Corp. 1999. United States Antimony Corp. Stibnite Hill Mine Project (Unpublished).

USDA (Department of Agriculture). 1994. National STATSGO Database: USDA-NRCS Soil Survey Division Data Access. U.S. Department of Agriculture, Natural Resources Conservation Service, Fort Worth, Texas. Website at http://www.ftw.nrcs.usda.gov/stat_data.html.

USGS. 1985. National water summary 1984. Hydrologic events, selected water-quality trends, and ground-water resources. In: *U.S. Geological Survey Water-Supply Paper 2275*. pp. 229-235, 397-402. United States Geological Survey, Washington, DC.

van der Leeden, F., F.L.Troise, and D.K.Todd 1990. *The Water Encyclopedia*. 2nd edition. Lewis Publishers, Chelsea, Michigan. p. 176.