Human and Ecological Risk Assessment of Coal Combustion Wastes

April 2010

DRAFT

U.S. Environmental Protection Agency
Office of Solid Waste and Emergency Response
Office of Resource Conservation and Recovery

Regulation Identifier Number (RIN) 2050-AE81

This draft document has been prepared by the Office of Resource Conservation and Recovery in the U.S. Environmental Protection Agency (EPA). Any opinions, findings, conclusions, or recommendations do not change or substitute for any statutory or regulatory provisions. This document does not impose legally binding requirements, nor does it confer legal rights, impose legal obligations, or implement any statutory or regulatory provisions. Mention of trade names or commercial products is not intended to constitute endorsement or recommendation for use. This document is being provided to other government agencies and to the public. As a draft, EPA may change any part of this document in the future, as appropriate.

Prepared under
Contract EP-W2-09-004
Work Assignment 1-02

Human and Ecological Risk Assessment of Coal Combustion Wastes

DRAFT

U.S. Environmental Protection Agency
Office of Solid Waste and Emergency Response
Office of Resource Conservation and Recovery

April 2010

Table of Contents

List of Figures

List of Tables

[This page intentionally left blank.]

List of Acronyms

Acronym	Definition
3MRA	multimedia, multiple exposure pathway, multiple receptor risk assessment
ADD	average daily dose
Ag	silver
Al	aluminum
As	arsenic
ATSDR	Agency for Toxic Substances and Disease Registry
B	boron
Ba	barium
BCF	bioconcentration factors
Be	beryllium
CAIR	Clean Air Interstate Rule
CalEPA	California EPA
CAMR	Clean Air Mercury Rule
CCW	coal combustion waste
Cd	cadmium
Co	cobalt
Cr	chromium
CSCL	chemical stressor concentration limit
CSF	cancer slope factor
Cu	copper
DAF	dilution attenuation factor
DBGS	depth below ground surface
DOE	U.S. Department of Energy
DWAK	Drinking Water Action Level
EFH	Exposure Factors Handbook
EIA	Energy Information Agency
EPA	U.S. Environmental Protection Agency
EPACMTP	EPA's Composite Model for Leachate Migration with Transformation Products
EPRI	Electric Power Research Institute
F	fluoride
FBC	fluidized-bed combustion
Fe	iron
FGD	flue gas desulfurization
GIS	geographic information system
HBN	health-based number
HDPE	high-density polyethylene
HEAST	Health Effects Assessment Summary Tables
HELP	Hydrologic Evaluation of Landfill Performance
Hg	mercury
HGDB	Hydrogeologic Database
HHRAP	Human Health Risk Assessment Protocol
HQ	hazard quotient

Acronym	Definition
ID	identification
IEM	Indirect Exposure Methodology
IRIS	Integrated Risk Information System
IWEM	Industrial Waste Evaluation Model
LADD	lifetime average daily dose
LDS	leak detection system
LF	landfill
LOAEL	lowest observed adverse exposure level
MCL	maximum concentration limit
Mn	manganese
MINTEQA2	a geochemical assessment model for environmental systems
Mo	molybdenum
MRL	minimal risk level
MSW	municipal solid waste
Ni	nickel
NOAEL	no observed adverse exposure level
NPDES	National Pollutant Discharge Elimination System
ORD	Office of Research and Development (EPA)
Pb	lead
PCS	Permit Compliance System
pH	a measure of hydrogen ion activity
RCRA	Resource Conservation and Recovery Act
RCRA	Resource Conservation and Recovery Act
RfD	reference dose
RME	reasonable maximum exposure
SAB	Science Advisory Board (EPA)
Sb	antimony
SD	standard deviation
Se	selenium
SI	surface impoundment
Sr	strontium
SW	solid waste
SWTSS	surface water total suspended solids
TCLP	toxicity characteristic leaching procedure
Tl	thallium
TL3	trophic level 3
TL4	trophic level 4
TSS	total suspended solids
UF	uncertainty factor
USGS	U.S. Geological Survey
V	vanadium
WMU	waste management unit
Zn	zinc

Human and Ecological Risk Assessment of Coal Combustion Wastes – Executive Summary

The Executive Summary of EPA's *Human and Ecological Risk Assessment of Coal Combustion Wastes* is organized into four parts. First, it presents **Background** for the regulation and study of coal combustion wastes. Next, it discusses the **Risk Assessment Methodology** used to evaluate these wastes' potential impact on human health and the environment. The Executive Summary continues with the presentation of the report's **Results and Characterization**. Finally, it discusses the overall **Conclusions** of the report.

Background

The U.S. Environmental Protection Agency (EPA) is evaluating management options for solid wastes from coal combustion: fly ash, bottom ash, boiler slag, flue gas desulphurization (FGD) residues, and fluidized bed combustion (FBC) wastes. In this report, these five types of coal combustion wastes are referred to as coal combustion waste (CCW). All coal-fired electric utility plants in the United States generate at least one of these wastes, and most generate more than one. For example, most electric utility plants generate fly ash and either bottom ash or boiler slag.[1] Some plants also generate FGD residues.[2] Coal-fired electric utility plants that use FBC technology generate both bottom ash (bed ash) and fly ash.

Depending on the coal-fired power plant boiler and air pollution control technologies employed at the power plants, these five types of CCW might be initially generated either as primarily dry or primarily wet material. Typically, the dry materials are disposed of in landfills, while the wet materials are disposed of, at least initially, in surface impoundments (the settled solids can be removed periodically and disposed of in landfills). Landfills and surface impoundments are referred to as waste management units (WMUs).

Coal-fired power plants typically conduct coal preparation activities before burning the coal in their boilers. Wastes from these coal preparation activities (such as coal handling by conveyor systems, coal washing for removing mineral matter, and coal "sizing"—for example, reducing particle sizes of coal for firing in a pulverized coal boiler) are not part of the Bevill exclusion under the federal Resource Conservation and Recovery Act. However, in the past, some U.S. coal-fired power plants have managed CCW together with these coal preparation wastes, or "coal refuse," in the same landfills and surface impoundments. Because the chemical characteristics of the coal refuse can affect the amount and behavior of chemical constituents in the CCW,[3] EPA designed this analysis specifically to estimate risks from CCW management that is conducted separately from coal refuse management, as well as to estimate risks from CCW that is comanaged with coal refuse.

This report describes the results of a national-scale analysis of groundwater impacts of managing CCW in five separate scenarios:

- CCW managed alone in landfills
- CCW managed alone in surface impoundments
- CCW comanaged with coal refuse in landfills

[1] U.S. EPA (1999a), Figure 3-2.
[2] U.S. EPA (1999a), Figure 3-3.
[3] U.S. EPA (1999a), page 3-18.

- CCW comanaged with coal refuse in surface impoundments
- FBC waste managed in landfills.

This risk assessment was designed and implemented to identify and quantify human health and ecological risks that may result from groundwater contamination from current management practices for high-volume CCWs. The risk assessment uses mathematical models to represent either a landfill or a surface impoundment, and to represent the movement of chemical constituents from the CCW placed into a landfill or surface impoundment through the environment, up to an exposure point where the chemical constituent comes into contact with a person (such as in a glass of drinking water from a well) or an aquatic organism (such as a fish swimming in surface water that has become contaminated by groundwater that discharges into the stream near a CCW landfill). In this analysis, EPA evaluated human health exposures that occur by the groundwater-to-drinking-water pathway, human health exposures that occur by fish consumption, ecological exposures of aquatic organisms in direct contact with contaminated surface water or sediment, and ecological exposures of organisms that eat contaminated food items from those contaminated nearby surface water bodies.

Because the infiltration from a landfill or surface impoundment can significantly influence how much, and how quickly, leachate flows out of a waste management unit, the models also account for three types of liner scenarios: unlined, clay-lined, and composite-lined. An unlined waste management unit has native soils as the bottom and sides; a clay-lined unit has a certain amount of clay present to slow the flow of leachate; and a composite-lined unit is constructed from various layers, including human-made materials, which are assumed to retard the leachate flow to a significantly greater extent than a clay liner.

The risk assessment provides a distribution of estimated risks for each of the five scenarios and three liner types. EPA modeled CCW waste management units that were located across the United States, in locations that represent a subset of the coal-fired power plants that were in use in the mid-1990s. The models used to represent the movement of chemical constituents from a landfill or surface impoundment through the environment rely on data such as weather patterns, soil types, and subsurface geology, which influence the speed and direction in which the chemical constituents move. Thus, the environmental setting, or geographic location, of a landfill or surface impoundment can influence the resulting estimated risk. By conducting the analysis at a national scale, EPA estimated risks at locations across the United States.

Risk Assessment Methodology

To estimate the risks posed by the onsite management of CCW, the risk assessment determined the release of CCW constituents from landfills and surface impoundments, estimated the concentrations of these constituents in environmental media, and estimated the risks that these concentrations pose to human and ecological receptors. To evaluate the significance of these risks, they were compared with a risk range or single criterion as follows:

- For constituents that cause cancer (carcinogens), the typical cancer risk evaluated was a range from 1 excess lifetime cancer case per 1,000,000 exposed individuals (i.e., 10^{-6} excess cancer risk) to 1 case per 10,000 exposed individuals (i.e., 10^{-4} excess lifetime cancer risk).[4]

[4] This is the typical cancer risk range used by the Office of Solid Waste and Emergency Response - (10^{-6} to 10^{-4}).

- For constituents that cause adverse, noncancer health effects (noncarcinogens), the criterion is a hazard quotient (HQ)[5] of greater than 1
- For constituents that cause adverse ecological effects, the criterion is an HQ of greater than 1.

In support of this risk assessment, EPA assembled a constituent database that includes leachate and porewater waste concentrations for 41 CCW constituents taken from more than 140 CCW disposal sites around the country. The CCW risk assessment subjected these waste and leachate constituent concentrations to a risk assessment methodology that implemented the following steps to assess the human and ecological risks posed by CCW:

- **Hazard Identification,** which collected existing human health and ecological benchmarks for the 41 CCW constituents to identify the 25 chemicals with benchmarks for constituent screening
- **Constituent Screening,** which compared very conservative estimates of exposure concentrations (e.g., whole waste concentrations, leachate concentrations) to health-based concentration benchmarks to quickly and simply eliminate constituents and exposure pathways that did not require further analysis
- **Full-Scale Analysis,** which used a Monte Carlo probabilistic analysis to characterize the risks to human health and ecological receptors from onsite disposal of CCW constituents that posed the greatest potential risks in the screening analysis.

To select the constituents for full-scale modeling, the screening analysis compared very conservative estimates of exposure concentrations (e.g., leachate concentrations) to health-based concentration benchmarks to quickly and simply identify constituents that do not appear to pose human or ecological health concerns, so that these constituents could be eliminated from further analysis. For example, leachate concentrations were compared directly to drinking water standards, which is equivalent to assuming that human receptors are drinking leachate.

During both the screening and probabilistic modeling stages, two exposure scenarios were evaluated for humans:

- Contaminated groundwater being transported to drinking water wells from a CCW landfill or surface impoundment
- Contaminated groundwater discharging into a surface waterbody where people catch and eat fish.

Constituents addressed in the full-scale analysis were those that posed the greatest potential risks to human or ecological health. The full-scale analysis was designed to characterize waste management scenarios based on two WMUs (landfills and surface impoundments), three waste types (CCW, CCW codisposed with coal refuse, and FBC waste), and three liner types (unlined, clay-lined, and composite-lined). Because FBC waste is not known to be disposed of in surface impoundments, this left 15 possible disposal options to model. These options provide a good representation of CCW disposal practices and waste chemistry conditions that affect the release of CCW constituents from WMUs.

The full-scale analysis was implemented using a probabilistic approach that produces a distribution of risks or hazards for each receptor by allowing the values of some of the parameters in the analysis to vary. This approach is ideal for this risk assessment because there are so many CCW facilities across the United States, and the approach

[5] The HQ is the ratio of the average daily exposure level to a protective exposure level corresponding to the maximum level at which no appreciable effects are likely to occur.

captures the variability in both waste management practices and environmental settings (e.g., hydrogeology, climate, hydrology). This probabilistic approach was implemented through the following steps:

1. Characterize the CCW constituents and waste chemistry, along with the WMUs in which each waste stream may be managed (i.e., the size and linear status of CCW landfills and surface impoundments).
2. Characterize the environmental settings for the sites where CCW landfills and surface impoundments are located (i.e., locations of coal-fired power plants).
3. Identify how contaminants are released from a WMU through leaching and transported to human and ecological receptors by groundwater and surface water.
4. Predict the fate, transport, and concentration of constituents in groundwater and surface water once they are released to groundwater from the WMUs and travel to receptors at each site.
5. Quantify the potential exposure of human and ecological receptors to the contaminant in the environment.
6. Estimate the potential risk to each receptor from the exposure and characterize this risk in terms of exposure pathways and health effects.

Based on this approach, EPA characterized the potential risks associated with the waste disposal scenarios and exposure pathways, including the uncertainties associated with the results.

Results and Characterization

The CCW risk assessment presented results at a typical exposure (50th percentile) as well as a high-end exposure (90th percentile). CCW risk assessment results at the 90th percentile suggest that managing CCW in unlined or clay-lined WMUs result in risks greater than the risk criteria of an HQ greater than 1 for noncancer effects to both human and ecological receptors (for humans drinking groundwater, 90th percentile HQs up to 3 for antimony, 7 for boron, 9 for lead, 8 for molybdenum, 20 for nitrate, and 4 for thallium; for ecological receptors, 90th percentile HQs up to 2,000 for boron, 300 for lead, 100 for arsenic, 30 for cadmium, and 12 for selenium). With respect to arsenic in CCW, the 90th percentile results suggest that managing CCW in unlined or clay-lined WMUs results in human excess cancer risks within or above a range of 1 in 1 million to 1 in 10,000 (i.e., ranging from 6 in 100,000 to 1 in 50 excess cancer risk). Clay-lined units tended to have lower risks than unlined units, but still had 90th percentile arsenic III excess cancer risks ranging from 6 in 100,000 to 7 in 1,000.However, it was the composite-lined units that effectively reduced risks from all pathways and constituents, below1 in 100,000 excess cancer risk or an HQ of one.

The tables that follow present selected risk results only for chemicals that exceed an excess cancer risk of 1 in 100,000 (arsenic only) or an HQ of 1.

As shown in **Table ES-1**, arsenic was the constituent with the highest risk for landfills. Clay-lined landfills presented 90th percentile arsenic III cancer risks as high as 1 in 5,000 and thallium HQs as high as 2. When landfills were unlined, they additionally presented arsenic III cancer risks as high as 1 in 2,000 and a maximum thallium HQ of 3. In addition to arsenic and thallium, clay-lined FBC landfills also presented 90th percentile risks above an HQ of 1 for antimony. However, unlined FBC landfills differed in that they only exceeded a 1 in 100,000 excess cancer risk for arsenic and did not exceed an HQ of 1 for any of the noncarcinogens modeled.[6] At the 50th percentile (see **Table ES-2**) arsenic

[6] Unlined FBC units showed less risk as modeled.

III from CCW codisposed with coal refuse exceeded an excess cancer risk of 10^{-5}, with cancer risks of 1 in 50,000.

As shown in **Table ES-3**, arsenic and cobalt were the constituents with the highest risks for surface impoundments. Clay-lined surface impoundments presented 90th percentile excess cancer risks above 1 in 10,000 for arsenic and exceed the HQ criterion of 1 for boron, cadmium, cobalt, molybdenum, and nitrate. Here, arsenic excess cancer risks were as high as 1 in 500, and cobalt had HQs as high as 200. When surface impoundments were unlined, they also showed risk above the HQ criterion for lead and selenium. Here, arsenic excess cancer risks were as high as 1 in 50, and cobalt had HQs as high as 500. As seen in **Table ES-4**, the 50th percentile surface impoundment results exceeded a 1 in 100,000 cancer risk for arsenic and only cobalt exceeded an HQ of 1. Here, unlined units had arsenic excess cancer risks as high as 6 in 10,000 while clay-lined units had arsenic excess cancer risks as high as 1 in 5,000. Cobalt HQs were as high as 20 and 6 for unlined and clay-lined surface impoundments, respectively.

For the groundwater-to-drinking-water pathway, composite liners, as modeled in this assessment, effectively reduced risks from all constituents to below a 10^{-5} cancer risk or HQ of 1 for both landfills and surface impoundments at the 90th and 50th percentiles.

For the groundwater-to-drinking-water pathway, arrival times of the peak concentrations at a receptor well are much longer for landfills (hundreds or thousands of years) than for surface impoundments (most less than 100 years).

For humans exposed via the groundwater-to-surface-water (fish consumption) pathway, unlined and clay-lined surface impoundments posed risks above the HQ criterion and an excess cancer risk of 1 in 100,000 at the 90th percentile (see **Table ES-5**). For CCW managed alone in surface impoundments, these exceedences came from selenium (HQs of 3 and 2 for unlined and clay-lined units, respectively), while for CCW comanaged with coal refuse, these exceedences came from arsenic (3 in 100,000 and 2 in 100,000 excess cancer risks for unlined and clay-lined units, respectively). All 50th percentile surface impoundment risks are below an HQ of 1 and an excess cancer risk of 1 in 100,000. No constituents pose risks above these levels for landfills (including FBC landfills) at the 90th or 50th percentile.

Waste type has a much larger effect when managed in surface impoundments than when managed in landfills. In the case of surface impoundments, some constituents presented higher risks from CCW managed alone (boron, molybdenum, nitrate, and selenium). However, others presented higher risks when CCW is comanaged with coal refuse (arsenic, cadmium, cobalt, and lead).

The higher risks for surface impoundments than landfills are likely due to higher waste leachate concentrations and the higher hydraulic head from the impounded liquid waste. This is consistent with damage cases reporting wet handling as a factor that can increase risks from CCW management.

Table ES-1. Selected[a] 90th Percentile Risk Results by CCW Type: Landfills, Groundwater-to-Drinking-Water Pathway

Chemical	90th Percentile HQ or Cancer Risk Value[b]		
	Unlined Units	Clay-Lined Units	Composite-Lined Units
Conventional CCW – 79 landfills			
Cancer			
Arsenic III	4E-04	2E-04	0
Arsenic V	2E-04	3E-05	0
Noncancer			
Antimony	2	0.8	0
Molybdenum	2	0.8	0
Thallium	3	2	0
Codisposed CCW and Coal Refuse – 41 landfills			
Cancer			
Arsenic III	5E-04	2E-04	0
Arsenic V	4E-04	6E-05	0
Noncancer			
Molybdenum	2	0.6	0
Thallium	2	1	0
FBC Waste – 7 landfills			
Cancer			
Arsenic III	3E-05	6E-05	0
Arsenic V	2E-05	2E-05	0
Noncancer			
Antimony	0.8	3	0
Thallium	1	4	0

[a] Values are presented only for chemicals that exceed an excess cancer risk of 1 in 100,000 (arsenic only) or an HQ of 1.

[b] Values are HQs for all chemicals except arsenic; arsenic values are cancer risk. Zero results indicate that contaminant infiltration rates were too small for the contaminant plume to reach the receptor during the 10,000-year period of the analysis.

Table ES-2. Selected[a] 50th Percentile Risk Results by CCW Type: Landfills, Groundwater-to-Drinking-Water Pathway

Chemical	50th Percentile HQ or Cancer Risk Value[b]		
	Unlined Units	Clay-Lined Units	Composite-Lined Units
Codisposed CCW and Coal Refuse – 41 landfills			
Cancer			
Arsenic III	2E-05	6E-06	0

[a] Values are presented only for chemicals that exceed an excess cancer risk of 1 in 100,000 (arsenic only) or an HQ of 1.

[b] Values are HQs for all chemicals except arsenic; arsenic values are cancer risk. Zero results indicate that contaminant infiltration rates were too small for the contaminant plume to reach the receptor during the 10,000-year period of the analysis.

Table ES-3. Selected[a] 90th Percentile Risk Results by CCW Type: Surface Impoundments, Groundwater-to-Drinking-Water Pathway

Chemical	90th Percentile HQ or Cancer Risk Value[b]		
	Unlined Units	Clay-Lined Units	Composite-Lined Units
Conventional CCW – 44 surface impoundments			
Cancer			
Arsenic III	2E-03	9E-04	2E-07
Arsenic V	7E-04	2E-04	0
Noncancer			
Boron	7	4	5E-03
Lead (MCL)[c]	3	0.7	1E-21
Molybdenum	8	5	7E-03
Nitrate/nitrite (MCL)[c]	20	10	9E-04
Selenium VI	2	1	1E-03
Codisposed CCW and Coal Refuse – 72 surface impoundments			
Cancer			
Arsenic III	2E-02	7E-03	4E-06
Arsenic V	2E-02	2E-03	3E-09
Noncancer			
Cadmium	9	3	5E-05
Cobalt	500	200	3E-06
Lead (MCL)[c]	9	1	1E-19
Molybdenum	3	2	4E-03

[a] Values are presented only for chemicals that exceed an excess cancer risk of 1 in 100,000 (arsenic only) or an HQ of 1.

[b] Values are HQs for all chemicals except arsenic; arsenic values are cancer risk. Zero results indicate that contaminant infiltration rates were too small for the contaminant plume to reach the receptor during the 10,000-year period of the analysis.

[c] Values are ratios of exposure concentration to MCL.

Table ES-4. Selected[a] 50th Percentile Risk Results by CCW Type: Surface Impoundments, Groundwater-to-Drinking-Water Pathway

Chemical	50th Percentile HQ or Cancer Risk Value[b]		
	Unlined Units	Clay-Lined Units	Composite–Lined Units
Conventional CCW – 44 surface impoundments			
Cancer			
Arsenic III	1E-04	6E-05	0
Arsenic V	2E-05	4E-06	0
Codisposed CCW and Coal Refuse – 72 surface impoundments			
Cancer			
Arsenic III	6E-04	2E-04	0
Arsenic V	3E-04	4E-05	0
Noncancer			
Cobalt	20	6	0

[a] Values are presented only for chemicals that exceed an excess cancer risk of 1 in 100,000 (arsenic only) or an HQ of 1.

[b] Values are HQs for all chemicals except arsenic; arsenic values are cancer risk. Zero results indicate that contaminant infiltration rates were too small for the contaminant plume to reach the receptor during the 10,000-year period of the analysis.

Table ES-5. Selected[a] 90th Percentile Risk Results by CCW Type: Surface Impoundments, Groundwater-to-Surface-Water Pathway

Chemical	90th Percentile HQ or Cancer Risk Value[b]		
	Unlined Units	Clay-Lined Units	Composite-Lined Units
Conventional CCW – 44 surface impoundments			
Noncancer			
Selenium VI	3	2	2E-06
Codisposed CCW and Coal Refuse – 72 surface impoundments			
Cancer			
Arsenic III	3E-05	2E-05	1E-14
Arsenic V	2E-05	8E-06	6E-19

[a] Values are presented only for chemicals that exceed an excess cancer risk of 1 in 100,000 (arsenic only) or an HQ of 1.

[b] Values are HQs for all chemicals except arsenic; arsenic values are cancer risk. Zero results indicate that contaminant infiltration rates were too small for the contaminant plume to reach the receptor during the 10,000-year period of the analysis.

For ecological receptors exposed via surface water, risks for landfills exceeded an HQ of 1 for boron and lead at the 90th percentile, but 50th percentile HQs were well below 1. For surface impoundments, 90th percentile risks for several constituents exceeded the risk criteria, with boron showing the highest risks (HQ = 2,000). Only boron exceeded an HQ of 1 at the 50th percentile (HQ = 7). Exceedances for boron and selenium are consistent with reported ecological damage cases, which include impacts to waterbodies through the groundwater-to-surface-water pathway.

For ecological receptors exposed via sediment, 90th percentile risks for lead, arsenic, and cadmium exceeded the risk criteria for both landfills and surface impoundments because these constituents strongly sorb to sediments in the waterbody. The 50th percentile risks were generally an order of magnitude or more below the risk criteria.

Sensitivity analysis results indicate that for more than 70 percent of the scenarios evaluated, the risk assessment model was most sensitive to parameters related to the contaminant source and groundwater flow and transport, including WMU infiltration rate, leachate concentration, and aquifer hydraulic conductivity and gradient. For the groundwater-to-surface water pathway, another sensitive parameter is the flow rate of the waterbody into which the contaminated groundwater is discharging. For strongly sorbing contaminants (such as lead and cadmium), variables related to sorption and travel time are also important (adsorption coefficient, depth to groundwater, and receptor well distance).

Although the best available data and techniques were used, there were several uncertainties associated with the CCW risk assessment. The major types of uncertainty were as follows:

- **Scenario Uncertainty** includes the assumptions and modeling decisions that are made to represent an exposure scenario.
- **Model Uncertainty** is associated with all models used in a risk assessment because mathematical expressions are simplifications of reality that approximate real-world conditions and processes.
- **Parameter Uncertainty** occurs when there is a lack of data about the values used in the equations, data available are not representative of the instance being modeled, or parameter values have not been measured precisely because of limitations in technology.

Scenario uncertainty has been minimized by basing the risk assessment on conditions around existing U.S. coal-fired power plants around the United States. Uncertainty in environmental setting parameters has been incorporated into the risk assessment by varying these inputs within reasonable ranges when the exact value is not known. Uncertainty in human exposure factors (such as exposure duration, body weight, and intake rates) has also been addressed through the use of national distributions.

Some uncertainties not addressed explicitly in the risk assessment have been addressed through comparisons with other studies and data sources. These include the appropriateness of the leachate data used for landfills, concentrations of mercury in current CCW, and the potential impacts of future mercury regulations.

Other uncertainties are not as easily addressed as the ones above. These include issues such as receptor well distance, liner conditions, ecological benchmarks, ecological receptors at risk, and synergistic risks. Detailed discussion of all the risk

assessment uncertainties is presented in **Section 4.4** of the report.

Conclusions

Given the results and characterization above, composite liners, as modeled in this risk assessment, effectively reduced risks from all pathways and constituents below the risk criteria for both landfills and surface impoundments. The CCW risk assessment suggests that the management of CCW in unlined landfills and unlined surface impoundments may present risks to human health and the environment. Selenium in certain types of WMUs managing certain types of CCW may present a risk of clinical selenosis to highly exposed groundwater users or fish consumers, or a risk of adverse effects to highly exposed aquatic receptors. Arsenic in certain types of WMUs managing certain types of CCW may present lifetime cancer risks above EPA's range of concern to highly exposed groundwater users. Estimated risks from clay-lined units are lower than the risks of unlined units, but are still above the risk criteria used for this analysis. In addition, surface impoundments typically showed higher risks than landfills, regardless of liner type. Finally, for surface impoundments, codisposal of CCW with coal refuse results in significantly higher risks from arsenic and certain other constituents than CCW disposed alone, while for other constituents, managing CCWs alone results in higher estimated risks than codisposed CCW.

These risk results are in many cases consistent with damage cases compiled by EPA (U.S. EPA, 2000, 2003e, 2007) and others (Lang and Schlictmann, 2004; Zillmer and Fauble, 2004; Carlson and Adriano, 1993; Rowe et al., 2002; Hopkins et al., 2006).[7] For example, the full-scale modeling of selenium released from unlined surface impoundments into groundwater suggests that certain fish consumers may be exposed to relatively high levels of selenium, consistent with fish consumption advisories at some of the proven damage case sites. These results suggest that with a higher prevalence of composite liners in new CCW disposal facilities, along with practices to prevent codisposal of coal refuse with CCW, future national risks from onsite CCW disposal are likely to be lower than those presented in this risk assessment (which is based on 1995 CCW WMUs).

References

Carlson, C.L., and D.C. Adriano. 1993. Environmental impacts of coal combustion residues. *J. Environ. Qual*. 22(227-247).

Hopkins, W.A., S.E. DuRant, B.P. Staub, C.L. Rowe, and B.P. Jackson. 2006. Reproduction, embryonic development, and maternal transfer of contaminants in the amphibian Gastrophryne carolinensis. *Environmental Health Perspectives* 114(5):661–666.

Lang, D.J., and J.R. Schlictmann. 2004. Leaching of arsenic from a fly ash dump in Beverly, Massachu-setts. In *Proceedings of State Regulation of Coal Combustion By-Product Placement at Mine Sites*. May 4–6, 2004, Harrisburg, Pennsylvania. U.S. Department of the Interior, Office of Surface Mining, Alton, IL. Available at http://www.mcrcc.osmre.gov/MCR/Resources/ccb/PDF/State_Regulation_of_CCB_Placement.pdf.

Rowe, C.L., W.A. Hopkins, and J.D. Congdon. 2002. Ecotoxicological

[7] See **Sections 4.1.5** and **4.2.5** for a more complete discussion of CCW damage cases and risk assessment results.

implications of aquatic disposal of coal combustion residues in the United States: a review. *Environmental Monitoring and Assessment* 80:207–276.

U.S. EPA (Environmental Protection Agency). 1999a. *Report to Congress: Wastes from the Combustion of Fossil Fuels*. EPA 530-R-99-010. Office of Solid Waste, Washington, DC. March.

U.S. EPA (Environmental Protection Agency). 2000. *Review of Causative Factors for Coal Combustion Waste Damage Cases*. Prepared by SAIC for U.S. EPA Office of Solid Waste under EPA contract 68-W-99-001, WA No. 121. Washington, DC. November 29. Available as document ID number EPA-HQ-RCRA-2006-0796-0020 at http://www.regulations.gov.

U.S. EPA (Environmental Protection Agency). 2003e. *Final Revised Report on Resolution of 18 Previously Indeterminate Candidate Damage Cases*. Prepared by SAIC for U.S. EPA Office of Solid Waste under EPA contract 68-W-02-036, WA No. 1-12. Washington, DC. March 5. Available as document ID number EPA-HQ-RCRA-2003-0003-0420 at http://www.regulations.gov.

U.S. EPA (Environmental Protection Agency). 2007. *Coal Combustion Waste Damage Case Assessments. Available from the docket to the Notice of Data Availability on the Disposal of Coal Combustion Wastes in Landfills and Surface Impoundments*, Docket # EPA-HQ-RCRA-2006-0796. June 19.

Zilmer, M., and P. Fauble. 2004. Groundwater impacts from coal combustion ash disposal sites in Wisconsin. In: *Proceedings of State Regulation of Coal Combustion By-Product Placement at Mine Sites*. May 4–6, 2004, Harrisburg, Pennsylvania. U.S. Department of Interior, Office of Surface Mining, Alton, IL. Available at http://www.mcrcc.osmre.gov/MCR/Resources/ccb/PDF/State_Regulation_of_CCB_Placement.pdf.

1.0 Introduction

1.1 Background

The U.S. Environmental Protection Agency (EPA) has evaluated the human health and environmental risks associated with coal combustion waste (CCW) management practices, including disposal in landfills and surface impoundments. In May 2000, EPA determined that regulation as hazardous wastes under Subtitle C of the Resource Conservation and Recovery Act (RCRA) was not warranted for certain CCWs, but that regulation as nonhazardous wastes under RCRA Subtitle D was appropriate. However, EPA did not specify regulatory options at that time. This risk assessment was designed and implemented to help EPA identify and quantify human health and ecological risks that may be associated with current management practices for high-volume CCWs. These wastes are fly ash, bottom ash, boiler slag, and flue gas desulfurization (FGD) sludge, along with wastes from fluidized-bed combustion (FBC) units and CCWs codisposed with coal refuse. This risk assessment will help EPA develop CCW management options for these high-volume waste streams. Details on EPA's CCW work can be found at http://www.epa.gov/epawaste/nonhaz/industrial/special/fossil/index.htm.

Note that the full-scale risk assessment described in this report was primarily conducted in 2003, meaning that the data collection efforts to support the risk assessment were based on the best information available to EPA at that time. As a result, more recent Agency efforts to characterize CCW wastes and management practices, such as the joint EPA and U.S. Department of Energy (DOE) survey of CCW waste management units (WMUs) (U.S. DOE, 2006) and EPA's recent study of CCW chemistries and leaching behavior (U.S. EPA, 2006c, 2008c), were not considered in the main analysis phase of this risk assessment. However, these more recent efforts are discussed as part of the risk characterization, and EPA is currently evaluating how to best incorporate and consider the results and findings of these studies in future efforts to address CCW management practices.

The Agency has revised this risk analysis document to address comments on the analytical methodology, data, and assumptions used in the risk assessment from an independent scientific peer review by experts outside EPA. Public comments (available in docket number EPA-HQ-RCRA-2006-0796[1]) were made available to the peer reviewers for their consideration during the review process. The peer review focused on technical aspects of the analysis, including the construction and implementation of the Monte Carlo analysis, the selection of models to estimate the release of constituents found in CCW from landfills and surface impoundments and their subsequent fate and transport in the environment, and the characterization of risks resulting from potential exposures to human and ecological receptors. EPA's responses to the peer-review comments, including descriptions of the revisions incorporated in this document to address those comments, are available in a separate response-to-comments document (U.S. EPA, 2009d).

[1] Available at http://www.regulations.gov.

1.2 Purpose and Scope of the Risk Assessment

The purpose of this risk assessment was to identify CCW constituents, waste types, exposure pathways, and receptors that may produce risks to human and ecological health, and to provide information about those scenarios that EPA could use to develop management options for CCW management.

The scope of this risk assessment was utility CCWs managed onsite at utility power plants. EPA's *Report to Congress: Wastes from the Combustion of Fossil Fuels* (U.S. EPA, 1999a) reports that there are 440 coal-fired utility power plants in the United States. Although these plants are concentrated in the East, they are found in nearly every state, with facility settings ranging from urban to rural. The large volumes of waste generated by these plants are typically managed onsite in landfills and surface impoundments. This risk assessment was designed to develop national human and ecological risk estimates that are representative of onsite CCW management settings throughout the United States.

1.3 Overview of Risk Assessment Methodology

To estimate the risks posed by the onsite management of CCW, this risk assessment estimated the release of CCW constituents from landfills and surface impoundments, the concentrations of these constituents in groundwater and surface water near coal-fired utility power plants, and the risks that these concentrations pose to human and ecological receptors.

1.3.1 Contaminant Sources

The size, design, and locations of the onsite CCW landfills and surface impoundments modeled in this risk assessment were based on data from a national survey of utility CCW disposal conducted by the Electric Power Research Institute (EPRI) in 1995 (EPRI, 1997). Data from this survey on facility area, volume, and liner characteristics were used in the CCW risk assessment because they were the most recent and complete data set available at the time the risk assessment was conducted (2003). However, as shown in **Table 1-1**, the EPA/DOE study conducted since then (U.S. DOE, 2006) shows a much higher proportion of lined facilities than do the 1995 EPRI data (see further discussion in **Section 4.4.1**).

Table 1-1. Liner Prevalence in EPRI and DOE Surveys

Liner Type	Landfills	Surface Impoundments
1995 EPRI Survey[a] – 181 facilities		
Unlined	40%	68%
Lined (compacted clay or composite [clay and synthetic])	60%	32%
2004 DOE Survey[b] – 56 facilities		
Unlined	3%	0%
Lined (compacted clay or composite [clay and synthetic])	97%	100%

[a] EPRI (1997)

[b] U.S. DOE (2006)

1.3.2 Exposure Pathways

The releases, and hence media concentrations and risk estimates in this report, were based on leaching to groundwater and groundwater transport to nearby wells and surface water bodies. This analysis did not address direct releases to surface water, which are permitted under the National Pollutant Discharge Elimination System (NPDES) of the Clean Water Act. Thus, the estimated media concentrations and risks do not take into account contributions from NPDES-permitted releases, including discharges due to flooding or heavy rainfall. Uncertainties associated with this decision are described in **Section 4.4.1** of this report.

EPA recognizes that there are exposure pathways in addition to the groundwater pathways addressed in this report that could be of concern to human health and ecological receptors, including fugitive dust eroded and transported by wind from uncovered CCW landfills, and erosion and transport of CCW constituents from uncovered landfills onto adjacent land and eventually into downslope waterbodies. These "aboveground" pathways were addressed in the 1998 risk assessment, and in 2002, EPA conducted a draft screening analysis (U.S. EPA, 2002a) to evaluate risks from these pathways.

1.3.3 Risk Levels

To evaluate the significance of the estimated risks from the pathways assessed in this assessment, EPA compared the risk estimates to a risk range (for carcinogens) or to a specific risk criterion (for noncarcinogens) that are protective of human health and the environment:

- An estimate of the excess lifetime cancer risk for individuals exposed to carcinogenic (cancer-causing) contaminants ranging from 1 chance in 1,000,000 (10^{-6} excess cancer risk) to 1 chance in 10,000 (10^{-4} excess cancer risk). For decisions made to screen out certain constituents from further consideration, a 1 in 100,000 (10^{-5}) excess lifetime cancer risk was used.[2]

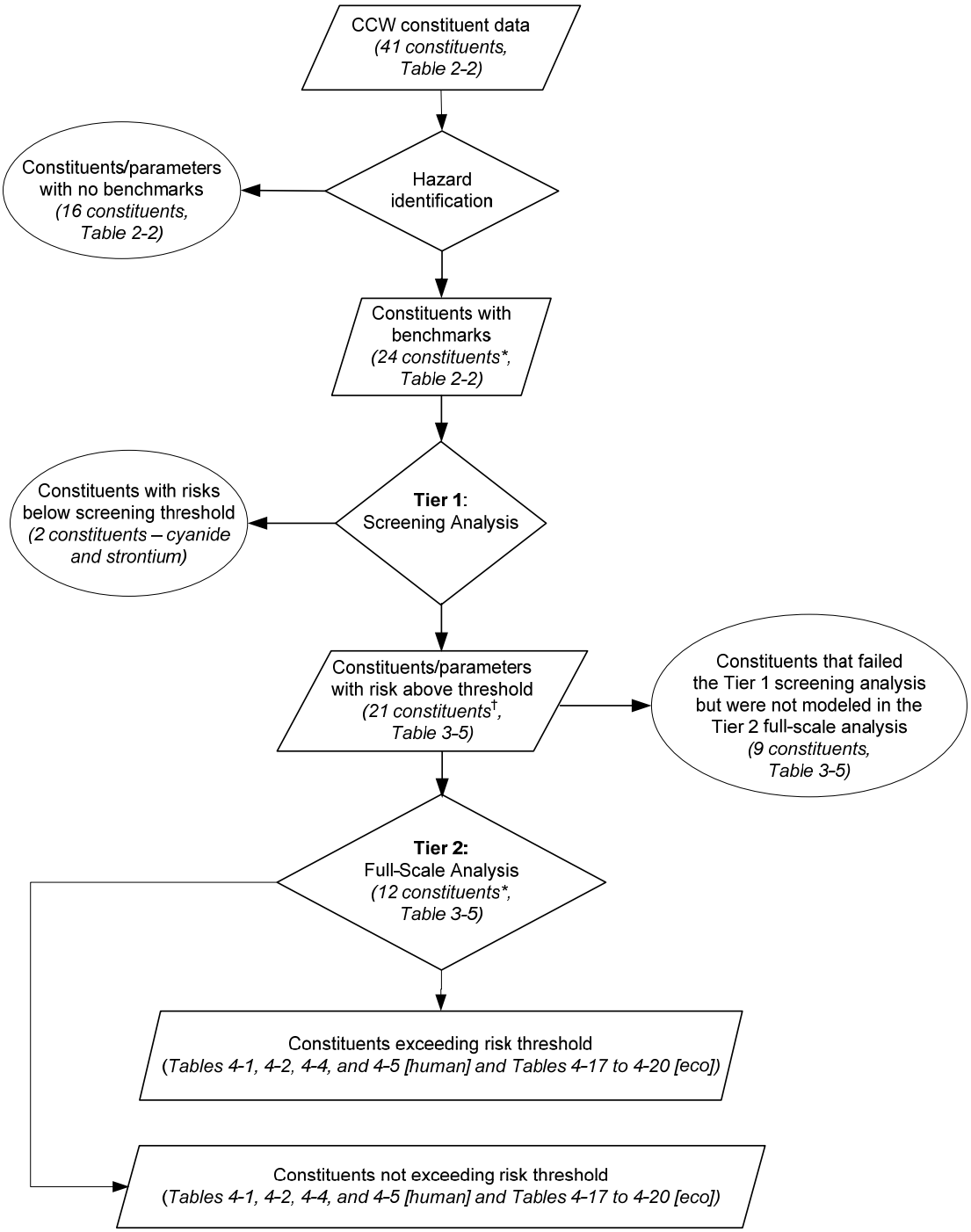
- A measure of safe intake levels to predicted intake levels, a hazard quotient (HQ) greater than 1 for constituents that can produce noncancer human health effects (an HQ of 1 is defined as the ratio of a potential exposure to a constituent to the highest exposure level at which no adverse health effects are likely to occur)

- An HQ greater than 1 for constituents with adverse effects to ecological receptors.

1.3.4 Methodology

In 1998, EPA conducted a risk assessment for fossil fuel combustion wastes (which include CCWs) to support the May 2000 RCRA regulatory determination (U.S. EPA, 1998a,b). Since then, EPA has added to the waste constituent database that was used in that effort, expanding the number of leachate and total waste analyses for 41 CCW constituents. The CCW risk assessment subjected these waste and leachate constituent concentrations to the tiered risk assessment methodology illustrated in **Figure 1-1**.

[2] The typical cancer risk range used by the Office of Solid Waste and Emergency Response is 10^{-4} to 10^{-6}.



* Does not include ammonia. Although ammonia was detected in CCW, data were insufficient to address it in the screening analysis or the full-scale analysis.

† Does not include mercury. Although mercury was considered in the screening analysis and modeled in the full-scale analysis, the results were not meaningful due to the high proportion of non-detect measurements.

Note: The risk threshold used for cancer risks was 1 in 100,000.

Figure 1-1. Overview of coal combustion waste risk assessment.

This methodology implemented the following steps to assess the human and ecological risk of CCWs:

- **Hazard Identification**, which collected existing human health and ecological benchmarks for the CCW constituents. Only constituents with benchmarks moved on to the next step, constituent screening.

- **Constituent Screening**, which compared very conservative estimates of exposure concentrations (e.g., using leachate concentrations) to health-based concentration benchmarks to quickly and simply identify constituents with risks below the screening criteria.

- **Full-Scale Analysis**, which characterized at a national level the human health and ecological risks for constituents in CCW disposed onsite in landfills and surface impoundments using a site-based probabilistic Monte Carlo risk analysis.

This document focuses on the full-scale Monte Carlo analysis, but includes a discussion of the hazard identification and screening analysis (in U.S. EPA, 2002a) that led to the full-scale assessment.

1.3.5 Waste Management Scenarios Addressed

The full-scale analysis was designed to characterize waste management scenarios based on two waste management options (disposal of CCW onsite in landfills and in surface impoundments) and three waste types, as follows:

- **Conventional CCW**, which includes fly ash, bottom ash, boiler slag, and FGD sludge

- **Codisposed CCWs and coal refuse,**[3] which are more acidic than conventional CCWs due to sulfide minerals in the mill rejects

- **FBC wastes**, which include fly ash and the fluidized bed ash, and which tend to be more alkaline than conventional CCW because of the limestone mixed in during fluidized bed combustion.

Conventional CCW and codisposed CCW and coal refuse are typically disposed of in landfills and surface impoundments that can be lined with clay or composite liners. FBC wastes are only disposed of in landfills in the United States; therefore, surface impoundment disposal was not modeled for FBC waste.

These three waste types, two waste management options, and three liner conditions (unlined, clay lined, composite lined) modeled in this analysis provide a good representation of CCW disposal practices and waste chemistry conditions that affect the release of CCW constituents from WMUs.

[3] Coal refuse is the waste coal produced from coal handling, crushing, and sizing operations, and tends to have a high sulfur content and low pH from high amounts of sulfide minerals (like pyrite). In the CCW constituent database, codisposed coal refuse includes "combined ash and coal gob," "combined ash and coal refuse," and "combined bottom ash and pyrites."

1.3.6 Modeling Approach

The full-scale analysis was implemented using a site-based probabilistic approach that produces a distribution of risks or hazard for each receptor by allowing the values of some of the parameters in the analysis to vary. Input parameters were varied in the analysis using data collected at or around CCW disposal facilities or, when site-based data were not available, from distributions representing the variability of parameters across the United States. This approach was ideal for this risk assessment because there are many CCW facilities across the United States, and site-based data collection can capture both the variability in waste management practices at these facilities and the differences in their environmental settings (e.g., hydrogeology, climate, hydrology).

This probabilistic approach was implemented through seven primary steps:

Problem Formulation

1. Characterize the CCW constituents and waste chemistry, along with the size and liner status of the WMUs in which each waste stream may be managed

2. Characterize the environmental settings for the sites where CCW landfills and surface impoundments are located

3. Identify scenarios under which contaminants are released from a WMU and transported to a human receptor

Analysis

4. Screen risks to select CCW constituents for full-scale analysis

5. Predict the fate and transport of constituents in the environment once they are released from the WMUs at each site

6. Quantify the exposure of human and ecological receptors to the contaminant in the environment and the risk associated with this exposure

Risk Characterization

7. Estimate the risk to receptors from the exposure and characterize this risk in terms of exposure pathways, health effects, and uncertainties

8. Identify the waste disposal scenarios and environmental conditions that pose risks of potential concern to human health or the environment. Evaluate risks at the 50th and 90th percentiles.

1.4 Document Organization

This document is organized into the following sections:

- **Section 2, Problem Formulation,** describes how the framework for the full-scale analysis was developed, including identification of the waste constituents, exposure pathways, and receptors of concern; selection and characterization of waste management practices and sites to model; and development of the conceptual site models for the modeling effort.
- **Section 3, Analysis,** describes the probabilistic modeling framework and the models and methods used to (1) screen CCW constituents for the full-scale analysis, (2) estimate constituent releases from CCW landfills and surface impoundments (source models), (3) model constituent concentrations in the environmental media of concern (groundwater and surface water), (4) calculate exposure, and (5) estimate risk to human and ecological receptors.
- **Section 4, Risk Characterization,** characterizes the human health and environmental risks posed by CCW, including (1) discussion of the methods used to account for variability and uncertainty and (2) identification of the scenarios and conditions modeled that resulted in higher and lower risks. Results are presented as national estimates for CCW landfills and CCW surface impoundments, as well as by waste type and liner status. This section characterizes the risks posed by CCW constituents and pathways under the conditions modeled, including factors (such as liners or facility environmental setting) that result in higher or lower risk levels. Finally, the risk characterization evaluates the risk results in light of more recent research on CCW waste management practices and the environmental behavior of CCW constituents.

The first three appendices provide detailed information on how wastes, WMUs, and settings were characterized for the risk assessment. **Appendix A** describes the chemical characteristics of CCW, including the CCW leachate concentration distributions used to represent disposal conditions in landfills and surface impoundments. **Appendix B** describes how EPA characterized the CCW landfills and surface impoundments, including locations, surface area, capacities, geometry, and liner status. **Appendix C** presents the methodologies and data used to characterize the environmental setting at each CCW site identified in Appendix B, including delineating the site layout and determining the environmental setting (e.g., meteorology, climate, soils, aquifers, and waterbodies).

The next five appendices provide detailed information on the specific models and data used to calculate risk, including the nonlinear sorption isotherms (**Appendix D**), the surface water fate and transport and intake equations (**Appendix E**), the exposure factors (**Appendix F**), benchmarks for human health (**Appendix G**), and benchmarks for ecological risk (**Appendix H**).

The next three appendices provide background and results for the screening analysis, including calculation of health-based numbers (HBNs, **Appendix I**), chemical-specific inputs used in the screening analysis (**Appendix J**), and the screening analysis results (**Appendix K**).

Finally, **Appendix L** provides figures showing, for selected CCW constituents, cumulative percentiles of the time it took for the peak concentration to reach a receptor well for each source type.

2.0 Problem Formulation

The CCW risk assessment was intended to evaluate, at a national level, risk to individuals who live near WMUs used for CCW disposal. This section describes how the conceptual framework for the full-scale risk assessment was developed, including

- Constituent selection to identify the CCW constituents, exposure pathways, and receptors to address in this analysis (**Section 2.1.1**)
- Location and characterization of the CCW landfills and surface impoundments to be modeled as the sources of CCW contaminants in the full-scale site-based analysis (**Section 2.1.2**)
- The conceptual site model used to represent CCW disposal facilities (**Section 2.2**)
- The general modeling approach and scope, including constituent screening (**Section 2.3**), and full-scale modeling (**Section 2.4**) to estimate exposure point concentrations, assess exposures, and calculate risks to human and environmental receptors.

2.1 Source Characterization

The main technical aspects of the CCW risk assessment were completed in 2003, and the waste management scenarios modeled in this assessment were based on the best data on waste compositions, industry operations, and waste management practices that were available at that time. These data sources included a 1995 industry survey on CCW management practices (the EPRI comanagement survey [EPRI, 1997]) and data collected from a variety of sources before the 2003 risk assessment (e.g., EPA's CCW constituent database). Since 2003, DOE and EPA have completed a survey to characterize CCW waste disposal practices from 1994 to 2004, with a focus on new facilities or facility expansions completed within that same time frame (U.S. DOE, 2006). In addition, EPA studies of CCW composition and leaching behavior are ongoing (U.S. EPA, 2006c, 2008c). Although these newer data were not available when this risk assessment was conducted, they are discussed in the risk characterization (**Section 4**) as an uncertainty with respect to how well the risk assessment represents CCW leachate composition and current WMU liner conditions.

This risk assessment provides a national characterization of waste management scenarios for wastes generated by coal-fired utility power plants. The sources modeled in these scenarios are onsite landfills and surface impoundments, which are the primary means by which CCW is managed in the United States. The characterization of these sources, in terms of their physical dimensions, operating parameters, location, environmental settings, and waste characteristics, is fundamental to the construction of scenarios for modeling. This section describes how the coal combustion waste streams and management practices were characterized (based on the above

data sources) and screened to develop the waste disposal scenarios modeled in the full-scale analysis.

2.1.1 Identification of Waste Types, Constituents, and Exposure Pathways

To identify the CCW constituents and exposure pathways to be addressed in this risk assessment, we relied on a database of CCW analyses that EPA had assembled over the past several years to characterize whole waste and waste leachate from CCW disposal sites across the country (see **Appendix A**). The 2003 CCW constituent database includes all of the CCW characterization data used by EPA in its previous risk assessments, supplemented with additional data collected from public comments, data from EPA Regions and state regulatory agencies, industry submittals, and literature searches up to 2003.

The CCW constituent database represents a significant improvement in the quantity and scope of waste characterization data available from the 1998 EPA risk assessment of CCWs (U.S. EPA, 1998a,b). For example, the constituent data set used for the previous risk assessments (U.S. EPA, 1999a) covered approximately 50 CCW generation and/or disposal sites, while the 2002 CCW constituent database covers approximately 140 waste disposal sites.[1] The 2002 database also has broader coverage of the major ion concentrations of CCW leachate (e.g., calcium, sulfate, pH), that can influence CCW impacts on human health and the environment.

2.1.1.1 Waste Types

Table 2-1 shows the waste types included in the 2002 CCW constituent database, along with counts of the number of sites with wastes of each type with constituent measurements in landfill leachate, surface impoundment porewater, and whole waste.

Comments received by EPA on the previous CCW risk assessment pointed out that the analysis did not adequately consider the impacts of CCW leachate on the geochemistry and mobility of metal constituents in the subsurface. Commenters stated that given the large size of the WMUs and the generally alkaline nature of CCW leachate, it is likely that the leachate affects the geochemistry of the soil and aquifers underlying CCW disposal facilities, which can impact the migration of metals in the subsurface. To address this concern, EPA statistically evaluated major ion porewater data from the CCW constituent database for the waste streams shown in Table 2-1. Based on this analysis and prevalent comanagement practices, EPA grouped the waste streams into three statistically distinct categories: conventional CCW (fly ash, bottom ash, slag, and FGD sludge), which has moderate to high pH; codisposed CCW and coal refuse, which tends to have low pH; and FBC waste, which tends to have high pH. As shown in Table 2-1, each of these waste types included several waste streams that are usually codisposed in landfills or surface impoundments. Note that some sites in the CCW database have more than one waste stream, so the site counts for the different waste streams in a waste type category sum to more than the total site count for that waste type.

[1] Although EPA believes that the 140 waste disposal sites do represent the national variability in CCW characteristics, they are not the same sites as in the EPRI survey. During full-scale modeling, data from the CCW constituent database were assigned to each EPRI site based on the waste types reported in the EPRI survey data.

Table 2-1. Waste Streams in CCW Constituent Database

Waste Type Waste Stream	Landfills		Surface Impoundments
	Landfill Leachate	Total Waste[b]	Pore Water
Conventional CCW	***97***	***62***	***13***
Ash (not otherwise specified)	43	30	0
Fly ash	61	33	2
Bottom ash and slag	24	23	3
Combined fly and bottom ash	7	4	4
FGD sludge	4	5	6
Codisposed Ash & Coal Refuse	***9***	***1***	***5***
FBC Waste	***58***	***54***	***0***
Ash (not otherwise specified)	18	10	0
Fly ash	33	32	0
Bottom and bed ash	26	25	0
Combined fly & bottom ash	20	22	0

[a] For waste types (shaded rows) the table gives the number of sites; for waste streams (unshaded rows), the table gives the number of samples.
[b] Whole waste concentration data.

Along with the type of WMU (landfill or surface impoundment), the three waste types in Table 2-1 defined the basic modeling scenarios to be addressed in the full-scale analysis. To characterize these waste types, the CCW constituent database was queried by waste type to develop the waste concentration data for the constituents and the major ion and pH conditions used to develop waste-type-specific metal sorption isotherms (see **Appendix D** for a more extensive discussion of the development of CCW waste chemistries and metal sorption isotherms).

2.1.1.2 CCW Constituents of Potential Concern

The CCW constituent database contains data on more than 40 constituents. During the hazard identification step of the CCW risk assessment, constituents of potential concern were identified from this list of constituents by searching EPA and other established sources for human health and ecological benchmarks (e.g., Agency for Toxic Substances and Disease Registry [ATSDR]; see **Section 3.1** and **Appendices G** and **H** for a full list of sources). **Table 2-2** shows the results of that search for each constituent. Benchmarks were found for 24 chemicals in the constituent database. The 16 constituents without human health or ecological benchmarks were not addressed further in the risk analysis.[2]

[2] The CCW constituents without human health benchmarks are limited to common elements, ions, and compounds (e.g., iron, magnesium, phosphate, silicon, sulfate, sulfide, calcium, pH, potassium, sodium, carbon, sulfur). These measurements were used to determine overall CCW chemistries modeled in the risk assessment (see Section 3). Although some of these chemicals or parameters (e.g., pH, sulfate, phosphate, chloride) can pose an ecological hazard if concentrations are high enough, they were not addressed in this risk assessment.

Table 2-2. Toxicity Assessment of CCW Constituents

Constituent	CAS ID	HHB[a]	EcoB[b]
Metals			
Aluminum	7429-90-5	✔	✔
Antimony	7440-36-0	✔	✔
Arsenic	7440-38-2	✔[c]	✔
Barium	7440-39-3	✔	✔
Beryllium	7440-41-7	✔[d]	✔
Boron	7440-42-8	✔	✔
Cadmium	7440-43-9	✔[d]	✔
Chromium	7440-47-3	✔[c]	✔
Cobalt	7440-48-4	✔	✔
Copper	7440-50-8	✔	✔
Iron	7439-89-6		
Lead	7439-92-1	✔[e]	✔
Magnesium	7439-95-4		
Manganese	7439-96-5	✔	
Mercury	7439-97-6	✔	✔
Molybdenum	7439-98-7	✔	✔
Nickel	7440-02-0	✔	✔
Selenium	7782-49-2	✔	✔
Silver	7440-22-4	✔	✔
Strontium	7440-24-6	✔	
Thallium	7440-28-0	✔	✔
Vanadium	7440-62-2	✔	✔
Zinc	7440-66-6	✔	✔
Inorganic Anions			
Chloride	16887-00-6		
Cyanide	57-12-5	✔	
Fluoride	16984-48-8	✔	
Nitrate/nitrite	14797-55-8/14797-65-0	✔	
Phosphate	14265-44-2		
Silicon	7631-86-9		
Sulfate	14808-79-8		
Sulfide	18496-25-8		
Inorganic Cations			
Ammonia	7664-41-7	✔	
Calcium	7440-70-2		
pH	12408-02-5		
Potassium	7440-09-7		
Sodium	7440-23-5		
Nonmetallic Elements			
Carbon	7440-44-0		
Sulfur	7704-34-9		

(continued)

Toxicity Assessment of CCW Constituents (continued)

Constituent	CAS ID	HHB[a]	EcoB[b]
Measurements			
Total Dissolved Solids	none		
Total Organic Carbon	none		
Dissolved Organic Carbon	none		

[a] HHB = human health effect benchmark
[b] EcoB = ecological benchmark
[c] Known carcinogen (for chromium VI, inhalation only); although arsenic can act as both a carcinogen and a noncarcinogen, the cancer risk exceeds the noncancer risk at any concentration, so the more protective cancer benchmark for human health was used throughout this assessment.
[d] Probable carcinogen
[e] Safe Drinking Water Act Action Level only

2.1.2 Waste Management Scenarios

The full-scale CCW risk assessment modeled landfills and surface impoundments managing wastes onsite at coal-fired utility power plants. Because EPA selected a site-based modeling approach for the full-scale analysis, it was necessary to locate these disposal sites across the country to provide the spatial foundation for this analysis. It was also necessary to characterize CCW WMUs to define the scope for source modeling.

Two primary sources of data on these were used to characterize this population:

- 1998 Energy Information Agency (EIA) data on coal-fired power plants, which identifies approximately 300 coal-fired power plants with onsite waste management
- The 1995 EPRI waste comanagement survey (EPRI, 1997), which contains detailed WMU data (i.e., area, capacity, liner status, and waste type) for 177 of those facilities.

Because of the completeness of the WMU data from the EPRI survey, the EPRI data were used to establish the plant locations and WMU data for the full-scale modeling effort for conventional CCW[3] and CCW codisposed with coal refuse, as well as to help define protective waste management settings for the screening analysis.

Note that although there is overlap, the 140-site CCW constituent database described in Appendix A and the EPRI survey used to characterize CCW landfills and surface impoundments were assembled under separate efforts and represent different populations of disposal sites. As described in Section 3.1.3, these data sets were sampled independently during the Monte Carlo analysis, and constituent data were not assigned to particular sites except by waste type.

Although there is a good amount of FBC data in the constituent database (58 sites; see Table 2-1), there were only 3 FBC landfill sites in the EPRI database and 4 additional sites added by EPA, for a total of 7 FBC sites with data on onsite WMUs. Because EPA believes that this

[3] Fly ash, bottom ash, boiler slag, and FGD sludge.

small sample is not sufficient to represent the universe of FBC disposal units and, if included in the overall analysis, could bias the Monte Carlo results towards the environmental conditions around these few landfill units, FBC wastes were addressed separately from the more conventional CCW types in the full-scale analysis and are not included with the conventional and codisposal CCW management scenarios in the overall results. **Section 4.1.3** compares the risk results for each of these waste types, including FBC.

Table 2-3 shows how the plants were distributed across the waste type/WMU scenarios modeled in the full-scale analysis. The distribution across the waste type/WMU scenarios, the geographic distribution of these facilities, and the size and liner status of the WMUs were assumed to be representative of all onsite CCW landfills and surface impoundments in the continental United States as of 1995. As mentioned previously, DOE and EPA have conducted a newer survey on CCW disposal facilities (U.S. DOE, 2006), but the scope of this survey was not as comprehensive as the EPRI survey (e.g., WMU areas and capacity data were not collected). Newer information (U.S. DOE, 2007a,b) suggest that there now may be up to approximately 500 coal-fired electric utility power plants in the United States, the majority of which would be expected to conduct some waste management activities in onsite landfills or surface impoundments (U.S. EPA, 2010).

Table 2-3. Coal Combustion Plants with Onsite CCW WMUs Modeled in the Full-Scale Assessment

Waste Type and Liner Status	Number of Plants in 1995 EPRI Survey[a] with Onsite:		
	Landfills	Surface Impoundments	Either WMU Type[b]
Conventional CCW[c]	71	38	103
unlined	*38*	*24*	*60*
clay-lined	*28*	*10*	*38*
composite-lined	*10*	*5*	*15*
Codisposed CCW and coal refuse	38	65	100
unlined	*20*	*52*	*69*
clay-lined	*10*	*11*	*21*
composite-lined	*9*	*2*	*11*
FBC waste[d]	7	-	7
unlined	*3*		*3*
clay-lined	*3*		*3*
composite-lined	*1*		*1*
All waste types	108	96	181

[a] EPRI (1997); note that some coal combustion plants have one or more onsite WMUs.
[b] Number of coal combustion plants with onsite landfill(s), surface impoundment(s), or both.
[c] Fly ash, bottom ash, boiler slag, and FGD sludge.
[d] Includes 3 EPRI Survey FBC landfills plus 4 additional FBC landfills added by EPA. FBC was treated separately in the full-scale assessment because of the small number of FBC sites.

2.2 Conceptual Model

The waste stream/WMU combinations discussed above provided the waste management scenarios evaluated in the risk assessment. The full-scale assessment used the EPRI survey data to place these scenarios at actual onsite CCW disposal sites across the country. These sites were

used as the basis for a national-scale site-based Monte Carlo assessment of risks posed by the onsite disposal of CCW at utility power plants across the United States. **Figure 2-1** maps the CCW disposal sites modeled in this analysis against long-term average precipitation levels for the country.

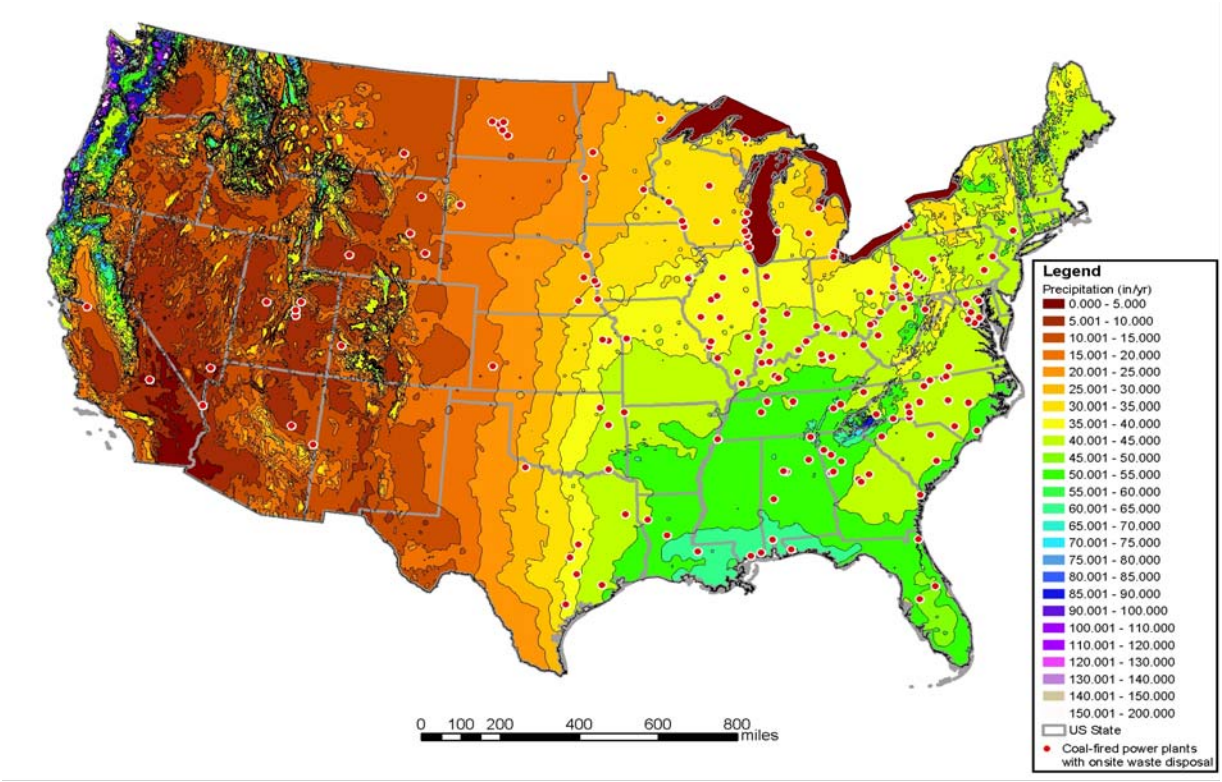


Figure 2-1. Coal combustion plants with onsite waste disposal modeled in CCW risk assessment.

2.2.1 Conceptual Site Model

Figure 2-2 depicts the conceptual site model for CCW disposal that was the basis for the national CCW risk assessment, including contaminant sources, exposure pathways, and receptors. The CCW conceptual site model includes the following exposure pathways:

Human Health:

- Groundwater to drinking water (drinking water ingestion)
- Groundwater to surface water (fish consumption)

Ecological Risk:

- Groundwater to surface water and subsequent direct contact with contaminated surface water and sediments
- Groundwater to surface water and subsequent ingestion of contaminated aquatic food items.

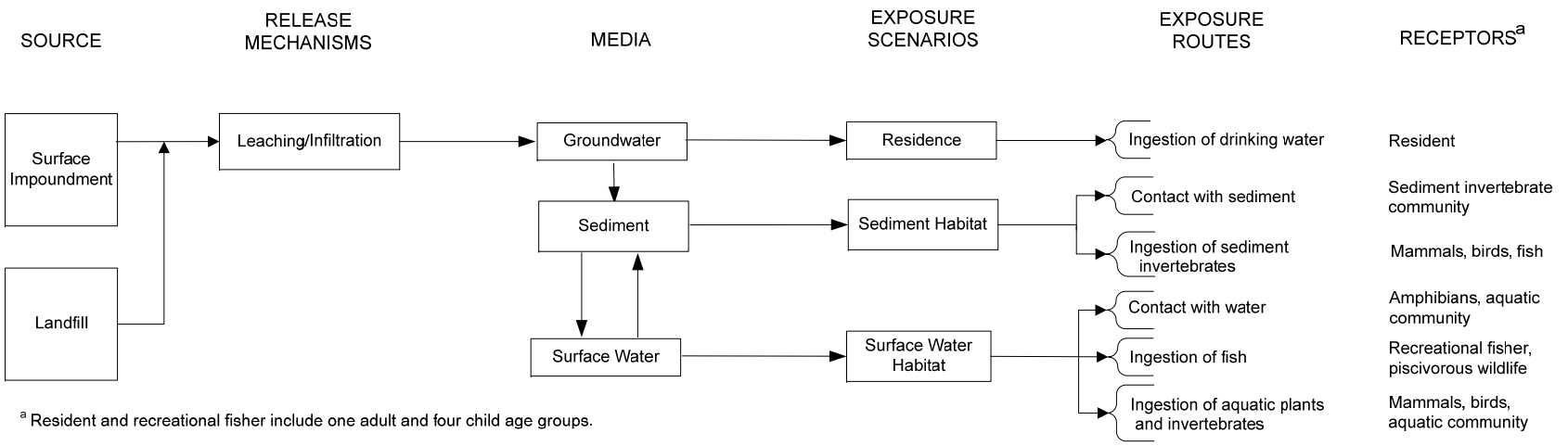


Figure 2-2. Conceptual site model of CCW risk assessment.

As shown in Figure 2-2, EPA focused full-scale modeling on groundwater-to-drinking-water and groundwater-to-surface-water exposure pathways for the national risk assessment. This groundwater pathway analysis evaluated exposures through drinking water ingestion and surface water contamination from groundwater discharge. For the groundwater-to-surface-water pathway, the analysis assumed that human exposure occurs through the consumption of contaminated fish and that ecological exposure occurs through direct contact with contaminated surface water and sediment or from the consumption of aquatic organisms.

2.2.2 Conceptual Site Layouts

This risk assessment was based on site layouts that are conceptual rather than site-specific. Although EPA had plant locations and some site-specific data on WMUs, we did not have the exact locations of each WMU or the residential wells surrounding each facility. Therefore, EPA had to develop conceptual layouts to place receptors around each WMU.

The conceptual site layouts capture possible relationships between a WMU and human and ecological receptors by locating, with respect to the WMU boundary, the geographic features (i.e., receptor wells, waterbodies) that are important for determining human and ecological exposures to chemicals released from CCW landfills and surface impoundments.

Two site layouts were used in the full-scale analysis to model the land use scenarios of most concern for CCW disposal facilities:

- Residential groundwater ingestion scenario
- Recreational fisher and aquatic ecological risk scenario.

These two conceptual site layouts are shown in the following two subsections, including WMU boundaries, waterbodies, and residential wells modeled in this analysis. In the conceptual site layouts, the WMU is represented as a square source. The size of the source was determined by the surface area of the WMU (CCW WMU areas were collected from the EPRI comanagement survey, as described in **Appendix B**). The WMU was assumed to be located at the property line of the facility to which it belongs.

Adjacent to the WMU is a buffer area within which there was assumed to be no human activity that would present human risk (i.e., there are no residences or waterbodies in the buffer). The buffer area lies between the WMU boundary and the residential well or waterbody, and represents the distance to well or waterbody discharge point modeled by the groundwater model. Each site layout must also be oriented in terms of direction.

2.2.2.1 Residential Groundwater-to-Drinking-Water Scenario

The residential groundwater-to-drinking-water scenario, shown in **Figure 2-3**, calculated exposure through residential use of well water as drinking water. In the Monte Carlo analysis, the receptor well was randomly placed up to 1 mile downgradient from the edge of the WMU (this radial well distance is labeled R_{rw} in Figure 2-3), based on a nationwide distribution of nearest downgradient residential wells from Subtitle D municipal landfills (U.S. EPA, 1988a; this distribution is provided in **Appendix C**). EPA assumed that this distribution was relevant to

onsite CCW landfills and surface impoundments at coal-fired utility power plants, but does not have data on typical distances (or the distributions of distances) of domestic drinking water wells from CCW disposal facilities. (The potential impact on the results of this assumption is discussed as an uncertainty in **Section 4.4.3.3**).

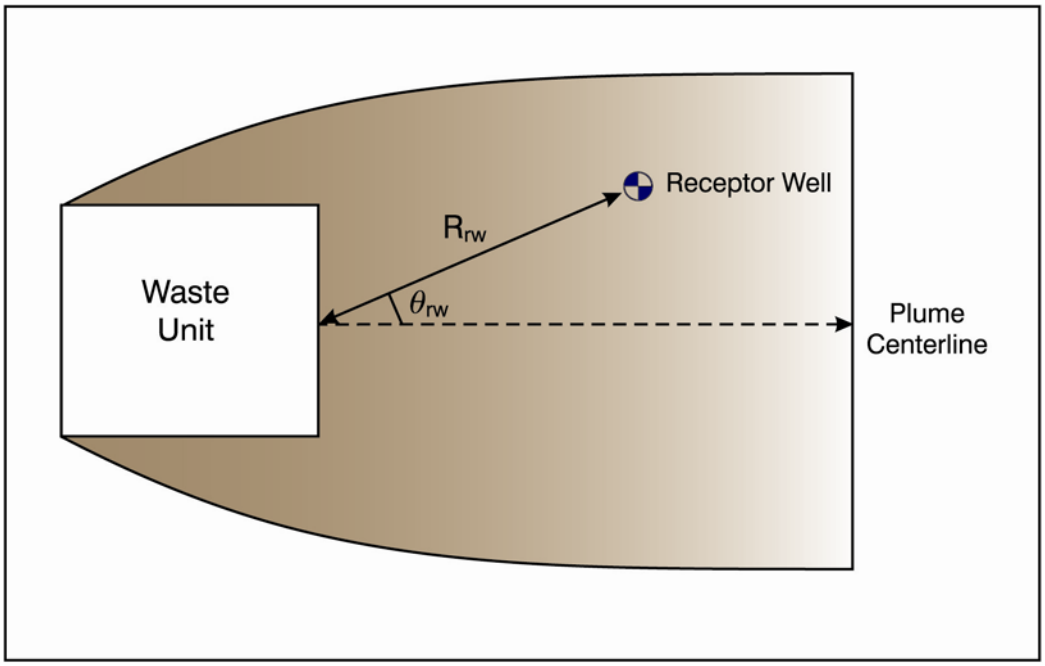


Figure 2-3. Conceptual site layout for residential groundwater ingestion scenario.

The angle off the contaminant plume centerline (θ_{rw} in Figure 2-3) was based on a uniform distribution ranging from 0 to 90°. The depth of the well below the water table was set within the groundwater model based on assumptions that are generally typical of average conditions for surficial aquifers across the United States. These limits are discussed in **Section 3.6.3**. In this assessment, receptors were always located within the lateral extent of the plume.

The soil and aquifer characteristics needed for the groundwater model were based on available data on soil and groundwater conditions collected around the 181 modeled sites, as described in **Appendix C**.

2.2.2.2 Recreational Fisher and Ecological Risk Scenario

The recreational fisher[4] scenario, shown in **Figure 2-4**, was used to estimate risks to recreational fishers (and their children) who live near the CCW landfills and surface impoundments and catch and consume fish from a waterbody located adjacent to the buffer and contaminated by CCW constituents through the groundwater to surface water pathway. The

[4] Only recreational fishers were considered as the reasonable maximum exposed individuals. Subsistence receptors who eat fish were not modeled, but could be expected to have higher risks than the recreational fishers for whom we present results.

potential for cumulative exposure from both contaminated fish and groundwater was not considered in the CCW risk assessment. One reason is that the exposures are likely to occur over different timeframes because of differences in transit time of the contaminant plume to wells versus surface waterbodies. As described in **Section 3.6.3**, for each model run in the Monte Carlo analysis, the distances to the downgradient well and surface water were independently sampled from national distributions presented in Appendix C, Tables C-1 (wells) and C-2 (surface waterbodies). Also, these exposures may involve different receptors because a resident exposed via groundwater may not be a recreational fisher. Thus, adding risks across pathways would not likely change the results.

The waterbody was assumed to be a stream or lake located downgradient from the WMU, beginning where the buffer area ends (see Figure 2-4), and was also used as the most impacted aquatic system for the ecological risk assessment. Waterbody characteristics were determined based on a combination of site-specific, regional, or national data (as described in **Appendix C**), except for the length of the stream impacted by the plume, which was determined by the width of the plume as it intersects the waterbody.

The downgradient distance to the surface water body was determined from a national distribution developed by measuring this distance (using scaled U.S. Geological Survey [USGS] maps and aerial photographs obtained from the Terraserver Web site [http://terraserver.usa.com/geographic.aspx]) at 59 CCW landfill and surface impoundment sites randomly selected from a larger data set of 204 CCW WMUs, including those modeled in this risk assessment. **Appendix C** presents that distribution and further details on how the distribution was developed and the sample of 59 facilities used to develop the distribution.

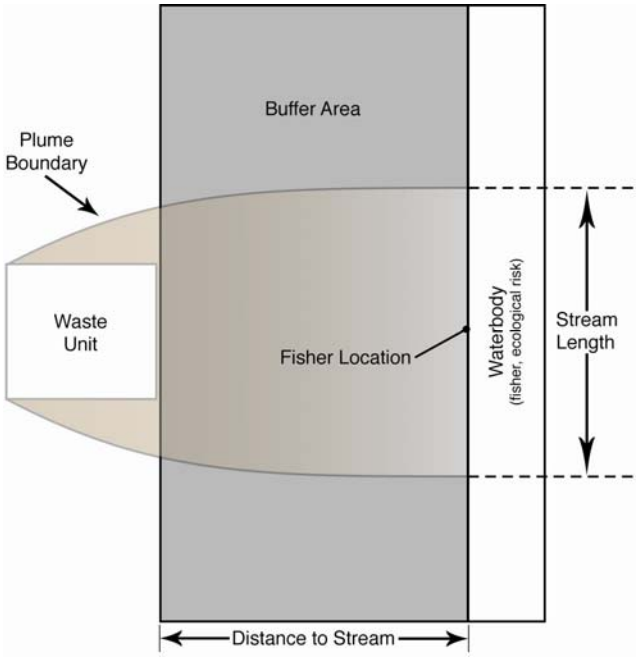


Figure 2-4. Conceptual site layout for residential fisher and aquatic ecological risk scenario.

2.3 Screening Analysis

To assist in selecting constituents for full-scale modeling, a screening analysis was conducted in 2002 (U.S. EPA, 2002a) that compared very conservative estimates of exposure concentrations (e.g., leachate concentrations) to health-based concentration benchmarks to quickly and simply identify constituents with risks that clearly do not exceed the risk criteria so that these could be eliminated from further analysis. For example, for the groundwater-to-drinking-water pathway, leachate concentrations were compared directly to drinking water standards, which is equivalent to assuming that human receptors are drinking leachate.[5] Similarly conservative estimates were used for ecological receptors (e.g., fish swimming directly in leachate). EPA made use of those screening results in this risk assessment, which was conducted in 2003 and documented in the August 6, 2007, draft report and its subsequent revisions, including the current document. **Section 3.2** provides further detail on how the CCW screening analysis was conducted to develop the list of CCW constituents modeled in the full-scale analysis.

2.4 Full-Scale Risk Assessment

Although the screening analysis identified the potential for risk for a subset of the constituents reported in CCW, the conservative assumptions used precluded an accurate quantitative estimate of these risks. The screening results were not intended to, and do not, characterize the risks that we expect would actually occur, because the purpose is not to characterize risks but rather to identify those constituent/pathway/receptor combinations that are unlikely to be problematic versus those that are most likely to be problematic. To gain a better understanding of the risks that may be posed by the constituents identified as likely to be problematic, EPA conducted a full-scale probabilistic (Monte Carlo) risk assessment to estimate the national distribution of the risks to human health and the environment posed by CCW disposal, and to provide the information needed to assess future management options for these wastes in the context of their risks to human health and the environment. The full-scale CCW Monte Carlo risk assessment was designed to characterize the national CCW risk profile in terms of WMU type, waste type, and constituent, and to use distributions in a probabilistic modeling framework to incorporate variability and uncertainty into the analysis.

The full-scale modeling approach used data about waste management practices and environmental conditions at 181 utility CCW disposal sites across the United States.[6] These sites were assumed to represent the universe of CCW onsite waste disposal sites at the time of the EPRI survey (1995) and defined the national framework for the risk assessment. One question related to this risk assessment is how CCW facilities may have changed since the 1995 EPRI survey. Although the DOE/EPA survey did not include all of the data needed to conduct a risk assessment (WMU area and capacity data were not collected), liner conditions were addressed, and by comparing the DOE/EPA survey results to the EPRI data, it is possible to assess how

[5] Note that RCRA waste disposal risk assessments do not address direct discharges from impoundments to surface waters because they are regulated as permitted point source discharges under the Clean Water Act by EPA's Office of Water.

[6] These 181 sites include177 sites from the EPRI survey and 4 additional CCW sites added by EPA to better represent FBC waste disposal facilities; see Section 2.1.2.

liner conditions have changed as CCW facilities were built or expanded since 1995. The 56 WMUs surveyed in the U.S. DOE (2006) study were commissioned between 1994 and 2004. Although the actual number of WMUs that were established in that timeframe cannot be verified, based on proxy data (i.e., CCW available for disposal in those states with identified new WMUs and coal-fired power plant generating capacity), the sample coverage was estimated to be at least 61 to 63 percent of the total population of the newly commissioned WMUs.[7] With the exception of one landfill, the newly constructed facilities are all lined, with either clay, synthetic, or composite liners. The single unlined landfill identified in the recent DOE report receives bottom ash, which is characterized as an inert waste by the state, and therefore, a liner is not required. There has been a marked trend away from unlined WMUs in favor of lined units, with a distinct preference for synthetic or composite liners. A comparison of the 26 coal combustion plants in both the EPRI survey and the DOE/EPA survey (U.S. DOE, 2006) showed that although most of those facilities (17 of 26) were using unlined WMUs in 1995, all 26 are now placing wastes in new or expanded landfills or surface impoundments that are lined with clay, synthetic, or composite liners. However, it is likely that the older unlined units were closed with wastes in place, and that these wastes therefore still pose a threat through groundwater pathways. In addition, the available data cannot be used to determine the number of unlined units that continue to operate in the United States. See further discussion of the uncertainty posed by the use of the EPRI liner data in **Section 4.4.1**.

The full-scale assessment was conducted using several modeling components: (1) EPA's Composite Model for Leachate Migration with Transformation Products (EPACMTP; U.S. EPA, 1997a) groundwater model, (2) a simple steady-state surface water and aquatic food web model, and (3) a multipathway exposure and risk modeling system.

2.4.1 Data Collection

For the sites representing each WMU and waste type combination selected for analysis, the Monte Carlo analysis began with input files that contain, for each Monte Carlo realization, the following variables collected at and around each of the 181 modeled sites:

- WMU area, depth, and capacity
- WMU liner status (no liner, clay liner, composite liner)
- Waste type (conventional CCW, CCW codisposed with coal refuse, FBC wastes)
- Soil texture (for vadose zone properties and infiltration rates)
- Soil pH and organic carbon
- Aquifer type
- Groundwater temperature
- Climate center (for infiltration rates)
- USGS Hydrologic Region (for surface water quality data)
- Surface water type and flow conditions.

[7] For additional details as to how these estimates were derived, the reader is referred to the DOE study, pages S-2 to S-3 of the Summary Section and Section 3.1.2.

Data sources and collection methods for these variables may be found in **Appendices B** and **C**.

CCW constituent data in the CCW constituent database were used as a national empirical distribution of the concentrations of the constituents of concern in CCW landfill leachate and surface impoundment porewater. Like the WMU database, the CCW constituent data include WMU type and waste type, which enabled constituent concentrations to be assigned to the 181 CCW sites by waste type and WMU type. The CCW constituent database was also used to assign (by waste type) the high, medium, and low leachate pH and ionic strength conditions needed to select the appropriate subsurface sorption isotherms for each model run (see **Appendix D**).

Because site-specific data were not readily available, national distributions were used to populate the following variables by model run:

- Distance to nearest drinking water well
- Distance to nearest surface waterbody
- Aquifer depth, thickness, gradient, and hydraulic conductivity (based on site-specific hydrogeologic setting)
- Soil hydrologic properties (based on site-specific soil type).

The data sources used to develop national distributions for these variables are described in **Appendix C**. Human exposure factors, such as exposure duration and drinking water and fish consumption rates, were also based on national distributions, which are provided in **Appendix F**.

2.4.2 Model Implementation

As a first step in the modeling process, the groundwater model (EPACMTP) read the site-based data files to estimate the following for each model run:

- Drinking water well peak concentration
- Time to drinking water peak concentration
- Peak surface water contaminant flux
- Time to peak surface water contaminant flux.

The groundwater model was run until contaminant concentrations at the receptor point returned to zero after the concentration peak or for the maximum simulation time of 10,000 years, whichever came first.

Groundwater model results were passed to the multimedia modeling system to estimate surface water and sediment concentrations and to calculate human and ecological exposure and risk. Additional inputs sent to the model at this stage included

- Site-based surface waterbody type, dimensions, flows, pH, and total suspended solids (TSS) concentration
- Chemical-specific fish bioconcentration factors (BCFs)
- Human exposure factors (from national distributions)
- Human and ecological health benchmarks.

For human health, the multimedia modeling system calculated risk from drinking water ingestion and fish consumption for each realization. For ecological risk, the model used surface water and sediment concentrations along with ecological benchmarks to estimate the risks to ecological receptors.

2.4.3 Exposure Assessment

Table 2-4 lists the human and ecological receptors considered in the CCW risk assessment, along with the specific exposure pathways that apply to each receptor. All of the receptors that EPA considered were assumed to live offsite, at a location near the WMU.

Table 2-4. Receptors and Exposure Pathways Addressed in the Full-Scale CCW Assessment

Receptor	Ingestion of Drinking Water	Fish Consumption	Direct Contact with Surface Water and Sediment	Ingestion of Aquatic Organisms
Human Receptors				
Adult resident	✔			
Child resident	✔			
Adult recreational fisher		✔		
Child recreational fisher		✔		
Ecological Receptors				
Aquatic and sediment organisms			✔	
Mammals and birds				✔

For human receptors, the exposure assessment estimated the dose to an individual receptor by combining modeled CCW constituent concentrations in drinking water or fish with intake rates for adult and child receptors. The full-scale CCW risk assessment considered exposures due to chemicals leaching from WMUs and contaminating groundwater. The groundwater exposures include drinking water ingestion and consumption of recreationally caught contaminated fish from surface waterbodies affected by contaminated groundwater. For the groundwater-to-drinking-water pathway, it was assumed that well water was the only source

of drinking water (although some households may drink bottled or treated water or may drink water outside the home, e.g., at work or at school).

For ecological receptors, exposure assumptions were incorporated into the development of ecological benchmarks (see **Appendix H**), which were surface water and sediment concentrations corresponding to an HQ of 1.

The time period for the exposure assessment was defined by the peak concentration in the media of concern and the exposure duration. For human receptors, annual average media concentrations were averaged over the randomly selected exposure duration around the peak concentration for each run. To protect against chronic effects to ecological receptors, EPA considered the exposure duration over a significant portion of the receptor's lifetime, and we believe that one year is the appropriate period of time for that. To be protective, we used the highest (peak) annual average concentration to estimate ecological exposure and risk.

2.4.4 Risk Estimation

Risk was estimated using several risk endpoints as particular measures of human health risk or ecological hazard. A risk endpoint is a specific type of risk estimate (e.g., an individual's excess cancer risk) that is used as the metric for a given risk category. The CCW risk assessment evaluated cancer and noncancer endpoints for humans and noncancer endpoints for ecological receptors. For human risk, the availability of toxicological benchmarks for cancer and noncancer effects determined which endpoints were evaluated for each constituent.

EPA used two risk endpoints to characterize risk for the human receptors and a single risk endpoint, total HQ, to characterize risk for ecological receptors. These endpoints are discussed in **Section 3.9**; in addition, uncertainty related to these endpoints is discussed in **Sections 4.4.2** (exposures to multiple constituents) and **4.4.3.4** (benchmark uncertainties).

From the distribution of risks for each risk endpoint generated by the Monte Carlo analysis, the 50th and 90th percentile risks were selected and compared to a risk range of 1 in 1,000,000 to 1 in 10,000 excess cancers and a hazard quotient greater than 1 for noncarcinogenic effects. A hazard quotient greater than 1 was also used for the ecological risk criterion in the full-scale risk assessment.

3.0 Analysis

The CCW risk analysis evaluated risks from CCWs disposed of in landfills and surface impoundments located onsite at coal-fired utility power plants across the United States based primarily on data collected in 1995 by EPRI (1997).[1] Chemical constituents found in CCW can be released from these WMUs into the surrounding environment by releases through leachate to the subsurface underlying the WMU. Leachate forms in both landfills and surface impoundments, migrates from the WMU through soil to groundwater, and is transported in groundwater to drinking water wells (groundwater-to-drinking-water pathway) and into surface waterbodies near the WMU (groundwater-to-surface-water pathway).

To select the constituents for full-scale modeling, the screening analysis compared very conservative estimates of exposure concentrations (e.g., leachate concentrations) to health-based concentration benchmarks to quickly, simply, and safely identify constituents with risks that clearly do not exceed the risk criteria so that these could be eliminated from further analysis. For example, leachate concentrations were compared directly to drinking water standards, which is equivalent to assuming that human receptors are drinking leachate.

For the full-scale analysis, EPA used computer-based models and sets of equations to estimate the risk to human health and the environment from current CCW disposal practices.[2] These models included

- **Source** models that simulate the release of CCW constituents in leachate from landfills and surface impoundments[3]
- **Fate and transport** models that estimate contaminant concentrations in environmental media such as groundwater and surface water
- **Exposure** models that estimate daily contaminant doses for humans and ecological receptors exposed to CCW constituents in environmental media that were not screened out
- **Risk** models that calculate risks to humans and ecological receptors.

This section describes the data, models, and equations used for CCW constituent screening, as well as those used to calculate exposure point concentrations and risk in the full-

[1] The selection and characterization of these CCW WMUs are described in more detail in Appendix B.

[2] As discussed in **Section 2**, the 1995 EPRI survey data was assumed to represent current CCW management practices. However, new data from a more recent DOE/EPA survey suggest that liners may be more prevalent in new and expanded units built since 1994. **Section 4** discusses implications of this uncertainty on the risk assessment results.

[3] EPA used source-term models integrated into EPACMTP to estimate environmental releases of constituents in leachate from landfills and surface impoundments.

scale analysis. **Section 3.1** describes the health benchmarks used to develop human and ecological risk estimates for screening and full-scale analysis. **Section 3.2** describes the screening analysis, along with how the screening results were used to select constituents for the full-scale analysis. **Section 3.3** provides the overall structure for the full-scale analysis, including the spatial and temporal framework and the probabilistic (Monte Carlo) framework for the model runs. **Sections 3.4** and **3.5** describe the landfill and surface impoundment source models used to predict environmental releases of constituents from CCW. **Sections 3.6** and **3.7** describe the fate and transport modeling used to predict contaminant concentrations in groundwater and surface water. **Section 3.8** describes the human exposure calculations and **Section 3.9** describes how risks were calculated for human and ecological receptors.

Supporting detail can be found in the following appendices:

- **Appendix A, CCW Constituent Data**, provides the CCW constituent concentrations used and describes how they were collected and processed for both the screening and full-scale analyses
- **Appendix B, Waste Management Unit Data**, describes the location and characteristics of each landfill and surface impoundment modeled and describes how the source model input parameter values were collected for the full-scale analysis
- **Appendix C, Site Data**, describes how environmental data around each CCW waste disposal site were collected to provide inputs for the groundwater and surface water modeling
- **Appendix D, MINTEQA2 Nonlinear Sorption Isotherms**, describes the development and application of the CCW-specific MINTEQ metal sorption isotherms used to model fate and transport in soils and groundwater
- **Appendix E, Surface Water, Fish Concentration, and Contaminant Intake Equations**, documents the algorithms used to calculate surface water concentrations, fish concentrations, and drinking water and fish intake rates
- **Appendix F, Human Exposure Factors**, documents the human exposure parameters and equations used for calculating the environmental exposure from CCW disposal
- **Appendix G, Human Health Benchmarks**, describes how the human toxicity benchmarks were selected and developed for CCW constituents
- **Appendix H, Ecological Benchmarks**, describes how the ecological toxicity benchmarks were selected and developed for CCW constituents
- **Appendix I, Calculation of HBNs,** describes how health-based numbers were calculated for the screening analysis
- **Appendix J, Chemical-Specific Inputs Used in the Screening Analysis,** describes additional chemical-specific data used in the screening analysis

- **Appendix K, Screening Analysis Results,** provides the results of the screening analysis for human and ecological receptors.

- **Appendix L, Time of Travel to Receptor Well,** provides figures showing, for selected CCW constituents, cumulative percentiles of the time it took for the peak concentration to reach a receptor well for each source type.

3.1 Toxicity Assessment

The assessment of human risks from disposal of a waste stream like CCW begins by assessing, for constituents in the waste, the ability of each chemical to cause an adverse human health effect, which depends on the toxicity of the chemical, the chemical's route of exposure to an individual (ingestion, inhalation, or direct contact), the duration of exposure, and the dose received (the amount that a human ingests or inhales). Similar principles apply to ecological receptors, although exposure duration is much shorter than for human receptors because humans generally live longer than ecological receptors. For a risk assessment, the toxicity of a constituent is defined by a human health or ecological benchmark for each route of exposure. A benchmark is a quantitative value used to predict a chemical's possible toxicity and ability to induce an adverse effect at certain levels of exposure. Because different chemicals cause different health effects at different doses, benchmarks are chemical-specific.

Appropriate human health and ecological benchmarks for the constituents of potential concern in CCW wastes were collected for use in the screening assessment and in the full-scale risk assessment. Although these assessments were conducted in 2002 and 2003, the benchmarks and risks presented in this 2009 report were updated to reflect current toxicity data.[4] The data sources and collection methodology for these benchmarks are described briefly in **Sections 3.1.1** (human health benchmarks) and **3.1.2** (ecological benchmarks), and in more detail in **Appendix G** (human health benchmarks) and **Appendix H** (ecological benchmarks).

3.1.1 Human Health Benchmarks

Human health benchmarks for chronic oral exposures were needed for the full-scale analysis. These health benchmarks were derived from toxicity data based on animal studies or human epidemiological studies. Each benchmark represents a dose-response estimate that relates the likelihood and severity of adverse health effects to exposure and dose. This section presents the noncancer and cancer benchmarks used to evaluate human health effects that may result from exposure to the constituents modeled.

Chronic human health benchmarks were used to evaluate potential noncancer and cancer risks. These include reference doses (RfDs) to evaluate noncancer risk from oral exposures and oral cancer slope factors (CSFs) to evaluate cancer risk from oral exposures. The benchmarks are chemical-specific and do not vary between age groups.

[4] Because the risk calculations are linear and occur at the end of the analysis, all screening and full-scale results can be simply scaled to accommodate any changes in human health and ecological benchmarks.

- The **RfD** is an estimate (with uncertainty spanning perhaps an order of magnitude) of a daily exposure to the human population (including sensitive subgroups) that is likely to be without appreciable risk of deleterious noncancer effects during a lifetime. The RfD provides a reference point to gauge the potential effects (U.S. EPA, 2002c). At exposures increasingly greater than the RfD, the potential for adverse health effects may increase, although this potential cannot be quantified. Lifetime exposure above the RfD does not imply that an adverse health effect would necessarily occur.

- The **CSF** is an upper-bound estimate (approximating a 95 percent confidence limit) of the increased human cancer risk from a lifetime exposure to an agent. Because this is an upper-bound estimate, true risk is likely lower. This estimate is usually expressed in units of proportion (of a population) affected per milligram of agent per kilogram body weight per day (mg/kg-d). Unlike RfDs, CSFs do not represent "safe" exposure levels; rather, they relate levels of exposure with a probability of effect or risk.

Human health benchmarks are available from several sources. Health benchmarks developed by EPA were used whenever they were available. Sources of human health benchmarks were used in the following order of preference:

- Integrated Risk Information System (IRIS) (U.S. EPA, 2002c)
- Superfund Technical Support Center Provisional Benchmarks
- Health Effects Assessment Summary Tables (HEAST) (U.S. EPA, 1997f)
- Various other EPA health benchmark sources
- ATSDR minimal risk levels (MRLs) (ATSDR, 2002).

These sources and the selection hierarchy are described in more detail in **Appendix G**.

The chronic human health benchmarks used in the screening and full-scale analysis are summarized in **Table 3-1**. For most constituents, human health benchmarks were available from IRIS. Benchmarks for a few constituents were obtained from ATSDR. For chemicals for which purely health-based benchmarks were not available (lead), a drinking water action level was used (U.S. EPA, 2002d).

Table 3-1. Human Health Benchmarks Used in the CCW Risk Assessment

Constituent	Type of Benchmark[a]	Value	Units	Source[b]
Cancer Benchmark				
Arsenic	CSF	1.5E+00	$(mg/kg\text{-}d)^{-1}$	IRIS
Noncancer Benchmarks				
Aluminum	RfD	1.0E+00	mg/kg-d	PPRTV
Antimony	RfD	4.0E-04	mg/kg-d	IRIS
Arsenic	RfD	3.0E-04	mg/kg-d	IRIS
Barium	RfD	2.0E-01	mg/kg-d	IRIS
Beryllium	RfD	2.0E-03	mg/kg-d	IRIS

(continued)

Human Health Benchmarks Used in the CCW Risk Assessment (continued)

Constituent	Type of Benchmark[a]	Value	Units	Source[b]
Boron	RfD	2.0E-01	mg/kg-d	IRIS
Cadmium	RfD (water)[c]	5.0E-04	mg/kg-d	IRIS
	RfD (food)[d]	1.0E-03	mg/kg-d	IRIS
Chromium III	RfD	1.5E-00	mg/kg-d	IRIS
Chromium VI	RfD	3.0E-03	mg/kg-d	IRIS
Cobalt	RfD	3.0E-04	mg/kg-d	PPRTV
Copper	RfD	1.0E-02	mg/kg-d	ATSDR
Cyanide	RfD	2.0E-02	mg/kg-d	IRIS
Fluoride	RfD	6.0E-02	mg/kg-d	IRIS
Lead	MCL	1.5E-02	mg/L	DWAL
Manganese	RfD (food)	1.4E-01	mg/kg-d	IRIS
	RfD (water, soil)	4.7E-02	mg/kg-d	IRIS
Mercury (divalent)	RfD (food, water, soil)	3.0E-04	mg/kg-d	HEAST
	RfD (fish)	1.0E-04	mg/kg-d	IRIS
Molybdenum	RfD	5.0E-03	mg/kg-d	IRIS
Nickel	RfD	2.0E-02	mg/kg-d	IRIS
Nitrate	MCL	1.0E+01	mg/L	DWAL
	RfD	1.6E+00	mg/kg-d	IRIS
Nitrite	RfD	1.0E-01	mg/kg-d	IRIS
Selenium	RfD	5.0E-03	mg/kg-d	IRIS
Silver	RfD	5.0E-03	mg/kg-d	IRIS
Strontium	RfD	6.0E-01	mg/kg-d	IRIS
Thallium	RfD	8.0E-05	mg/kg-d	IRIS
Vanadium	RfD	7.0E-03	mg/kg-d	HEAST
Zinc	RfD	3.0E-01	mg/kg-d	IRIS

[a] MCL = maximum concentration limit

[b] References:
ATSDR: Minimal Risk Levels, ATSDR (2009)
DWAL: Drinking Water Action Level, U.S. EPA (2002d)
HEAST: U.S. EPA (1997f)
IRIS: U.S. EPA (2009a)
PPRTV: Provisional peer-reviewed toxicity value (U.S. EPA, 2006a, 2006b, 2008a)

[c] Used for drinking water ingestion.

[d] Used for fish ingestion.

Cadmium has two RfDs, one for exposures via water and one for exposures via food. The RfD for water was used for drinking water ingestion and the RfD for food was used for fish consumption.

3.1.2 Ecological Benchmarks

The ecological risk assessment addressed two routes of exposure for ecological receptors, direct contact with contaminated media and ingestion of contaminated food items. For each constituent for which ecological effect data were available, HQs were calculated using chemical-specific media concentrations assumed to be protective of ecological receptors of concern. To calculate ecological HQs, these media concentrations (also known as chemical stressor concentration limits [CSCLs]) were divided by the estimated media concentrations. The CSCLs are media-specific environmental quality criteria intended to represent a protective threshold value for adverse effects to various ecological receptors in aquatic ecosystems (surface water and sediment). The CSCLs were developed to be protective of the assessment endpoints chosen for this assessment. An HQ greater than 1 indicates that the predicted concentration exceeds the CSCL, and therefore, the potential for adverse ecological effects exists. In this regard, the use of CSCLs to calculate an ecological HQ is analogous to the use of the reference concentration (RfC) for human health where the air concentration is compared to the health-based concentration (the RfC), and an HQ greater than 1 is considered to indicate the potential for adverse health effects.

Table 3-2 shows the receptor types assessed for each exposure route (direct contact and ingestion) in each environmental medium addressed by the full-scale CCW risk assessment.

Table 3-2. Ecological Receptors Assessed by Exposure Route and Medium (Surface Water or Sediment)

Receptor Type	Surface Water (water column)	Surface Water Sediment
Direct Contact Exposure		
Aquatic Community	✔	
Sediment Community		✔
Amphibians	✔	
Aquatic Plants and Algae	✔	
Terrestrial Plants		
Ingestion Exposure		
Mammals	✔	
Birds	✔	

Ecological receptors that live in close contact with contaminated media are considered to be potentially at risk. For the screening and full-scale analysis, these receptors are exposed through direct contact with contaminants in surface water and sediment. The benchmarks for receptor communities (aquatic or sediment communities) are not truly *community-level* concentration limits in that they do not consider predator-prey interactions. Rather, they are based on the theory that protection of 95 percent of the species in the community will provide a sufficient level of protection for the community (see, for example, Stephan et al., 1985, for additional detail). **Appendix H** summarizes the benchmark derivation methods for each receptor assessed for the direct contact route of exposure.

For surface water and sediments, the ingestion route of exposure addresses the exposure of terrestrial mammals and birds through ingestion of aquatic plants and prey. Thus, the benchmarks for ingestion exposure represent media concentrations that, based on certain assumptions about receptor diet and foraging behavior, are expected to be protective of populations of mammals and birds feeding and foraging in contaminated areas.

For birds and mammals, the derivation of ingestion benchmarks required the selection of appropriate ecotoxicological data based on a hierarchy of sources. The assessment endpoint chosen for birds and mammals was population viability and therefore, the ingestion benchmarks were based on study data for physiological effects that are relevant to populations. These data included measures of reproductive fitness, developmental success, survival, and other toxicological effects that could have an impact on the population rather than just the health of an individual animal. Choosing these measures of effect provided the basis to evaluate the potential for adverse effects at the population level by inference; this analysis did not evaluate the effects on population dynamics in the sense that a reduction in the population was predicted over time in response to exposure to constituents released from CCW. Population-level modeling was beyond the scope of this risk assessment.

Once an appropriate ingestion exposure study was identified, a benchmark was calculated. **Appendix H** describes the basic technical approach used to convert avian or mammalian benchmarks (in daily doses) to the CSCLs (in units of concentration) used to assess ecological risks for contaminated surface water and sediment. The methods reflect exposure through the ingestion of contaminated plants, prey, and various media, and include parameters on accumulation (e.g., BCFs), uptake (e.g., consumption rates), and dietary preferences.

Where multiple ecological benchmarks were available for a pathway of interest, the benchmark that produced the lowest (most sensitive) CSCL for each chemical in each medium was used. For example, several types of receptors (the aquatic community, amphibians, aquatic plants, mammals, birds) can be exposed to contaminants in surface water. The surface water criterion for a given constituent represents the lowest CSCL for these receptors, and thus gives the highest (most protective) HQ. The CSCLs used to assess ecological endpoints in the full-scale analysis and the associated receptor are summarized in **Table 3-3**. Additional details on the CCW ecological benchmarks and CSCLs and their development can be found in **Appendix H** and in U.S. EPA (1998a).

Table 3-3. Ecological Risk Criteria Used for Surface Water and Sediment

Constituent	Medium[a]	Exposure Route	CSCL	Units	Receptor
Aluminum	Surface Water	Direct contact	0.09	mg/L	Aquatic biota
Arsenic total	Sediment	Ingestion	0.51	mg/kg	Spotted sandpiper
Arsenic III	Surface Water	Direct contact	0.15	mg/L	Aquatic biota
Arsenic IV	Surface Water	Direct contact	8.10E-03	mg/L	Aquatic biota
Barium	Sediment	Ingestion	190	mg/kg	Spotted sandpiper
	Surface Water	Direct contact	4.00E-03	mg/L	Aquatic biota
Boron	Surface Water	Direct contact	1.60E-03	mg/L	Aquatic biota

(continued)

Ecological Risk Criteria Used for Surface Water and Sediment (continued)

Constituent	Medium[a]	Exposure Route	CSCL	Units	Receptor
Cadmium	Sediment	Direct contact	0.68	mg/kg	Sediment biota
	Surface Water	Direct contact	2.50E-03	mg/L	Aquatic biota
Cobalt	Surface Water	Direct contact	0.02	mg/L	Aquatic biota
Lead	Sediment	Ingestion	0.22	mg/kg	Spotted sandpiper
	Surface Water[b]	Ingestion	3.00E-04	mg/L	River otter
Selenium total	Surface Water	Direct contact	5.00E-03	mg/L	Aquatic biota
Selenium IV	Surface Water	Direct contact	0.03	mg/L	Aquatic biota
Selenium VI	Surface Water	Direct contact	9.5E-03	mg/L	Aquatic biota

Source: U.S. EPA (1998a)

[a] If a medium (surface water or sediment) is not listed, there were insufficient data to develop a benchmark for it.

[b] Includes ingestion of fish.

3.2 Constituent Screening

The screening risk analysis was designed to select the CCW constituents for full-scale exposure modeling. The groundwater pathway screening evaluated exposure through drinking and surface water contamination[5] from groundwater. The analysis considered risks to both human and ecological receptors. Waste constituents that passed the screen (i.e., were below target risk/hazard criteria) were assumed to pose *de minimis* risks and were not addressed in the full-scale modeling.

3.2.1 Waste Constituent Concentrations

The CCW screening analysis addressed metals and inorganic compounds identified as described in **Section 2.1.1.2**. Waste concentrations were available for most of these constituents from the CCW constituent database described in **Section 2.1.1** and **Appendix A**. The CCW constituent database includes waste analysis data for CCW leachate, surface impoundment and landfill porewater, and whole waste samples, and was used in the screening analysis as follows:

- Analyte concentrations (in mg/L) porewater sampled from surface impoundment sediments represent surface impoundment leachate affecting the groundwater pathways
- To represent landfill leachate, the different types of landfill leachate and porewater data in the CCW constituent database were selected based on a hierarchy developed to best represent CCW landfill waste concentrations at a wide variety of sites and waste disposal conditions.

To allow screening decisions to be made by waste constituent, waste stream, and exposure pathway, CCW data were processed to produce a single concentration per analyte and waste

[5] For the groundwater-to-surface-water pathway, the analysis assumed that human exposure occurs through the consumption of contaminated fish. Ecological exposure occurs through direct contact to contaminated surface water and sediment and consumption of aquatic organisms.

stream (surface impoundment porewater and landfill leachate) for comparison with health-based numbers (HBNs) and CSCLs. Data processing to create these analyte concentrations involved two steps:

- **Calculation of average constituent concentrations by site for landfill leachate, surface impoundment porewater, and total ash concentrations.** Site averaging avoids potential bias toward sites with many analyses per analyte. During site averaging, separate waste disposal scenarios at a site (e.g., non-FBC and FBC ash; FGD sludge and ash) were treated as separate "sites" and averaged independently. Nondetects were averaged at one-half the reported detection limit.[6]

- **Selection of screening concentrations from site-averaged values.** For the screening calculations, the analysis used the 90th percentile of the site-averaged concentrations across all sites for landfill leachate and surface impoundment porewater.

Appendix A describes the CCW constituent database and how the waste constituent concentrations were selected and processed for the screening analysis and full-scale risk assessment.

3.2.2 Media-Specific Exposure Concentrations for Screening

The screening analysis required media concentrations for groundwater, surface water, and sediment to compare with the HBNs and CSCLs. As a simple first screen of risk, the analysis used waste concentrations as protective estimates of offsite groundwater and surface water concentrations.

For groundwater-to-drinking-water exposures, the analysis used the 90th percentile waste porewater[7] and leachate concentrations to represent groundwater contamination from the surface impoundment and landfill, respectively. No dilution or attenuation was assumed between the WMU and the drinking water well because the large size range of CCW units precluded the use of a dilution attenuation factor (DAF)[8] for a nearby well. Similarly, surface water concentrations were assumed to be equivalent to waste leachate and porewater concentrations.

3.2.3 Screening Methodology

The CCW screening approach compared protective health-based concentrations in each medium of concern with estimated offsite media concentrations of CCW constituents described in **Section 3.2.2**. Both human and ecological receptors were addressed. HBNs are media concentrations developed to protect human health, and CSCLs are media concentrations developed to protect ecological receptors. HBNs were calculated based on the target risk criteria

[6] Appendix A contains figures showing how site-averaged 90th percentile concentrations and 90th percentile concentrations taken across all analyses (nonaveraged concentrations) compare with HBNs for surface impoundment porewater, toxicity characteristic leaching procedure (TCLP) leachate, and whole waste concentrations.

[7] Although the 95th percentile was used in 1998, the 90th percentile was used in this analysis as a reasonably conservative value considering the protective screening analysis assumptions and the larger 2002 constituent data set.

[8] A DAF is the waste concentration divided by the media concentration at the point of exposure.

for the screening analysis: an HQ of 1 (for noncarcinogens) or an excess cancer risk level of 10^{-5}. CSCLs were calculated based on an HQ of 1. A full description of the development of the HBNs can be found in **Appendix I**. Development of the CSCLs used for screening based on ecological risks is provided in **Appendix H**.

Screening involved developing these HBNs and CSCLs, as well as developing the waste constituent or media concentrations to be used in the comparison and estimating the risk associated with these concentrations. Pathways and waste streams evaluated in the analysis include those summarized in **Table 3-4**, along with the basic assumptions and methods used to evaluate each pathway in the screening analysis.

Table 3-4. Exposure Pathways Evaluated In CCW Constituent Screening

Exposure Pathway	Methodology
Groundwater-to-drinking-water	Compared drinking water HBNs to landfill leachate and surface impoundment porewater concentrations
Groundwater-to-surface-water (fish consumption; ecological)	Compared surface water HBNs and CSCLs to landfill leachate and surface impoundment porewater concentrations
Direct exposure to surface impoundment CCW (ecological only)	Compared surface water CSCLs to CCW surface impoundment constituent concentrations from the 1998 CCW risk assessment

3.2.3.1 HBN Calculations

HBNs represent media concentrations that are protective of human health from exposure pathways that are relevant to that particular medium. The exposure scenarios assumed for CCW management (see **Section 2.2**) defined the media of concern for the analysis. Human exposure scenarios included the following:

- Drinking of groundwater contaminated by leachate from CCW landfills and surface impoundments
- Consumption of fish by recreational fishers fishing in streams and lakes contaminated by CCW leachate through the groundwater-to-surface-water pathway

The CCW screening analysis used HBNs calculated for groundwater and surface water exposure. The CCW HBNs represent reasonable maximum exposure (RME) scenarios for an offsite receptor:

- Groundwater HBNs are protective for residential drinking water exposure from a domestic well immediately downgradient from a CCW landfill or surface impoundment
- Surface water HBNs are protective for fish caught (and consumed) by a recreational fisher from a river, lake, or stream adjacent to a CCW landfill or surface impoundment.

Key features and assumptions of the HBN calculations included the following:

- HBNs were calculated based on a target cancer risk of 10^{-5} or target HQ of 1

- The analysis considered exposures for three child receptor cohorts and one adult receptor cohort; exposure for these cohorts was assumed to start at ages 3, 8, 15, and 20, respectively

- Chemical properties (bio-uptake and bioaccumulation factors) were collected from best available literature values (see **Appendix J**)

- Human exposure factors (e.g., body weight, exposure duration, exposure frequency, consumption rates) were set at central tendency values.

Appendix I describes the methodology used to develop the CCW HBNs and provides the HBNs used in the screening analysis.

3.2.3.2 CSCL Calculations

The CCW ecological screening analysis paralleled the human health screening analysis and addressed two routes of exposure for ecological receptors: direct contact with contaminated media and ingestion of contaminated food items. Ecological exposure scenarios occurring near CCW landfills or surface impoundments and addressing these exposure routes included the following:

- Direct contact with surface water contaminated by CCW leachate through the groundwater-to-surface-water pathway

- Ingestion of aquatic organisms in streams and lakes contaminated by CCW leachate through the groundwater-to-surface-water pathway.

CSCLs for the contaminated media in each of these exposure scenarios were calculated as described in **Section 3.1.2** and **Appendix H** (the same CSCLs were used for both screening and the full-scale analysis). As with the HBNs, CSCLs were compared directly to concentrations of constituents found in CCW and CCW leachate and porewater, or to protective offsite media concentrations to estimate risk for screening.

3.2.4 Screening Results

The screening analysis conducted in 2002 (U.S. EPA, 2002a) was used in this risk assessment to help narrow the list of constituents to be addressed in the full scale analysis for the groundwater-to-drinking-water and groundwater-to-surface-water pathways. Detailed human and ecological screening results for these pathways are provided in **Appendix K**. The groundwater-to-drinking-water and groundwater-to-surface-water pathways (human fish consumption and ecological risks) did show risks above the screening criteria for several CCW constituents in the screening analysis. **Table 3-5** lists the 21 constituents that had 90th percentile screening analysis groundwater pathway risks greater than a cancer risk of 1 in 100,000 or a noncancer risk with an HQ greater than 1 for human health and 10 for ecological risk.[9]

[9] An HQ of 10 was used for screening ecological risks to account for conservatism of ecological benchmarks and exposure estimates used in the screening analysis (see Section 4.4.3.4).

Table 3-5. Screening Analysis Results: Selection and Prioritization of CCW Constituents for Further Analysis[a]

Constituent	Human Health – Drinking Water		Human Health – Surface Water[b]		Ecological Risk - Surface Water	
	LF HQ (*Cancer Risk*)	SI HQ (*Cancer Risk*)	LF HQ (*Cancer Risk*)	SI HQ (*Cancer Risk*)	LF HQ	SI HQ
Constituents Modeled in Full-scale Assessment						
Carcinogen						
Arsenic[c]	($1.4x10^{-3}$)	($1.8x10^{-2}$)	($2.2x10^{-4}$)	($1.7x10^{-5}$)	49	640
Noncarcinogens						
Boron	4.0	28	-	-	6,600	47,000
Cadmium	3.4	8.9	1.4	3.7	20	52
Lead	16	12	-	-	790	590
Selenium	1.2	2.4	4.7	9.5	35	71
Thallium	21	19	6.3	5.7	-	-
Aluminum	-	-	-	-	120	270
Antimony	22	5.5	-	-	-	-
Barium	-	-	-	-	400	75
Cobalt	-	11	-	-	-	270
Molybdenum	4.2	6.8	-	-	-	-
Nitrate/ Nitrite	- /1.2	60/1.2	-	-	-	-
Constituents Not Modeled in Full-scale Assessment[d]						
Noncarcinogens						
Chromium VI	2.3	4.2	-	-	18	33
Fluoride	1.8	5.2	-	-	-	-
Manganese	1	5.6	-	-	-	-
Vanadium	2.2	2.3	-	-	23	24
Beryllium	-	-	-	-	24	-
Copper	-	-	-	-	16	31
Nickel	-	1.3	-	-	-	14
Silver	-	-	-	-	110	14
Zinc	-	-	-	-	16	-

HQ = screening hazard quotient.
LF = landfill.
SI = surface impoundment.

[a] A dash in a cell indicates that the screening HQ was less than 1 (or 10 for ecological risk), so the risk did not exceed the screening criteria for the indicated pathway.

[b] Fish consumption pathway.

[c] Although arsenic can act as both a carcinogen and a noncarcinogen, the cancer risk exceeds the noncancer risk at any concentration, so the more protective cancer benchmark for human health was used throughout this assessment.

[d] These constituents were addressed using risk attenuation factors developed from full-scale results from modeled constituents (see **Section 4.1.5**).

Note that although mercury was originally addressed in both the 2002 screening and 2003 full-scale analyses, results were removed from the 2007 draft and this version of the risk assessment report because subsequent evaluation found that the very high proportion of mercury nondetects in the CCW constituent database, along with the use of one-half the detection limit for the nondetect measurements, led to the results being driven by the detection limit, rather than the actual (but unknown) levels in CCW leachate and porewater. Therefore, the results were not meaningful in terms of the actual risks mercury in CCW poses to human and ecological health. Similarly, a large number of nondetects (or a very small number of measurements) prevented accurate screening or full-scale analysis for antimony, thallium, and cobalt in surface impoundments. These uncertainties are discussed in **Section 4.4.3.1**.

Full-scale modeling was not conducted for all 21 constituents that had 90th percentile risks above the screening criteria for the groundwater pathways. Instead, those 21 constituents were ranked and divided into two groups to focus the full-scale analysis on the CCW constituents that were likely to pose relatively higher risks to human and ecological receptors. The ranking was based on the magnitude of the HQs and the number of HQs exceeding the screening criteria, and was used to select chemicals for full-scale modeling. Constituents with at least one human health HQ greater than 6 or with ecological HQs greater than 100 for both landfills and surface impoundments were modeled. Arsenic, with cancer risks greater than 1 in 1,000, exceeded the cancer risk criterion by a factor of 100 and was also modeled in the full-scale analysis. Constituents with no human health HQs greater than 6 and only one or no ecological HQs greater than 100 were not modeled, but were addressed in a separate analysis using results from the modeled constituents.

Table 3-5 shows the 21 constituents and which of these constituents exceeded the screening criteria and thus were modeled in the full-scale analysis. As shown, 12 constituents were subjected to the full-scale probabilistic risk assessment described in this document. Another 9 constituents exceeded the screening criteria and were addressed using risk factors developed from comparing the screening and full-scale results for the modeled constituents, as described in **Section 4.1.5** of this document.

3.3 Full-Scale Modeling Approach

This section describes the framework, general assumptions, and constraints for the full-scale probabilistic analysis. **Section 3.3.1** describes the temporal and spatial framework. **Section 3.3.2** describes the probabilistic framework, and **Section 3.3.3** describes how the assessment was implemented within the probabilistic framework.

3.3.1 Spatial and Temporal Framework

The spatial framework for the analysis was determined by the geographic distribution of CCW facilities modeled and by the site layout assumed as the conceptual site model for risk assessment. As described in **Section 2.1.2**, the geographic distribution of landfills and surface impoundments managing wastes onsite at coal-fired utility power plants was determined from the 177 sites in the 1995 EPRI survey of the onsite management of CCW (EPRI, 1997). The assessment assumes that these 177 sites and their locations were representative of the

approximately 300 coal-fired power plants identified by EIA data as having onsite waste management of conventional CCW and CCW codisposed with coal refuse throughout the United States. For FBC wastes, these 177 sites include only 3 FBC landfills. EPA was able to add 4 additional FBC landfill sites to better represent FBC waste management, for an overall total of 181 sites in this analysis.

The conceptual site layouts applied to each of the sites are described and pictured in **Section 2.2.2**. Two site layouts were used to define the relationship between a landfill or surface impoundment and (1) a drinking water well (for human risk via the groundwater-to-drinking-water pathway) and (2) a surface water body (for human and ecological risk via the groundwater-to-surface-water pathway). In each case, the receptor point (well or waterbody) was assumed to lie within the boundaries of the groundwater contaminant plume. The distance from the edge of the WMU to the well or waterbody was varied for each model run based on national distributions, with well distance taken from a national distribution for Subtitle D municipal landfills (U.S. EPA, 1988a) and distance to surface water taken from a set of measured distances for CCW landfills and surface impoundments developed for this assessment. **Appendix C** presents additional details on these distributions.

The temporal framework was mainly defined by the time of travel from the modeled WMU to the well or waterbody, which can be up to one mile away from the edge of the unit, and the exposure duration over which risks were calculated. The subsurface migration of some CCW constituents (e.g., lead) may be very slow; therefore, it may take a long time for the contaminant plume to reach the receptor well or nearest waterbody, and the maximum concentration may not occur until a very long time after the WMU ceases operations. This time delay may be on the order of thousands of years. To avoid excessive model run time while not missing significant risk at the receptor point, the groundwater model was run until the observed groundwater concentration of a contaminant at the receptor point dropped below a minimum concentration (10^{-16} mg/L) or until the model had been run for a time period of 10,000 years. The minimum concentration used for all fate and transport simulations (10^{-16} mg/L) was at least a million times below any risk- or health-based criteria.

For the groundwater-to-drinking-water pathway (human health risk), risks were calculated based on a maximum time-averaged concentration around the peak concentration at each receptor well. The exposure duration (which varies from 1 to 50 years)[10] was applied around the peak drinking well concentration to obtain the maximum time-averaged concentration.

For the groundwater-to-surface-water pathway, the groundwater model produced surface water contaminant loads (based on groundwater concentration and flow) for a stream that penetrates the aquifer. Because the surface water model is a steady-state model, there is no temporal component to it and the receptor is exposed to the same concentration over the entire exposure duration. For human health risk, the loadings from groundwater to surface water were averaged over the exposure duration, bracketing the time of the peak groundwater concentration.

[10] Distributions of exposure duration and other exposure variables were obtained from the *Exposure Factors Handbook* (U.S. EPA, 1997c,d,e) as described in **Section 3.8.2** and **Appendix F**.

The exposure duration for sensitive ecological receptors was generally a year or less; therefore, for ecological risk, a single peak annual average surface water concentration was used.

For all scenarios, if the groundwater model predicted that the maximum groundwater concentration had not yet occurred after 10,000 years, the actual groundwater concentration at 10,000 years was used in the exposure calculations instead of a maximum time-weighted average concentration around the peak.

3.3.2 Probabilistic Approach

The full-scale analysis evaluated risk in a probabilistic manner and was based on a Monte Carlo simulation that produced a distribution of exposures and risks. The general Monte Carlo approach is shown in **Figure 3-1**. The foundation of the Monte Carlo simulation was the source data derived from the EPRI survey. These were combined with data from the national CCW constituent database to conduct a Monte Carlo simulation of 10,000 iterations per waste type/WMU type/constituent combination.

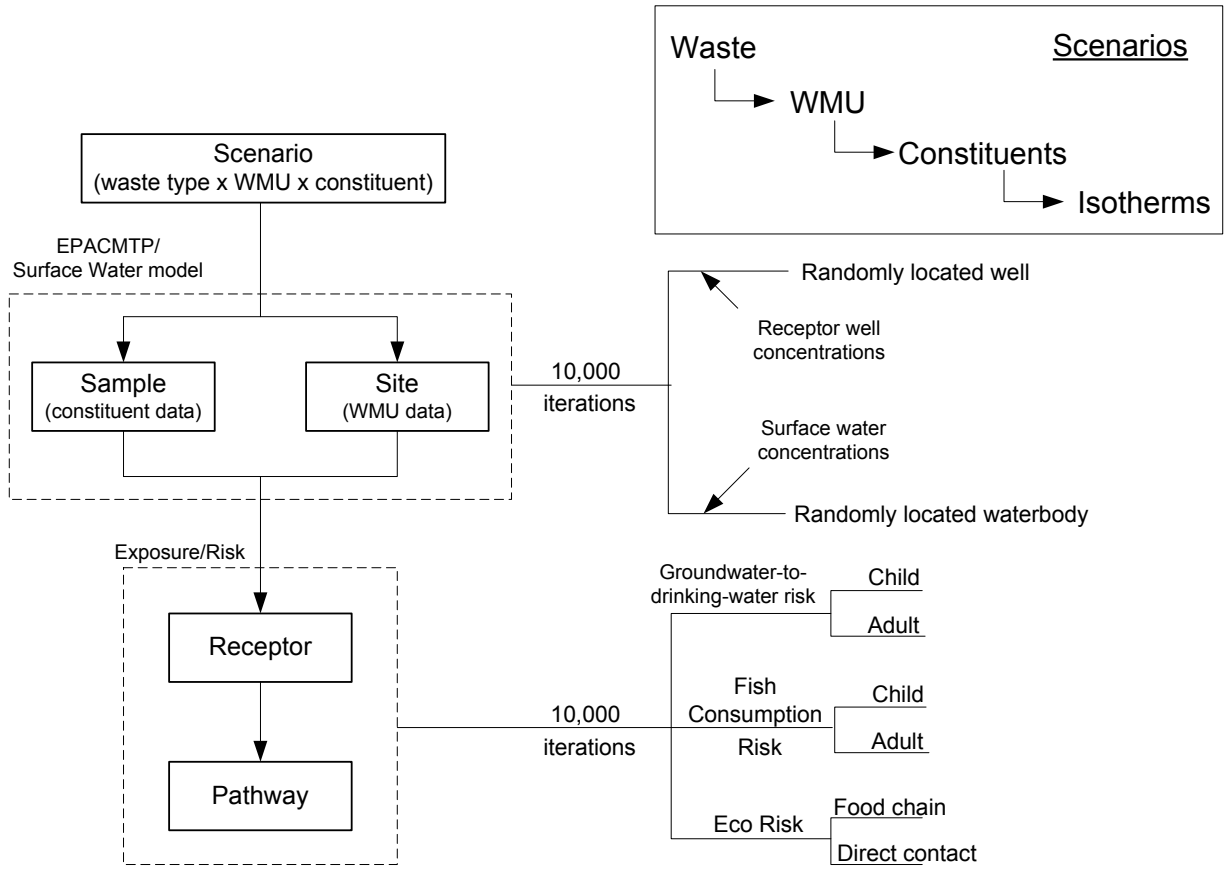


Figure 3-1. Overview of the Monte Carlo approach.

The detailed looping structure for the Monte Carlo analysis is shown in **Figure 3-2**. For each waste type/WMU combination, two separate loops were run. The first loop (shown with dashed lines in Figure 3-2) prepared a set of input files containing 10,000 sets of WMU and site data (as described in **Section 3.3.3**). The second loop (shown with solid lines in Figure 3-2) used

those input files to run 10,000 iterations of the source, fate and transport, exposure, and risk models for each constituent.

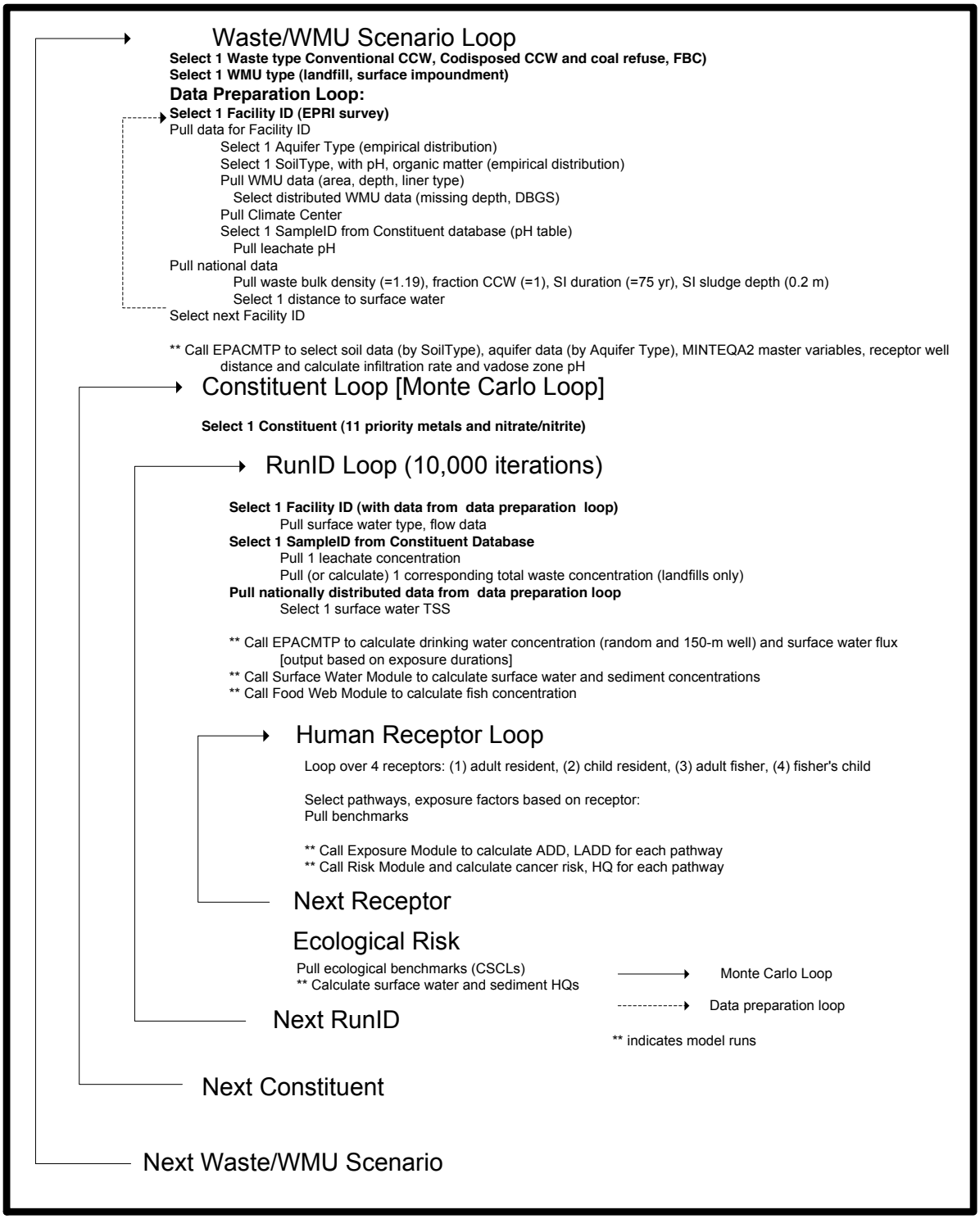


Figure 3-2. Monte Carlo looping structure.

3.3.3 Implementation of Probabilistic Approach

Table 3-6 lists the five waste disposal scenarios addressed in the full-scale analysis. FBC waste landfills were modeled and treated as a separate scenario in the analysis because of the limited number (7) of FBC landfill sites. Each waste disposal scenario modeled in the full-scale assessment included unlined, clay-lined, and composite-lined WMUs. Additional detail on these scenarios can be found in **Section 2.1** and **Appendix A**.

Table 3-6. CCW Waste Management Scenarios Modeled in Full-Scale Assessment

WMU Type	Waste Type
Conventional CCW and CCW Codisposed with Coal Refuse (main analysis)	
Landfill	Conventional CCW (fly ash, bottom ash, boiler slag, FGD sludge)
Landfill	Codisposed CCW and coal refuse
Surface impoundment	Conventional CCW
Surface impoundment	Codisposed CCW and coal refuse
FBC Waste (separate analysis)	
Landfill	FBC waste (fly ash and bottom [bed] ash)

To capture the national variation in waste management practices for the Monte Carlo analysis, an input database was created with approximately 10,000 iterations for each of the waste type/WMU combinations. This input database provided the source data for 10,000 iterations of the source modeling and the fate and transport modeling. **Figure 3-3** provides an overview of the process used to compile these data, which were organized into source data files. As shown in Figure 3-3, seven tasks, some parallel and some sequential, were required to construct these data files, one file for each waste management scenario.

Constructing the source data files for use in the probabilistic analysis involved first developing a 10,000-record data file for each waste type-WMU scenario. This was accomplished by selecting from the EPRI survey data the landfills and surface impoundments that manage each type of waste. Within a scenario, a list of the EPRI plants with that WMU type and waste type was repeated to produce around 10,000 records. For each record, site-based, regional, and national inputs were randomly selected from distributions developed to characterize the regional or national variability in these inputs. Each record in the source data files was identified by a model run identification number (RunID).

The EPRI survey provided most of the WMU data needed, including area, capacity, liner type, and waste type. Additional data were collected to characterize the height and depth below ground surface of typical CCW landfills and surface impoundments (see **Appendix B**).

The environmental setting in which waste disposal occurs was characterized based on the location of the 181 power plants used in the full-scale analysis. These locations were used to characterize climate, soils, aquifers, and surface water bodies at each site as follows (see **Appendix C** for details):

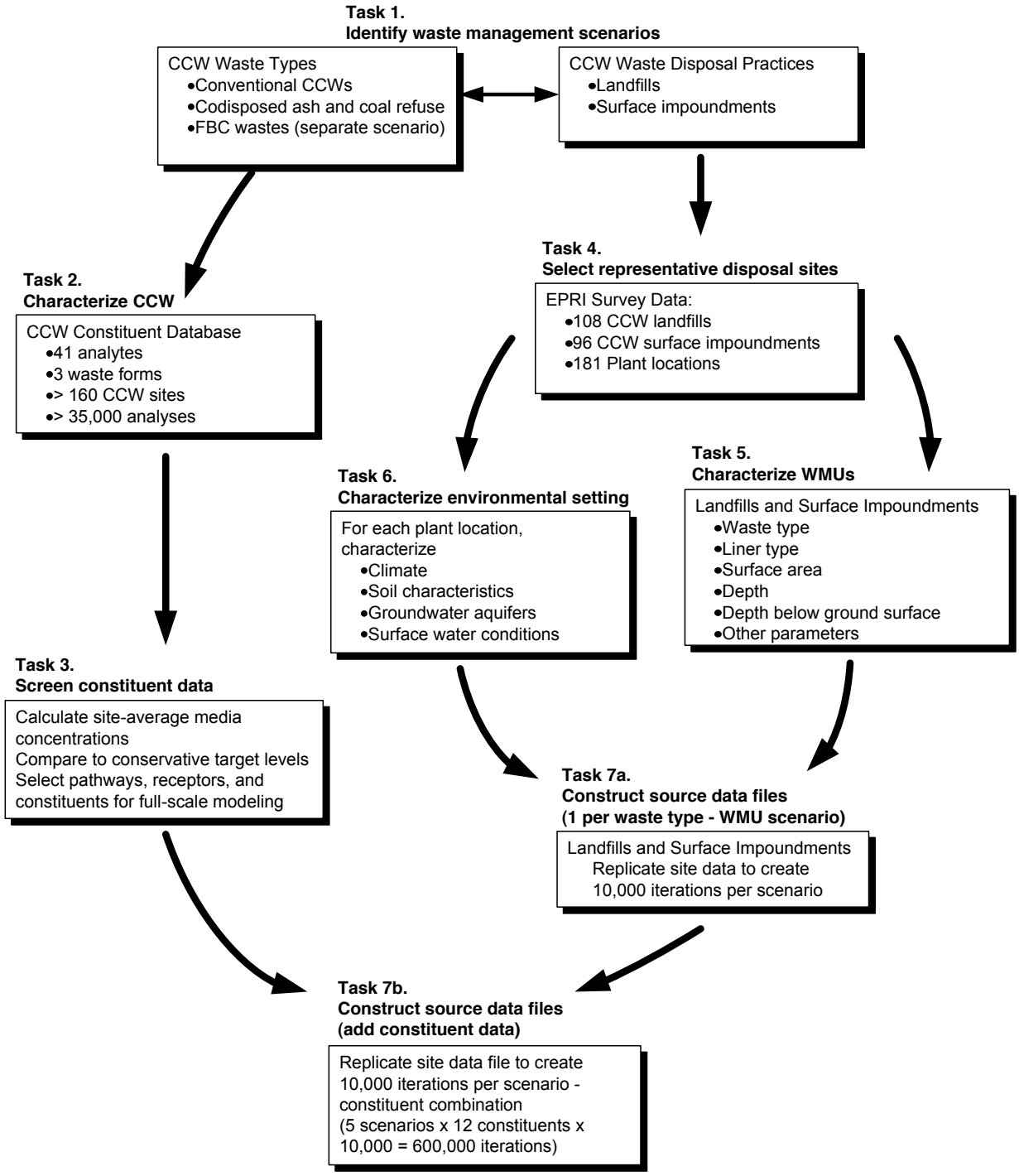


Figure 3-3. Process used to construct the Monte Carlo input database.

- Climatic data, including precipitation and infiltration rates, were collected by assigning each site to a nearby HELP climate center (see **Section 3.4.2.1** for a discussion of the HELP landfill infiltration model).

- Soil and aquifer type were collected within a 5-km radius of each site location to account for locational uncertainty for the WMUs (site location are often facility centroids or front gate locations).

- Surface water type and flows were collected using a geographic information system (GIS) to identify the nearest stream and by matching plants to the Permit Compliance System (PCS) database to get the stream segment for each plant's NPDES discharge point.

These site-based data were supplemented with regional data on surface water quality and with national distributions of receptor distances (i.e., distance to drinking water well and distance to nearest surface waterbody). **Appendix C** describes the site-based approach and data sources used for these site-specific, regional, and national-scale data collection efforts.

The five 10,000-record scenario-specific source data files were then combined with the CCW constituent data for each constituent in the appropriate waste type to develop the final source data files for each scenario. With 12 constituents modeled for most scenarios, this resulted in over 600,000 records in the final input data set.

3.4 Landfill Model

Releases from landfills were modeled using a landfill source-term model contained in EPACMTP. EPA has used EPACMTP and its predecessor models for almost 20 years to conduct groundwater risk assessments in support of regulations for land disposal of hazardous and nonhazardous wastes. In that context, EPACMTP has undergone numerous peer reviews, including multiple reviews by EPA's Science Advisory Board (SAB). Each of these reviews has supported and approved the use of this model for developing national regulations and guidance, including verification that the model and model code are scientifically sound and properly executed. Some of the more important reviews include

- A 1989 review by SAB of the component saturated zone (groundwater) model used in EPACMTP

- A 1993 review by EPA's Office of Research and Development (ORD) of EPACMTP for potential Hazardous Waste Identification Rule applications, which resulted in a number of improvements in the computational modules of EPACMTP

- A 1994 consultation with SAB on the use of EPACMTP for determination of dilution-attenuation factors for EPA's *Soil Screening Guidance*

- A 1994 review by expert modelers Dr. Fred Molz (Auburn University) and Mr. Chris Neville (SS Papadopoulos & Associates), who verified that the mathematical formulation of the model and the code verification testing are scientifically sound

- The peer-reviewed publication of EPACMTP in the *Journal of Contaminant Hydrology* (Kool et al., 1994)

- An in-depth review by SAB related to the use of EPACMTP in the proposed/draft 1995 Hazardous Waste Identification Rule (U.S. EPA, 1995)

- A 1999 peer review by leading modelers of the implementation of EPACMTP in EPA's multimedia, multiple exposure pathway, multiple receptor risk assessment (3MRA) model (U.S. EPA, 1999c)

- A 2003 SAB review of the 3MRA implementation of EPACMTP (SAB, 2004).

An overview and statement of assumptions for the landfill model is presented here, followed by a listing of inputs to the landfill source-term model and a brief discussion of the output generated by the model.

3.4.1 Conceptual Model

The landfill model treats a landfill as a permanent WMU with a rectangular footprint and a uniform depth (see **Figure 3-4**). If only the area is known (which is the case for the CCW landfills), the landfill source-term model assumes a square footprint. The model assumes that the landfill is filled with waste during the unit's operational life and that upon closure of the landfill, the waste is left in place and a final soil cover is installed.

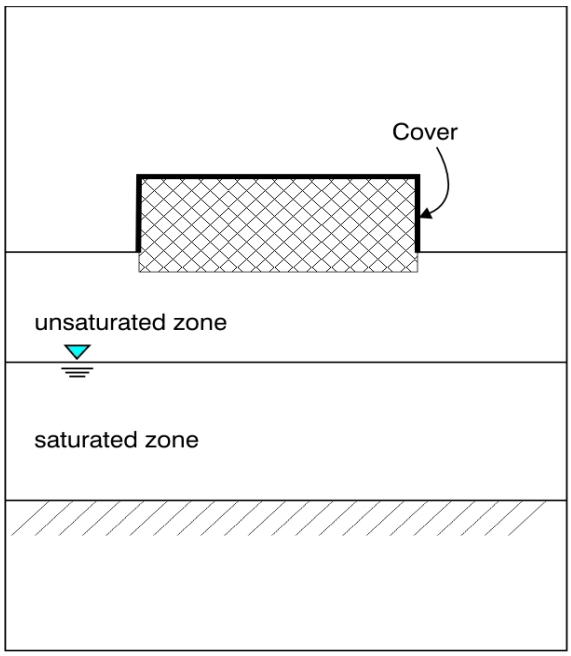


Figure 3-4. Conceptualization of a landfill in the landfill source-term model.

Three liner scenarios were modeled: a no-liner (unlined) scenario, a compacted clay liner, and a composite liner that combines a high-density polyethylene (HDPE) membrane with either geosynthetic or natural clays.

In the unlined scenario, waste is placed directly on local soils, either on grade or excavated to some design depth and without a leachate collection system. After the landfill has been filled to capacity, a 2-foot native soil cover (the minimum required by Subtitle D regulations) is installed and assumed to support vegetation.

In the clay liner scenario, waste is placed directly on a 3-foot compacted clay liner, which is installed on the local soils, either on grade or excavated to some design depth and without a leachate collection system. After the landfill has been filled to capacity, a 3-foot clay cover is installed and covered with 1 foot of loam to support vegetation and drainage. The hydraulic conductivity of both the liner and cover clays is assumed to be $1x10^{-7}$ cm/sec, the typical design specification for compacted clay liners (U.S. EPA, 1988c).

In the composite liner scenario, wastes are placed on a liner system that consists of a 60 mil HDPE membrane with either an underlying geosynthetic clay liner or a 3-foot compacted clay liner. A leachate collection system is also assumed to exist between the waste and the liner system. After the landfill has been filled to capacity, a 3-foot clay cover is assumed to be installed and covered with 1 foot of loam to support vegetation and drainage (U.S. EPA, 2002b).

As described in **Section 3.4.3** (and **Appendix B**), one of these three liner types was assigned to each CCW landfill or surface impoundment modeled based on the liner type data from the 1995 EPRI Survey (EPRI, 1997).

3.4.2 Modeling Approach and Assumptions

The starting point for the landfill source-term model simulation was the time when the landfill is closed (i.e., when the unit is filled with CCW).[11] As described in detail below, the full-scale analysis modeled contaminants leaching from CCW into precipitation infiltrating the landfill, which exits the landfills as leachate. Contaminant loss in leachate was taken into account at closure by subtracting the cumulative amount of contaminant mass loss that occurred during the unit's active life from the amount of contaminant mass present at the time of landfill closure. Loss calculations in the landfill source-term model continued after closure until the contaminant was depleted from the waste mass in the landfill. This is a conservative assumption, as some metal will not leach from the waste mass.

3.4.2.1 Infiltration and Leaching

The average rate at which water percolates through a landfill over time (the long-term infiltration rate) drives the leaching process in the landfill, which results from partitioning of the constituent from the waste into the infiltrating water. The methodology, assumptions, and data

[11] The simple landfill model used in this assessment cannot model a landfill as it is being filled prior to closure. Although leaching does occur during a landfill's operating life, risks from these releases are insignificant when compared to postclosure releases, given the long time it takes metal-bearing wastes to leach and reach peak concentrations in groundwater wells surrounding the landfill. For lined units, the liner system would be functional and governing during the active period of the landfill. For the unlined case, the landfill model assumes that the cap soils are no less permeable than the ambient soils around and under the landfill. So the majority of the cases would not have greater infiltration before closure. For these reasons, EPA does not believe that the additional risks from the preclosure period justify the additional complexity, data, and effort required to model an operating landfill.

used to determine infiltration rates for each CCW liner scenario were consistent with the approach used in EPA's Industrial D guidance, as described in Section 4.3 and Appendix A of the *EPACMTP Parameter/Data Background Document* (U.S. EPA, 2003a) and Section 4.2.2 of the *Industrial Waste Evaluation Model (IWEM) Technical Background Document* (U.S. EPA, 2002b). EPA developed the IWEM model as part of a guide for managing nonhazardous industrial wastes in landfills and surface impoundments (http://www.epa.gov/industrialwaste). To help ensure that it was technically sound, the model (including the liner scenarios and algorithms used in the CCW risk assessment) was developed with a large stakeholder working group, including representatives from industry. The model was also subjected to a peer review in 1999 (64 FR 54889–54890, October 8, 1999, *Peer Reviews Associated with the Guide for Industrial Waste Management*), and the model was updated and improved in response to those comments before its final release in 2003. That update included the addition of a more robust liner leakage database to support the existing algorithms for calculating infiltration rates through composite liner systems.

No-Liner (Unlined) Scenario. For the no-liner scenario, infiltration rates were selected from a database in EPACMTP that contains 306 infiltration rates already calculated using EPA's Hydrologic Evaluation of Landfill Performance (HELP) water balance model (Schroeder, et al., 1994a, b). HELP is a product of an interagency agreement between EPA and the U.S. Army Corps of Engineers Waterways Experiment Station, and was subjected to the Agency's peer and administrative review. All of the infiltration rates were calculated based on the single typical landfill design described in **Section 3.4.1**, with the only variables that changed between HELP simulations being the meteorological data associated with 102 nationwide climate centers and the type of cover soil applied at closure. Three cover soil categories representing coarse-grained soils, medium-grained soils, and fine-grained soils were used. The selection of an infiltration rate from the database depends on the type of cover soil selected for the landfill and the assignment of the landfill to a HELP climate center. The unlined HELP-derived infiltration rates are presented in U.S. EPA (2003a) by climate center. The assignment of HELP climate centers and soil categories to each CCW site modeled is described in **Appendix C**.

Clay Liner Scenario. The clay liner scenario is very similar to the unlined scenario in that previously calculated HELP infiltration rates for a single clay-lined, clay-capped landfill design were used. The scenario was based on a typical engineered compacted clay liner that is 3 feet thick with a design hydraulic conductivity of $1x10^{-7}$ cm/sec. The one difference from the unlined case is that the clay liner and cover control the rate of water percolation through the landfill and thus infiltration rate does not vary with cover soil (i.e., there is one clay liner infiltration rate per climate center). The clay liner HELP-derived infiltration rates are provided in U.S. EPA (2003a).

Composite Liner Scenario. Composite liner infiltration rates were compiled from monthly average leak detection system (LDS) flow rates for industrial landfill cells reported by TetraTech (2001). The liner configurations are consistent with the composite liner design assumptions presented in **Section 3.4.1** and are the same as those assumed for defaults in EPA's Industrial D landfill guidance (U.S. EPA, 2002b). The LDS flow rates were taken from 27 municipal landfill cells and used in the IWEM model (U.S. EPA, 2002b). As shown in **Table 3-7**, these LDS flow rates included 22 operating landfill cells and 5 closed landfill cells

located in eastern United States: 23 in the northeastern region, 1 in the mid-Atlantic region, and 3 in the southeastern region. Each of the landfill cells is underlain by a geomembrane/ geosynthetic clay liner which consists of a high-density polyethylene geomembrane of thickness between 1 and 1.5 mm, overlying a 6-mm composite geosynthetic clay layer consisting of two geotextile outer layers with a uniform core of bentonite clay to form a hydraulic barrier. Each liner system is underlain by an LDS.

Table 3-7. Leak Detection System Flow Rate Data Used to Develop Landfill Composite Liner Infiltration Rates

Cell ID	Status	Flow Rate (m/y)	Location
G228	Operating	2.1E-04	Mid-Atlantic
G232	Operating	4.0E-04	Northeast
G232	Closed	7.3E-05	Northeast
G233	Operating	0	Northeast
G233	Closed	0	Northeast
G234	Operating	7.3E-05	Northeast
G234	Closed	0	Northeast
G235	Operating	1.5E-04	Northeast
G235	Closed	3.7E-05	Northeast
G236	Operating	3.7E-05	Northeast
G236	Closed	0	Northeast
G237	Operating	7.3E-05	Northeast
G238	Operating	0	Northeast
G239	Operating	7.3E-05	Northeast
G240	Operating	0	Northeast
G241	Operating	0	Northeast
G242	Operating	0	Northeast
G243	Operating	0	Northeast
G244	Operating	0	Northeast
G245	Operating	0	Northeast
G246	Operating	0	Northeast
G247	Operating	0	Northeast
G248	Operating	0	Northeast
G249	Operating	7.3E-05	Northeast
G250	Operating	2.2E-04	Southeast
G251	Operating	0	Southeast
G252	Operating	0	Southeast

Source: U.S. EPA (2002a); original data from TetraTech (2001).

As described in U.S. EPA (2002b), only a subset of the TetraTech (2001) flow rates were used to develop the composite liner infiltration rates. LDS flow rates for geomembrane/ compacted clay composite-lined landfill cells were not used in the distribution because compacted clay liners (including composite geomembrane/compacted clay liners) can release water during consolidation and contribute an unknown amount of water to LDS flow, which makes it difficult to determine how much of the LDS flow is due to liner leakage versus clay

consolidation. Also, LDS flow rates from three geomembrane/geosynthetic clay lined-cells were not used. For one cell, postclosure flow rates were very high, and were more than twice as high as those recorded during the cell's operating period. Data were not used for two other cells because of inconsistencies with the data for the 27 landfill cells used to develop composite liner infiltration rates (U.S. EPA, 2002b). The composite liner infiltration rates were specified as an empirically distributed input to the landfill model (see U.S. EPA, 2003a).

3.4.2.2 Source Depletion and Mass Balance

For this assessment**,** the landfill source-term model represented releases from landfills as a finite source where the mass of a constituent in a landfill is finite and depleted over time by leaching. The landfill source-term model was set as a pulse source, where the leachate concentration is constant over a prescribed period of time and then goes to zero when the constituent is depleted from the landfill. A pulse source is appropriate for metals and other constituents whose sorption behavior is nonlinear. Because all but one (nitrate/nitrite) of the constituents addressed in the full-scale analysis were metals, releases from landfills were modeled as pulse sources.

For a pulse source, basic mass balance considerations require leaching from the landfill to stop when all of the constituent mass has leached from the landfill. For the constant concentration pulse source condition, the pulse duration is given by

$$TSOURC = \frac{CWASTE \times DEPTH \times FRACT \times CTDENS}{CZERO \times SINFIL} \quad (3\text{-}1)$$

where

TSOURC = Pulse duration (yr)
CWASTE = Constituent concentration in the waste (mg/kg)
DEPTH = Depth of landfill (m)
FRACT = Volume fraction of the landfill occupied by the waste (unitless)
CTDENS = Waste density (g/cm^3)
CZERO = Initial waste leachate concentration (mg/L)
SINFIL = Annual areal infiltration rate (m/yr).

The landfill source-term model uses the above relationship to determine the leaching duration. More details regarding the waste concentration and WMU parameters in Equation 3-1 are provided below and in **Appendices A** and **B**.

3.4.3 Landfill Model Input Parameters

Input parameters required by the landfill source-term model are discussed below. Additional details on how data for these inputs were collected for the CCW risk assessment are provided in **Appendix A** for leachate and waste concentrations and **Appendix B** for landfill dimensions and characteristics.

- **Landfill Area.** The model uses landfill area to determine the area over which infiltration rate occurs and, along with landfill depth and waste concentration, to calculate the total

contaminant mass in the landfill. CCW landfill area data were obtained from the EPRI comanagment survey (EPRI, 1997). The landfill was assumed to be square.

- **Landfill Depth.** Landfill depth is one of several parameters used by the landfill source-term model to calculate the contaminant mass in the landfill. For CCW landfills, average waste depth was estimated by dividing landfill capacity by landfill area. CCW landfill capacity data were taken from the EPRI comanagement survey (EPRI, 1997).

- **Depth Below Grade.** The depth of the bottom of the landfill below the surrounding ground surface is used, along with depth to groundwater, to determine the thickness of the unsaturated zone. For CCW landfills, depth below grade was determined from a national distribution based on available measurements from a number of CCW landfills (see **Appendix B**).

- **Waste Fraction.** The landfills were assumed to be CCW monofills, which corresponds to a waste fraction of 1.0.

- **Waste Density.** The average waste bulk density, as disposed, is used to convert waste volume to waste mass. The waste bulk density for all CCW waste types was assumed to be 1.19 g/cm^3 (U.S. EPA, 1998b).

- **Leachate Concentration.** The concentration of waste constituents in leachate was assumed to be constant until all of the contaminant mass initially present in the landfill has leached out, after which the leachate concentration was assumed to be zero. The constant value used for leachate concentration is from EPA's CCW Constituent Database, described in **Appendix A**.

- **Waste Concentration.** In the finite-source scenario modeled, the total waste concentration is used, along with the waste bulk density and landfill area and depth, to determine the total amount of a constituent available for leaching. Measured total CCW concentrations were paired with leachate concentrations, as described in **Appendix A** and provided in **Attachment A-2**.

- **Liner Type.** The type of liner is used to determine the infiltration/leaching scenario used to calculate leachate flux from the landfill. **Table 3-8** shows the crosswalk used to assign one of the three liner scenarios to each facility based on the liner data in the 1995 EPRI survey (EPRI, 1997). **Attachment B-2** to **Appendix B** provides these assignments, along with the original EPRI liner type, for each CCW landfill facility modeled. One uncertainty in these liner assumptions is how representative the EPRI survey data are of current conditions at coal combustion facilities.

Table 3-8. Crosswalk Between EPRI and CCW Source Model Liner Types

EPRI Liner Type	Model Liner Code	Description
Compacted ash	0	no liner
Compacted clay	1	clay

(continued)

Crosswalk Between EPRI and CCW Source Model Liner Types (continued)

EPRI Liner Type	Model Liner Code	Description
Composite clay/membrane	2	composite
Double	2	composite
Geosynthetic membrane	2	composite
None/natural soils	0	no liner

3.4.4 Model Outputs

For each year in the simulation, the landfill source-term model uses the average annual leachate concentration and infiltration rate to calculate a constituent flux through the bottom of the landfill. This time series was used as an input for the EPACMTP unsaturated zone model.

3.5 Surface Impoundment Model

Releases from surface impoundments were modeled using a surface impoundment source-term model contained in EPACMTP. An overview and statement of assumptions for the surface impoundment model are presented here, followed by a listing of inputs to the surface impoundment source-term model and a brief discussion of the output generated by the model. The primary differences between the treatment of landfills and surface impoundments are (1) the integration of the surface impoundment source term into the unsaturated flow solution, and (2) clean closure of the impoundment after the operating period is over.

3.5.1 Conceptual Model

The surface impoundment model treats a surface impoundment as a temporary WMU with a prescribed operational life. Unlike the landfill model, clean closure is assumed; that is, at the end of the unit's operational life, the model assumes that all wastes are removed and there is no further release of waste constituents to groundwater. Although this simplifying assumption limits the length of potential exposure, and is not consistent with the practice to close CCW surface impoundments with these wastes in place, the peak annual leachate concentrations on which the CCW risk results are based are not likely to be affected, because they are highest when the surface impoundment is in operation due to the higher hydraulic head in an operating impoundment, which drives leachate into the underlying soil with greater force than infiltration after the impoundment is covered and closed. This higher head results in a greater flux of contaminants to groundwater during the active life of the surface impoundment, especially in unlined units. These assumptions are discussed further in **Section 3.5.3.**

Following the unit's closure, the surface impoundment model assumes that the contaminated liquid and sediment in the surface impoundment are replaced by uncontaminated liquid and sediment with otherwise identical configurations and properties. The contaminants that have migrated to the unsaturated zone during operation continue to migrate towards the water table with the same infiltration rate as during operation. By continuing infiltration after the wastes are removed, the infiltration through the surface impoundment unit can be modeled as a

single steady-state flow regime until concentrations in groundwater are no longer affected by constituents released from the surface impoundment during its operation.

The EPACMTP surface impoundment model assumes a square footprint and a constant ponding depth during the impoundment's operational life (**Figure 3-5**). For an unlined impoundment, the model assumes that while the impoundment is in operation, a consolidated layer of sediment accumulates at the bottom of the impoundment. The leakage (infiltration) rate through the unlined impoundment is a function of the ponding depth in the impoundment and the thickness and effective permeability of the consolidated sediment layer at the bottom of the impoundment. The rate of leakage is constrained to ensure that there is not a physically unrealistic high rate of leakage, which would cause groundwater mounding beneath the unit to rise above the ground surface. Underlying the assumption of a constant ponding depth, the surface impoundment source-term model assumes that wastewater in the impoundment is continually replenished while the impoundment is in operation. It also assumes, from the beginning of the unit's operation, that the sediment is always in equilibrium with the wastewater (i.e., the presence of sediment does not alter the concentration of leachate). Accordingly, the surface impoundment source-term model also assumes that the leachate concentration is constant during the impoundment operational life and equal to the concentration in the porewater in the sediments at the bottom of the impoundment.

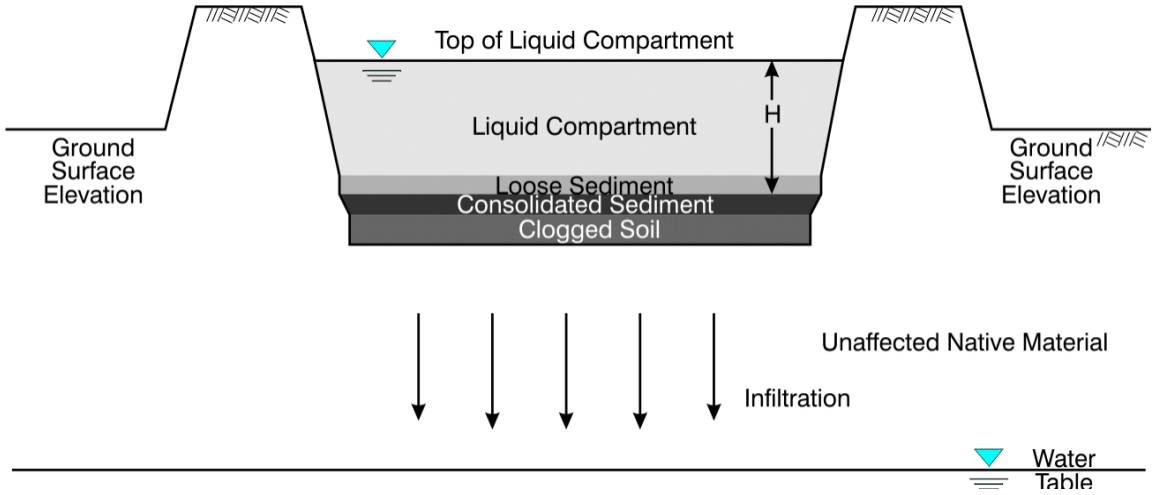


Figure 3-5. Schematic cross-section view of surface impoundment.

Three liner scenarios were modeled: a no-liner (unlined) scenario, a compacted clay liner, and a composite liner.

In the unlined scenario, wastewater is placed directly on local soils and the depth of water is constant over the entire life of the impoundment, pre- and post-closure. As described above, sediments accumulate and consolidate at the bottom of the impoundment and migrate into the underlying native soils, where they clog pore spaces and provide some barrier to flow. The surface impoundment model assumes that the thickness of the consolidated sediments is equal to one-half of the total sediment thickness, which is an input to the model. The sediment thickness was assumed to be 0.2 m for all simulations. The model also assumes that the thickness of the

clogged region of native soils is always 0.5 m and has a hydraulic conductivity 10 percent of that of the native soil underlying the impoundment.

In the clay liner scenario, wastewater is placed on a compacted clay liner, which is installed on the local soils. The assumptions for an unlined impoundment also apply to the compacted clay liner scenario, except that a compacted clay liner filters out the sediments that clog the native soils in the unlined case, so the effect of clogging the native materials is not included in the calculation of the infiltration rate. The thickness of the compacted clay liner was assumed to be 3 feet and the hydraulic conductivity was assumed to be $1x10^{-7}$ cm/sec (U.S. EPA, 1988c).

In the composite liner scenario, wastewater is placed on a synthetic membrane with an underlying geosynthetic or natural compacted clay liner with a hydraulic conductivity of $1x10^{-7}$ cm/sec. The membrane liner was assumed to have a number of pinhole leaks of uniform size (6 mm^2). The distribution of leak densities (expressed as number of leaks per hectare) was compiled from 26 leak density values reported in TetraTech (2001), the best available data on liner leaks. These leak densities are based on liners installed with formal construction quality assurance programs. The 26 sites with leak density data are mostly located outside the United States: 3 in Canada, 7 in France, 14 in the United Kingdom, and 2 in unknown locations; EPA assumed that these are representative of U.S. conditions. The WMUs at these sites (8 landfills, 4 surface impoundments, and 14 of unknown type) are underlain by a layer of geomembrane with a thickness varying from 1.14 mm to 3 mm. The majority of the geomembranes (23 of 26) are made from HDPE, and the remaining 3 are made from prefabricated bituminous geomembrane or polypropylene. One of the sites has a layer of compacted clay liner beneath the geomembrane; however, for 25 of the 26 sites, material types below the geomembrane layer are not reported. The empirical distribution used in the analysis can be found in IWEM (U.S. EPA, 2002b), along with a table showing details about the 26 liners used to develop the distribution.

3.5.2 Modeling Approach and Assumptions

Figure 3-5 illustrates a compartmentalized surface impoundment with stratified sediment. Shown in the figure are the liquid compartment, the sediment compartment (with loose and consolidated sediments), and the unsaturated zone (with clogged and unaffected native materials). The model assumes that all sediment layer thicknesses remain unchanged throughout the life of the unit.

The EPACMTP surface impoundment model uses the unsaturated zone flow model to calculate the infiltration rate out of the bottom of the impoundment. This model is designed to simulate steady-state downward flow through an unsaturated (vadose) zone consisting of one or more soil layers. Steady-state means that the rate of flow does not change with time. In the case of flow out of an unlined surface impoundment, the model simulates flow through a system consisting of three layers: a consolidated sediment layer, a clogged soil layer, and a native soil layer.

The native unsaturated soil extends downward to the water table. The steady-state infiltration rate out of the surface impoundment is driven by the head gradient between the water ponded in the impoundment and the head at the water table. The pressure head at the top of the

consolidated sediment layer is equal to the water depth in the impoundment plus the thickness of the unconsolidated sediment.

The *EPACMTP Technical Background Document* (U.S. EPA, 2003c) describes the algorithms used in this model to calculate the infiltration rate from surface impoundment units, and discusses in detail the maximum allowable infiltration rate based on the groundwater mounding condition. This information is summarized here.

The EPACMTP surface impoundment source-term model calculates infiltration through the accumulated sediment at the bottom of an impoundment, accounting for clogging of the native soil materials underlying the impoundment, liner conditions, and mounding due to infiltration. The modeled infiltration is governed by the depth of liquid in the impoundment and the following limiting factors:

- **Effective hydraulic conductivity and thickness of the consolidated 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. The consolidation process reduces the hydraulic conductivity of the sediment layer, and the layer of consolidated sediment will act as a restricting layer for flow out of the impoundment. By contrast, the layer of loose, unconsolidated sediment that overlies the consolidated sediment layer is assumed not to restrict the flow rate out of the unit, so it is not explicitly considered in the surface impoundment flow model.

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

- **Effective hydraulic conductivity and thickness of a clay liner.** When the surface impoundment is underlain by a compacted clay liner, the rate of infiltration is also determined by simulating flow through a three-layer system, substituting the characteristics of the clay liner for those of the clogged soil layer.

- **Leak rate of a composite liner.** For cases where the surface impoundment is underlain by a composite liner (a geomembrane underlain by a low permeability liner such as a compacted clay liner or a geosynthetic clay liner), the surface impoundment source-term model uses a modified equation of Bonaparte et al. (1989) to calculate the infiltration rate. The equation uses, among other inputs, the head generated by the water and unconsolidated sediments in the unit, a leak density selected from an empirical distribution derived from a TetraTech (2001) study of liner leakage, a uniform leak size of 6 mm^2, and an assumed hydraulic conductivity of $1x10^{-7}$ cm/sec for the 3 feet of underlying compacted clay material.

- **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. The model accounts for groundwater mounding when calculating the infiltration rate from the

surface impoundment unit and, if necessary, constrains the value to ensure that the groundwater mound does not rise to the bottom of the surface impoundment unit.

3.5.3 Surface Impoundment Model Input Parameters

Input parameters required by the surface impoundment source-term model are discussed below. Additional details on how data for these inputs were collected for the CCW risk assessment are provided in **Appendix A** for waste concentrations and **Appendix B** for surface impoundment dimensions and characteristics.

- **Surface Impoundment Area.** The model uses surface impoundment area to determine the area over which infiltration occurs. CCW surface impoundment area data were obtained from the EPRI comanagement survey (EPRI, 1997). The impoundment was assumed to be square.

- **Areal Infiltration Rate.** The surface impoundment leachate infiltration rate (or flux) is computed internally by the surface impoundment source-term model, as described in **Section 3.5.2**.

- **Depth Below Grade.** The depth of the bottom of the impoundment below the surrounding ground surface is used, along with depth to groundwater, to determine the thickness of the unsaturated zone beneath the impoundment. For CCW impoundments, depth below grade was sampled from an empirical distribution based on available measurements from a number of CCW surface impoundments (see **Appendix B**).

- **Operating Depth.** The operating (or ponding) depth is the long-term average depth of wastewater and sediment in the impoundment, measured from the base of the impoundment. For CCW surface impoundments, depth was estimated by dividing impoundment capacity by impoundment area. CCW impoundment capacity data were taken from the EPRI comanagement survey (EPRI, 1997).

- **Total Thickness of Sediment.** By default, EPACMTP models unlined surface impoundments with a layer of "sludge" or sediment above the base of the unit. The sediment layer is divided into two sublayers: an upper, loose sediment sublayer and a lower, consolidated sediment sublayer. The consolidated sediment has a relatively low hydraulic conductivity and acts to impede flow. The calculated infiltration rate is inversely related to the thickness of the consolidated sediment sublayer. A thinner consolidated sediment layer will result in a higher infiltration rate and a greater rate of constituent loss from the impoundment. The surface impoundment source-term model uses the total sediment thickness as an input parameter and assumes that it consists of equal thicknesses of loose and consolidated material. Because data were not available on CCW sediment layer thicknesses, the CCW risk assessment used the Tier 1 IWEM model assumption: a total (unconsolidated plus consolidated) sediment layer thickness of 0.2 meters (U.S. EPA, 2002b). It is not known how representative this assumption is with respect to unlined CCW surface impoundments, but it is reasonable to assume that a sediment layer would accumulate and restrict flow from the bottom of a CCW impoundment.

- **Distance to the Nearest Surface Water Body.** The distance to the nearest waterbody is used to determine the location of a fully penetrating surface waterbody at which groundwater mass and water fluxes will be calculated and reported. The distance to the nearest surface waterbody is also used as a surrogate for the distance to the nearest point at which the water table elevation is kept at a fixed value. That distance is used to calculate the estimated height of groundwater mounding underneath the impoundment to ensure that excessively high infiltration rates, which may be calculated for deep, unlined impoundments, do not occur. If necessary, the model reduces the infiltration rate to ensure the predicted water table does not rise above the ground surface. For the CCW sites, distance to surface water was sampled from an empirical distribution developed from aerial photo measurements at 59 coal-fired power plants with onsite landfills or surface impoundments (**Appendix C**).

- **Leachate Concentration.** The annual average leachate concentration is modeled as a constant concentration pulse with a defined duration. For a particular model run, the leachate concentration was assumed to be constant during the operation of the unit; there is no reduction in leachate concentration until the impoundment ceases operation. Leachate concentrations for CCW impoundments were obtained by waste type from surface impoundment porewater data from EPA's CCW Constituent Database, as described in **Appendix A**.

- **Source Leaching Duration.** For surface impoundments, the addition and removal of waste during the operational life period are more or less balanced, without significant net accumulation of waste. In the finite-source implementation used for CCW surface impoundments, the duration of the leaching period is assumed to be the same as the operational life of the surface impoundment. Based on industry data (see **Appendix B**) for CCW surface impoundments, EPA used a high-end (90th percentile) fixed surface impoundment operating life of 75 years. A high-end value was appropriate because CCW surface impoundments are typically closed with waste in place, while the surface impoundment source-term model assumes clean closure (waste removed). In addition, operating life is not a particularly sensitive parameter in this analysis: the difference between the 50th percentile value (40 years) and the 90th percentile value used (75 years) is less than a factor of two.

- **Liner Type, Thickness, Hydraulic Conductivity, and Leak Density.** The type of liner is used to calculate leachate flux from the impoundment. To assign one of the three liner scenarios to each facility in the EPRI survey (EPRI, 1997), EPA used the same crosswalk as for landfills (see Table 3-7). Attachment B-2 to **Appendix B** provides these assignments, along with the original EPRI liner type, for each CCW surface impoundment modeled.

As with IWEM (U.S. EPA, 2002b), clay liners were assumed to be 3 feet thick and to have a constant hydraulic conductivity of 10^{-7} cm/s, reflecting typical design specifications for clay liners. For composite liners, infiltration was assumed to result from defects (pin holes) in the geomembrane. The pin holes were assumed to be circular and uniformly sized (6 mm^2). The leak density was defined as the average number of circular pin holes per square meter and was

obtained from a study of industrial surface impoundment membrane liner leak rates by Tetra Tech (2001).

3.5.4 Surface Impoundment Model Outputs

For each year in the simulation, the surface impoundment source-term model uses the average annual leachate concentration and calculates an infiltration rate to estimate the constituent flux through the bottom of the impoundment. This time series was used as an input to the EPACMTP unsaturated zone model.

3.6 Groundwater Model

This section describes the methodology and the models that were used to predict the fate and transport of chemical constituents in soil and groundwater to determine impacts on drinking water wells and surface water that is connected to groundwater. The surface water model used to address the groundwater-to-surface water pathways is described in **Section 3.7**.

3.6.1 Conceptual Model

The groundwater pathway was modeled to determine the receptor well concentrations and contaminant flux to surface water resulting from the release of waste constituents from a WMU. The release of a constituent occurs when liquid percolating through the WMU becomes leachate as it infiltrates from the bottom of the WMU into the subsurface. For landfills, the liquid percolating through the landfill is from water in the waste and precipitation. For surface impoundments, the percolating liquid is primarily the wastewater managed in the impoundments.

Waste constituents dissolved in the leachate are transported through the unsaturated zone (the soil layer under the WMU) to the underlying saturated zone (i.e., groundwater). Once in the groundwater, contaminants are transported downgradient to a hypothetical receptor well or waterbody. For this analysis, the groundwater concentration was evaluated for two receptor locations, each at a specified distance from the downgradient edge of the WMU:

- The intake point of a hypothetical residential drinking water well (the receptor well), which was used for the residential drinking water pathway
- A nearby river, stream, or lake, which is modeled as a fully penetrating surface waterbody and was used for the fish ingestion and ecological pathways.

Figure 3-6 shows the conceptual model of the groundwater fate and transport of contaminant releases from a WMU to a downgradient receptor well.

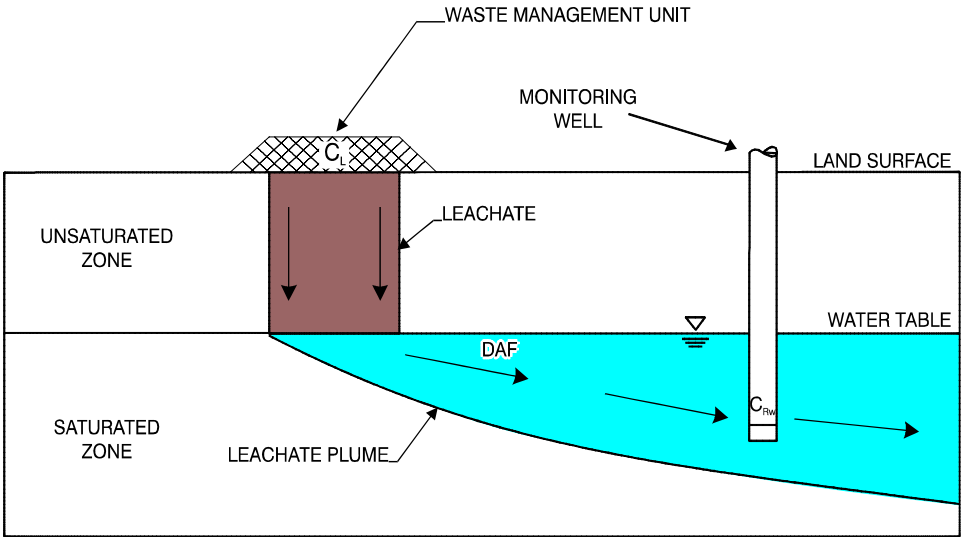


Figure 3-6. Conceptual model of the groundwater modeling scenario.

3.6.2 Modeling Approach and Assumptions

The transport of leachate from the WMU through the unsaturated and saturated zones was modeled using EPACMTP (U.S. EPA, 1996a, 1997a, 2003a,c,d). EPACMTP is a composite model consisting of two coupled modules: (1) a one-dimensional module that simulates infiltration and dissolved contaminant transport through unsaturated soils, and (2) a 3-dimensional saturated zone flow and transport module to model groundwater fate and transport. EPACMTP has been used by EPA to make regulatory decisions for wastes managed in land disposal units (including landfills and surface impoundments) for a number of solid waste and hazardous waste regulatory efforts, and as noted earlier, has undergone extensive peer review. EPACMTP simulates the concentration arriving at a specified receptor location (such as a well or stream).

The primary subsurface transport mechanisms modeled by EPACMTP are (1) downward (1-dimensional) movement along with infiltrating water flow in the unsaturated zone soils and (2) movement and dispersion along with ambient groundwater flow in the saturated zone. EPACMTP models soils and aquifer as uniform porous media and does not account for preferential pathways such as fractures and macropores or for facilitated transport, which may affect migration of strongly sorbing constituents such as metals.

In the unsaturated zone, flow is gravity driven and prevails in the downward direction. Therefore, the flow is modeled in the unsaturated zone as one-dimensional in the vertical direction. The model also assumes that transverse (sideways) dispersion (from both mechanical and molecular diffusion processes) is negligible in the unsaturated zone because the scale of lateral migration due to transverse dispersion is negligible compared with the size of the WMUs. This assumption is also environmentally protective because it allows the leading front of the contaminant plume to arrive at the water table with greater peak concentration in the case of a finite source.

In the saturated zone, the EPACMTP model assumes that movement of chemicals is driven primarily by ambient groundwater flow, which in turn is controlled by a regional hydraulic gradient and hydraulic conductivity in the aquifer formation. The model does take into account the effects of infiltration through the WMU, as well as regional recharge into the aquifer around the WMU. Infiltration through the WMU increases the groundwater flow in all directions under and near the WMU and may result in groundwater mounding. This 3-dimensional flow pattern enhances the horizontal and vertical spreading of the contaminant plume. The effect of recharge (outside the WMU) is to cause a downward (vertical) movement of the contaminant plume as it travels along groundwater flow direction. In addition to advective movement with the groundwater flow, the model simulates mixing of contaminants with groundwater due to hydrodynamic dispersion, which acts along the groundwater flow direction, as well as vertically and in the horizontal transverse direction.

To model sorption of CCW constituents in the unsaturated zone, soil-water partitioning coefficients (K_d values) for metal constituents were selected from nonlinear sorption isotherms generated from the equilibrium geochemical speciation model MINTEQA2 (U.S. EPA, 2001a). Chemicals with low K_d values will have low retardation factors, which means that they will move at nearly the same velocity as the groundwater. Chemicals with high K_d values will have high retardation factors and may move many times slower than groundwater. As described in **Appendix D**, CCW-specific partition coefficients were developed with MINTEQA2 considering CCW leachate chemistry, including the highly alkaline chemistries that are characteristic of some CCWs. Although a complete listing of all K_d values available in the MINTEQA2 isotherms used in these analyses would not be practicable, Table D-1 presents a sampling of the K_d values used.

MINTEQA2 is a product of ORD, and like EPACMTP, has a long history of peer- and SAB-review during its development, use, and continued improvement for regulatory support over the past two decades. These reviews largely focused on the use of MINTEQA2 to generate sorption isotherms for metals for EPACMTP, which is how it was used in the CCW risk assessment. Two of the more recent peer reviews include one for application within the 3MRA model (U.S. EPA, 1999d) and a review of its use and application to RCRA rulemaking and guidance support, including revisions made to the model to support IWEM and the CCW rulemaking efforts (U.S. EPA, 2003f). In the latter review, three experts found that the revisions made to the MINTEQA2 model were appropriate, but also suggested further improvements in how the model addresses environments with highly alkaline leachate (such as CCW sites). As explained in **Appendix D**, these comments were addressed in this application of MINTEQA2 to CCW waste transport by the development of sorption isotherms that are specific to geochemical conditions encountered in CCW landfills and surface impoundments.

3.6.3 Model Inputs and Receptor Locations

EPACMTP requires information about soil and aquifer properties as model inputs. For soils, EPACMTP uses soil texture to generate consistent hydrological properties for the unsaturated zone model, and soil pH and organic matter to select appropriate sorption coefficients to model contaminant sorption in the soil. As described in **Appendix C, Attachment C-2**, site-specific soil texture, pH, and organic carbon data were collected around each site from the STATSGO soils database. Similarly, the hydrogeological setting around each

WMU was used to select appropriate aquifer conditions from EPACMTP's Hydrogeologic Database (HGDB; see **Appendix C**).

Recharge is water percolating through the soil to the aquifer outside the footprint of the WMU. The recharge rate is determined by precipitation and soil texture. For the CCW landfills and surface impoundments, recharge rates were selected by soil texture and meteorological station assignment from a database of HELP model–derived recharge rates for climate stations across the country that is included in the EPACMTP input files. Further details about how these rates were determined and other options for determining recharge rates outside of the EPACMTP model can be found in the *EPACMTP Parameters/Data Background Document* (U.S. EPA, 2003a).

One of the most important inputs for EPACMTP is receptor location, which for this risk assessment includes residential drinking water wells and surface water bodies. **Figure 3-7** shows a schematic of how residential well drinking water intakes were defined in terms of their radial downgradient distance from the WMU and the angle off the contaminant plume centerline. The shaded areas in Figure 3-7 represent the horizontal extent of the contaminant plume.

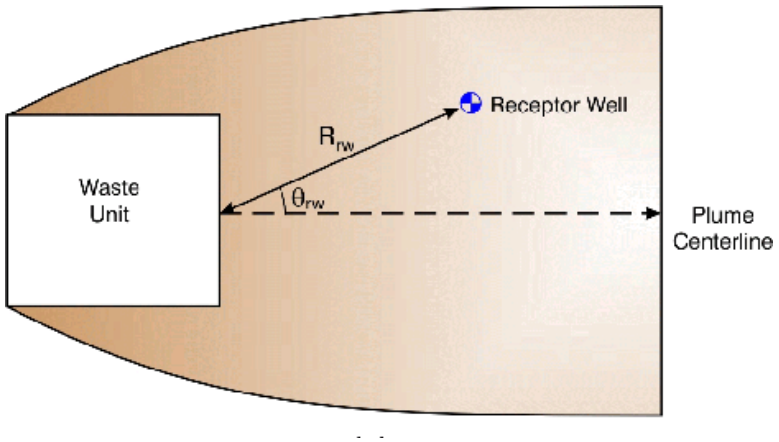


Figure 3-7. Schematic plan view showing idealized maximum lateral contaminant plume extent and receptor well location.

In this analysis, receptor wells were located randomly within the contaminant plume, as follows:

- Because residential well distance data are not available for CCW WMUs, EPA based the radial downgradient distance on a nationwide distribution of the nearest downgradient residential or municipal wells from a survey of Subtitle D municipal solid waste landfills (U.S. EPA, 1988a; see **Appendix C**). The maximum radial distance in this survey was 1 mile. EPA believes that this distribution is protective of CCW WMUs. A well distance, (R_{rw} in Figure 3-7) was randomly selected from this distribution.

- The angle off the contaminant plume centerline (θ_{rw} in Figure 3-7) was selected from a uniform distribution ranging from 0 to 90 degrees.

- The receptor well was located based on R_{rw} and θ_{rw} as shown in Figure 3-7.

- The maximum lateral extent of a groundwater plume, based on lateral dispersion, was calculated using the dimensions of the WMU sampled for that simulation, a sampled value for lateral dispersivity in the groundwater, and the downgradient distance to the receptor well.

- If the receptor well was located *inside* the idealized maximum plume extent, the shaded portion in Figure 3-7 (the distance from the well to the centerline was less than the lateral extent of the calculated in the previous step), the well location was used for that simulation. Otherwise, new values for R_{rw} and θ_{rw} were sampled and the process repeated for the same WMU. The depth of the well intake point was based on a uniform distribution with limits of 0 (i.e., well at the water table) to 10 meters (or the total saturated aquifer thickness if the aquifer is less than 10 meters thick).

The location of the surface waterbody intercepting groundwater flow was specified for each flow and transport simulation. The waterbody was constrained to lie across the contaminant plume centerline and perpendicular to the groundwater flow direction. The waterbody is assumed to fully penetrate the aquifer thickness. Downgradient distance to the surface waterbody was determined from an empirical distribution of distances measured for CCW landfills and surface impoundments (see **Appendix C**), which was randomly sampled to develop the distances used in EPACMTP to calculate groundwater concentrations at those distances in the Monte Carlo analysis.

3.6.4 Groundwater Model Outputs

The output of EPACMTP is a prediction of the contaminant concentration arriving at a downgradient groundwater receptor location (either a well or a surface water body). Because a finite-source scenario was used, the concentration is time-dependent. A maximum time-averaged concentration was calculated for each constituent across the exposure duration selected in each Monte Carlo iteration.

3.7 Surface Water Models

For the groundwater-to-surface-water pathway, chemical contaminants leach out of WMUs and into groundwater, and this contaminated groundwater then discharges into a surface waterbody through groundwater discharge. Once in the waterbody, the continued fate and transport of the contaminants is modeled with a surface water model, which uniformly mixes the contaminants in a single stream segment. Surface water flows in and out of the stream segment. Surface water flowing into the stream segment is assumed to have zero constituent concentration, and surface water flowing out has nonzero constituent concentrations due to the groundwater contamination. The primary simplifying assumptions in EPACMTP are as follows: (1) the groundwater–surface water interface is assumed to be perpendicular to the regional groundwater flow direction; (2) the interface is infinite in its lateral extent so as to intercept the entire width of the dissolved contaminant plume; and (3) the intercepting surface water body fully penetrates the saturated region of the subsurface. Therefore, all of the mass in the contaminated groundwater is available to be transferred to the surface water model. If stream

flow is greater than the available groundwater flow, then all of the mass available in the groundwater is assumed to be transferred to the surface waterbody. It is important to note that while a mass transfer is assumed to take place between the two systems, mass is not actually removed from the groundwater—it is still available to be observed at a receptor well placed beyond the groundwater-surface water interface.

To ensure that an unrealistic transfer of mass from the contaminated groundwater into the surface waterbody does not occur, the available groundwater flow is compared to the stream flow. If the groundwater flow exceeds the stream flow, all of the stream flow is assumed to be from groundwater discharge and the total concentration in the stream is equal to the groundwater concentration.

The waterbody considered in the CCW risk assessment is a river, stream, or lake located downgradient of the WMU. As described in **Appendix C**, the flow characteristics and dimensions for this waterbody were determined by site-specific stream flow data, the width of the groundwater contaminant plume as it intersects the waterbody, and established relationships between flow and stream depth. The stream segment modeled in this assessment was assumed to be homogeneously mixed.

Simple equilibrium partitioning models were used to estimate contaminant concentrations in the water column, suspended and bed sediments (see **Section 3.7.1**), and aquatic organisms (see **Section 3.7.2**). Special modeling provisions for aluminum are described in **Section 3.7.3**.

3.7.1 Equilibrium Partitioning Model

The primary surface water model used to estimate groundwater impacts on waterbodies is a simple steady-state equilibrium-partitioning model adapted from models in EPA's Indirect Exposure Methodology (IEM; U.S. EPA, 1998c) and Human Health Risk Assessment Protocol (HHRAP; U.S. EPA, 1998d). This model is based on the concept that dissolved and sorbed concentrations can be related through equilibrium partitioning coefficients. This model was used for all constituents except aluminum, which was modeled based on a solubility approach (see **Section 3.7.3**). Although these models have not been specifically peer reviewed in this application, they have been subject to the Agency's peer review process as part of the development of the IEM and HHRAP.

The model partitions the total mass of chemical contaminant in the waterbody into four compartments:

- Constituents dissolved in the water column
- Constituents sorbed onto suspended solids
- Constituents sorbed onto sediment particles at the bottom of the waterbody
- Constituents dissolved in porewater in the sediment layer.

Table 3-9 provides the partitioning coefficients used by the surface water model to estimate contaminant partitioning between water and suspended solids in the water column and between

sediment and porewater in the sediment layer. These distributions were derived from published empirical data as described in U.S. EPA (1999b).

Table 3-9. Sediment/Water Partition Coefficients: Empirical Distributions[a]

Chemical	Distribution Type	Minimum	Mean	Maximum	SD
Aluminum	not used				
Antimony	log normal	0.6	3.6	4.8	1.8
Arsenic	log normal	1.6	2.4	4.3	0.7
Barium	log normal	0.9	2.5	3.2	0.8
Boron	log normal	−0.5	0.8	1.4	0.5
Cadmium	log normal	0.5	3.3	7.3	1.8
Cobalt	log normal	2.2	3.9	5.3	0.8
Lead	log normal	2.0	4.6	7.0	1.9
Molybdenum	log normal	1.3	2.2	3.2	0.9
Selenium IV	log normal	1.0	3.6	4.0	1.2
Selenium VI	log normal	−1.4	0.6	3.0	1.2
Thallium	log normal	−0.5	1.3	3.5	1.1
Total Nitrate Nitrogen	constant	0	0	0	0

Source: U.S. EPA (1999b).
SD = standard deviation.
[a] All values are log values.

Following calculation of the constituent loading and loss rates, the surface water model estimates steady-state, equilibrium waterbody contaminant concentrations in each compartment using equations presented in Attachment E-1 to **Appendix E**. For evaluating risks to human health from fish consumption, the model calculates waterbody concentrations using groundwater loadings that are explicitly averaged over the exposure period for the each human receptor (i.e., adult and child fishers). These average waterbody concentrations are then used to calculate fish concentrations as described in **Section 3.7.2**. Ecological risks were based on waterbody concentrations calculated using the peak annual groundwater loading value from EPACMTP. The equilibrium–partitioning model, as implemented, is conservative because there are no loss mechanisms (e.g., burial) for any of the constituents.

3.7.2 Aquatic Food Web Model

An aquatic food web model was used to estimate the concentration of CCW constituents that accumulate in fish. This risk assessment assumed that fish are a food source for a recreational fisher. Trophic level three (TL3) and four (TL4) fish[12] were considered in this analysis because most of the fish that humans eat are T4 fish (e.g., salmon, trout, walleye, bass) and medium to large T3 fish (e.g., carp, smelt, perch, catfish, sucker, bullhead, sauger). The aquatic food web model has been peer reviewed as part of the 3MRA model development effort (see http://www.epa.gov/osw/hazard/wastetypes/wasteid/hwirwste/peer03/aquatic/aqtfoods.pdf).

[12] TL3 fish are those that consume invertebrates and plankton; TL4 fish are those that consume other fish.

The aquatic food web model calculates the concentration in fish from the concentration calculated for the waterbody downgradient from the CCW disposal site. The contaminants in the water column consist of dissolved constituents and constituents sorbed to suspended solids. For all constituents, the contaminant concentrations in fish were calculated from the total waterbody concentration (i.e., dissolved plus sorbed to suspended solids) using BCFs, which are presented in **Table 3-10**. The equations used to model fish tissue concentrations are provided in Attachment E-2 to **Appendix E**.

Table 3-10. Bioconcentration Factors for Fish

CAS	Chemical	TL3 Value	TL4 Value	Units	Reference
7429-90-5	Aluminum	ND	ND	L/kg	
7440-36-0	Antimony	0	0	L/kg	Barrows et al. (1980)
22569-72-8	Arsenic (III)	4.0E+00	4.0E+00	L/kg	Barrows et al. (1980)
15584-04-0	Arsenic (V)	4.0E+00	4.0E+00	L/kg	Barrows et al. (1980)
7440-39-3	Barium	ND	ND	L/kg	
7440-42-8	Boron	ND	ND	L/kg	
7440-43-9	Cadmium	2.7E+02	2.7E+02	L/kg	Kumada et al. (1972)
7440-48-4	Cobalt	ND	ND	L/kg	
7439-92-1	Lead	4.6E+01	4.6E+01	L/kg	Stephan (1993)
7439-98-7	Molybdenum	4.0E+00	4.0E+00	L/kg	Eisler (1989)
10026-03-6	Selenium (IV)	4.9E+02	1.7E+03	L/kg	Lemly (1985)
7782-49-2	Selenium (VI)	4.9E+02	1.7E+03	L/kg	Lemly (1985)
7440-28-0	Thallium	3.4E+01	1.3E+02	L/kg	T3: Barrows et al. (1980) T4: Stephan (1993)
14797-55-8	Total Nitrate Nitrogen	ND	ND	L/kg	

ND = No Data. Fish concentrations were not calculated for constituents with no BCF data.

3.7.3 Aluminum Precipitation Model

Aluminum is generally solubility limited in natural waters; therefore, a simple precipitation model was used for aluminum in lieu of the equilibrium-partitioning model. The MINTEQA2 model was used to estimate total soluble aluminum concentrations as a function of pH for a typical surface waterbody (Stumm and Morgan, 1996; Drever, 1988). By assuming the common aluminum silicate mineral gibbsite was the equilibrium solid phase, the computed values of total dissolved aluminum were interpreted as the maximum expected for each pH. If more aluminum were added to the system, it would be expected to precipitate as the mineral gibbsite for the system to maintain equilibrium. **Table 3-11** shows the maximum dissolved aluminum concentrations as a function of waterbody pH.

The precipitation model initially calculates the aluminum concentration in the surface water column by assuming that all aluminum in the groundwater flux is dissolved. If this concentration exceeds the maximum soluble concentration based on pH, the dissolved concentration is capped and the excess aluminum is assumed to precipitate as the mineral gibbsite and settle to the benthic sediment layer. The equations used in this model are presented in **Appendix E**.

Table 3-11. Aluminum Solubility as a Function of Waterbody pH[a]

Minimum pH	Maximum pH	Solubility (mg/L)
3.5	4.5	26.2
4.5	5	1.84
5	5.5	0.196
5.5	6	0.0112
6	6.5	0.00143
6.5	7	0.000662
7	7.5	0.000915
7.5	8	0.00229
8	8.5	0.00682
8.5	9	0.0212
9	9.5	0.0666
9.5	10	0.211
10	10.5	0.668

[a] Computed using MINTEQA2

Only the water column concentration for aluminum was used in subsequent exposure and risk calculations, because there is no available ecological benchmark for aluminum in sediment. The water column concentration was used to calculate human exposure via drinking water ingestion, as well as risk to ecological receptors exposed via direct contact.

3.8 Human Exposure Assessment

The human exposure component of the full-scale analysis assessed the magnitude, frequency, duration, and route of exposure to CCW contaminants that an individual may experience. The term "exposure," as defined by the EPA exposure guidelines (U.S. EPA, 1992), as the condition that occurs when a contaminant comes into contact with the outer boundary of the body. The exposure of an individual to a contaminant completes an exposure pathway (i.e., the course a constituent takes from the WMU to an exposed individual). Once the body is exposed, the constituent can cross the outer boundary and enter the body. The amount of contaminant that crosses and is available for adsorption at internal exchange boundaries is referred to as the "dose" (U.S. EPA, 1992).

This risk assessment evaluated the risk from CCW contaminants to receptors in the vicinity of a WMU. The individuals evaluated were those residents closest to the WMU. The distances from the WMU to the residents were taken from a distribution of distances to the nearest residential drinking water well measured for municipal landfills and, for the recreational fisher, a distribution of the distance of the nearest surface water body from CCW landfills and surface impoundments (see **Appendix C**).

Section 3.8.1 presents an overview of the receptors and selected exposure pathways considered for this assessment, including a discussion of how childhood exposure was considered in the analysis. **Section 3.8.2** presents exposure factors (i.e., values needed to calculate human exposure) used in the analysis. **Section 3.8.3** describes the methods used to estimate dose, including average daily dose (ADD) and lifetime average daily dose (LADD).

3.8.1 Receptors and Exposure Pathways

Human receptors may come into contact with constituents present in environmental media through a variety of pathways. The exposure pathways considered in the full-scale analysis were ingestion of drinking water from contaminated groundwater sources and ingestion of fish from surface water contaminated by groundwater.

- **Ingestion of Drinking Water.** Groundwater from an offsite well was assumed to be used for drinking water for residents (adult and child).
- **Ingestion of Fish.** Fish are exposed to constituents via uptake of contaminants from surface water. Adult recreational fishers and their children were assumed to consume fish caught in local waterbodies contaminated by CCW constituents through the groundwater-to-surface-water pathway. EPA considers this assumption to be reasonable and protective for fishers relying on locally caught fish as a food source.

Table 3-12 lists each human receptor type considered in this analysis along with the specific exposure pathways that apply to that receptor. Both adult and child residents are exposed by drinking groundwater, and adult fishers and their children are exposed by eating fish caught in streams and lakes impacted by CCW.

Table 3-12. Receptors and Exposure Pathways

Receptor	Ingestion of Drinking Water	Ingestion of Fish
Adult resident	✓	
Child resident	✓	
Adult recreational fisher		✓
Child of recreational fisher		✓

Children are an important subpopulation to consider in a risk assessment because they may be more sensitive to exposures than adults. Compared with adults, children may eat more food and drink more fluids per unit of body weight. This higher intake-rate-to-body-weight ratio can result in a higher ADD for children than adults.

As children mature, their physical characteristics and behavior patterns change. To capture these changes in the analysis, the life of a child was considered in stages represented by the following cohorts: cohort 1 (ages 1 to 5), cohort 2 (ages 6 to 11), cohort 3 (ages 12 to 19), and cohort 4 (ages 20 to 70). Associated with each cohort are distributions of exposure parameters that reflect the physical characteristics and behavior patterns of that age range. These exposure parameters are required to calculate exposure to an individual. The distributions for the 20- to 70-year-old cohort were the same as those used for adult receptors.

To capture the higher intake-rate-to-body-weight ratio of children, a start age of 1 year was selected for the child receptors. The exposure duration distribution for cohort 1 (a 1- to 5-year-old) was used to define exposure duration for the child receptors for each of the 10,000 iterations in the probabilistic analysis. For each individual iteration, the child receptor was aged through the age cohorts as appropriate until the age corresponding to the selected exposure duration was reached (e.g., if an exposure duration of 25 years was selected for an iteration, the

child was aged from 1 year to 25 years, spending 5 years in cohort 1, 6 years in cohort 2, 8 years in cohort 3, and 6 years in cohort 4, for a total of 25 years).

3.8.2 Exposure Factors

The exposure factors used in the risk assessment are listed in **Table 3-13**, along with their data sources and variable type (i.e., whether they were represented as a distribution or a fixed value in the Monte Carlo analysis). These exposure factors were used to calculate the dose of a chemical based on contact with contaminated media or food, the duration of that contact, and the body weight of the exposed individuals.

Table 3-13. Human Exposure Factor Input Parameters and Data Sources

Parameter	Variable Type	Data Source
Body weight (adult, child)	Distribution	U.S. EPA (1997c)
Ingestion rate: fish (adult, child)	Distribution	U.S. EPA (1997d)
Ingestion rate: drinking water (adult, child)	Distribution	U.S. EPA (1997c)
Exposure duration (adult, child)	Distribution	U.S. EPA (1997e)
Exposure frequency (adult, child)	Fixed (constant)	U.S. EPA policy
Fraction contaminated: drinking water	Fixed (constant)	U.S. EPA policy
Fraction contaminated: fish	Fixed (constant)	U.S. EPA policy
Fraction of TL3 fish consumed	Fixed (constant)	U.S. EPA (1997d)
Fraction of TL4 fish consumed	Fixed (constant)	U.S. EPA (1997d)
Human lifetime (used in carcinogenic risk calculation)	Fixed (constant)	U.S. EPA policy

The primary data source of human exposure model inputs used in this risk assessment was EPA's *Exposure Factors Handbook* (EFH; U.S. EPA, 1997c-e). The EFH summarizes data on human behaviors and characteristics related to human exposure from relevant key studies and provides recommendations and associated confidence estimates on the values of exposure factors. These data were carefully reviewed and evaluated for quality before being included in the EFH. EPA's evaluation criteria included peer review, reproducibility, pertinence to the United States, currency, adequacy of the data collection period, validity of the approach, representativeness of the population, characterization of variability, lack of bias in study design, and measurement error (U.S. EPA, 1997c-e). For exposure factors that were varied in the Monte Carlo analysis, probability distribution functions were developed from the values in the EFH.

The data sources and assumptions for intake and other human exposure factors used in this analysis are described below. **Appendix F** presents the exposure factors used and describes the rationale and data used to select the form of the distributions (e.g., normal, lognormal, gamma, Weibull) for those exposure factors that were varied in the probabilistic analysis. Data for three child cohorts (ages 1–5, 6–11, and 12–19 years) and adults were used. However, as most infants are breastfed and therefore are not exposed to fish or water, they were excluded from the risk assessment (i.e., modeling start age for a child is 1 year).

- **Body Weight.** Distributions of body weight were developed for adult and child receptors based on data from the EFH.

- **Fish Ingestion Rate.** Fish ingestion rates were based on a recreational angler who catches and eats some fish from a waterbody impacted by contaminants released from CCW WMUs. Distributions of fish intake rates were developed for adult fishers based on data from the 1997 EFH. At the time the risk assessment was conducted (May-June 2003), separate fish ingestion rates for children of recreational anglers were not available.

- **Drinking Water Ingestion Rate.** Distributions of drinking water intake rates were developed for the adult and child resident based on data from the EFH.

- **Exposure Duration.** Exposure duration refers to the amount of time that a receptor is exposed to a contaminant source. Exposure duration was assumed to correspond with the receptor's residence time in the same house. Exposure durations were determined using data on residential occupancy from the EFH. The data used to develop parameter information for resident receptors were age-specific. Thus, separate exposure duration distributions were developed for adult and child residents. For children, the modeling start age is 1 year, and exposure duration was used to determine the amount of time spent in each cohort (e.g., if exposure duration was 2 years, consumption rates and body weights were based only on cohort 1 data; however, if exposure duration was 21 years, the child spends 5 years in cohort 1, 6 years in cohort 2, 8 years in cohort 3, and 2 years in cohort 4/adult). Infants between birth and 1 year are not modeled because they are assumed to either breastfeed or consume commercial formula.

- **Exposure Frequency.** Exposure frequency is the frequency with which the receptor is exposed to the contaminated source during the exposure duration. Exposure frequency is not expected to vary much, so distributions were not developed for exposure frequency. All receptors were assumed to be exposed to the contaminant source 350 days/year. This value was based on the assumption that individuals are away from their homes (e.g., on vacation) approximately 2 weeks out of the year, but are otherwise exposed daily.

- **Lifetime and Averaging Time.** Averaging time is the period of time over which a receptor's dose is averaged. To evaluate carcinogens, total dose was averaged over the lifetime of the individual, assumed to be 70 years. To evaluate noncarcinogens, dose was averaged over the last year of exposure because noncancer effects may become evident during less-than-lifetime exposure durations if toxic thresholds are exceeded. Essentially, this amounts to setting exposure duration and averaging time equal so that they cancel each other out in the equation for ADD. Thus, neither exposure duration nor averaging time is included in the ADD equation.

3.8.3 Dose Estimates

An exposure assessment estimates the dose to each receptor from the contaminant concentration in the exposure medium (e.g., drinking water, fish) and the intake rate for that medium (e.g., ingestion rate of drinking water, ingestion rate of fish). For this assessment, exposure estimates were based on the *potential* dose (e.g., the dose ingested) rather than the *applied* dose (e.g., the dose delivered to the gastrointestinal tract) or the *internal* dose (e.g., the dose delivered to the target organ). Doses from groundwater or fish ingestion were calculated by multiplying the contaminant concentration in groundwater or fish by the respective intake rate on

a per kilogram body weight basis. Doses were then summed over the exposure duration, resulting in an ADD received from ingestion exposure. The ADD was used to assess noncancer risk from ingestion exposures and is defined as

$$ADD = C \times IR \tag{3-2}$$

where

C = average concentration (mass/volume or mass/mass)
IR = intake rate (mass/body weight mass/time, or volume/body weight mass/time).

Contaminant concentration represents the concentration of a chemical in a medium that contacts the body. The ADD was calculated from concentrations averaged over the exposure duration for each receptor.

For cancer effects, where the biological response is described in terms of lifetime probabilities even though exposure may not occur over the entire lifetime, dose is presented as a LADD. The LADD was used to assess cancer risks from each exposure route (i.e., ingestion) and is defined as

$$LADD = \frac{C \times IR \times ED \times EF}{AT \times 365} \tag{3-3}$$

where

C = average concentration (mass/mass or mass/volume)
IR = intake rate (mass/body weight mass/time, or volume/body weight mass/time)
ED = exposure duration (yr)
EF = exposure frequency (d/yr)
AT = averaging time (yr)
365 = units conversion factor (d/yr).

As with the ADD, contaminant concentration represents the concentration of a chemical in a medium that contacts the body. Intake rate depends on the route of exposure; for example, it might be an inhalation rate or an ingestion rate. Exposure frequency is the number of days per year the receptor is exposed to the contaminated source during the exposure duration.

For cancer effects, biological responses are described in terms of lifetime probabilities, even though exposure may not be lifelong; consequently, the exposure duration (the length of time of contact with a contaminant) was used to average the ADD over a lifetime (70 years). The media concentrations used were averaged over the duration of exposure.

3.9 Risk Estimation

The final step of the risk assessment process is to estimate the risk posed to human and ecological receptors (e.g., residents, fishers; aquatic organisms). In this step, estimates of toxicity

(the human health and ecological benchmarks) and exposure doses or exposure concentrations are integrated into quantitative expressions of risk. For the CCW constituents modeled in the full-scale assessment, the CCW human risk assessment used estimates of dose and toxicity to calculate individual excess lifetime carcinogenic risk estimates and noncancer HQs (**Section 3.9.1**). The risk calculations for ecological receptors differ from those for humans because the ecological benchmarks are developed as media concentrations (i.e., they are calculated considering ecological exposure). Thus the CCW risk assessment used estimates of exposure (media) concentrations and toxicity (media-specific concentration limits) to calculate an ecological HQ (**Section 3.9.2**).

3.9.1 Human Health Risk Estimation

The full-scale analysis focused on two human health exposure pathways: groundwater-to-drinking-water and groundwater-to-surface-water via fish consumption by recreational fishers. The cancer and noncancer health impacts of ingesting groundwater and fish contaminated by CCW leachate were estimated using the risk endpoints shown in **Table 3-14**. These endpoints were generated for each iteration of the Monte Carlo analysis. Only the cancer endpoint was used for arsenic, because it is the more sensitive endpoint compared to noncancer effects. For the other 11 constituents, only noncancer HQs were calculated, using the appropriate noncancer endpoint.

Table 3-14. Risk Endpoints Used for Human Health

Risk Category	Risk Endpoints	Definition
Cancer Effects (arsenic only)	Lifetime excess cancer risk by pathway and chemical	Lifetime excess cancer risk resulting from single pathway exposure
Noncancer Effects	Ingestion HQ by pathway and chemical	Ingestion HQ resulting from single pathway exposure
	Ingestion HQ based on drinking water action level for lead and copper	Lead and copper ingestion HQ resulting from drinking water pathway
	Average daily dose for fish consumption for lead	Lead exposure resulting from fish ingestion pathway

Cancer risks for arsenic were characterized using lifetime excess cancer risk estimates, which represent the excess probability of developing cancer over a lifetime as a result of exposure to the chemical of interest. Lifetime excess cancer risk estimates use the LADD (see **Section 3.8.3**) as the exposure metric. Lifetime excess cancer risk estimates are the product of the LADD for a specific receptor and the corresponding cancer slope factor, as shown in Equation 3-4.

$$Lifetime\ excess\ cancer\ risk_i = LADD_i \times CSF \quad (3\text{-}4)$$

where

LADD = lifetime average daily dose for ingestion pathway i (mg/kg BW/d)
i = pathway index
CSF = cancer slope factor $(mg/kg\ BW/d)^{-1}$.

Noncancer risk was characterized through the use of HQs, which are generated by dividing an ADD (see **Section 3.8.3**) for ingestion pathways by the corresponding RfD.[13] An HQ establishes whether a particular individual has experienced exposure above a threshold for a specific health effect. Therefore, unlike cancer risk estimates, HQs are not probability statements. Rather, the RfD represents an estimate (with uncertainty spanning perhaps an order of magnitude) of a daily oral exposure to the human population (including sensitive subgroups) that is likely to be without an appreciable risk of deleterious effects during a lifetime. It can be derived from a no observed adverse exposure level (NOAEL), lowest observed adverse exposure level (LOAEL), or benchmark dose, with uncertainty factors generally applied to reflect limitations of the data used. Equation 3-5 shows the calculation for the ingestion HQ. This calculation was completed for each pathway considered (i.e., drinking water ingestion and fish consumption).

$$HQ_i = \frac{ADD_i}{RfD} \tag{3-5}$$

where

ADD_i = average daily dose for ingestion pathway i (mg/kg-d)
i = pathway index
RfD = reference dose (mg/kg-d).

The risk results address risk from exposure via the groundwater-to-drinking-water and groundwater-to-surface-water pathway separately. This is appropriate because the resident consuming contaminated groundwater may not be the recreational fisher who is consuming contaminated fish. Also, the arrival time of the contaminant plume to the stream and the human receptor may not be the same for a particular iteration.[14] However, a resident may consume fish caught from a nearby stream or lake and contaminated drinking water if the travel times are similar, so that possibility should be considered as an uncertainty in this analysis (see **Section 4.4.1**).

For each receptor type, lifetime excess cancer risk estimates for arsenic were calculated separately for the drinking water and fish consumption pathways.

3.9.2 Ecological Risk Estimation

The full-scale analysis addressed two routes of exposure for ecological receptors: direct contact with contaminated media and ingestion of contaminated food items. HQs were calculated using chemical-specific media concentrations assumed to be protective of ecological receptors of concern through either exposure route (CSCLs). As described in **Section 3.1.2**, these ecological benchmarks were developed for representative organisms and communities in each environmental medium of concern.

[13] HQs calculated for lead in drinking water were based on the drinking water action level (0.015 mg/L); lead exposures from fish ingestion are reported as an ADD.
[14] Stream distance and well distance were sampled independently in the Monte Carlo analysis.

For a particular Monte Carlo iteration, HQs were calculated for sediment and surface water as the ratio between the media concentration and the ecological benchmark. Because the CSCLs were derived for an HQ of 1 (for relevant ecological endpoints), the ratio of a constituent concentration in a media to the media-specific CSCL represents the HQ for that constituent and pathway. For surface water, the HQ was calculated as follows:

$$HQ_{surface\ water} = C_{sw} / CSCL_{sw} \quad (3\text{-}6)$$

where

C_{sw} = total concentration in surface water column (mg/L)
$CSCL_{sw}$ = ecological benchmark for surface water (mg/L).

Similarly, for sediment, the HQ was calculated as

$$HQ_{sediment} = C_{sediment} / CSCL_{sediment} \quad (3\text{-}7)$$

where

$C_{sediment}$ = total concentration in sediment (mg/kg)
$CSCL_{sediment}$ = ecological benchmark for sediment (mg/kg).

Because the sediment and surface water benchmarks were based on separate receptor communities, it is not appropriate to add HQs across pathways.

4.0 Risk Characterization

This section summarizes the results of the full-scale Monte Carlo analysis and characterizes those results in terms of significant uncertainties and the scenarios and factors that influence risks to human health and the environment. Results are presented at a high-end (90th percentile) and typical (50th percentile) exposure for both pathways under each combination of WMU type, ash type, and liner type.

An overview of the assessment on which these results were based (e.g., waste management scenarios, analysis framework) is provided in **Section 2**. **Section 3** provides more details on analysis methodologies, parameter values, and assumptions. In this section, **Section 4.1** presents results from the human health risk assessment and includes an analysis of how liner conditions influence results. **Section 4.2** presents the results from the ecological risk assessment. Tables summarizing the human and ecological results are presented in each section. **Section 4.3** describes the sensitivity analysis conducted for the CCW risk assessment, and **Section 4.4** discusses how variability and uncertainty have been addressed, including a semi-quantitative review of the potential impact of some of the more significant uncertainties on results.

The probabilistic results were based on a Monte Carlo simulation in which many model input parameter values were varied over 10,000 iterations of the model per waste management scenario to yield a statistical distribution of exposures and risks. Per the Guidance for Risk Characterization developed by the EPA Science Policy Council in 1995 (http://www.epa.gov/OSA/spc/pdfs/rcguide.pdf), EPA defined the high end of the risk distribution at the 90th percentile risk or hazard estimate generated during the Monte Carlo simulation. Thus, the 90th percentile risk results are shown in this section as the high-end estimate of the risk distribution generated during the Monte Carlo simulation of constituent release, fate and transport, and exposure associated with CCW disposal in landfills and surface impoundments. In addition, the 50th percentile results are presented as the central tendency estimate of that risk distribution.

For exposure scenarios describing the waste management unit type (e.g., lined landfill; unlined surface impoundment), waste type (e.g., conventional CCW, ash mixed with coal refuse), receptor (i.e., child, adult, ecological), and health endpoint (i.e., cancer, noncancer, ecological), the 90th percentile risk represents the high-end estimate of cancer or noncancer risk that was used to help determine whether CCW disposal practices are protective of public health. To evaluate the significance of the estimated cancer risks or noncancer hazards that are attributable to CCW disposal for the exposure pathways assessed in this assessment, EPA compared the risk estimates to a risk range (for carcinogens) or to a specific risk criterion (for noncarcinogens) that are protective of human health and the environment:

- An estimate of the excess lifetime cancer risk for individuals exposed to carcinogenic (cancer-causing) contaminants ranging from 1 chance in 1,000,000 (10^{-6} excess cancer risk) to 1 chance in 10,000 (10^{-4} excess cancer risk). For decisions made to screen out

certain constituents from further consideration, a 1 in 100,000 (10^{-5}) excess lifetime cancer risk) was used.[1]

- For constituents that cause adverse, noncancer health effects (noncarcinogens), the criterion is an HQ of greater than 1, with the HQ being the ratio of the average daily exposure level to a protective exposure level corresponding to the maximum level at which no appreciable effects are likely to occur.

- An HQ greater than 1 for was used to identify constituents with adverse effects to ecological receptors.

In general, the full-scale analysis showed lower risks than the screening analysis, but still showed risks within or above the cancer risk range or above an HQ of 1 for certain CCW constituents, WMU types, pathways, and receptors at the 90th percentile. At the 50th percentile, risks are still above these levels for both WMU types, but for fewer constituents and pathways. The results presented herein are subject to further interpretation, as EPA queries the CCW risk inputs and outputs to investigate how the results may be affected by (1) waste types and environmental and waste management conditions, (2) assumptions made about these conditions in designing the probabilistic analysis, and (3) the availability of newer facility data.

4.1 Human Health Risks

This section presents the 90th and 50th percentile risk results for the two human exposure pathways evaluated in the full-scale analysis: (1) groundwater-to-drinking-water and (2) groundwater-to-surface-water (fish consumption). Results are presented for the two WMU types addressed in the analysis: landfills and surface impoundments, and show the distribution of risks across all waste types by liner type from the EPRI survey data (see **Section 4.1.3** for further discussion of liners).

4.1.1 Groundwater-to-Drinking-Water Results by Waste Type/WMU Scenario

As described in **Section 3.3**, the CCW risk assessment was organized by waste type so that different waste chemistries could be accounted for in the fate and transport modeling. The results discussed so far in this report address conventional CCW (fly ash, bottom ash, boiler slag, FGD sludge) and conventional CCW codisposed with coal refuse.[2] **Section 4.1.1.1** presents these results by waste type. FBC wastes were also modeled in this assessment. However, there was a very small number of FBC waste disposal sites (seven) in the EPRI/EPA database. For this reason, the FBC results are treated separately in **Section 4.1.1.2**. Groundwater results are reported for a resident's child because these consistently led to higher HQs, with the exception of arsenic cancer values, which were consistently higher in adults. Thus, the cancer risks reported are for adults.

[1] The typical cancer risk range used by the Office of Solid Waste and Emergency Response is 10^{-4} to 10^{-6}.

[2] Coal refuse is the waste coal produced from coal handling, crushing, and sizing operations, and tends to have a high sulfur content and low pH. In the CCW constituent database, codisposed coal refuse includes "combined ash and coal gob," "combined ash and coal refuse," and "combined bottom ash and pyrites."

Note that only the chemicals for which constituent data were adequate to model and assess risks were modeled in the full-scale assessment, and only those modeled chemical/pathway/WMU scenarios are shown in the tables and figures. For example, antimony and thallium risks are not presented for surface impoundments and mercury is not shown for either landfills or surface impoundments because more than 90% of the measurements were nondetects. For further discussion of how nondetects were treated, see **Section 4.4.3.1**. Although screening-level human health risks for aluminum and barium were below the screening criteria, they were modeled in the risk assessment due to their potential to cause ecological harm. Additionally, there were nine constituents that failed the screen but were not modeled. Instead, these constituents were dealt with using risk attenuation factors, as described in **Section 4.1.4**. The screening analysis results in **Section 3.2.4** and **Table 3-6** show which CCW constituents were modeled.

Results for two constituents (arsenic and selenium) also varied based on chemical speciation. An earlier draft of this document showed results assuming 100% trivalent arsenic (arsenic III) and 100% hexavalent selenium (selenium VI) because these forms are more mobile in soil and groundwater, and thus would show higher estimated risks than either arsenic V or selenium IV. This revised draft also presents results for arsenic V and selenium IV. The results for the two species of arsenic and selenium bound the range of possible risks for these two constituents. For further discussion of speciation, see **Section 4.4.2**.

4.1.1.1 Conventional CCW and CCW Codisposed with Coal Refuse

Tables 4-1 and **4-2** show 90th and 50th percentile risk results, respectively, by waste type and liner type for CCW landfills for the drinking water pathway. Although some risks were higher for conventional CCW and others for codisposed CCW, there was generally little difference in results between waste types for landfills. Although risks are greater for unlined landfills than for clay-lined landfills, those with composite liners show zero, or near-zero, risks for all constituents modeled in this assessment (see **Section 4.1.3** for a further discussion of risks by liner type).

Tables 4-3 and **4-4** show the 90th and 50th percentile risk results, respectively, by waste type and liner type for CCW surface impoundments for the drinking water pathway. The difference in risks between waste types is greater for surface impoundments than for landfills. For surface impoundments, some constituents present higher risks from CCW managed alone (boron, molybdenum, nitrate, and selenium). However, others presented higher risks when CCW is comanaged with coal refuse (arsenic, cadmium, cobalt, and lead). This result is likely due to the higher metal concentrations and the acidity of coal refuse leachate[3] for surface impoundments in the CCW database, which in turn result from the association of these elements (and acidity) with the sulfide minerals[4] that are concentrated in coal refuse (Finkelman, 1995). As with landfills, clay-lined units show lower risks than unlined units, and composite liners show zero, or near-zero, risks for either waste type.

[3] Many metals tend to be more soluble and mobile in acidic leachate.
[4] Arsenic: pyrite, cadmium: sphalerite, lead: galena, cobalt: pyrite.

When viewing the results in Tables 4-1 through 4-4, readers should note that these risks assume that the contaminated groundwater plume will intercept a receptor well. Because approximately two-thirds of the model runs showed surface water bodies intersecting the groundwater plume, there could be a significant number of instances where a well is either not contaminated or is less contaminated than the results below would indicate. This uncertainty is discussed further in **Section 4.4.3.3**.

Table 4-1. 90th Percentile Risk Results by CCW Type: Landfills, Groundwater-to-Drinking-Water Pathway

Chemical	90th Percentile HQ or Cancer Risk Value[a]		
	Unlined Units	Clay-Lined Units	Composite-Lined Units
Conventional CCW – 79 landfills			
Cancer			
Arsenic III	4E-04	2E-04	0
Arsenic V	2E-04	3E-05	0
Noncancer			
Aluminum	2E-03	1E-04	0
Antimony	2	0.8	0
Barium	3E-03	7E-04	0
Boron	0.7	0.4	0
Cadmium	0.7	0.4	0
Cobalt	1	0.4	0
Lead (MCL)[b]	1	0.3	0
Molybdenum	2	0.8	0
Nitrate/nitrite (MCL)[b]	0.1	0.06	2E-06
Selenium IV	0.01	3E-3	0
Selenium VI	0.2	0.1	0
Thallium	3	2	0
Codisposed CCW and Coal Refuse – 41 landfills			
Cancer			
Arsenic III	5E-04	2E-04	0
Arsenic V	4E-04	6E-05	0
Noncancer			
Aluminum	0.02	4E-04	0
Antimony	0.8	0.3	0
Barium	0.04	4E-03	0
Boron	0.3	0.1	0
Cadmium	0.2	0.07	0
Cobalt	0.8	0.09	0
Lead (MCL)[b]	0.7	0.09	0
Molybdenum	2	0.6	0

(continued)

90th Percentile Risk Results by CCW Type: Landfills, Groundwater-to-Drinking-Water Pathway (continued)

Chemical	90th Percentile HQ or Cancer Risk Value[a]		
	Unlined Units	Clay-Lined Units	Composite-Lined Units
Nitrate/nitrite (MCL)[b]	0.2	0.1	3E-06
Selenium IV	0.1	0.04	0
Selenium VI	0.7	0.3	0
Thallium	2	1	0

[a] Values are HQs for all chemicals except arsenic; arsenic values are cancer risk. Zero results indicate that contaminant infiltration rates were too small for the contaminant plume to reach the receptor during the 10,000 year period of the analysis.
[b] Values are ratios of exposure concentration to MCL.

Table 4-2. 50th Percentile Risk Results by CCW Type: Landfills, Groundwater-to-Drinking-Water Pathway

Chemical	50th Percentile HQ or Cancer Risk Value[a]		
	Unlined Units	Clay-Lined Units	Composite-Lined Units
Conventional CCW – 79 landfills			
Cancer			
Arsenic III	6E-06	4E-06	0
Arsenic V	6E-10	3E-14	0
Noncancer			
Aluminum	5E-07	3E-07	0
Antimony	0.04	0.02	0
Barium	0	0	0
Boron	0.01	0.01	0
Cadmium	0.01	8E-03	0
Cobalt	3E-03	8E-06	0
Lead (MCL)[b]	4E-04	2E-08	0
Molybdenum	0.1	0.04	0
Nitrate/nitrite (MCL)[b]	0.004	0.003	0
Selenium IV	0	0	0
Selenium VI	9E-03	6E-03	0
Thallium	0.2	0.1	0
Codisposed CCW and Coal Refuse – 41 landfills			
Cancer			
Arsenic III	2E-05	6E-06	0
Arsenic V	6E-06	7E-10	0

(continued)

50th Percentile Risk Results by CCW Type: Landfills, Groundwater-to-Drinking-Water Pathway (continued)

Chemical	50th Percentile HQ or Cancer Risk Value[a]		
	Unlined Units	Clay-Lined Units	Composite-Lined Units
Noncancer			
Aluminum	4E-06	2E-09	0
Antimony	0.05	0.02	0
Barium	5E-05	7E-07	0
Boron	8E-03	3E-03	0
Cadmium	0.02	4E-03	0
Cobalt	2E-05	0	0
Lead (MCL)[b]	0.01	2E-07	0
Molybdenum	0.02	6E-03	0
Nitrate/nitrite (MCL)[b]	0.04	0.009	0
Selenium IV	2E-09	2E-15	0
Selenium VI	0.03	0.01	0
Thallium	0.2	0.07	0

[a] Values are HQs for all chemicals except arsenic; arsenic values are cancer risk. Zero results indicate that contaminant infiltration rates were too small for the contaminant plume to reach the receptor during the 10,000 year period of the analysis.
[b] Values are ratios of exposure concentration to MCL.

Table 4-3. 90th Percentile Risk Results by CCW Type: Surface Impoundments, Groundwater-to-Drinking-Water Pathway

Chemical	90th Percentile HQ or Cancer Risk Value[a]		
	Unlined Units	Clay-Lined Units	Composite-Lined Units
Conventional CCW – 44 surface impoundments			
Cancer			
Arsenic III	2E-03	9E-04	2E-07
Arsenic V	7E-04	2E-04	0
Noncancer			
Aluminum	2E-03	1E-03	2E-07
Barium	5E-03	3E-03	2E-11
Boron	7	4	5E-03
Cadmium	0.5	0.3	4E-11
Cobalt	0.9	0.4	0
Lead (MCL)[b]	3	0.7	1E-21
Molybdenum	8	5	7E-03
Nitrate/nitrite (MCL)[b]	20	10	9E-04
Selenium IV	0.4	0.1	1E-04

(continued)

90th Percentile Risk Results by CCW Type: Surface Impoundments, Groundwater-to-Drinking-Water Pathway (continued)			
	90th Percentile HQ or Cancer Risk Value[a]		
Chemical	Unlined Units	Clay-Lined Units	Composite-Lined Units
Selenium VI	2	1	1E-03
Codisposed CCW and Coal Refuse – 72 surface impoundments			
Cancer			
Arsenic III	2E-02	7E-03	4E-06
Arsenic V	2E-02	2E-03	3E-09
Noncancer			
Aluminum	0.3	0.07	6E-07
Barium	7E-03	3E-03	9E-07
Boron	1	0.5	2E-03
Cadmium	9	3	5E-05
Cobalt	500	200	3E-06
Lead (MCL)[b]	9	1	1E-19
Molybdenum	3	2	4E-03
Nitrate/nitrite (MCL)[b]	0.4	0.2	1E-04
Selenium IV	0.3	0.1	3E-10
Selenium VI	0.8	0.4	1E-03

[a] Values are HQs for all chemicals except arsenic; arsenic values are cancer risk. Zero results indicate that contaminant infiltration rates were too small for the contaminant plume to reach the receptor during the 10,000 year period of the analysis.
[b] Values are ratios of exposure concentration to MCL.

Table 4-4. 50th Percentile Risk Results by CCW Type: Surface Impoundments, Groundwater-to-Drinking-Water Pathway

	50th Percentile HQ or Cancer Risk Value[a]		
Chemical	Unlined Units	Clay-Lined Units	Composite–Lined Units
Conventional CCW – 44 surface impoundments			
Cancer			
Arsenic III	1E-04	6E-05	0
Arsenic V	2E-05	4E-06	0
Noncancer			
Aluminum	2E-05	1E-05	8E-20
Barium	1E-04	1E-04	0
Boron	0.4	0.2	3E-11
Cadmium	0.05	0.02	0
Cobalt	0.2	0.05	0

(continued)

50th Percentile Risk Results by CCW Type: Surface Impoundments, Groundwater-to-Drinking-Water Pathway (continued)

Chemical	50th Percentile HQ or Cancer Risk Value[a]		
	Unlined Units	Clay-Lined Units	Composite–Lined Units
Lead (MCL)[b]	0.05	0.007	0
Molybdenum	1	0.5	2E-11
Nitrate/nitrite (MCL)[b]	0.1	0.05	7E-08
Selenium IV	8E-04	4E-10	0
Selenium VI	0.1	0.07	2E-11
Codisposed CCW and Coal Refuse – 72 surface impoundments			
Cancer			
Arsenic III	6E-04	2E-04	0
Arsenic V	3E-04	4E-05	0
Noncancer			
Aluminum	5E-04	4E-05	0
Barium	4E-04	2E-04	0
Boron	0.1	0.06	5E-15
Cadmium	0.1	0.05	0
Cobalt	20	6	0
Lead (MCL)[b]	0.1	0.01	0
Molybdenum	0.8	0.3	3E-18
Nitrate/nitrite (MCL)[b]	0.03	0.01	4E-08
Selenium IV	3E-03	9E-05	0
Selenium VI	0.1	0.03	5E-15

[a] Values are HQs for all chemicals except arsenic; arsenic values are cancer risk. Zero results indicate that contaminant infiltration rates were too small for the contaminant plume to reach the receptor during the 10,000 year period of the analysis.

[b] Values are ratios of exposure concentration to MCL.

4.1.1.2 FBC Wastes

Tables 4-5 and **4-6** show the 90th and 50th percentile risk results for FBC landfills by liner type. These results suggest lower risks than for conventional CCW and CCW codisposed with coal refuse. The difference may be attributed to lower FBC leachate concentrations and the alkaline nature of FBC waste. Note that clay-lined FBC landfills show higher risks than unlined facilities, which is counterintuitive considering how clay-lined and unlined units are designed and operated. This result reflects the characteristics of the limited number and locations of FBC landfills.[5] When the risk results of an exposure pathway are viewed at a resolution finer than the analysis design, a small sample size, along with the interactions of liner type with other site-

[5] FBC WMU data were available for only seven landfills (3 unlined, 3 clay-lined, and 1 composite-lined), and it is not known how representative these data are with respect to WMU characteristics and locations throughout the United States.

based inputs, can produce unexpected results. In the case of FBC wastes, the characteristics of the three unlined landfills (primarily infiltration rate and areas) were such that their risks were lower than the three clay-lined FBC landfills.

Table 4-5. 90th Percentile Risk Results for FBC Wastes: Landfills, Groundwater-to-Drinking-Water Pathway

Chemical	90th Percentile HQ or Cancer Risk Value[a]		
	Unlined Units	Clay-Lined Units	Composite-Lined Units
FBC Waste – 7 landfills			
Cancer			
Arsenic III	3E-05	6E-05	0
Arsenic V	2E-05	2E-05	0
Noncancer			
Aluminum	4E-06	2E-05	0
Antimony	0.8	3	0
Barium	4E-04	2E-03	0
Boron	0.02	0.07	0
Cadmium	0.1	0.3	0
Cobalt	0.4	0.8	0
Lead (MCL)[b]	0.4	0.6	0
Molybdenum	0.2	0.5	0
Nitrate/nitrite (MCL)[b]	0.03	0.07	5E-08
Selenium IV	3E-14	0.05	0
Selenium VI	0.08	0.1	0
Thallium	1	4	0

[a] Values are HQs for all chemicals except arsenic; arsenic values are cancer risk. Zero results indicate that contaminant infiltration rates were too small for the contaminant plume to reach the receptor during the 10,000-year period of the analysis.
[b] Values are ratios of exposure concentration to MCL.

Table 4-6. 50th Percentile Risk Results for FBC Wastes: Landfills, Groundwater-to-Drinking-Water Pathway

Chemical	50th Percentile HQ or Cancer Risk Value[a]		
	Unlined Units	Clay-Lined Units	Composite-Lined Units
FBC Waste – 7 landfills			
Cancer			
Arsenic III	0	4E-07	0
Arsenic V	0	5E-10	0
Noncancer			
Aluminum	0	0	0
Antimony	0	0.09	0

(continued)

50th Percentile Risk Results for FBC Wastes: Landfills, Groundwater-to-Drinking-Water Pathway (continued)

Chemical	50th Percentile HQ or Cancer Risk Value[a]		
	Unlined Units	Clay-Lined Units	Composite-Lined Units
Barium	0	0	0
Boron	0	0.003	0
Cadmium	0	0.01	0
Cobalt	0	3E-03	0
Lead (MCL)[b]	0	2E-04	0
Molybdenum	0	0.04	0
Nitrate/nitrite (MCL)[b]	3E-08	0.004	0
Selenium IV	0	5E-15	0
Selenium VI	0	0.01	0
Thallium	0	0.2	0

[a] Values are HQs for all chemicals except arsenic; arsenic values are cancer risk. Zero results indicate that contaminant infiltration rates were too small for the contaminant plume to reach the receptor during the 10,000-year period of the analysis.
[b] Values are ratios of exposure concentration to MCL.

4.1.1.3 Comparing Landfills and Surface Impoundments

The full-scale analysis produced lower risks for landfills than for surface impoundments. The higher risks for surface impoundments as compared to landfills reflect higher constituent concentrations in the surface impoundment wastes and a higher hydraulic head in an impoundment that drives leachate into the underlying soil with greater force than infiltration in landfills. This higher head results in a greater flux of contaminants to groundwater during the active life of the surface impoundment, especially in unlined units. In combination with the higher CCW constituent concentrations in surface impoundment porewater and a greater proportion of unlined units, these factors lead to more and higher risk exceedances for surface impoundments than for landfills.

4.1.1.4 The Effect of Liners

The analysis demonstrates that the presence of liners, especially composite liners, reduce leaching and risks from CCW landfills and surface impoundments. Note that 90th percentile risks from composite liners are zero for most constituents for landfills, which means that in 90 percent of the cases, the contaminant did not reach the receptor well in the 10,000 year limit for this analysis. Composite liners also reduced risks for surface impoundments for several constituents at the 90th percentile by 4 to 10 orders of magnitude and generated risk results well below the cancer risk range or noncancer risk criterion used for this analysis. Infiltration rates for composite-lined surface impoundments are largely controlled by leak density (see **Section 3.5**), which is an empirical distribution from the same source as the landfill infiltration rates (U.S. EPA, 2002b), and are subject to similar uncertainties.

Zero values reflect the liner leakage rates in the empirical data set used to develop composite landfill liner infiltration rates used in this risk assessment (from U.S. EPA, 2002b; see **Section 3.4.2**), which are mostly zero values or very low in terms of infiltration rate. Although these infiltration rates are based on the best data available to EPA, these data are not specific to CCW facilities. This represents an uncertainty in the analysis (see **Sections 3.4.2** and **4.4.3.2**).

4.1.1.5 Modeled Peak Concentration Arrival Times

Arrival times for the peak well concentrations used to calculate groundwater to drinking water risks for selected CCW constituents (arsenic, boron, cobalt, selenium, and thallium) are plotted as cumulative distributions for surface impoundments and landfills in the figures in **Appendix L.** These constituents were selected to represent the chemicals with the highest risks and to span the range of mobility in the subsurface. **Table 4-7** summarizes these time of travel results by showing selected percentiles from these distributions for each WMU/liner combination modeled in the risk assessment.

As can be seen in Table 4-7, the peak arrival times for most constituents in unlined surface impoundment is less than 100 years (i.e., peak concentration occurs before or shortly after surface impoundment closure). The 10th percentile ranges from 70 years (for arsenic III, boron, and selenium VI) to 76 years (for selenium IV). The 50th percentile arrival times remain under 100 years for most constituents, with only the less mobile forms of arsenic and selenium having 50th percentile arrival times later than 100 years.

Arrival times for unlined landfills are much longer, ranging up to thousands of years. For boron and selenium IV, the 50th percentiles are 2,000 and 10,000 years respectively. However, even at the 10th percentile, arrival times ranged from 300 years (for boron) to 4,600 years (for selenium IV).

At the higher percentiles, arrival times shown as greater than 10,000 years indicate that the contaminant plume did not reach the well before the simulation ended. Although the plume might eventually reach the well in these cases, EPA does not believe that extending the simulation beyond 10,000 years would have captured any significant risk beyond what was captured by the selection of the 90th percentile values, which reflect cases where the plume did reach the well. In other words, the 90th percentile values would not be influenced by whether lower percentile concentrations were zero or the concentration at a peak beyond 10,000 years.

Table 4-7. Time to Peak Well Concentration by WMU and Liner Type as Modeled

Liner	Time to Peak (years)[a,b]							
	Percentile	Arsenic III	Arsenic IV	Boron	Cobalt	Selenium IV	Selenium VI	Thallium[c]
Landfills (all waste types)								
Unlined	10	400	2,000	300	1,200	4,600	400	580
	30	1,100	7,100	880	4,100	9,400	1,000	1,100
	50	2,800	9,700	2,000	7,800	10,000	2,600	2,300
	70	6,400	10,000	4,300	10,000	>10,000	5,500	4,400
	90	>10,000	>10,000	9,400	>10,000	>10,000	10,000	9,700

(continued)

Time to Peak Well Concentration by WMU and Liner Type as Modeled (continued)

Liner	Time to Peak (years)[a,b]							
	Percentile	Arsenic III	Arsenic IV	Boron	Cobalt	Selenium IV	Selenium VI	Thallium[c]
Compacted clay	10	400	1,900	550	1,400	8,100	400	570
	30	1,400	8,200	1,400	5,900	>10,000	1,300	1,200
	50	4,000	10,000	5,600	10,000	>10,000	5,100	4,300
	70	>10,000	>10,000	10,000	>10,000	>10,000	10,000	10,000
	90	>10,000	>10,000	>10,000	>10,000	>10,000	>10,000	>10,000
Synthetic or composite (clay and synthetic)	10	10,000	>10,000	9,600	>10,000	10,000	9,000	>10,000
	30	>10,000	>10,000	>10,000	>10,000	>10,000	>10,000	>10,000
	50	>10,000	>10,000	>10,000	>10,000	>10,000	>10,000	>10,000
	70	>10,000	>10,000	>10,000	>10,000	>10,000	>10,000	>10,000
	90	>10,000	>10,000	>10,000	>10,000	>10,000	>10,000	>10,000
Surface Impoundments (all waste types)								
Unlined	10	70	73	70	71	76	70	N/A
	30	73	97	72	78	610	72	N/A
	50	78	220	74	97	4,400	74	N/A
	70	91	890	80	190	10,000	80	N/A
	90	170	6,500	110	970	>10,000	110	N/A
Compacted clay	10	75	95	75	86	81	75	N/A
	30	86	350	80	140	3,000	80	N/A
	50	110	1,300	90	270	7,900	90	N/A
	70	150	5,000	110	690	10,000	110	N/A
	90	340	10,000	150	3,100	>10,000	150	N/A
Synthetic or composite (clay and synthetic)	10	1,300	10,000	960	9,500	1,900	990	N/A
	30	3,900	>10,000	2,800	>10,000	6,900	2,800	N/A
	50	8,600	>10,000	4,400	>10,000	>10,000	4,600	N/A
	70	>10,000	>10,000	7,000	>10,000	>10,000	7,300	N/A
	90	>10,000	>10,000	10,000	>10,000	>10,000	10,000	N/A

[a] Arrival times have been rounded to two significant digits.
[b] >10,000 indicates that the contaminant plume did not reach the receptor well during the modeled period.
[c] N/A = Not Applicable. Thallium was not modeled for surface impoundments (see **Section 4.1.1** above).

As with the higher constituent concentrations that are characteristic of surface impoundments, the shorter arrival times for surface impoundments are primarily due to the hydraulic head of the waste liquids in the unit; by contrast, landfill leaching is driven solely by infiltration of precipitation through the cap and liner of the unit and the peak concentration takes much longer to reach the well.

The arrival times presented in Table 4-7 correspond to the arrival of the maximum estimated risks for each model run. However, for model runs where the risk range or HQ criterion was exceeded, the first exceedence would sometimes occur earlier than the maximum risk arrivals reported in Table 4-7. This is consistent with the appearance of damage cases described in U.S. EPA (2007), which were sometimes observed sooner than the time-to-peak estimates in Table 4-7.

4.1.2 Groundwater-to-Surface-Water (Fish Consumption) Pathway

Like the drinking water results above, the fish consumption results are organized by waste type so that different waste chemistries could be accounted for. **Section 4.1.2.1** presents the results for conventional CCW and codisposed CCW by WMU and liner type. FBC wastes were also modeled for the surface water pathway, and these results are treated separately in **Section 4.1.2.2**. Note that only the four constituents that failed the surface water screen were probabilistically modeled for this scenario. Of those, thallium risks are not presented for surface impoundments because of a high proportion (>90%) of nondetects in the surface impoundment data (see **Section 4.4.3.1** for further discussion). The screening analysis results in **Section 3.2.4** and **Table 3-6** show which CCW constituents exceeded the surface water screening criteria.

4.1.2.1 Conventional CCW and CCW Codisposed with Coal Refuse

Tables 4-8 and **4-9** present the 90th and 50th percentile risk results, respectively, by waste type and liner type for CCW landfills for the fish consumption pathway. The results presented are for a fisher's child because those risks were consistently higher than the risks for the adult fisher. As seen in these tables, the results for landfills that codispose of CCW are not drastically different from those that handle only conventional CCW. At the 90th percentile, only unlined landfills that comanage CCW present risks at an HQ of 1 (for selenium). The remainder of the modeled constituents had risks below an excess cancer risk of 1 in 100,000 or an HQ of 1 at the 90th percentile. 50th percentile results were all well below these levels for both cancer and noncancer risks.

Tables 4-10 and **4-11** present the 90th and 50th percentile risk results, respectively, by waste type and liner for CCW surface impoundments for the fish consumption pathway. Again, risks are higher for surface impoundments than for landfills because of the higher waste concentrations and the higher hydraulic head in these units, as discussed previously for the drinking water pathway. Results at that 90th percentile exceeded an HQ of 1 for selenium in unlined (HQ of 3) and clay-lined (HQ of 2) impoundments managing conventional CCW, and also exhibited excess cancer risks just above 1 in 100,000 for arsenic in unlined (3 in 100,000) and clay-lined (2 in 100,000) impoundments comanaging CCW. Fish consumption pathway 50th percentile results are well below an excess cancer risk of 1 in 1,000,000 and an HQ of 1 for all constituents, waste management scenarios, and liner types.

Table 4-8. 90th Percentile Risk Results by CCW Type: Landfills, Groundwater-to-Surface-Water Pathway

Chemical[b]	90th Percentile HQ or Cancer Risk Value[a]		
	Unlined Units	Clay-Lined Units	Composite-Lined Units
Conventional CCW – 79 landfills			
Cancer			
Arsenic III	1E-06	1E-07	0
Arsenic V	4E-07	3E-09	0

(continued)

90th Percentile Risk Results by CCW Type: Landfills, Groundwater-to-Surface-Water Pathway (continued)

Chemical[b]	90th Percentile HQ or Cancer Risk Value[a]		
	Unlined Units	Clay-Lined Units	Composite-Lined Units
Noncancer			
Cadmium	0.09	6E-03	0
Selenium IV	6E-05	1E-04	0
Selenium VI	0.3	0.04	0
Thallium	0.4	0.04	0
Codisposed CCW and Coal Refuse – 41 landfills			
Cancer			
Arsenic III	2E-06	8E-07	0
Arsenic V	2E-06	2E-07	0
Noncancer			
Cadmium	0.05	0.01	0
Selenium IV	0.03	9E-03	0
Selenium VI	1	0.4	0
Thallium	0.4	0.2	0

[a] Values are HQs for all chemicals except arsenic; arsenic values are cancer risk. Zero results indicate that contaminant infiltration rates were too small for the contaminant plume to reach the receptor during the 10,000 year period of the analysis.

[b] Note that only the chemicals with adequate data that were identified in the screening analysis as needing further assessment (see **Section 3.2.4**) were modeled.

Table 4-9. 50th Percentile Risk Results by CCW Type: Landfills, Groundwater-to-Surface-Water Pathway

Chemical[b]	50th Percentile HQ or Cancer Risk Value[a]		
	Unlined Units	Clay-Lined Units	Composite-Lined Units
Conventional CCW – 79 landfills			
Cancer			
Arsenic III	3E-10	7E-11	0
Arsenic V	4E-14	1E-18	0
Noncancer			
Cadmium	2E-05	3E-06	0
Selenium IV	0	0	0
Selenium VI	2E-04	4E-05	0
Thallium	1E-04	5E-05	0

(continued)

50th Percentile Risk Results by CCW Type: Landfills, Groundwater-to-Surface-Water Pathway (continued)

Chemical[b]	50th Percentile HQ or Cancer Risk Value[a]		
	Unlined Units	Clay-Lined Units	Composite-Lined Units
Codisposed CCW and Coal Refuse – 41 landfills			
Cancer			
Arsenic III	4E-09	3E-09	0
Arsenic V	2E-10	8E-14	0
Noncancer			
Cadmium	1E-04	6E-05	0
Selenium IV	6E-10	1E-15	0
Selenium VI	3E-03	3E-03	0
Thallium	1E-03	1E-03	0

[a] Values are HQs for all chemicals except arsenic; arsenic values are cancer risk. Zero results indicate that contaminant infiltration rates were too small for the contaminant plume to reach the receptor during the 10,000 year period of the analysis.

[b] Note that only the chemicals with adequate data that were identified in the screening analysis as needing further assessment (see **Section 3.2.4**) were modeled.

Table 4-10. 90th Percentile Risk Results by CCW Type: Surface Impoundments, Groundwater-to-Surface-Water Pathway

Chemical[b]	90th Percentile HQ or Cancer Risk Value[a]		
	Unlined Units	Clay-Lined Units	Composite-Lined Units
Conventional CCW – 44 surface impoundments			
Cancer			
Arsenic III	8E-06	4E-06	1E-12
Arsenic V	2E-06	4E-07	0
Noncancer			
Cadmium	0.09	0.04	2E-15
Selenium IV	0.6	0.04	1E-07
Selenium VI	3	2	2E-06
Codisposed CCW and Coal Refuse – 72 surface impoundments			
Cancer			
Arsenic III	3E-05	2E-05	1E-14
Arsenic V	2E-05	8E-06	6E-19
Noncancer			
Cadmium	0.5	0.3	8E-13
Selenium IV	0.2	0.05	0

(continued)

90th Percentile Risk Results by CCW Type: Surface Impoundments, Groundwater-to-Surface-Water Pathway (continued)

Chemical[b]	90th Percentile HQ or Cancer Risk Value[a]		
	Unlined Units	Clay-Lined Units	Composite-Lined Units
Selenium VI	1	0.8	7E-10

[a] Values are HQs for all chemicals except arsenic; arsenic values are cancer risk. Zero results indicate that contaminant infiltration rates were too small for the contaminant plume to reach the receptor during the 10,000 year period of the analysis.

[b] Note that only the chemicals with adequate data that were identified in the screening analysis as needing further assessment (see **Section 3.2.4**) were modeled.

Table 4-11. 50th Percentile Risk Results by CCW Type: Surface Impoundments, Groundwater-to-Surface-Water Pathway

Chemical[b]	50th Percentile HQ or Cancer Risk Value[a]		
	Unlined Units	Clay-Lined Units	Composite–Lined Units
Conventional CCW – 44 surface impoundments			
Cancer			
Arsenic III	4E-08	6E-10	0
Arsenic V	7E-09	2E-11	0
Noncancer			
Cadmium	6E-04	6E-06	0
Selenium IV	5E-05	1E-11	0
Selenium VI	0.02	3E-04	0
Codisposed CCW and Coal Refuse – 72 surface impoundments			
Cancer			
Arsenic III	6E-08	1E-08	0
Arsenic V	3E-08	2E-09	0
Noncancer			
Cadmium	1E-03	2E-04	0
Selenium IV	8E-05	4E-07	0
Selenium VI	3E-03	8E-04	0

[a] Values are HQs for all chemicals except arsenic; arsenic values are cancer risk. Zero results indicate that contaminant infiltration rates were too small for the contaminant plume to reach the receptor during the 10,000 year period of the analysis.

[b] Note that only the chemicals with adequate data that were identified in the screening analysis as needing further assessment (see **Section 3.2.4**) were modeled.

As with the groundwater-to-drinking-water pathway analysis, the absence of risk from composite-lined units indicates that the liners modeled in this analysis are effective at preventing contaminants from reaching the surface waterbodies of interest. One should keep in mind that all surface water results are calculated assuming that constituents are being added to the waterbodies only via groundwater. However, for surface impoundment operation, effluent is constantly being discharged directly into that same waterbody. These discharges are regulated

under the Clean Water Act, and although they pose an uncertainty in the analysis, they are outside the scope of the risk assessment (see **Section 4.4.1** for further discussion).

4.1.2.2 FBC Wastes

Tables 4-12 and **4-13** show the 90th and 50th percentile risk results for FBC landfills by liner type. These results are much lower than those for conventional CCW and comanaged CCW landfills seen above, and suggest that releases from FBC landfills do not present a hazard to surface waters. This difference may be attributed to lower FBC leachate concentrations and the alkaline nature of FBC wastes. However, as with the FBC results reported for drinking water, the results here are strongly influenced by the small sample size of site data available. Thus, the limitation of only having seven sites may present an uncertainty in the analysis.

Table 4-12. 90th Percentile Risk Results for FBC Wastes: Landfills, Groundwater-to-Surface-Water Pathway

Chemical	90th Percentile HQ or Cancer Risk Value[a]		
	Unlined Units	Clay-Lined Units	Composite-Lined Units
FBC Waste – 7 landfills			
Cancer			
Arsenic III	4E-12	7E-08	0
Arsenic V	3E-12	3E-08	0
Noncancer			
Cadmium	7E-07	0.02	0
Selenium IV	3E-17	8E-03	0
Selenium VI	5E-06	0.1	0
Thallium	5E-06	0.2	0

[a] Values are HQs for all chemicals except arsenic; arsenic values are cancer risk. Zero results indicate that contaminant infiltration rates were too small for the contaminant plume to reach the receptor during the 10,000 year period of the analysis.

[b] Note that only the chemicals with adequate data that were identified in the screening analysis as needing further assessment (see Section 3.2.4) were modeled.

Table 4-13. 50th Percentile Risk Results for FBC Wastes: Landfills, Groundwater-to-Surface-Water Pathway

Chemical	50th Percentile HQ or Cancer Risk Value[a]		
	Unlined Units	Clay-Lined Units	Composite-Lined Units
FBC Waste – 7 landfills			
Cancer			
Arsenic III	0	3E-13	0
Arsenic V	0	6E-14	0

(continued)

50th Percentile Risk Results for FBC Wastes: Landfills, Groundwater-to-Surface-Water Pathway (continued)

Chemical	50th Percentile HQ or Cancer Risk Value[a]		
	Unlined Units	Clay-Lined Units	Composite-Lined Units
Noncancer			
Cadmium	0	2E-05	0
Selenium IV	0	8E-16	0
Selenium VI	0	1E-03	0
Thallium	0	1E-03	0

[a] Values are HQs for all chemicals except arsenic; arsenic values are cancer risk. Zero results indicate that contaminant infiltration rates were too small for the contaminant plume to reach the receptor during the 10,000 year period of the analysis.

[b] Note that only the chemicals with adequate data that were identified in the screening analysis as needing further assessment (see Section 3.2.4) were modeled.

4.1.3 Results by Liner Type

The effect of liner type on human health risk for the groundwater-to-drinking-water pathways can be seen in Tables 4-1 through 4-6 and for the groundwater-to-surface water pathway in Tables 4-8 through 4-13, which present risks for WMUs that are unlined, clay lined, and lined with composite liners from the 1995 EPRI survey data (EPRI, 1997). At the 90th percentile, lined units produced lower risk estimates than unlined units for all constituents modeled. Composite liners produced very low to zero risk estimates as compared to clay liners for all constituents modeled for both landfills and surface impoundments. For surface impoundments, clay liners produced higher risk estimates for all constituents as compared to clay liners in landfills. Similar trends are evident at the 50th percentile, where composite liners produced risk estimates of zero or near zero for all constituents for surface impoundments.

Table 4-14 shows how frequent each of the liner types is in the 1995 EPRI survey data modeled in this analysis, and it compares these data with the liner type frequency in the more recent DOE/EPA study (U.S. DOE, 2006). The 56 WMUs surveyed in the U.S. DOE 2006 study were commissioned between 1994 and 2004. Although the actual number of WMUs that were established in that timeframe cannot be verified, based on proxy data (i.e., CCW available for disposal in those states with identified, new WMUs and coal-fired power plant generating capacity), the sample coverage is estimated to be at least 61–63 percent of the total population of the newly commissioned WMUs.[6] With the exception of one landfill, the newly constructed facilities are all lined, with either clay, synthetic, or composite liners. The single unlined landfill identified in the recent DOE report receives bottom ash, which is characterized as an inert waste by the state, and therefore, a liner is not required.

[6] For additional details as to how these estimates were derived, the reader is referred to the DOE study, pages S-2 – S-3 of the Summary Section and Section 3.1.2..

Table 4-14. Liner Types in EPRI Survey

Liner Type	Landfills	Surface Impoundments
1995 EPRI Survey[a] – 181 facilities		
Unlined	40%	68%
Compacted clay	45%	27%
Synthetic or composite (clay and synthetic)	16%	5%
Total	100%	100%
2004 DOE Survey[b] – 56 Facilities		
Unlined	3%	0%
Compacted clay	29%	17%
Synthetic or composite (clay and synthetic)	68%	83%
Total	100%	100%

[a] EPRI (1997)
[b] U.S.DOE (2006)

As Table 4-14 shows, there is a marked trend away from unlined WMUs in favor of lined units, with a distinct preference for synthetic or composite liners. Comparison of the 26 coal combustion plants in both the EPRI survey and the DOE/EPA survey (U.S. DOE, 2006) shows that although most of those facilities (17 of 26) were using unlined WMUs in 1995, all 26 are now placing wastes in new or expanded landfills or surface impoundments that are lined with clay, synthetic, or composite liners. However, it is likely that the older unlined units were closed with wastes in place, and that these wastes could therefore still pose a threat through groundwater pathways. Also, the number of unlined unit that continue to operate in the United States cannot be determined from the available data.

As described in **Sections 3.4.1** and **3.5.1**, the characteristics of the liners used in the CCW risk were taken from the IWEM model as representative of the general performance of each liner type. For landfills, an engineered compacted clay liner (3 feet thick, with a hydraulic conductivity of 1×10^{-7} cm/s) reduced the 90th percentile risk by a factor of about 2 to 4 compared to no liner, but did not change the constituents at or above an excess cancer risk of 1 in 100,000 (arsenic, excess cancer risk of 1 in 5,000) or an HQ of 1 (thallium, HQ of 2). For surface impoundments, clay liners did reduce the risk to just below an HQ of 1 for cobalt, lead, and selenium.

Composite (clay and synthetic) liners, as modeled in this risk assessment (see **Sections 3.4** and **3.5**), were much more effective at reducing risk for all constituents; 90th (and 50th) percentile risks with composite liners for landfills were zero[7] for arsenic and metals and very low or zero for reactive nitrogen compounds (nitrate and nitrite), and were well below an excess cancer risk of 1 in 100,000 or an HQ of 1 for all constituents for surface impoundments. The analysis used data collected for composite liner performance at industrial waste management

[7] The absence of risk indicates that contaminant infiltration rates were too small for the contaminant plume to reach the receptor well during the 10,000 year period of the analysis. See Section 3.2.2 for a discussion of the empirical liner infiltration data used in this analysis.

facilities, including liner leakage rate for landfills and the number of liner perforations for surface impoundments (TetraTech, 2001). Because data on CCW liner leakage rates were not available, there is some uncertainty in applying these Industrial D Guidance liner performance data to CCW disposal units. Still, these rates do reflect actual performance data from liners under real WMUs. They demonstrate that composite liners can be effective in reducing leaching from CCW WMUs and suggest that there will be a decrease in risk from CCW disposal if more facilities line their WMUs with composite or clay liners. Information from the more recent DOE/EPA study (U.S. DOE, 2006) indicates that composite liners are becoming more prevalent in newly constructed facilities, so the risks from CCW disposal should be lower for newer CCW landfills and surface impoundments.

4.1.4 Constituents Not Modeled in the Full-Scale Assessment

As described in **Section 3.2.4**, full-scale modeling was not conducted for all 21 constituents that were above the screening criteria in the initial screening analysis; only constituents that were judged likely to have generally higher risks to human health and ecological health were modeled in the full-scale risk assessment.[8] Five chemicals (chromium, fluoride, manganese, vanadium, and nickel) had drinking water pathway HQs in the screening analysis ranging from 1 to less than 6 for surface impoundments, and three (chromium, fluoride, and vanadium) had screening HQs of 2 for landfills.

To address these unmodeled constituents, EPA developed surrogate risk attenuation factors by dividing the screening risk results by the full-scale risk results, across all unit (liner) types combined, for the constituents modeled in the full-scale assessment. This comparison was done only for the drinking water exposure pathway, the only human health exposure pathway for which the risks for these constituents were above the screening criteria. **Table 4-15** shows the risk attenuation factor statistics for the modeled constituents, and **Table 4-16** shows the results of applying the median and 10th percentile attenuation factors to the screening risk results for the marginal constituents. Differences in attenuation among the modeled constituents reflect differences in contaminant sorption and mobility. To be conservative, the 10th percentile attenuation factor was selected as a high-end value representing the more mobile constituents, such as arsenic, selenium, and molybdenum. The 50th percentile (or median) risk represents a central tendency value.

Table 4-15. Risk Attenuation Factor[a] Statistics for Modeled Constituents—Groundwater to Drinking Water Pathway

Statistic	Landfill	Surface Impoundment
10th percentile	7	1.6
50th percentile	12	2.6
Average	16	3.3
Maximum	40	9.3

(continued)

[8] These constituents of concern had human health HQs greater than 6 or both ecological HQs greater than 100 at the 90th percentile.

Risk Attenuation Factor[a] Statistics for Modeled Constituents— Groundwater to Drinking Water Pathway (continued)

Statistic	Landfill	Surface Impoundment
Number of data points	9	8

[a] The risk attenuation factor is the ratio of the full-scale analysis risk and screening analysis risk for a constituent modeled in the full-scale analysis.

Table 4-16. Summary of Risk Screening Values for Unmodeled Constituents Using Risk Attenuation Factors—Groundwater-to-Drinking-Water Pathway

WMU/Pathway	Landfill			Surface Impoundment		
	Screening HQ	HQ with Median Attenuation	HQ with 10th Percentile Attenuation	Screening HQ	HQ with Median Attenuation	HQ with 10th Percentile Attenuation
Chromium VI	2.3	0.2	0.3	4.2	1.6	2.6
Fluoride	1.8	0.2	0.3	5.2	2.0	3.3
Manganese	1	0.1	0.1	5.6	2.2	3.5
Vanadium	2.2	0.2	0.3	2.3	0.9	1.4
Nickel	-	-	-	1.3	0.5	0.8

For landfills, the risk attenuation factors ranged from 6 to 40, with the lower attenuation factors mainly representing the more mobile constituents (i.e., those with lower soil sorption potential). Both the median and 10th percentile risk attenuation factors were adequate to reduce risks for all nine constituents below an HQ of 1.

For surface impoundments, risk attenuation factors were considerably lower, ranging from 1 to 9, reflecting higher contaminant mobility due to the higher hydraulic head in surface impoundments (as compared to landfills) and a lower proportion of liners. For the same reason, the screening HQs for surface impoundments were higher than the landfill HQs. As a result of this combination of higher HQs and lower risk attenuation factors, only the HQ for nickel was reduced to below 1 by applying the attenuation factors. The other constituents (chromium, fluoride, manganese, and vanadium) still show risks slightly above an HQ of 1, with HQs ranging from 1.4 to 3.5 at 10th percentile attenuation. This is consistent with the general trend in this analysis of surface impoundments showing higher risks than CCW landfills.

4.1.5 Human Health (Groundwater and Fish Consumption) Damage Case Review

Table 4-17 summarizes the proven damage cases from U.S. EPA (2007) that showed an impact on groundwater, usually through an exceedence of an MCL or state groundwater standard for one or more metals. As detailed in U.S. EPA (2007), these facilities represent worst-case disposal conditions: all are unlined, several represent fills in old quarries, and many have wastes disposed of below the water table. Groundwater standard exceedences are usually onsite or closely offsite. As one can see in the table, the same metals showing risk exceedences for unlined facilities (arsenic, boron, cadmium, lead, molybdenum, and selenium) in this analysis were reported as exceedences in the groundwater damage cases. Other incidents of groundwater

contamination supporting the conclusions of this risk assessment can be found in the published literature in references such as Lang and Schlictmann (2004) and Zilmer and Fauble (2004).

Table 4-18 summarizes the five proven damage cases from U.S. EPA (2007) that showed a fish consumption advisory for selenium. Although these were all cases where CCW surface impoundments directly discharged to a lake, and hence larger fluxes of surface impoundment waters into the waterbody of interest than through the groundwater-to-surface-water pathway, they do support the finding of this risk assessment that the fish consumption pathway is of potential concern for selenium in CCW.

Table 4-17. Summary of Proven Damage Cases with Groundwater Impacts (U.S. EPA, 2007)

Proven NODA Damage Case[a]	Reported Groundwater Impacts
1. Salem Acres Site, MA (lagoons and fly ash pile)	Minor – As, Cr, Pb
2. City of Beverly/ Vitale Brothers Fly Ash Pit, MA (quarry fill)	Al, As, Fe, Mn, Se over MCLs
3. Don Frame Trucking, Inc. Fly Ash Landfill, NY	Pb, Mn over MCLs
4. Virginia Electric Power Co. (VEPCO) Possum Point, VA (ash ponds)	Cd, Ni over MCLs
5. PEPCO Morgantown Generating Station Faulkner Off-site Disposal Facility, MD (landfills and settling ponds)	Low pH, iron staining
6. Virginia Power Yorktown Power Station Chisman Creek Disposal Site, VA (quarry fill)	Se, sulfate over MCL; green staining; As, Be, Cr, Cu, Mo, Ni, V over background
9. DOE Oak Ridge Y-12 Plant Chestnut Ridge Operable Unit 2, TN (ash pond)	Al, As, Fe, Pb, Mn over MCL
10. South Carolina Electric & Gas Canadys Plant, SC (ash ponds)	As above MCL outside compliance boundary; NI above state standard
13. Dairyland Power Cooperative E.J. Stoneman Generating Station Ash Disposal Pond, WI	Cd, Cr, sulfate, Mn, Fe, and Zn over MCLs onsite; B over background offsite
14. WEPCO Highway 59 Landfill, WI	As, Se, sulfate, B, Mn, Cl-, Fe over state standards
15. Alliant Nelson Dewey Ash Disposal Facility, WI	As, Se, sulfate, B, F- over state standards
16. WEPCO Cedar-Sauk Landfill, WI	Se, sulfate over MCLs; B over state standard
17. WEPCO Port Washington Facility, WI (quarry fill)	B over state standard; elevated Se
18. Lansing Board of Water & Light North Lansing Landfill, MI (quarry fill)	Li, Mn, Se above state standards
19. Northern Indiana Public Service Corp. Yard 520 Landfill Site, IN	As, B, Mn,, Mo, Pb contaminated residential wells
23. Basin Electric Power Cooperative W.J. Neal Station Surface Impoundment, ND	Al, As, Cd, Cr, Zn above MCL
24. Cooperative Power Association/United Power Coal Creek Station Surface Impoundments, ND	As, Se, sulfate, Cl above MCL; elevated B

[a] Numbers represent original case numbers in U.S. EPA (2007)

Table 4-18. Summary of Proven Damage Cases with Fish Consumption Advisories (U.S. EPA, 2007)

Proven NODA Damage Case	Reported Fish Consumption Advisory
7. Hyco Lake, Roxboro, North Carolina (surface impoundment discharge)	Selenium fish consumption advisory
11. Belews Lake, NC (surface impoundment discharge)	Selenium fish consumption advisory
20. Brandy Branch Reservoir, Texas (ash pond discharge)	Selenium fish consumption advisory
21. Southwestern Electric Power Company Welsh Reservoir, TX (ash pond discharge)	Selenium fish consumption advisory
22. Texas Utilities Electric Martin Lake Reservoir, TX (ash pond discharge)	Selenium fish consumption advisory; elevated selenium in birds

EPA has also found that CCW contaminants of concern in the damage cases agree with those exceeding a 1 in 100,000 excess cancer risk or an HQ of 1 in this analysis, building confidence that the risk assessment captures national conditions. **Table 4-19** compares the results from the 2007 draft risk assessment with the damage cases reported in the *Coal Combustion Waste Damage Case Assessments* (U.S. EPA, 2007) for the groundwater pathway.

Table 4-19. Modeled 90th and 50th Percentile Risk Results vs. Reported Groundwater Exceedences

Constituent	2007 Risk Assessment[a]		Damage Cases[b]			Consistent Results as of 2007[f]
	90th %ile	50th %ile	Human Health Effects[c]	Cosmetic/ Aesthetic Effects[d]	State Standard[e]	
Aluminum	–	–	–	✓	–	Yes
Antimony	✓	–	–	–	–	No
Arsenic	✓	T	✓	–	✓	Yes
Barium	–	–	–	–	–	Yes
Beryllium	Screened		–	–	–	Yes
Boron	✓	–	✓	–	✓	Yes
Cadmium	✓	–	✓	–	–	Yes
Chloride	Not Screened		–	✓	✓	N/A
Chromium	RAF		✓	–	–	Uncertain
Cobalt	✓	T	–	–	–	No
Copper	Screened		–	–	–	Yes
Fluoride	RAF		–	–	✓	Uncertain
Iron	Not Screened		–	✓	✓	N/A
Lead	✓	–	✓	–	–	Yes
Lithium	Not Screened		–	–	✓	N/A
Manganese	RAF		✓	✓	✓	Uncertain
Molybdenum	✓	–	✓	–	–	Yes
Nickel	RAF		✓	–	✓	Uncertain
Nitrate/Nitrite	✓	–	–	–	–	No
Selenium	✓	–	✓	–	✓	Yes
Silver	Screened		–	–	–	Yes
Sulfate	Not Screened		–	✓	✓	N/A
Thallium	✓	–	–	–	–	No
Vanadium	RAF		–	–	–	Uncertain

(continued)

Modeled 90th and 50th Percentile Risk Results vs. Reported Groundwater Exceedences (continued)

Constituent	2007 Risk Assessment[a]		Damage Cases[b]			Consistent Results as of 2007[f]
	90th %ile	50th %ile	Human Health Effects[c]	Cosmetic/ Aesthetic Effects[d]	State Standard[e]	
Zinc	Screened		–	✓	–	Yes

[a] Not Screened = Constituent was not considered due to lack of health-based benchmarks.
Screened = Constituent showed no risk potential in the screening assessment.
RAF = Constituent showed risk potential in the screening assessment, and was analyzed with risk attenuation factors.
✓ = Constituent underwent full probabilistic modeling and was shown to pose a risk to human health in the landfill scenario, the surface impoundment scenario, or both.
– = Constituent underwent full probabilistic modeling and was not shown to pose a risk to human health

[b] ✓ = At least one proven damage case showed an exceedence of this constituent.
– = No proven damage cases have yet shown an exceedence of this constituent.

c ✓ = Exceedences of primary maximum contaminant levels (MCLs) or other health-based numbers published by EPA.

[d] ✓ = Exceedences of secondary MCLs, which would not result in harm to human health.

e ✓ = Exceedences of a relevant state standard.

f Yes = Results of risk assessment and damage cases either both indicated a risk to human health or both indicated no risk to human health.
No = The risk assessment indicated risks where none have yet been found in a proven damage case.
Uncertain = It is possible that the results were consistent, but due to lack of probabilistic modeling, no definitive conclusion can be made.
N/A = Constituent was not examined at any stage in the 2007 risk assessment, so it was not possible to draw any conclusions as to consistency.

The first category of constituents is those for which the risk assessment and the damage cases agree, either because both the risk assessment results and the damage cases indicated risks, or because both the risk assessment results and damage cases did not indicate risks. The former group had model results exceeding the cancer risk range or an HQ of 1, and also appeared in the damage cases with exceedances of maximum contaminant levels (MCLs), state groundwater standards, or other health-based numbers (arsenic, boron, cadmium, lead, molybdenum, and selenium). The latter group did not show the potential for risks above an HQ of 1 from the risk assessment and did not appear in the damage case literature (aluminum, barium, beryllium, copper, silver, and zinc).

The second category of constituents is those for which the risk assessment and the damage cases did not agree. Four modeled constituents (antimony, cobalt, thallium, and nitrate/nitrite) showed risk at the 90th percentile but no damage cases had been proven as of 2007. This could indicate that (1) the risk assessment was conservative for these constituents, (2) not enough time has passed to see the remaining constituents appear in damage cases, (3) corrective action was taken when the first constituent(s) was observed, so further constituents that would have appeared at the same site were never seen, or (4) these constituents are not tested for as frequently as the constituents found in the proven damage cases.

The third category of constituents is those that were not screened out, and were analyzed using risk attenuation factors (chromium, fluoride, manganese, nickel, and vanadium). Because all that is known is that these constituents have the potential to pose a risk to human health, they cannot currently be compared to the damage case results.

The final category of constituents is those that were not evaluated at either the screening or modeling stages because no health-based values were available for comparison. These four constituents (chlorine, iron, lithium, and sulfate) appeared in damage cases because of exceedences of aesthetic or state levels, not because of a known risk to human health.

Table 4-20 compares the results from the 2007 draft risk assessment with the damage cases reported in U.S. EPA (2007) for the fish consumption pathway. The only fish consumption advisories documented in CCW damage cases are for selenium. This is consistent with the risk assessment for selenium. The two constituents that do not pose a risk in the risk assessment (cadmium and thallium) were also not part of any fish consumption advisories in the damage cases. The one inconsistency is arsenic, for which the risk assessment shows a cancer risk of 1 in 50,000, slightly exceeding an excess cancer risk of 1 in 100,000. However, no arsenic fish consumption advisories exist at proven damage case sites. This inconsistency could indicate that (1) the risk assessment was conservative with respect to arsenic, (2) not enough time has passed to see arsenic appear in fish advisories at these sites, or (3) the arsenic exceedences have not been detected in random fish tissue samples thus far.

Table 4-20. Modeled 90th and 50th Percentile Risk Results vs. Reported Fish Consumption Exceedences

Constituent	2007 Risk Assessment[a]		Damage Cases[b]	Consistent Results as of 2007[c]
	90th %ile	50th %ile		
Arsenic	✓	–	–	No
Cadmium	–	–	–	Yes
Selenium	✓	–	✓	Yes
Thallium	–	–	–	Yes

[a] ✓ = Constituent underwent full probabilistic modeling and was shown to pose a risk to human health in the landfill scenario, the surface impoundment scenario, or both.
– = Constituent underwent full probabilistic modeling and was not shown to pose a risk to human health

[b] ✓ = At least one proven damage case showed a fish consumption advisory for this constituent.
– = No proven damage cases have yet shown a fish consumption advisory for this constituent.

[c] Yes = Results of risk assessment and damage cases either both indicated a risk to human health or both indicated no risk to human health.
No = The risk assessment indicated risks where none have yet been found in a proven damage case.

4.2 Ecological Risks

EPA defines ecological risk characterization in terms of (1) the risk estimation, which integrates the exposure and stressor-response profile to estimate the likelihood of adverse ecological effects and (2) the risk description, which synthesizes the overall conclusion of the assessment and addresses assumptions, uncertainty, and limitations.

For assessments that are based on a HQ approach, as this one was, the comparison of modeled exposure concentrations to CSCLs to estimate risk has a binary outcome: either the constituent concentration is above the concentration corresponding to an HQ of 1 or the concentration is less than or equal to the concentration corresponding to an HQ of 1. For the full-scale analysis, an ecological HQ greater than 1 was selected by EPA as a criterion for decision making. Because the CSCLs were based on *de minimis* ecological effects, it is generally presumed that an HQ at or below 1 indicates a low potential for adverse ecological effects for those receptors included in the analysis for which data are available. However, it is important to recognize that although this method provides important insight into the potential for adverse ecological effects, the results are relevant only to those receptors that were included in the assessment and for which data were available. The results have limited utility in interpreting the ecological significance of predicted effects, and caution should be exercised in extrapolating to ecosystems (e.g., wetlands) and receptors (e.g., threatened and endangered species) not explicitly modeled.

This section presents risk results for direct surface impoundment exposure (as evaluated in the 1998 CCW risk assessment, U.S. EPA, 1998a,b), screening results for boron that indicate risks to plants from aboveground exposure, and the two groundwater-to-surface-water ecological exposure pathways investigated in the full-scale analysis: (1) receptors exposed to CCW constituents in the water column (surface water receptors) and (2) receptors exposed to CCW constituents in bed sediment (sediment receptors). Results are presented for the two WMU types addressed in the analysis: landfills and surface impoundments, and are broken out separately for the different unit (liner) types. Finally, ecological damage case reports from U.S. EPA (2007) and from the published literature are summarized as field evidence supporting the conclusions of this risk assessment.

The ecological risk results and damage cases suggest the potential for adverse ecological effects to plants, terrestrial organisms, and aquatic systems from CCW releases into the subsurface and subsequent connection with surface waters, particularly for CCW managed in unlined surface impoundments. As with human health risks, the higher prevalence of liners in newer facilities should result in lower risks in current and future CCW disposal facilities than those presented in this risk assessment.

4.2.1 Direct Surface Impoundment Exposure

The current risk assessment addresses exposure to receptors in offsite surface waterbodies impacted by groundwater, where both the aquatic communities and upper trophic level terrestrial receptors would need to be protected.[7] The 2003 CCW constituent database used

[7] The 2002 CCW constituent database does not include impoundment water samples, and the direct exposure pathway was not addressed.

in this analysis does not include impoundment water samples, and the direct exposure pathway could not be addressed for ecological risk. However, the CCW risk assessment conducted in 1998 (U.S. EPA, 1998a,b) did consider direct exposure of ecological receptors to surface impoundment waters. The approach in the 1998 study restricted the analysis to terrestrial receptors that obtain food and prey from the surface impoundments and excluded aquatic receptors living in the water column because surface impoundments are not intended to be a habitat for aquatic species. For the terrestrial and aquatic receptors considered, the 1998 analysis used the same CSCLs and a similar methodology to that used in the CCW screening analysis (e.g., comparison of 90th percentile waste concentrations with CSCLs).

The 1998 direct exposure results are provided in **Figure 4-1** and show HQs greater than 100 for boron, selenium, lead, barium, and cadmium. This, along with the damage case results presented in **Section 4.2.4**, show a clear likelihood of risks to terrestrial organisms that obtain food and prey from CCW surface impoundments. It is probable that ecological receptors eat and drink from CCW surface impoundments in some settings. In addition, ecological receptors, particularly amphibians who may lay their eggs in surface impoundments, are probably exposed through chronic contact with wastewater. Because amphibians are prey to a large variety of animals (e.g., raptors; wading birds; mammalian omnivores, such as foxes, raccoons, and weasels), this exposure is transferred up the food chain. Aquatic plants, although not often a focus of this ecological risk assessment, are directly exposed in surface impoundments. Plants, in turn, may be ingested by vertebrates and invertebrates at higher trophic levels.

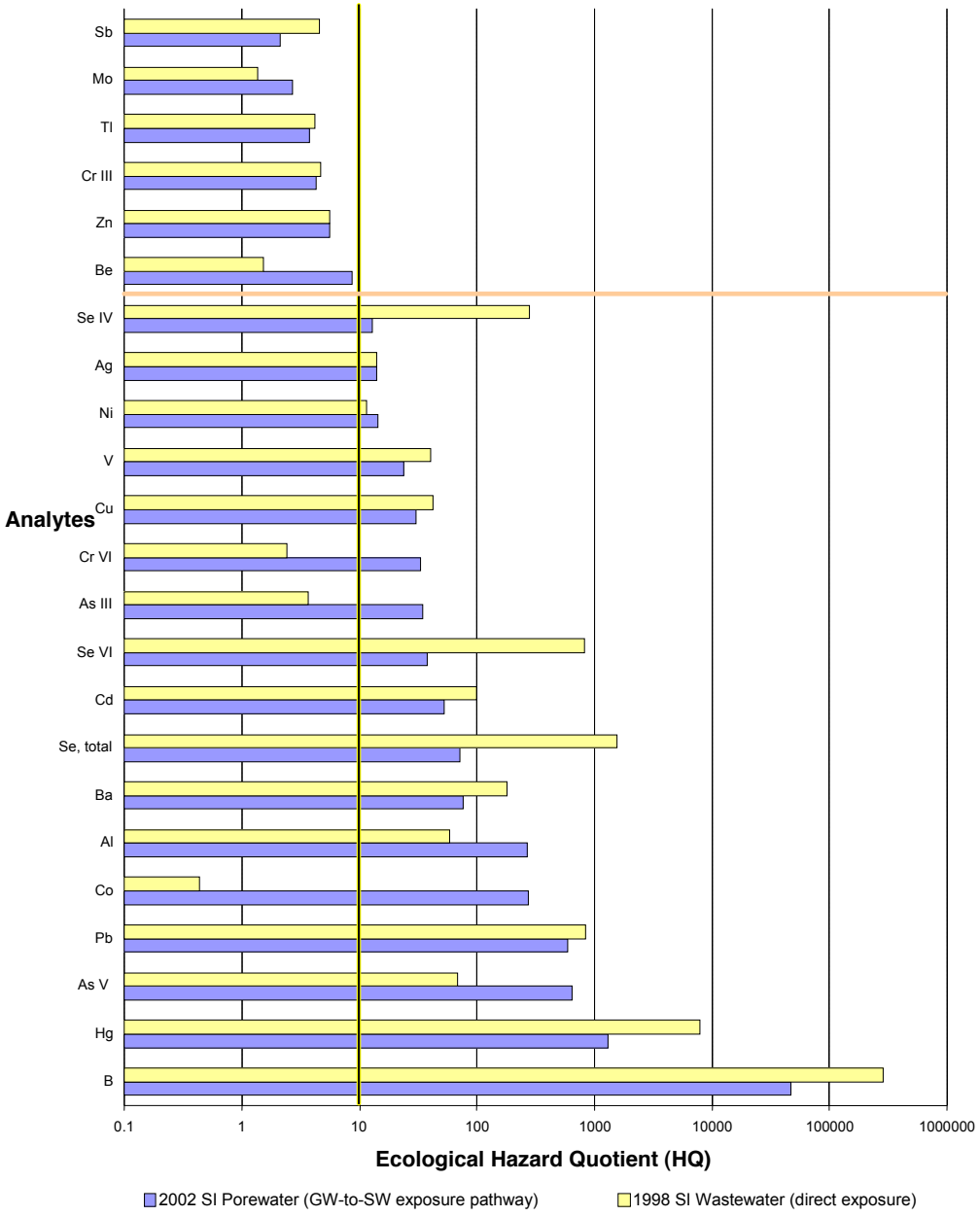


Figure 4-1. CCW surface impoundment ecological screening risks: Direct exposure to surface impoundment wastewater.

4.2.2 Surface Water Receptors (Full-Scale Analysis)

Tables 4-21 and **4-22** present the 90th and 50th percentile results from the full-scale ecological risk assessment of the groundwater-to-surface-water pathway for surface water receptors for CCW landfills and surface impoundments. For landfills, boron (HQ of 281) , lead (HQ of 8), selenium (HQ of 2), arsenic (HQ of 2), and barium (HQ of 2) show risks above an HQ of 1 at the 90th percentile for the unlined units. Clay liners reduce the risks below an HQ of 1 for all constituents except for boron, which still has a very high HQ (78 for the clay liner versus 281 for unlined). For surface impoundments, all modeled constituents except cadmium

and aluminum showed 90th percentile risks above the ecological risk criterion, with boron showing an HQ over 2,000 for the unlined units, and other HQs ranging from 3 to 22 for unlined units. The 50th percentile results are all well below an HQ of 1 for landfills and only exceed an HQ of 1 for boron in unlined surface impoundments (HQ = 7).

As with other pathways and receptors, the difference in the number and magnitude of ecological HQs that exceed the risk criterion between landfills and surface impoundments is likely the result of (1) higher CCW constituent concentrations in surface impoundment porewater and (2) the greater flux of contaminants to groundwater predicted during the active life of the surface impoundment. As discussed in **Section 4.1**, the higher infiltration rates for surface impoundments result from a higher hydraulic head in the impoundment and a higher proportion of unlined surface impoundments than landfills in the 1995 EPRI survey data used for this risk assessment.

Table 4-21. Summary of 90th Percentile Full-Scale CCW Ecological Risk Results: Groundwater-to-Surface-Water Pathway, Aquatic Receptors[a]

Chemical	90th Percentile Ecological HQ			Exposure Pathway	Receptor
	Unlined Units	Clay-Lined Units	Composite-Lined Units		
Landfills					
Boron	281	78	0.07	direct contact	aquatic biota
Lead	8	0.4	2E-06	ingestion	river otter
Selenium (VI)	2	0.7	3E-04	direct contact	aquatic biota
Arsenic (V)	2	0.1	4E-08	direct contact	aquatic biota
Barium	2	0.2	0	direct contact	aquatic biota
Cadmium	0.5	0.1	3E-05	direct contact	aquatic biota
Aluminum	0.01	0.003	1E-07	direct contact	aquatic biota
Surface Impoundments					
Boron	2,375	854	257	direct contact	aquatic biota
Lead	22	7	2	ingestion	river otter
Arsenic (V)	13	4	5	direct contact	aquatic biota
Selenium (VI)	12	4	1	direct contact	aquatic biota
Cobalt	6	3	5	direct contact	aquatic biota
Barium	3	1	0.8	direct contact	aquatic biota
Cadmium	1	0.7	0.4	direct contact	aquatic biota
Aluminum	0.03	0.01	0.008	direct contact	aquatic biota

[a] Zero results indicate that contaminant infiltration rates were too small for the contaminant plume to reach the receptor during the 10,000 year period of the analysis.

Table 4-22. Summary of 50th Percentile Full-Scale CCW Ecological Risk Results: Groundwater-to-Surface-Water Pathway, Aquatic Receptors[a]

Chemical	50th Percentile Ecological HQ			Exposure Pathway	Receptor
	Unlined Units	Clay-Lined Units	Composite-Lined Units		
Landfills					
Boron	0.2	0.1	0	direct contact	aquatic biota
Lead	7E-05	4E-08	0	ingestion	river otter
Selenium (VI)	0.002	0.001	0	direct contact	aquatic biota
Arsenic (V)	4E-06	5E-09	0	direct contact	aquatic biota
Barium	1E-10	4E-12	0	direct contact	aquatic biota
Cadmium	2E-04	9E-05	0	direct contact	aquatic biota
Aluminum	3E-07	8E-09	0	direct contact	aquatic biota
Surface Impoundments					
Boron	7	0.4	5E-05	direct contact	aquatic biota
Lead	0.05	0.0008	0	ingestion	river otter
Arsenic (V)	0.03	0.0007	0	direct contact	aquatic biota
Selenium (VI)	0.03	0.002	4E-07	direct contact	aquatic biota
Cobalt	0.01	0.001	0	direct contact	aquatic biota
Barium	0.006	0.0004	0	direct contact	aquatic biota
Cadmium	0.008	0.0003	0	direct contact	aquatic biota
Aluminum	0.0007	4E-05	4E-11	direct contact	aquatic biota

[a] Zero results indicate that contaminant infiltration rates were too small for the contaminant plume to reach the receptor during the 10,000 year period of the analysis.

4.2.3 Sediment Receptors (Full-Scale Analysis)

Tables 4-23 and 4-24 present the 90th and 50th percentile results of the ground-water-to-surface-water pathway for sediment receptors for landfills and surface impoundments. For unlined landfills, lead (HQ of 58), arsenic (HQ of 11), cadmium (HQ of 5), and antimony (HQ of 2) show 90th percentile risks above the ecological risk criterion. For clay lined landfills, only arsenic (HQ of 3) has an ecological HQ greater than 1. For surface impoundments, lead, arsenic, and cadmium showed 90th percentile HQs above 1 for unlined, clay-lined, and composite-lined units (with HQs ranging from 2 to 311). Although cadmium was not above the risk criterion in surface water, it did have an HQ of 30 in sediments at the 90th percentile for unlined surface impoundments and HQs of 9 and 2 for clay- and composite-lined impoundments respectively. None of the constituents modeled showed sediment risks at or above an HQ of 1 at the 50th percentile.

Table 4-23. Summary of 90th Percentile Full-Scale CCW Ecological Risk Results: Groundwater-to-Surface-Water Pathway, Sediment Receptors[a]

Chemical	90th Percentile Ecological HQ			Exposure Pathway	Receptor
	Unlined Units	Clay-Lined Units	Composite-Lined Units		
Landfills					
Lead	58	1	1E-06	direct contact	aquatic biota
Arsenic (III)	11	3	5E-04	ingestion	river otter
Cadmium	5	1	6E-05	direct contact	aquatic biota
Antimony	2	0.5	7E-05	direct contact	aquatic biota
Molybdenum	0.1	0.03	2E-05	direct contact	aquatic biota
Barium	0.006	6e-04	0	direct contact	aquatic biota
Surface Impoundments					
Lead	311	58	4	direct contact	aquatic biota
Arsenic (III)	127	55	31	ingestion	river otter
Cadmium	30	9	2	direct contact	aquatic biota
Molybdenum	0.9	0.3	0.1	direct contact	aquatic biota
Barium	0.008	0.004	0.002	direct contact	aquatic biota

[a] Zero results indicate that contaminant infiltration rates were too small for the contaminant plume to reach the receptor during the 10,000 year period of the analysis.

Table 4-24. Summary of 50th Percentile Full-Scale CCW Ecological Risk Results: Groundwater-to-Surface-Water Pathway, Sediment Receptors[a]

Chemical	50th Percentile Ecological HQ			Exposure Pathway	Receptor
	Unlined Units	Clay-Lined Units	Composite-Lined Units		
Landfills					
Lead	6E-05	9E-08	0	ingestion	spotted sandpiper
Arsenic (III)	4E-03	0.002	0	ingestion	spotted sandpiper
Cadmium	5E-04	2E-04	0	direct contact	sediment biota
Antimony	3E-04	1E-04	0	direct contact	sediment biota
Molybdenum	5E-05	3E-05	0	ingestion	spotted sandpiper
Barium	3E-13	8E-15	0	ingestion	spotted sandpiper
Surface Impoundments					
Lead	0.1	0.001	0	ingestion	spotted sandpiper
Arsenic (III)	0.4	0.02	4E-09	ingestion	spotted sandpiper
Cadmium	0.02	0.0007	0	direct contact	sediment biota
Molybdenum	0.004	0.0002	2E-08	ingestion	spotted sandpiper
Barium	1E-05	1E-06	0	ingestion	spotted sandpiper

[a] Zero results indicate that contaminant infiltration rates were too small for the contaminant plume to reach the receptor during the 10,000 year period of the analysis.

4.2.4 Constituents Not Modeled in the Full-Scale Assessment

As described in **Section 3.2.4**, full-scale modeling was not conducted for 6 constituents with generally lower risks to ecological receptors.[9] These chemicals (chromium, vanadium, beryllium, copper, silver, and zinc), had surface water pathway HQs in the screening analysis ranging from 16 to 110 for landfills, and four (chromium, vanadium, copper, and silver) had screening HQs ranging from 14 to 33 for surface impoundments.

These constituents were addressed using risk attenuation factors developed by dividing the screening risk results by the full-scale risk results for the constituents that were modeled in the full-scale assessment. **Tables 4-25** and **4-26** show the results of this comparison for the surface water ecological risk exposure pathway. Table 4-23 shows the risk attenuation factors for the modeled constituents, and Table 4-24 shows the results of applying the median (central tendency) and 10th percentile (conservative) attenuation factors to the screening risk results for constituents that were not modeled.

For landfills, the risk attenuation factors ranged from 50 to 2,000. Both the median and 10th percentile risk attenuation factors were adequate to reduce risks to an HQ below 1 for all constituents except for silver. Although silver shows an HQ of 1.5 using the 10th percentile attenuation factor, silver's low mobility would probably result in a higher attenuation factor (i.e., at the median or greater).

For surface impoundments, risk attenuation factors ranged from 7.1 to 64, reflecting higher contaminant mobility from the higher hydraulic head in the surface impoundments and a lower prevalence of liners (compared to landfills) in the 1995 EPRI data. HQs were reduced below 1 for all four unmodeled constituents with the median attenuation factor (38), and the HQ for silver was reduced to 0.8 by applying the 10th percentile attenuation factor (17). The other three constituents (chromium, vanadium, and copper) show HQs slightly above 1 with the10th percentile attenuation (HQs ranging from 1.4 to 1.9). Note that the risks for chromium are based on the protective assumption of 100 percent hexavalent chromium in CCW.

Table 4-25. Risk Attenuation Factor[a] Statistics for Modeled Constituents— Ecological Risk, Surface Water Pathway (all unit types combined)

Statistic	Landfill	Surface Impoundment
10th percentile	75	17
50th percentile	178	38
Average	483	38
Maximum	2,000	64
Number of data points	6	7

[a] The risk attenuation factor is the ratio of the full-scale analysis risk and screening analysis risk for a constituent modeled in the full-scale analysis.

[9] These constituents had only one or no ecological HQs greater than 100.

Table 4-26. Summary of Risk Screening Values for Unmodeled Constituents Using Risk Attenuation Factors—Ecological Risk, Surface Water Pathway

WMU/Pathway	Landfill			Surface Impoundment		
	Screening HQ	HQ with Median Attenuation	HQ with 10th Percentile Attenuation	Screening HQ	HQ with Median Attenuation	HQ with 10th Percentile Attenuation
Chromium VI	18	0.1	0.2	33	0.9	1.9
Vanadium	23	0.1	0.3	24	0.6	1.4
Beryllium	24	0.1	0.3	-	-	-
Copper	16	0.09	0.2	31	0.8	1.8
Silver	110	0.6	1.5	14	0.4	0.8
Zinc	16	0.09	0.2	-	-	-

4.2.5 Ecological Damage Cases

Cases of damages to terrestrial and aquatic organisms from improperly managed CCW are common in the literature. For example, Carlson and Adriano (1993) summarize such damage incidents, including those resulting from alkaline CCW effluent discharge to surface waterbodies and boron toxicity to plants. Rowe et al. (2002) provide a more comprehensive review, assessment, and meta-analysis of the ecotoxicity of CCW, focusing on aquatic disposal (i.e., CCW surface impoundments) and tabulating damages from over 20 years of field and laboratory studies in the published literature. Selenium and arsenic are most commonly associated with CCW damages to terrestrial and aquatic organisms. Cadmium, boron, chromium, and lead are also associated with CCW ecological risk. Hopkins et al. (2006) show deformities and reproductive effects in amphibians living on or near CCW disposal sites in Georgia, which are mainly attributed to selenium exposure.

Table 4-27 summarizes the proven CCW ecological damage cases from U.S. EPA (2007). Most of these cases are from surface impoundments and direct discharge into lakes and other water bodies. Along with the published results discussed in **Section 4.1.5**, these cases clearly support selenium and arsenic in coal ash as risks to aquatic ecosystems, as well as the adverse impacts of coal ash on terrestrial vegetation.

Table 4-27. Summary of Proven Damage Cases with Ecological Impacts (U.S. EPA, 2007)

Proven NODA Damage Case	Reported Ecological Impacts
2. City of Beverly/ Vitale Brothers Fly Ash Pit, MA (quarry fill)	Contamination of wetlands and surface waters
5. PEPCO Morgantown Generating Station Faulkner Off-site Disposal Facility, MD (landfills and settling ponds)	Vegetative damages, contamination of stream and wetland by GW
6. Virginia Power Yorktown Power Station Chisman Creek Disposal Site, VA (quarry fill)	As, Be, Cr, Cu, Mo, Ni, Se, V contamination of onsite ponds and offsite creek
7. Hyco Lake, Roxboro, North Carolina (surface impoundment discharge)	Se fish advisory; fish reproduction and population effects
8. Georgia Power Company, Plant Bowen, Cartersville, GA (ash pond over sinkhole)	Ash slurry release damaged creek
9. DOE Oak Ridge Y-12 Plant Chestnut Ridge Operable Unit 2, TN (ash pond)	Se, As, Tl elevated in bass; As over screening criteria; deformed fish; stress on aquatic ecosystem; Se plant and mammal uptake
11. Belews Lake, NC (surface impoundment discharge)	Fish advisory for Se; 16 of 20 fish species eliminated from lake
12. U.S. Department of Energy Savannah River Project, SC (landfill)	Impacts on amphibians (deformities) and snake (metabolic effects)
16. WEPCO Cedar-Sauk Landfill, WI	Wetland vegetative damage from B in groundwater
20. Brandy Branch Reservoir, Texas (ash pond discharge)	Se fish consumption advisory
21. Southwestern Electric Power Company Welsh Reservoir, TX (ash pond discharge)	Se fish consumption advisory
22. Texas Utilities Electric Martin Lake Reservoir, TX (ash pond discharge)	Se fish consumption advisory; elevated Se in birds

4.3 Sensitivity Analysis

EPA conducted a sensitivity analysis (U.S. EPA, 2009b) on the probabilistic risk assessment to determine which model inputs were most important to risk, which in turn helped focus additional analyses and data collection efforts on the most important drivers of risk, and helped identify the important factors to consider when evaluating regulatory and management options for CCW. The sensitivity analysis also helped identify parameters that are both sensitive and highly uncertain, which affects the confidence in the results.

The CCW sensitivity analysis used a response-surface regression method that derives a statistical model for risk (as the dependent variable) based on the input parameters from the probabilistic analysis (as independent variables). Environmental concentration (rather than risk) was chosen as the dependent variable for the sensitivity analysis because (1) there is a direct, linear relationship between environmental concentrations and risks and (2) the additional inputs used to calculate risk from environmental concentration (i.e., exposure factors, such as body weight, ingestion rates) are lifestyle variables that are not amenable to regulation to reduce or manage risk. Furthermore, these variables have well-established, peer-reviewed, national

distributions, which are regularly used in the probabilistic national risk analyses conducted by EPA. Therefore, the contribution of the exposure factors to the variability in risk was not particularly useful for the primary purposes of the sensitivity analysis, to better understand sources of uncertainty in the CCW risk results and to help focus regulatory development on sensitive variables that can be addressed through the RCRA regulatory process.

The outputs from the sensitivity analysis were goodness-of-fit values for the regression models and the relative importance of each input parameter in determining environmental concentrations across different WMU, waste type, and constituent scenarios. The goodness-of-fit values of the regression models were moderate to very good for the drinking water pathway (R^2=0.53–0.90) and good to very good for fish consumption (R^2=0.76–0.90). In general, the drinking water pathway had more input parameters that were significant (seven) than the fish consumption pathway (three). The most sensitive parameters for most (over 70 percent) of the drinking water scenarios[10] evaluated were parameters impacting groundwater flow:

- Infiltration rate within the WMU footprint
- Leachate concentration from the WMU
- Aquifer hydraulic conductivity and groundwater gradient (i.e., groundwater velocity).

For many (over 30 percent) of the scenarios, including those corresponding to strongly sorbing contaminants (i.e., metals with high soil/water partition coefficients), sorption and travel time parameters are also important, including

- Adsorption isotherm coefficient
- Depth to groundwater
- Receptor well distance.

For the fish consumption pathway, only three variables were consistently significant across scenarios:

- Infiltration rate within the WMU footprint
- Leachate concentration from the WMU
- Waterbody flow rate.

Additional detail on how the CCW sensitivity analysis was conducted can be found in U.S. EPA (2009b). In terms of the model inputs, the sensitivity analysis found that the most consistent drivers of the risk results were constituent concentration in waste leachate (i.e., the source term for the risk assessment and infiltration rate through the WMU), which is largely controlled by the liner conditions and, to a lesser extent, soil type and (for landfills only) precipitation. These variables and their uncertainties are discussed in the following section.

[10] Scenarios represent unique combinations of WMU, waste type, chemical, exposure pathway, and receptor.

4.4 Variability and Uncertainty

Variability and uncertainty are different conceptually in their relevance to a probabilistic risk assessment. Variability represents true heterogeneity in characteristics, such as body weight differences within a population or differences in pollutant levels in the environment. It accounts for the distribution of risk within the exposed population. Although variability may be known with great certainty (e.g., age distribution of a population may be known and represented by the mean age and its standard deviation), it cannot be eliminated and needs to be treated explicitly in the assessment. Uncertainty is a description of the imperfection in knowledge of the true value of a particular parameter. In contrast to variability, uncertainty can be reduced through additional information-gathering or analysis (i.e., better data, better models). EPA typically classifies the major areas of uncertainty in risk assessments as scenario uncertainty, model uncertainty, and parameter uncertainty. Scenario uncertainty refers to missing or incomplete information needed to fully define exposure and dose. Model uncertainty is a measure of how well the model simulates reality. Parameter uncertainty is the lack of knowledge regarding the true value of a parameter used in the assessment.

Variability arises from true heterogeneity in characteristics, such as body weight differences within a population or differences in contaminant levels in the environment.

Uncertainty represents a lack of knowledge about factors such as the nature of adverse effects from exposure to constituents, which may be reduced with additional research to improve data or models.

Uncertainty and variability can be addressed two ways:

- By varying parameter values in a probabilistic assessment such as a Monte Carlo analysis
- By comparing the data or results to other data or other studies such as damage cases or alternative results based on different assumptions.

In planning this assessment, EPA addressed as much of the variability as possible, either directly in the Monte Carlo analysis or through aggregation of the data into discrete elements of the analysis. For example, spatial variability in soil, aquifer, and climate data was accounted for by using distributions for soil and aquifer properties around the facility when the actual environmental characteristics around a WMU are uncertain. Conversely, variability in waste leachate concentrations was represented by a national database of CCW constituent concentrations from disposal sites around the country. These data were aggregated by waste and WMU types that were defined by statistically significant differences in concentration. Variability in human exposure factors (e.g., body weight, ingestion rates) was accounted for using national distributions that represent the range of possible values.

Because CCW is generated nationwide, its disposal may occur anywhere in the United States. Thus, this assessment characterized environmental conditions that influence the fate and transport of constituents in the environment using site-specific data collected around coal-fired power plants with onsite CCW disposal facilities. Spatial variability in environmental setting was accounted for by the site-to-site variables for the 181 CCW disposal sites modeled in the

analysis using 41 different climate regions and 9 different hydrologic regions throughout the contiguous 48 states.

In summary, a distribution of exposures was developed that included specific consideration of the variability in the following sensitive model parameters

- WMU characteristics, in particular liner type (which strongly influences infiltration rate)
- CCW constituent concentrations in waste leachate
- Distance to nearest well
- Site-specific environmental conditions (especially groundwater flow conditions)
- Human exposure factors.

Uncertainty was also considered in the analysis by using reasonable ranges and distributions when variables were not known exactly. For example, when a soil texture or groundwater flow conditions could not be precisely assigned at a site, multiple soil types or hydrogeologic environments were sampled based on the soil and aquifer types that were likely to be present at the site.

The treatment of variability and uncertainty in model parameters using a Monte Carlo simulation formed the basis for the national exposure distributions used in this analysis to estimate risk. Previous sections of this document describe how EPA generated distributions and estimated input parameter values and then used these values in models to estimate risk. The discussion in this section focuses on how this treatment of variability and uncertainty affects the analysis results and on various comparisons we performed on the results or critical input data to evaluate uncertainty. **Table 4-28** lists the more important uncertainties described in this section, along with whether the uncertainty is likely to underestimate or overestimate risk, or if its effect on the risk results is uncertain.

Table 4-28. Summary of CCW Uncertainties and Their Effect on Risk Estimates

Uncertainty	Likely Effect on Risk		
	Overestimates	Uncertain	Underestimates
Scenario Uncertainties			
CCW Management Unit Data (1995 EPRI Survey)		✓	
Liner type (as built, 1995; liners more prevalent today)	✓		
Direct discharge from CCW impoundments (not addressed in CCW risk assessment; covered by NPDES)			✓
Effect of the 10,000-year timeframe for groundwater (complete leaching, long timeframe)	✓		
Receptor populations evaluated (high-end receptor and child living near CCW WMU)	✓		
Additive risks across pathways (not considered)			✓
Co-occurrence of ecological receptors and constituents	✓		
Ecosystems and receptors at risk		✓	

(continued)

Summary of CCW Uncertainties and Their Effect on Risk Estimates (continued)

Uncertainty	Likely Effect on Risk		
	Overestimates	**Uncertain**	**Underestimates**
Model Uncertainties			
Clean closure of surface impoundments			✓
Arsenic and selenium speciation	✓		
100% bioavailability of constituents to ecological receptors	✓		
Compaction of landfilled waste	✓		
Landfills above water table			✓
Indirect ecological effects (not considered)			✓
Full mixing effects on aquifer pH (full mixing assumed; effect depends on constituent)	✓	✓	✓
Goethite versus hydrous ferric oxide sorbent	✓		
Multiple constituent exposures (not considered)			✓
Parameter Uncertainties			
Waste concentrations (2002 CCW constituent database)		✓	
Appropriateness of leachate data (TCLP results)		✓ (noncancer)	✓ (cancer)
Constituents with many nondetect analyses (e.g., mercury)		✓	
Treatment of nondetect analyses at half detection limit		✓	
WMU locations (1995 EPRI survey data)		✓	
WMU characteristics (1995 EPRI survey liner types, unit sizes)	✓		
Well location (MSW landfill survey data)		✓	
Well location (well always within plume)	✓		
Location and characteristics of waterbodies		✓	
Soil and aquifer characteristics		✓	
Waterbodies intercepting the groundwater plume	✓		
Human exposure factors		✓	
All drinking water from CCW-contaminated well)	✓		
Human health benchmarks	✓ (cancer)	✓ (noncancer)	
Ecological benchmarks		✓	

4.4.1 Scenario Uncertainty

Sources of scenario uncertainty include the assumptions and modeling decisions that are made to represent an exposure scenario. Because this risk assessment attempted to characterize current conditions by estimating risks from actual CCW disposal sites across the country, it was subject to less scenario uncertainty than risk assessments that rely on hypothetical conceptual models. However, certain aspects of the scenario are uncertain.

CCW Management Unit Data. The landfills and surface impoundments modeled in this risk assessment were placed, sized, and lined according to data from the 1995 EPRI survey (EPRI, 1997). New data collected by EPA and DOE since this risk assessment was conducted (U.S. DOE, 2006) indicate that liners are much more prevalent in WMUs constructed or expanded from 1994 through 2004 than in units in place before that. This suggests that the risks

may be lower for future CCW disposal facilities (although most of the unlined WMUs have been closed with wastes remaining in the units).

Liner-related questions are especially important because liner configurations greatly influence infiltration rates, one of the most sensitive parameters in the risk assessment. In terms of risks through groundwater pathways, this risk assessment has shown that liners, in particular composite (combined clay and synthetic) liners, can limit risks through subsurface exposure pathway, and the DOE/EPA survey shows that liners are more prevalent in newly constructed WMUs and WMU expansions. Although the DOE/EPA survey does not shed light on how many unlined facilities are still operating today, it does indicate that more units are lined today than were in the 1995 EPRI survey data set on which this risk assessment was based.

Although it would have been possible to address this uncertainty by evaluating different hypothetical liner scenarios for each facility, such an approach was outside the original scope of this risk assessment, which was to evaluate current CCW management activities, not hypothetical management scenarios. Furthermore, this approach likely would not have changed the general conclusion of the risk assessment that composite lined landfills pose less risk than clay lined landfills and that unlined landfills pose the greatest risk.

Direct Discharge of CCW Impoundments into Surface Water. Because this risk assessment addressed CCW disposal under RCRA, it did not include risks from the direct discharge of wastes into waterbodies, which are regulated under the Clean Water Act. Although not relevant for the management of RCRA waste disposal, EPA recognizes that CCW surface impoundment effluent may pose additional risks.

Effect of the 10,000-Year Timeframe for Groundwater. The risk assessment assumed that contaminant concentrations in the leachate remain constant throughout the 10,000-year modeling timeframe, although leaching may or may not persist for 10,000 years, depending on model inputs. The waste concentration model input was assumed to be a portion of the total waste concentration available to be leached, and it was assumed that 100% of the constituent in the waste could leach from the landfill. The nonlinear fate and transport solution used for metallic constituents in the unsaturated zone module of EPACMTP is based on the assumption that the leachate concentration released from the waste management unit is constant over time (see Section 3.3.5.3 of U.S. EPA, 2003b). Although a leaching profile that changes over time might be more realistic, the simplified leaching profile used by the model does not lead to a poorer estimate of risk associated with groundwater exposures. The adoption of a simplified leaching profile to support a non-linear sorption approach in the unsaturated zone offered a greater benefit and defensibility to the overall approach than assuming linear partitioning and a depleting leachate profile would have.

Receptor Populations Evaluated. The human receptors evaluated for the CCW risk analysis were a family with children residing near the CCW disposal facility, drinking from a private well screened in a surficial aquifer or eating fish caught from a nearby stream or lake impacted by CCW leachate. Additionally, except for a 15-day vacation, it was assumed that adults and children were exposed daily and that the private well was the only source of drinking water. Although it is possible for other types of individuals to be exposed, the use of the resident adult and child as protective of other receptors and pathways is a high-end, simplifying

assumption of the analysis. The lack of information to define and model actual exposure conditions also introduces uncertainty into this assessment, but EPA believes that the national distribution of exposure factors used is appropriate for a national assessment.

In addition, not all possible exposure pathways were evaluated. For example, the risk assessment did not consider potential indirect exposure to humans through game species that may have been exposed to surface impoundment waste (e.g., deer drinking surface impoundment water). This represents a potential uncertainty in the analysis.

Additive Risks Across Pathways. The human receptors evaluated in the CCW risk assessment were assumed not to consume both contaminated fish and drinking water from the same waterbody because untreated surface water is not considered potable water (municipal water treatment facilities were assumed to reduce contaminant levels prior to consumption). EPA also did not consider the potential cumulative exposure from contaminated fish and groundwater in the CCW risk assessment, because the exposures are likely to occur over different timeframes (because of differences in transit time of the contaminant plume to wells versus surface waterbodies) and may involve different receptors (because a resident near a CCW surface impoundment or landfill and exposed via groundwater may not be a recreational fisher). Although this could potentially miss some higher exposures for a maximally exposed individual, analysis of the individual pathway results does not indicate that adding such risks would change the conclusions of this risk assessment in terms of the constituents exceeding the risk criteria. Also, risks were high enough for single chemicals for human exposure pathways (notably arsenic) that this would not change the basic conclusion of the risk assessment that there are potentially significant risks to human health from CCW disposal in landfills and surface impoundments.

Co-Occurrence of Ecological Receptors and Constituents. As a simplification for national-scale analyses in the absence of site-based data, co-occurrence of the ecological receptors and the constituents of concern is typically assumed. However, the prior probability that a receptor will be found in waterbodies affected by constituent releases from CCW WMUs is not known, nor is it known whether a receptor will forage for food in contaminated areas or if those areas do, in fact, support the type of habitat needed by the receptor. Although the assumption of co-occurrence was necessary for this analysis, relatively few field studies are available to demonstrate the relationship between adverse ecological effects and constituent releases from CCW as it is currently managed.

Ecosystems and Receptors at Risk. One challenge in conducting a predictive ecological risk assessment intended to reflect risks at a national scale is representing *all* of the receptors and ecosystems at risk. In *Wastes from the Combustion of Coal by Electric Utility Power Plants - Report to Congress* (U.S. EPA, 1988b), the authors pointed out that plants or animals of concern were located within a 5-km radius of the CCW WMUs at 12 to 32 percent of the sites. Although these figures are of limited spatial resolution, they suggest the possibility that threatened and endangered species or critical habitats may be at risk from CCW constituents. Examples of other critical assessment endpoints not evaluated in this analysis include the following:

- **Managed Lands:** Because protected lands play a critical role in preserving plant and animal species, managed areas in the United States represent well-recognized ecological

values. Managed lands refer to a variety of lands designated by the federal government as worthy of protection, including National Wildlife Refuges, National Forests, Wilderness areas, and National Recreation areas.

- **Critical Habitats:** Although critical habitats may be defined in a number of ways (e.g., presence of threatened species, decreasing habitat area), wetlands are widely recognized as serving critical ecological functions (e.g., maintenance of water quality). The U.S. Fish and Wildlife Service estimates that approximately 45 percent of the Nation's threatened and endangered species directly depend on aquatic and wetland habitats. Consequently, impacts of chemical stressors on wetland habitats may have high ecological (and societal) significance. The presence of critical habitats such as wetlands is also used to inform the selection of ecological receptors (e.g., amphibians, waterfowl) and the construction of appropriate food webs.

- **Threatened and Endangered Species:** For most ecological risk assessments of chemical stressors, available data on toxicity and biological uptake are sufficient to support the evaluation of effects on representative species populations or generalized communities (e.g., the aquatic community). However, despite their obvious value, threatened and endangered species are frequently excluded from the analytical framework for national rulemakings. The assessment of threatened and endangered species requires a site-specific approach in which locations, habitats, and species of concern are identified and characterized with respect to the spatial scale of constituent releases.

Although these classes of receptors and potential ecological hazards are not explicitly considered in the analysis, conditions represented by simulations in the upper end of the risk distribution (higher risk scenarios) should reasonably characterize many situations with such sensitive species or habitats.

Impact on Groundwater as a Resource. The risk assessment did not explicitly consider potential impacts on the availability of groundwater in the future (e.g., contaminated groundwater becoming unsuitable for consumption), but the results do clearly indicate that there can be a reduction in resource availability if CCW is improperly disposed. However, the scope of the risk assessment was to evaluate human health and ecological effects associated with current waste disposal practices and conditions, and a quantitative evaluation of potential future reductions in groundwater availability as a consequence of CCW disposal practices was not conducted as part of this analysis.

4.4.2 Model Uncertainty

Model uncertainty is associated with all models used in a risk assessment because models and their mathematical expressions are simplifications of reality that are used to approximate real-world conditions and processes and their relationships. Computer models are simplifications of reality, requiring exclusion of some variables that influence predictions but that cannot be included in models either because of their complexity or because data are lacking on a particular parameter. 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. Because this was a probabilistic assessment that predicted

what may occur with the management of CCW under actual scenarios, it is possible to compare the results of these models to specific situations.

The risk assessor needs to consider the importance of excluded variables on a case-by-case basis, because a given variable may be important in some instances and not important in others. A similar problem can occur when a model that is applicable under one set of conditions is used for a different set of conditions. In addition, in some instances, choosing the correct model form is difficult when conflicting theories seem to explain a phenomenon equally well. In other instances, EPA does not have established model forms from which to choose to address certain phenomena, such as facilitated groundwater transport.

The models used in this analysis were selected based on science, policy, and professional judgment. These models were selected because they provide the information needed for this assessment and because they are generally considered to reflect the state of the science. Even though the models used in this analysis are used widely and have been accepted for numerous applications, they each retain significant sources of uncertainty. These limitations are well documented in the model development references cited in **Section 3**.

Although the sources of model uncertainty in this assessment could result in either an overestimation or an underestimation of risk, the models used in this assessment have been developed over many years to support regulatory applications. As a result, they have been designed to be protective of the impacted populations that they represent. In other words, where simplifying assumptions are necessary, the assumptions are made in a way that will not underestimate risk.

Assumption of Clean Closure of Surface Impoundments. As described in **Section 3.5.1**, the surface impoundment model treats a surface impoundment as a temporary waste management unit with a set operational life. At the end of this life, clean closure is assumed; all wastes are removed and there is no further release of waste constituents to groundwater. Although this simplifying assumption is not consistent with the practice to close CCW surface impoundments with wastes in place, and it limits the length of potential exposure, the peak annual leachate concentrations on which the CCW risk results were based are not likely to be affected. Releases to groundwater are much higher during surface impoundment operation because the higher hydraulic head in an operating impoundment drives wastewater into the underlying soil with greater force than infiltration through the impoundment cover after the impoundment is closed. This higher head results in a greater flux of contaminants to groundwater during the active life of the surface impoundment, especially in unlined units. Thus, even if the post-closure period were modeled, the corresponding results would not be as high as the peak annual leachate concentrations used in the analysis.

Arsenic and Selenium Speciation. Because the models used in this assessment do not speciate metals during soil or groundwater transport, arsenic and selenium speciation in the subsurface is a significant groundwater modeling uncertainty in this analysis. Arsenic can occur in either a +3 (arsenic III) or +5 (arsenic V) oxidation state in groundwater, with arsenic III being the more mobile form. Selenium can occur in either a +4 (selenium IV) or +6 (selenium VI) oxidation state in groundwater, with selenium VI being the more mobile form. Because the soil and groundwater models assume one form for each model run, the risk results presented for

arsenic and selenium were originally based on 100% arsenic III and selenium VI, which is a high-end assumption (i.e., arsenic III has higher risks than arsenic V and selenium VI has higher risks than selenium IV). Although arsenic is generally thought to occur in the +3 form in leachate, there is evidence from damage cases at CCW disposal sites that suggests that arsenic III is converted to arsenic V during subsurface transport at some sites (see, for example, U.S. EPA, 2000, 2003e; Lang and Schlictmann, 2004; Zillmer and Fauble, 2004). To address the uncertainty of running the model with 100% arsenic III and selenium VI, the models were also run assuming 100% arsenic V and selenium IV. The results from the two species should bracket the results expected given some mixing of oxidation states.

Bioavailability of Constituents to Ecological Receptors. For the purposes of this analysis, the model assumed that all forms of a constituent were equally bioavailable to ecological receptors, and therefore, the actual exposures that may occur in the field tend to be overestimated, thus making this a high-end assumption. Both the chemical form and the environmental conditions influence bioavailability and ultimately the expression of adverse effects. For example, as discussed above, the form of arsenic has been shown to profoundly influence mobility and toxicity.

Compaction of Landfilled Waste. The source model did not consider potential compaction of CCW waste over time. Such compaction could decrease the hydraulic conductivity and the associated water infiltration. However, no readily available data were identified to support an analysis of the influence of CCW compaction on infiltration rates. The current approach would tend to overestimate infiltration rates compared to a model that would adjust the hydraulic conductivity over time due to compaction. EPA believes this is an appropriately conservative assumption given the lack of the information needed to accurately model the effects of waste compaction.

Landfills Assumed to be Above Water Table. The landfill source model and EPACMTP assume that the source is above the water table. However, some actual CCW disposal units do extend below the water table. Because waste intersecting the saturated zone may increase groundwater concentrations, the approach may underestimate risk in some cases. However, including this effect would strengthen a general conclusion of the analysis that potentially unacceptable risks exist in some cases with unlined and clay lined CCW landfills.

Indirect Ecological Effects. Indirect ecological effects (e.g., depletion of food resources) were not considered in the analysis. For any given facility, the spatial scale of potential contamination would affect a very small proportion of the home range for typical species; determining impacts on food supply and habitat quality with regard to the landscape and overall health of the animals is not currently possible in a national-level assessment (and difficult to understand or estimate in the majority of site-specific assessments). In addition, many species are opportunistic feeders and will seek other areas if food sources decline, regardless of the source of the stress to the food supply. For these reasons, EPA does not believe that it is possible to consider indirect ecological effects in a national risk assessment like CCW.

Aquifer pH. As explained in **Section 3.4**, aquifer pH was used to select the metal sorption coefficients that were in turn used to calculate retardation coefficients for groundwater transport of the CCW constituents. To estimate pH in an aquifer impacted by CCW leachate, the

CCW risk analysis assumed that, after entering the aquifer, the leachate plume thoroughly mixes with the ambient, uncontaminated groundwater. However, because this mixing zone is largely at the periphery of the groundwater plume, thorough mixing may or may not occur at actual sites. The full mixing assumption results in higher receptor point concentrations for most metals, because metal sorption and precipitation tend to increase (i.e., Kd goes up) with higher pH and full mixing tends to reduce the pH of CCW leachate, which is normally alkaline (i.e., assuming full mixing results in a lower groundwater pH and lower sorption).

To assess the effect of this simplifying assumption on the risk results, we compared two landfill Monte Carlo simulations for coal ash waste containing As(III) and coal ash waste containing As(V): (1) the fully mixed aquifer assumption and (2) an assumption that no mixing occurs in the aquifer and the leachate pH is the governing pH for Kd selection. These two metal species were selected because their sorption isotherm behavior with pH change differs; Kds derived from As(III) isotherms tend to decrease as pH increases (which is typical of most metal species examined in the risk assessment), while Kds derived from As(V) isotherms tend to increase with increasing pH.

Percentiles of peak receptor well concentration from the As (III) and As (V) simulations were selected and compared by calculating the percent change with mixing assumption as follows:

$$\%\,Change = \frac{C_{\text{No Mix}} - C_{\text{Full Mix}}}{C_{\text{Full Mix}}} \times 100$$

where

$C_{\text{No Mix}}$ = Simulated peak receptor well concentration for a select percentile based on a no mixing assumption (mg/L)

$C_{\text{Full Mix}}$ = Simulated peak receptor well concentration for a select percentile based on a fully mixed assumption (mg/L)

Table 4-29 compares the percent change in peak receptor well As (III) and As (V) concentrations between the well mixed and no mixing scenarios over a range of peak well percentiles. The results indicate that As(V) has a sensitivity to pH that leads to increased receptor well concentrations under the no mixing assumption (i.e., when the leachate pH is used to determine Kd in the saturated zone) relative to the well-mixed assumption used in the risk assessment. These results suggest that a change in the complete leachate mixing assumption could raise the receptor well concentrations (and therefore risks) for metal constituents whose Kd values decrease with increasing pH.

Table 4-29. Change in Peak Receptor Well Concentrations for Ash Disposed in Landfills Assuming Leachate Does Not Mix in Aquifer

Percentile of Peak Concentration	Percent Change in Peak Concentration	
	As(III)	As(V)
10	0.00%	0.00%
20	0.00%	0.00%
30	0.91%	0.00%
40	0.25%	0.00%
50	0.31%	2.28%
60	0.00%	15.57%
70	0.23%	57.97%
80	0.00%	18.31%
90	0.00%	11.75%

Goethite Versus Hydrous Ferric Oxide Sorbent. The choice of iron sorbent is important because goethite is a much poorer adsorbent than hydrous ferric oxide and will result in larger leachate contaminant concentration. With respect to the use of goethite versus the use of hydrous ferric oxide, EPA had discussions with Dr. David Dzombak and Dr. Samir Mathur (developer of the goethite database). In these discussions, the group discussed the sorbent question extensively, and EPA chose to use goethite rather than hydrous ferric oxide as a best estimate that would not underestimate risk. However, because actual CCW disposal sites could have hydrous ferric oxide present in their soils, the risks for arsenic could be overestimated.

Multiple Constituent Exposures. The individual human risk from each CCW constituent was considered separately in this analysis. However, the CCW waste constituent database and recent field studies such as U.S. EPA (2006c) and U.S. EPA (2008c) suggest that exposure to multiple constituents is highly likely. Because multiple constituent exposure may be synergistic depending on the constituents, certain constituent combinations may cause adverse health impacts that a single-constituent approach may underestimate. However, the quantitative human health benchmarks used by EPA are based on the toxicity of individual chemicals. With only one carcinogen present in CCW (arsenic), it was not necessary to add carcinogenic risks. Noncarcinogenic risks can be added only for chemicals with toxic effects on the same target organs, and this could have been done for fish and drinking water ingestion risks by accounting for transit time and adding HQs for contaminants with noncancer effects on the same target organs that arrive at the same time to the receptor point.

However additivity across chemicals was not considered in this risk assessment; neither was synergism or antagonism. Noncancer hazard may, therefore, be under- or overestimated. Nevertheless, risks were high enough from human exposure to single chemicals (notably arsenic, the single carcinogen) that this would not have changed the basic conclusion of the risk assessment: that there are potentially significant risks to human health from CCW disposal in landfills and surface impoundments.

4.4.3 Parameter Uncertainty and Variability

Parameter uncertainty occurs when (1) there is a lack of data about the values used in the equations, (2) the data that are available are not representative of the particular instance being modeled, or (3) parameter values have not been measured precisely or accurately because of limitations in measurement technology. Random, or sample, errors are a common source of parameter uncertainty that is especially critical for small sample sizes, as illustrated by the FBC waste results discussed in **Section 4.1.3.2**. More difficult to recognize and address are nonrandom or systematic errors that can bias the analyses from sampling errors, faulty experimental designs, or bad assumptions.

Spatial and temporal variability in parameters used to model exposure account for the distribution in the exposed population. For example, the rainfall or precipitation rates used to calculate infiltration and recharge to groundwater are measured daily by the National Weather Service at many locations throughout the United States, and statistics about these parameters are well documented. Although the distributions of these parameters may be well known, their actual values vary spatially and temporally and cannot be predicted exactly. Thus, the annual average infiltration rates used in the source model for a particular climate station provide information on average conditions appropriate for this analysis. Additionally, using data from multiple climate stations located throughout the United States can account for some, but not all, spatial variability.

4.4.3.1 Waste Concentrations

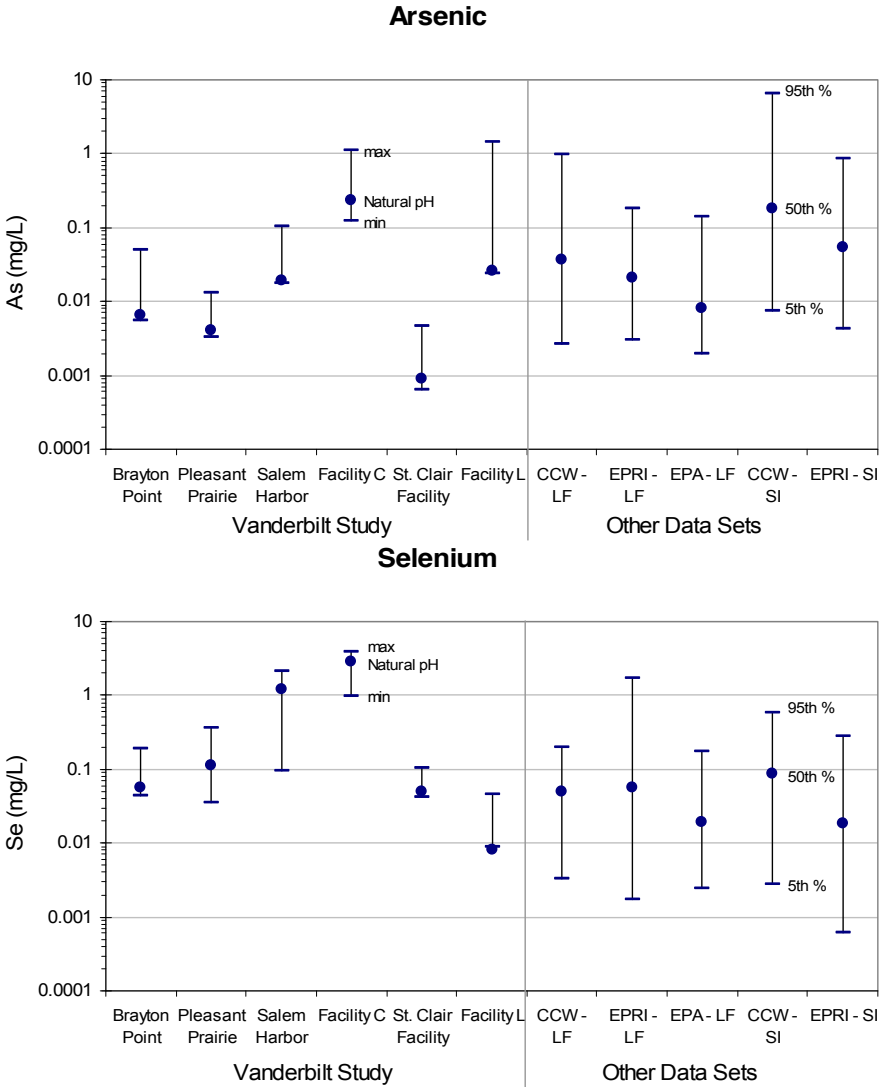
The CCW constituent database used to represent CCW total waste and waste leachate concentrations is arguably the most important data set in terms of driving the risk assessment results. The constituent data are subject to two primary uncertainties beyond the normal sampling and analysis uncertainty associated with environmental measurements: (1) the appropriateness of the landfill leachate data used in the analysis and (2) high percentages of nondetect analyses for some CCW constituents.

Appropriateness of Leachate Data. The CCW leachate data were collected from a varying number of sites using a variety of methods. The available landfill data were largely derived from the TCLP, a laboratory test designed to estimate leachate concentrations in municipal solid waste (MSW) landfills. The TCLP has been shown to both over- and underpredict leachate concentrations for other waste disposal scenarios, so the use of the TCLP data to represent CCW leachate is another source of uncertainty. However, as noted below, the TCLP data do appear to encompass the range of variability in CCW leachate concentrations that have been measured in more recent studies.

Surface impoundment leachate is represented by porewater measurements taken beneath actual impoundments, which should more closely represent the leachate seeping from the bottom of the impoundment than would bulk surface impoundment waste concentrations. The porewater is in direct contact with the waste, so these concentrations should typically be at least as great as concentrations in the bulk surface impoundment. However, although these porewater data arguably should better represent leachate concentrations, they are fewer in number than the landfill data and therefore subject to uncertainty as to how representative they are of all CCW

wastes. Results for surface impoundments for antimony, mercury, and thallium are not presented due to the paucity of leachate data (1 or 2 sites, and 11 or fewer values).

Since the CCW risk assessment was conducted in 2003, EPA-sponsored research conducted by Vanderbilt University has improved the scientific understanding of the generation of leachate from CCW, in particular for mercury, arsenic, and selenium (U.S. EPA, 2006c; U.S. EPA, 2008c). **Figure 4-2** plots the results from U.S. EPA (2006c) for arsenic and selenium, along with data from EPA's Leach2000 database and EPRI (as provided in U.S. EPA, 2006c) against the data used for landfills and surface impoundments used in the CCW analysis.



Key to data sets:

Vanderbilt	=	U.S. EPA (2006c)
CCW	=	CCW Constituent Database (this analysis)
EPRI	=	EPRI Leachate data (from U.S. EPA, 2006c)
EPA	=	Leach 2000 data (from U.S. EPA, 2000)
LF	=	landfills
SI	=	surface impoundments

Figure 4-2. Comparison of CCW leachate data with other leachate data in U.S. EPA (2006c).

For the 2006 Vanderbilt leaching study report, data are provided for each ash tested, with the minimum, maximum, and value at natural pH plotted on the chart. Percentile values (95th, 50th, 5th) are plotted for the compiled data sets (EPA, EPRI, and CCW), and mercury was not modeled for landfills because of a high number of nondetects.

For arsenic, the CCW values bracket the range found in the other studies. Selenium values also agree fairly well for CCW landfill data, although the CCW landfill values appear to be lower than some of the values from the other studies, suggesting that selenium risks may have been somewhat underestimated for landfills in this analysis. This is significant even though selenium risks from landfills were not above an HQ of 1 in this analysis, because selenium is often reported as a constituent of concern (along with arsenic and boron) in CCW damage cases (U.S. EPA, 2000, 2003e; Lang and Schlictmann, 2004; Zillmer and Fauble, 2004).

U.S. EPA (2008c) extends the work in U.S. EPA (2006c) to include laboratory leaching studies of 23 CCWs sampled from 8 coal combustion power plants. Wastes tested included fly ash, scrubber sludges, and gypsum. All of the metals addressed in this risk assessment were measured in the laboratory leaching tests.

Similar to Figure 4-2 above, Figures 46–59 on pages 77–86 in U.S. EPA (2008c) compare constituent concentration ranges in their laboratory CCW extracts to ranges reported by other CCW leachate data compilations, including the constituent data from this risk assessment. These graphs are not repeated here, but the conclusions are similar to the U.S. EPA (2006c) comparisons, in that the ranges of metals concentrations generally plot within the range reported for the laboratory tests, especially with fly ash and flue gas desulfurization sludges. For ash codisposed with coal refuse metal, concentrations tend to be an order of magnitude or more greater than the wastes studied in U.S. EPA (2008c), which did not include such codisposed wastes. Only two CCW metals plot largely outside the range for fly ash. Barium fly ash concentrations from the CCW risk assessment are an order of magnitude or more lower than those reported by U.S. EPA (2008c), and lead concentrations in the fly ash and FGD wastes modeled in this risk assessment are one to two orders of magnitude above those plotted in U.S. EPA (2008c). The latter may be an artifact of the predominance of TCLP measurements in the CCW constituent database, because the acetate buffer in the TCLP can be especially effective in complexing lead compounds into the extract solution. Finally, a few of the Vanderbilt measurements for molybdenum and selenium are above the range modeled in the CCW risk assessment.

The fact that the 2006 and 2008 Vanderbilt results are in general agreement with the CCW arsenic and selenium levels does help allay concerns that the TCLP CCW leachate values used in the analysis markedly overestimate or underestimate the concentrations actual CCW leachate.

Mercury and Nondetect Analyses. For certain of the CCW constituents addressed in this analysis, the CCW leachate database contains a large number of nondetect measurements (concentrations below an analytical instrument's ability to measure). **Table 4-30** illustrates this point by showing, by WMU type and chemical, the overall percent of nondetect values for each

chemical and the percent of site-averaged values[11] that are composed entirely of nondetect measurements. Although some constituents have a large number of nondetect values, many of those could still be modeled (substituting half the detection limit for nondetect values). Where there are detections for a chemical, the specific substitute value used for nondetect values does not affect the upper percentile risks, because the upper percentile risks are associated with the higher, detectable source concentrations in the distribution rather than the lower source concentrations associated with nondetect values. Values for nondetects will be in the lower percentiles whether they are half the detection limit or some other value.

Table 4-30. Proportion of Nondetect Analyses for Modeled CCW Constituents

Chemical[a]	Measurements		Sites	
	Number	% nondetects	Number	% with all nondetects
Landfills				
Aluminum	397	18%	61	5%
Antimony	496	50%	66	41%
Arsenic	1,182	49%	128	20%
Barium	1,225	11%	126	5%
Boron	930	8%	83	2%
Cadmium	1,237	50%	124	31%
Cobalt	559	56%	52	19%
Lead	1,109	60%	125	30%
Mercury	***974***	***91%***	***101***	***58%***
Molybdenum	373	24%	58	10%
Nitrate/Nitrite	141	48%	20	15%
Selenium	1,227	49%	131	17%
Thallium	402	60%	40	45%
Surface Impoundments				
Aluminum	158	10%	16	6%
Antimony	***11***	***100%***	***2***	***100%***
Arsenic	155	16%	16	6%
Barium	161	14%	16	13%
Boron	164	7%	171	6%
Cadmium	164	68%	16	50%
Cobalt	49	59%	4	50%
Lead	138	78%	14	36%
Mercury	***1***	***100%***	***1***	***100%***
Molybdenum	161	37%	17	24%
Nitrate/Nitrite	267	59%	14	7%
Selenium	140	33%	15	20%

(continued)

[11] As explained in **Appendix A**, the CCW risk assessment used site-averaged constituent concentrations. That is, an average value was used when there were multiple measurements for a chemical at a particular site.

Proportion of Nondetect Analyses for Modeled CCW Constituents (continued)

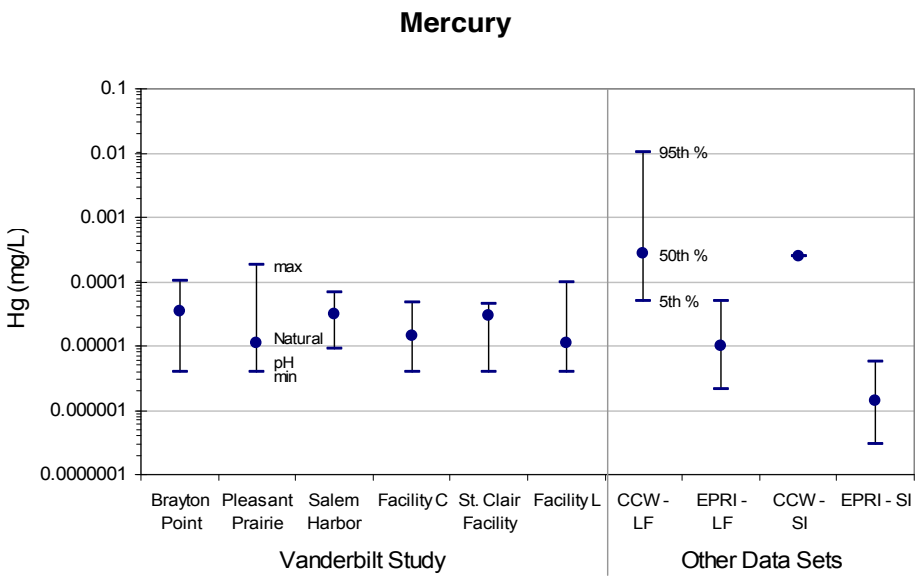
Chemical[a]	Measurements		Sites	
	Number	% nondetects	Number	% with all nondetects
Thallium	***11***	***100%***	***2***	***100%***

[a] Results for constituents shown in ***bold italics*** were not presented in this report because of high detection limits or limited data.

Constituents that could not be addressed in this analysis because of a very high number of nondetects (i.e., more than 90 percent of measurements) included mercury (for landfills and surface impoundments) and thallium and antimony (for surface impoundments only). Mercury is of particular interest because it is the only constituent with significant concern through the fish consumption pathway, and because there is the potential for mercury concentrations in CCW to increase as flue gas mercury controls are installed on coal-fired power plants in response to the Clean Air Interstate Rule (CAIR) and the Clean Air Mercury Rule (CAMR). However, analysis of the effect of mercury emission controls was outside the scope of the risk assessment, which was to evaluate current waste disposal conditions, not potential future changes due to emission controls.

Recent work by Vanderbilt University (U.S. EPA, 2006c, 2008c) sheds some light on mercury concentrations in leachate from some CCWs. **Figure 4-3** plots the CCW distribution of mercury concentrations (assuming half the detection limit for mercury values below detection) against results from the Vanderbilt work and recent data collected by EPRI (from U.S. EPA, 2006c; results are similar in U.S. EPA, 2008c). Assuming half the detection limit, the CCW mercury leachate values are about an order of magnitude or more higher than the Vanderbilt or EPRI data. With a single CCW leachate analysis available for surface impoundments, it is difficult to draw firm conclusions, but the concentration value is above the maximum value shown in the other studies. In short, the mercury levels in the CCW database are not useful because of high detection limits. In addition, the Vanderbilt study found that older mercury analyses, such as the ones in the CCW database, could be biased high because of cross-contamination issues.

Finally, U.S. EPA (2006c) and preliminary results of ongoing EPA studies (e.g., U.S. EPA, 2008c) suggest that both mercury levels and mercury leachability in CCW can vary depending on the flue gas mercury controls used at a power plant.



Key to data sets:

Vanderbilt = U.S. EPA (2006c)
CCW = CC Constituent Database (this analysis)
EPRI = EPRI Leachate data (from U.S. EPA, 2006c)
EPA = Leach 2000 data (from U.S. EPA, 2000, as cited in U.S. EPA, 2006c)
LF = landfills
SI = surface impoundments

Figure 4-3. CCW mercury concentrations compared with other leachate data.

4.4.3.2 WMU Locations and Characteristics

The locations of the specific sites in the United States where CCW is disposed are known, and EPA used the soil and climatic characteristics of these sites in the Monte Carlo analysis. Because most locations were facility front gates or centroids, the exact location of the CCW landfill or surface impoundment was not known. To account for this uncertainty, soil data were collected for an area around the plant and soil type distributions were sampled in the Monte Carlo analysis. Climate center assignments were combined with the soil texture distributions to select infiltration and recharge rates to use in the analysis.

WMU area, depth, volume, and liner type were not varied in the Monte Carlo analysis because values for these variables were known from the EPRI survey data. More uncertain parameters, such as depth below grade, were varied within reasonable ranges. These data were used in the source model calculations to generate the distribution of environmental releases used by the fate and transport modeling.

Three standard WMU liner scenarios (clay, composite, and unlined) were assigned to each facility based on best matches to data in the EPRI survey on liner type. Infiltration through these liners was then modeled using assumptions, models, and data developed in support of EPA's Industrial Subtitle D guidance. How well these assumptions and models represent the performance of CCW WMU landfills and surface impoundments is an uncertainty in this analysis.

With respect to the clay liners, the 2009 risk assessment used the assumption that compact clay liners were designed to have a hydraulic conductivity of 1×10^{-7} cm/sec. This is consistent with EPA's Industrial D Guidance, which states that "clay liners should be at least 2 feet thick and have a maximum hydraulic conductivity of 1×10^{-7} cm/sec" (U.S. EPA, 2006d). However, clay liners designed to meet a 1×10^{-7} cm/sec hydraulic conductivity could perform differently in practice. In one liner study (Moo-Young et al., 2004), a small set of clay-lined landfills were found to have field hydraulic conductivities ranging from 2×10^{-9} to 4.4×10^{-8} cm/sec and a small set of surface impoundments were found to have field hydraulic conductivities ranging from 3×10^{-6} to 3.2×10^{-5} cm/sec. Thus, the assumption of clay liners performing at 1×10^{-7} cm/sec could lead to an under- or over-estimate of actual risks.

Composite liners would also not be expected to perform consistently over 10,000 years as was assumed in the model. Instead, the liner would eventually perform at the level of the clay layer once the synthetic layer had deteriorated. This simplification is likely to lead to an underestimate of composite liner risks.

4.4.3.3 Fate and Transport Model Variables

The parameter values required to model contaminant fate and transport in groundwater were obtained from site-specific, regional, and national databases. Hydrogeologic environment was assigned to each site, based on geologic maps and soil conditions; where assignments were uncertain, two or three settings might be used in the Monte Carlo analysis. Because aquifer properties are highly variable and uncertain, reasonable sets of aquifer properties were selected, based on hydrogeologic environment, from a hydrogeologic database.

Receptor Location (Drinking Water Wells). The sensitivity analysis (**Section 4.3**) showed that distance of a receptor from the contaminant source is an important influence on media concentration, especially for contaminants that strongly sorb to soil and aquifer materials. For the groundwater-to-drinking-water pathway, receptor location was represented as the distance and position, relative to a contaminant plume, of residential drinking water wells from the WMU. Because no data were readily available on the distance of CCW disposal sites from residential wells, EPA used data from a survey of well distances from MSW landfills. Whether or not this is an accurate representation of well distance for CCW landfills and surface impoundment is an uncertainty in this analysis. EPA believes that the MSW well distance distribution used is protective for CCW landfills and surface impoundments. See **Appendix C**, **Section C.2**, for more details.

Location and Characteristics of Waterbodies. One aspect of the site configuration of particular relevance to the aquatic food chain modeling is the locations and characteristics of the waterbodies. The size of the waterbodies (and the distance from the WMU) affects constituent concentrations and loadings predicted for that waterbody. The distance from the WMU to the waterbody was based on an empirical distribution of measurements, taken from actual CCW sites, of the distance from the edge of the WMU to the nearest stream or lake. The uncertainty posed in this analysis is the sampling of this distribution as compared to a more certain measurement of the actual distance at each CCW site. Surface water variables, including flow and water quality parameters, were collected for the stream reach being modeled, or for a larger hydrologic region where data were not available for a particular reach.

Waterbodies Intercepting the Groundwater Plume. As discussed in **Section 3.7**, mass is not actually removed from the groundwater when the plume is intercepted by a surface waterbody. Therefore, in cases where wells are located beyond an intersecting surface water body, the draft risk assessment may not account for interactions between surface water and groundwater. Examining the input database, EPA notes that approximately two-thirds (69%) of the Monte Carlo runs contained such an intersecting surface waterbody. Thus, the 50th percentile results may overestimate groundwater risks to these receptors. However, because the WMUs with closer receptor wells exhibited higher risks on average, the 90th percentile results are not likely to be significantly affected.

Environmental Parameters. Uncertainties related to environmental parameters (soil, aquifer, surface water, climate data) have already been mentioned. The parameters with the largest impact on results are aquifer hydraulic conductivity and gradient, which were selected from a national database of aquifer properties.

Fish Bioconcentration and Bioaccumulation Factors. For fish consumption, exposure dose was calculated using BCFs to estimate the transfer of pollutants from environmental media into fish. Uncertainty is associated with models used to estimate BCFs for aquatic biota. Aquatic BCFs are developed by dividing measured concentrations in aquatic biota by total surface water concentrations. **Appendix J** lists the bioconcentration and bioaccumulation parameters used in the risk assessment, along with their sources.

4.4.3.4 Exposure and Risk Modeling Variables

Exposure parameters and benchmarks for human and ecological risk also contribute to parameter variability and uncertainty.

Human Exposure Factors. Individual physical characteristics, activities, and behavior are quite different, and thus the exposure factors that influence the exposure of an individual, including ingestion rate, body weight, and exposure duration, are quite variable. Exposure modeling relies heavily on default assumptions concerning population activity patterns, mobility, dietary habits, body weights, and other factors. The probabilistic assessment for the adult and child exposure scenario addressed the possible variability in the exposure modeling by using statistical distributions for these variables for each receptor in the assessment: adult and child resident and adult and child recreational fisher. Data on fish consumption rates were not available for children of recreational anglers; thus the adult recreational angler data were used for children in this analysis, which could overestimate risk from this pathway for children. For all exposure factors varied, a single exposure factor distribution was used for adults for both males and females. For child exposures, one age (age 1) was used to represent the age at the start of exposure, because this age group was considered to be most sensitive for most health effects.

The *Exposure Factors Handbook* (U.S. EPA, 1997c,d,e) provides the current state of the science concerning exposure assumptions and represents EPA's current guidance on exposure data, and it was used throughout this assessment to establish statistical distributions of values for each exposure parameter for each receptor. The *Exposure Factors Handbook* has been carefully reviewed and evaluated for quality. EPA's evaluation criteria included peer review, reproducibility, pertinence to the United States, currency, adequacy of the data collection period,

validity of the approach, representativeness of the population, characterization of the variability, lack of bias in study design, and measurement error. There are some uncertainties, however, in the data that were used.

Site-specific fish consumption rate data were not available, but the Maine study data, where anglers fished from streams, rivers, and ponds, were consistent with the modeling scenarios used in this risk analysis and provided the detailed percentile data required for a probabilistic analysis. However, applying Maine angler consumption rates to other parts of the country may under- or overestimate exposures.

EPA's child-specific exposure guidance has been recently finalized (U.S. EPA, 2008b) but was not used in the risk assessment because the water consumption rates and body weights provided in the *Child-Specific Exposure Factors Handbook* (U.S. EPA, 2008b) do not differ significantly from those found in the 1997 *Exposure Factors Handbook* and would not have changed the results, but the use of the 1997 values may contribute some parameter uncertainty. One exception is the distribution of child fish consumption rates used. Here, U.S. EPA (2008b) consumption rates are higher than the 1997 rates used in the analysis. This introduces uncertainty into the analysis, and likely underestimates risks in the fish consumption pathway.

As is customary for EPA's RCRA risk assessments, human exposure factor data were not correlated (i.e., for each modeling run, each exposure factor was selected from its distribution independently), introducing some uncertainty because it is possible to select, for example, a high drinking water rate with a small body weight. However, although a specific modeling run may have had an unrealistic combination of exposure factors, the large number of Monte Carlo iterations performed (10,000) ensures that this is unlikely to significantly affect the risk assessment results.

Diet Assumptions for Ecological Receptors. National-scale assessments often assume maximum intake of contaminated prey in the diets of primary and secondary consumers (i.e., 100 percent of the diet originates from the contaminated area). Under field conditions, many receptors are opportunistic feeders with substantial variability in both the type of food items consumed and the geospatial patterns of feeding and foraging. The actual proportion of wildlife receptors' diets that would be contaminated depends on a number of factors such as the species' foraging range, quality of food source, season, intra- and interspecies competition. Consequently, the exclusive diet of contaminated food items tends to provide a very high-end estimate of potential risks.

Human Health Benchmarks. The uncertainties generally associated with human health benchmarks are discussed in detail in EPA's *Guidelines for Carcinogen Risk Assessment* (U.S. EPA, 2005), and IRIS (U.S. EPA, 2009a). EPA defines the RfD as "an estimate (with uncertainty spanning perhaps an order of magnitude) of a daily exposure to the human population (including sensitive subgroups) that is likely to be without appreciable risk of deleterious effects during a lifetime" (U.S. EPA, 1994, 2009a). RfDs are based on an assumption of lifetime exposure and may not be appropriate when applied to less-than-lifetime exposure situations (U.S. EPA, 2009a). The CSF is an upper-bound estimate of the human cancer risk per mg of chemical per kg body weight per day. Because exposures were often less than lifetime, some uncertainty was introduced in the noncancer hazard and cancer risk estimates.

EPA routinely accounts for uncertainty in their development of RfDs and other human health benchmarks. Uncertainty and variability in the toxicological and epidemiological data from which RfDs were derived are accounted for by applying uncertainty factors. Some of these uncertainties include those associated with extrapolation from animals to humans, from LOAELs to NOAELs, and from subchronic to chronic data, and to account for sensitive subpopulations. If certain toxicological data are missing from the overall toxicological database (e.g., reproductive data), EPA accounts for this by applying an uncertainty factor.

Table 4-31 presents IRIS uncertainty factors for the RfDs for the CCW constituents that showed HQs greater than 1 in the risk assessment, along with the highest HQ observed and the disposal scenario for which this HQ was observed. IRIS defines uncertainty factors as follows:

> "Uncertainty factors (UFs) are one of several, generally 10-fold, default factors used in operationally deriving the RfD from experimental data. The factors are intended to account for (1) variation in susceptibility among the members of the human population (i.e., inter-individual or intraspecies variability); (2) uncertainty in extrapolating animal data to humans (i.e., interspecies uncertainty); (3) uncertainty in extrapolating from data obtained in a study with less-than-lifetime exposure (i.e., extrapolating from subchronic to chronic exposure); (4) uncertainty in extrapolating from a LOAEL rather than from a NOAEL; and (5) uncertainty associated with extrapolation when the database is incomplete."[12]

The constituent-specific uncertainty factors for the CCW constituents in Table 4-31 are discussed further in the source documents (e.g., IRIS) for the individual human health benchmarks used in the analysis, which are referenced in **Appendix G**. In general, EPA human health benchmarks are derived using a health-protective approach. These uncertainty factors can be considered when evaluating the constituent-specific risks presented in this document, but only in the context of the above definitions and the information presented in IRIS for each chemical.

The hierarchy of data sources that was implemented for this analysis was based largely on the rigor of review that a benchmark has received. Methodologies evolve over time, with improvements in existing methods and the development of new health benchmark practices (e.g., benchmark dose methodology). As a result, the magnitude of a given benchmark can either increase or decrease, or a given benchmark can appear or disappear in a toxicity benchmark database. An example of the latter situation, disappearance of a toxicity benchmark, occurred during the development of this report. The human health benchmark for thallium was withdrawn from IRIS in late September 2009. The modeling results, including the noncancer human health effects estimates, were retained in this document to reflect the potential for thallium releases from CCW WMUs. EPA has decided to retain these estimates, in light of the National Academy of Sciences' (NAS's) 2008 report entitled *Science and Decisions: Advancing Risk Assessment* (NAS, 2008). In that report's recommendations, the authors noted that absence of certain information from a risk characterization can result in the missing information being overlooked during the decision making process. Evidence that relatively small quantities of thallium can be

[12] http://www.epa.gov/ncea/iris/help_gloss.htm#u

fatal to humans[13] leads EPA to conclude that omitting the thallium results from this report might cause thallium's existence in coal combustion residues to be overlooked during the risk management decision making, and thus EPA has chosen to retain those modeling results in this report.

Table 4-31. RfD Uncertainty Factors for and Benchmark Confidence for CCW Constituents with HQs Over 1

Constituent	RfD (mg/kg-day)	Source	Uncertainty Factor	Benchmark Confidence	Highest CCW HQ	CCW Scenario for Highest HQ
Antimony	4.0E-04	IRIS	1,000	low	3	GW-DW, FBC wastes, clay-lined landfills
Boron	2.0E-01	IRIS	66	high	7	GW-DW, Conventional CCW, unlined SIs
Cadmium	5.0E-04	IRIS	10	high	9	GW-DW, Codisposed CCW, unlined SIs
Cobalt	3.0E-04	PPRTV	1,000	low	500	GW-DW, Codisposed CCW, unlined SIs
Molybdenum	5.0E-03	IRIS	30	medium	8	GW-DW, Conventional CCW, unlined SIs
Selenium	5.0E-03	IRIS	3	high	3	GW-SW, Conventional CCW, unlined SIs
Thallium	8.0E-05	IRIS	3,000	low	4	GW-DW, FBC wastes, clay-lined landfills

Most health benchmarks used in this analysis were from IRIS. Human health benchmarks in IRIS have been subjected to rigorous internal and external reviews and represent Agency-wide consensus human health risk information. However, some benchmarks in IRIS are quite dated. Provisional human health benchmarks derived by the Superfund Technical Support Center have been peer reviewed and are used where there is no IRIS value.

Chemical-specific health benchmarks were used for all constituents assessed in the analyses. However, the RfD for fluoride was based on fluorine; the RfDs for mercuric chloride and methyl mercury were used as surrogates for elemental mercury from food, soil, and water ingestion, and fish ingestion, respectively; and the RfD for thallium was based on thallium chloride. The use of these surrogate data is not thought to have introduced any significant uncertainty. Human health benchmarks are not age-specific, and therefore, were applied to both child and adult receptors, thereby introducing some uncertainty.

EPA used the drinking water MCL for lead to estimate risks from drinking water exposure. The IEUBK model may better quantify risk for a young child exposed to lead; therefore, use of the MCL may introduce some uncertainty. However, risks from lead exposure were relatively low, well below the risk criterion for landfills and at or slightly above the risk criterion for surface impoundments, and did not drive the risk assessment conclusions.

[13] "Temporary hair loss, vomiting, and diarrhea can also occur and death may result after exposure to large amounts of thallium for short periods. Thallium can be fatal from a dose as low as 1 gram." (ATSDR, 1992)

Ecological Criteria. CSCLs were developed for constituents when sufficient data were available. In many cases, sufficient data were unavailable for a receptor/constituent combination, and therefore, the potential risk to a receptor could not be assessed. In particular, insufficient data were available to derive chronic effects CSCLs for amphibians. Because the risk results can only be interpreted within the context of available data, the absence of data cannot be construed to mean that adverse ecological effects will not occur.

In addition to the effects of data gaps on ecological benchmarks, the ecological criteria tend to be fairly conservative because the overall approach is based on "no effects" or "lowest effects" study data. In site-specific assessments, a *de minimis* effects approach is often replaced with an effects level similar to natural population variability (e.g., sometimes as high as a 20 percent effects level). As a result, the CSCLs used in this analysis are likely to overestimate risks for representative species and communities assumed to live in surface waters impacted by CCW WMUs. Because the difference between a LOAEL and a NOAEL is often about a factor of 10, an HQ exceedance of roughly 10 may not be ecologically significant. In contrast, CSCLs based on no effects data that are developed for the protection of threatened and endangered species are presumed to be protective.

4.5 Summary and Conclusions

CCW risk assessment results at the 90th percentile suggest that the management of CCW in unlined or clay-lined WMUs result in risks greater than 1 in 100,000 for excess cancer risk to humans or an HQ greater than 1 for noncancer effects to both human and ecological receptors. Key risk findings include the following:

- For humans exposed via the groundwater-to-drinking-water pathway, risks from clay-lined units that dispose CCW or CCW comanaged with coal refuse are lower than those for unlined units. However, the 90th percentile risks for clay-lined units are still well within or above the range of concern (10^{-6} to 10^{-4}) for cancer risk and above an HQ of 1 for noncarcinogens. For example, arsenic III cancer risks in clay-lined units range from 1 in 5,000 for landfills to 9 in 10,000 in surface impoundments. The thallium HQ was as high as 2 for clay-lined landfills, and the clay-lined surface impoundment HQ was as high as 200 for cobalt and 4 for boron.

- Arsenic was the constituent with the highest risk for landfills. Clay-lined landfills presented 90th percentile risks above an excess cancer risk of 1 in 100,000 for arsenic (risks as high as 1 in 5,000) and an HQ of 1 for thallium (HQ of 2). When landfills are unlined, they also present risk above an HQ of 1 for antimony and molybdenum, each with an HQ of 2. Here, arsenic cancer risks were as high as 1 in 2,000. Clay-lined FBC landfills also presented 90th percentile risks above and HQ of 1 for antimony (HQ = 3) and thallium (HQ = 4) and showed excess cancer risks of 3 in 50,000 for arsenic. However, unlined FBC landfills differed in that they only exceeded a 1 in 100,000 excess cancer risk for arsenic.[14] At the 50th percentile, arsenic III from CCW codisposed with

[14] As modeled, unlined FBC units showed less risk than clay-line FBC units.

coal refuse unlined landfills showed an excess cancer risk of 1 in 50,000: all noncarcinogenic constituents were well below an HQ of 1.

- Arsenic and cobalt were the constituents with the highest risks for surface impoundments, with risks as high as 1 in 50 and an HQ of 500, respectively, for unlined units. Clay-lined surface impoundments presented 90th percentile cancer risks above 1 in 100,000 for arsenic (7 in 1,000 cancer risk), HQs above 1 for boron (HQs as high as 4), cadmium (HQ as high as 3), cobalt (HQ as high as 200), molybdenum (HQ as high as 5), and nitrate (an MCL-based HQ as high as 10). When surface impoundments are unlined, they also show risk above an HQ of 1 for lead (HQ of 9) and selenium (HQ of 2). Here, arsenic cancer risks are as high as 1 in 50, and cobalt had HQs as high as 500. The only 50th percentile surface impoundment results that exceeded the risk range or HQ criterion were arsenicand cobalt. Here, unlined units had arsenic cancer risks as high as 6 in 10,000 while clay-lined units had arsenic cancer risks as high as 1 in 5,000. Cobalt HQs were as high as 20 and 6 for unlined and clay-lined surface impoundments, respectively.

- For the groundwater-to-drinking-water pathway, composite liners, as modeled in this assessment, effectively reduce risks from all constituents to below a cancer risk of 1 in 100,000 and an HQ of 1 for both landfills and surface impoundments at the 90th and 50th percentiles.

- For the groundwater-to-drinking-water pathway, arrival times of the peak concentrations at a receptor well are much longer for landfills (hundreds or thousands of years) than for surface impoundments (most less than 100 years).

- For humans exposed via the groundwater-to-surface-water (fish consumption) pathway, unlined and clay-lined surface impoundments posed risks above an excess cancer risk of 1 in 100,000 and an HQ of 1 at the 90th percentile. For CCW managed alone in surface impoundments, these exceedences came from selenium (HQs of 3 and 2), while for CCW comanaged with coal refuse these exceedences came from arsenic (3 in 100,000 and 2 in 100,000 excess cancer risks for unlined and clay-lined units). All 50th percentile surface impoundment risks are below a cancer risk of 1 in 100,000 and an HQ of 1. No constituents pose risks above these risk levels for landfills (including FBC landfills) at the 90th or 50th percentile for the fish consumption pathway.

- Waste type has a much larger effect when wastes are managed in surface impoundments than when they are managed in landfills. In the case of surface impoundments, some constituents (boron, molybdenum, nitrate, and selenium) presented higher risks from CCW managed alone. However, others (arsenic, cadmium, cobalt, and lead) presented higher risks when CCW is comanaged with coal refuse, because of their association with the sulfide minerals concentrated in the refuse.

- The higher risks for surface impoundments than landfills are likely due to higher waste leachate concentrations and the higher hydraulic head from the impounded liquid waste. This is consistent with damage cases reporting wet handling as a factor that can increase risks from CCW management.

- For ecological receptors exposed via surface water, risks for landfills exceed an HQ of 1 for boron (HQ of 281 for unlined and 78 for clay-lined), lead (HQ of 8 for unlined), and selenium, arsenic, and barium (HQs of 2) at the 90th percentile, but 50th percentile HQs are well below 1. For surface impoundments, 90th percentile risks for several constituents (boron, lead, arsenic, selenium, cobalt, and barium) exceed an HQ of 1, with boron showing the highest risks (HQ over 2,000). Only boron exceeds an HQ of 1 at the 50th percentile (HQ = 7 for unlined surface impoundments). The HQs over 1 for boron and selenium are consistent with reported ecological damage cases, which include impacts to waterbodies through the groundwater-to-surface-water pathway.

- For ecological receptors exposed via sediment, 90th percentile risks exceed an HQ of 1 for both landfills and surface impoundments because certain CCW constituents strongly sorb to sediments in the waterbody. Here, the 90th percentile HQ for lead was 58 for unlined landfills and clay-lined surface impoundments, and 311 for unlined surface impoundments. For arsenic, HQs were 11 and 3 for unlined and clay-lined landfills, and 127 and 55 for unlined and clay-lined surface impoundments. Cadmium had HQs of 5 for unlined landfills, and 30 and 9 for unlined and clay-lined surface impoundments. Antimony had an HQ of 2 for unlined landfills. Composite lined surface impoundments also had risks above an HQ of 1 for lead (HQ of 4), arsenic (HQ of 31), and cadmium (HQ of 2). The 50th percentile risks are an order of magnitude or more below an HQ of 1 for ecological receptors exposed via sediments.

Sensitivity analysis results indicate that for most of the scenarios evaluated (over 70 percent), the risk assessment model was most sensitive to parameters related to the contaminant source and groundwater flow and transport: WMU infiltration rate, leachate concentration, and aquifer hydraulic conductivity and gradient. For strongly sorbing contaminants (such as lead and cadmium), variables related to sorption and travel time (adsorption coefficient, depth to groundwater, receptor well distance) are also important.

One of the most sensitive parameters in the risk assessment (infiltration rate) is greatly influenced by whether and how a WMU is lined. The 1994–2004 DOE/EPA survey results (U.S. DOE, 2006) do not include information on how many unlined facilities are still operating today, but do indicate that more facilities are lined today than were in the 1995 EPRI survey data set on which this risk assessment was based. This suggests that the risks from future CCW disposal facilities are likely to be lower than the results presented in this report.

There are uncertainties associated with the CCW risk assessment, but scenario uncertainty (i.e., uncertainty about the environmental setting around the plant) has been minimized by basing the risk assessment on conditions around existing U.S. coal-fired power plants around the United States. Uncertainty in environmental setting parameters has been incorporated into the risk assessment by varying these inputs within reasonable ranges when the exact value is not known. Uncertainty in human exposure factors (such as exposure duration, body weight, and intake rates) has also been addressed through the use of national distributions.

Some uncertainties not addressed explicitly in the risk assessment have been addressed through comparisons with other studies and data sources.

- **Appropriateness of CCW leachate data.** Data on another highly sensitive parameter, leachate (porewater) constituent concentration, were available and used for CCW surface impoundments. However, available data for landfills were mainly TCLP analyses, which may not be representative of actual CCW leachate. Comparisons with recent (2006 and 2008) studies of coal ash leaching processes show very good agreement for arsenic. However, although the selenium CCW data are within the range of the 2006 and 2008 data, some of the higher concentrations in both Vanderbilt data sets are not represented by the TCLP data, and U.S. EPA (2008c) show similar trends for barium and molybdenum. This suggests that risks for these metals may be underestimated, which is consistent with selenium as a common driver of the damage cases.

- **Impacts of mercury rules (CAIR and CAMR).** While CAIR and CAMR will reduce emissions of mercury and other metals from coal-fired power plants, mercury and other more volatile metals will be transferred from the flue gas to fly ash and other air pollution control residues, including the sludge from wet scrubbers. EPA ORD has research underway to evaluate changes to CCW characteristics and leaching of mercury and other metals from CAIR and CAMR. Data from the first report (U.S. EPA, 2006c) suggest that although total mercury will increase in CCW from the use of sorbents as mercury controls, the leachability of mercury may be reduced. Data from U.S. EPA (2008c) add to this assessment by supporting similar findings.

- **Mercury and nondetect analyses.** Because of a high proportion of nondetect values and a limited number of measurements, the risks from mercury in CCW could not be evaluated for either landfills or surface impoundments and for antimony and thallium in surface impoundments. The 2006 leaching study data suggest that mercury levels are fairly low in fly ash from coal combustion, a conclusion generally confirmed by the 2008 study report (U.S. EPA, 2008c), although that study did find higher mercury leachate concentrations from scrubber sludge than other coal wastes and found that blending fly ash and lime can increase mercury leaching from scrubber sludge.

Uncertainties that are more difficult to evaluate with respect to CCW risk results include the following:

- **Well distance.** Nearest well distances were taken from a survey of MSW landfills, as data were not available from CCW sites. EPA believes that this is a protective assumption because MSW landfills generally tend to be in more populated areas, but there are little data available to test this hypothesis.

- **Liner conditions.** Liner design and performance for CCW WMUs were based on data and assumptions EPA developed to be appropriate for nonhazardous industrial waste landfills. EPA believes that CCW landfills should have similar performance characteristics, but does not have the quantitative data to verify that.

- **Data gaps for ecological receptors.** Insufficient data were available to develop screening levels and quantitative risk estimates for terrestrial amphibians, but EPA acknowledges that damage cases indicate risk to terrestrial amphibian and plant communities through exposure to selenium and boron.

- **Ecosystems and receptors at risk.** Certain critical assessment endpoints were not evaluated in this analysis, including impacts on managed lands, critical habitats, and threatened and endangered species.

- **Synergistic risk.** The impact of exposures of multiple contaminants to human and ecological risks was not evaluated in this analysis. EPA recognizes that a single-constituent analysis may underestimate risks associated with multiple chemical exposures.

These are potentially the more significant uncertainties associated with the CCW risk assessment. Other uncertainties are discussed in **Section 4.4**.

Given the results and characterization above, composite liners, as modeled in this risk assessment, effectively reduce risks from all pathways and constituents to levels below an excess cancer risk of 1 in 100,000 or an HQ of 1 for both landfills and surface impoundments. The CCW risk assessment suggests that the management of CCW in unlined landfills and unlined surface impoundments may present risks to human health and the environment. From the perspective of what is known about toxic effects in humans, arsenic, nitrates, cadmium, and selenium appear to be among the constituents that may present risks of concern depending on the specific waste management practices employed. From the perspective of what is known about toxic effects in ecological receptors, arsenic, boron, lead, and selenium emerge as having documented adverse effects on ecological receptors.

The estimated human health arsenic risks from clay-lined units are lower than the risks of unlined units, but are still above a 1 in 100,000 excess cancer risk or an HQ of 1. In addition, surface impoundments typically showed higher risks than landfills, regardless of liner type. These risk results are largely consistent with damage cases compiled by EPA (U.S. EPA, 2000, 2003e, 2007) and others (Lang and Schlictmann, 2004; Zillmer and Fauble, 2004; Carlson and Adriano, 1993; Rowe et al., 2002; Hopkins et al., 2006). These results suggest that with a higher prevalence of composite liners in new CCW disposal facilities, future national risks from onsite CCW disposal are likely to be lower than those presented in this risk assessment (which is based on 1995 CCW WMUs).

5.0 References

ATSDR (Agency for Toxic Substances and Disease Registry). 1992. *Toxicological Profile for Thallium*. Available at http://www.atsdr.cdc.gov/toxprofiles/tp54.html.

Barrows, M.E., S.R. Petrocelli, K.J. Macek, and J.J. Carroll. 1980. Chapter 24: Bioconcentration and elimination of selected water pollutants by bluegill sunfish (*Lepomis macrochirus*). In *Dynamics, Exposure and Hazard Assessment of Toxic Chemicals*. Edited by R. Haque. Ann Arbor, MI: Ann Arbor Science.

Bonaparte, R., J.P. Giroud, and B.A. Gross. 1989. Rates of leakage through landfill liners. *Geosynthetics 1989 Conference*. San Diego, CA.

CalEPA (California Environmental Protection Agency). 2000. *Air Toxics Hot Spots Program Risk Assessment Guidelines: Part III. Technical Support Document for the Determination of Noncancer Chronic Reference Exposure Levels*. Office of Environmental Health Hazard Assessment, Berkeley, CA. Available (in four sections) at http://www.oehha.org/air/chronic_rels/22RELS2k.html http://www.oehha.org/air/chronic_rels/42kChREL.html http://www.oehha.org/air/chronic_rels/Jan2001ChREL.html http://www.oehha.org/air/chronic_rels/1201Crels.html.

CalEPA (California Environmental Protection Agency). 2008. *Air Toxics Hot Spots Program Risk Assessment Guidelines. Technical Support Document for the Determination of Noncancer Chronic Reference Exposure Levels*. Office of Environmental Health Hazard Assessment, Berkeley, CA. Available at http://www.oehha.org/air/hot_spots/2008/NoncancerTSD_final.pdf.

Carlson, C.L., and D.C. Adriano. 1993. Environmental impacts of coal combustion residues. *J. Environ. Qual.* 22(227-247).

Drever, J.I. 1988. *The Geochemistry of Natural Waters*, 2d ed. Englewood Cliffs NJ: Prentice-Hall.

Eisler, R. 1989. *Molybdenum Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review. Contaminant Hazard Reviews*, Report No. 19, Biological Report 85(1.19). Laurel, MD. August. Available at http://www.pwrc.usgs.gov/infobase/eisler/CHR_19_Molybdenum.pdf.

EPRI (Electric Power Research Institute). 1997. *Coal Combustion By-Products and Low-Volume Wastes Comanagement Survey*. Palo Alto, CA. June.

Finkelman, R.B. 1995. Modes of occurrence of environmentally sensitive trace elements in coal. Chapter 3 in: Dalway, J.S., and F. Goodarzi (eds.), *Environmental Aspects of Trace Elements in Coal*. Springer.

Hopkins, W.A., S.E. DuRant, B.P. Staub, C.L. Rowe, and B.P. Jackson. 2006. Reproduction, embryonic development, and maternal transfer of contaminants in the amphibian Gastrophryne carolinensis. *Environmental Health Perspectives*. *114*(5):661–666.

Kool, J.B., P.S. Huyakorn, E.A. Sudicky, and Z.A. Saleem. 1994. A composite modeling approach for subsurface transport of degrading contaminants from land-disposal sites. *J. Contaminant Hydrology 17*:69–90.

Kumada, H., S. Kimura, M. Yokote, and Y. Matida. 1972. Acute and chronic toxicity, uptake, and retention of cadmium in freshwater organisms. *Bulletin of Freshwater Fisheries Research Laboratory 22*(2):157–165. December 20.

Lang, D.J., and J.R. Schlictmann. 2004. Leaching of arsenic from a fly ash dump in Beverly, Massachusetts. In *Proceedings of State Regulation of Coal Combustion By-Product Placement at Mine Sites.* May 4–6, 2004, Harrisburg, Pennsylvania. U.S. Department of the Interior, Office of Surface Mining, Alton, IL. Available at http://www.mcrcc.osmre.gov/MCR/Resources/ccb/PDF/State_Regulation_of_CCB_Placement.pdf.

Lemly, A.D. 1985. Toxicology of selenium in a freshwater reservoir: Implications for environmental hazard evaluation and safety. *Ecotoxicology and Environmental Safety 10*:314–338.

Moo-Young, H., B. Johnson, A. Johnson, D. Carson, C. Lew, S. Liu, and K. Hancock. 2004. Characterization of infiltration rates from landfills: supporting groundwater modeling efforts. *Environmental Monitoring and Assessment* 96:283–311.

NAS (National Academies of Science). 2008. *Science and Decisions: Advancing Risk Assessment*. National Academy Press, Washington, DC.

Newell, C.J., L.P. Hopkins, and P.B. Bedient. 1989. *Hydrogeologic Database for Ground Water Modeling*. API Publication No. 4476. American Petroleum Institute, Washington, DC. Available as document ID number EPA-HQ-RCRA-2001-0022-0604 and also in several other dockets at http://www.regulations.gov.

Newell, C,J., L.P. Hopkins, and P.B. Bedient. 1990. A hydrogeologic database for ground water modeling. *Ground Water 28*(5):703–714.

Rowe, C.L., W.A. Hopkins, and J.D. Congdon. 2002. Ecotoxicological implications of aquatic disposal of coal combustion residues in the United States: a review. *Environmental Monitoring and Assessment.* 80:207–276.

SAB (Science Advisory Board). 2004. *Review of EPA's Multimedia, Multipathway, and Multireceptor Risk Assessment (3MRA) Modeling System.* Report of the 3MRA Review Panel of the EPA Science Advisory Board. EPA-SAB-05-003, November.

Schroeder, P.R., T.S. Dozier, P.A. Zappi, B.M. McEnroe, J.W. Sjostrom, and R.L. Peton. 1994a. *The Hydrologic Evaluation of Landfill Performance Model (HELP): User's Guide for Version 3*. EPA/600/R-94/168a. U.S. EPA Risk Reduction Engineering Laboratory, Cincinnati, OH. Available at http://nepis.epa.gov/EPA/html/Pubs/pubtitleORD.htm.

Schroeder, P.R., T.S. Dozier, P.A. Zappi, B.M. McEnroe, J.W. Sjostrom, and R.L. Peton. 1994b. *The Hydrologic Evaluation of Landfill Performance (HELP) Model: Engineering Documentation for Version 3*. EPA/600/R-94/168b. U.S. EPA Risk Reduction Engineering Laboratory, Cincinnati, OH. Available at http://nepis.epa.gov/EPA/html/Pubs/pubtitleORD.htm.

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 Ambient Water Quality Criteria for the Protection of Aquatic Organisms and Their Use*. NTIS PB85-220749. U.S. Environmental Protection Agency, Office of Research and Development, Springfield, VA.

Stephan, C.E. 1993. *Derivation of Proposed Human Health and Wildlife Bioaccumulation Factors for the Great Lakes Initiative (Draft)*. PB93-154672. Environmental Research Laboratory, Office of Research and Development, Duluth, MN. March.

Stumm, W., and J.J. Morgan. 1996. *Aquatic Chemistry, Chemical Equilibria and Rates in Natural Waters*. 3d ed. New York: John Wiley & Sons, Inc.

TetraTech. 2001. *Characterization of Infiltration Rate Data to Support Groundwater Modeling Efforts (Draft)*. Prepared for the Office of Solid Waste, U.S. Environmental Protection Agency, Washington, DC. Contract No. 68-W6-0061. May. Available as document ID numbers EPA-HQ-RCRA-2003-0001-0115 and EPA-HQ-RCRA-2003-0001-0116 at http://www.regulations.gov.

U.S. DOE (Department of Energy). 2006. *Coal Combustion Waste Management at Landfills and Surface Impoundments, 1994-2004*. DOE/PI-0004, ANL-EVS/06-4. Office of Scientific and Technical Information, Oak Ridge, TN. August. Available at http://www.ipd.anl.gov/anlpubs/2006/09/57081.pdf.

U.S. DOE (Department of Energy). 2007a. Electric Generating Capacity: Existing Generating Units in the United States, 2007. Excel spreadsheet file named existingunits2007.xls. Available at http://www.eia.doe.gov/cneaf/electricity/page/capacity/capacity.html. Washington DC: U.S. Department of Energy, Energy Information Administration.

U.S. DOE (Department of Energy). 2007b. Form EIA-860 Database Annual Electric Generator Report for 2007. Available at http://www.eia.doe.gov/cneaf/electricity/page/eia860.html. Washington DC: U.S. Department of Energy, Energy Information Administration.

U.S. EPA (Environmental Protection Agency). 1988a. *National Survey of Solid Waste (Municipal) Landfill Facilities (Draft)*. EPA/530-SW88-034. Office of Solid Waste, Washington, DC. September. Available as document ID number EPA-HQ-RCRA-2003-0004-0466 and also in other dockets at http://www.regulations.gov.

U.S. EPA (Environmental Protection Agency). 1988b. *Wastes from the Combustion of Coal by Electric Utility Power Plants—Report to Congress*. EPA 530-SW-88-002. U.S. EPA Office of Solid Waste and Emergency Remediation. Washington, DC. February.

U.S. EPA (Environmental Protection Agency). 1988c. *Design, Construction and Evaluation of Clay Liners for Waste Management Facilities*. EPA-530-SW-86-007F. Office of Solid Waste and Emergency Response. Washington, DC. November. Available at http://nepis.epa.gov/EPA/html/Pubs/pubtitleOSWER.htm.

U.S. EPA (Environmental Protection Agency). 1992. Guidelines for exposure assessment. Final guidelines. *Federal Register* 57 FR 22888-22893. Washington, DC. May 29.

U.S. EPA (Environmental Protection Agency). 1995. *Review of EPA's Composite Model for Leachate Migration with Transformation Products-EPACMTP*. EPA-SAB-EEC-95-010. Science Advisory Board Exposure Model Subcommittee of the Environmental Engineering Committee. Washington, DC.

U.S. EPA (Environmental Protection Agency). 1996a. *EPA's Composite Model for Leachate Migration with Transformation Products. EPACMTP: Background Document*. Office of Solid Waste, Washington, DC.

U.S. EPA (Environmental Protection Agency). 1996b. *Soil Screening Guidance: Technical Background Document*. EPA/540/R-95/128. Office of Solid Waste and Emergency Response, Washington, DC. July. Available at http://www.epa.gov/superfund/health/conmedia/soil/index.htm.

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. *Mercury Study Report to Congress. Volume III, Fate and Transport of Mercury in the Environment*. EPA 452/R-97/005. Office of Air Quality Planning and Standards and Office of Research and Development, Washington, DC.

U.S. EPA (Environmental Protection Agency). 1997c. *Exposure Factors Handbook, Volume I, General Factors.* EPA/600/P-95/002Fa. Office of Research and Development, Washington, DC. August.

U.S. EPA (Environmental Protection Agency). 1997d. *Exposure Factors Handbook, Volume II, Food Ingestion Factors.* EPA/600/P-95/002Fa. Office of Research and Development, Washington, DC. August.

U.S. EPA (Environmental Protection Agency). 1997e. *Exposure Factors Handbook, Volume III, Activity Factors.* EPA/600/P-95/002Fa. Office of Research and Development, Washington, DC. August.

U.S. EPA (Environmental Protection Agency). 1997f. Health Effects Assessment Summary Tables (HEAST). EPA-540-R-97-036. FY 1997 Update. Office of Solid Waste and Emergency Response, Washington, DC. Available at http://nepis.epa.gov/EPA/html/Pubs/pubtitleOSWER.htm.

U.S. EPA (Environmental Protection Agency). 1998a. *Non-Groundwater Pathways, Human Health and Ecological Risk Analysis for Fossil Fuel Combustion Phase 2 (FFC2): Draft Final Report*. Office of Solid Waste, Washington, DC. June. Available at http://www.epa.gov/waste/nonhaz/industrial/special/fossil/fsltech.htm.

U.S. EPA (Environmental Protection Agency). 1998b. *Technical Background Document for the Supplemental Report to Congress on Remaining Fossil Fuel Combustion Wastes: Ground-Water Pathway Human Health Risk Assessment. Revised Draft Final*. Office of Solid Waste, Washington, DC. June. Available at http://www.epa.gov/waste/nonhaz/industrial/special/fossil/ffc2_395.pdf.

U.S. EPA (Environmental Protection Agency). 1998c. *Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions. Update to Methodology for Assessing Health Risks Associated with Indirect Exposure to Combustor Emissions*. EPA-600/R-98/137. National Center for Environmental Assessment, Cincinnati, OH. December. Available at http://cfpub.epa.gov/ncea/cfm/recordisplay.cfm?deid=55525.

U.S. EPA (Environmental Protection Agency). 1998d. *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities, Volume One*. EPA-530-D-98-001A. Office of Solid Waste and Emergency Response, Washington, DC. Available at http://nepis.epa.gov/EPA/html/Pubs/pubtitleOSWER.htm.

U.S. EPA (Environmental Protection Agency). 1999a. *Report to Congress: Wastes from the Combustion of Fossil Fuels*. EPA 530-R-99-010. Office of Solid Waste, Washington, DC. March.

U.S. EPA (Environmental Protection Agency). 1999b. *Partition Coefficients for Metals in Surface Water, Soil, and Waste*. Office of Solid Waste, Washington, DC. Available at http://www.epa.gov/osw/hazard/wastetypes/wasteid/hwirwste/pdf/risk/reports/s0524.pdf.

U.S. EPA (Environmental Protection Agency). 1999c. *Peer Review of EPA's Hazardous Waste Identification Rule Risk Assessment Model - The Vadose Zone and Saturated Zone Modules Extracted From EPACMTP for HWIR99*. Prepared by Eastern Research Group under EPA Contract No. 68-W-99-001, Work Assignment No. 17, for Office of Solid Waste. Washington, DC. Available at http://www.epa.gov/osw/hazard/wastetypes/wasteid/hwirwste/peer03/vadoze/vz-aquif.pdf.

U.S. EPA (Environmental Protection Agency). 1999d. *Peer Review of EPA's Hazardous Waste Identification Rule Risk Assessment Model - MINTEQA2/PRODEFA2 Version 4.0*. Prepared by Eastern Research Group under EPA Contract No. 68-W-99-001, Work Assignment No. 17, for Office of Solid Waste. Washington, DC. Available at

http://www.epa.gov/osw/hazard/wastetypes/wasteid/hwirwste/peer03/minteqa2/minteqa2.pdf.

U.S. EPA (Environmental Protection Agency). 1999e. *Revised Risk Assessment for the Air Characteristic Study*. EPA-530-R-99-019b. Volume 2. Office of Solid Waste, Washington, DC. November. Available at http://nepis.epa.gov/EPA/html/Pubs/pubtitleOSWER.htm.

U.S. EPA (Environmental Protection Agency). 2000. *Review of Causative Factors for Coal Combustion Waste Damage Cases*. Prepared by SAIC for U.S. EPA Office of Solid Waste under EPA contract 68-W-99-001, WA No. 121. Washington, DC. November 29. Available as document ID number EPA-HQ-RCRA-2006-0796-0020 at http://www.regulations.gov.

U.S. EPA (Environmental Protection Agency). 2001a. *Revisions in Input Data and Modeling Procedures for Using MINTEQA2 in Estimating Metal Partition Coefficients*. Prepared by Allison Geoscience Consultants, Inc., for the Office of Solid Waste, Washington, DC.

U.S. EPA (Environmental Protection Agency). 2001b. *WATER9, Air Emission Models for Waste and Wastewater*. Technology Transfer Network Clearinghouse for Inventories & Emission Factors. Available at http://www.epa.gov/ttn/chief/software/water.

U.S. EPA (Environmental Protection Agency). 2001c. *Supplemental Guidance For Developing Soil Screening Levels For Superfund Sites*. OSWER 9355.4-24. Office of Solid Waste and Emergency Response, Washington, DC. December. Available at http://www.epa.gov/superfund/health/conmedia/soil/index.htm.

U.S. EPA (Environmental Protection Agency). 2002a. *Constituent Screening for Coal Combustion Wastes*. Draft Report prepared by Research Triangle Institute for Office of Solid Waste, Washington, DC, under EPA Contract No. 68-W-98-085. October.

U.S. EPA (Environmental Protection Agency). 2002b. *Industrial Waste Management Evaluation Model (IWEM) Technical Background Document*. EPA530-R-02-012. Office of Solid Waste, Washington, DC. August. Available at http://nepis.epa.gov/EPA/html/Pubs/pubtitleOSWER.htm.

U.S. EPA (Environmental Protection Agency). 2002d. *Current Drinking Water Standards. National Primary and Secondary Drinking Water Regulations.* Office of Ground Water and Drinking Water, Washington, DC. Available at http://www.epa.gov/safewater/mcl.html.

U.S. EPA (Environmental Protection Agency). 2003a. *EPA's Composite Model for Leachate Migration with Transformation Products (EPACMTP)*: *Parameters/Data Background Document*. Office of Solid Waste, Washington, DC. April. Available at http://www.epa.gov/epawaste/nonhaz/industrial/tools/cmtp/index.htm.

U.S. EPA (Environmental Protection Agency). 2003b. *Risk Assessment Technical Background Document for the Dye and Pigment Industry Hazardous Waste Listing Determination. OMB Review Draft*. Office of Solid Waste, Washington, DC.

U.S. EPA (Environmental Protection Agency). 2003c. *EPACMTP Technical Background Document*. Office of Solid Waste, Washington, DC. April. Available at http://www.epa.gov/epawaste/nonhaz/industrial/tools/cmtp/index.htm.

U.S. EPA (Environmental Protection Agency). 2003d. *Addendum to the EPACMTP Technical Background Document*. Office of Solid Waste, Washington, DC. Available as document ID number EPA-HQ-RCRA-2003-0001-0166 at http://www.regulations.gov.

U.S. EPA (Environmental Protection Agency). 2003e. *Final Revised Report on Resolution of 18 Previously Indeterminate Candidate Damage Cases*. Prepared by SAIC for U.S. EPA Office of Solid Waste under EPA contract 68-W-02-036, WA No. 1-12. Washington, DC. March 5. Available as document ID number EPA-HQ-RCRA-2003-0003-0420 at http://www.regulations.gov.

U.S. EPA (Environmental Protection Agency). 2003f. *Peer Review Of Changes To EPA's MINTEQA2 Model.* Prepared by Industrial Economics for the Office of Solid Waste, Washington, DC.

U.S. EPA (Environmental Protection Agency). 2005. *Guidelines for Carcinogen Risk Assessment.* EPA/630/P-03/001B. Risk Assessment Forum, U.S. Environmental Protection Agency, Washington, DC. March. Available at http://nepis.epa.gov/EPA/html/Pubs/pubtitleORD.htm.

U.S. EPA (Environmental Protection Agency). 2006a. *Provisional Peer Reviewed Toxicity Values for Aluminum (CASRN 7429-90-5).* 10-23-2006. National Center for Environmental Assessment. Superfund Technical Support Center, Cincinnati, OH.

U.S. EPA (Environmental Protection Agency). 2006b. *Provisional Peer Reviewed Toxicity Values for Iron and Compounds (CASRN 7439-89-6). Derivation of Subchronic and Chronic Oral RfDs.* 9-11-2006. National Center for Environmental Assessment. Superfund Technical Support Center, Cincinnati, OH.

U.S. EPA (Environmental Protection Agency). 2006c. *Characterization of Mercury-Enriched Coal Combustion Residues from Electric Utilities Using Enhanced Sorbents for Mercury Control*. EPA 600/R-06/008. Office of Research and Development. Research Triangle Park, NC. January. Available at http://nepis.epa.gov/EPA/html/Pubs/pubtitleORD.htm.

U.S. EPA (Environmental Protection Agency). 2006d. *Guide for Industrial Waste Management*. Office of Solid Waste, Washington, DC. Available at http://www.epa.gov/waste/nonhaz/industrial/guide/index.htm.

U.S. EPA (Environmental Protection Agency). 2007. *Coal Combustion Waste Damage Case Assessments*. Available from the docket to the Notice of Data Availability on the

Disposal of Coal Combustion Wastes in Landfills and Surface Impoundments, Docket # EPA-HQ-RCRA-2006-0796. June 19.

U.S. EPA (Environmental Protection Agency). 2008a. *Provisional Peer Reviewed Toxicity Values for Cobalt (CASRN 7440-48-4).* 8-25-2008. National Center for Environmental Assessment. Superfund Technical Support Center, Cincinnati, OH.

U.S. EPA (Environmental Protection Agency). 2008b. *Child-Specific Exposure Factors Handbook.* EPA/600/R-06-096F. National Center for Environmental Assessment, Cincinnati, OH.

U.S. EPA (Environmental Protection Agency). 2008c. *Characterization of Coal Combustion Residues from Electric Utilities Using Wet Scrubbers for Multi-Pollutant Control*. EPA/600/R-08/077. Office of Research and Development, Air Pollution Control Division. Research Triangle Park, NC. July. Available at http://www.epa.gov/nrmrl/pubs/600r08077/600r08077.pdf.

U.S. EPA (Environmental Protection Agency). 2009a. Integrated Risk Information System (IRIS). National Center for Environmental Assessment, Office of Research and Development, Washington, DC. Available at http://www.epa.gov/iris/.

U.S. EPA (Environmental Protection Agency). 2009b. *Sensitivity Analysis for the Coal Combustion Waste Risk Assessment*. Draft Technical Report. Prepared by RTI International for U.S. EPA, Office of Solid Waste, Washington, DC.

U.S. EPA (Environmental Protection Agency). 2009c. *Inhalation of Fugitive Dust: A Screening Assessment of the Risks Posed by Coal Combustion Waste Landfills.* Prepared by Office of Solid Waste, Washington, DC.

U.S. EPA (Environmental Protection Agency). 2009d. *Responses to Review Comments on Human and Ecological Risk Assessment for Coal Combustion Wastes.* Prepared by RTI International for U.S. EPA, Office of Solid Waste, Washington, DC.

U.S. EPA (Environmental Protection Agency). 2010. *Regulatory Impact Analysis For EPA's Proposed Regulation of Coal Combustion Wastes Generated by the Electric Utility Industry.* Office of Resource Conservation and Recovery, Washington, DC. April.

Zilmer, M., and P. Fauble. 2004. Groundwater impacts from coal combustion ash disposal sites in Wisconsin. In: *Proceedings of State Regulation of Coal Combustion By-Product Placement at Mine Sites.* May 4–6, 2004, Harrisburg, Pennsylvania. U.S. Department of Interior, Office of Surface Mining, Alton, IL. Available at http://www.mcrcc.osmre.gov/MCR/Resources/ccb/PDF/State_Regulation_of_CCB_Placement.pdf.

Appendix A. Constituent Data

The coal combustion waste (CCW) risk assessment addressed metals and inorganic constituents identified by EPA as potential constituents of concern in CCW (**Table A-1**). EPA-derived waste concentrations for these constituents from the CCW constituent database, which includes analyte concentration data in three tables representing different types of waste samples: landfill leachate analyses (in mg/L), surface impoundment and landfill porewater analyses (in mg/L), and analyses of whole waste samples (in mg/kg). Each database table specifies, for most samples, the type of waste sampled and the type of coal burned at the facility.

Table A-1. Constituents Addressed in the CCW Risk Assessment

Constituent	CAS ID	Constituent	CAS ID
Metals		***Inorganic Anions***	
Aluminum	7429-90-5	Chloride	16887-00-6
Antimony	7440-36-0	Cyanide	57-12-5
Arsenic	7440-38-2	Fluoride	16984-48-8
Barium	7440-39-3	Total Nitrate Nitrogen	14797-55-8
Beryllium	7440-41-7	Phosphate	14265-44-2
Boron	7440-42-8	Silicon	7631-86-9
Cadmium	7440-43-9	Sulfate	14808-79-8
Chromium	7440-47-3	Sulfide	18496-25-8
Cobalt	7440-48-4	***Inorganic Cations***	
Copper	7440-50-8	Ammonia	7664-41-7
Iron	7439-89-6	Calcium	7440-70-2
Lead	7439-92-1	pH	12408-02-5
Magnesium	7439-95-4	Potassium	7440-09-7
Manganese	7439-96-5	Sodium	7440-23-5
Mercury	7439-97-6	***Nonmetallic Elements***	
Molybdenum	7439-98-7	Inorganic Carbon	7440-44-0
Nickel	7440-02-0	Total Elemental Sulfur	7704-34-9
Selenium	7782-49-2	***Measurements***	
Silver	7440-22-4	Total Dissolved Solids	none
Strontium	7440-24-6	Total Organic Carbon	none
Thallium	7440-28-0	Dissolved Organic Carbon	none
Vanadium	7440-62-2		
Zinc	7440-66-6		

A.1 Data Sources

EPA prepared the CCW constituent database in 2002. The 2002 CCW constituent database includes all of the waste characterization data used by EPA in its risk assessments in support of the March 1999 *Report to Congress: Wastes from the Combustion of Fossil Fuels* (the RTC) (U.S. EPA, 1999). In addition to the data set from the March 1999 RTC, EPA supplemented the database with the following data:

- Data submitted with public comments to EPA on the 1999 RTC
- Data submitted with public comments to EPA concerning the May 22, 2000, Final Regulatory Determination
- Data collected by and provided to EPA since the end of the public comment period on the Final Regulatory Determination
- Data identified from literature searches.

The primary sources of these additional data include the electric power industry, state and federal regulatory agencies, and scientific literature. **Attachment A-1** is a complete list of the sources of data contained in the 2002 CCW constituent database.

The additional data represent a significant expansion in the quantity of characterization data available to EPA for analysis. For example, the data set used for the risk assessments supporting the RTC covered approximately 50 CCW generation and/or disposal sites. With the addition of the supplemental data, the 2002 CCW constituent database now covers more than 160 sites. The 1999 data set included approximately 10,000 individual samples of CCW. The 2002 CCW constituent database now includes more than 35,000 individual samples.

The additional data also represent an expansion in the scope of characterization data available to EPA for analysis. The 1999 data were obtained exclusively from the electric power industry. As shown in Attachment A-1, the 2002 data set includes data from other sources, such as scientific literature and state and federal regulatory agencies. The 1999 data set included analyses of whole waste samples, surface impoundment and landfill porewater analyses, and analyses of extracts obtained using the Toxicity Characteristic Leaching Procedure (TCLP), the Synthetic Precipitation Leaching Procedure (SPLP), and Extraction Procedure (EP) Toxicity leaching methods. The 2002 data set added analyses of actual landfill leachate (e.g., obtained from leachate collection systems), analyses of extracts obtained using other leaching methods (including higher retention time leaching methods), and porewater analyses.

The 2002 CCW constituent database represents CCW characteristics across a broad cross-section of the generating universe. Not only does the database include data from a large number of sites, but these sites are distributed throughout the United States, as shown in **Table A-2**. The database includes data for all major types of CCW (i.e., fly ash, bottom ash, flue gas desulfurization [FGD] sludge, fluidized bed combustion [FBC] fly ash, and FBC bed ash), from mixtures of CCW types that are commonly created during disposal operations (e.g., combined fly ash and bottom ash), and from CCW mixed with coal refuse (a common disposal practice). **Section A.2** discusses waste types in more detail.

Table A-2. States Included in the CCW Constituent Database

Alaska	Illinois	Maryland
Arkansas	Indiana	Michigan
California	Kentucky	Ohio
Colorado	Missouri	Oklahoma
Connecticut	North Carolina	Pennsylvania
Florida	North Dakota	Tennessee
Georgia	Nebraska	Texas
Hawaii	New Mexico	Wisconsin
Iowa	Louisiana	West Virginia

The database also includes data for CCW generated from combustion of all major coal ranks: bituminous, sub-bituminous, lignite, and anthracite. Although the database does include coal type designations for most of the entries, in many cases the type is not specified. In addition, many coal plants mix coal from different sources (e.g., eastern and western coals), depending on prices and the need to reduce sulfur levels. As a result, correlations of risk results with coal types may be difficult and may not produce significant results.

A.2 Data Preparation

Table A-3 lists the waste types evaluated in the CCW risk assessment, along with the number of sites representing each waste type in the CCW constituent database. Key steps in preparing these data for risk assessment include (1) selection and grouping of waste types to be addressed, (2) selection of the analyte data to be used, and (3) processing of these data to develop the analyte concentrations for the screening analysis and full-scale risk assessment.

Table A-3. Waste Streams in CCW Constituent Database

Waste Type Waste Streams	Number of Sites by Waste Type[a]		
	Landfill Leachate	Surface Impoundment Porewater	Total Waste
Conventional Combustion Waste	*97*	*13*	*62*
Ash (not otherwise specified)	43	0	30
Fly ash	61	2	33
Bottom ash & slag	24	3	23
Combined fly & bottom ash	7	4	4
FGD sludge	4	6	5
Codisposed Ash & Coal Refuse	9	5	1
Fluidized Bed Combustion Waste	*58*	*0*	*54*
Ash (not otherwise specified)	18	0	10
Fly ash	33	0	32
Bottom and bed ash	26	0	25
Combined fly & bottom ash	20	0	22

[a] Site counts by waste type from leachate, porewater, and whole waste data tables in the 2002 CCW constituent database.

A.2.1 Selection and Grouping of Waste Types of Concern

The CCW constituent database contains a variety of waste types. Some selection and grouping of these types was appropriate so that the risk assessment could evaluate risks consistently for groups of wastes that are expected to behave similarly when disposed in landfills and surface impoundments.

Combustion ash types in the CCW constituent database include fly ash, bottom ash, bed ash, slag, combined fly and bottom ash, and coal ash not otherwise specified. Based on a statistical analysis that showed no significant difference in leachate and porewater chemistry, the analysis combines data for these ash types for landfills and surface impoundments. FGD sludge was also combined with these conventional combustion ash types based on insignificant differences in porewater chemistry and the fact that FGD sludge is usually codisposed with varying amounts of fly ash and bottom ash.

CCW porewater constituent data did show that FBC wastes and codisposed ash and coal refuse (coal waste from coal crushers and other coal preparation and handling operations[1]) differ significantly from coal combustion ash in their composition and leachate chemistry, so these wastes were addressed separately in the risk analysis. FBC waste chemistry is impacted by the limestone injected with coal in FBC units for sulfur capture and tends to be very alkaline with high levels of calcium and sulfate. Coal refuse is high in pyrite, which generates sulfuric acid when disposed. As a result, combustion wastes exhibit a lower pH when codisposed with coal refuse.

A.2.2 Selection of Appropriate Analyte Data

CCW analyte concentration data represent leachate from landfills and surface impoundments and whole waste in landfills, as follows:

- Whole waste analyte concentrations (in mg/kg) represent landfill waste.
- Analyte concentrations (in mg/L) in porewater sampled from surface impoundment sediments represent surface impoundment leachate.
- Analyte concentrations for extracts from leaching methods, analyses of actual landfill leachate, and landfill porewater analyses represent landfill leachate. Because the CCW constituent database includes analyte concentrations from several leaching methods, a decision hierarchy was used to select leachate analyses to use in the risk assessment (**Table A-4**).

As shown in Table A-4, the methods thought to best represent long-term waste monofill porewater composition (i.e., methods with long equilibration times and low liquid-to-solid ratios) represent only a few sites, with most sites having TCLP and/or SPLP measurements. To best represent CCW landfill waste concentration at a wide variety of sites, the hierarchy rank shown in Table A-4 was used to select the best method for a particular site. For sites where two or more

[1] Coal refuse is the waste coal produced from coal handling, crushing, and sizing operations. In the CCW constituent database, codisposed coal refuse includes "combined ash and coal gob", "combined ash and coal refuse", and "combined bottom ash and pyrites".

methods were available in the same rank (which often occurs for SPLP and TCLP analyses), the screening analysis used the method with the highest analyte concentrations. This ensured that the data used in the risk assessment were the best that were available and represent a broad variety of waste disposal conditions.

Table A-4. Comparison/Hierarchy of Leaching Methods for Landfills Represented in CCW Constituent Database

Method (Rank)	Description	Advantages	Disadvantages
Landfill leachate (1)	Direct samples of landfill leachate	Most representative of leachate chemistry	Low number of sites represented
Landfill porewater (1)	Direct porewater samples from landfill	Most representative of leachate chemistry	Low number of sites represented
High retention time and low liquid-to-solid ratio (L:S) methods (2)	Waste extractions with long equilibration times (days to weeks) and low L:S	Better representation of landfill equilibration times and L:S	Low number of sites represented
Low L:S methods (3)	Waste extractions with low L:S	Better representation of landfill L:S	Low number of sites represented; equilibrium times relatively short
High retention time methods (3)	Waste extractions with long equilibration times (days to weeks)	Better representation of landfill equilibration times	Low number of sites represented; L:S relatively high
TCLP (4)	Toxicity Characteristic Leaching Procedure waste extractions	Most representative in terms of number of sites, waste types covered	High L:S (20:1) can dilute leachate concentrations; short equilibration time (18 hours) may not allow equilibrium to develop; Na-acetate buffer can overestimate leaching for some constituents (e.g., Pb)
SPLP (4)	Synthetic Precipitation Leaching Procedure and other dilute water waste extractions	More representative in terms of number of sites, waste types covered; extract similar to precipitation	High L:S (20:1) can dilute leachate concentrations; short equilibration time (18 hours) may not allow equilibrium to develop

A.2.3 Development of Waste Constituent Concentrations

To allow risk assessment results to be organized by waste constituent and waste type, CCW data were processed to produce a single concentration per waste stream (surface impoundment porewater, landfill leachate, and landfill whole waste), analyte, and site for use in the risk assessment. Data processing to prepare these analyte concentrations for the CCW risk assessment involved two steps:

1. **Calculation of average constituent concentrations by site for landfill leachate, surface impoundment porewater, and total ash concentrations.** Site averaging avoids potential bias toward sites with many analyses per analyte. During site averaging, any separate waste disposal scenarios occurring at a site (e.g., non-FBC and FBC ash) were treated as separate "sites" and were averaged independently. This approach is consistent with that used in the 1998 CCW risk analysis. As in 1998, nondetects were averaged at one-half the reported detection limit.

2. **Selection of waste concentrations from site-averaged values.** For the Monte Carlo analysis, the analysis randomly selected, by waste type/waste management unit (WMU) scenario, site-averaged leachate concentrations. For landfills, a corresponding total waste analysis was pulled from the database or calculated from a constituent-specific relationship between landfill leachate and total waste analyses.

A.3 Results

Attachment A-2 provides the site-averaged constituent data used (sampled) in the full-scale CCW risk assessment by waste type/WMU scenario. **Attachment A-3** presents summary statistics, by constituent and WMU type, including the 90th percentile waste concentrations used in the screening analysis. Attachment A-3 also includes figures (**Figures A-3-1** to **A-3-3**) that illustrate the differences between site-averaged and non-site-averaged waste concentrations for surface impoundment porewater (Figure A-3-1), landfill leachate (Figure A-3-2), and total waste analyses (Figure A-3-3).

A.4 References

U.S. EPA (Environmental Protection Agency). 1999. *Report to Congress: Wastes from the Combustion of Fossil Fuels*. EPA 530-5-99-010. Office of Solid Waste and Emergency Response, Washington, DC. March.

Attachment A-1: Sources of Data

General

Alaska Department of Environmental Conservation. 2000. Comments on Regulatory Determination on Wastes from the Combustion of Fossil Fuels, Docket No. F-2000-FF2F-FFFFF and attachments.

ARIPPA (Anthracite Region Independent Power Producers Association). 2000. Occurrence and Fate of Selected Trace Elements in Circulating Fluidized Bed Combustion Byproducts, Volume II of II, Prepared by Earthtech Inc., Johnstown, PA, for ARIPPA. Volume I is available in EPA docket EPA-HQ-RCRA-2008-0329 as document number EPA-HQ-RCRA-2008-0329-0324.

Bhattacharja, S., J.L. Evanko, J.J. Gajda, and R.H. Carty. 2001. Ceramic tiles using Illinois fly ash. In Proceedings of the American Coal Ash Association 14th International Symposium on Management & Use of Coal Combustion Products (CCPs), January 22–26, 2001, San Antonio, TX. Palo Alto, CA: Electric Power Research Institute, EPRI 1001183, Volume 1: Management and Use of Coal Combustion Products (CCPs). Available at http://my.epri.com/portal/server.pt?space=CommunityPage&cached=true&parentname=ObjMgr&parentid=2&control=SetCommunity&CommunityID=404&RaiseDocID=000000000001001158&RaiseDocType=Abstract_id.

Bland, A.E., T.H. Brown, and D.O. Steen. 2001. Flowable fill materials made from lignite fired fluidized bed combustion ash. In *Proceedings of the American Coal Ash Association 14th International Symposium on Management & Use of Coal Combustion Products (CCPs)*, January 22–26, 2001, San Antonio, TX. Palo Alto, CA: Electric Power Research Institute, EPRI 1001183, Volume 1: Management and Use of Coal Combustion Products (CCPs). Available at http://my.epri.com/portal/server.pt?space=CommunityPage&cached=true&parentname=ObjMgr&parentid=2&control=SetCommunity&CommunityID=404&RaiseDocID=00000000001001158&RaiseDocType=Abstract_id.

Butler, R.D., D.F. Pflughoeft-Hassett, B.A. Dockter, and H.J. Foster. 1995. *Stabilization of Underground Mine Voids by Filling with Coal Conversion Residuals*. Energy & Environmental Research Center, University of North Dakota, Grand Forks, ND. 43 pp. Available in EPA docket EPA-HQ-RCRA-2003-0003 as document number EPA-HQ-RCRA-2003-0003-0123.

Dairyland Power Cooperative. 1991. Letter to Wisconsin Department of Natural Resources. May 22, 1991. Available in EPA docket EPA-HQ-RCRA-2003-0003 as document number EPA-HQ-RCRA-2003-0003-0111.

Electric Power Research Institute (EPRI). 1983. *Pilot Study of Time Variability of Elemental Ash Concentrations in Power-Plant Ash*. EPRI-EA-2959, EPRI, Palo Alto, CA. Available in RCRA docket number F-2003-FF4P-FFFFF.

Electric Power Research Institute (EPRI). 1995. *Laboratory Characterization of Atmospheric Fluidized-Bed Combustion By-Products, Final Report*. EPRI-TR-105527, EPRI, Palo Alto, CA. Available at http://my.epri.com/portal/server.pt?Abstract_id=TR-105527.

Electric Power Research Institute (EPRI). 1995. *Land Application of Coal Combustion By-Products: Use in Agriculture and Land Reclamation, Final Report*. EPRI-TR-103298, EPRI, Palo Alto, CA. Available in EPA docket EPA-HQ-RCRA-1999-0022 as document number EPA-HQ-RCRA-1999-0022-0931.

Electric Power Research Institute (EPRI). 1996. *Fluid Placement of Fixated Scrubber Sludge in Abandoned Deep Mines to Abate Surface Subsidence and Reduce Acid Mine Drainage*. EPRI-TR-107053, EPRI, Palo Alto, CA. Available for viewing in EPA docket EPA-HQ-RCRA-2003-0003 as document number EPA-HQ-RCRA-2003-0003-0127.

Electric Power Research Institute (EPRI). 1996. *Mixtures of a Coal Combustion By-Product and Composted Yard Wastes for Use as Soil Substitutes and Amendments, Final Report*. EPRI-TR-106682, EPRI, Palo Alto, CA. Available at http://my.epri.com/portal/server.pt?Abstract_id=TR-106682

Electric Power Research Institute (EPRI). 1998. *Identification of Arsenic Species in Coal Ash Particles*. EPRI-TR-109002, EPRI, Palo Alto, CA. Available at http://www.regulations.gov as document # EPA-HQ-RCRA-2003-0003-0406.

Electric Power Research Institute (EPRI). 1999. *Investigation of Ammonia Adsorption on Fly Ash and Potential Impacts of Ammoniated Ash*. EPRI-TR-113777, EPRI, Palo Alto, CA. Available for viewing in EPA docket #F-2003-FF4P-FFFFF.

Electric Power Research Institute (EPRI). 1999. *Utilization of Coal Combustion By-Products in Agriculture and Land Reclamation, Final Report*. EPRI-TR-112746, EPRI, Palo Alto, CA. Available in EPA docket EPA-HQ-RCRA-1999-0022 as document number EPA-HQ-RCRA-1999-0022-0074.

Electric Power Research Institute (EPRI). 1999. *Guidance for Comanagement of Mill Rejects at Coal-Fired Power Plants*. EPRI-TR-108994, EPRI, Palo Alto, CA. Available for viewing in EPA docket number EPA-HQ-RCRA-1999-0022 as document number EPA-HQ-RCRA-1999-0022-1078.

Electric Power Research Institute (EPRI). 2001. *Environmental Evaluation for Use of Ash in Soil Stabilization Applications*. WO9227-01, EPRI, Palo Alto, CA. Available in EPA docket number F-2003-FF4P-FFFFF.

Finkelman, R.B., C.A. Palmer, and C.F. Eble. 1998. Characterization of hazardous trace elements in solid waste products from a coal burning power plant in Kentucky. In *Characterization of Coal and Coal Combustion Products from a Coal Burning Power Plant – Preliminary Report and Results of Analysis*. Edited by G.N. Breit and R.B. Finkelman. U.S. Geological Survey, Open File Report 98-342, 101 pp. Available at http://pubs.er.usgs.gov/usgspubs/ofr/ofr98342.

Freeman United Coal Mining Company. 1991–2001. Quarterly Ash Analyses. Available in EPA docket number F-2003-FF4P-FFFFF and at http://www.regulations.gov as document numbers EPA-HQ-RCRA-2003-0003-0119, EPA-HQ-RCRA-2003-0003-0411, EPA-

HQ-RCRA-2003-0003-0412, EPA-HQ-RCRA-2003-0003-0413, EPA-HQ-RCRA-2003-0003-0414, EPA-HQ-RCRA-2003-0003-0415, EPA-HQ-RCRA-2003-0003-0416.

Haefner, R.J. 2000. Water quality at an abandoned mine site treated with coal combustion by-products. In *Proceedings of the Use and Disposal of Coal Combustion By-Products at Coal Mines: Technical Interactive Forum*, K.C. Vories and D. Throgmorton, eds. Morgantown, WV, April 10–13, 2000. Available at http://www.techtransfer.osmre.gov/NTTMainSite/Library/proceed/ccb2000/front.pdf.

Hassett, D.J., D.F. Pflughoeft-Hassett, D.L. Laudal, and J.H. Pavlish. 1999. Mercury release from coal combustion by-products to the environment. In *Mercury in the Environment, Proceedings of a Specialty Conference*, Minneapolis, MN, September 15–17, Air & Waste Management Association, pp. 85–93. Available in EPA docket EPA-HQ-RCRA-2003-0003-0114 as document number EPA-HQ-RCRA-2003-0003-0114.

Hower, J.C., J.D. Robertson, and J.M. Roberts. 2001. Coal combustion by products from the co-combustion of coal, tire-derived fuel, and petroleum coke at a western Kentucky cyclone-fired unit. In *Proceedings of the American Coal Ash Association 14th International Symposium on Management & Use of Coal Combustion Products (CCPs)*, January 22–26, 2001, San Antonio, TX. Palo Alto, CA: Electric Power Research Institute, EPRI 1001183, *Volume 1: Management and Use of Coal Combustion Products (CCPs)*. Available at http://my.epri.com/portal/server.pt?space=CommunityPage&cached=true&parentname=ObjMgr&parentid=2&control=SetCommunity&CommunityID=404&RaiseDocID=000000000001001158&RaiseDocType=Abstract_id.

Hower, J.C., J.D. Robertson, U.M. Graham, G.A. Thomas, A.S. Wong, and W.H. Schram. 1993. Characterization of Kentucky coal-combustion by-products: Compositional variations based on sulfur content of feed coal. *Journal of Coal Quality 12*:150–155.

Hower, J.C., A.S. Trimble, C.F. Eble, C.A. Palmer, and A. Kolker. 1997. Characterization of fly ash from low-sulfur and high-sulfur coal sources: Partitioning of carbon and trace elements with particle size. *Energy Sources Part A: Recovery, Utilization and Environmental Effects* 21(6):511–525.

Indiana Department of Natural Resources. 1999. Response to the U.S. Environmental Protection Agency's March 1999 *Report to Congress: Wastes from the Combustion of Fossil Fuels*, Volume 2 of 3. Available in EPA Docket number EPA-HQ-RCRA-1999-0022 as document number EPA-HQ-RCRA-1999-0022-0147.

Kim, A.G., and G. Kazonich. 2001. Release of trace elements from CCB: Maximum extractable fraction. 2001. In *Proceedings of the American Coal Ash Association 14th International Symposium on Management & Use of Coal Combustion Products (CCPs)*, January 22–26, 2001, San Antonio, TX. Palo Alto, CA: Electric Power Research Institute, EPRI 1001183, *Volume 1: Management and Use of Coal Combustion Products (CCPs)*. Available at http://my.epri.com/portal/server.pt?space=CommunityPage&cached=true&parentname=ObjMgr&parentid=2&control=SetCommunity&CommunityID=404&RaiseDocID=000000000001001158&RaiseDocType=Abstract_id.

Kost, D.A., J.P. Vimmerstedt, and R.C. Stehouwer. 1997. Reclamation of Acid, Toxic Coal Spoils Using Wet Flue Gas Desulfurization By-Product, Fly Ash and Sewage Sludge.

Available in EPA docket number EPA-HQ-RCRA-2003-0003 as document number EPA-HQ-RCRA-2003-0003-0113.

Longley, R.D. Jr., Silsbee, M.R., and Elsworth, D. 1997. Acid mine drainage remediation utilizing fly ash based grouts. Presented at the '97 International Ash Utilization Symposium. Available at http://www.ems.psu.edu/~elsworth/publications/proc/1997_p_ash_utilization_conf_longley.pdf.

Loop, C.M. 2000. The Impact of Ash Placement in a Surface Mine Pool on the Chemistry of the Silverbrook Basin. M.S. Thesis, University Park, PA: The Pennsylvania State University.

Mafi, S., M.T. Damian, R.E. Senita, W.C. Jewitt, S. Bair, Y-C. Chin, E. Whitlatch, S. Traina, and W. Wolfe. 1997. *Injection of FGD Grout to Abate Acid Mine Drainage in Underground Coal Mines*. Presented at the Advanced Coal-Based Power and Environmental Systems '97 Conference, Pittsburgh, PA, July 22–24. Publication Number DOE/MC/33092-98/C0946. Washington, DC: U.S. Department of Energy. Available at http://www.osti.gov/bridge/purl.cover.jsp?purl=/611785-iz9wzM/webviewable/.

Naik, T.R., R.N. Kraus, R.F. Sturzl, and B.W. Ramme. 1998. Design and testing controlled low-strength materials (CLSM) using clean coal ash. In *The Design and Application of Controlled Low-Strength Materials (Flowable Fill)*, Edited by A.K. Howard and J.L. Hitch, ASTM STP 1331.

Pennsylvania Department of Environmental Protection (PADEP). 1999. *Pennsylvania Department of Environmental Quality Comments on EPA Report to Congress*, *Volume 2: Coal Ash Used as Minefill – Placement Not in Contact with Groundwater*. Available from Roderick A. Fletcher, Director, Bureau of Mining and Reclamation, PADEP, Rachel Carson State Office Building, P.O. Box 8461, Harrisburg, PA 17105-8461.

Pennsylvania Department of Environmental Protection (PADEP). 1999. *Pennsylvania Department of Environmental Quality Comments on EPA Report to Congress*, *Volume 3: Coal Ash Used as Minefill – Placement in Contact with Groundwater*. Available from Roderick A. Fletcher, Director, Bureau of Mining and Reclamation, PADEP, Rachel Carson State Office Building, P.O. Box 8461, Harrisburg, PA 17105-8461.

Pennsylvania Department of Environmental Protection (PADEP). 2001. Coal Ash Data CD.

Pflughoeft-Hassett, D.F. 1997. *IGCC and PFBC By-Products: Generation, Characteristics, and Management Practices*. Report Number 97-EERC-09-06. Prepared by the Energy & Environmental Research Center, University of North Dakota, for the U.S. Environmental Protection Agency, Office of Solid Waste. Available at http://www.undeerc.org/carrc/Assets/IGCCandPFBC.pdf.

Pflughoeft-Hassett, D.F., B.A. Dockter, D.J. Hassett, K.E. Eylands, and L.L. Hamre. 2000. *Use of Bottom Ash and Fly Ash in Rammed-Earth Construction*. Report No. 2000-EERC-09-03, Energy & Environmental Research Center, University of North Dakota, Grand Forks, ND. Available at http://www.undeerc.org/carrc/Assets/RammedEarth.pdf.

PPL Generation, LLC. 1994–2001. Coal Ash Quality Results. Available at http://www.regulations.gov as document number EPA-HQ-RCRA-2003-0003-0110.

Radian Corporation. 1994. *Field Study for Disposal of Solid Wastes from Advanced Coal Processes: Ohio LIMB Site Assessment, Volume I: Text, Final Report*. Available at http://regulations.gov as document number EPA-HQ-RCRA-2003-0002-0442.

Radian Corporation. 1994. *Field Study of Circulating Fluidized-Bed Ash, Final Report*. Available at http://www.regulations.gov as document numbers EPA-HQ-RCRA-2003-0002-0035, EPA-HQ-RCRA-2003-0002-0436, and EPA-HQ-RCRA-2003-0002-0437.

Radian Corporation. 1997. *Field Monitoring of Advanced Coal By-Products: Illinois Site, Volume II: Appendices, Final Report*. Available at http://www.regulations.gov as document numbers EPA-HQ-RCRA-2003-0002-0040, EPA-HQ-RCRA-2003-0002-0423, and EPA-HQ-RCRA-2003-0002-0424

Reliant Energy. 2001. *Coal Ash Analyses for Conemaugh, Keystone, Portland, Titus, Seward, and Shawville Facilities & Groundwater Assessments for Disposal Sites at Keystone, Conemaugh, Portland, and Shawville*.

Science Applications International Corporation (SAIC). 1999. Memorandum to ERG, Regarding Maryland/West Virginia Mine Site Visit Summary. Available in EPA docket EPA-HQ-RCRA-2003-0003-0107 as document number EPA-HQ-RCRA-2003-0003-0107.

Sellakumar, K.M., R. Conn, and A. Bland. 1999. A comparison study of ACFB and PCFB ash characteristics. Paper presented at the 6th International Conference on Circulating Fluidized Beds, Wurzburg, Germany, August 22–27, 1999. Available at http://www.fwc.com/publications/tech_papers/powgen/pdfs/cfb6ash.pdf.

Turris Coal Company. 1993–2001. Quarterly Ash Analyses. Available at http://www.regulations.gov as document numbers EPA-HQ-RCRA-2003-0120, EPA-HQ-RCRA-2003-0120, EPA-HQ-RCRA-2003-0409, and EPA-HQ-RCRA-2003-0410.

Zeigler Coal Holding Company. 1994–2000. Quarterly Ash Analyses.

Report to Congress

FBC Leachate:

Council of Industrial Boiler Owners. 1997. Responses to Fossil Fuel Fluidized Bed Combustion By-Products Survey. November.

FBC Total:

Council of Industrial Boiler Owners. 1997. Responses to Fossil Fuel Fluidized Bed Combustion By-Products Survey. November.

Porewater:

Electric Power Research Institute (EPRI). 1991. *Comanagement of Coal Combustion By-Products and Low-Volume Wastes: A Southeastern Site. Final Report*. EPRI-EN-7545. EPRI, Palo Alto, CA. November. (“L” Site).

Electric Power Research Institute (EPRI). 1992. *Comanagement of Coal Combustion By-Products and Low-Volume Wastes: A Midwestern Site. Interim Report*. EPRI-TR-100955, EPRI, Palo Alto, CA. August. (“C” Site).

Electric Power Research Institute (EPRI). 1996. *Comanagement of Coal Combustion By-Products and Low-Volume Wastes: A Southwestern Site. Final Report*. EPRI-TR-105673, EPRI, Palo Alto, CA. June. ("O" Site).

Electric Power Research Institute (EPRI). 1996. *Evaluation of the Co-Management of High Volume Combustion By-Products with Other Utility Wastes: CL Site. Draft Report*. EPRI, Palo Alto, CA. May (updated December).

Electric Power Research Institute (EPRI). 1997. *Evaluation of the Co-Management of High Volume Combustion By-Products with Other Utility Wastes: MO Site. Draft Report*, EPRI, Palo Alto, CA. April.

Electric Power Research Institute (EPRI). 1997. *Evaluation of the Co-Management of High Volume Combustion By-Products with Other Utility Wastes: AP Site. Draft Report*. EPRI, Palo Alto, CA. June.

Electric Power Research Institute (EPRI). 1997. *Evaluation of the Co-Management of High Volume Combustion By-Products with Other Utility Wastes: SX Site. Draft Report*. EPRI-TR-001015, EPRI, Palo Alto, CA. June.

Electric Power Research Institute (EPRI). 1997. *Evaluation of the Co-Management of High Volume Combustion By-Products with Other Utility Wastes: P4 Site. Final Report*. EPRI-TR-108420. July.

Electric Power Research Institute (EPRI). 1997. *Evaluation of the Co-Management of High Volume Combustion By-Products with Other Utility Wastes: OK Site. Draft Report*. EPRI-TR-108421. EPRI, Palo Alto, CA. July.

Electric Power Research Institute (EPRI). 1997. *Evaluation of the Co-Management of High Volume Combustion By-Products with Other Utility Wastes: HA Site. Draft Report*. EPRI, Palo Alto, CA. August.

Electric Power Research Institute (EPRI). 1997. *Evaluation of the Co-Management of High Volume Combustion By-Products with Other Utility Wastes: FC Site. Draft Report*. EPRI-TR-108425. EPRI, Palo Alto, CA. August.

Electric Power Research Institute (EPRI). 1997. *Evaluation of the Co-Management of High Volume Combustion By-Products with Other Utility Wastes: LS Site. Final Report*. EPRI-TR-108422. EPRI, Palo Alto, CA. August.

Electric Power Research Institute (EPRI). 1997. *Evaluation of the Co-Management of High Volume Combustion By-Products with Other Utility Wastes: PA Site. Final Report*. EPRI-TR-108427. EPRI, Palo Alto, CA. August.

Electric Power Research Institute (EPRI). 1997. *Evaluation of the Co-Management of High Volume Combustion By-Products with Other Utility Wastes: CY Site. Final Report*. EPRI-TR-108426. EPRI, Palo Alto, CA. August.

Electric Power Research Institute (EPRI). 1994. *Calcium-Based Flue Gas Desulfurization Sludge Disposal Sites. Final Report*. EPRI-TR-103914. EPRI, Palo Alto, CA. April.

Electric Power Research Institute (EPRI). 1994. *Sodium-Based Flue Gas Desulfurization Sludge Disposal Sites. Final Report*. EPRI-TR-103915. EPRI, Palo Alto, CA. April.

TCLP&SPLP:

Electric Power Research Institute (EPRI). 1998. *TCLP & SPLP Data for Comanaged Coal Combustion Wastes*. EPRI, Palo Alto, CA. August.

[This page intentionally left blank.]

Attachment A-2: CCW Constituent Data

Site/Waste Type	WMU Type	Chemical	Leachate (mg/L)	No. of Leachate Measure-ments	No. of Leachate Non-detects	Total (mg/kg)
11 - FBC	LF	Arsenic	0.002916667	3	3	51
11 - FBC	LF	Barium	0.339166667	3	3	174.5
11 - FBC	LF	Cadmium	0.0005	4	4	6.91875
11 - FBC	LF	Lead	0.0025	4	4	39.5
11 - FBC	LF	Mercury	0.00125	4	4	0.1325
11 - FBC	LF	Selenium	0.00225	4	2	45.5
12 - FBC	LF	Aluminum	3.4	1	0	35874.6
12 - FBC	LF	Antimony	0.27	1	0	18
12 - FBC	LF	Arsenic	0.02205	2	0	57.64333333
12 - FBC	LF	Barium	0.196	2	1	203.805
12 - FBC	LF	Boron	0.05	1	1	20.324
12 - FBC	LF	Cadmium	0.005625	2	1	0.279375
12 - FBC	LF	Lead	0.025	1	1	45.66666667
12 - FBC	LF	Mercury	0.00005	2	2	1.2575
12 - FBC	LF	Molybdenum	0.21	1	0	15.5
12 - FBC	LF	Selenium	0.04355	2	0	7.365833333
17 - FBC	LF	Aluminum	4.788	5	0	46194.8
17 - FBC	LF	Antimony	0.0708	5	2	14.60333333
17 - FBC	LF	Arsenic	0.1378	5	0	71.46666667
17 - FBC	LF	Barium	0.3512	5	1	134.975
17 - FBC	LF	Boron	0.4404	5	1	34.06333333
17 - FBC	LF	Cadmium	0.0434	5	2	3.058333333
17 - FBC	LF	Lead	0.2372	5	2	49.65
17 - FBC	LF	Mercury	0.01022	5	5	1.60345
17 - FBC	LF	Molybdenum	0.097	5	1	3.515
17 - FBC	LF	Selenium	0.06315	5	2	3.301666667
18 - FBC	LF	Aluminum	1.333333333	3	0	23501.33333
18 - FBC	LF	Antimony	0.025	3	3	5
18 - FBC	LF	Arsenic	0.025	3	3	53.33333333
18 - FBC	LF	Barium	0.175	3	1	211.3333333
18 - FBC	LF	Boron	1.341666667	3	1	532.3333333
18 - FBC	LF	Cadmium	0.025	3	3	2.5
18 - FBC	LF	Cobalt	0.025	3	3	11
18 - FBC	LF	Lead	0.025	3	3	22
18 - FBC	LF	Mercury	0.0005	3	2	0.268333333
18 - FBC	LF	Molybdenum	0.175	3	1	7.666666667

(continued)

CCW Constituent Data (continued)

Site/Waste Type	WMU Type	Chemical	Leachate (mg/L)	No. of Leachate Measure-ments	No. of Leachate Non-detects	Total (mg/kg)
18 – FBC	LF	Selenium	0.108333333	3	1	0.5
18 - FBC	LF	Thallium	0.025	3	3	1
19 - FBC	LF	Arsenic	0.0875	2	1	6.25
19 - FBC	LF	Barium	0.27	2	1	39.2
19 - FBC	LF	Cadmium	0.01375	2	2	2.5
19 - FBC	LF	Lead	0.0675	2	2	3.75
19 - FBC	LF	Mercury	0.00125	2	1	0.125
19 - FBC	LF	Selenium	0.06875	2	2	6.25
20 - FBC	LF	Aluminum	10.81	12	0	34329.16522
20 - FBC	LF	Antimony	0.787	10	0	46.28125
20 - FBC	LF	Arsenic	0.035	12	0	15.03130435
20 - FBC	LF	Barium	0.381818182	11	0	255.4608696
20 - FBC	LF	Boron	0.457142857	7	0	28.0025
20 - FBC	LF	Cadmium	0.03625	8	0	2.089166667
20 - FBC	LF	Lead	0.301111111	9	0	36.20052632
20 - FBC	LF	Mercury	0.29	1	0	0.454
20 - FBC	LF	Molybdenum	0.392857143	7	0	12.10111111
20 - FBC	LF	Selenium	0.088571429	7	0	4.177333333
21 - FBC	LF	Aluminum	1.91	3	0	14677.33167
21 - FBC	LF	Antimony	0.001833333	3	3	1.083333333
21 - FBC	LF	Arsenic	0.012	3	0	10.76666667
21 - FBC	LF	Barium	0.022333333	3	2	176.2666667
21 - FBC	LF	Boron	0.036666667	3	2	14.38333333
21 - FBC	LF	Cadmium	0.002083333	3	3	0.145833333
21 - FBC	LF	Cobalt	0.008333333	3	2	5.756666667
21 - FBC	LF	Lead	0.009166667	3	3	27.3
21 - FBC	LF	Mercury	0.000133333	3	2	0.431666667
21 - FBC	LF	Molybdenum	0.0125	3	3	3.708333333
21 - FBC	LF	Selenium	0.016666667	3	0	10.9
2-18 - Ash	LF	Arsenic	0.41794375	16	3	
2-18 - Ash	LF	Barium	0.4305625	16	0	
2-18 - Ash	LF	Boron	1.0160625	16	0	
2-18 - Ash	LF	Cadmium	0.05825	16	11	
2-18 - Ash	LF	Lead	0.2819375	16	11	
2-18 - Ash	LF	Mercury	0.000115625	16	16	
2-18 - Ash	LF	Selenium	0.01534375	16	8	
22 - FBC	LF	Arsenic	0.055	5	3	
22 - FBC	LF	Barium	0.5405	5	1	
22 - FBC	LF	Cadmium	0.003	5	5	
22 - FBC	LF	Lead	0.015	5	5	
22 - FBC	LF	Mercury	0.0002	5	3	

(continued)

CCW Constituent Data (continued)

Site/Waste Type	WMU Type	Chemical	Leachate (mg/L)	No. of Leachate Measure-ments	No. of Leachate Non-detects	Total (mg/kg)
22 - FBC	LF	Molybdenum	0.0125	2	2	

(continued)

22 - FBC	LF	Selenium	0.032	5	5	
23 - FBC	LF	Barium	0.81	4	0	
25 - FBC	LF	Arsenic	0.125	1	1	
25 - FBC	LF	Barium	2.5	1	1	
25 - FBC	LF	Cadmium	0.025	1	1	
25 - FBC	LF	Lead	0.125	1	1	
25 - FBC	LF	Mercury	0.005	1	1	
25 - FBC	LF	Selenium	0.025	1	1	
28 - FBC	LF	Barium	2.525	2	0	235.11875
30 - FBC	LF	Aluminum	6.894555556	18	7	28246.46923
30 - FBC	LF	Antimony	0.548082353	17	2	61.49315385
30 - FBC	LF	Arsenic	0.050694444	18	3	48.55980769
30 - FBC	LF	Barium	0.286388889	18	6	120.0687692
30 - FBC	LF	Boron	0.31759375	16	7	30.83913462
30 - FBC	LF	Cadmium	0.023125	14	3	1.916230769
30 - FBC	LF	Lead	0.240805556	18	4	39.36092308
30 - FBC	LF	Mercury	0.000744444	18	17	10.91689923
30 - FBC	LF	Molybdenum	0.138125	16	10	14.50257692
30 - FBC	LF	Selenium	0.10475	16	10	5.603596154
31 - FBC	LF	Aluminum	0.28	1	0	29437.5
31 - FBC	LF	Antimony	0.00065	1	1	5.0325
31 - FBC	LF	Arsenic	0.0687	4	2	26.825
31 - FBC	LF	Barium	0.58275	4	0	170.25
31 - FBC	LF	Boron	26.7	1	0	930
31 - FBC	LF	Cadmium	0.02775	4	3	5.45
31 - FBC	LF	Cobalt	0.0065	1	0	6.42
31 - FBC	LF	Lead	0.03025	4	3	1.19
31 - FBC	LF	Mercury	0.00095	4	1	0.61
31 - FBC	LF	Molybdenum	0.085	1	0	8
31 - FBC	LF	Selenium	0.06485	4	2	7.54
32 - FBC	LF	Arsenic	0.35	1	1	1.4
32 - FBC	LF	Barium	0.085	1	0	
32 - FBC	LF	Cadmium	0.005	1	1	0.009
32 - FBC	LF	Lead	0.05	1	1	0.45
32 - FBC	LF	Mercury	0.0001	1	1	0.03
32 - FBC	LF	Selenium	0.175	1	1	3.5
33 - FBC	LF	Arsenic	0.015	1	1	
33 - FBC	LF	Barium	42	1	0	
33 - FBC	LF	Boron	0.06	1	0	

(continued)

CCW Constituent Data (continued)

Site/Waste Type	WMU Type	Chemical	Leachate (mg/L)	No. of Leachate Measure-ments	No. of Leachate Non-detects	Total (mg/kg)
33 - FBC	LF	Cadmium	0.00125	1	1	
33 - FBC	LF	Cobalt	0.0025	1	1	
33 - FBC	LF	Mercury	0.00005	1	1	
33 - FBC	LF	Selenium	0.01	1	1	
35 - FBC	LF	Arsenic	0.015	1	1	
35 - FBC	LF	Barium	2.6	1	0	
35 - FBC	LF	Cadmium	0.009	1	0	
35 - FBC	LF	Lead	0.035	1	1	
35 - FBC	LF	Mercury	0.00025	1	1	
35 - FBC	LF	Selenium	0.2	1	0	
37 - FBC	LF	Arsenic	0.011102941	17	9	5.79
37 - FBC	LF	Barium	2.104705882	17	2	
37 - FBC	LF	Boron	1.125	5	1	15.9
37 - FBC	LF	Cadmium	0.046176471	17	4	4.183333333
37 - FBC	LF	Cobalt	0.246	5	0	
37 - FBC	LF	Lead	0.287352941	17	6	55
37 - FBC	LF	Mercury	0.001314706	17	4	0.01125
37 - FBC	LF	Selenium	0.01075	17	9	3.42
38 - FBC	LF	Aluminum	2.256666667	9	2	26711.25
38 - FBC	LF	Antimony	0.213069444	9	6	11.27770833
38 - FBC	LF	Arsenic	0.024554444	9	3	25.136075
38 - FBC	LF	Barium	0.178888889	9	4	181.0083333
38 - FBC	LF	Boron	0.346555556	9	2	26.98916667
38 - FBC	LF	Cadmium	0.007388889	9	5	0.71625
38 - FBC	LF	Cobalt	0.008566667	3	2	4.515
38 - FBC	LF	Lead	0.0565	9	6	28.54166667
38 - FBC	LF	Mercury	0.000344444	9	8	0.18195
38 - FBC	LF	Molybdenum	0.177375	8	2	14.1875
38 - FBC	LF	Selenium	0.088561111	9	4	7.682450833
39 - FBC	LF	Arsenic	0.075	1	1	14.5
39 - FBC	LF	Barium	0.395	2	1	590
39 - FBC	LF	Boron	0.76	1	0	
39 - FBC	LF	Cadmium	0.005	1	1	0.5
39 - FBC	LF	Lead	0.025	1	1	15
39 - FBC	LF	Mercury	0.00025	1	1	0.17
39 - FBC	LF	Molybdenum	0.14	1	0	13.5
39 - FBC	LF	Selenium	0.025	1	1	21.5
4 - FBC	LF	Aluminum	13.556	5	0	16084.68429
4 - FBC	LF	Antimony	0.2236	5	2	26.78817857
4 - FBC	LF	Arsenic	0.271	5	0	28.03585714
4 - FBC	LF	Barium	0.6346	5	1	154.95

(continued)

CCW Constituent Data (continued)

Site/Waste Type	WMU Type	Chemical	Leachate (mg/L)	No. of Leachate Measure-ments	No. of Leachate Non-detects	Total (mg/kg)
4 - FBC	LF	Boron	0.693	4	0	13.026
4 - FBC	LF	Cadmium	0.0115	5	2	0.646539286
4 - FBC	LF	Mercury	0.00005	5	5	0.087192857
4 - FBC	LF	Lead	0.1834	5	1	18.35671429
4 - FBC	LF	Molybdenum	0.286666667	3	0	16.18257143
4 - FBC	LF	Selenium	0.0620625	4	2	1.505421429
41 - FBC	LF	Antimony	0.025	5	5	1.551333333
41 - FBC	LF	Arsenic	0.035471698	53	50	13.72255319
41 - FBC	LF	Barium	0.095694444	54	25	19.05490196
41 - FBC	LF	Cadmium	0.022355769	52	51	0.427826087
41 - FBC	LF	Lead	0.017548077	52	51	0.935208333
41 - FBC	LF	Mercury	0.000596154	52	50	0.119542553
41 - FBC	LF	Selenium	0.024433962	53	51	1.505744681
41 - FBC	LF	Thallium	0.031	5	4	3.662790698
42 - FBC	LF	Arsenic	0.0125	2	2	
42 - FBC	LF	Barium	0.1625	2	1	
42 - FBC	LF	Cadmium	0.005	2	2	
42 - FBC	LF	Lead	0.0075	2	2	
42 - FBC	LF	Mercury	0.0005	2	2	
42 - FBC	LF	Selenium	0.0125	2	2	
43 - FBC	LF	Arsenic	0.0125	2	2	
43 - FBC	LF	Barium	0.0875	2	1	
43 - FBC	LF	Cadmium	0.005	2	2	
43 - FBC	LF	Lead	0.0075	2	2	
43 - FBC	LF	Mercury	0.0005	2	2	
43 - FBC	LF	Selenium	0.08625	2	1	
6 - FBC	LF	Aluminum	0.1525	2	1	42736.5
6 - FBC	LF	Antimony	0.05	2	2	16.25
6 - FBC	LF	Arsenic	0.09125	2	1	126.6
6 - FBC	LF	Barium	0.285	2	0	221.5
6 - FBC	LF	Boron	0.1425	2	1	73.8
6 - FBC	LF	Cadmium	0.0025	2	2	1.29625
6 - FBC	LF	Lead	0.01375	2	2	8.1125
6 - FBC	LF	Mercury	0.00005	2	2	1.16
6 - FBC	LF	Molybdenum	0.09	2	0	1.425
6 - FBC	LF	Selenium	0.1025	2	1	84.5625
Amerikohl - FBC	LF	Aluminum	0.753333333	3	0	51600
Amerikohl - FBC	LF	Antimony	0.345	3	3	20
Amerikohl - FBC	LF	Arsenic	0.024166667	3	3	114
Amerikohl - FBC	LF	Barium	0.1	3	3	140
Amerikohl - FBC	LF	Boron	0.346666667	3	1	60

(continued)

CCW Constituent Data (continued)

Site/Waste Type	WMU Type	Chemical	Leachate (mg/L)	No. of Leachate Measure-ments	No. of Leachate Non-detects	Total (mg/kg)
Amerikohl - FBC	LF	Cadmium	0.004166667	3	3	0.15
Amerikohl - FBC	LF	Cobalt	0.175	3	3	30
Amerikohl - FBC	LF	Lead	0.009166667	3	3	23
Amerikohl - FBC	LF	Mercury	0.0005	3	3	0.15
Amerikohl - FBC	LF	Molybdenum	0.266666667	3	1	10
Amerikohl - FBC	LF	Nitrate/Nitrite	3.15	3	3	
Amerikohl - FBC	LF	Selenium	0.044166667	3	3	3.5
Arkwright - Ash	LF	Arsenic	0.07	1	0	
Arkwright - Ash	LF	Barium	0.4	1	0	
Arkwright - Ash	LF	Cadmium	0.01	1	0	
Arkwright - Ash	LF	Lead	0.04	1	0	
Arkwright - Ash	LF	Selenium	0.02	1	0	
Barry - Ash	LF	Arsenic	1	1	0	
Barry - Ash	LF	Barium	0.7	1	0	
Barry - Ash	LF	Cadmium	0.005	1	0	
Barry - Ash	LF	Lead	0.04	1	0	
Barry - Ash	LF	Selenium	0.07	1	0	
Belle Ayr - Ash	LF	Aluminum	0.036666667	3	0	
Belle Ayr - Ash	LF	Antimony	0.021	2	0	
Belle Ayr - Ash	LF	Arsenic	0.181	3	0	
Belle Ayr - Ash	LF	Barium	1.163333333	3	0	
Belle Ayr - Ash	LF	Cobalt	0.0075	2	0	
Belle Ayr - Ash	LF	Molybdenum	0.325	3	0	
Belle Ayr - Ash	LF	Selenium	0.652333333	3	0	
Big Gorilla Pit - FBC	LF	Aluminum	3.774166667	12	0	18440.58824
Big Gorilla Pit - FBC	LF	Antimony	0.037166667	12	1	1.244485294
Big Gorilla Pit - FBC	LF	Arsenic	0.023181818	22	21	7.534117647
Big Gorilla Pit - FBC	LF	Barium	0.243636364	11	3	147.7320588
Big Gorilla Pit - FBC	LF	Boron	0.677916667	12	2	29.64058824
Big Gorilla Pit - FBC	LF	Cadmium	0.015227273	22	22	0.58728125
Big Gorilla Pit - FBC	LF	Cobalt	0.008553571	14	11	2.374214286
Big Gorilla Pit - FBC	LF	Lead	0.08125	12	7	19.51823529
Big Gorilla Pit - FBC	LF	Mercury	0.001704545	22	19	0.302990909
Big Gorilla Pit - FBC	LF	Molybdenum	0.1202	10	1	6.429333333
Big Gorilla Pit - FBC	LF	Nitrate/Nitrite	1.755857143	14	3	
Big Gorilla Pit - FBC	LF	Selenium	0.10975	12	1	7.159397059
Bowen - Ash	LF	Arsenic	0.6	1	0	68
Bowen - Ash	LF	Barium	0.3	1	0	974
Bowen - Ash	LF	Cadmium	0.01	1	0	0.7
Bowen - Ash	LF	Lead	0.04	1	0	63.9
Bowen - Ash	LF	Selenium	0.1	1	0	

(continued)

CCW Constituent Data (continued)

Site/Waste Type	WMU Type	Chemical	Leachate (mg/L)	No. of Leachate Measure-ments	No. of Leachate Non-detects	Total (mg/kg)
Branch - Ash	LF	Arsenic	0.04	1	0	
Branch - Ash	LF	Barium	0.5	1	0	
Branch - Ash	LF	Cadmium	0.01	1	0	
Branch - Ash	LF	Lead	0.04	1	0	
Branch - Ash	LF	Selenium	0.06	1	0	
Buckheart Mine - Ash	LF	Antimony	0.01854	40	14	
Buckheart Mine - Ash	LF	Arsenic	0.122357143	42	13	
Buckheart Mine - Ash	LF	Barium	0.364809524	42	0	
Buckheart Mine - Ash	LF	Boron	9.998738095	42	0	
Buckheart Mine - Ash	LF	Cadmium	0.0235	42	8	
Buckheart Mine - Ash	LF	Cobalt	0.048047619	42	17	
Buckheart Mine - Ash	LF	Lead	0.27887619	42	9	
Buckheart Mine - Ash	LF	Mercury	0.000107143	42	40	
Buckheart Mine - Ash	LF	Selenium	0.118266667	42	26	
Buckheart Mine - Ash	LF	Thallium	0.017875	40	10	
Buckheart Mine - FBC	LF	Antimony	0.0018125	8	8	
Buckheart Mine - FBC	LF	Arsenic	0.0465	8	5	
Buckheart Mine - FBC	LF	Barium	0.560125	8	1	
Buckheart Mine - FBC	LF	Boron	3.157	8	0	
Buckheart Mine - FBC	LF	Cadmium	0.0033125	8	7	
Buckheart Mine - FBC	LF	Cobalt	0.02875	8	7	
Buckheart Mine - FBC	LF	Lead	0.036	8	4	
Buckheart Mine - FBC	LF	Mercury	0.0005	8	4	
Buckheart Mine - FBC	LF	Selenium	0.050625	8	5	
Buckheart Mine - FBC	LF	Thallium	0.001	8	8	
CAER - Ash	LF	Arsenic	1.132	5	0	77.32222222
CAER - Ash	LF	Barium	0.315	5	0	537.6666667
CAER - Ash	LF	Cadmium	0.0942	5	0	
CAER - Ash	LF	Lead	0.1	5	2	73.62375
CAER - Ash	LF	Mercury	0.00025	5	5	
CAER - Ash	LF	Selenium	0.103	5	0	
Canton Site - Ash	LF	Aluminum	9.818127778	36	0	
Canton Site - Ash	LF	Arsenic	0.0025	2	2	
Canton Site - Ash	LF	Barium	3.0156	10	0	
Canton Site - Ash	LF	Boron	18.62468571	35	0	
Canton Site - Ash	LF	Cadmium	0.0005	2	2	
Canton Site - Ash	LF	Cobalt	0.02	1	1	
Canton Site - Ash	LF	Lead	0.1865	2	0	
Canton Site - Ash	LF	Mercury	0.0001	1	1	
Canton Site - Ash	LF	Molybdenum	30.9359	20	0	
Canton Site - Ash	LF	Nitrate/Nitrite	0.095	1	0	

(continued)

CCW Constituent Data (continued)

Site/Waste Type	WMU Type	Chemical	Leachate (mg/L)	No. of Leachate Measure-ments	No. of Leachate Non-detects	Total (mg/kg)
Canton Site - Ash	LF	Selenium	0.0374	1	0	
Canton Site - FBC	LF	Aluminum	2.461866667	24	0	
Canton Site - FBC	LF	Arsenic	0.005	1	1	
Canton Site - FBC	LF	Barium	0.02	1	0	
Canton Site - FBC	LF	Boron	1.5602625	16	0	
Canton Site - FBC	LF	Cadmium	0.066	1	0	
Canton Site - FBC	LF	Lead	0.062	1	0	
Canton Site – FBC	LF	Mercury	0.0005	1	1	
Canton Site – FBC	LF	Molybdenum	1.768009524	21	0	
Canton Site – FBC	LF	Selenium	0.005	1	1	
Central Cleaning Plant - Ash	LF	Antimony	0.008205882	17	17	
Central Cleaning Plant - Ash	LF	Arsenic	0.005	17	17	
Central Cleaning Plant - Ash	LF	Barium	0.168164706	17	0	
Central Cleaning Plant - Ash	LF	Boron	7.213823529	17	0	
Central Cleaning Plant - Ash	LF	Cadmium	0.004117647	17	16	
Central Cleaning Plant - Ash	LF	Cobalt	0.019588235	17	15	
Central Cleaning Plant - Ash	LF	Lead	0.022782353	17	11	
Central Cleaning Plant - Ash	LF	Mercury	0.000568824	17	11	
Central Cleaning Plant - Ash	LF	Selenium	0.040211765	17	0	
Central Cleaning Plant - Ash	LF	Thallium	0.005	17	17	
CL - Ash and Coal Refuse	LF	Aluminum	2.58	3	0	
CL - Ash and Coal Refuse	LF	Antimony	0.0041	3	0	
CL - Ash and Coal Refuse	LF	Arsenic	0.121266667	3	0	
CL - Ash and Coal Refuse	LF	Barium	3.63	3	0	
CL - Ash and Coal Refuse	LF	Boron	0.103133333	3	0	
CL - Ash and Coal Refuse	LF	Cadmium	0.001	3	0	
CL - Ash and Coal Refuse	LF	Cobalt	0.006066667	3	1	
CL - Ash and Coal Refuse	LF	Lead	0.003533333	3	0	
CL - Ash and Coal Refuse	LF	Mercury	0.00005	6	6	
CL - Ash and Coal Refuse	LF	Selenium	0.0452	3	0	
CL - Ash and Coal Refuse	LF	Thallium	0.003483333	3	1	
Coal Creek - Ash	LF	Arsenic	0.0109	2	0	0.086
Coal Creek - Ash	LF	Barium	0.6105	2	0	4.76
Coal Creek - Ash	LF	Boron	6.22	2	0	1.1105
Coal Creek - Ash	LF	Cadmium	0.00015	2	2	0.00045
Coal Creek - Ash	LF	Lead	0.001	2	2	0.02025
Coal Creek - Ash	LF	Mercury	0.000005	2	2	0.0006
Coal Creek - Ash	LF	Selenium	0.0555	2	1	0.00505
Colver Site - FBC	LF	Aluminum	0.248333333	6	1	78878.83333
Colver Site - FBC	LF	Antimony	0.196666667	6	2	166.5
Colver Site - FBC	LF	Arsenic	0.0875	6	1	124.2

(continued)

CCW Constituent Data (continued)

Site/Waste Type	WMU Type	Chemical	Leachate (mg/L)	No. of Leachate Measure-ments	No. of Leachate Non-detects	Total (mg/kg)
Colver Site - FBC	LF	Barium	0.291666667	6	0	443.8333333
Colver Site - FBC	LF	Boron	0.261666667	6	1	62.6
Colver Site - FBC	LF	Cadmium	0.016666667	6	2	9.994166667
Colver Site - FBC	LF	Lead	0.190833333	6	2	192.075
Colver Site - FBC	LF	Mercury	0.00015	6	5	0.586666667
Colver Site - FBC	LF	Molybdenum	0.143333333	6	0	30.65833333
Colver Site - FBC	LF	Selenium	0.48	6	1	68.70833333
Conemaugh - Ash	LF	Aluminum	1.245	2	0	
Conemaugh - Ash	LF	Antimony	0.075	1	1	
Conemaugh - Ash	LF	Arsenic	0.388333333	3	1	
Conemaugh - Ash	LF	Barium	0.331666667	3	0	
Conemaugh - Ash	LF	Boron	0.91	1	0	
Conemaugh - Ash	LF	Cadmium	0.01	3	0	
Conemaugh - Ash	LF	Cobalt	0.026	1	0	
Conemaugh - Ash	LF	Lead	0.1	2	2	
Conemaugh - Ash	LF	Mercury	0.00055	2	2	
Conemaugh - Ash	LF	Molybdenum	0.355	2	0	
Conemaugh - Ash	LF	Selenium	0.295	2	1	
Conemaugh - Ash	LF	Thallium	0.024	1	0	
Conemaugh - Ash and Coal Refuse	LF	Aluminum	1.467666667	3	0	
Conemaugh - Ash and Coal Refuse	LF	Antimony	0.075	3	3	
Conemaugh - Ash and Coal Refuse	LF	Arsenic	0.625	2	2	
Conemaugh - Ash and Coal Refuse	LF	Barium	0.145666667	3	0	
Conemaugh - Ash and Coal Refuse	LF	Boron	0.095	2	0	
Conemaugh - Ash and Coal Refuse	LF	Cadmium	0.002	3	3	
Conemaugh - Ash and Coal Refuse	LF	Cobalt	0.009	1	0	
Conemaugh - Ash and Coal Refuse	LF	Lead	0.073333333	3	2	
Conemaugh - Ash and Coal Refuse	LF	Mercury	0.0004	3	2	
Conemaugh - Ash and Coal Refuse	LF	Molybdenum	0.01	1	0	
Conemaugh - Ash and Coal Refuse	LF	Selenium	0.179833333	3	1	
Conemaugh - Ash and Coal Refuse	LF	Thallium	0.005	1	0	
Crist - Ash	LF	Arsenic	0.02	1	0	

(continued)

CCW Constituent Data (continued)

Site/Waste Type	WMU Type	Chemical	Leachate (mg/L)	No. of Leachate Measure-ments	No. of Leachate Non-detects	Total (mg/kg)
Crist - Ash	LF	Barium	0.1	1	0	
Crist - Ash	LF	Cadmium	0.02	1	0	
Crist - Ash	LF	Lead	0.003	1	0	
Crist - Ash	LF	Selenium	0.05	1	0	
Crown III - Ash	LF	Antimony	0.071159259	54	10	
Crown III - Ash	LF	Arsenic	0.352503226	62	29	
Crown III - Ash	LF	Barium	0.279112903	62	3	
Crown III - Ash	LF	Boron	22.93277419	62	0	
Crown III - Ash	LF	Cadmium	0.128258065	62	3	
Crown III - Ash	LF	Cobalt	0.101225806	62	17	
Crown III - Ash	LF	Lead	0.605616935	62	19	
Crown III - Ash	LF	Mercury	0.000104839	62	61	
Crown III - Ash	LF	Molybdenum	0.588888889	9	4	
Crown III - Ash	LF	Selenium	0.03946129	62	46	
Crown III - Ash	LF	Thallium	0.0645	54	18	
Crown III - FBC	LF	Antimony	0.0135	17	9	
Crown III - FBC	LF	Arsenic	0.034822581	31	26	3.766666667
Crown III - FBC	LF	Barium	0.346774194	31	2	150
Crown III - FBC	LF	Boron	2.815296296	27	1	
Crown III - FBC	LF	Cadmium	0.011241935	31	22	2.17
Crown III - FBC	LF	Cobalt	0.02475	24	16	
Crown III - FBC	LF	Lead	0.068645161	31	17	8.233333333
Crown III - FBC	LF	Mercury	0.000164516	31	27	0.381
Crown III - FBC	LF	Molybdenum	0.1522	10	2	
Crown III - FBC	LF	Selenium	0.061467742	31	27	3.3
Crown III - FBC	LF	Thallium	0.004941176	17	11	
CTL-V - Ash	LF	Antimony	0.26	1	0	
CTL-V - Ash	LF	Arsenic	0.037	1	0	
CTL-V - Ash	LF	Barium	0.247	1	0	
CTL-V - Ash	LF	Cadmium	0.04	1	0	
CTL-V - Ash	LF	Lead	0.072	1	0	
CTL-V - Ash	LF	Mercury	0.001	1	0	
CTL-V - Ash	LF	Selenium	0.014	1	0	
CTL-V - Ash	LF	Thallium	0.01	1	0	
CY - Ash	LF	Aluminum	4.735	2	0	
CY - Ash	LF	Antimony	0.0078	2	0	
CY - Ash	LF	Arsenic	0.04825	2	0	
CY - Ash	LF	Barium	1.2395	2	0	
CY - Ash	LF	Boron	6.13	2	0	
CY - Ash	LF	Cadmium	0.0002075	2	1	
CY - Ash	LF	Cobalt	0.001915	4	4	

(continued)

CCW Constituent Data (continued)

Site/Waste Type	WMU Type	Chemical	Leachate (mg/L)	No. of Leachate Measure-ments	No. of Leachate Non-detects	Total (mg/kg)
CY - Ash	LF	Lead	0.003555	2	1	
CY - Ash	LF	Mercury	0.000265	2	0	
CY - Ash	LF	Selenium	0.004825	2	1	
CY - Ash	LF	Thallium	0.00196	4	4	
Dairyland Power Coop - Ash	LF	Arsenic	0.0328625	8	0	
Dairyland Power Coop - Ash	LF	Barium	0.058740741	27	0	
Dairyland Power Coop - Ash	LF	Boron	68.03979592	49	0	
Dairyland Power Coop - Ash	LF	Cadmium	0.00539	34	0	
Dairyland Power Coop - Ash	LF	Lead	0.0046	7	2	
Dairyland Power Coop - Ash	LF	Mercury	0.000223	2	1	
Dairyland Power Coop - Ash	LF	Selenium	0.0696375	8	0	
Daniel - Ash	LF	Arsenic	0.2	1	0	
Daniel - Ash	LF	Barium	0.4	1	0	
Daniel - Ash	LF	Cadmium	0.001	1	1	
Daniel - Ash	LF	Lead	0.001	1	1	
Daniel - Ash	LF	Selenium	0.001	1	1	
Deer Ridge Mine - Ash	LF	Aluminum	0.5941	10	1	64681.487
Deer Ridge Mine - Ash	LF	Arsenic	0.0029	10	6	21.29419
Deer Ridge Mine - Ash	LF	Barium	0.1448	10	2	258.468
Deer Ridge Mine - Ash	LF	Boron	1.228	10	2	179.354
Deer Ridge Mine - Ash	LF	Cadmium	0.01365	10	1	0.94425
Deer Ridge Mine - Ash	LF	Lead	0.0253	10	2	58.48
Deer Ridge Mine - Ash	LF	Mercury	0.00011025	10	10	0.1158
Deer Ridge Mine - Ash	LF	Molybdenum	0.0756	10	4	6.6287
Deer Ridge Mine - Ash	LF	Nitrate/Nitrite	0.095	3	2	
Deer Ridge Mine - Ash	LF	Selenium	0.01022	10	2	13.1061
DPC - Ash	LF	Antimony	0.04	2	1	0.475
DPC - Ash	LF	Arsenic	0.051	2	0	55.085
DPC - Ash	LF	Barium	0.28	2	0	37.7
DPC - Ash	LF	Boron	27.945	2	0	404.05
DPC - Ash	LF	Cadmium	0.005	4	4	0.56
DPC - Ash	LF	Lead	0.025	4	4	28.7
DPC - Ash	LF	Mercury	0.001	2	2	0.127
DPC - Ash	LF	Nitrate/Nitrite	2.5	2	0	0.2425
DPC - Ash	LF	Selenium	0.046	2	0	3.4445
EERC - Ash	LF	Mercury	0.000025	4	4	
Elkhart Mine - Ash	LF	Antimony	0.025192308	52	46	
Elkhart Mine - Ash	LF	Arsenic	0.043571429	77	71	
Elkhart Mine - Ash	LF	Barium	0.495324675	77	23	
Elkhart Mine - Ash	LF	Boron	6.88961039	77	0	
Elkhart Mine - Ash	LF	Cadmium	0.022551948	77	41	

(continued)

CCW Constituent Data (continued)

Site/Waste Type	WMU Type	Chemical	Leachate (mg/L)	No. of Leachate Measure-ments	No. of Leachate Non-detects	Total (mg/kg)
Elkhart Mine - Ash	LF	Cobalt	0.012785714	77	57	
Elkhart Mine - Ash	LF	Lead	0.027987013	77	66	
Elkhart Mine - Ash	LF	Mercury	0.000148052	77	68	
Elkhart Mine - Ash	LF	Selenium	0.036649351	77	64	
Elkhart Mine - Ash	LF	Thallium	0.015942308	52	48	
Elkhart Mine - FBC	LF	Antimony	0.021875	16	15	
Elkhart Mine - FBC	LF	Arsenic	0.034512195	41	37	
Elkhart Mine - FBC	LF	Barium	0.525365854	41	5	
Elkhart Mine - FBC	LF	Boron	13.13829268	41	0	
Elkhart Mine - FBC	LF	Cadmium	0.003536585	41	41	
Elkhart Mine - FBC	LF	Cobalt	0.007219512	41	39	
Elkhart Mine - FBC	LF	Lead	0.017195122	41	34	
Elkhart Mine - FBC	LF	Mercury	0.000104878	41	40	
Elkhart Mine - FBC	LF	Selenium	0.035365854	41	33	
Elkhart Mine - FBC	LF	Thallium	0.02390625	16	15	
FBX - Ash	LF	Arsenic	0.0025	2	2	
FBX - Ash	LF	Barium	29.6225	2	1	
FBX - Ash	LF	Cadmium	0.2	2	2	
FBX - Ash	LF	Lead	0.5	2	2	
FBX - Ash	LF	Mercury	0.00025	2	2	
FBX - Ash	LF	Selenium	0.01375	2	2	
FC - Ash and Coal Refuse	LF	Aluminum	13.8	2	0	
FC - Ash and Coal Refuse	LF	Antimony	0.00105	4	4	
FC - Ash and Coal Refuse	LF	Arsenic	0.005	2	0	
FC - Ash and Coal Refuse	LF	Barium	0.602	2	0	
FC - Ash and Coal Refuse	LF	Boron	2.54	2	0	
FC - Ash and Coal Refuse	LF	Cadmium	0.00015	4	4	
FC - Ash and Coal Refuse	LF	Cobalt	0.0029	2	0	
FC - Ash and Coal Refuse	LF	Lead	0.00345	2	0	
FC - Ash and Coal Refuse	LF	Mercury	0.00005	4	4	
FC - Ash and Coal Refuse	LF	Selenium	0.01765	2	0	
FC - Ash and Coal Refuse	LF	Thallium	0.00185	4	4	
Florence Mine - Ash	LF	Aluminum	0.03	1	0	
Florence Mine - Ash	LF	Antimony	0.005	1	1	
Florence Mine - Ash	LF	Arsenic	0.07	1	0	
Florence Mine - Ash	LF	Barium	2.23	1	0	
Florence Mine - Ash	LF	Boron	0.01	1	1	
Florence Mine - Ash	LF	Cadmium	0.01	1	1	
Florence Mine - Ash	LF	Lead	0.001	1	0	
Florence Mine - Ash	LF	Mercury	0.002	1	0	
Florence Mine - Ash	LF	Molybdenum	0.01	1	1	

(continued)

CCW Constituent Data (continued)

Site/Waste Type	WMU Type	Chemical	Leachate (mg/L)	No. of Leachate Measure-ments	No. of Leachate Non-detects	Total (mg/kg)
Florence Mine - Ash	LF	Nitrate/Nitrite	1.2	1	0	
Florence Mine - Ash	LF	Selenium	0.06	1	0	
Fran Site - FBC	LF	Aluminum	0.32	1	0	
Fran Site - FBC	LF	Antimony	0.005	1	1	
Fran Site - FBC	LF	Arsenic	0.02	1	0	
Fran Site - FBC	LF	Barium	0.08	1	0	
Fran Site - FBC	LF	Boron	0.43	1	0	
Fran Site - FBC	LF	Cadmium	0.005	1	1	
Fran Site - FBC	LF	Lead	0.005	1	1	
Fran Site - FBC	LF	Nitrate/Nitrite	1.22	1	0	
Fran Site - FBC	LF	Selenium	0.03	1	0	
FW - FBC	LF	Arsenic	0.02525	4	3	
FW - FBC	LF	Barium	0.304	4	0	
FW - FBC	LF	Cadmium	0.005	4	4	
FW - FBC	LF	Lead	0.05	4	4	
FW - FBC	LF	Mercury	0.001	4	4	
FW - FBC	LF	Selenium	0.1	4	4	
Gadsden - Ash	LF	Arsenic	0.2	1	0	
Gadsden - Ash	LF	Barium	0.3	1	0	
Gadsden - Ash	LF	Cadmium	0.01	1	0	
Gadsden - Ash	LF	Lead	0.04	1	0	
Gadsden - Ash	LF	Selenium	0.03	1	0	
Gale - Ash	LF	Aluminum	3.1	1	0	13630
Gale - Ash	LF	Antimony	0.03	1	0	3
Gale - Ash	LF	Arsenic	0.42	1	0	51.5
Gale - Ash	LF	Barium	1.7	1	0	143
Gale - Ash	LF	Boron	0.22	1	0	25
Gale - Ash	LF	Cadmium	0.01	1	0	1
Gale - Ash	LF	Lead	0.23	1	0	21
Gale - Ash	LF	Molybdenum	0.05	1	0	5
Gale - Ash	LF	Selenium	0.1	1	0	4.4
Gaston - Ash	LF	Arsenic	1.8	1	0	
Gaston - Ash	LF	Barium	0.3	1	0	
Gaston - Ash	LF	Cadmium	0.01	1	0	
Gaston - Ash	LF	Lead	0.05	1	0	
Gaston - Ash	LF	Selenium	0.003	1	0	
Gorgas - Ash	LF	Arsenic	1.6	1	0	
Gorgas - Ash	LF	Barium	0.3	1	0	
Gorgas - Ash	LF	Cadmium	0.01	1	0	
Gorgas - Ash	LF	Lead	0.04	1	0	
Gorgas - Ash	LF	Selenium	0.002	1	0	

(continued)

CCW Constituent Data (continued)

Site/Waste Type	WMU Type	Chemical	Leachate (mg/L)	No. of Leachate Measure-ments	No. of Leachate Non-detects	Total (mg/kg)
Greene Co - Ash	LF	Arsenic	1.1	1	0	
Greene Co - Ash	LF	Barium	0.4	1	0	
Greene Co - Ash	LF	Cadmium	0.01	1	0	
Greene Co - Ash	LF	Lead	0.04	1	0	
Greene Co - Ash	LF	Selenium	0.003	1	0	
HA - Ash and Coal Refuse	LF	Aluminum	1.71925	4	0	5666.666667
HA - Ash and Coal Refuse	LF	Antimony	0.003905	4	2	
HA - Ash and Coal Refuse	LF	Arsenic	0.024975	4	0	9.666666667
HA - Ash and Coal Refuse	LF	Barium	1.01675	4	0	186.6666667
HA - Ash and Coal Refuse	LF	Boron	0.64545	4	0	14
HA - Ash and Coal Refuse	LF	Cadmium	0.0039275	4	0	0.25
HA - Ash and Coal Refuse	LF	Cobalt	0.01517875	4	1	
HA - Ash and Coal Refuse	LF	Lead	0.00378	4	2	8.7
HA - Ash and Coal Refuse	LF	Mercury	0.0001	4	0	0.065
HA - Ash and Coal Refuse	LF	Selenium	0.005025	4	0	0.534166667
HA - Ash and Coal Refuse	LF	Thallium	0.00196	8	8	
Hammond - Ash	LF	Arsenic	0.1	1	0	
Hammond - Ash	LF	Barium	0.3	1	0	
Hammond - Ash	LF	Cadmium	0.01	1	0	
Hammond - Ash	LF	Lead	0.05	1	0	
Hammond - Ash	LF	Selenium	0.02	1	0	
Harrim 3019 - Ash	LF	Aluminum	5.21	1	0	46577
Harrim 3019 - Ash	LF	Antimony	0.0058	1	0	646.4
Harrim 3019 - Ash	LF	Arsenic	0.178	1	0	50.43172727
Harrim 3019 - Ash	LF	Barium	0.32	1	0	319.89
Harrim 3019 - Ash	LF	Molybdenum	0.594	1	0	17.9
Harrim 3019 - Ash	LF	Nitrate/Nitrite	1.99	1	0	
Harrim 3019 - Ash	LF	Selenium	0.0468	1	0	1.405714286
Harrim 3019 - FBC	LF	Aluminum	0.67375	8	0	
Harrim 3019 - FBC	LF	Antimony	0.002	1	0	
Harrim 3019 - FBC	LF	Barium	0.465888889	9	0	
Harrim 3019 - FBC	LF	Boron	0.07	1	0	
Harrim 3019 - FBC	LF	Cobalt	0.1385	6	0	
Harrim 3019 - FBC	LF	Lead	0.24	5	0	
Harrim 3019 - FBC	LF	Molybdenum	0.347714286	7	0	
Harrim 3019 - FBC	LF	Nitrate/Nitrite	0.199333333	3	0	
Harrim 3019 - FBC	LF	Selenium	0.019	2	0	
Industry Mine - Ash	LF	Antimony	0.031597143	70	12	
Industry Mine - Ash	LF	Arsenic	0.050248454	97	51	
Industry Mine - Ash	LF	Barium	0.328329897	97	13	
Industry Mine - Ash	LF	Boron	4.719969072	97	0	

(continued)

CCW Constituent Data (continued)

Site/Waste Type	WMU Type	Chemical	Leachate (mg/L)	No. of Leachate Measure-ments	No. of Leachate Non-detects	Total (mg/kg)
Industry Mine - Ash	LF	Cadmium	0.059061856	97	7	
Industry Mine - Ash	LF	Cobalt	0.120010309	97	30	
Industry Mine - Ash	LF	Lead	3.610544845	97	16	
Industry Mine - Ash	LF	Mercury	0.000284536	97	92	
Industry Mine - Ash	LF	Selenium	0.052408247	97	64	
Industry Mine - Ash	LF	Thallium	0.016984286	70	12	
Industry Mine - FBC	LF	Antimony	0.017077778	9	4	
Industry Mine - FBC	LF	Arsenic	0.031111111	9	7	
Industry Mine - FBC	LF	Barium	9.515666667	9	0	
Industry Mine - FBC	LF	Boron	2.813888889	9	2	
Industry Mine - FBC	LF	Cadmium	0.015888889	9	7	
Industry Mine - FBC	LF	Cobalt	0.029333333	9	8	
Industry Mine - FBC	LF	Lead	0.051877778	9	6	
Industry Mine - FBC	LF	Mercury	0.000222222	9	8	
Industry Mine - FBC	LF	Selenium	0.080388889	9	4	
Industry Mine - FBC	LF	Thallium	0.002288889	9	6	
Key West - Ash	LF	Arsenic	0.005	1	1	
Key West - Ash	LF	Barium	1	2	0	
Key West - Ash	LF	Boron	0.2	1	0	
Key West - Ash	LF	Cadmium	0.07	1	0	
Key West - Ash	LF	Lead	0.4	1	0	
Key West - Ash	LF	Mercury	0.18	1	0	
Key West - Ash	LF	Selenium	0.005	1	1	
Keystone - Ash	LF	Aluminum	2.059	4	0	
Keystone - Ash	LF	Antimony	0.036	1	0	
Keystone - Ash	LF	Arsenic	0.30925	4	0	
Keystone - Ash	LF	Barium	0.40375	4	0	
Keystone - Ash	LF	Boron	0.72	1	0	
Keystone - Ash	LF	Cadmium	0.009625	4	1	
Keystone - Ash	LF	Cobalt	0.023	1	0	
Keystone - Ash	LF	Lead	0.045375	4	1	
Keystone - Ash	LF	Mercury	0.001	1	1	
Keystone - Ash	LF	Molybdenum	0.32	1	0	
Keystone - Ash	LF	Selenium	0.0525	4	2	
Keystone - Ash	LF	Thallium	0.083	1	0	
Keystone - Ash and Coal Refuse	LF	Aluminum	0.842	4	0	
Keystone - Ash and Coal Refuse	LF	Antimony	0.0015	2	2	
Keystone - Ash and Coal Refuse	LF	Arsenic	0.01875	4	4	

(continued)

CCW Constituent Data (continued)

Site/Waste Type	WMU Type	Chemical	Leachate (mg/L)	No. of Leachate Measure-ments	No. of Leachate Non-detects	Total (mg/kg)
Keystone - Ash and Coal Refuse	LF	Barium	0.1925	4	0	
Keystone - Ash and Coal Refuse	LF	Boron	0.06	1	0	
Keystone - Ash and Coal Refuse	LF	Cadmium	0.00225	4	4	
Keystone - Ash and Coal Refuse	LF	Cobalt	0.022	1	0	
Keystone - Ash and Coal Refuse	LF	Lead	0.01875	4	4	
Keystone - Ash and Coal Refuse	LF	Mercury	0.001	1	1	
Keystone - Ash and Coal Refuse	LF	Molybdenum	0.01	2	2	
Keystone - Ash and Coal Refuse	LF	Selenium	0.02	4	4	
Keystone - Ash and Coal Refuse	LF	Thallium	0.028	1	0	
Kraft - Ash	LF	Arsenic	0.02	1	0	
Kraft - Ash	LF	Barium	0.3	1	0	
Kraft - Ash	LF	Cadmium	0.01	1	0	
Kraft - Ash	LF	Lead	0.04	1	0	
Kraft - Ash	LF	Selenium	0.04	1	0	
LIMB Site - Ash	LF	Aluminum	0.102894737	38	37	
LIMB Site - Ash	LF	Antimony	0.29	5	1	25
LIMB Site - Ash	LF	Arsenic	0.033594737	38	6	63
LIMB Site - Ash	LF	Barium	0.036552632	38	0	255
LIMB Site - Ash	LF	Boron	0.521842105	38	31	400
LIMB Site - Ash	LF	Cadmium	0.001031579	38	33	0.31
LIMB Site - Ash	LF	Cobalt	0.005131579	38	37	
LIMB Site - Ash	LF	Lead	0.012789474	38	25	14.5
LIMB Site - Ash	LF	Mercury	0.0001	2	2	
LIMB Site - Ash	LF	Molybdenum	1.527342105	38	1	2.5
LIMB Site - Ash	LF	Nitrate/Nitrite	26	2	0	
LIMB Site - Ash	LF	Selenium	0.0199	38	24	0.25
LIMB Site - Ash	LF	Thallium	0.05	5	5	
Little Sandy #10 Mine - Ash	LF	Aluminum	1.078	6	2	4541.666667
Little Sandy #10 Mine - Ash	LF	Arsenic	0.032336364	11	8	38.293
Little Sandy #10 Mine - Ash	LF	Barium	0.264454545	11	6	48.81
Little Sandy #10 Mine - Ash	LF	Boron	2.630909091	11	3	157.76
Little Sandy #10 Mine - Ash	LF	Cadmium	0.008290909	11	9	1.198
Little Sandy #10 Mine - Ash	LF	Lead	0.022009091	11	10	56.84
Little Sandy #10 Mine - Ash	LF	Mercury	0.000486364	11	10	0.24435

(continued)

CCW Constituent Data (continued)

Site/Waste Type	WMU Type	Chemical	Leachate (mg/L)	No. of Leachate Measure-ments	No. of Leachate Non-detects	Total (mg/kg)
Little Sandy #10 Mine - Ash	LF	Molybdenum	0.177272727	11	5	6.354
Little Sandy #10 Mine - Ash	LF	Selenium	0.059527273	11	9	6.531
Lone Mtn - Ash	LF	Aluminum	28.615	2	0	
Lone Mtn - Ash	LF	Antimony	0.033	2	0	
Lone Mtn - Ash	LF	Arsenic	0.185	2	0	76
Lone Mtn - Ash	LF	Barium	0.167	2	0	1483.2
Lone Mtn - Ash	LF	Cadmium	0.572	2	0	11.86
Lone Mtn - Ash	LF	Cobalt	0.142	2	0	87.3
Lone Mtn - Ash	LF	Mercury	0.0019	1	0	
Lone Mtn - Ash	LF	Molybdenum	0.4295	2	0	
Lone Mtn - Ash	LF	Selenium	0.328	2	0	
LS - Ash and Coal Refuse	LF	Aluminum	1.18	7	0	
LS - Ash and Coal Refuse	LF	Antimony	0.0107	4	0	
LS - Ash and Coal Refuse	LF	Arsenic	0.0104525	16	3	
LS - Ash and Coal Refuse	LF	Barium	0.13220625	16	0	
LS - Ash and Coal Refuse	LF	Boron	18.93125	16	0	
LS - Ash and Coal Refuse	LF	Cadmium	0.00148	16	15	
LS - Ash and Coal Refuse	LF	Cobalt	0.011125	4	0	
LS - Ash and Coal Refuse	LF	Lead	0.0025	16	16	
LS - Ash and Coal Refuse	LF	Mercury	0.00007	4	3	
LS - Ash and Coal Refuse	LF	Molybdenum	0.886875	16	0	
LS - Ash and Coal Refuse	LF	Nitrate/Nitrite	3.045	32	16	
LS - Ash and Coal Refuse	LF	Selenium	1.05343125	16	0	
LS - Ash and Coal Refuse	LF	Thallium	0.00185	8	8	
Martins Creek - Ash	LF	Aluminum	3.18335	20	2	114229.3889
Martins Creek - Ash	LF	Antimony	0.005021053	19	11	10.315
Martins Creek - Ash	LF	Arsenic	0.2314	20	1	50.50530556
Martins Creek - Ash	LF	Barium	0.1969	20	2	641.5466667
Martins Creek - Ash	LF	Boron	3.5089	20	1	304.1266667
Martins Creek - Ash	LF	Cadmium	0.0032	20	20	2.025
Martins Creek - Ash	LF	Cobalt	0.024722222	18	18	66.37611111
Martins Creek - Ash	LF	Lead	0.014	20	19	
Martins Creek - Ash	LF	Mercury	0.0001	19	19	
Martins Creek - Ash	LF	Molybdenum	0.195157895	19	10	
Martins Creek - Ash	LF	Nitrate/Nitrite	0.636428571	14	9	
Martins Creek - Ash	LF	Selenium	0.05717	20	8	4.043888889
Martins Creek - Ash	LF	Thallium	0.003263158	19	19	
McCloskey Site - FBC	LF	Aluminum	0.5	2	2	27450
McCloskey Site - FBC	LF	Arsenic	0.001	2	2	45.355
McCloskey Site - FBC	LF	Barium	0.1	2	2	32.55
McCloskey Site - FBC	LF	Boron	0.022	2	1	0.092

(continued)

CCW Constituent Data (continued)

Site/Waste Type	WMU Type	Chemical	Leachate (mg/L)	No. of Leachate Measure-ments	No. of Leachate Non-detects	Total (mg/kg)
McCloskey Site - FBC	LF	Cadmium	0.0375	2	1	0.025
McCloskey Site - FBC	LF	Lead	0.05	2	2	50
McCloskey Site - FBC	LF	Mercury	0.25	2	2	0.4465
McCloskey Site - FBC	LF	Molybdenum	0.15	2	2	0.15
McCloskey Site - FBC	LF	Selenium	0.0515675	2	2	52.315
McDonough - Ash	LF	Arsenic	0.9	1	0	
McDonough - Ash	LF	Barium	0.5	1	0	
McDonough - Ash	LF	Cadmium	0.01	1	0	
McDonough - Ash	LF	Lead	0.04	1	0	
McDonough - Ash	LF	Selenium	0.2	1	0	
McIntosh - Ash	LF	Arsenic	0.09	1	0	
McIntosh - Ash	LF	Barium	0.2	1	0	
McIntosh - Ash	LF	Cadmium	0.6	1	0	
McIntosh - Ash	LF	Lead	0.03	1	0	
McIntosh - Ash	LF	Selenium	0.03	1	0	
McKay Site - FBC	LF	Aluminum	0.105	2	0	30000
McKay Site - FBC	LF	Antimony	0.01	2	2	2.5
McKay Site - FBC	LF	Arsenic	0.025	2	2	51.5
McKay Site - FBC	LF	Barium	0.27	2	0	215
McKay Site - FBC	LF	Boron	0.265	2	0	41.5
McKay Site - FBC	LF	Cadmium	0.005	2	2	2.5
McKay Site - FBC	LF	Lead	0.03	2	1	49
McKay Site - FBC	LF	Mercury	0.0001	2	2	0.345
McKay Site - FBC	LF	Molybdenum	0.13	2	0	6.25
McKay Site - FBC	LF	Nitrate/Nitrite	0.0175	2	1	
McKay Site - FBC	LF	Selenium	0.0355	2	1	1
Miller - Ash	LF	Arsenic	1.3	1	0	18
Miller - Ash	LF	Barium	0.1	1	0	7140
Miller - Ash	LF	Cadmium	0.09	1	0	1.6
Miller - Ash	LF	Lead	0.002	1	0	38
Miller - Ash	LF	Selenium	0.03	1	0	
Miller Creek Mine - Ash	LF	Aluminum	4.78597619	42	4	22486.5969
Miller Creek Mine - Ash	LF	Arsenic	0.075817021	47	16	60.54551064
Miller Creek Mine - Ash	LF	Barium	0.147255319	47	0	87.49382979
Miller Creek Mine - Ash	LF	Boron	2.343829787	47	3	167.0508511
Miller Creek Mine - Ash	LF	Cadmium	0.009771277	47	31	1.850959894
Miller Creek Mine - Ash	LF	Lead	0.034382979	47	24	51.50851064
Miller Creek Mine - Ash	LF	Mercury	0.000255319	47	46	0.06780663
Miller Creek Mine - Ash	LF	Molybdenum	0.166808511	47	17	9.819680851
Miller Creek Mine - Ash	LF	Selenium	0.047102128	47	23	6.492617021
Mine 26 - Ash	LF	Antimony	0.0125	6	6	

(continued)

CCW Constituent Data (continued)

Site/Waste Type	WMU Type	Chemical	Leachate (mg/L)	No. of Leachate Measure-ments	No. of Leachate Non-detects	Total (mg/kg)
Mine 26 - Ash	LF	Arsenic	0.022333333	9	8	
Mine 26 - Ash	LF	Barium	0.388111111	9	1	
Mine 26 - Ash	LF	Boron	9.266666667	9	0	
Mine 26 - Ash	LF	Cadmium	0.008555556	9	4	
Mine 26 - Ash	LF	Cobalt	0.021744444	9	5	
Mine 26 - Ash	LF	Lead	0.148111111	9	6	
Mine 26 - Ash	LF	Mercury	0.0003	9	9	
Mine 26 - Ash	LF	Selenium	0.026388889	9	6	
Mine 26 - Ash	LF	Thallium	0.006833333	6	5	
Mine 26 - Ash and Coal Refuse	LF	Antimony	0.01	2	2	
Mine 26 - Ash and Coal Refuse	LF	Arsenic	0.054285714	7	5	
Mine 26 - Ash and Coal Refuse	LF	Barium	0.615714286	7	0	
Mine 26 - Ash and Coal Refuse	LF	Boron	3.504285714	7	0	
Mine 26 - Ash and Coal Refuse	LF	Cadmium	0.010142857	7	4	
Mine 26 - Ash and Coal Refuse	LF	Cobalt	0.032857143	7	2	
Mine 26 - Ash and Coal Refuse	LF	Lead	0.047142857	7	4	
Mine 26 - Ash and Coal Refuse	LF	Mercury	0.0001	7	7	
Mine 26 - Ash and Coal Refuse	LF	Selenium	0.02	7	7	
Mine 26 - Ash and Coal Refuse	LF	Thallium	0.005	2	2	
Mine 26 - FBC	LF	Arsenic	0.03	1	1	
Mine 26 - FBC	LF	Barium	0.51	1	0	
Mine 26 - FBC	LF	Boron	1.3	1	0	
Mine 26 - FBC	LF	Cadmium	0.0025	1	1	
Mine 26 - FBC	LF	Cobalt	0.005	1	1	
Mine 26 - FBC	LF	Lead	0.01	1	1	
Mine 26 - FBC	LF	Mercury	0.0001	1	1	
Mine 26 - FBC	LF	Selenium	0.08	1	0	
Mitchell - Ash	LF	Arsenic	1.3	1	0	
Mitchell - Ash	LF	Barium	0.3	1	0	
Mitchell - Ash	LF	Cadmium	0.01	1	0	
Mitchell - Ash	LF	Lead	0.06	1	0	
Mitchell - Ash	LF	Selenium	0.06	1	0	
MO - Ash and Coal Refuse	LF	Aluminum	4.49	2	0	

(continued)

CCW Constituent Data (continued)

Site/Waste Type	WMU Type	Chemical	Leachate (mg/L)	No. of Leachate Measure-ments	No. of Leachate Non-detects	Total (mg/kg)
MO - Ash and Coal Refuse	LF	Antimony	0.0125	2	0	
MO - Ash and Coal Refuse	LF	Arsenic	0.2855	2	0	
MO - Ash and Coal Refuse	LF	Barium	1.845	2	0	
MO - Ash and Coal Refuse	LF	Boron	0.219	2	0	
MO - Ash and Coal Refuse	LF	Cadmium	0.006	2	0	
MO - Ash and Coal Refuse	LF	Cobalt	0.012	2	0	
MO - Ash and Coal Refuse	LF	Lead	0.0065	2	0	
MO - Ash and Coal Refuse	LF	Mercury	0.00005	4	4	
MO - Ash and Coal Refuse	LF	Selenium	0.1312	2	0	
MO - Ash and Coal Refuse	LF	Thallium	0.01415	2	0	
Murdock Mine - Ash	LF	Antimony	0.0076875	8	8	
Murdock Mine - Ash	LF	Arsenic	0.0080875	8	6	
Murdock Mine - Ash	LF	Barium	0.258625	8	0	
Murdock Mine - Ash	LF	Boron	9.38775	8	0	
Murdock Mine - Ash	LF	Cadmium	0.0458	8	2	
Murdock Mine - Ash	LF	Cobalt	0.0225625	8	2	
Murdock Mine - Ash	LF	Lead	0.00555	8	2	
Murdock Mine - Ash	LF	Mercury	0.0004375	8	8	
Murdock Mine - Ash	LF	Selenium	0.0053875	8	4	
Murdock Mine - Ash	LF	Thallium	0.02325	8	2	
Murdock Mine - FBC	LF	Antimony	0.004	3	3	
Murdock Mine - FBC	LF	Arsenic	0.005	3	3	
Murdock Mine - FBC	LF	Barium	0.368333333	3	0	
Murdock Mine - FBC	LF	Boron	0.436666667	3	0	
Murdock Mine - FBC	LF	Cadmium	0.0015	3	3	
Murdock Mine - FBC	LF	Cobalt	0.0025	3	3	
Murdock Mine - FBC	LF	Lead	0.0015	3	3	
Murdock Mine - FBC	LF	Mercury	0.0004	3	3	
Murdock Mine - FBC	LF	Selenium	0.003533333	3	2	
Murdock Mine - FBC	LF	Thallium	0.005	3	3	
Nepco - FBC	LF	Arsenic	0.025	2	2	21
Nepco - FBC	LF	Cadmium	0.01	1	0	0.5
Nepco - FBC	LF	Lead	0.025	2	2	39
Nepco - FBC	LF	Mercury	0.0002	2	2	0.01
Nepco - FBC	LF	Selenium	0.05	2	2	12.6
No. 1 Contracting Corp - FBC	LF	Aluminum	0.935	2	0	
No. 1 Contracting Corp - FBC	LF	Antimony	0.018	1	0	
No. 1 Contracting Corp - FBC	LF	Arsenic	0.046	2	0	

(continued)

CCW Constituent Data (continued)

Site/Waste Type	WMU Type	Chemical	Leachate (mg/L)	No. of Leachate Measure-ments	No. of Leachate Non-detects	Total (mg/kg)
No. 1 Contracting Corp - FBC	LF	Barium	0.1315	2	0	
No. 1 Contracting Corp - FBC	LF	Boron	0.05	1	0	
No. 1 Contracting Corp - FBC	LF	Cadmium	0.005	1	0	
No. 1 Contracting Corp - FBC	LF	Lead	0.06	1	0	
No. 1 Contracting Corp - FBC	LF	Mercury	0.0002	1	0	
No. 1 Contracting Corp - FBC	LF	Molybdenum	0.105	2	0	
No. 1 Contracting Corp - FBC	LF	Selenium	0.1395	2	0	
Northampton40000201 - Ash	LF	Aluminum	0.38	1	0	24500
Northampton40000201 - Ash	LF	Antimony	0.01	1	0	20
Northampton40000201 - Ash	LF	Arsenic	0.005	1	0	40.6
Northampton40000201 - Ash	LF	Barium	0.21	1	0	242
Northampton40000201 - Ash	LF	Boron	0.2	1	0	17.3
Northampton40000201 - Ash	LF	Cadmium	0.012	1	0	0.5
Northampton40000201 - Ash	LF	Lead	0.1	1	0	18
Northampton40000201 - Ash	LF	Mercury	0.0002	1	0	0.535
Northampton40000201 - Ash	LF	Molybdenum	0.1	1	0	10
Northampton40000201 - Ash	LF	Selenium	0.015	1	0	8.9
Nucla - FBC	LF	Aluminum	0.1	2	2	110050
Nucla - FBC	LF	Arsenic	0.0025	4	4	7.4
Nucla - FBC	LF	Barium	0.08	2	1	190
Nucla - FBC	LF	Boron	0.485	2	1	57.5
Nucla - FBC	LF	Cadmium	0.00055	2	2	1.95
Nucla - FBC	LF	Cobalt	0.005	2	2	10
Nucla - FBC	LF	Lead	0.0016	2	1	35.5
Nucla - FBC	LF	Mercury	0.0001	2	2	
Nucla - FBC	LF	Molybdenum	0.2045	2	0	83
Nucla - FBC	LF	Nitrate/Nitrite	0.1125	2	2	
Nucla - FBC	LF	Selenium	0.00485	2	1	9.35

(continued)

CCW Constituent Data (continued)

Site/Waste Type	WMU Type	Chemical	Leachate (mg/L)	No. of Leachate Measure-ments	No. of Leachate Non-detects	Total (mg/kg)
Nucla2 - FBC	LF	Aluminum	7.18	3	0	100000
Nucla2 - FBC	LF	Antimony	0.1	6	6	46
Nucla2 - FBC	LF	Arsenic	0.00375	6	5	27.93333333
Nucla2 - FBC	LF	Barium	0.093	3	0	246
Nucla2 - FBC	LF	Boron	3.1	3	1	69.16666667
Nucla2 - FBC	LF	Cadmium	0.000475	6	4	0.263333333
Nucla2 - FBC	LF	Cobalt	0.012	3	1	6.1
Nucla2 - FBC	LF	Lead	0.0062	3	0	8.296666667
Nucla2 - FBC	LF	Mercury	0.000566667	6	5	0.214166667
Nucla2 - FBC	LF	Molybdenum	0.303333333	3	0	3.316666667
Nucla2 - FBC	LF	Nitrate/Nitrite	6.591666667	6	4	
Nucla2 - FBC	LF	Selenium	0.048666667	6	2	1.395
Nucla2 - FBC	LF	Thallium	0.05	3	3	6.416666667
OK - Ash	LF	Aluminum	11.895	2	0	
OK - Ash	LF	Antimony	0.001575	2	1	
OK - Ash	LF	Arsenic	0.003225	2	1	
OK - Ash	LF	Barium	0.686	2	0	
OK - Ash	LF	Boron	2.68	2	0	
OK - Ash	LF	Cadmium	0.00027	2	1	
OK - Ash	LF	Cobalt	0.00745	2	0	
OK - Ash	LF	Lead	0.00355	2	0	
OK - Ash	LF	Mercury	0.0001	2	1	
OK - Ash	LF	Selenium	0.037	2	0	
OK - Ash	LF	Thallium	0.00185	4	4	
P4 - Ash	LF	Aluminum	6.2196875	8	0	
P4 - Ash	LF	Antimony	0.00105	4	4	
P4 - Ash	LF	Arsenic	0.00420375	8	5	
P4 - Ash	LF	Barium	0.254375	8	0	
P4 - Ash	LF	Boron	1.142697917	8	0	
P4 - Ash	LF	Cadmium	0.00125	8	8	
P4 - Ash	LF	Cobalt	0.00315	2	0	
P4 - Ash	LF	Lead	0.0025	8	8	
P4 - Ash	LF	Mercury	0.00005	4	4	
P4 - Ash	LF	Molybdenum	0.2114375	8	4	
P4 - Ash	LF	Nitrate/Nitrite	1.92075	16	8	
P4 - Ash	LF	Selenium	0.01	8	8	
P4 - Ash	LF	Thallium	0.002775	2	2	
PA - Ash	LF	Aluminum	26.16153846	13	0	
PA - Ash	LF	Antimony	0.0031	2	0	
PA - Ash	LF	Arsenic	0.005991923	13	9	
PA - Ash	LF	Barium	1.043838462	13	0	

(continued)

CCW Constituent Data (continued)

Site/Waste Type	WMU Type	Chemical	Leachate (mg/L)	No. of Leachate Measure-ments	No. of Leachate Non-detects	Total (mg/kg)
PA - Ash	LF	Boron	0.736153846	13	0	
PA - Ash	LF	Cadmium	0.001758462	13	12	
PA - Ash	LF	Cobalt	0.001915	2	2	
PA - Ash	LF	Lead	0.005993077	13	10	
PA - Ash	LF	Mercury	0.000175	2	0	
PA - Ash	LF	Molybdenum	0.138461538	13	4	
PA - Ash	LF	Nitrate/Nitrite	2.544596154	26	15	
PA - Ash	LF	Selenium	0.084376923	13	5	
PA - Ash	LF	Thallium	0.00196	4	4	
Pitt - FBC	LF	Antimony	0.0219	1	0	
Pitt - FBC	LF	Arsenic	0.05	1	1	
Pitt - FBC	LF	Barium	1.167333333	3	1	
Pitt - FBC	LF	Cadmium	0.033333333	3	3	
Pitt - FBC	LF	Lead	0.183333333	3	3	
Pitt - FBC	LF	Mercury	0.005	1	1	
Pitt - FBC	LF	Selenium	0.05	1	1	
Pitt - FBC	LF	Thallium	0.0025	3	3	
Plant 10 - FBC	LF	Arsenic	0.14875	4	0	71.3
Plant 10 - FBC	LF	Cadmium	0.05425	4	1	2.418181818
Plant 10 - FBC	LF	Lead	0.2965	4	1	39.63636364
Plant 10 - FBC	LF	Mercury	0.05005	4	4	1.174
Plant 10 - FBC	LF	Selenium	0.1285	4	0	4.011818182
Plant 12 - FBC	LF	Arsenic	0.004125	8	4	98.62222222
Plant 12 - FBC	LF	Cadmium	0.02	8	8	2.188888889
Plant 12 - FBC	LF	Lead	0.28375	8	2	47.83333333
Plant 12 - FBC	LF	Mercury	0.0004	8	8	1.047777778
Plant 12 - FBC	LF	Selenium	0.006125	8	8	4.263888889
Plant 8 - FBC	LF	Arsenic	0.019868421	19	18	42.04210526
Plant 8 - FBC	LF	Cadmium	0.016826923	52	43	2.288947368
Plant 8 - FBC	LF	Lead	0.007211538	52	37	27.62105263
Plant 8 - FBC	LF	Mercury	0.000289474	19	19	0.065789474
Plant 8 - FBC	LF	Selenium	0.053026316	19	9	33.02263158
Plant 9 - FBC	LF	Arsenic	0.058666667	3	0	2.8
Plant 9 - FBC	LF	Lead	0.105454545	11	8	57.67142857
Plant 9 - FBC	LF	Mercury	0.00025	11	11	0.604285714
Plant 9 - FBC	LF	Selenium	0.065333333	3	0	5.115714286
Portland - Ash	LF	Aluminum	2.648555556	9	0	
Portland - Ash	LF	Antimony	0.075	2	2	
Portland - Ash	LF	Arsenic	0.178666667	9	6	
Portland - Ash	LF	Barium	0.28475	8	0	
Portland - Ash	LF	Boron	4.799333333	3	0	

(continued)

CCW Constituent Data (continued)

Site/Waste Type	WMU Type	Chemical	Leachate (mg/L)	No. of Leachate Measure-ments	No. of Leachate Non-detects	Total (mg/kg)
Portland - Ash	LF	Cadmium	0.006	9	7	
Portland - Ash	LF	Cobalt	0.014	2	1	
Portland - Ash	LF	Lead	0.058333333	9	8	
Portland - Ash	LF	Mercury	0.001	4	4	
Portland - Ash	LF	Molybdenum	0.178666667	3	1	
Portland - Ash	LF	Selenium	0.25625	4	4	
Portland - Ash	LF	Thallium	0.005	4	4	
PP - Ash	LF	Aluminum	2.422	2	0	
PP - Ash	LF	Antimony	0.00245	2	0	
PP - Ash	LF	Arsenic	0.0273375	2	1	
PP - Ash	LF	Barium	0.2435	2	0	
PP - Ash	LF	Boron	6.605	2	0	
PP - Ash	LF	Cadmium	0.0023975	2	1	
PP - Ash	LF	Cobalt	0.0049575	2	1	
PP - Ash	LF	Lead	0.001155	2	1	
PP - Ash	LF	Mercury	0.00028	2	0	
PP - Ash	LF	Selenium	0.0364	2	0	
PP - Ash	LF	Thallium	0.01518	2	1	
Revloc Site - FBC	LF	Aluminum	0.58	2	1	
Revloc Site - FBC	LF	Antimony	0.002	2	2	
Revloc Site - FBC	LF	Arsenic	0.002	2	1	
Revloc Site - FBC	LF	Barium	0.44	2	2	
Revloc Site - FBC	LF	Boron	0.2585	2	1	
Revloc Site - FBC	LF	Cadmium	0.02	2	2	
Revloc Site - FBC	LF	Cobalt	0.0825	2	1	
Revloc Site - FBC	LF	Lead	0.25	2	0	
Revloc Site - FBC	LF	Mercury	0.0005	2	2	
Revloc Site - FBC	LF	Molybdenum	0.0545	2	1	
Revloc Site - FBC	LF	Selenium	0.0025	2	1	
Scherer - Ash	LF	Arsenic	0.01	1	0	
Scherer - Ash	LF	Barium	0.7	1	0	
Scherer - Ash	LF	Cadmium	0.001	1	0	
Scherer - Ash	LF	Lead	0.001	1	0	
Scherer - Ash	LF	Selenium	0.06	1	0	
Scholz - Ash	LF	Arsenic	0.02	1	0	
Scholz - Ash	LF	Barium	0.2	1	0	
Scholz - Ash	LF	Cadmium	0.04	1	0	
Scholz - Ash	LF	Lead	0.04	1	0	
Scholz - Ash	LF	Selenium	0.02	1	0	
Scrubgrass - FBC	LF	Arsenic	0.025	2	2	59
Scrubgrass - FBC	LF	Cadmium	0.0025	1	0	0.7

(continued)

CCW Constituent Data (continued)

Site/Waste Type	WMU Type	Chemical	Leachate (mg/L)	No. of Leachate Measure-ments	No. of Leachate Non-detects	Total (mg/kg)
Scrubgrass - FBC	LF	Lead	0.025	2	2	50
Scrubgrass - FBC	LF	Mercury	0.0002	2	2	0.01
Scrubgrass - FBC	LF	Selenium	0.05	2	2	21.7
Seward - Ash	LF	Aluminum	2.965	2	0	
Seward - Ash	LF	Antimony	0.075	2	2	
Seward - Ash	LF	Arsenic	0.288666667	3	2	
Seward - Ash	LF	Barium	0.473333333	3	0	
Seward - Ash	LF	Boron	0.57	1	0	
Seward - Ash	LF	Cadmium	0.005833333	3	1	
Seward - Ash	LF	Cobalt	0.014	1	0	
Seward - Ash	LF	Lead	0.1875	1	1	
Seward - Ash	LF	Mercury	0.003733333	3	3	
Seward - Ash	LF	Molybdenum	0.53	1	0	
Seward - Ash	LF	Selenium	0.196666667	3	2	
Seward - Ash	LF	Thallium	0.012	1	0	
Shawnee - FBC	LF	Aluminum	0.231	5	3	38240
Shawnee - FBC	LF	Antimony	0.296	5	2	15.6
Shawnee - FBC	LF	Arsenic	0.219	10	6	17.3
Shawnee - FBC	LF	Barium	2.001	10	0	799.4
Shawnee - FBC	LF	Boron	0.97	5	3	116.2
Shawnee - FBC	LF	Cadmium	0.005555	10	7	0.622
Shawnee - FBC	LF	Cobalt	0.07	5	2	2.75
Shawnee - FBC	LF	Lead	0.0897	10	5	6.4
Shawnee - FBC	LF	Mercury	0.00029	10	8	0.365
Shawnee - FBC	LF	Molybdenum	0.382	5	0	6.4
Shawnee - FBC	LF	Nitrate/Nitrite	3.786666667	8	4	
Shawnee - FBC	LF	Selenium	0.13005	10	6	0.73
Shawnee - FBC	LF	Thallium	0.197	5	3	8.9
Shawville - Ash	LF	Aluminum	2.0958	5	0	
Shawville - Ash	LF	Antimony	0.075	2	2	
Shawville - Ash	LF	Arsenic	0.4384	5	1	
Shawville - Ash	LF	Barium	0.2172	5	0	
Shawville - Ash	LF	Boron	0.56	1	0	
Shawville - Ash	LF	Cadmium	0.0059	5	2	
Shawville - Ash	LF	Cobalt	0.021	1	0	
Shawville - Ash	LF	Lead	0.1875	1	1	
Shawville - Ash	LF	Mercury	0.001	2	2	
Shawville - Ash	LF	Molybdenum	0.09	1	0	
Shawville - Ash	LF	Selenium	0.191	5	2	
Shawville - Ash	LF	Thallium	0.005	2	2	
Sibley Quarry - Ash	LF	Aluminum	0.6	4	4	

(continued)

CCW Constituent Data (continued)

Site/Waste Type	WMU Type	Chemical	Leachate (mg/L)	No. of Leachate Measure-ments	No. of Leachate Non-detects	Total (mg/kg)
Sibley Quarry - Ash	LF	Arsenic	0.018	4	0	
Sibley Quarry - Ash	LF	Barium	0.265	4	4	
Sibley Quarry - Ash	LF	Cadmium	0.00114125	4	2	
Sibley Quarry - Ash	LF	Lead	0.00305	4	4	
Sibley Quarry - Ash	LF	Mercury	0.0001	4	4	
Sibley Quarry - Ash	LF	Molybdenum	0.725	3	1	
Sibley Quarry - Ash	LF	Selenium	0.18425	4	1	
Silverton - Ash	LF	Aluminum	3.1	1	0	16870
Silverton - Ash	LF	Arsenic	0.375	2	0	48.5
Silverton - Ash	LF	Barium	1.7	1	0	181.5
Silverton - Ash	LF	Boron	0.22	1	0	20.5
Silverton - Ash	LF	Lead	0.23	1	0	29.5
Silverton - Ash	LF	Molybdenum	0.1	1	0	5
Silverton - Ash	LF	Selenium	0.12	2	0	6.7
Smith - Ash	LF	Arsenic	0.02	1	0	
Smith - Ash	LF	Barium	0.2	1	0	
Smith - Ash	LF	Cadmium	0.04	1	0	
Smith - Ash	LF	Lead	0.01	1	0	
Smith - Ash	LF	Selenium	0.01	1	0	
SW - Ash	LF	Arsenic	0.006679487	195	53	29.495189
SW - Ash	LF	Barium	0.81082716	243	0	2538.862069
SW - Ash	LF	Cadmium	0.003400769	195	47	1.230670103
SW - Ash	LF	Lead	0.001570707	99	97	35.39886598
SW - Ash	LF	Mercury	0.000217677	99	98	0.039255034
SW - Ash	LF	Selenium	0.003534884	172	46	0.6
SX - Ash	LF	Aluminum	1.862	2	0	
SX - Ash	LF	Antimony	0.003275	2	1	
SX - Ash	LF	Arsenic	0.0365	2	0	
SX - Ash	LF	Barium	0.959	2	0	
SX - Ash	LF	Boron	4.5223	2	0	
SX - Ash	LF	Cadmium	0.04425	2	0	
SX - Ash	LF	Cobalt	0.0167	2	0	
SX - Ash	LF	Lead	0.00675	2	0	
SX - Ash	LF	Mercury	0.00005	4	4	
SX - Ash	LF	Selenium	0.048725	2	1	
SX - Ash	LF	Thallium	0.013625	2	1	
Tidd - FBC	LF	Aluminum	0.105	3	1	
Tidd - FBC	LF	Antimony	0.03	5	5	
Tidd - FBC	LF	Arsenic	0.028333333	3	2	
Tidd - FBC	LF	Barium	0.184	2	0	
Tidd - FBC	LF	Boron	0.82	3	0	

(continued)

CCW Constituent Data (continued)

Site/Waste Type	WMU Type	Chemical	Leachate (mg/L)	No. of Leachate Measure-ments	No. of Leachate Non-detects	Total (mg/kg)
Tidd - FBC	LF	Cadmium	0.0015	3	3	
Tidd - FBC	LF	Cobalt	0.021	3	0	
Tidd - FBC	LF	Lead	0.015833333	3	3	
Tidd - FBC	LF	Mercury	0.006733333	3	3	
Tidd - FBC	LF	Molybdenum	0.082	3	0	
Tidd - FBC	LF	Selenium	0.101666667	3	2	
Titus - Ash	LF	Aluminum	4.4135	4	0	
Titus - Ash	LF	Antimony	0.04375	4	4	
Titus - Ash	LF	Arsenic	0.346	2	1	
Titus - Ash	LF	Barium	0.3	4	0	
Titus - Ash	LF	Boron	7.345	2	0	
Titus - Ash	LF	Cadmium	0.0115	4	0	
Titus - Ash	LF	Cobalt	0.027	2	0	
Titus - Ash	LF	Lead	0.19375	2	2	
Titus - Ash	LF	Mercury	0.001	2	2	
Titus - Ash	LF	Molybdenum	0.34	2	0	
Titus - Ash	LF	Selenium	0.144	4	3	
Titus - Ash	LF	Thallium	0.01	2	0	
Tracy Vein Slope - Ash	LF	Aluminum	0.533833333	6	0	11090
Tracy Vein Slope - Ash	LF	Antimony	0.05	5	0	24.215
Tracy Vein Slope - Ash	LF	Arsenic	0.065166667	6	0	61.33333333
Tracy Vein Slope - Ash	LF	Barium	0.148833333	6	0	99.31666667
Tracy Vein Slope - Ash	LF	Boron	1.4486	5	0	122.4333333
Tracy Vein Slope - Ash	LF	Cadmium	0.044833333	6	0	1.070166667
Tracy Vein Slope - Ash	LF	Lead	0.075	6	0	18.90833333
Tracy Vein Slope - Ash	LF	Mercury	0.001	2	0	1.5888
Tracy Vein Slope - Ash	LF	Molybdenum	0.1662	5	0	7.721666667
Tracy Vein Slope - Ash	LF	Selenium	0.0524	5	0	8.608
Tracy Vein Slope - FBC	LF	Aluminum	1.32	1	0	7240
Tracy Vein Slope - FBC	LF	Arsenic	0.052	1	0	6.97
Tracy Vein Slope - FBC	LF	Barium	0.056	1	0	68.9
Tracy Vein Slope - FBC	LF	Boron	0.043	1	0	7.43
Tracy Vein Slope - FBC	LF	Molybdenum	0.027	1	0	0.84
Tracy Vein Slope - FBC	LF	Selenium	0.039	1	0	3.22
UAPP - Ash	LF	Arsenic	0.0025	2	2	
UAPP - Ash	LF	Barium	0.4	2	1	
UAPP - Ash	LF	Cadmium	0.04	2	2	
UAPP - Ash	LF	Lead	0.1	2	2	
UAPP - Ash	LF	Mercury	0.025	2	2	
UAPP - Ash	LF	Selenium	0.00275	2	1	
Universal - Ash	LF	Aluminum	2.057777778	9	0	6000.222222

(continued)

CCW Constituent Data (continued)

Site/Waste Type	WMU Type	Chemical	Leachate (mg/L)	No. of Leachate Measure-ments	No. of Leachate Non-detects	Total (mg/kg)
Universal - Ash	LF	Arsenic	0.277818182	11	2	41.50909091
Universal - Ash	LF	Barium	0.090181818	11	1	71
Universal - Ash	LF	Boron	2.754545455	11	0	180.2954545
Universal - Ash	LF	Cadmium	0.003227273	11	9	2.115909091
Universal - Ash	LF	Lead	0.022145455	11	7	33.00909091
Universal - Ash	LF	Mercury	0.000386364	11	11	0.137272727
Universal - Ash	LF	Molybdenum	0.134363636	11	1	3.554545455
Universal - Ash	LF	Selenium	0.160090909	11	2	7.106363636
Wansley - Ash	LF	Arsenic	0.05	1	0	
Wansley - Ash	LF	Barium	0.2	1	0	
Wansley - Ash	LF	Cadmium	0.09	1	0	
Wansley - Ash	LF	Lead	0.02	1	0	
Wansley - Ash	LF	Selenium	0.06	1	0	
WEPCO CALEDONIA LANDFILL - Ash	LF	Barium	0.225	2	0	
WEPCO CALEDONIA LANDFILL - Ash	LF	Boron	16.90454545	22	0	
WEPCO CALEDONIA LANDFILL - Ash	LF	Cadmium	0.000045	3	3	
WEPCO CALEDONIA LANDFILL - Ash	LF	Lead	0.003566667	3	3	
WEPCO CALEDONIA LANDFILL - Ash	LF	Molybdenum	0.77500575	4	3	
WEPCO CALEDONIA LANDFILL - Ash	LF	Selenium	0.046794118	34	0	
WEPCO HWY 32 LANDFILL - Ash	LF	Boron	83.41666667	12	0	
WEPCO HWY 32 LANDFILL - Ash	LF	Selenium	0.006675	12	4	
WEPCO SYSTEMS CONTROL CENTER A - Ash	LF	Arsenic	0.0055	2	0	
WEPCO SYSTEMS CONTROL CENTER A - Ash	LF	Barium	0.1195	2	0	
WEPCO SYSTEMS CONTROL CENTER A - Ash	LF	Boron	14.02134483	29	0	
WEPCO SYSTEMS CONTROL CENTER A - Ash	LF	Cadmium	0.010266667	3	1	
WEPCO SYSTEMS CONTROL CENTER A - Ash	LF	Lead	0.00625	2	1	

(continued)

CCW Constituent Data (continued)

Site/Waste Type	WMU Type	Chemical	Leachate (mg/L)	No. of Leachate Measure-ments	No. of Leachate Non-detects	Total (mg/kg)
WEPCO SYSTEMS CONTROL CENTER A - Ash	LF	Mercury	0.0002	1	0	
WEPCO SYSTEMS CONTROL CENTER A - Ash	LF	Molybdenum	0.000022375	4	4	
WEPCO SYSTEMS CONTROL CENTER A - Ash	LF	Nitrate/Nitrite	1.866666667	3	0	
WEPCO SYSTEMS CONTROL CENTER A - Ash	LF	Selenium	0.06332275	28	0	
Wilton Site - Ash	LF	Aluminum	3	1	0	
Wilton Site - Ash	LF	Arsenic	0.027	1	0	
Wilton Site - Ash	LF	Barium	0.51	1	0	
Wilton Site - Ash	LF	Boron	25	1	0	
Wilton Site - Ash	LF	Cadmium	0.0025	2	2	
Wilton Site - Ash	LF	Lead	0.0025	2	2	
Wilton Site - Ash	LF	Mercury	0.001	2	2	
Wilton Site - Ash	LF	Molybdenum	0.34	1	0	
Wilton Site - Ash	LF	Nitrate/Nitrite	0.5	1	1	
Wilton Site - Ash	LF	Selenium	0.09	1	0	
WIS PUBLIC SERV CORP-WESTON AS - Ash	LF	Arsenic	0.0014	3	2	
WIS PUBLIC SERV CORP-WESTON AS - Ash	LF	Barium	0.183025	4	1	
WIS PUBLIC SERV CORP-WESTON AS - Ash	LF	Boron	6.363333333	21	1	
WIS PUBLIC SERV CORP-WESTON AS - Ash	LF	Cadmium	0.0047595	8	0	
WIS PUBLIC SERV CORP-WESTON AS - Ash	LF	Lead	0.00668375	8	0	
WIS PUBLIC SERV CORP-WESTON AS - Ash	LF	Mercury	0.000082	5	5	
WIS PUBLIC SERV CORP-WESTON AS - Ash	LF	Selenium	0.011077619	21	1	
Yates1 - Ash	LF	Arsenic	0.1	1	0	
Yates1 - Ash	LF	Barium	0.3	1	0	
Yates1 - Ash	LF	Cadmium	0.02	1	0	
Yates1 - Ash	LF	Lead	0.05	1	0	
Yates1 - Ash	LF	Selenium	0.02	1	0	
Yates2 - Ash	LF	Arsenic	0.09	1	0	
Yates2 - Ash	LF	Barium	0.2	1	0	
Yates2 - Ash	LF	Cadmium	0.02	1	0	
Yates2 - Ash	LF	Lead	0.03	1	0	

(continued)

CCW Constituent Data (continued)

Site/Waste Type	WMU Type	Chemical	Leachate (mg/L)	No. of Leachate Measure-ments	No. of Leachate Non-detects	Total (mg/kg)
Yates2 - Ash	LF	Selenium	0.05	1	0	
AP - Ash	SI	Aluminum	0.553384615	13	0	
AP - Ash	SI	Antimony	0.01	1	1	
AP - Ash	SI	Arsenic	0.070933333	15	0	
AP - Ash	SI	Barium	0.063066667	15	1	
AP - Ash	SI	Boron	12.50986667	15	0	
AP - Ash	SI	Cadmium	0.001042857	14	7	
AP - Ash	SI	Cobalt	0.01	1	1	
AP - Ash	SI	Lead	0.001723333	15	14	
AP - Ash	SI	Molybdenum	0.486733333	15	2	
AP - Ash	SI	Nitrate/Nitrite	0.254809524	29	22	
AP - Ash	SI	Selenium	0.044326667	15	1	
AP - Ash	SI	Thallium	0.0025	1	1	
BR - Ash and Coal Refuse	SI	Aluminum	89.12777778	18	0	
BR - Ash and Coal Refuse	SI	Arsenic	0.775383333	15	4	
BR - Ash and Coal Refuse	SI	Barium	0.188055556	18	14	
BR - Ash and Coal Refuse	SI	Boron	3.857694444	18	2	
BR - Ash and Coal Refuse	SI	Cadmium	0.175	18	7	
BR - Ash and Coal Refuse	SI	Cobalt	0.204722222	18	11	
BR - Ash and Coal Refuse	SI	Molybdenum	0.5	18	18	
C - Ash	SI	Aluminum	4.192307692	13	0	
C - Ash	SI	Antimony	0.07	10	10	
C - Ash	SI	Arsenic	0.15	10	0	
C - Ash	SI	Barium	0.113769231	13	0	
C - Ash	SI	Boron	10.96428571	14	0	
C - Ash	SI	Cadmium	0.0025	10	10	
C - Ash	SI	Cobalt	0.005	10	10	
C - Ash	SI	Lead	0.00229	10	5	
C - Ash	SI	Molybdenum	0.585384615	13	0	
C - Ash	SI	Nitrate/Nitrite	10.85474359	16	3	
C - Ash	SI	Selenium	0.0175	10	2	
C - Ash	SI	Thallium	0.05	10	10	
CADK - Ash	SI	Aluminum	0.165	2	0	
CADK - Ash	SI	Arsenic	0.0075	2	2	
CADK - Ash	SI	Barium	0.02	2	2	
CADK - Ash	SI	Boron	60.05	2	0	
CADK - Ash	SI	Cadmium	0.001	2	2	
CADK - Ash	SI	Lead	0.1	2	2	
CADK - Ash	SI	Molybdenum	1.165	2	0	
CADK - Ash	SI	Nitrate/Nitrite	11.135	4	0	
CADK - Ash	SI	Selenium	0.125	2	0	

(continued)

CCW Constituent Data (continued)

Site/Waste Type	WMU Type	Chemical	Leachate (mg/L)	No. of Leachate Measure-ments	No. of Leachate Non-detects	Total (mg/kg)
CASJ - Ash	SI	Aluminum	0.1108	5	4	
CASJ - Ash	SI	Arsenic	5.37225	4	0	
CASJ - Ash	SI	Barium	0.0214	5	2	
CASJ - Ash	SI	Boron	46.02	5	0	
CASJ - Ash	SI	Cadmium	0.0156	5	3	
CASJ - Ash	SI	Lead	0.21	5	4	
CASJ - Ash	SI	Molybdenum	0.13	5	5	
CASJ - Ash	SI	Nitrate/Nitrite	1.882	10	8	
CASJ - Ash	SI	Selenium	0.40575	4	0	
CATT - Ash	SI	Aluminum	0.28	2	0	
CATT - Ash	SI	Arsenic	0.206	2	0	
CATT - Ash	SI	Barium	0.085	2	0	
CATT - Ash	SI	Boron	110.5	2	0	
CATT - Ash	SI	Cadmium	0.002	2	1	
CATT - Ash	SI	Lead	0.2275	2	0	
CATT - Ash	SI	Molybdenum	0.655	2	0	
CATT - Ash	SI	Nitrate/Nitrite	0.01	2	0	
CATT - Ash	SI	Selenium	1.025	2	0	
CL - Ash and Coal Refuse	SI	Aluminum	4.680970556	30	2	
CL - Ash and Coal Refuse	SI	Arsenic	0.493663408	30	2	
CL - Ash and Coal Refuse	SI	Barium	0.550251717	30	0	
CL - Ash and Coal Refuse	SI	Boron	1.092075	30	0	
CL - Ash and Coal Refuse	SI	Cadmium	0.001680507	30	27	
CL - Ash and Coal Refuse	SI	Lead	0.003384333	30	29	
CL - Ash and Coal Refuse	SI	Molybdenum	0.377590556	30	0	
CL - Ash and Coal Refuse	SI	Nitrate/Nitrite	0.6303	60	13	
CL - Ash and Coal Refuse	SI	Selenium	0.147525085	30	9	
CY - Ash	SI	Aluminum	6.0975	4	0	
CY - Ash	SI	Arsenic	0.1975	4	0	
CY - Ash	SI	Barium	0.179725	4	0	
CY - Ash	SI	Boron	0.025	4	4	
CY - Ash	SI	Cadmium	0.0040625	4	4	
CY - Ash	SI	Lead	0.008125	4	4	
CY - Ash	SI	Molybdenum	0.655	4	0	
CY - Ash	SI	Nitrate/Nitrite	750.2625	8	5	
CY - Ash	SI	Selenium	0.086575	4	1	
FC - Ash and Coal Refuse	SI	Aluminum	11.433	10	0	
FC - Ash and Coal Refuse	SI	Arsenic	0.00752	10	8	
FC - Ash and Coal Refuse	SI	Barium	0.14918	10	0	
FC - Ash and Coal Refuse	SI	Boron	0.7445	10	1	
FC - Ash and Coal Refuse	SI	Cadmium	0.001956	10	9	

(continued)

CCW Constituent Data (continued)

Site/Waste Type	WMU Type	Chemical	Leachate (mg/L)	No. of Leachate Measure-ments	No. of Leachate Non-detects	Total (mg/kg)
FC - Ash and Coal Refuse	SI	Lead	0.0025	10	10	
FC - Ash and Coal Refuse	SI	Molybdenum	0.2275	10	10	
FC - Ash and Coal Refuse	SI	Nitrate/Nitrite	0.2	20	20	
FC - Ash and Coal Refuse	SI	Selenium	0.02174	10	0	
HA - Ash	SI	Aluminum	2.830833333	9	2	
HA - Ash	SI	Arsenic	0.086774333	9	2	
HA - Ash	SI	Barium	0.471945556	9	0	
HA - Ash	SI	Boron	2.283583333	9	0	
HA - Ash	SI	Cadmium	0.00125	9	9	
HA - Ash	SI	Lead	0.003503333	9	8	
HA - Ash	SI	Molybdenum	0.107333333	9	4	
HA - Ash	SI	Nitrate/Nitrite	1.968222222	18	10	
HA - Ash	SI	Selenium	0.01	9	9	
HA - Ash and Coal Refuse	SI	Aluminum	0.65	1	0	
HA - Ash and Coal Refuse	SI	Arsenic	0.18	1	0	
HA - Ash and Coal Refuse	SI	Barium	0.11	1	0	
HA - Ash and Coal Refuse	SI	Boron	1.7	1	0	
HA - Ash and Coal Refuse	SI	Cadmium	0.0025	1	1	
HA - Ash and Coal Refuse	SI	Lead	0.025	1	1	
HA - Ash and Coal Refuse	SI	Mercury	0.00025	1	1	
HA - Ash and Coal Refuse	SI	Molybdenum	0.075	1	1	
HA - Ash and Coal Refuse	SI	Selenium	0.0025	1	1	
L - Ash	SI	Aluminum	0.015	2	2	
L - Ash	SI	Barium	0.001	2	2	
L - Ash	SI	Boron	0.62	2	0	
L - Ash	SI	Cadmium	0.001	2	2	
L - Ash	SI	Molybdenum	0.1675	2	1	
MO - Ash	SI	Aluminum	0.894458333	6	0	
MO - Ash	SI	Arsenic	0.011755993	6	3	
MO - Ash	SI	Barium	0.019379487	6	0	
MO - Ash	SI	Boron	0.085041667	6	2	
MO - Ash	SI	Cadmium	0.00125	6	6	
MO - Ash	SI	Lead	0.003666667	6	5	
MO - Ash	SI	Molybdenum	0.928770833	6	3	
MO - Ash	SI	Nitrate/Nitrite	0.1205	12	10	
MO - Ash	SI	Selenium	0.005	6	6	
MO - Ash and Coal Refuse	SI	Aluminum	296.2888026	19	6	
MO - Ash and Coal Refuse	SI	Arsenic	11.67554177	20	0	
MO - Ash and Coal Refuse	SI	Barium	0.039930301	20	1	
MO - Ash and Coal Refuse	SI	Boron	15.49313158	19	2	
MO - Ash and Coal Refuse	SI	Cadmium	0.124406392	27	9	

(continued)

CCW Constituent Data (continued)

Site/Waste Type	WMU Type	Chemical	Leachate (mg/L)	No. of Leachate Measure-ments	No. of Leachate Non-detects	Total (mg/kg)
MO - Ash and Coal Refuse	SI	Cobalt	4.8377	20	7	
MO - Ash and Coal Refuse	SI	Lead	0.321181411	20	11	
MO - Ash and Coal Refuse	SI	Molybdenum	0.402184211	19	15	
MO - Ash and Coal Refuse	SI	Nitrate/Nitrite	5.165	39	37	
MO - Ash and Coal Refuse	SI	Selenium	0.103823054	20	9	
O - Ash	SI	Arsenic	0.234766667	3	0	
O - Ash	SI	Boron	6.166666667	3	0	
O - Ash	SI	Molybdenum	0.0179	1	0	
O - Ash	SI	Nitrate/Nitrite	461	1	0	
O - Ash	SI	Selenium	0.0029	3	0	
OK - Ash	SI	Aluminum	40.45955556	9	0	
OK - Ash	SI	Arsenic	0.060628889	9	2	
OK - Ash	SI	Barium	0.159055556	9	1	
OK - Ash	SI	Boron	3.148333333	9	0	
OK - Ash	SI	Cadmium	0.01	9	9	
OK - Ash	SI	Lead	0.02	9	9	
OK - Ash	SI	Molybdenum	0.721694444	9	0	
OK - Ash	SI	Nitrate/Nitrite	7.62	18	17	
OK - Ash	SI	Selenium	0.282377778	9	2	
SX - Ash	SI	Aluminum	3.866609827	15	0	
SX - Ash	SI	Arsenic	0.054834273	15	2	
SX - Ash	SI	Barium	0.079191593	15	0	
SX - Ash	SI	Boron	32.70433889	15	0	
SX - Ash	SI	Cadmium	0.019243353	15	5	
SX - Ash	SI	Lead	0.001228153	15	5	
SX - Ash	SI	Molybdenum	11.40518778	15	0	
SX - Ash	SI	Nitrate/Nitrite	1.6328	30	12	
SX - Ash	SI	Selenium	0.239368793	15	6	

[This page intentionally left blank.]

Attachment A-3: CCW Constituent Data Used in Screening Analysis

Table A-3-1. CCW Landfill Waste Analyte Concentrations Used in Screening: Total Waste Analyses (mg/kg)

Analyte	Sites[1]	ND Sites[2]	2002 CCW Total Waste Concentrations						1998 Total Waste 95th
			Minimum	Maximum	50th	75th	90th	95th	
Aluminum	71	0	1.45E+01	1.37E+05	2.53E+04	4.17E+04	8.57E+04	9.76E+04	1.43E+05
Antimony	64	19	1.25E-01	3.10E+02	1.56E+01	2.94E+01	4.62E+01	7.93E+01	4.67E+01
Arsenic	111	3	4.70E-02	3.70E+02	2.79E+01	6.18E+01	1.05E+02	1.25E+02	1.54E+02
Barium	94	1	4.76E+00	7.14E+03	2.22E+02	4.49E+02	1.05E+03	2.59E+03	8.38E+03
Beryllium	37	6	1.19E-01	2.85E+01	4.10E+00	1.00E+01	1.76E+01	2.25E+01	1.56E+01
Boron	70	4	2.50E-02	2.47E+03	5.35E+01	1.50E+02	3.46E+02	5.54E+02	4.17E+02
Cadmium	102	21	1.65E-04	7.60E+02	1.08E+00	2.26E+00	5.43E+00	1.12E+01	2.37E+01
Chromium	108	2	5.00E-03	1.38E+03	4.45E+01	7.62E+01	1.66E+02	1.81E+02	2.91E+02
Cobalt	67	8	5.00E-03	1.35E+02	1.02E+01	3.26E+01	6.22E+01	7.93E+01	4.16E+01
Copper	95	3	5.00E-03	8.90E+02	3.61E+01	8.24E+01	2.28E+02	2.99E+02	1.55E+02
Cyanide	2	1	1.25E-01	2.48E-01	1.86E-01	2.17E-01	2.35E-01	2.41E-01	-
Fluoride	8	0	2.50E+00	7.61E+02	1.08E+01	2.07E+01	2.49E+02	5.05E+02	-
Lead	107	6	1.30E-02	1.37E+03	2.87E+01	4.97E+01	8.06E+01	1.25E+02	1.52E+02
Manganese	87	2	5.00E-02	9.81E+03	1.11E+02	2.41E+02	5.10E+02	6.37E+02	8.17E+02
Mercury	86	12	6.00E-04	6.43E+01	3.28E-01	6.00E-01	1.63E+00	8.22E+00	-
Molybdenum	73	7	4.43E-02	1.26E+02	1.20E+01	2.23E+01	3.47E+01	5.38E+01	4.31E+01
Nickel	106	5	4.90E-02	5.41E+04	4.23E+01	1.30E+02	3.29E+02	6.79E+02	1.55E+02
Nitrate	1	1	2.43E-01	2.43E-01	2.43E-01	2.43E-01	2.43E-01	2.43E-01	-
Nitrite	0	0	-	-	-	-	-	-	-
Selenium	94	11	5.05E-03	6.73E+02	5.12E+00	1.03E+01	2.14E+01	4.79E+01	3.24E+02
Silver	69	26	5.00E-02	1.90E+03	1.72E+00	3.30E+00	1.37E+01	2.66E+01	1.36E+01
Strontium	15	1	5.60E+00	1.23E+03	2.63E+02	7.63E+02	1.05E+03	1.20E+03	4.76E+03
Thallium	20	10	9.00E-02	1.00E+02	3.23E+00	1.05E+01	2.08E+01	4.21E+01	4.80E+01
Vanadium	43	1	3.30E+00	4.55E+03	2.24E+02	3.48E+02	9.07E+02	2.95E+03	3.46E+02
Zinc	98	1	3.40E-02	1.82E+04	4.58E+01	1.44E+02	2.93E+02	1.43E+03	8.56E+02

[1] Number of sites with analyte data (2002)

[2] Number of sites with only nondetect analyte data (2002)

Table A-3-2. CCW Surface Impoundment Waste Analyte Concentrations Used in Screening: Porewater Analyses (mg/L)

Analyte	Sites[1]	ND Sites[2]	2002 SI Porewater Concentrations						1998 Porewater 95th[3]
			Minimum	Maximum	50th	75th	90th	95th	
Aluminum	17	2	1.50E-02	8.91E+01	1.18E+00	4.68E+00	2.30E+01	5.02E+01	2.70E+02
Antimony	2	2	1.00E-02	7.00E-02	4.00E-02	5.50E-02	6.40E-02	6.70E-02	-
Arsenic	17	2	7.50E-03	6.77E+00	1.80E-01	4.94E-01	5.18E+00	5.65E+00	9.64E+00
Barium	17	2	1.00E-03	5.50E-01	1.10E-01	1.59E-01	3.02E-01	4.88E-01	2.74E+01
Beryllium	2	1	1.00E-03	6.20E-03	3.60E-03	4.90E-03	5.68E-03	5.94E-03	-
Boron	18	1	2.50E-02	3.37E+02	5.01E+00	2.92E+01	7.52E+01	1.44E+02	3.42E+02
Cadmium	17	9	1.00E-03	2.50E-01	2.50E-03	1.56E-02	1.31E-01	1.90E-01	1.56E-01
Chromium	18	8	9.00E-04	5.78E-01	3.56E-02	1.13E-01	3.66E-01	5.29E-01	7.46E-01
Cobalt	4	2	5.00E-03	8.87E+00	1.07E-01	2.37E+00	6.27E+00	7.57E+00	-
Copper	16	5	6.40E-04	7.22E-01	3.63E-02	1.26E-01	2.84E-01	4.90E-01	6.90E-01
Cyanide	0	0	-	-	-	-	-	-	-
Fluoride	15	2	5.05E-02	4.10E+02	8.96E-01	4.99E+00	1.91E+01	1.39E+02	4.10E+02
Lead	14	5	1.23E-03	2.28E-01	5.90E-03	4.53E-02	1.77E-01	2.16E-01	4.68E-01
Manganese	16	2	4.24E-03	1.82E+02	1.69E-01	1.20E+00	7.67E+00	5.15E+01	1.03E+02
Mercury	1	1	2.50E-04	2.50E-04	2.50E-04	2.50E-04	2.50E-04	2.50E-04	7.96E-04
Molybdenum	18	6	1.79E-02	1.14E+01	4.73E-01	6.55E-01	1.00E+00	2.70E+00	1.14E+01
Nickel	17	4	5.00E-03	1.23E+01	4.61E-02	2.75E-01	7.49E-01	3.09E+00	8.33E+00
Nitrate	13	3	8.05E-02	1.17E+03	1.85E+00	4.73E+00	6.02E+02	9.17E+02	1.17E+03
Nitrite	15	4	7.00E-03	4.61E+02	1.89E-01	1.39E+00	5.22E+00	1.43E+02	4.61E+02
Selenium	15	3	2.50E-03	1.03E+00	6.97E-02	1.93E-01	3.56E-01	5.92E-01	1.03E+00
Silver	8	8	5.00E-05	5.00E-03	2.06E-03	4.25E-03	5.00E-03	5.00E-03	
Strontium	17	0	4.20E-01	1.61E+01	4.25E+00	7.00E+00	8.74E+00	1.06E+01	1.61E+01
Thallium	2	2	2.50E-03	5.00E-02	2.63E-02	3.81E-02	4.53E-02	4.76E-02	
Vanadium	15	1	1.25E-03	6.61E-01	1.03E-01	3.15E-01	4.78E-01	5.81E-01	8.00E-01
Zinc	17	5	1.16E-02	2.34E+01	1.00E-01	1.20E-01	6.70E-01	5.40E+00	2.31E+01

[1] Number of sites with analyte data (2002)
[2] Number of sites with only nondetect analyte data (2002)
[3] Includes both landfill and surface impoundment (SI) porewater data

Table A-3-3. CCW Landfill Waste Analyte Concentrations Used in Screening: Leachate Analyses (mg/L)

Analyte	Sites[1]	ND Sites[2]	Leachate Concentrations						1998 TCLP 95th
			Minimum	Maximum	50th	75th	90th	95th	
Aluminum	54	3	3.00E-02	2.86E+01	2.06E+00	4.47E+00	1.05E+01	1.36E+01	-
Antimony	60	27	6.50E-04	7.87E-01	2.19E-02	7.50E-02	2.61E-01	2.98E-01	-
Arsenic	119	26	1.00E-03	1.80E+00	3.65E-02	1.31E-01	3.94E-01	1.01E+00	2.40E-01
Barium	115	7	2.00E-02	4.20E+01	3.04E-01	5.71E-01	1.60E+00	2.55E+00	-
Beryllium	47	15	5.00E-05	2.80E-01	2.14E-03	5.37E-03	1.58E-02	2.96E-02	-
Boron	72	3	1.00E-02	2.79E+01	1.07E+00	4.57E+00	1.06E+01	2.07E+01	-
Cadmium	117	38	1.50E-04	6.00E-01	1.00E-02	2.24E-02	4.94E-02	9.00E-02	-
Chromium	118	17	1.00E-03	7.64E-01	3.40E-02	1.00E-01	2.00E-01	3.50E-01	5.90E-02
Cobalt	51	10	1.92E-03	2.46E-01	1.52E-02	2.55E-02	8.25E-02	1.31E-01	-
Copper	72	13	1.60E-03	3.27E+00	4.14E-02	9.46E-02	1.50E-01	4.55E-01	-
Cyanide	24	14	3.50E-03	1.20E-01	7.23E-03	2.03E-02	6.32E-02	8.67E-02	-
Fluoride	33	1	8.00E-02	5.99E+01	8.19E-01	1.90E+00	6.34E+00	3.09E+01	-
Lead	116	38	1.00E-03	3.61E+00	3.23E-02	7.23E-02	2.39E-01	2.90E-01	-
Manganese	72	13	1.25E-03	3.27E+00	1.63E-01	4.39E-01	1.37E+00	1.56E+00	-
Mercury	97	60	5.00E-06	2.90E-01	2.89E-04	1.00E-03	2.69E-03	1.32E-02	-
Molybdenum	49	5	1.00E-02	3.09E+01	1.77E-01	3.40E-01	6.16E-01	1.27E+00	-
Nickel	80	19	2.00E-03	3.88E+00	5.17E-02	1.41E-01	3.09E-01	5.70E-01	5.00E-02
Nitrate	17	3	1.75E-02	2.60E+01	1.59E+00	2.50E+00	2.83E+00	7.72E+00	-
Nitrite	5	4	5.00E-02	5.00E+00	8.33E-01	1.17E+00	3.47E+00	4.23E+00	-
Selenium	119	23	1.00E-03	1.05E+00	4.87E-02	8.74E-02	1.76E-01	2.06E-01	4.40E-01
Silver	109	60	0.00E+00	2.50E-01	8.70E-03	1.75E-02	3.95E-02	5.02E-02	-
Strontium	20	0	6.35E-02	4.28E+01	2.95E+00	4.87E+00	9.70E+00	1.36E+01	-
Thallium	40	18	1.00E-03	1.97E-01	8.29E-03	2.34E-02	5.00E-02	6.54E-02	-
Vanadium	40	5	1.00E-03	1.20E+01	1.07E-01	1.82E-01	4.50E-01	1.50E+00	-
Zinc	75	9	2.00E-03	5.83E+01	1.30E-01	6.09E-01	1.94E+00	1.01E+01	-

[1] Number of sites with analyte data (2002)
[2] Number of sites with only nondetect analyte data (2002)
TCLP = Toxicity Characteristic Leaching Procedure

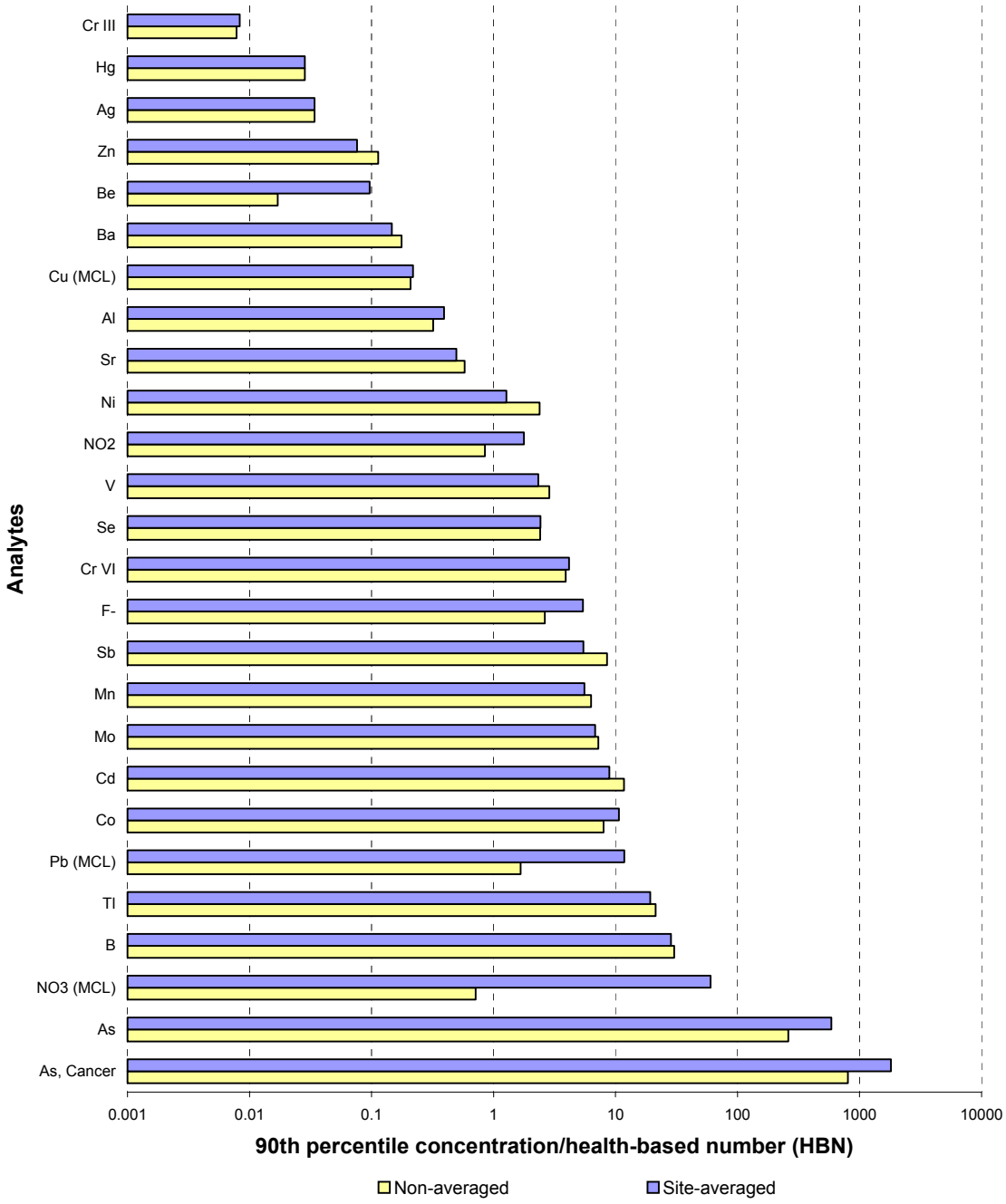


Figure A-3-1. Comparison of site-averaged and nonaveraged results for surface impoundment porewater screening, groundwater-to-drinking-water pathway.

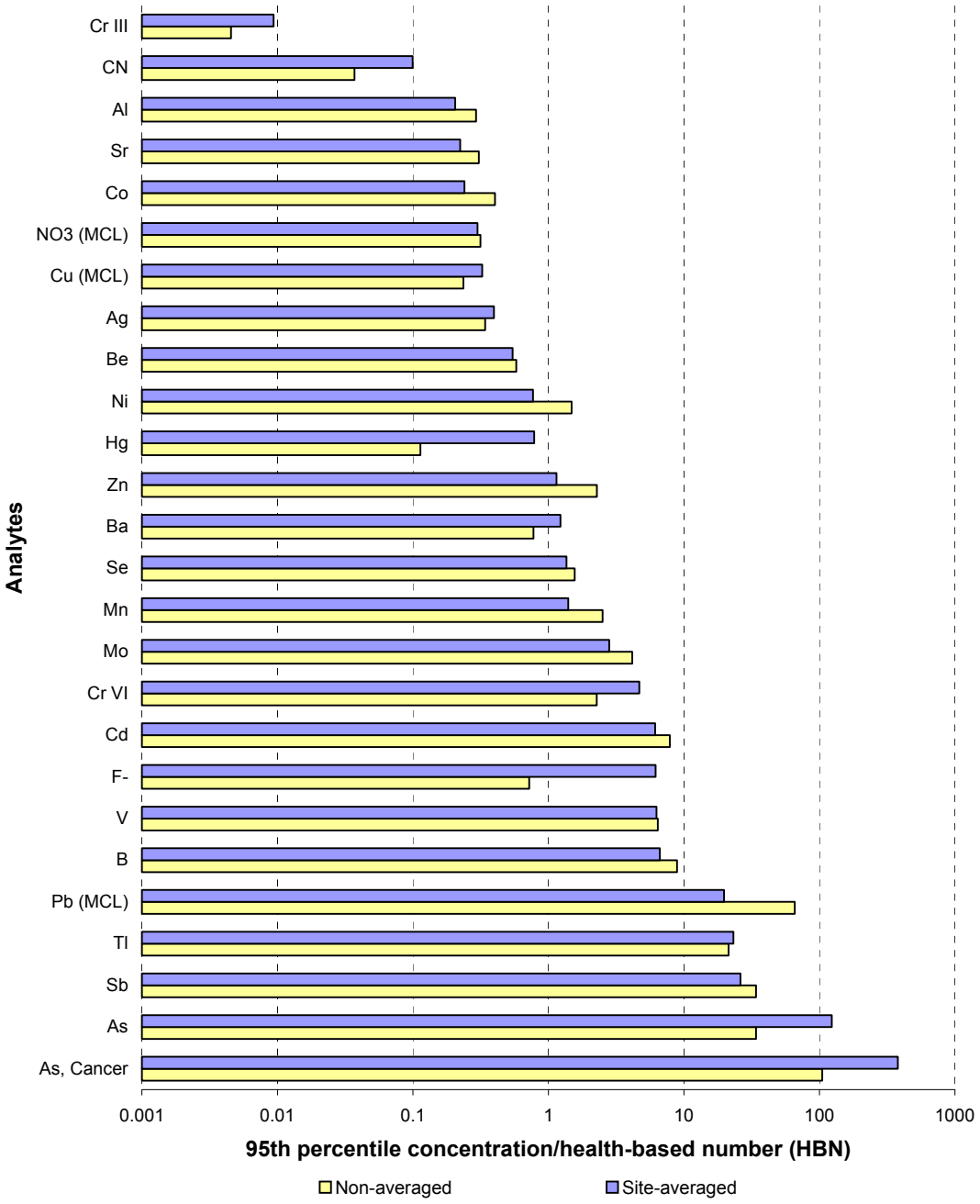


Figure A-3-2. Comparison of site-averaged and nonaveraged results for landfill leachate screening, groundwater-to-drinking-water pathway.

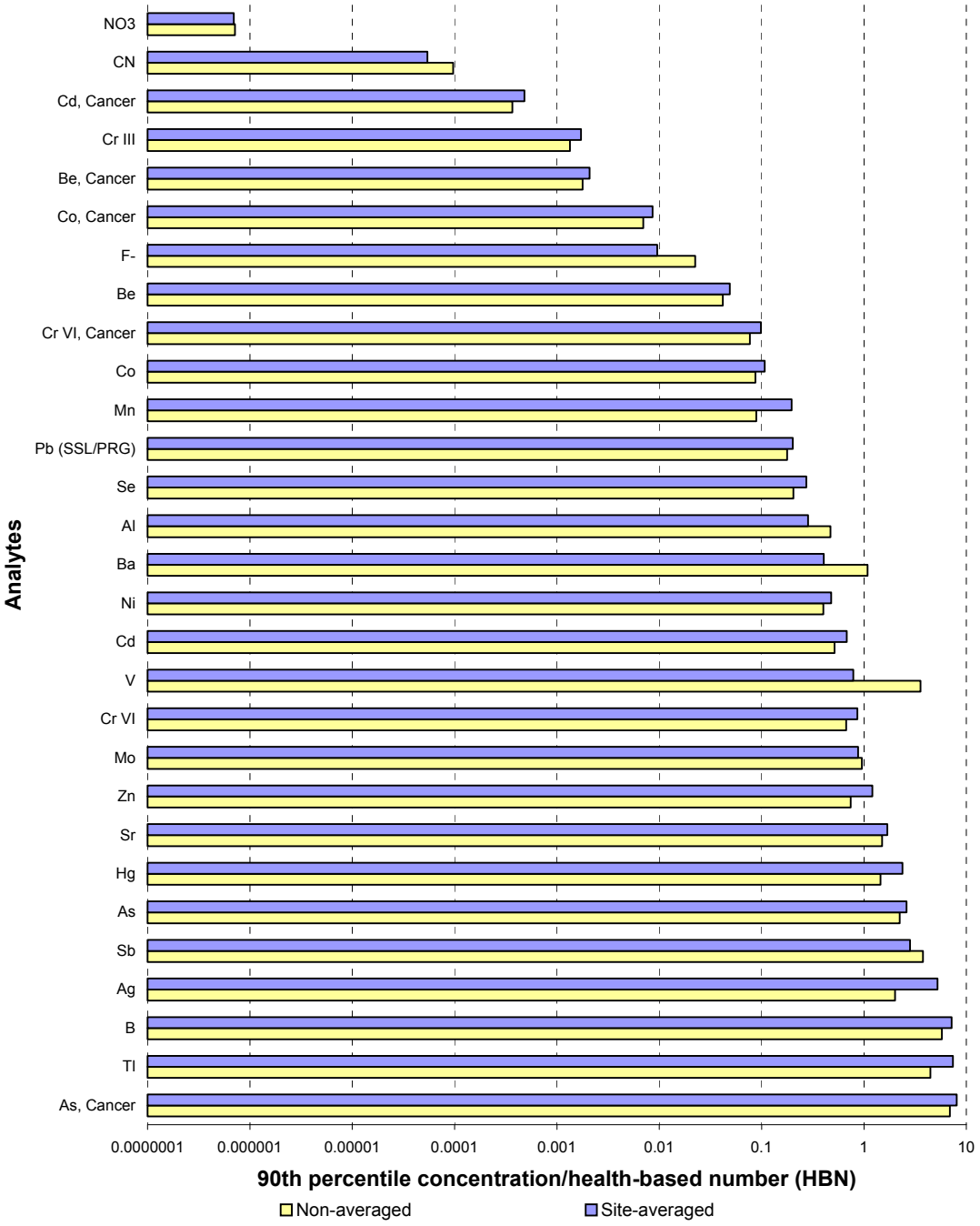


Figure A-3-3. Comparison of site-averaged and nonaveraged results for landfill total waste screening, aboveground pathways.

Appendix B. Waste Management Units

The source models supporting the CCW risk assessment require inputs describing the characteristics of CCW waste management units (WMUs). To satisfy this requirement, the assessment used a data set of WMU area, capacity, liner type, geometry, and waste type managed for a set of individual CCW landfills and surface impoundments that are representative of the national population of coal combustion facilities that are managing their wastes onsite.

The sources for these data sets were responses to two voluntary industry surveys: an Electric Power Research Institute (EPRI) comanagement survey (for conventional utility coal combustion WMUs units) and a Council of Industrial Boiler Owners (CIBO) fluidized bed combustion (FBC) survey (for FBC WMUs). In addition to the individual WMU data, certain assumptions were required regarding (1) liner types and characteristics, (2) surface impoundment operating life, and (3) above- and below-grade geometries for WMUs. The sections below describe the two industry surveys, then discuss the data sources and assumptions made. **Attachment B-1** lists the 181 CCW disposal sites modeled in this risk assessment and their locations. **Attachment B-2** presents the WMU data used in the CCW risk assessment for each of the 108 landfills and 96 surface impoundments at these coal combustion facilities.

B.1 EPRI Comanagement Survey

For conventional utility coal combustion WMUs, the source of data for area, capacity, liner type, and waste type managed was the EPRI Coal Combustion By-Products and Low-Volume Wastes Comanagement Survey (EPRI, 1997a). In 1995, EPRI sent a 4-page questionnaire to all electric utilities with more than 100 megawatts (MW) of coal-fired generating capacity. The survey gathered data on the design of coal combustion management units and the types and volumes of waste managed. From the survey responses, EPRI prepared an electronic database and provided it to EPA in support of the March 1999 *Report to Congress: Wastes from the Combustion of Fossil Fuels* (the RTC) (U.S. EPA, 1999a). EPRI also published a report (EPRI, 1997a) documenting the survey format and providing a brief summary of the results.

The EPRI survey responses include information on 323 waste management facilities serving 238 power plants located in 36 states. The total annual volume of CCW reported disposed by respondents to the EPRI comanagement survey was nearly 62 million tons. This quantity was two-thirds of the total generation of CCW in 1995. Therefore, the survey sample encompasses the majority of CCW disposed in terms of volume. Based on comparison with data from other sources, the EPRI survey sample appears representative of the population of coal combustion WMUs in terms of the types of units included (i.e., landfills and surface impoundments). The EPRI survey sample also is believed to be generally geographically representative of the population of conventional utility WMUs, although it may under-represent certain management practices in a few states. The EPA document, *Technical Background Document for the Supplemental Report to Congress on Remaining Fossil Fuel Combustion*

Wastes: Industry Statistics and Waste Management Practices (U.S. EPA, 1999b), discusses the representativeness of the EPRI survey in greater detail and provides extensive summary statistics on the survey responses.

The EPRI comanagement survey included questions requesting the respondent to report the location of the WMU (by state) and the WMU area, capacity, liner type, and waste type managed. Therefore, the data set used for modeling these variables was extracted directly from the EPRI database for all active landfills and surface impoundments responding to the EPRI survey. Mine placement sites and closed WMUs were excluded from the data set. Also excluded from the data set were three responding WMUs that managed FBC waste. Data for these units were instead combined with the data set for FBC WMUs from the CIBO FBC survey (described below).

The EPRI survey data were provided in blinded form. That is, the original database did not report the identity of each respondent and identified WMU location only by state. To provide a more complete identification of the EPRI waste management locations, each unit in the EPRI database had to be matched with a specific electric utility facility. This matching was accomplished by applying professional judgment in comparing the state, waste quantity, and waste management practice information in the EPRI database with similar data from responses to the U.S. Department of Energy's Energy Information Administration (EIA) Form EIA-767 (Steam-Electric Plant Operation and Design Report) for the same year as the EPRI survey (1995). The latitude and longitude plant locations in the EIA database allowed the pairing of the EPRI WMU data with environmental setting information.

B.2 CIBO Fluidized Bed Combustion Survey

For FBC WMUs, the primary source of data for area, capacity, liner type, and waste type managed was the CIBO Fossil Fuel Fluidized Bed Combustion (FBC) Survey. In 1996, CIBO sent a voluntary questionnaire to every fossil-fuel-fired FBC plant, both utility and nonutility, in the United States. This survey collected general facility information, characterized process inputs and outputs, gathered data on waste generation and characteristics, and captured details of FBC waste management practices. From the survey responses, CIBO prepared an electronic database and provided it to EPA in support of the March 1999 RTC. CIBO also published a report (CIBO, 1997) that includes documentation of the survey format and provides a brief summary of the results.

CIBO reports a total of 84 facilities using FBC technology. Forty-five of these responded to the CIBO FBC survey, with 20 of the respondents providing information about waste management practices. The facilities with waste management data cover 24 percent of all U.S. facilities using FBC. The CIBO sample is geographically representative of the full population, with the exception of two states that appear under-represented in the sample—Pennsylvania and Illinois. EPA's technical background document on industry statistics and waste management practices (U.S. EPA, 1999b) discusses the representativeness of the EPRI survey in greater detail and provides extensive summary statistics on the survey responses.

The CIBO survey includes questions requesting the respondent to report WMU area, capacity, liner type, and waste type managed. Therefore, the data set used for modeling these

variables was extracted directly from the CIBO database. The CIBO respondents included both utility and nonutility (i.e., industrial or institutional facilities that burn coal, but are not primarily engaged in the business of selling electricity) facilities. Because nonutilities are outside the scope of this risk assessment, nonutilities were excluded from the data set. Three additional utility facilities were excluded from the data set because their responses contained insufficient data on the variables of interest (area, capacity, liner type, and waste type). Mine placement sites also were excluded from the data set. Data for the FBC units responding to the EPRI survey (see **Section B.1**) were added to the data set. This resulted in a sample of seven FBC landfills and one FBC surface impoundment for modeling. **Table B-1** compares this sample to the waste management practices of the full utility FBC population.

As shown in Table B-1, FBC facilities frequently avoid waste disposal units by directing all of their waste to mine placement or beneficial use. Therefore, although only 8 of the 41 utility FBC facilities were included in the model data set, these 8 facilities represent nearly all of the known FBC landfills and surface impoundments.

Table B-1. Utility FBC Waste Management Practices and Units Modeled

Number of Facilities...	Total	Landfill	Surface Impoundment	Minefill or Beneficial Use	Unknown
in the full population	41	11	1	16	13
modeled	8	7	1	Not applicable	Not applicable

The CIBO survey database identified the location of each WMU in detail (latitude and longitude). Therefore, no additional analysis was necessary to pair the WMU data with environmental setting information.

B.3 Liner Type

The EPRI survey data included information on the liner (if any) for each WMU. For this assessment, the WMUs were assigned to one of three liner scenarios based on the EPRI liner data: an unlined (no liner) scenario, a compacted clay liner, and a composite liner that combines a plastic (e.g., high-density polyethylene (HDPE) membrane) over either geosynthetic or natural clays. These three scenarios correspond to the following conceptual liner scenarios, developed in support of EPA's Industrial Subtitle D guidance (U.S. EPA, 2002), which can be selected in the landfill and surface impoundment models used in this assessment.

- **Unlined Scenario.** For landfills, waste is placed directly on local soils, either on grade or excavated to some design depth and without a leachate collection system. After the landfill has been filled to capacity, a 2-foot native soil cover (the minimum required by Subtitle D regulations) is installed and assumed to support vegetation. For surface impoundments, wastewater is placed directly on local soils, and the depth of water is constant over the entire life of the impoundment, pre- and post-closure. Sediments accumulate and consolidate at the bottom of the impoundment and migrate into the underlying native soils, where they clog pore spaces and provide some barrier to flow.

- **Clay Liner Scenario.** For landfills, waste is placed directly on a 3-foot compacted clay liner, which is installed on the local soils, either on grade or excavated to some design depth and without a leachate collection system. After the landfill has been filled to capacity, a 3-foot clay cover is installed and covered with 1 foot of loam to support vegetation and drainage. The hydraulic conductivity of both the liner and cover clays is assumed to be $1x10^{-7}$ cm/sec. For surface impoundments, wastewater is placed on a compacted clay liner, which is installed on the local soils. The assumptions for an unlined impoundment also apply to the compacted clay liner scenario, except that a compacted clay liner filters out the sediments that clog the native soils in the unlined case, so the effect of clogging the native materials is not included in the calculation of the infiltration rate. The thickness of the compacted clay liner was assumed to be 3 feet and the hydraulic conductivity was assumed to be $1x10^{-7}$ cm/sec.

- **Composite Liner Scenario.** For landfills, wastes are placed on a liner system that consists of a 60 mil HDPE membrane with either an underlying geosynthetic clay liner with a maximum hydraulic conductivity of $5x10^{-9}$ cm/sec, or a 3-foot compacted clay liner with a maximum hydraulic conductivity of $1x10^{-7}$ cm/sec. A leachate collection system is also assumed to exist between the waste and the liner system. After the landfill has been filled to capacity, a 3-foot clay cover is assumed to be installed and covered with 1 foot of loam to support vegetation and drainage. For surface impoundments, wastewater is placed on a synthetic membrane with an underlying geosynthetic or natural compacted clay liner with a hydraulic conductivity of $1x10^{-7}$ cm/sec. The membrane liner was assumed to have a number of pinhole leaks of uniform size (6 mm^2). The number of these leaks was based on an empirical distribution of membrane leak density values obtained from TetraTech (2001), as described in the *IWEM Technical Background Document* (U.S. EPA, 2002).

Table B-2 shows the crosswalk used to assign one of the three liner scenarios to each facility based on the liner data in the EPRI survey data (EPRI, 1997a). **Attachment B-2** provides these assignments, along with the original EPRI liner type, for each CCW landfill facility modeled.

Table B-2. Crosswalk Between EPRI and CCW Source Model Liner Types

EPRI Liner Type	Model Liner Code	Description
Compacted ash	0	no liner
Compacted clay	1	clay
Composite clay/membrane	2	composite
Double	2	composite
Geosynthetic membrane	2	composite
None/natural soils	0	no liner

B.4 Surface Impoundment Operating Life

The model runs for surface impoundments required a general assumption about the length of the operating life for these WMUs. Of the surface impoundments in the EPRI comanagement survey, 86 provided responses to questions about both the unit's opening date and expected closure date. From these two dates, an expected operating life for each impoundment could be calculated. An additional 30 impoundments provided an opening date, but no closure date. One possible interpretation of these responses is that these facilities do not expect to close in the foreseeable future, corresponding to a very long or indefinite operating life with dredging of waste to maintain capacity. **Figure B-1** shows the distribution of the calculated operating lives, along with a bar showing the facilities with no closure date.

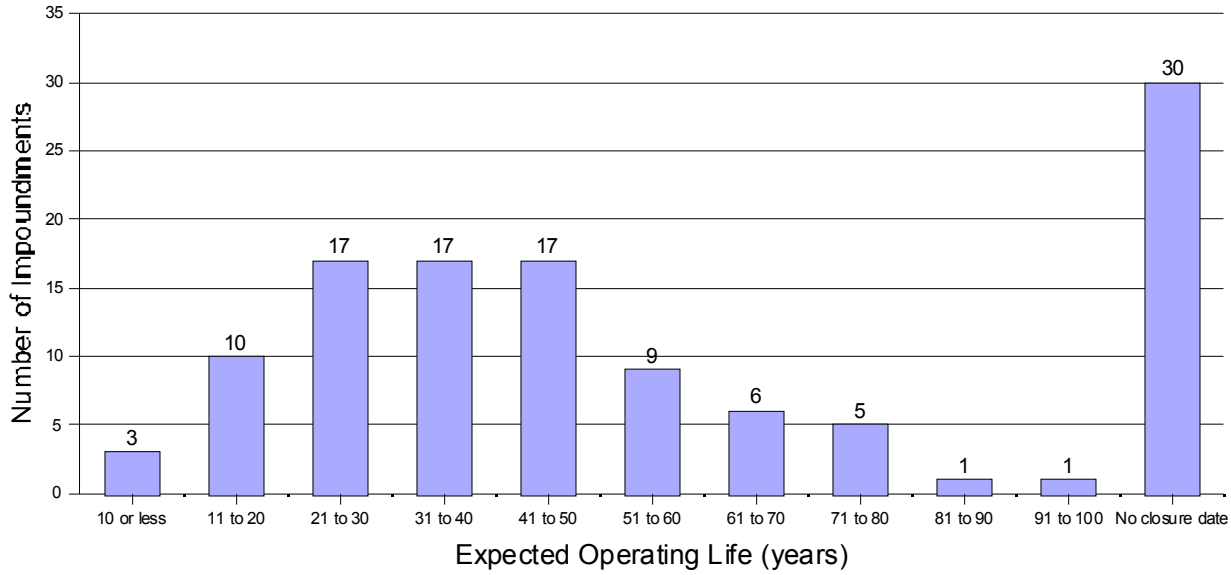


Figure B-1. Operating life of impoundments in the EPRI survey.

Based on these data, a 75-year operating life was chosen. This value corresponds to the 95th percentile of the observed distribution. While the use of a 95th percentile value may appear conservative, if many of the facilities with no closure date do, in fact, plan to operate indefinitely, 75 years would correspond to a much lower percentile in the distribution. More significantly, many CCW surface impoundments close with wastes in place. The selection of 75 years minimizes the underestimation of chronic risks for this scenario, given that EPA's Composite Model for Leachate Migration with Transformation Products (EPACMTP) surface impoundment model assumes clean closure after the operating life.

B.5 Above- and Below-Grade Geometry

The model runs for surface impoundments and landfills required general assumptions about the geometry of these units with respect to the ground surface (i.e., how much of the unit's depth is below grade). The CIBO FBC survey included data on this geometry, so, for FBC units, these data were extracted directly from the database along with the other individual WMU data

(e.g., capacity). The EPRI comanagement survey did not contain data describing above- and below-grade geometry. Therefore, for conventional utility coal combustion WMUs, EPA reviewed 17 site-characterization reports published by EPRI (EPRI 1991; 1992; 1994a,b; 1996a,b; 1997b-k) and determined an above- versus below-grade geometry for each unit described in those reports based on schematic diagrams and site descriptions. EPA also extracted data from another CIBO voluntary survey that covered conventional (non-FBC) nonutility coal combustors. **Figures B-2** and **B-3** display the distributions of the data thus collected.

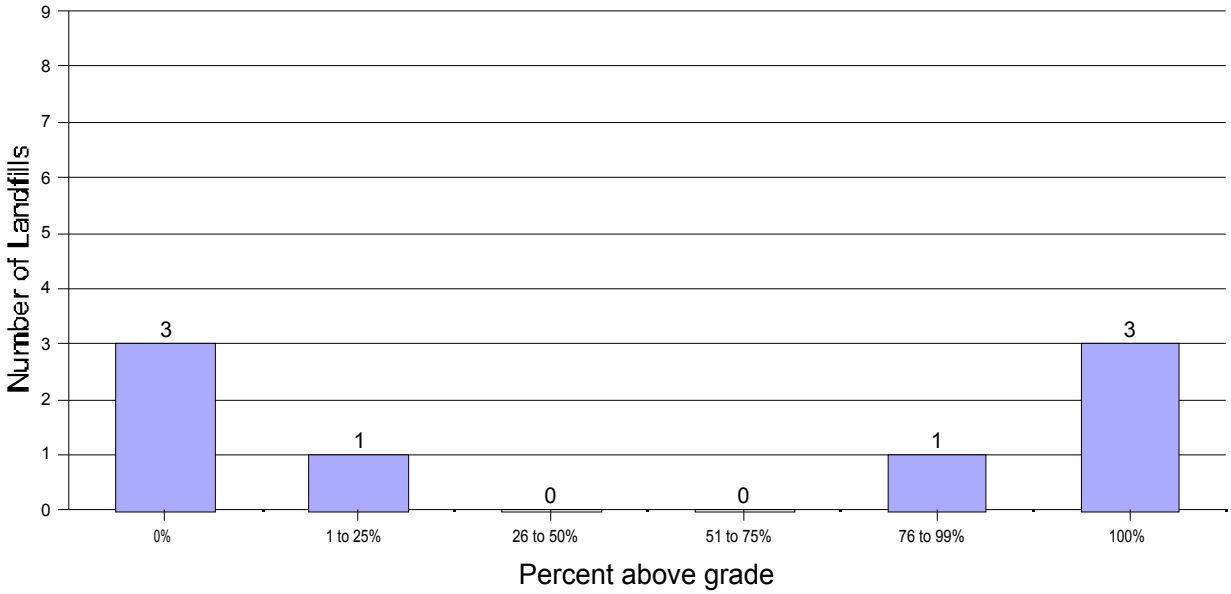


Figure B-2. Above- and below-grade geometry for landfills.

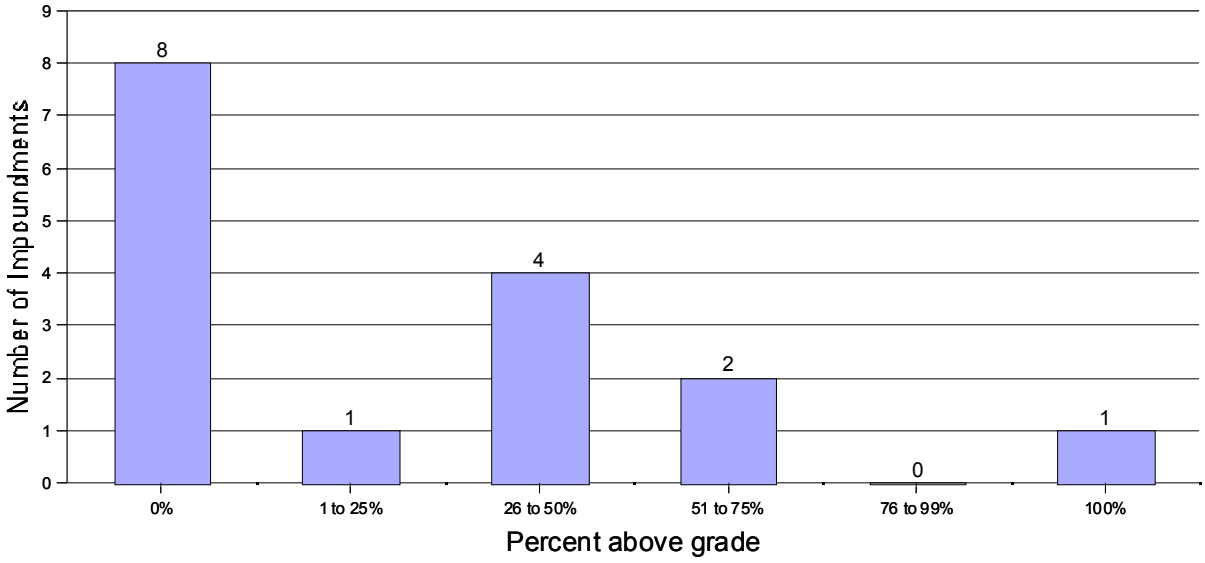


Figure B-3. Above- and below-grade geometry for impoundments.

For landfills, because the data were limited (8 sites), the model runs assumed that the percent below grade ranged from 1 to 100 and was uniformly distributed. For each landfill iteration, a random value for percent below grade was picked and applied to the landfill depth to determine depth below ground surface. This value was constrained to be no deeper than the water table and was checked to see that EPACMTP groundwater mounding constraints were not violated.

For surface impoundments, more data were available (16 sites), with 8 sites being constructed entirely below grade and the remaining 8 sites ranging from 7.5 to 45 feet above grade. For each surface impoundment iteration, height above grade at these 15 sites was randomly sampled as an empirical distribution and applied to the overall surface impoundment depth to determine depth below ground surface.

B.6 Calculation of WMU Depth and Imputation of Missing WMU Data

The EPRI survey includes information on the total area and total waste capacity of each landfill and surface impoundment included in the survey. To calculate average depth for each WMU (a necessary EPACMTP model input), the total waste capacity was divided by the area. The resulting depths were then checked for reasonableness. For surface impoundments, one depth (1 foot) was culled as being unrealistically low and one (700 feet) as too high. Two landfill depths less than 2 feet and one depth greater than 350 feet were also removed from the database. In these cases the EPRI waste capacity data were culled and replaced using the regressions described below (i.e., WMU areas are considered more reliable than the capacity estimates in the survey data), and new capacities were estimated as described below.

In addition, four landfills and six surface impoundments had neither area nor capacity data in the EPRI survey. In these cases, the EIA facility locations were used to find the plants and their WMUs on aerial photos from the Terraserver Web site (http://terraserver-usa.com/geographic.aspx), and a geographic information system (GIS) was used to measure the areas of the units in question. Capacities were then estimated as described below.

To impute data for facilities missing either area or capacity data in the EPRI survey, linear regression equations were developed based on WMUs with both area and capacity data, one to predict area from capacity, and one to predict capacity from area. The final regression equations are shown in **Figures B-4** and **B-5** for landfills and **Figures B-6** and **B-7** for surface impoundments. In each case, a standard deviation around the regression line was also computed and used during source data file preparation to randomly vary the area or capacity from iteration to iteration within the bounds of the existing data set.

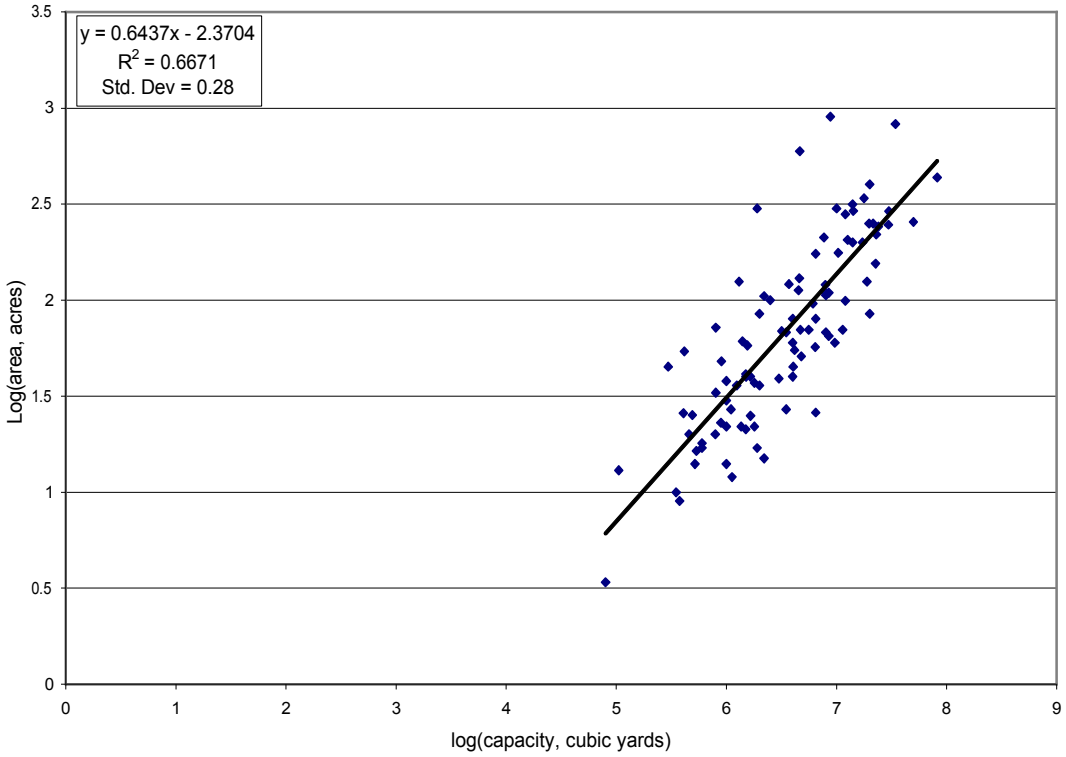


Figure B-4. Linear regression to impute landfill area from capacity.

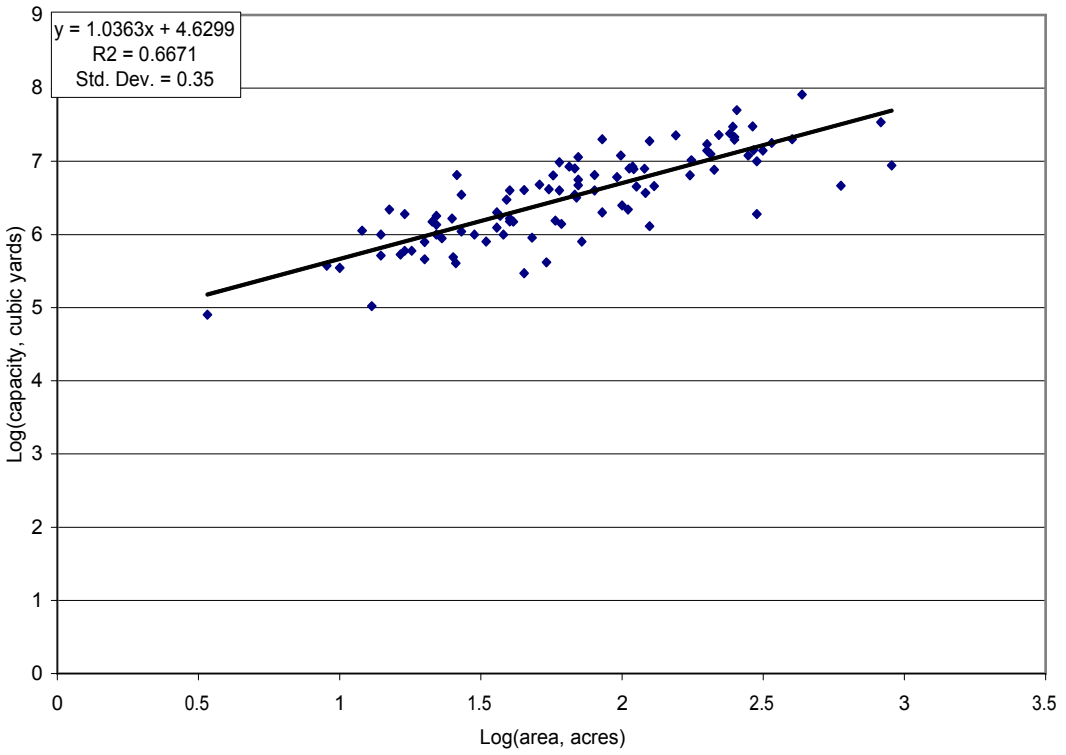


Figure B-5. Linear regression to impute landfill capacity from area.

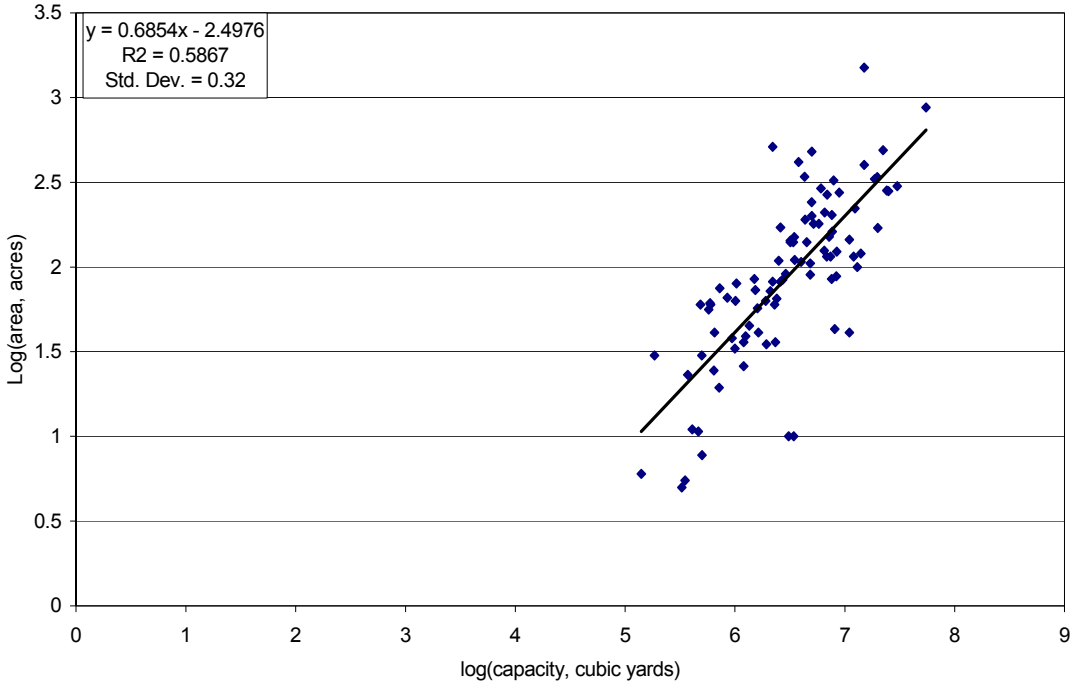


Figure B-6. Linear regression to impute surface impoundment area from capacity.

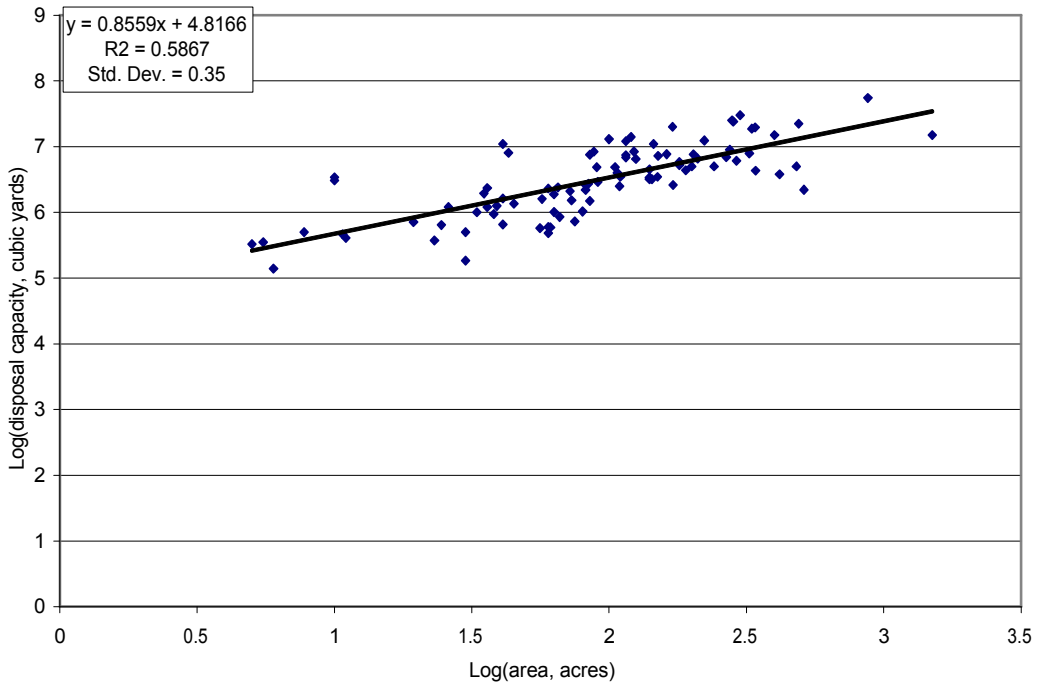


Figure B-7. Linear regression to impute surface impoundment capacity from area.

B.7 Results

Attachment B-1 lists the 181 CCW disposal sites modeled in this risk assessment and their locations. The WMU data used in the CCW risk assessment for each of the 108 landfills and 96 surface impoundments at these coal combustion facilities are presented in **Attachment B-2**. Missing data that were randomly replaced as described above are not represented in the table (i.e., the fields are left blank).

B.8 References

CIBO (Council of Industrial Boiler Owners). 1997. *Report to the U.S. Environmental Protection Agency on Fossil Fuel Combustion Byproducts from Fluidized Bed Boilers.* Prepared by the CIBO Special Project on Non-Utility Fossil Fuel Ash Classification and ICF Kaiser Consulting Group. November.

EPRI (Electric Power Research Institute). 1991. *Comanagement of Coal Combustion By-Products and Low-Volume Wastes: A Southeastern Site*. Final Report. EN-7545. EPRI, Palo Alto, CA. November 1991. ("L" Site).

EPRI (Electric Power Research Institute). 1992. *Comanagement of Coal Combustion By-Products and Low-Volume Wastes: A Midwestern Site*. Interim Report. TR-100955. EPRI, Palo Alto, CA. August 1992. ("C" Site).

EPRI (Electric Power Research Institute). 1994a. *Calcium-Based Flue Gas Desulfurization Sludge Disposal Sites.* Final Report. TR-103914. EPRI, Palo Alto, CA. April 1994.

EPRI (Electric Power Research Institute). 1994b. *Sodium-Based Flue Gas Desulfurization Sludge Disposal Sites.* Final Report. TR-103915. EPRI, Palo Alto, CA. April 1994.

EPRI (Electric Power Research Institute). 1996a. *Comanagement of Coal Combustion By-Products and Low-Volume Wastes: A Southwestern Site.* Final Report. TR-105673. EPRI, Palo Alto, CA. June 1996. ("O" Site).

EPRI (Electric Power Research Institute). 1996b. *Evaluation of the Co-Management of High Volume Combustion By-Products with Other Utility Wastes: CL Site*. Draft Report. EPRI, Palo Alto, CA. May 1996 (updated December).

EPRI (Electric Power Research Institute). 1997a. *Coal Combustion By-Products and Low-Volume Wastes Comanagement Survey*. EPRI, Palo Alto, CA. June.

EPRI (Electric Power Research Institute). 1997b. *Evaluation of the Co-Management of High Volume Combustion By-Products with Other Utility Wastes: AP Site*. Draft Report. EPRI, Palo Alto, CA. June 1997.

EPRI (Electric Power Research Institute). 1997c. *Evaluation of the Co-Management of High Volume Combustion By-Products with Other Utility Wastes: CY Site.* Final Report. TR-108426. EPRI, Palo Alto, CA. August 1997.

EPRI (Electric Power Research Institute). 1997d. *Evaluation of the Co-Management of High Volume Combustion By-Products with Other Utility Wastes: FC Site*. Draft Report. TR-108425. EPRI, Palo Alto, CA. August 1997.

EPRI (Electric Power Research Institute). 1997e. *Evaluation of the Co-Management of High Volume Combustion By-Products with Other Utility Wastes: HA Site*. Draft Report. EPRI, Palo Alto, CA. August 1997.

EPRI (Electric Power Research Institute). 1997f. *Evaluation of the Co-Management of High Volume Combustion By-Products with Other Utility Wastes: LS Site.* Final Report. TR-108422. EPRI, Palo Alto, CA. August 1997.

EPRI (Electric Power Research Institute). 1997g. *Evaluation of the Co-Management of High Volume Combustion By-Products with Other Utility Wastes: MO Site*. Draft Report. EPRI, Palo Alto, CA. April 1997.

EPRI (Electric Power Research Institute). 1997h. *Evaluation of the Co-Management of High Volume Combustion By-Products with Other Utility Wastes: OK Site*. Draft Report. TR-108421. EPRI, Palo Alto, CA. July 1997.

EPRI (Electric Power Research Institute). 1997i. *Evaluation of the Co-Management of High Volume Combustion By-Products with Other Utility Wastes: P4 Site.* Final Report. TR-108420. EPRI, Palo Alto, CA. July 1997.

EPRI (Electric Power Research Institute). 1997j. *Evaluation of the Co-Management of High Volume Combustion By-Products with Other Utility Wastes: PA Site.* Final Report. TR-108427. EPRI, Palo Alto, CA. August 1997.

EPRI (Electric Power Research Institute). 1997k. *Evaluation of the Co-Management of High Volume Combustion By-Products with Other Utility Wastes: SX Site*. Draft Report. TR-001015. EPRI, Palo Alto, CA. June 1997.

TetraTech. 2001. *Characterization of infiltration rate data to support groundwater modeling efforts (Draft)*. Prepared for the Office of Solid Waste, U.S. Environmental Protection Agency, Contract No. 68-W6-0061. May.

U.S. EPA (Environmental Protection Agency). 1999a. *Report to Congress: Wastes from the Combustion of Fossil Fuels*. EPA 530-5-99-010. Office of Solid Waste and Emergency Response, Washington, DC. March.

U.S. EPA (Environmental Protection Agency). 1999b. *Technical Background Document for the Supplemental Report to Congress on Remaining Fossil Fuel Combustion Wastes: Industry Statistics and Waste Management Practices*. Office of Solid Waste, Washington, DC. March 15, 1999. Docket number F-1999-FF2P-S0398.

U.S. EPA (Environmental Protection Agency). 2002. *Industrial Waste Management Evaluation Model (IWEM) Technical Background Document*. EPA 530-R-02-012. Office of Solid Waste, Washington, DC. August.

[This page intentionally left blank.]

Attachment B-1: CCW Disposal Sites (Plants)

Plant Name	Utility Name	County	State	Latitude	Longitude
A B Brown	Southern Indiana Gas & Elec. Co.	Posey	IN	37.9053	87.715
A/C Power - Ace Operations	A.C.E. Cogeneration Co.	San Bernardino	CA	35.75	117.3667
Allen	Tennessee Valley Authority	Shelby	TN	35.0742	90.1492
Alma	Dairyland Power Coop	Buffalo	WI	44.3078	91.905
Antelope Valley	Basin Electric Power Coop	Mercer	ND	47.37	101.8353
Arkwright	Georgia Power Co.	Bibb	GA	32.9269	83.6997
Asheville	Carolina Power & Light Co.	Buncombe	NC	35.4714	82.5431
Baldwin	Illinois Power Co.	Randolph	IL	38.205	89.8544
Barry	Alabama Power Co.	Mobile	AL	31.0069	88.0103
Bay Front	Northern States Power Co.	Ashland	WI	43.4833	89.4
Bay Shore	Toledo Edison Co.	Lucas	OH	41.6925	83.4375
Belews Creek	Duke Power Co.	Stokes	NC	36.2811	80.0603
Ben French	Black Hills Corp.	Pennington	SD	44.0872	103.2614
Big Cajun 2	Cajun Electric Power Coop, Inc.	Pointe Coupee	LA	30.7283	91.3686
Big Sandy	Kentucky Power Co.	Lawrence	KY	38.1686	82.6208
Big Stone	Otter Tail Power Co.	Grant	SD	45.3047	96.5083
Black Dog Steam Plant	Northern States Power Company	Dakota	MN	44.8167	93.25
Blue Valley	Independence, City of	Jackson	MO	39.0919	94.3364
Bowen	Georgia Power Co.	Bartow	GA	34.1256	84.9192
Brandon Shores	Baltimore Gas & Electric Co.	Anne Arundel	MD	39.18	76.5333
Buck	Duke Power Co.	Rowan	NC	35.7133	80.3767
Bull Run	Tennessee Valley Authority	Anderson	TN	36.0211	84.1567
C D McIntosh Jr.	Lakeland, City of	Polk	FL	28.075	81.9292
C P Crane	Baltimore Gas & Electric Co.	Baltimore City	MD	39.2845	76.6207
Cape Fear	Carolina Power & Light Co.	Chatham	NC	35.5989	79.0492
Carbon	PacifiCorp	Carbon	UT	39.7264	110.8639
Cardinal	Cardinal Operating Co.	Jefferson	OH	40.2522	80.6486
Cayuga	PSI Energy, Inc.	Vermillion	IN	39.9008	87.4136
Chalk Point	Potomac Electric Power Co.	Prince Georges	MD	38.5639	76.6806
Cholla	Arizona Public Service Co.	Navajo	AZ	34.9414	110.3003
Cliffside	Duke Power Co.	Cleveland	NC	35.22	81.7594
Clover	Virginia Electric & Power Co.	Halifax	VA	36.8667	78.7
Coal Creek	Coop Power Assn.	McLean	ND	47.3789	101.1572
Coleto Creek	Central Power & Light Co.	Goliad	TX	28.7128	97.2142

(continued)

CCW Disposal Sites (Plants) (continued)

Plant Name	Utility Name	County	State	Latitude	Longitude
Colstrip	Montana Power Co.	Rosebud	MT	45.8844	106.6139
Conemaugh	GPU Service Corporation	Indiana	PA	40.3842	79.0611
Conesville	Columbus Southern Power Co.	Coshocton	OH	40.1842	81.8811
Council Bluffs	MidAmerican Energy Co.	Pottawattamie	IA	41.18	95.8408
Crawford	Commonwealth Edison Co.	Cook	IL	39.8225	90.5681
Crist	Gulf Power Co.	Escambia	FL	30.5658	87.2239
Cross	South Carolina Pub Serv. Auth.	Berkeley	SC	33.3694	80.1119
Cumberland	Tennessee Valley Authority	Stewart	TN	36.3942	87.6539
Dale	East Kentucky Power Coop, Inc.	Clark	KY	37.875	84.25
Dallman	Springfield, City of	Sangamon	IL	39.7547	89.6008
Dan E Karn	Consumers Energy Co.	Bay	MI	43.645	83.8414
Dan River	Duke Power Co.	Rockingham	NC	36.4861	79.7244
Danskammer	Central Hudson Gas & Elec. Corp.	Orange	NY	41.5719	73.9664
Dave Johnston	PacifiCorp	Converse	WY	42.8333	105.7667
Dickerson	Potomac Electric Power Co.	Montgomery	MD	39.144	77.2059
Dolet Hills	CLECO Corporation	De Soto	LA	32.0308	93.5644
Duck Creek	Central Illinois Light Co.	Fulton	IL	40.4644	89.9825
Dunkirk	Niagara Mohawk Power Corp.	Chautauqua	NY	42.4919	79.3469
E D Edwards	Central Illinois Light Co.	Peoria	IL	40.5961	89.6633
E W Brown	Kentucky Utilities Co.	Mercer	KY	37.7911	84.7147
Eckert Station	Lansing, City of	Ingham	MI	42.7189	84.5583
Edgewater	Wisconsin Power & Light Co.	Sheboygan	WI	43.7181	87.7092
Elmer W Stout	Indianapolis Power & Light Co.	Marion	IN	39.7122	86.1975
F B Culley	Southern Indiana Gas & Elec. Co.	Warrick	IN	37.91	87.3267
Fayette Power Prj.	Lower Colorado River Authority	Fayette	TX	29.9172	96.7506
Flint Creek	Southwestern Electric Power Co.	Benton	AR	36.2625	94.5208
Fort Martin	Monongahela Power Co.	Monongalia	WV	39.7	79.9167
Frank E Ratts	Hoosier Energy R E C, Inc.	Pike	IN	38.5186	87.2725
G G Allen	Duke Power Co.	Gaston	NC	35.1897	81.0122
Gadsden	Alabama Power Co.	Etowah	AL	34.0136	85.9703
Gallatin	Tennessee Valley Authority	Sumner	TN	36.3156	86.4006
Gen J M Gavin	Ohio Power Co.	Gallia	OH	38.9358	82.1164
Genoa	Dairyland Power Coop	Vernon	WI	43.5592	91.2333
Gibson	PSI Energy, Inc.	Gibson	IN	38.3589	87.7783
Gorgas	Alabama Power Co.	Walker	AL	33.5111	87.235
Green River	Kentucky Utilities Co.	Muhlenberg	KY	37.3636	87.1214
Greene County	Alabama Power Co.	Greene	AL	32.6	87.7667
H B Robinson	Carolina Power & Light Co.	Darlington	SC	34.4	80.1667
Hammond	Georgia Power Co.	Floyd	GA	34.3333	85.2336

(continued)

CCW Disposal Sites (Plants) (continued)

Plant Name	Utility Name	County	State	Latitude	Longitude
Harllee Branch	Georgia Power Co.	Putnam	GA	33.1942	83.2994
Harrison	Monongahela Power Co.	Harrison	WV	39.3833	80.3167
Hatfield's Ferry	West Penn Power Co.	Greene	PA	39.85	79.9167
Hennepin	Illinois Power Co.	Putnam	IL	41.3028	89.315
Heskett	Montana-Dakota Utilities Co.	Morton	ND	46.8669	100.8839
Holcomb	Sunflower Electric Power Corp.	Finney	KS	37.9319	100.9719
Homer City	GPU Service Corporation	Indiana	PA	40.5142	79.1969
Hoot Lake	Otter Tail Power Co.	Otter Tail	MN	46.29	96.0428
Hugo	Western Farmers Elec. Coop, Inc.	Choctaw	OK	34.0292	95.3167
Hunter	PacifiCorp	Emery	UT	39.1667	111.0261
Huntington	PacifiCorp	Emery	UT	39.3792	111.075
Intermountain	Los Angeles, City of	Millard	UT	39.5108	112.5792
J H Campbell	Consumers Energy Co.	Ottawa	MI	42.9103	86.2031
J M Stuart	Dayton Power & Light Co.	Adams	OH	38.6364	83.7422
J R Whiting	Consumers Energy Co.	Monroe	MI	41.7914	83.4486
Jack McDonough	Georgia Power Co.	Cobb	GA	33.8244	84.475
Jack Watson	Mississippi Power Co.	Harrison	MS	30.4392	89.0264
James H Miller Jr.	Alabama Power Co.	Jefferson	AL	33.6319	87.0597
Jim Bridger	PacifiCorp	Sweetwater	WY	41.75	108.8
John E Amos	Appalachian Power Co.	Putnam	WV	38.4731	81.8233
John Sevier	Tennessee Valley Authority	Hawkins	TN	36.3767	82.9639
Johnsonville	Tennessee Valley Authority	Humphreys	TN	36.0278	87.9861
Joliet 29	Commonwealth Edison Co.	Will	IL	41.4892	88.0844
Keystone	GPU Service Corporation	Armstrong	PA	40.6522	79.3425
Killen Station	Dayton Power & Light Co.	Adams	OH	38.6903	83.4803
Kingston	Tennessee Valley Authority	Roane	TN	35.8992	84.5194
Kraft	Savannah Electric & Power Co	Chatham	GA	32.1333	81.1333
L V Sutton	Carolina Power & Light Co.	New Hanover	NC	34.2831	77.9867
Lansing	Interstate Power Co.	Allamakee	IA	43.3386	91.1667
Laramie R Station	Basin Electric Power Coop	Platte	WY	42.1086	104.8711
Lawrence EC	KPL Western Resources Co.	Douglas	KS	39.0078	95.2681
Lee	Carolina Power & Light Co.	Wayne	NC	35.3778	78.1
Leland Olds	Basin Electric Power Coop	Mercer	ND	47.2833	101.4
Lon Wright	Fremont, City of	Dodge	NE	41.45	96.5167
Louisa	MidAmerican Energy Co.	Louisa	IA	41.3181	91.0931
Marion	Southern Illinois Power Coop	Williamson	IL	37.6167	88.95
Marshall	Duke Power Co.	Catawba	NC	35.5975	80.9658
Martin Lake	Texas Utilities Electric Co.	Rusk	TX	32.2606	94.5708
Mayo	Carolina Power & Light Co.	Person	NC	36.5278	78.8919
Meramec	Union Electric Co.	St Louis	MO	38.6522	90.2397

(continued)

CCW Disposal Sites (Plants) (continued)

Plant Name	Utility Name	County	State	Latitude	Longitude
Merom	Hoosier Energy R E C, Inc.	Sullivan	IN	39.0694	87.5108
Miami Fort	Cincinnati Gas & Electric Co.	Hamilton	OH	39.1111	84.8042
Milton R Young	Minnkota Power Coop, Inc.	Oliver	ND	47.0664	101.2139
Mitchell - PA	West Penn Power Co.	Washington	PA	40.2167	79.9667
Mitchell - WV	Ohio Power Co.	Marshall	WV	39.8297	80.8153
Mohave	Southern California Edison Co.	Clark	NV	35.1667	114.6
Monroe	Detroit Edison Co.	Monroe	MI	41.8911	83.3444
Morgantown	Potomac Electric Power Co.	Charles	MD	38.3611	76.9861
Mountaineer (1301)	Appalachian Power Co.	Mason	WV	38.9794	81.9344
Mt Storm	Virginia Electric & Power Co.	Grant	WV	39.2014	79.2667
Muscatine Plant #1	Muscatine, City of	Muscatine	IA	41.3917	91.0569
Muskogee	Oklahoma Gas & Electric Co.	Muskogee	OK	35.7653	95.2883
Neal North	MidAmerican Energy Co.	Woodbury	IA	42.3167	96.3667
Neal South	MidAmerican Energy Co.	Woodbury	IA	42.3022	96.3622
Nebraska City	Omaha Public Power District	Otoe	NE	40.625	95.7917
New Castle	Pennsylvania Power Co.	Lawrence	PA	40.9383	80.3683
Newton	Central Illinois Pub Serv. Co.	Jasper	IL	38.9364	88.2778
North Omaha	Omaha Public Power District	Douglas	NE	41.33	95.9467
Northeastern	Public Service Co. of Oklahoma	Rogers	OK	36.4222	95.7047
Nucla	Tri-State G & T Assn., Inc.	Montrose	CO	38.2386	108.5072
Oklaunion	West Texas Utilities Co.	Wilbarger	TX	34.0825	99.1753
Paradise	Tennessee Valley Authority	Muhlenberg	KY	37.2608	86.9783
Petersburg	Indianapolis Power & Light Co.	Pike	IN	38.5267	87.2522
Pleasant Prairie	Wisconsin Electric Power Co.	Kenosha	WI	42.5381	87.9033
Port Washington	Wisconsin Electric Power Co.	Ozaukee	WI	43.3908	87.8686
Portland	Metropolitan Edison Co.	Northampton	PA	40.7525	75.3324
Possum Point	Virginia Electric & Power Co.	Prince William	VA	38.5367	77.2806
Potomac River	Potomac Electric Power Co.	Alexandria	VA	38.8078	77.0372
Presque Isle	Wisconsin Electric Power Co.	Marquette	MI	46.5694	87.3933
R Gallagher	PSI Energy, Inc.	Floyd	IN	38.2631	85.8378
R M Schahfer	Northern Indiana Pub. Serv. Co.	Jasper	IN	41.2167	87.0222
Reid Gardner	Nevada Power Co.	Clark	NV	36.6606	114.625
Richard Gorsuch	American Mun. Power-Ohio, Inc.	Washington	OH	39.3672	81.5208
Riverbend	Duke Power Co.	Gaston	NC	35.36	80.9742
Rodemacher	CLECO Corporation	Rapides	LA	31.395	92.7167
Roxboro	Carolina Power & Light Co.	Person	NC	36.4831	79.0711
Sandow	Texas Utilities Electric Co.	Milam	TX	30.5642	97.0639
Scherer	Georgia Power Co.	Monroe	GA	33.0583	83.8072
Shawnee	Tennessee Valley Authority	McCracken	KY	37.1517	88.775
Shawville	GPU Service Corporation	Clearfield	PA	41.0681	78.3661

(continued)

CCW Disposal Sites (Plants) (continued)

Plant Name	Utility Name	County	State	Latitude	Longitude
Sheldon	Nebraska Public Power District	Lancaster	NE	40.5589	96.7842
South Oak Creek	Wisconsin Electric Power Co.	Milwaukee	WI	42.8014	87.8314
Springerville	Tucson Electric Power Co	Apache	AZ	34.3186	109.1636
St Johns River Power	JEA	Duval	FL	30.4308	81.5508
Stanton Energy Ctr.	Orlando Utilities Comm.	Orange	FL	28.4822	81.1678
Stockton Cogen Company	Stockton Cogen Co (operator: Air Products)	San Joaquin	CA	37.9778	121.2667
Syl Laskin	Minnesota Power, Inc.	St Louis	MN	47.53	92.1617
Tecumseh EC	KPL Western Resources Co.	Shawnee	KS	39.0528	95.5683
Texas-New Mexico	Texas-New Mexico Power Company/Sempra Energy	Robertson	TX	31.0928	96.6933
Titus	Metropolitan Edison Co.	Berks	PA	40.3047	75.9072
Trimble County	Louisville Gas & Electric Co.	Trimble	KY	38.5678	85.4139
Tyrone	Kentucky Utilities Co.	Woodford	KY	38.0213	84.7456
Valley	Wisconsin Electric Power Co.	Milwaukee	WI	43.0303	87.925
Vermilion	Illinois Power Co.	Vermilion	IL	40.1781	87.7481
Victor J Daniel Jr.	Mississippi Power Co.	Jackson	MS	30.5322	88.5569
W A Parish	Houston Lighting & Power Co.	Fort Bend	TX	29.4833	95.6331
W H Weatherspoon	Carolina Power & Light Co.	Robeson	NC	34.5889	78.975
W S Lee	Duke Power Co.	Anderson	SC	34.6022	82.435
Wabash River	PSI Energy, Inc.	Vigo	IN	39.5278	87.4222
Walter C Beckjord	Cincinnati Gas & Electric Co.	Clermont	OH	38.9917	84.2972
Wansley	Georgia Power Co.	Heard	GA	33.4167	85.0333
Warrick	Southern Indiana Gas & Elec. Co.	Warrick	IN	37.915	87.3319
Waukegan	Commonwealth Edison Co.	Lake	IL	42.3833	87.8083
Weston	Wisconsin Public Service Corp.	Marathon	WI	44.8617	89.655
Widows Creek	Tennessee Valley Authority	Jackson	AL	34.8825	85.7547
Will County	Commonwealth Edison Co.	Will	IL	38.8639	90.1347
Wyodak	PacifiCorp	Campbell	WY	44.2833	105.4
Yates	Georgia Power Co.	Coweta	GA	33.4631	84.955

[This page intentionally left blank.]

Attachment B-2: CCW WMU Data

Plant	Facility ID	WMU Type	Area (acres)	Capacity (cubic yards)	Waste Type	Original Liner	Liner Type
A B Brown	42	LF	176	10360000	Ash	compacted clay	clay
A/C Power - Ace Operations	3000	LF	18	1030815	FBC	none/natural soils	no liner
Allen	293	SI	85	1500000	Ash	none/natural soils	no liner
Alma	7	LF	85	2000000	Ash and Coal Waste	composite clay/membrane	composite
Antelope Valley	57	LF	27	3500000	Ash	none/natural soils	no liner
Arkwright	198	LF	54	415907	Ash and Coal Waste	none/natural soils	no liner
Asheville	159	SI	140	3200000	Ash	none/natural soils	no liner
Baldwin	2	SI	107	4000000	Ash and Coal Waste	none/natural soils	no liner
Barry	301	SI	63	1900000	Ash and Coal Waste	none/natural soils	no liner
Bay Front	81	LF	10	350000	Ash	none/natural soils	no liner
Bay Shore	32	LF	85		Ash	none/natural soils	no liner
Belews Creek	167	SI	512	2200000	Ash and Coal Waste	none/natural soils	no liner
Belews Creek	168	LF	315	14000000	Ash	compacted ash	no liner
Ben French	14	LF	4.61		Ash	compacted clay	clay
Big Cajun 2	186	SI	241	4990003	Ash	compacted clay	clay
Big Sandy	138	SI	115	12052100	Ash and Coal Waste	none/natural soils	no liner
Big Stone	15	LF	3.4	80000	Ash	compacted clay	clay
Big Stone	41	LF	106	8000000	Ash	none/natural soils	no liner
Black Dog Steam Plant	2700	LF	96	8936296	FBC	compacted clay	clay
Blue Valley	176	SI	23.1	372000	Ash and Coal Waste	compacted clay	clay
Bowen	143	LF	25.24	491400	Ash	compacted ash	no liner
Bowen	144	LF	25.77	406971	Ash	compacted ash	no liner
Brandon Shores	339	LF	246	5600000	Ash and Coal Waste	none/natural soils	no liner

(continued)

CCW WMU Data (continued)

Plant	Facility ID	WMU Type	Area (acres)	Capacity (cubic yards)	Waste Type	Original Liner	Liner Type
Buck	235	SI	90	4840000	Ash and Coal Waste	none/natural soils	no liner
Bull Run	296	SI	41	650000	Ash and Coal Waste	none/natural soils	no liner
C D McIntosh Jr.	223	LF	26		Ash and Coal Waste	compacted ash	no liner
C P Crane	338	LF	35	800000	Ash	none/natural soils	no liner
Cape Fear	161	SI	60	2300000	Ash	none/natural soils	no liner
Carbon	263	lf	11.7739066		Ash and Coal Waste	none/natural soils	no liner
Cardinal	126	SI	123	8437500	Ash	none/natural soils	no liner
Cayuga	325	SI	280	25000000	Ash and Coal Waste	none/natural soils	no liner
Chalk Point	292	LF	596	4634000	Ash and Coal Waste	none/natural soils	no liner
Cholla	107	SI	171	2600000	Ash	none/natural soils	no liner
Cliffside	163	SI	82	2200000	Ash	compacted clay	clay
Clover	139	LF	22	1000000	Ash	geosynthetic membrane	composite
Coal Creek	29	LF	70	4700000	Ash	compacted clay	clay
Coal Creek	30	LF	220	23000000	Ash	composite clay/membrane	composite
Coleto Creek	190	si	314.6135409		Ash and Coal Waste	compacted clay	clay
Colstrip	89	LF	9		Ash	none/natural soils	no liner
Conemaugh	101	LF	434	82000000	Ash	geosynthetic membrane	composite
Conesville	250	LF	300	10000000	Ash	compacted clay	clay
Conesville	251	LF	100	2500000	Ash and Coal Waste	none/natural soils	no liner
Council Bluffs	94	SI	200		Ash	none/natural soils	no liner
Crawford	272	SI	24.5	642000	Ash and Coal Waste	compacted clay	clay
Crist	157	LF	12		Ash and Coal Waste	none/natural soils	no liner
Cross	264	LF	320		Ash	compacted ash	no liner
Cross	265	LF	30		Ash and Coal Waste	none/natural soils	no liner
Cross	266	LF	30		Ash and Coal Waste	none/natural soils	no liner
Cross	267	LF	230		Ash and Coal Waste	none/natural soils	no liner
Cross	268	LF	60		Ash and Coal Waste	compacted clay	clay

(continued)

CCW WMU Data (continued)

Plant	Facility ID	WMU Type	Area (acres)	Capacity (cubic yards)	Waste Type	Original Liner	Liner Type
Cumberland	294	SI	75	1750000	Ash and Coal Waste	none/natural soils	no liner
Cumberland	303	SI	295	9500000	Ash	none/natural soils	no liner
Dale	151	SI	115	7408274	Ash and Coal Waste	none/natural soils	no liner
Dallman	178	LF	22	1800000	Ash	compacted clay	clay
Dallman	179	SI	417	3800000	Ash	none/natural soils	no liner
Dan E Karn	6	LF	40	1650000	Ash and Coal Waste	geosynthetic membrane	composite
Dan River	234	SI	72	2097000	Ash and Coal Waste	none/natural soils	no liner
Danskammer	24	LF	14	517265	Ash and Coal Waste	geosynthetic membrane	composite
Dave Johnston	13	LF	45	296100	Ash	compacted clay	clay
Dickerson	290	LF	206	12600000	Ash	none/natural soils	no liner
Dolet Hills	245	SI	66	850000	Ash and Coal Waste	none/natural soils	no liner
Dolet Hills	246	LF	109	8500000	Ash	compacted clay	clay
Duck Creek	11	LF	21.3	1500000	Ash	compacted clay	clay
Dunkirk	49	LF	12	1126080	Ash	compacted clay	clay
E D Edwards	276	SI	145	11000000	Ash and Coal Waste	none/natural soils	no liner
E W Brown	313	SI	33	1000000	Ash	none/natural soils	no liner
E W Brown	314	SI	84	2710000	Ash	none/natural soils	no liner
Eckert Station	113	LF	174	6460000	Ash	none/natural soils	no liner
Eckert Station	114	SI	151	7200000	Ash	none/natural soils	no liner
Edgewater	289	LF	25	1655700	Ash and Coal Waste	none/natural soils	no liner
Elmer W Stout	130	SI	10	3420000	Ash	geosynthetic membrane	composite
F B Culley	183	SI	82	2600000	Ash and Coal Waste	none/natural soils	no liner
Fayette Power Prj.	195	SI	190	4351644	Ash	compacted clay	clay
Fayette Power Prj.	196	LF	23	890560	Ash	geosynthetic membrane	composite
Flint Creek	191	LF	40	1508250	Ash and Coal Waste	none/natural soils	no liner
Flint Creek	192	si	35.73857178		Ash and Coal Waste	none/natural soils	no liner
Fort Martin	213	LF	17	1900000	Ash	none/natural soils	no liner

(continued)

CCW WMU Data (continued)

Plant	Facility ID	WMU Type	Area (acres)	Capacity (cubic yards)	Waste Type	Original Liner	Liner Type
Fort Martin	214	LF	61	1400000	Ash	double	composite
Fort Martin	215	LF	121	3700000	Ash and Coal Waste	composite clay/membrane	composite
Frank E Ratts	182	SI	39	1250000	Ash and Coal Waste	none/natural soils	no liner
G G Allen	237	SI	210	6545000	Ash and Coal Waste	none/natural soils	no liner
Gadsden	283	SI	60	484000	Ash and Coal Waste	compacted clay	clay
Gallatin	304	SI	341	4300000	Ash and Coal Waste	none/natural soils	no liner
Gen J M Gavin	135	LF	255	50000000	Ash	composite clay/membrane	composite
Gen J M Gavin	136	SI	300	30000000	Ash and Coal Waste	none/natural soils	no liner
Gen J M Gavin	137	LF	99	12000000	Ash	compacted clay	clay
Genoa	244	LF	100		Ash and Coal Waste	none/natural soils	no liner
Gibson	327	SI	875	55000000	Ash and Coal Waste	none/natural soils	no liner
Gibson	329	LF	85	20000000	Ash	compacted clay	clay
Gorgas	280	SI	250		Ash and Coal Waste	compacted clay	clay
Gorgas	281	SI	283	24100000	Ash and Coal Waste	compacted clay	clay
Gorgas	282	SI	1500	15000000	Ash and Coal Waste	compacted clay	clay
Green River	147	SI	36	2331219	Ash and Coal Waste	none/natural soils	no liner
Greene County	279	SI	480	5000000	Ash	compacted clay	clay
H B Robinson	169	SI	30		Ash and Coal Waste	none/natural soils	no liner
Hammond	203	SI	56	576256	Ash and Coal Waste	none/natural soils	no liner
Harllee Branch	204	SI	324	7898277	Ash and Coal Waste	none/natural soils	no liner
Harllee Branch	205	SI	203	7634000	Ash and Coal Waste	none/natural soils	no liner
Harrison	211	LF	79	18000000	Ash and Coal Waste	composite clay/membrane	composite
Harrison	330	SI	300	28000000	Ash	none/natural soils	no liner
Hatfield's Ferry	112	LF	20	790000	Ash and Coal Waste	compacted ash	no liner
Hennepin	274	SI	150	3460600	Ash and Coal Waste	none/natural soils	no liner
Heskett	87	LF	58	1550000	FBC	compacted clay	clay
Holcomb	65	LF	8		Ash	compacted ash	no liner

(continued)

CCW WMU Data (continued)

Plant	Facility ID	WMU Type	Area (acres)	Capacity (cubic yards)	Waste Type	Original Liner	Liner Type
Homer City	118	LF	247	29636550	Ash and Coal Waste	geosynthetic membrane	composite
Hoot Lake	40	LF	72	800000	Ash and Coal Waste	none/natural soils	no liner
Hugo	193	LF	40	4000000	Ash	compacted ash	no liner
Hugo	194	si	151.0232271		Ash and Coal Waste	compacted clay	clay
Hunter	256	LF	280	12000000	Ash	none/natural soils	no liner
Huntington	255	LF	70	11400000	Ash	none/natural soils	no liner
Intermountain	224	SI	105	4840000	Ash and Coal Waste	geosynthetic membrane	composite
Intermountain	225	LF	339	17800000	Ash	compacted ash	no liner
Intermountain	226	SI	180	5200000	Ash	geosynthetic membrane	composite
J H Campbell	115	SI	267	6900000	Ash and Coal Waste	none/natural soils	no liner
J M Stuart	125	SI	88	8357000	Ash	none/natural soils	no liner
J R Whiting	129	SI	6	140000	Ash	none/natural soils	no liner
Jack McDonough	202	SI	73	1531893	Ash and Coal Waste	none/natural soils	no liner
Jack Watson	220	SI	100		Ash	none/natural soils	no liner
James H Miller Jr.	300	SI	200	5500000	Ash	compacted clay	clay
Jim Bridger	257	LF	120	7940941	Ash	none/natural soils	no liner
Jim Bridger	258	LF	241	24000000	Ash and Coal Waste	none/natural soils	no liner
Jim Bridger	259	SI	140	3400000	Ash and Coal Waste	none/natural soils	no liner
Jim Bridger	262	SI	125	6500000	Ash and Coal Waste	none/natural soils	no liner
John E Amos	120	SI	100	13000000	Ash	none/natural soils	no liner
John E Amos	121	LF	200	14000000	Ash and Coal Waste	compacted clay	clay
John E Amos	122	SI	10	3078000	Ash	none/natural soils	no liner
John Sevier	297	SI	57	1600000	Ash and Coal Waste	none/natural soils	no liner
John Sevier	298	LF	51	4800000	Ash	compacted clay	clay
John Sevier	309	SI	105	7000000	Ash and Coal Waste	none/natural soils	no liner
Johnsonville	306	SI	91	2900000	Ash and Coal Waste	none/natural soils	no liner
Joliet 29	275	SI	63.1	1012000	Ash and Coal Waste	none/natural soils	no liner

(continued)

CCW WMU Data (continued)

Plant	Facility ID	WMU Type	Area (acres)	Capacity (cubic yards)	Waste Type	Original Liner	Liner Type
Keystone	106	LF	155	22663120	Ash and Coal Waste	none/natural soils	no liner
Killen Station	254	SI		99935	Ash and Coal Waste	compacted clay	clay
Kingston	311	SI	41	11000000	Ash and Coal Waste	none/natural soils	no liner
Kingston	312	SI	275	8900000	Ash and Coal Waste	none/natural soils	no liner
Kraft	206	si	59.87027428		Ash and Coal Waste	none/natural soils	no liner
L V Sutton	231	SI	162	7696000	Ash and Coal Waste	none/natural soils	no liner
Lansing	64	SI	15		Ash	compacted clay	clay
Laramie R Station	260	SI	10.7	464156	Ash and Coal Waste	compacted clay	clay
Laramie R Station	261	SI	38	939605	Ash	geosynthetic membrane	composite
Lawrence EC	109	LF	825	34300000	Ash	compacted clay	clay
Lawrence EC	110	LF	22	1360000	Ash	compacted clay	clay
Lawrence EC	111	LF	30	1000000	Ash	compacted clay	clay
Lee	240	SI	35	1936000	Ash and Coal Waste	none/natural soils	no liner
Leland Olds	103	LF	37	1800000	Ash	compacted clay	clay
Leland Olds	104	LF	20	458000	Ash and Coal Waste	none/natural soils	no liner
Lon Wright	98	LF		170000	Ash	none/natural soils	no liner
Louisa	63	SI	30	500000	Ash	compacted clay	clay
Marion	52	LF	105	2200000	Ash	none/natural soils	no liner
Marion	53	LF	38	1000000	Ash	compacted clay	clay
Marshall	232	LF	110	7826000	Ash	none/natural soils	no liner
Marshall	233	SI	340	19689000	Ash and Coal Waste	none/natural soils	no liner
Martin Lake	152	LF	290	30000000	Ash	compacted clay	clay
Mayo	171	SI	30	185000	Ash	none/natural soils	no liner
Mayo	172	SI	65	2400000	Ash	none/natural soils	no liner
Meramec	175	SI	61.1	591200	Ash and Coal Waste	none/natural soils	no liner
Merom	184	LF	65	8500000	Ash	none/natural soils	no liner
Miami Fort	39	LF	80	4000000	Ash	compacted clay	clay

(continued)

CCW WMU Data (continued)

Plant	Facility ID	WMU Type	Area (acres)	Capacity (cubic yards)	Waste Type	Original Liner	Liner Type
Milton R Young	100	LF	80	6500000	Ash	compacted clay	clay
Mitchell - PA	208	LF	70	5600000	Ash	none/natural soils	no liner
Mitchell - WV	131	SI		12030000	Ash and Coal Waste	none/natural soils	no liner
Mohave	72	LF	250	21500000	Ash	none/natural soils	no liner
Monroe	26	LF	400	20000000	Ash and Coal Waste	none/natural soils	no liner
Monroe	27	SI	400	15000000	Ash	none/natural soils	no liner
Morgantown	291	LF	212	7700000	Ash and Coal Waste	none/natural soils	no liner
Mountaineer (1301)	212	LF	60	9700000	Ash	composite clay/membrane	composite
Mt Storm	73	LF	125	18920000	Ash	composite clay/membrane	composite
Mt Storm	134	LF	900	8800000	Ash and Coal Waste	compacted clay	clay
Muscatine Plant #1	70	LF	36	2000000	Ash	compacted clay	clay
Muskogee	51	LF	36	1247112	Ash	compacted clay	clay
Neal North	92	SI	150		Ash and Coal Waste	none/natural soils	no liner
Neal North	93	LF	200		Ash	none/natural soils	no liner
Neal South	284	LF	150		Ash	none/natural soils	no liner
Nebraska City	20	LF	17	600000	Ash and Coal Waste	compacted clay	clay
New Castle	66	LF	27	1100000	Ash and Coal Waste	geosynthetic membrane	composite
Newton	180	LF	309		Ash	none/natural soils	no liner
North Omaha	17	LF	13	105000	Ash and Coal Waste	compacted clay	clay
Northeastern	142	LF	69	3185190	Ash	none/natural soils	no liner
Nucla	96	LF	41.2	1500000	FBC	none/natural soils	no liner
Oklaunion	228	SI	11	408940	Ash and Coal Waste	none/natural soils	no liner
Oklaunion	229	SI	19.4	718060	Ash	none/natural soils	no liner
Oklaunion	230	SI	290.8	6056820	Ash	none/natural soils	no liner
Paradise	146	SI	85	7582510	Ash	composite clay/membrane	composite
Paradise	316	SI	200	5000000	Ash	none/natural soils	no liner
Petersburg	155	LF	250	19750000	Ash	compacted clay	clay

(continued)

CCW WMU Data (continued)

Plant	Facility ID	WMU Type	Area (acres)	Capacity (cubic yards)	Waste Type	Original Liner	Liner Type
Petersburg	156	si	156.6901408		Ash	none/natural soils	no liner
Pleasant Prairie	243	LF	26	6500000	Ash and Coal Waste	geosynthetic membrane	composite
Port Washington	242	LF	300	1900000	Ash and Coal Waste	compacted clay	clay
Portland	67	LF	15	2200000	Ash and Coal Waste	none/natural soils	no liner
Possum Point	77	SI	56		Ash and Coal Waste	none/natural soils	no liner
Potomac River	140	LF	33	802000	Ash	geosynthetic membrane	composite
Presque Isle	116	LF	292	14200000	Ash	none/natural soils	no liner
R Gallagher	326	SI	170	20000000	Ash and Coal Waste	compacted clay	clay
R M Schahfer	84	SI	80	1030000	Ash and Coal Waste	none/natural soils	no liner
R M Schahfer	85	LF	200	17200000	Ash	none/natural soils	no liner
Reid Gardner	95	LF	112.5	4520000	Ash	none/natural soils	no liner
Richard Gorsuch	36	LF		3003600	Ash	compacted clay	clay
Riverbend	165	SI	143	3200000	Ash	none/natural soils	no liner
Rodemacher	247	SI	36	1200000	Ash	compacted clay	clay
Rodemacher	248	SI	109	2500000	Ash	compacted clay	clay
Roxboro	239	LF	55	4165000	Ash	none/natural soils	no liner
Sandow	153	LF	125	1300000	Ash	compacted clay	clay
Sandow	187	LF	48	903467	Ash and Coal Waste	none/natural soils	no liner
Sandow	188	SI	45	1351973	Ash and Coal Waste	none/natural soils	no liner
Scherer	199	SI	490	22262030	Ash and Coal Waste	none/natural soils	no liner
Shawnee	317	SI	180	5810000	Ash and Coal Waste	none/natural soils	no liner
Shawnee	318	LF	96	6100000	FBC	none/natural soils	no liner
Shawville	209	LF	68	8000000	Ash	none/natural soils	no liner
Sheldon	23	LF	9	375000	Ash	compacted clay	clay
South Oak Creek	3	LF	45	4050000	Ash and Coal Waste	compacted clay	clay
South Oak Creek	4	LF	130	4600000	Ash	none/natural soils	no liner
Springerville	154	LF	57	6400000	Ash	none/natural soils	no liner

(continued)

CCW WMU Data (continued)

Plant	Facility ID	WMU Type	Area (acres)	Capacity (cubic yards)	Waste Type	Original Liner	Liner Type
St Johns River Power	158	lf	128.624166		Ash and Coal Waste	compacted clay	clay
Stanton Energy Ctr.	117	LF	312		Ash	none/natural soils	no liner
Stockton Cogen Company	2000	LF	4	533333	FBC	composite clay/membrane	composite
Syl Laskin	68	SI	75	726000	Ash and Coal Waste	none/natural soils	no liner
Tecumseh EC	177	LF	540		Ash	compacted clay	clay
Texas-New Mexico	3900	LF	61	6142473	FBC	compacted clay	clay
Titus	207	LF	39	3000000	Ash and Coal Waste	composite clay/membrane	composite
Trimble County	69	SI	115	6856667	Ash	compacted clay	clay
Tyrone	148	SI	5.5	351699	Ash	none/natural soils	no liner
Tyrone	149	SI	5	327500	Ash and Coal Waste	none/natural soils	no liner
Tyrone	150	SI	7.75	500123	Ash and Coal Waste	none/natural soils	no liner
Valley	8	LF	16.4	534000	Ash and Coal Waste	compacted clay	clay
Vermilion	55	SI	43	8100000	Ash and Coal Waste	none/natural soils	no liner
Victor J Daniel Jr	287	lf	49.20163084		Ash	compacted clay	clay
Victor J Daniel Jr	288	si	20.03879417		Ash and Coal Waste	composite clay/membrane	composite
W A Parish	189	lf	28.68322214		Ash	compacted clay	clay
W H Weatherspoon	236	SI	26	1200000	Ash and Coal Waste	none/natural soils	no liner
W S Lee	238	SI	41	1634000	Ash and Coal Waste	none/natural soils	no liner
Wabash River	324	SI	120	14000000	Ash and Coal Waste	none/natural soils	no liner
Walter C Beckjord	123	LF	14	1000000	Ash	compacted ash	no liner
Walter C Beckjord	124	SI		2000000	Ash	none/natural soils	no liner
Wansley	200	SI	330	18712850	Ash and Coal Waste	none/natural soils	no liner
Wansley	201	SI	43		Ash	none/natural soils	no liner
Warrick	181	SI	140	4500000	Ash and Coal Waste	compacted clay	clay
Waukegan	54	LF	60	4000000	Ash and Coal Waste	compacted clay	clay
Weston	241	LF	18	600000	Ash	none/natural soils	no liner
Widows Creek	320	SI	110	3500000	Ash and Coal Waste	none/natural soils	no liner

(continued)

CCW WMU Data (continued)

Plant	Facility ID	WMU Type	Area (acres)	Capacity (cubic yards)	Waste Type	Original Liner	Liner Type
Widows Creek	321	SI	222	12400000	Ash	compacted clay	clay
Will County	277	SI	60	599256	Ash and Coal Waste	compacted clay	clay
Wyodak	71	LF	68	3500000	Ash	geosynthetic membrane	composite
Yates	197	SI	4.7	115000	Ash	composite clay/membrane	composite

Appendix C. Site Data

The site characteristics used in this analysis were based on site-specific, regional, and national data sources to provide the environmental parameters necessary for modeling the fate and transport of coal combustion waste (CCW) constituents released in landfill or surface impoundment leachate. Site-specific data were collected for the area in the immediate vicinity of the waste management unit (WMU), and included the geographic relationship among important features such as the WMU boundary, residential well location, and streams and lakes. These data were collected at each of the 181 coal-fired power plants selected for the analysis. These 181 locations across the continental United States were intended to represent the geographic distribution of onsite WMUs used for disposal of CCW and were used to capture national variability in meteorology, soils, climate, aquifers, and surface waterbodies at the disposal sites.

C.1 Data Collection Methodology

The CCW risk assessment employed site-specific, regional, and national data. Site-specific data were collected around CCW plant locations from the Energy Information Administration (EIA) database to obtain data for each facility that were representative of the environment immediately surrounding the plant. When site-specific data were not available, regional or national scale data sources were used. Where appropriate, distributions were used in the Monte Carlo analysis to capture site-to-site, within-site, and national variability in the parameters collected.

Data were collected around each CCW site using a geographic information system (GIS) that allowed (1) site-specific data to be assembled from the area immediately surrounding the facility and (2) the site to be assigned to a region to collect regional data. To account for locational uncertainty for the CCW WMUs[1], a 5-km radius was used to define the data collection area for aquifer type and soil data. If multiple soil or aquifer types occurred within this radius, multiple types were sent to the model, weighted by the fraction of the collection area that they occupied. Surface waterbody type and stream flows also were collected for each site by identifying the nearest stream segment.

Climate and water quality data were collected by assigning each site to a Hydrologic Evaluation of Landfill Performance (HELP) model climate station and a U.S. Geological Survey (USGS) hydrologic region. The EPA STOrage and RETrieval (STORET) database was used as the source for water quality data, with parameters selected from distributions queried from this database for each region.

Because the EIA locations were not exact for the WMUs being modeled, a national distribution of stream distances was developed by manually measuring the distance between the

[1] The EIA latitudes and longitudes usually represent a facility centroid or front-gate location for each power plant. Because these facilities are often large, the WMUs are frequently located some distance from the plant itself and not at the EIA location.

WMU and the waterbody at a random sample of the CCW sites. Similarly, a national distribution was used to represent the distance of the nearest residential wells from the CCW WMUs being modeled.

C.2 Receptor Location (National Data)

The residential scenario for the CCW groundwater pathway analysis calculates exposure through use of well water as drinking water. During the Monte Carlo analysis, the receptor well was placed at a distance of up to 1 mile from the edge of the WMU, by sampling a nationwide distribution of nearest downgradient residential well distances taken from a survey of municipal solid waste landfills (U.S. EPA, 1988).

EPA believes that this MSW well-distance distribution (presented in **Table C-1**) is protective for onsite CCW landfills and surface impoundments at coal-fired utility power plants, but recognizes that this is an uncertainty in this analysis. Because CCW plants tend to be in more isolated areas than MSW landfills and because CCW WMUs tend to be larger than municipal landfills, EPA believes that the MSW well distance distribution is a protective representation of actual well distances at CCW disposal sites. As discussed in **Section 3.4.3**, the groundwater model used in the CCW risk assessment placed limits on the lateral direction from the plume centerline (i.e., angle off plume centerline) and depth below the water table to ensure that the well remained within the plume and at a depth appropriate for surficial aquifers across the United States. These limits were consistent with other recent national risk assessments conducted by EPA OSW and provided a protective approach to siting wells for this analysis.

Table C-1. Distribution of Receptor Well Distance

Percentile	x-distance (m)
Minimum	0.6
10	104
20	183
30	305
40	366
50 (Median)	427
60	610
70	805
80	914
90	1,220
Maximum	1,610

Source: U.S. EPA (1988).

C.2.1 Recreational Fisher and Ecological Risk Scenario (Distance to Waterbody)

The recreational fisher scenario was used to estimate risks to recreational fishers and their children who live in the vicinity of the CCW landfills and surface impoundments and catch and consume fish from a waterbody located adjacent to the buffer. The waterbody was assumed to be a stream or lake located downwind of the WMU, beginning where the buffer area ends (see Figure 2-4), and was also used as the reasonable worst case aquatic system for the ecological risk

assessment. Waterbody characteristics were determined based on site-specific, regional, or national data (as described in **Section C.6**), except for stream length, which was determined by the width of the plume as it intersects the waterbody.

The downgradient distance to the surface water body was determined from a national distribution developed by measuring this distance at 59 CCW landfill and surface impoundment sites randomly selected from the 204 WMUs modeled in this risk assessment. **Table C-2** presents this distribution. **Figure C-1** provides a map and aerial photo of one of the facilities used to develop this distribution. The development of this distribution is described in **Section C.6.4**.

Table C-2. Distribution of Surface Water Distances

Percentile	Distance (m)
Minimum	10
0.03	10
0.05	20
0.07	20
0.09	20
0.10	20
0.13	20
0.15	30
0.20	40
0.25	50
0.30	50
0.35	60
0.40	70
0.45	100
0.50 (Median)	120
0.55	130
0.60	150
0.65	250
0.70	400
0.75	440
0.80	500
0.85	700
0.87	775
0.90	800
0.91	1,000
0.93	1,500
0.95	2,125
0.97	2,750
Maximum	3,000

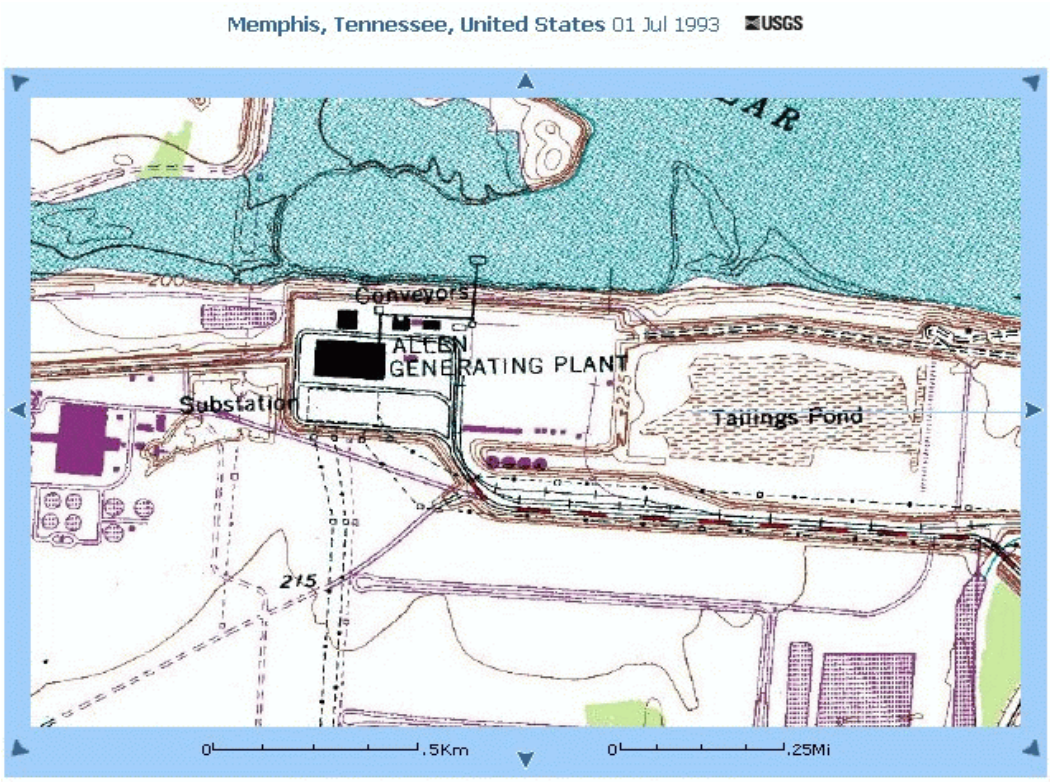


Image courtesy of the U.S. Geological Survey

Image courtesy of the U.S. Geological Survey

Figure C-1. Example CCW site used to develop waterbody distance distribution.

C.3 Soil Data

The groundwater model used in the CCW risk assessment—EPA's Composite Model for Leachate Migration with Transformation Products (EPACMTP)—requires soil properties for the entire soil column to model leachate transport through the vadose zone to groundwater. As with aquifer type, soil data were collected within a 5-km radius of each CCW plant. A GIS was used to identify soil map units within a 20-mile radius around each meteorological station. Database programs were then used to assemble and process soil texture, pH, and soil organic matter data for these map units from the State Soil Geographic (STATSGO) database. Both pH and soil organic matter were processed and indexed by the soil textures present within the 5-km radius. Soil properties are listed by texture for each of the 181 CCW plants in **Attachment C-1**.

C.3.1 Data Sources

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

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

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

- **CONUS.** The Conterminous United States Multi-Layer Soil Characteristics (CONUS) data set (Miller and White, 1998) provides average STATSGO data by map unit and a set of 11 standardized soil layers.

Soil organic matter and pH were derived directly from USSOILS and STATSGO data. A complete set of hydrological soil properties[2] was not available from STATSGO. To ensure consistent and realistic values, EPACMTP relies on established, nationwide relationships between hydrologic properties and soil texture. Peer-reviewed publications by Carsel and Parrish (1988) and Carsel et al. (1988) provide a consistent set of correlated hydrologic properties for each soil texture. Soil texture data for the entire soil column were collected from the CONUS database.

C.3.2 Methodology

The soil data collection methodology began with GIS programs (in Arc Macro Language [AML]). These programs overlaid a 5-km radius around each CCW plant location on the STATSGO map unit coverage to determine the STATSGO map units and their area within the radius. These data were then passed to data processing programs that derive soil properties for

[2] Hydrological soil properties required by EPACMTP include bulk density, saturated water content, saturated hydraulic conductivity, and the van Genuchten soil moisture retention parameters alpha and beta.

each site, either through direct calculations or by applying established relationships in lookup tables.

EPACMTP utilizes three soil textures to represent variability in hydrologic soil properties and (along with climate data) to assign infiltration rates to each site. Because STATSGO soils are classified into the 12 U.S. Soil Conservation Service (SCS) soil textures, the crosswalk shown in **Table C-3** was used to assign the SCS textures to the EPACMTP megatextures and to calculate the percentage of each megatexture within the 5-km data collection radius. These percentages were sampled for each site when preparing the source data file for each site.

Both soil pH and soil organic matter were derived for each EPACMTP soil megatexture at a site. During source data file preparation, when a megatexture was picked for a particular iteration of a site, the corresponding pH and organic matter values were selected as well.

Table C-3. EPACMTP Soil Texture Crosswalk

STATSGO Texture	EPACMTP Megatexture
Sand	Sandy loam
Loamy sand	
Sandy loam	
Silt loam	Silt loam
Silt	
Loam	
Sandy clay loam	
Clay loam	
Silty clay loam	Silty clay loam
Sandy clay	
Silty clay	
Clay	

C.3.3 Results

Attachment C-1 lists the STATSGO soil textures and EPACMTP megatexture assignments and percentages for each CCW disposal site.

C.4 Hydrogeologic Environments (Aquifer Type)

To assign aquifer properties used by EPACMTP, it was necessary to designate hydrogeologic environments (or aquifer types) for each of the locations modeled so that correlated, national aquifer property data could be used in the analysis. EPACMTP uses the Hydrogeologic Database (HGDB) developed by the American Petroleum Institute (API) (Newell et al., 1989; Newell et al., 1990) to specify correlated probability distributions, which were used to populate the following four hydrogeologic parameters during the Monte Carlo analysis:

- Unsaturated zone thickness
- Aquifer thickness

- Hydraulic gradient
- Saturated hydraulic conductivity.

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). The *EPACMTP User's Guide* (U.S. EPA, 1997) provides the empirical distributions of the four hydrogeologic parameters for each of the hydrogeologic environments.

Average aquifer/vadose zone temperature was also required for the groundwater model and was obtained from a digitized map of groundwater temperatures for the continental United States from the *Water Encyclopedia* (van der Leeden et al., 1990).

The hydrogeologic environment approach to assigning EPACMTP aquifer variables relied upon a hydrogeologic framework originally developed for an attempt by EPA to classify and score groundwater environments according to their potential to be polluted by pesticide application. Although this DRASTIC[3] scoring system was not widely applied to determining groundwater vulnerability to pesticide pollution, the hydrogeologic framework established for the effort has proven very useful in categorizing geologic settings in terms of the aquifer characteristics needed for groundwater modeling. The major components of this modeling framework are Groundwater Regions, hydrogeologic settings, and hydrogeologic environments, as described below:

- The fifteen **Groundwater Regions**, defined by Heath (1984), provide a regional framework that groups hydrogeologic features (i.e., nature and extent of dominant aquifers and their relationship to other geologic units) that influence groundwater occurrence and availability.

- **Hydrogeologic settings** were developed within each Heath region by Aller et al. (1987)[4] to create mappable geological units that are at the proper scale to capture differences in aquifer conditions. Note that there may be the same or similar settings across different regions (e.g., the alluvial settings). Within each region, Aller et al. (1987) describe each setting with a written narrative and provide a block diagram to visualize the geology, geomorphology, and hydrogeology.

- **Hydrogeologic environments** were developed by Newell et al. (1990) as the geologic framework for the API's HGDB. To create the 12 environments, Newell et al. rolled up similar hydrologic settings across the Groundwater Regions to group settings with similar aquifer characteristics (hydraulic conductivity, gradient, thickness, and depth-to-water). **Table C-4** shows the crosswalk between hydrologic environment and hydrogeologic setting, organized by Groundwater Region.

[3] The DRASTIC scoring factors are Depth to water, net Recharge, Aquifer media, Soil media, Topography, Impact of the vadose zone media, and aquifer hydraulic Conductivity.

[4] Aller et al. (1987, p. 14) did not develop settings for Region 15 (Puerto Rico and the Virgin Islands) and reincorporated Region 12 (Alluvial Valleys) into each of the other regions as "river alluvium with overbank deposits" and "river alluvium without overbank deposits."

Because EPACMTP uses the HGDB for national and regional analyses (using a regional site-based approach), it was necessary to assign the CCW sites to a hydrogeologic environment so that the correct HGDB data set would be used for modeling each site. The data sources and methodology used to make these assignments are described below.

C.4.1 Data Sources

Data sources used to make hydrogeologic assignments for the sites included:

- A USGS inventory of state groundwater resources (Heath, 1985)
- GIS coverages from *Digital Data Sets Describing Principal Aquifers, Surficial Geology, and Ground-Water Regions of the Conterminous United States* (Clawges and Price, 1999a-d)
- GIS coverages of principal aquifers from the USGS *Groundwater Atlas* (Miller, 1998)
- STATSGO soil texture data (described in Section C.3.2).

These coverages were used in a GIS overlay process to determine the principal aquifers, surficial geologic units, groundwater region, productive aquifers, and general hydrogeologic settings for a 5-km radius around each CCW facility location. Attributes for each of these items were passed to a database for use in assigning hydrogeologic environments.

C.4.2 Assignment Methodology

For each CCW site, hydrogeologic environments were assigned by a professional geologist as follows:

- Determine Heath Groundwater Region (for the Alluvial Valleys region, determine the region in which the alluvial valley is located)
- Assign hydrogeologic setting using state geological descriptions from Heath (1985); aquifer, soil, and surficial geology information obtained using GIS; and narratives and block diagrams from Aller et al. (1987)
- Using the look-up table from Newell et al. (1990), determine hydrogeologic environment from hydrogeologic setting.

In general, the surficial geology coverage had better resolution than the aquifer coverages and was used to develop setting percentages for the 5-km radius. In most cases, there were two settings per site. In cases where a single setting accounted for over 80 percent of the 5-km area, a single setting was assigned.

Because Newell et al. (1990) define two alluvial environments (6, River alluvium with overbank deposits, and 7, River alluvium without overbank deposits), it was necessary to determine which environment an alluvial site fell into. The survey soil layer information was used to distinguish between these two settings by determining whether there were significant fine-grained overbank deposits in the soil column.

Quality assurance/quality control (QA/QC) measures included independent review of the assignments by other geologists with expertise in assigning settings.

C.4.3 Data Processing

HGDB hydrogeologic environment fractions (i.e., the portion of the region assigned to each of the 12 hydrogeological environments) were defined and used in the CCW risk assessment as follows. If the 5-km radius around a site contained only one HGDB environment, the fraction assigned was 1.0 and all groundwater model runs for that location were associated with that hydrological environment. If more than one HGDB environment was present, each environment was assigned a fraction based on the areal percentages of each setting within the 5-km radius.

These fractions were used to generate the hydrogeologic environment for that location for each iteration of the Monte Carlo groundwater modeling analysis. For example, if two hydrogeologic environments were assigned to a CCW site with a fraction of 0.5, half of the realizations were modeled with the first hydrogeologic environment and half with the second.

Once the hydrogeologic environments were assigned, a preprocessing run of EPACMTP was conducted to construct a set of randomly generated but correlated hydrogeologic parameter values for each occurrence of the hydrogeologic environments in the source data files. Missing values in the HGDB data set were filled using correlations, as described in U.S. EPA (1997).

C.4.4 Results

Attachment C-2 lists the hydrogeologic environment assignments for each CCW disposal site. **Table C-4** summarizes these results, showing the crosswalk between Groundwater Regions, hydrogeologic settings, and hydrogeologic environments used to make the assignments, along with the number of CCW sites for each setting. **Table C-5** totals the number of CCW disposal sites for each hydrogeologic environment sent to EPACMTP.

Table C-4. Groundwater Regions, Hydrogeologic Settings, and Hydrogeologic Environments: CCW Disposal Sites

Hydrogeologic Setting		Hydrogeologic Environment	Number of CCW Sites
Alluvial Basins			
2C	Alluvial Fans	5	1
2E	Playa Lakes	5	1
2Ha	River Alluvium With Overbank Deposits	6	1
Colorado Plateau and Wyoming Basin			
4B	Consolidated Sedimentary Rock	2	7
4C	River Alluvium	7	3
High Plains			
5Gb	River Alluvium Without Overbank Deposits	7	1

(continued)

Groundwater Regions, Hydrogeologic Settings, and Hydrogeologic Environments: CCW Disposal Sites. (continued)

Hydrogeologic Setting		Hydrogeologic Environment	Number of CCW Sites
Nonglaciated Central Region			
6Da	Alternating Sandstone, Limestone, and Shale – Thin Soil	2	22
6Db	Alternating Sandstone, Limestone, and Shale – Deep Regolith	2	6
6E	Solution Limestone	12	9
6Fa	River Alluvium With Overbank Deposits	6	37
6Fb	River Alluvium Without Overbank Deposits	7	4
6H	Triassic Basins	2	4
Glaciated Central Region			
7Aa	Glacial Till Over Bedded Sedimentary Rock	3	12
7Ac	Glacial Till Over Solution Limestone	12	6
7Ba	Outwash	8	1
7Bb	Outwash Over Bedded Sedimentary Rock	2	3
7Bc	Outwash Over Solution Limestone	12	2
7D	Buried Valley	4	11
7Ea	River Alluvium With Overbank Deposits	6	24
7Eb	River Alluvium Without Overbank Deposits	7	6
7F	Glacial Lake Deposits	4	3
7G	Thin Till Over Bedded Sedimentary Rock	3	5
7H	Beaches, Beach Ridges, and Sand Dunes	11	1
Piedmont and Blue Ridge			
8B	Alluvial Mountain Valleys	5	1
8C	Mountain Flanks	2	2
8D	Regolith	1	13
8E	River Alluvium	6	6
Northeast and Superior Uplands			
9E	Outwash	8	3
9F	Moraine	4	1
9Ga	River Alluvium With Overbank Deposits	6	1
Atlantic and Gulf Coastal Plain			
10Aa	Regional Aquifers	4	1
10Ab	Unconsolidated/Semiconsolidated Shallow Surficial Aquifers	10	20
10Ba	River Alluvium With Overbank Deposits	6	7
10Bb	River Alluvium Without Overbank Deposits	7	6
Southeast Coastal Plain			
11A	Solution Limestone and Shallow Surficial Aquifers	12	3
11B	Coastal Deposits	4	1

Table C-5. Hydrogeologic Environments for CCW Disposal Sites

Hydrogeologic Environment		Number of CCW Sites
1	Metamorphic and Igneous	13
2	Bedded Sedimentary Rock	44
3	Till Over Sedimentary Rock	17
4	Sand and Gravel	17
5	Alluvial Basins Valleys and Fans	3
6	River Valleys and Floodplains With Overbank Deposit	76
7	River Valleys and Floodplains Without Overbank Deposits	20
8	Outwash	4
9	Till and Till Over Outwash	0
10	Unconsolidated and Semiconsolidated Shallow Aquifers	20
11	Coastal Beaches	1
12	Solution Limestone	20

C.5 Climate Data

The CCW risk assessment selected EPACMTP meteorological (or climate) stations for each CCW disposal site to collect the climatic data necessary for fate and transport modeling. For each station, the following data were compiled:

- Mean annual windspeed
- Mean annual air temperature
- Mean annual precipitation.

With respect to precipitation, EPACMTP uses the climate station, along with soil texture, to select the HELP-modeled infiltration rates to use in the landfill source model and recharge rates to use in EPACMTP (see **Section 3.2.2**). The surface water model uses mean annual windspeed and average air temperature to estimate volatilization losses from the surface waterbodies modeled in the analysis.

To assign the EPACMTP/HELP climate centers to each CCW site, a GIS was used to determine the three meteorological stations closest to the plant. These assignments were passed to a meteorologist, who reviewed the closest stations against plots of the CCW sites and the climate centers on a downloadable map (http://www.nationalatlas.gov) of annual average precipitation rates for the period from 1961 to 1990 across the contiguous United States. (**Figure C-2**). The meteorologist compared the 5-year average precipitation range for each EPACMTP climate center to precipitation ranges for each plant from the map. In most cases, the precipitation rate for the nearest climate center fell within the site's expected precipitation range, and the nearest climate center was assigned in those cases. In some cases, the precipitation rates from the nearest climate center did not fall within the site's expected range. When this occurred, the second or third closest climate center was examined and matched based on:

- A 5-year precipitation average within or close to the site's predicted precipitation range

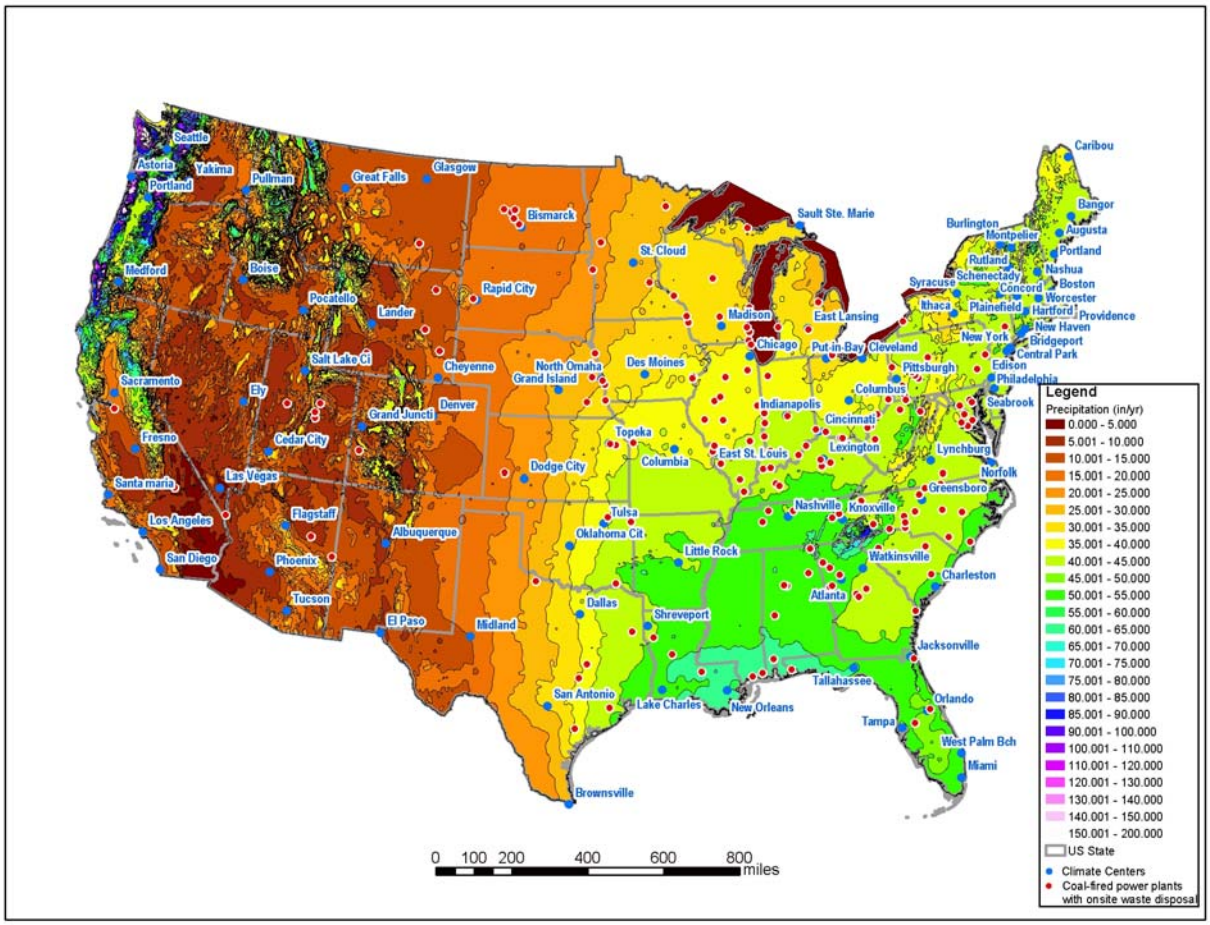


Figure C-2. EPACMTP climate centers, precipitation ranges, and CCW disposal sites.

- Confirmation of a site's average annual rainfall on http://www.weather.com and van der Leeden et al. (1990)
- Geographic similarities between plant and climate center locations
- Best professional judgment.

In a few cases, the three closest climate centers did not reflect the average precipitation rates for a plant's location. In these cases, other nearby stations were examined and the plant was assigned to the closest climate center with similar geography and average precipitation rates. Each assignment was independently checked for accuracy. **Attachment C-3** lists the climate center assigned to each CCW disposal site, along with notes for plants not assigned to the nearest center. **Table C-6** lists all the climate centers used in the CCW risk assessment along with the number of CCW sites assigned to each station.

Table C-6. EPACMTP Climate Centers Assigned to CCW Disposal Sites

Climate Center		State	Number of CCW Sites
4	Grand Junction	CO	2
6	Glasgow	MT	1
7	Bismarck	ND	5
10	Cheyenne	WY	2
11	Lander	WY	1
13	Sacramento	CA	1
16	Ely	NV	1
17	Rapid City	SD	2
18	Cedar City	UT	1
19	Albuquerque	NM	1
20	Las Vegas	NV	3
21	Phoenix	AZ	1
26	Salt Lake City	UT	1
29	Dodge City	KS	1
31	St. Cloud	MN	3
32	East Lansing	MI	3
33	North Omaha	NE	7
34	Tulsa	OK	2
37	Oklahoma City	OK	1
39	Pittsburgh	PA	12
42	Chicago	IL	8
48	Sault Ste. Marie	MI	1
49	Put-in-Bay	OH	3
50	Madison	WI	9
51	Columbus	OH	2
53	Des Moines	IA	2
54	East St. Louis	IL	8
55	Columbia	MO	1
56	Topeka	KS	3
58	San Antonio	TX	4
66	Ithaca	NY	1
69	Lynchburg	VA	2
71	Philadelphia	PA	2
72	Seabrook	NJ	5
73	Indianapolis	IN	12
74	Cincinnati	OH	11
75	Bridgeport	CT	1
76	Orlando	FL	2
77	Greensboro	NC	11

(continued)

EPACMTP Climate Centers Assigned to CCW Disposal Sites. (continued)

Climate Center		State	Number of CCW Sites
78	Jacksonville	FL	1
79	Watkinsville	GA	4
80	Norfolk	VA	2
81	Shreveport	LA	4
85	Knoxville	TN	4
87	Lexington	KY	3
89	Nashville	TN	4
90	Little Rock	AR	1
91	Tallahassee	FL	4
93	Charleston	SC	4
95	Atlanta	GA	9
96	Lake Charles	LA	2

C.6 Surface Water Data

The surface water model used in the CCW risk assessment requires information on surface waterbody type (river or lake), flow conditions, dimensions, and water quality. In addition, the groundwater model requires the distance between the waterbody and the WMU being modeled. Surface waterbody data were collected on a site-based, regional, or national basis depending on the variable and data availability. Collection methods are described below by data source. **Attachment C-4** provides a summary of waterbody assignments, waterbody types, and flow conditions.

C.6.1 Waterbody Type, Stream Flow Conditions, and Dimensions

Waterbody type and flow parameters were obtained by matching the CCW plants to stream segments in the Reach File Version 1.0 (RF1) database (U.S. EPA, 1990). Stream flow estimates for all RF1 flowing reaches were estimated in the early 1980s. Statistics developed for each flowing reach were mean annual flow, low flow (approximately 7Q10),[5] and mean monthly flow. RF1 also contains velocities corresponding to mean annual and low flow, estimated from a compendium of time-of-travel studies. For streams and rivers, the CCW risk assessment used the low flow statistic and the corresponding flow velocity, along with a waterbody type also included in the RF1 database. All RF1 data are indexed by USGS cataloging unit and stream segment (CUSEG).

To assign the CCW plants to the nearest downgradient reach (i.e., the nearest waterbody in the direction of groundwater flow), a GIS was used to identify the closest RF1 stream segment to each CCW plant location. Because of several uncertainties in the nearest reach approach (i.e.,

[5] The 7Q10 is the minimum 7-day average flow expected to occur within a 10-year return period (i.e., at least once in 10 years).

inaccurate WMU location, unknown direction of groundwater flow, and limited lake coverages), the CCW plants also were matched to standard industrial classification (SIC) code 4911 facilities in EPA's Permit Compliance System (PCS) database (http://www.epa.gov/enviro/html/pcs/index.html), to obtain the PCS information (e.g., name, CUSEG) on the receiving waterbody for the plants' National Pollutant Discharge Elimination System (NPDES) discharge point(s). When the two sources matched, the reach was selected for modeling. When they differed, the PCS data were used, because it was judged more likely that the NPDES receiving waterbody would also be receiving loads from the WMU through the groundwater-to-surface-water pathway. CCW plants that could not be matched to the PCS database were simply assigned the nearest RF1 waterbody.

The next step in the assignment process was to review the waterbody names (especially those from PCS) to identify lakes and reservoirs. Finally, visual review, using aerial photos and topographic maps from the Terraserver Web site (http://www.terraserver.com), was used to check all low-flow streams and RF1 reaches whose identity was not clear. **Attachment C-4** provides the RF1 stream assignments, flows, and waterbody types for the CCW disposal sites.

With respect to waterbody type, the RF1 data include several types of waterbodies, including streams and rivers, and types with zero flows such as lakes, Great Lakes, wide rivers, and coastline features. Each of these waterbody types needed to be designated as a river or a lake for the simple waterbody model used in the full-scale CCW risk assessment. Because only the streams and rivers have flow data in RF1 (i.e., are flowing reaches), all other types were assigned to the lake modeling category. Modeling these features as a simple model lake is an uncertainty in the CCW risk assessment **Table C-7** lists the RF1 waterbody types for the waterbodies assigned to the CCW disposal sites, along with the number of CCW plants assigned to each type and the crosswalk to the river (R) or lake (L) waterbody type used in this risk assessment.

Table C-7. RF1 Reach Types Assigned to CCW Disposal Sites

RF1 Code	RF1Name	Description	Reach Model Type[a]	Number of CCW Plants
Flowing Reaches				
M	Artificial Open Water Reach	An artificial reach within any open water, other than a lake or reservoir, to provide connection between input and output reaches of the open water.	R	1
R	Regular Reach	A reach that has upstream and downstream reaches connected to it and that is not classified as another type of reach.	R	106
S	Start Reach	A headwater reach that has no reaches above it and either one or two transport reaches connected to its downstream end.	R	16
T	Terminal Reach	A reach downstream of which there is no other reach (for example, a reach that terminates into an ocean, a land-locked lake, or the ground). This type of reach has either one or two reaches connected to its upstream end.	R	2

(continued)

RF1 Reach Types Assigned to CCW Disposal Sites. (continued)

RF1 Code	RF1Name	Description	Reach Model Type[a]	Number of CCW Plants
Reaches with Zero RF1 Flow				
C	Coastal/Continental Shoreline Segment	A reach that represents a segment of a shoreline of a gulf, sea, or ocean.	L	3
G	Great Lakes Shoreline Segment	A reach that represents a segment of a shoreline of the Great Lakes.	L	12
L	Lake Shoreline Segment	A segment that follows the shoreline of a lake other than one of the Great Lakes.	L	36
W	Wide-River Shoreline Segment	A reach that represents a segment of the left or right bank of a stream.	L	5

[a] R = river; L = lake.

Stream dimensions were calculated from the flow data as follows. First, the length of the modeled stream segment was set to be the width of the groundwater plume as it enters the waterbody. Stream width was then determined from flow (Q) using a liner regression equation derived from empirical data by Kocher and Sartor (1997):

$$Width = 5.1867Q^{0.4559} \quad \text{(C-1)}$$

Water column depth (dwc) was derived from width, velocity (V), and flow using the continuity equation:

$$dwc = \frac{Q}{v \times Width} \quad \text{(C-2)}$$

C.6.2. Lake Flow Conditions and Dimensions

Areas and depths for many of the lakes assigned to the CCW plant sites were not readily available from RF1, Reach File Version 3 (RF3), the National Hydrography Dataset (NHD), or other sources. In addition, many plants were located on very large waterbodies (e.g., the Great Lakes, wide rivers, or coastlines), where applying the simple steady-state, single-compartment model used in this analysis to the entire lake would not be appropriate. For these reasons, a model lake approach was used to represent all lakes and other nonflowing waterbodies assigned to the CCW disposal site.

The model lake chosen was Shipman City Lake in Illinois, a well-characterized 13-acre lake that EPA has chosen as the index reservoir for modeling drinking water exposures to pesticides (Jones et al., 1998). The parameter values shown in **Table C-8** for Shipman City Lake were used to model all lakes in this initial analysis. Given that many of the lakes assigned to CCW plants were much larger than 13 acres, this produced high-end risk results. However, given that many of the plants were located on very large waterbodies, this necessary simplification is an uncertainty in defining the environmental settings for the CCW risk assessment.

Table C-8. Model Lake Used in CCW Risk Assessment

Parameter	Value
Area [a]	13 acres
Water column depth (dwc)[a]	9 feet
Hydraulic residence time (HRT)	Random, triangular distribution: Minimum = 1 month Mean = 6 months Maximum = 24 months
Annual flow mixing volume	= (Area × dwc) / HRT

[a] Source: Shipman City Lake, IL (Jones et al., 1998).

C.6.3 Water Quality Data

Surface water temperature, total suspended solids (TSS), and pH data were collected by USGS hydrologic region from the STORET database. EPA's STORET system is the largest single source of water quality data in the country. The Legacy STORET database contains over 275 million analyses performed on more than 45 million samples collected from 800,000 stations across the United States for the period 1960 through 1998. STORET can be accessed from the Web at http://www.epa.gov/STORET.

STORET water quality data are notoriously "noisy" because they are influenced by hydrology, point sources, nonpoint sources, stream/lake morphology, and varying data quality. The following issues in using STORET data must be considered before using the data:

- Not all of the data have undergone rigorous QA/QC.
- STORET site locations can be biased, especially to known "problem" waters.
- The sample times are often at critical periods, such as summer low flows.

Statistical analysis techniques were employed taking into account the above issues (including coordination with gage statistical analysis and Reach Files, the use of median values to avoid bias in central tendency estimates, and specification of a minimum number of measurements to estimate median values). As a result of these techniques, which can be thought of as extracting the underlying "signal" of water quality from the inherent "noise" of water quality data, the above issues were manageable.

Surface water temperature data were collected as median values for each hydrologic region. These data are shown in **Table C-9** along with the number of the modeled CCW plants in each region.

Table C-9. Regional Surface Water Temperatures: CCW Disposal Sites

Hydrologic Region	Surface Water Temperature (°C)	Number of CCW Plants
2	16	12
3	21	37
4	14	14
5	17	43
6	18	6
7	15	20
8	20	2
9	10	1
10	13	20
11	17	8
12	21	6
14	9	5
15	17	4
16	9	1
18	15	2

Data source: Legacy STORET database.

Total suspended solids data were collected separately for streams/rivers and lakes because lakes tend to have lower TSS levels. Annual median values were used to develop statistics. For rivers, the minimum, maximum, and geometric mean values were used to define log triangular distributions for each hydrologic region (**Table C-10**); these distributions were then sampled during the preparation of the source data files. (The geometric means were weighted by the annual number of measurements.) For lakes, data were limited and national statistics were developed, with the geometric mean of the median values being weighted by the number of measurements per year and the number of annual values in each region.

Table C-10. Surface Water Total Suspended Solids (TSS) Distributions

Hydrologic Region	Number of CCW Plants	No. of Measure-ments	No. of Annual Medians	Annual Median TSS (log triangular distribution)			Geometric Mean
				Minimum	Maximum	Weighted Geometric Mean	
1	0	9,007	33	3.2	40	8.0	6.0
2	12	47,202	38	10	316	32	40
3	37	43,395	36	6.3	79	25	25
4	14	29,577	37	6.3	794	25	25
5	43	39,900	38	4.0	100	25	25
6	6	4,137	28	5.0	316	16	20

(continued)

Surface Water Total Suspended Solids (TSS) Distributions. (continued)

Hydrologic Region	Number of CCW Plants	No. of Measure-ments	No. of Annual Medians	Annual Median TSS (log triangular distribution)			Geometric Mean
				Minimum	Maximum	Weighted Geometric Mean	
7	20	34,494	37	32	1,585	63	100
8	2	46,231	38	50	316	158	126
9	1	3,254	35	13	3,162	32	63
10	20	62,791	38	10	398	126	126
11	8	48,969	38	25	794	200	126
12	6	7,280	35	40	1,995	79	126
13	0	13,974	37	32	79,433	200	398
14	5	26,699	38	16	5,012	158	251
15	4	9,162	37	20	19,953	200	398
16	1	19,965	33	4	2,512	16	25
17	0	173,136	37	2	316	6.0	10
18	2	42,022	37	13	398	63	50
Lakes (national)	56	4,360	99	1	398	25	25

Data source: Legacy STORET database.

For **surface water pH**, the minimum, maximum, and weighted average annual median values were used to specify triangular distributions for each hydrologic region. **Table C-11** provides these regional statistics, which were applied to both rivers and lakes.

To prepare the water quality data for the source datafile, the 181 CCW disposal sites were assigned to a hydrogeologic region using a GIS. For each region, 10,000-record TSS and pH data sets were created by sampling the distributions shown in Tables C-10 and C-11. During source data file preparation, TSS data were pulled from the appropriate regional data set sequentially for each iteration at a site.

Table C-11. Regional Surface Water pH Distributions

Hydrologic Region	Number of CCW Plants	No. of Measurements	No. of Annual Median Values	Annual Median pH (triangular distribution)			Average Median pH
				Minimum	Maximum	Weighted Average	
1	0	232,025	38	5.9	7.7	6.5	6.8
2	12	447,166	39	7.2	7.6	7.4	7.4
3	37	1,595,237	39	6.3	7.2	7.0	7.0
4	14	335,261	39	7.6	8.2	8.1	8.0
5	43	684,235	41	3.5	7.5	7.2	7.1
6	6	382,915	39	6.3	7.7	7.2	7.4

(continued)

Regional Surface Water pH Distributions. (continued)

Hydrologic Region	Number of CCW Plants	No. of Measurements	No. of Annual Median Values	Annual Median pH (triangular distribution)			Average Median pH
				Minimum	Maximum	Weighted Average	
7	20	234,589	39	7.6	8.1	7.9	7.8
8	2	171,643	39	6.9	7.8	7.1	7.2
9	1	23,038	38	7.5	8.4	7.9	7.9
10	20	269,570	39	7.6	8.2	8.0	8.0
11	8	311,768	39	7.4	8.1	7.8	7.8
12	6	178,990	39	7.0	7.9	7.8	7.6
13	0	35,355	39	7.0	8.1	8.0	7.9
14	5	77,041	39	7.9	8.3	8.1	8.1
15	4	75,145	38	7.7	8.3	8.0	8.0
16	1	68,581	38	7.5	8.3	8.0	8.0
17	0	293,909	39	6.9	8.0	7.5	7.4
18	2	182,049	38	7.4	8.6	7.8	7.8

Data source: Legacy STORET database.

C.6.4 Distance to Surface Water

Because the CCW plant locations were not accurate in terms of locating the WMUs, a national empirical distribution of distances between the WMU and the nearest downgradient surface waterbodies (discussed in **Section C.2.1**) was developed using manual measurements on online maps and aerial photographs for a random selection of 30 CCW landfills and 29 CCW surface impoundments. Scaled USGS maps and aerial photographs were obtained from the Terraserver Web site (http://www.terraserver.com) by entering each plant's longitude and latitude. Labels on the maps, features on the photographs, and best professional judgment were used to identify the power plant and the surface impoundment or landfill in question, along with the nearest downgradient waterbody.

The nearest waterbody matching one of the following descriptions was used in the analysis:

- Lakes or rivers beyond the facility boundary
- Streams originating in or passing through the facility boundary and then coursing downstream beyond the property boundary
- Streams with an order of 3 or greater (i.e., fishable waterbodies).

Stream order was determined by tracing the convergence of tributaries with order 1 assigned to the furthest upstream segment indicated on the map (both ephemeral and perennial streams were assigned as order 1). Topography on the map was used to determine if the waterbody was downgradient of the plant. Many CCW WMUs in the sample were located on a large waterbody.

Once the waterbody was identified, the scale provided on the maps and photos was used to measure the horizontal distance between the CCW impoundment or landfill and the waterbody. All assignments and measurements were independently checked for accuracy.

The two distributions (landfills and surface impoundments) were statistically compared using (1) a Wilcoxon Rank Sum Test (to determine whether one distribution is shifted to the right or left of the other distribution) and (2) a Quantile Test (to test for differences, that is, differing numbers of observations) between the two distributions for the values above a given percentile. The results of the Wilcoxon test showed a p value of 0.64, indicating no significant difference in the shape of the distributions. The Quantile Test evaluated every decile from 0.1 to 0.9, with adjustments to the lower percentiles to be estimated for large numbers of ties in the ranks for the lower end of the data. The nonsignificant p values ranged from 0.33 (for 90th percentile) to 0.17 (for the 40th percentile). One significant p value indicating differences between the two distributions occurred at the 17th percentile (p value = 0.066), but the remainder of the tests showed no significant differences. Based on these results, the distributions were judged to be similar and combined to produce the single distribution of 59 values used to produce a single empirical distribution (previously shown in Table C-2) that was applied nationally to both landfills and surface impoundments at the CCW sites.

C.5 References

Aller, L., T. Bennett, J.H. Lehr, R.J. Perry, and G. Hackett. 1987. *DRASTIC: A Standardized System for Evaluating Pollution Potential Using Hydrogeologic Settings.* EPA-600/2-87-035. U.S. Environmental Protection Agency. Office of Research and Development. Ada, OK. April.

Carsel, R.F., and R.S. Parrish. 1988. Developing joint probability distributions of soil water retention characteristics. *Water Resources Research 24*(5):755–769.

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.

Clawges, R.M., and C.V. Price. 1999a. *Digital Data Sets Describing Principal Aquifers, Surficial Geology, and Ground-Water Regions of the Conterminous United States*. Open-File Report 99-77. U.S. Geological Survey, Rapid City, SD. Available at http://pubs.usgs.gov/of/1999/ofr99-77/.

Clawges, R.M., and C.V. Price. 1999b. *Ground-water regions of the conterminous United States with unconsolidated watercourses.* Edition: 1.0 (map). In: Open-File Report 99-77 (Clawges and Price, 1999a). U.S. Geological Survey, Rapid City, SD. Available at http://water.usgs.gov/lookup/getspatial?ofr99-77_gwreguw.

Clawges, R.M., and C.V. Price. 1999c. *Productive aquifers of the conterminous United States. Version 1.0* (map). In Open-File Report 99-77(Clawges and Price, 1999a). U.S. Geological Survey, Rapid City, SD. Available at http://water.usgs.gov/lookup/getspatial?ofr99-77_aquif75m.

Clawges, R.M., and C.V. Price. 1999d. *Surficial geology of the conterminous United States. Version 1.0*. In Open-File Report 99-77 (Clawges and Price, 1999a). U.S. Geological Survey, Rapid City, SD. Available at http://water.usgs.gov/lookup/getspatial?ofr99-77_geol75m.

Heath, R.C. 1984. *Ground-Water Regions of the United States*. Water-Supply Paper 2242, U.S. Geological Survey, Washington, DC. Available at http://pubs.usgs.gov/wsp/wsp2242/.

Heath, R.C. 1985. *National Water Summary 1984*. *State Summaries of Groundwater Resources*. Water-Supply Paper 2275. U.S. Geological Survey, Washington, DC.

Jones, R.D., S.W. Abel, W. Effland, R. Matzner, and R. Parker. 1998. An index reservoir for use in assessing drinking water exposures. Chapter IV in *Proposed Methods for Basin-Scale Estimation of Pesticide Concentrations in Flowing Water and Reservoirs for Tolerance Reassessment*, presented to the FIFRA Scientific Advisory Panel. July. Available at http://www.epa.gov/scipoly/sap/meetings/1998/july/1part4.pdf.

Kocher, K., and R. Sartor. 1997. Derivation and use of data and table to determine stream flow values for water bodies. Memorandum to File, RTI Project 6720-005. Research Triangle Institute, Research Triangle Park, NC. June 16.

Miller, J.A. 1998. *Principal Aquifers of the 48 Contiguous United States*. Version 1.0. United States Geological Survey, Madison, WI. Available at http://nationalatlas.gov/.

Miller, D.A., and R.A. White. 1998. A Conterminous United States Multi-Layer Soil Characteristics Data Set for Regional Climate and Hydrology Modeling. *Earth Interactions* 2(2): 1–26.

Newell, C.J., L.P. Hopkins, and P.B. Bedient. 1989. *Hydrogeologic Database for Ground Water Modeling*. API Publication No. 4476. American Petroleum Institute, Washington, DC.

Newell, C,J., L.P. Hopkins, and P.B. Bedient. 1990. A hydrogeologic database for ground water modeling. *Ground Water 28*(5):703–714.

Schwarz, G.E., and R.B. Alexander. 1995. *State Soil Geographic (STATSGO) Data Base for the Conterminous United States. Edition: 1.1*. Reston, VA. Available at http://water.usgs.gov/GIS/metadata/usgswrd/XML/muid.xml.

USDA (U.S. Department of Agriculture). 1994. *State Soil Geographic (STATSGO) Data Base. Data Use Information.* Miscellaneous Publication No. 1492. Natural Resources Conservation Service, Fort Worth, TX. Available at http://www.nrcs.usda.gov/technical/techtools/statsgo_db.pdf.

U.S. EPA (Environmental Protection Agency). 1988. *National Survey of Solid Waste (Municipal) Landfill Facilities* (Draft). EPA/530-SW88-034. Office of Solid Waste and Emergency Response, Washington, DC. September. Available in docket number EPA-HQ-RCRA-2003-0004 as document number EPA-HQ-RCRA-2003-0004-0466.

U.S. EPA (Environmental Protection Agency). 1990. *Office of Water Environmental and Program Information Systems Compendium. FY 1990.* EPA 500/9-90-002. Office of Water, August.

U.S. EPA (Environmental Protection Agency). 1997. *EPA's Composite Model for Leachate Migration with Transformation Products (EPACMTP) User's Guide*. Office of Solid Waste, Washington, DC. Available at http://www.epa.gov/waste/nonhaz/industrial/tools/cmtp/index.htm.

van der Leeden, F., F.L. Troise, and D.K. Todd. 1990. *The Water Encyclopedia.* 2nd Ed. Chelsea, MI: Lewis Publishers. 176 pp.

[This page intentionally left blank.]

Attachment C-1: Soil Data

Plant	Percent Composition	Megatexture Code	Average pH	Average % Organic Material
A B Brown	43.9	SCL	6.0	1.2
A B Brown	51.1	SLT	6.5	1.6
A B Brown	5.0	SNL	6.9	1.4
A/C Power- Ace Operations	8.9	SCL	8.9	0.21
A/C Power- Ace Operations	32.0	SLT	8.4	0.46
A/C Power- Ace Operations	59.1	SNL	8.0	0.46
Allen	48.9	SCL	7.1	0.98
Allen	19.2	SLT	6.2	1.1
Allen	32.0	SNL	7.1	1.1
Alma	18.9	SCL	6.6	1.7
Alma	59.4	SLT	6.5	3.4
Alma	21.7	SNL	5.6	0.69
Antelope Valley	8.4	SCL	7.6	3.2
Antelope Valley	68.5	SLT	7.6	1.7
Antelope Valley	23.1	SNL	7.8	2.4
Arkwright	50.7	SCL	5.4	0.5
Arkwright	24.7	SLT	5.6	0.88
Arkwright	24.5	SNL	5.4	0.64
Asheville	6.3	SCL	5.4	0.43
Asheville	77.8	SLT	5.2	0.99
Asheville	15.8	SNL	5.4	1
Baldwin	39.5	SCL	6.2	1.3
Baldwin	58.6	SLT	6.0	1.6
Baldwin	1.9	SNL	6.5	1.4
Barry	35.8	SCL	4.8	3.6
Barry	23.5	SLT	4.8	7
Barry	40.7	SNL	4.8	4.4
Bay Front	11.7	SCL	7.3	4
Bay Front	21.1	SLT	7.1	3.8
Bay Front	67.2	SNL	7.1	1.4
Bay Shore	90.8	SCL	7.1	4.1
Bay Shore	4.3	SLT	7.2	2.6
Bay Shore	4.9	SNL	7.7	9.3
Belews Creek	69.2	SCL	5.2	0.34
Belews Creek	14.0	SLT	5.4	1

(continued)

Soil Data (continued)

Plant	Percent Composition	Megatexture Code	Average pH	Average % Organic Material
Belews Creek	16.8	SNL	5.2	0.4
Ben French	25.3	SCL	8.0	0.87
Ben French	59.7	SLT	7.7	1.8
Ben French	15.0	SNL	7.1	1.7
Big Cajun 2	66.4	SCL	7.1	1.1
Big Cajun 2	28.4	SLT	6.3	1.2
Big Cajun 2	5.2	SNL	6.0	1.3
Big Sandy	54.8	SCL	5.4	1.6
Big Sandy	41.5	SLT	5.3	1.9
Big Sandy	3.7	SNL	5.1	2.6
Big Stone	7.3	SCL	7.5	5.7
Big Stone	45.0	SLT	7.7	3.1
Big Stone	47.7	SNL	7.5	1.1
Black Dog Steam Plant	8.2	SCL	6.9	4.2
Black Dog Steam Plant	41.4	SLT	6.8	2.5
Black Dog Steam Plant	50.4	SNL	6.9	1.8
Blue Valley	63.8	SCL	6.3	1.5
Blue Valley	31.6	SLT	6.6	2.8
Blue Valley	4.6	SNL	6.5	1.1
Bowen	18.1	SCL	5.0	1.2
Bowen	81.9	SLT	5.0	0.74
Brandon Shores	18.2	SCL	4.5	0.47
Brandon Shores	16.8	SLT	4.6	3.4
Brandon Shores	64.9	SNL	4.8	0.88
Buck	79.1	SCL	5.4	0.39
Buck	18.9	SLT	5.6	1
Buck	2.0	SNL	5.3	0.6
Bull Run	76.7	SCL	5.2	0.92
Bull Run	18.2	SLT	5.6	1.7
Bull Run	5.1	SNL	5.0	0.67
C D McIntosh Jr	6.5	SCL	8.1	2.3
C D McIntosh Jr	93.5	SNL	5.5	1.8
C P Crane	34.1	SCL	4.8	0.52
C P Crane	34.3	SLT	4.7	1
C P Crane	31.6	SNL	4.9	1.1
Cape Fear	67.6	SCL	5.1	0.97
Cape Fear	24.7	SLT	5.4	1.5
Cape Fear	7.7	SNL	5.2	0.66
Carbon	0.4	SCL	6.3	7.4

(continued)

Soil Data (continued)

Plant	Percent Composition	Megatexture Code	Average pH	Average % Organic Material
Carbon	95.8	SLT	7.8	3.4
Carbon	3.8	SNL	8.2	1.4
Cardinal	69.1	SCL	5.8	1
Cardinal	30.4	SLT	5.7	1.7
Cardinal	0.5	SNL	6.4	2
Cayuga	32.3	SCL	6.6	1.9
Cayuga	48.7	SLT	7.1	1.4
Cayuga	19.0	SNL	6.8	1.1
Chalk Point	6.9	SCL	4.6	0.58
Chalk Point	16.4	SLT	4.8	8.8
Chalk Point	76.7	SNL	4.6	1.1
Cholla	27.3	SCL	8.4	1.9
Cholla	61.0	SLT	8.1	0.62
Cholla	11.6	SNL	8.3	0.75
Cliffside	66.4	SCL	5.2	0.31
Cliffside	13.6	SLT	5.5	0.77
Cliffside	20.0	SNL	5.2	0.27
Clover	71.0	SCL	5.3	0.71
Clover	23.3	SLT	5.3	1.3
Clover	5.7	SNL	5.1	0.65
Coal Creek	6.1	SCL	6.8	3
Coal Creek	82.7	SLT	7.6	1.7
Coal Creek	11.2	SNL	8.2	2.8
Coleto Creek	12.1	SCL	7.0	1.1
Coleto Creek	86.0	SLT	7.4	0.78
Coleto Creek	1.8	SNL	6.2	0.75
Colstrip	9.0	SCL	8.0	0.79
Colstrip	63.0	SLT	8.2	0.73
Colstrip	27.9	SNL	8.3	0.54
Conemaugh	11.8	SCL	5.0	2.7
Conemaugh	81.4	SLT	4.8	1.3
Conemaugh	6.8	SNL	4.5	1.8
Conesville	44.0	SCL	5.4	2.2
Conesville	45.5	SLT	5.6	1.9
Conesville	10.5	SNL	5.0	2.2
Council Bluffs	43.3	SCL	7.5	1.5
Council Bluffs	47.2	SLT	7.6	1.2
Council Bluffs	9.6	SNL	7.7	0.74
Crawford	48.4	SCL	6.8	1.9

(continued)

Soil Data (continued)

Plant	Percent Composition	Megatexture Code	Average pH	Average % Organic Material
Crawford	23.6	SLT	6.7	1.4
Crawford	28.0	SNL	6.7	0.82
Crist	18.8	SCL	5.4	4.5
Crist	32.3	SLT	5.3	1.1
Crist	48.8	SNL	5.4	3.3
Cross	3.0	SCL	5.0	1.3
Cross	46.0	SLT	4.6	0.58
Cross	51.0	SNL	4.9	1.2
Cumberland	61.1	SCL	5.3	1.6
Cumberland	34.2	SLT	5.7	0.98
Cumberland	4.8	SNL	5.2	1.3
Dale	91.7	SCL	6.4	1.9
Dale	8.2	SLT	6.4	2
Dale	0.1	SNL	6.7	1.3
Dallman	66.2	SCL	6.4	1.8
Dallman	33.3	SLT	6.7	1.2
Dallman	0.5	SNL	7.0	1.1
Dan E Karn	0.01	SCL	7.0	3
Dan E Karn	53.6	SLT	7.9	4.2
Dan E Karn	46.3	SNL	7.8	5.4
Dan River	73.3	SCL	5.0	0.39
Dan River	12.0	SLT	5.3	1.4
Dan River	14.7	SNL	5.1	0.6
Danskammer	89.8	SLT	5.8	2.9
Danskammer	10.2	SNL	6.9	2.8
Dave Johnston	2.2	SCL	8.9	0.96
Dave Johnston	36.6	SLT	8.2	1.2
Dave Johnston	61.2	SNL	8.2	1.1
Dickerson	6.1	SCL	5.1	0.52
Dickerson	93.9	SLT	5.2	0.68
Dolet Hills	65.7	SCL	4.8	0.97
Dolet Hills	21.6	SLT	5.0	0.77
Dolet Hills	12.7	SNL	5.1	1.1
Duck Creek	65.5	SCL	6.4	0.82
Duck Creek	33.6	SLT	6.5	0.6
Duck Creek	0.9	SNL	7.0	0.98
Dunkirk	8.8	SCL	7.3	5.4
Dunkirk	79.6	SLT	6.9	4.6
Dunkirk	11.6	SNL	6.5	2.7

(continued)

Soil Data (continued)

Plant	Percent Composition	Megatexture Code	Average pH	Average % Organic Material
E D Edwards	49.5	SCL	6.4	1.1
E D Edwards	29.8	SLT	6.3	1.2
E D Edwards	20.6	SNL	6.8	1.1
E W Brown	92.9	SCL	6.4	3.7
E W Brown	7.1	SLT	6.6	3.8
Eckert Station	4.8	SCL	7.2	4.5
Eckert Station	82.0	SLT	6.9	1.2
Eckert Station	13.2	SNL	6.7	0.5
Edgewater	58.5	SCL	7.3	3.3
Edgewater	3.7	SLT	7.3	1.2
Edgewater	37.8	SNL	6.8	2.2
Elmer W Stout	29.9	SCL	6.7	1.9
Elmer W Stout	56.7	SLT	7.0	1.2
Elmer W Stout	13.3	SNL	6.8	0.8
F B Culley	45.3	SCL	5.9	0.93
F B Culley	48.9	SLT	6.5	2
F B Culley	5.8	SNL	6.9	1.1
Fayette Power Prj	51.9	SCL	7.7	3.8
Fayette Power Prj	35.7	SLT	7.6	1.2
Fayette Power Prj	12.5	SNL	7.1	1
Flint Creek	62.2	SCL	4.9	0.87
Flint Creek	37.8	SLT	5.3	0.69
Fort Martin	45.9	SCL	5.6	1.2
Fort Martin	54.1	SLT	5.2	1.9
Fort Martin	0.04	SNL	4.6	2.5
Frank E Ratts	30.9	SCL	5.8	1.5
Frank E Ratts	58.0	SLT	6.3	1.1
Frank E Ratts	11.1	SNL	7.0	0.73
G G Allen	85.9	SCL	5.3	0.36
G G Allen	11.9	SLT	5.6	1.1
G G Allen	2.2	SNL	5.2	0.28
Gadsden	45.2	SCL	4.8	0.68
Gadsden	46.4	SLT	5.3	1.3
Gadsden	8.5	SNL	5.1	0.97
Gallatin	56.1	SCL	5.6	0.94
Gallatin	43.9	SLT	5.4	0.94
Gen J M Gavin	35.9	SCL	6.0	1.4
Gen J M Gavin	46.1	SLT	5.6	2.1
Gen J M Gavin	18.0	SNL	5.1	1.3

(continued)

Soil Data (continued)

Plant	Percent Composition	Megatexture Code	Average pH	Average % Organic Material
Genoa	14.3	SCL	6.1	2.3
Genoa	64.6	SLT	6.6	1.8
Genoa	21.0	SNL	6.1	0.97
Gibson	55.3	SCL	6.6	1.5
Gibson	43.2	SLT	6.4	1.1
Gibson	1.5	SNL	7.3	0.67
Gorgas	17.0	SCL	4.6	0.42
Gorgas	53.0	SLT	5.1	0.77
Gorgas	30.0	SNL	5.2	0.73
Green River	48.4	SCL	5.9	1
Green River	51.6	SLT	6.0	1.4
Greene County	19.5	SCL	5.1	1.8
Greene County	72.6	SLT	5.2	1.4
Greene County	7.9	SNL	4.9	1.6
H B Robinson	0.1	SCL	5.2	0.75
H B Robinson	32.6	SLT	4.8	1
H B Robinson	67.3	SNL	5.3	0.6
Hammond	54.7	SCL	5.1	0.74
Hammond	33.8	SLT	5.3	1.3
Hammond	11.5	SNL	5.0	0.75
Harllee Branch	54.7	SCL	5.3	0.49
Harllee Branch	15.3	SLT	5.6	0.97
Harllee Branch	30.0	SNL	5.3	0.47
Harrison	48.8	SCL	5.6	1
Harrison	51.2	SLT	5.0	2.1
Hatfield's Ferry	39.3	SCL	5.7	1.8
Hatfield's Ferry	60.4	SLT	5.3	1.6
Hatfield's Ferry	0.3	SNL	4.6	2.5
Hennepin	44.6	SCL	6.4	1.5
Hennepin	38.2	SLT	6.7	1.1
Hennepin	17.2	SNL	7.0	1.3
Heskett	39.9	SCL	8.0	2.1
Heskett	44.1	SLT	7.6	2.4
Heskett	16.0	SNL	7.7	1.9
Holcomb	4.4	SLT	7.9	0.67
Holcomb	95.6	SNL	7.3	0.75
Homer City	11.0	SCL	4.9	2.9
Homer City	84.5	SLT	4.8	1.6
Homer City	4.5	SNL	4.5	2.1

(continued)

Soil Data (continued)

Plant	Percent Composition	Megatexture Code	Average pH	Average % Organic Material
Hoot Lake	3.1	SCL	7.5	5.4
Hoot Lake	38.9	SLT	7.7	2.6
Hoot Lake	58.1	SNL	7.5	1.3
Hugo	55.1	SCL	6.6	1.4
Hugo	35.8	SLT	6.7	1.6
Hugo	9.2	SNL	5.3	0.7
Hunter	90.8	SCL	8.3	0.73
Hunter	3.5	SLT	8.2	2
Hunter	5.7	SNL	8.5	0.75
Huntington	4.5	SCL	8.6	1.5
Huntington	79.5	SLT	8.0	2.4
Huntington	15.9	SNL	8.6	1.3
Intermountain	46.9	SCL	8.6	0.7
Intermountain	8.3	SLT	8.9	0.51
Intermountain	44.8	SNL	8.8	0.44
J H Campbell	5.0	SLT	7.1	1.8
J H Campbell	95.0	SNL	5.9	1.2
J M Stuart	73.5	SCL	6.5	1.6
J M Stuart	24.8	SLT	6.8	2.4
J M Stuart	1.7	SNL	5.5	2
J R Whiting	80.6	SCL	7.1	4.2
J R Whiting	17.1	SLT	7.1	2.1
J R Whiting	2.3	SNL	6.8	2.8
Jack McDonough	58.9	SCL	5.2	0.46
Jack McDonough	7.8	SLT	5.6	1.1
Jack McDonough	33.3	SNL	5.3	0.37
Jack Watson	20.5	SCL	6.7	11
Jack Watson	46.8	SLT	4.8	3
Jack Watson	32.8	SNL	4.9	3.8
James H Miller Jr	17.0	SCL	4.6	0.42
James H Miller Jr	53.0	SLT	5.1	0.77
James H Miller Jr	30.0	SNL	5.2	0.73
Jim Bridger	1.4	SCL	8.7	0.75
Jim Bridger	37.9	SLT	8.6	0.52
Jim Bridger	60.6	SNL	8.2	0.64
John E Amos	35.8	SCL	6.3	1.6
John E Amos	64.2	SLT	5.1	2.2
John Sevier	43.2	SCL	6.2	1.6
John Sevier	56.7	SLT	5.8	1.2

(continued)

Soil Data (continued)

Plant	Percent Composition	Megatexture Code	Average pH	Average % Organic Material
John Sevier	0.2	SNL	5.0	0.67
Johnsonville	39.2	SCL	5.1	1.7
Johnsonville	57.3	SLT	5.2	1.3
Johnsonville	3.5	SNL	4.7	1.5
Joliet 29	52.8	SCL	7.1	2.7
Joliet 29	43.5	SLT	7.0	2.1
Joliet 29	3.7	SNL	7.1	1.8
Keystone	7.7	SCL	4.9	2.8
Keystone	90.1	SLT	4.9	1.4
Keystone	2.2	SNL	4.5	2.2
Killen Station	74.3	SCL	6.0	1.9
Killen Station	24.0	SLT	6.3	2.2
Killen Station	1.8	SNL	6.2	1.7
Kingston	66.7	SCL	5.0	1.2
Kingston	21.0	SLT	5.5	1.7
Kingston	12.3	SNL	5.0	0.67
Kraft	57.1	SCL	7.2	11
Kraft	22.8	SLT	5.0	1.3
Kraft	20.1	SNL	5.0	1.4
L V Sutton	18.0	SCL	6.1	3.9
L V Sutton	32.4	SLT	5.0	3.7
L V Sutton	49.6	SNL	5.0	1.6
Lansing	9.0	SCL	5.8	2.6
Lansing	67.7	SLT	6.8	2.1
Lansing	23.3	SNL	6.2	1.4
Laramie R Station	41.1	SLT	8.1	0.87
Laramie R Station	58.9	SNL	7.9	1.2
Lawrence EC	51.5	SCL	6.6	1.9
Lawrence EC	47.7	SLT	6.8	2.9
Lawrence EC	0.8	SNL	7.5	0.75
Lee	16.4	SCL	5.0	1.3
Lee	51.1	SLT	5.0	1.3
Lee	32.5	SNL	5.1	0.96
Leland Olds	13.5	SCL	7.8	2.6
Leland Olds	52.9	SLT	7.6	1.9
Leland Olds	33.6	SNL	7.5	2
Lon Wright	25.7	SCL	7.5	1.5
Lon Wright	8.4	SLT	7.0	2.1
Lon Wright	65.9	SNL	7.8	1.4

(continued)

Soil Data (continued)

Plant	Percent Composition	Megatexture Code	Average pH	Average % Organic Material
Louisa	35.5	SCL	6.7	1.8
Louisa	16.6	SLT	6.3	1.5
Louisa	47.9	SNL	6.6	0.96
Marion	10.9	SCL	5.6	0.96
Marion	88.8	SLT	5.2	0.95
Marion	0.3	SNL	6.6	1
Marshall	72.1	SCL	5.2	0.33
Marshall	12.9	SLT	5.5	0.87
Marshall	15.0	SNL	5.2	0.27
Martin Lake	34.3	SCL	4.9	1
Martin Lake	25.1	SLT	5.1	0.8
Martin Lake	40.6	SNL	5.1	0.73
Mayo	71.9	SCL	5.6	0.61
Mayo	27.9	SLT	5.6	1
Mayo	0.2	SNL	5.2	0.76
Meramec	87.9	SCL	6.4	1.3
Meramec	12.1	SLT	6.5	1.3
Merom	30.2	SCL	5.5	0.84
Merom	59.2	SLT	5.8	0.96
Merom	10.6	SNL	6.4	0.77
Miami Fort	69.6	SCL	6.5	1.7
Miami Fort	27.3	SLT	6.8	2
Miami Fort	3.1	SNL	6.7	1.2
Milton R Young	4.6	SCL	7.6	3.1
Milton R Young	92.9	SLT	7.7	1.5
Milton R Young	2.5	SNL	7.5	1.8
Mitchell - PA	19.1	SCL	5.9	2.1
Mitchell - PA	80.9	SLT	5.5	1.4
Mitchell - WV	39.9	SCL	6.0	1.7
Mitchell - WV	59.9	SLT	5.2	2
Mitchell - WV	0.2	SNL	6.0	1.3
Mohave	29.0	SLT	8.1	0.26
Mohave	71.0	SNL	8.1	0.31
Monroe	38.5	SCL	7.0	3
Monroe	49.5	SLT	7.2	3.1
Monroe	12.0	SNL	6.8	3.5
Morgantown	21.7	SCL	4.6	1.2
Morgantown	39.3	SLT	4.7	3.2
Morgantown	39.0	SNL	4.9	1.3

(continued)

Soil Data (continued)

Plant	Percent Composition	Megatexture Code	Average pH	Average % Organic Material
Mountaineer (1301)	56.1	SCL	6.0	1.6
Mountaineer (1301)	34.2	SLT	5.9	2.2
Mountaineer (1301)	9.8	SNL	4.9	2.5
Mt Storm	4.1	SCL	5.0	2.9
Mt Storm	65.3	SLT	4.7	1.4
Mt Storm	30.6	SNL	4.4	1
Muscatine Plant #1	46.8	SCL	6.6	1.8
Muscatine Plant #1	27.4	SLT	6.4	1.4
Muscatine Plant #1	25.8	SNL	6.6	0.84
Muskogee	30.9	SCL	6.5	1.7
Muskogee	53.1	SLT	6.8	1.1
Muskogee	16.0	SNL	6.7	1
Neal North	36.7	SCL	7.9	1.1
Neal North	46.5	SLT	7.9	0.67
Neal North	16.9	SNL	7.7	0.73
Neal South	34.0	SCL	7.8	1.1
Neal South	50.7	SLT	7.8	0.69
Neal South	15.3	SNL	7.7	0.73
Nebraska City	55.5	SCL	7.4	1.4
Nebraska City	35.5	SLT	7.3	1.7
Nebraska City	9.0	SNL	7.7	0.74
New Castle	5.1	SCL	7.7	0.73
New Castle	81.6	SLT	5.9	2.8
New Castle	13.2	SNL	6.1	1.5
Newton	37.9	SCL	5.5	0.54
Newton	61.3	SLT	5.5	0.53
Newton	0.7	SNL	6.5	0.85
North Omaha	29.0	SCL	7.4	1.5
North Omaha	60.1	SLT	7.7	0.82
North Omaha	11.0	SNL	7.7	0.74
Northeastern	76.9	SCL	6.7	2.1
Northeastern	21.3	SLT	6.3	2.2
Northeastern	1.8	SNL	5.6	2
Nucla	61.2	SLT	7.9	0.98
Nucla	38.8	SNL	8.1	0.55
Oklaunion	92.2	SCL	8.0	1.7
Oklaunion	7.0	SLT	7.9	0.94
Oklaunion	0.7	SNL	7.3	1.5
Paradise	14.8	SCL	5.6	1.4

(continued)

Soil Data (continued)

Plant	Percent Composition	Megatexture Code	Average pH	Average % Organic Material
Paradise	85.2	SLT	5.9	1.2
Petersburg	29.7	SCL	5.9	1.5
Petersburg	62.9	SLT	6.3	1.2
Petersburg	7.5	SNL	7.2	0.59
Pleasant Prairie	97.2	SCL	7.1	1.7
Pleasant Prairie	2.8	SNL	7.3	1.5
Port Washington	86.3	SCL	7.3	3.3
Port Washington	7.7	SLT	7.5	0.68
Port Washington	6.1	SNL	7.3	3
Portland	8.7	SCL	5.8	0.58
Portland	90.8	SLT	5.5	1.1
Portland	0.5	SNL	6.0	1.8
Possum Point	6.3	SCL	4.6	0.58
Possum Point	43.0	SLT	4.9	3
Possum Point	50.7	SNL	4.9	0.8
Potomac River	13.3	SCL	4.5	0.56
Potomac River	35.5	SLT	4.9	2.8
Potomac River	51.2	SNL	5.0	1.1
Presque Isle	18.7	SLT	5.2	2.5
Presque Isle	81.3	SNL	5.3	3.1
R Gallagher	40.4	SCL	5.6	1.5
R Gallagher	59.0	SLT	5.9	2.1
R Gallagher	0.5	SNL	6.9	1.4
R M Schahfer	2.1	SCL	7.1	3.8
R M Schahfer	6.5	SLT	6.9	2.9
R M Schahfer	91.4	SNL	6.6	1.5
Reid Gardner	13.3	SCL	8.4	0.29
Reid Gardner	21.6	SLT	8.3	0.58
Reid Gardner	65.1	SNL	8.4	0.34
Richard Gorsuch	69.9	SCL	6.1	1.7
Richard Gorsuch	27.0	SLT	5.9	2.4
Richard Gorsuch	3.0	SNL	5.1	2.6
Riverbend	77.4	SCL	5.3	0.37
Riverbend	20.1	SLT	5.7	1.1
Riverbend	2.5	SNL	5.2	0.45
Rodemacher	42.9	SCL	6.5	0.96
Rodemacher	51.4	SLT	6.5	0.92
Rodemacher	5.7	SNL	5.3	0.85
Roxboro	40.3	SCL	5.5	0.47

(continued)

Soil Data (continued)

Plant	Percent Composition	Megatexture Code	Average pH	Average % Organic Material
Roxboro	55.7	SLT	6.0	0.79
Roxboro	4.0	SNL	5.5	1.4
Sandow	0.8	SCL	6.9	0.5
Sandow	37.4	SLT	6.3	0.66
Sandow	61.8	SNL	6.3	0.64
Scherer	58.5	SCL	5.3	0.39
Scherer	12.8	SLT	5.5	0.97
Scherer	28.7	SNL	5.3	0.42
Shawnee	9.5	SCL	5.8	1
Shawnee	84.2	SLT	5.6	1.4
Shawnee	6.3	SNL	6.5	1.1
Shawville	5.2	SCL	5.0	3
Shawville	82.6	SLT	4.9	1.1
Shawville	12.2	SNL	4.4	1.2
Sheldon	62.7	SCL	6.8	2.3
Sheldon	33.2	SLT	7.0	1.6
Sheldon	4.1	SNL	6.9	2
South Oak Creek	95.5	SCL	7.1	1.9
South Oak Creek	4.5	SNL	7.3	1.6
Springerville	10.0	SLT	8.1	0.79
Springerville	90.0	SNL	7.9	0.79
St Johns River Power	27.1	SCL	6.9	49
St Johns River Power	0.4	SLT	5.0	1.3
St Johns River Power	72.5	SNL	5.2	1.1
Stanton Energy Ctr	0.8	SCL	7.0	10
Stanton Energy Ctr	2.4	SLT	7.7	1
Stanton Energy Ctr	96.8	SNL	5.3	4.8
Stockton Cogen Company	89.9	SCL	7.6	1.8
Stockton Cogen Company	6.6	SLT	7.5	1.5
Stockton Cogen Company	3.5	SNL	6.8	0.51
Syl Laskin	8.5	SCL	6.5	3.2
Syl Laskin	4.6	SLT	6.3	6.3
Syl Laskin	86.9	SNL	5.8	3.1
Tecumseh EC	55.2	SCL	6.6	2
Tecumseh EC	41.9	SLT	6.9	2.6
Tecumseh EC	2.9	SNL	7.6	0.62
Texas-New Mexico	4.4	SCL	7.0	0.61
Texas-New Mexico	43.5	SLT	6.3	0.67
Texas-New Mexico	52.1	SNL	6.0	0.77

(continued)

Soil Data (continued)

Plant	Percent Composition	Megatexture Code	Average pH	Average % Organic Material
Titus	31.8	SCL	6.0	0.76
Titus	63.6	SLT	5.6	1.4
Titus	4.6	SNL	5.0	0.98
Trimble County	57.3	SCL	6.3	2
Trimble County	41.9	SLT	6.5	1.9
Trimble County	0.8	SNL	5.9	1.7
Tyrone	92.1	SCL	6.3	3.7
Tyrone	7.9	SLT	6.6	3.9
Valley	98.5	SCL	6.9	1.2
Valley	0.2	SLT	7.5	0.45
Valley	1.3	SNL	7.4	1.3
Vermilion	82.5	SCL	6.9	1.3
Vermilion	16.6	SLT	7.0	1.2
Vermilion	0.8	SNL	7.2	1.1
Victor J Daniel Jr	46.2	SCL	4.6	2.2
Victor J Daniel Jr	27.7	SLT	4.7	2.3
Victor J Daniel Jr	26.1	SNL	4.7	16
W A Parish	95.8	SCL	7.4	1.4
W A Parish	4.2	SLT	7.9	0.74
W H Weatherspoon	7.4	SCL	5.5	1.9
W H Weatherspoon	50.4	SLT	4.7	2.2
W H Weatherspoon	42.2	SNL	4.8	1.3
W S Lee	68.0	SCL	5.3	0.48
W S Lee	9.0	SLT	5.7	1
W S Lee	23.0	SNL	5.3	0.41
Wabash River	22.0	SCL	6.4	1.6
Wabash River	48.5	SLT	6.9	1.2
Wabash River	29.5	SNL	6.7	1.2
Walter C Beckjord	71.6	SCL	6.3	1.4
Walter C Beckjord	26.5	SLT	6.7	2
Walter C Beckjord	1.9	SNL	6.6	1.1
Wansley	46.3	SCL	5.2	0.52
Wansley	18.1	SLT	5.6	1.2
Wansley	35.5	SNL	5.4	0.5
Warrick	45.8	SCL	6.0	0.95
Warrick	48.6	SLT	6.5	1.9
Warrick	5.6	SNL	7.0	1.1
Waukegan	43.9	SCL	6.6	1
Waukegan	18.1	SLT	6.6	1.4

(continued)

Soil Data (continued)

Plant	Percent Composition	Megatexture Code	Average pH	Average % Organic Material
Waukegan	38.0	SNL	6.7	0.8
Weston	33.5	SLT	5.6	1.7
Weston	66.5	SNL	6.0	1.4
Widows Creek	64.5	SCL	5.3	0.88
Widows Creek	20.0	SLT	5.2	1.4
Widows Creek	15.5	SNL	5.4	1.2
Will County	40.0	SCL	6.8	1.8
Will County	52.7	SLT	7.0	0.96
Will County	7.2	SNL	7.1	0.98
Wyodak	1.3	SCL	8.1	0.38
Wyodak	40.2	SLT	7.9	1.1
Wyodak	58.5	SNL	7.9	0.93
Yates	47.8	SCL	5.2	0.48
Yates	17.7	SLT	5.6	1.2
Yates	34.5	SNL	5.3	0.48

Attachment C-2: Hydrogeologic Environment

Plant	Hydrogeologic Setting		Hydrogeologic Environment		Percentage	Comment
	Code	Description	Code	Description		
Big Cajun 2	10Ba	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	100	Predominant alluvial setting (100% alluvium); soils have significant fines (SCL+SLT = 95%)
A B Brown	6Fa	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	100	Predominant alluvial setting; soils have significant fines (SCL+SLT = 95%)
A/C Power-Ace Operations	2C	Alluvial Fans	5	Alluvial Basins Valleys and Fans	100	Based on surficial geology; consistent with alluvial fan setting
Allen	10Ba	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	100	Setting based on aquifer coverages, surficial geology; Heath (1985) and soils indicate overbank deposits
Alma	7Ea	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	50	Percentage based on SNL/SCL soils; setting based on productive aquifers and surficial geology
Alma	7Eb	River Alluvium Without Overbank Deposits	7	River Valleys and Floodplains without Overbank Deposits	50	Percentage based on SNL/SCL soils; setting based on productive aquifers and surficial geology
Antelope Valley	7G	Thin Till Over Bedded Sedimentary Rock	3	Till Over Sedimentary Rock	100	Based on principal aquifer and surficial geology coverages
Arkwright	8D	Regolith	1	Metamorphic and Igneous	100	Most common Piedmont setting (residuum)
Asheville	8B	Alluvial Mountain Valleys	5	Alluvial Basins Valleys and Fans	100	Appropriate for alluvial blue ridge valley (colluvium)
Baldwin	7Ea	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	70	Percentage based on surficial geology (74% Floodplain and alluvium gravel terraces)
Baldwin	7G	Thin Till Over Bedded Sedimentary Rock	3	Till Over Sedimentary Rock	30	Percentage based on surficial geology (74% Floodplain and alluvium gravel terraces)

(continued)

Hydrogeologic Environment (continued)

Plant	Hydrogeologic Setting		Hydrogeologic Environment		Percentage	Comment
	Code	Description	Code	Description		
Barry	10Ba	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	100	Predominant alluvial setting, significant fine grained soils = overbank deposits
Bay Front	7Bb	Outwash Over Bedded Sedimentary Rock	2	Bedded Sedimentary Rock	70	Percentage based on productive aquifers
Bay Front	7D	Buried Valley	4	Sand and Gravel	30	Percentage based on productive aquifers
Bay Shore	7Ac	Glacial Till Over Solution Limestone	12	Solution Limestone	100	Closest setting considering carbonate aquifers, high SCL soils, and lake deposits surficial geology
Belews Creek	6H	Triassic Basins	2	Bedded Sedimentary Rock	50	Sources somewhat dissimilar; fraction based on surficial geology; Triassic basin
Belews Creek	8D	Regolith	1	Metamorphic and Igneous	50	Sources somewhat dissimilar; fraction based on surficial geology
Ben French	6Da	Alternating Sandstone, Limestone and Shale - Thin Soil	2	Bedded Sedimentary Rock	60	Percentage, thin soils based on surficial geology
Ben French	6Fa	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	40	Percentage based on surficial geology; significant fine soils (25% SCL)
Big Sandy	6Da	Alternating Sandstone, Limestone and Shale - Thin Soil	2	Bedded Sedimentary Rock	50	Percentage based on surficial geology; thin soils inferred from colluvium
Big Sandy	6Fa	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	50	Percentage based on surficial geology; soils have significant fines (SCL+SLT = 95%)
Big Stone	7Ba	Outwash	8	Outwash	100	Based on surficial geology
Black Dog Steam Plant	7Bb	Outwash Over Bedded Sedimentary Rock	2	Bedded Sedimentary Rock	100	Based on surficial geology, aquifer coverages
Blue Valley	7Aa	Glacial Till Over Bedded Sedimentary Rock	3	Till Over Sedimentary Rock	80	Percentage based on Heath (1985), productive aquifers

(continued)

Hydrogeologic Environment (continued)

Plant	Hydrogeologic Setting		Hydrogeologic Environment		Percentage	Comment
	Code	Description	Code	Description		
Blue Valley	7Ea	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	20	Percentage based on Heath (1985), productive aquifers
Bowen	6Db	Alternating Sandstone, Limestone and Shale - Deep Regolith	2	Bedded Sedimentary Rock	100	Based on aquifers, surficial residuum (massive red clay); metamorphic surficial geology not consistent with Valley and Ridge
Brandon Shores	10Ab	Unconsolidated and Semi-Consolidated Shallow Surficial Aquifer	10	Unconsolidated and Semiconsolidated Shallow Aquifers	100	Assigned based on location and aquifer and surficial geology coverages; Heath region incorrect (it's Atlantic Coastal Plain, not Piedmont)
Buck	8E	River Alluvium	6	River Valleys and Floodplains with Overbank Deposit	100	Based on productive aquifer & Heath region coverages
Bull Run	6Db	Alternating Sandstone, Limestone and Shale - Deep Regolith	2	Bedded Sedimentary Rock	60	Percentage based on surficial geology
Bull Run	6Fa	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	40	Percentage based on surficial geology; high SCL (77%) = overbank deposits
C D McIntosh Jr	11A	Solution Limestone and Shallow Surficial Aquifers	12	Solution Limestone	100	Based on both aquifer coverages
C P Crane	10Aa	Regional Aquifers	4	Sand and Gravel	50	Appears to be on border between Piedmont and Coastal Plain
C P Crane	8D	Regolith	1	Metamorphic and Igneous	50	Appears to be on border between Piedmont and Coastal Plain
Cape Fear	6H	Triassic Basins	2	Bedded Sedimentary Rock	20	Percentage based on productive aquifer & Heath region coverages; Triassic basin
Cape Fear	8E	River Alluvium	6	River Valleys and Floodplains with Overbank Deposit	80	Percentage based on productive aquifer & Heath region coverages
Carbon	4B	Consolidated Sedimentary Rock	2	Bedded Sedimentary Rock	100	Setting based on aquifer and surficial geology coverages

(continued)

Hydrogeologic Environment (continued)

Plant	Hydrogeologic Setting		Hydrogeologic Environment		Percentage	Comment
	Code	Description	Code	Description		
Cardinal	6Da	Alternating Sandstone, Limestone and Shale - Thin Soil	2	Bedded Sedimentary Rock	30	Percentage based on surficial geology; soils with low (<1%) SNL
Cardinal	6Fa	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	70	Percentage based on surficial geology; soils with low (<1%) SNL
Cayuga	7D	Buried Valley	4	Sand and Gravel	100	Glaciofluvial aquifer overlaid by alluvial deposits
Chalk Point	10Ab	Unconsolidated and Semi-Consolidated Shallow Surficial Aquifer	10	Unconsolidated and Semiconsolidated Shallow Aquifers	100	Predominant setting
Cholla	4B	Consolidated Sedimentary Rock	2	Bedded Sedimentary Rock	20	Percentage based on surficial geology (83% Floodplain and alluvium gravel terraces)
Cholla	4C	River Alluvium	7	River Valleys and Floodplains without Overbank Deposits	80	Percentage based on surficial geology (83% Floodplain and alluvium gravel terraces)
Cliffside	8D	Regolith	1	Metamorphic and Igneous	100	Based on surficial geology
Clover	6H	Triassic Basins	2	Bedded Sedimentary Rock	20	Percentage based on surficial geology; Triassic Basin from Heath (1985) and principal aquifer coverage
Clover	8E	River Alluvium	6	River Valleys and Floodplains with Overbank Deposit	80	Percentage based on surficial geology
Coal Creek	7G	Thin Till Over Bedded Sedimentary Rock	3	Till Over Sedimentary Rock	100	Based on principal aquifer and surficial geology coverages
Coleto Creek	10Ab	Unconsolidated and Semi-Consolidated Shallow Surficial Aquifer	10	Unconsolidated and Semiconsolidated Shallow Aquifers	100	Setting based on aquifer and surficial geology coverages
Colstrip	6da	Alternating Sandstone, Limestone and Shale - Thin Soil	2	Bedded Sedimentary Rock	100	Based on all coverages

(continued)

Hydrogeologic Environment (continued)

Plant	Hydrogeologic Setting		Hydrogeologic Environment		Percentage	Comment
	Code	Description	Code	Description		
Conemaugh	6Da	Alternating Sandstone, Limestone and Shale - Thin Soil	2	Bedded Sedimentary Rock	100	Setting based on aquifer coverages & Heath (1985); thin regolith inferred from colluvium
Conesville	6Da	Alternating Sandstone, Limestone and Shale - Thin Soil	2	Bedded Sedimentary Rock	40	Percentage based on surficial geology; soils with low (10%) SNL
Conesville	6Fa	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	60	Percentage based on surficial geology; soils with low (10%) SNL
Council Bluffs	7Ea	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	100	Setting based on productive aquifers
Crawford	7Ea	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	100	Assigned based on predominant surficial geology (98% Floodplain and alluvium gravel terraces), productive aquifer coverage
Crist	10Bb	River Alluvium Without Overbank Deposits	7	River Valleys and Floodplains without Overbank Deposits	100	Assigned based on predominant surficial geology (96% Floodplain and alluvium gravel terraces), coarse-grained soil (49% SNL)
Cross	10Ab	Unconsolidated and Semi-Consolidated Shallow Surficial Aquifer	10	Unconsolidated and Semiconsolidated Shallow Aquifers	100	Setting based on aquifers, surficial geology, soils, Heath (1985)
Cumberland	6Fa	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	100	Setting based on surface geology; high (61%) SCL = overbank deposits
Dale	6E	Solution Limestone	12	Solution Limestone	20	Percentage based on surficial geology; setting from principal aquifers (carbonate)
Dale	6Fa	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	80	Percentage based on surficial geology; soils have significant fines (SNL = 0.1%)
Dallman	7Aa	Glacial Till Over Bedded Sedimentary Rock	3	Till Over Sedimentary Rock	100	Based on soils, surficial geology, principal aquifer
Dan E Karn	7F	Glacial Lake Deposits	4	Sand and Gravel	100	Based on surficial geology, soils

(continued)

Hydrogeologic Environment (continued)

Plant	Hydrogeologic Setting		Hydrogeologic Environment		Percentage	Comment
	Code	Description	Code	Description		
Dan River	6H	Triassic Basins	2	Bedded Sedimentary Rock	100	Based on surfucial geology, principal aquifers; Triassic basin
Danskammer	7D	Buried Valley	4	Sand and Gravel	100	Based on predominant Heath region, productive aquifers; little coarse-grained soils
Dave Johnston	4C	River Alluvium	7	River Valleys and Floodplains without Overbank Deposits	100	Based on aquifer and surficial geology coverages, Heath (1985)
Dickerson	8D	Regolith	1	Metamorphic and Igneous	100	Predominant setting
Dolet Hills	10Ab	Unconsolidated and Semi-Consolidated Shallow Surficial Aquifer	10	Unconsolidated and Semiconsolidated Shallow Aquifers	100	Predominant shallow unconsolidated aquifer system
Duck Creek	7Ea	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	100	Assigned based on predominant surficial geology (100% Floodplain and alluvium gravel terraces), Heath Alluvial Valley Region
Dunkirk	7H	Beaches, Beach Ridges and Sand Dunes	11	Coastal Beaches	100	Based on location, surficial geology
E D Edwards	7Aa	Glacial Till Over Bedded Sedimentary Rock	3	Till Over Sedimentary Rock	20	Percentage based on surficial geology (83% Floodplain and alluvium gravel terraces)
E D Edwards	7Ea	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	80	Percentage based on surficial geology (83% Floodplain and alluvium gravel terraces)
E W Brown	6E	Solution Limestone	12	Solution Limestone	20	Percentage based on surficial geology (76% alluvium, 23% clay); soils have significant fine-grained (0% SNL)
E W Brown	6Fa	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	80	Percentage based on surficial geology (76% alluvium, 23% clay); soils have significant fine-grained (0% SNL)
Eckert Station	7Bb	Outwash Over Bedded Sedimentary Rock	2	Bedded Sedimentary Rock	30	Percentage based on productive aquifer coverage, Heath regions

(continued)

Hydrogeologic Environment (continued)

Plant	Hydrogeologic Setting		Hydrogeologic Environment		Percentage	Comment
	Code	Description	Code	Description		
Eckert Station	7Eb	River Alluvium Without Overbank Deposits	7	River Valleys and Floodplains without Overbank Deposits	70	Percentage based on productive aquifer coverage, Heath regions
Edgewater	7Bc	Outwash Over Solution Limestone	12	Solution Limestone	100	Setting based on aquifer and surficial geology coverages
Elmer W Stout	7D	Buried Valley	4	Sand and Gravel	100	Glaciofluvial aquifer overlaid by alluvial deposits
F B Culley	6Fa	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	100	Predominant alluvial setting; soils have significant fines (SCL+SLT = 94%)
Fayette Power Prj	10Ab	Unconsolidated and Semi-Consolidated Shallow Surficial Aquifer	10	Unconsolidated and Semiconsolidated Shallow Aquifers	100	Setting based on aquifer and surficial geology coverages
Flint Creek	6Db	Alternating Sandstone, Limestone and Shale - Deep Regolith	2	Bedded Sedimentary Rock	100	Ozark plateau; Heath (1985) indicates dolomite, sandy dolomite, sandstone, with no indication of solutioning. Surficial geology (cherty red clay) noted as thick regolith in Aller et al. (1987)
Fort Martin	6Fa	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	100	Setting based on surficial geology; low SNL (< 1%) = overbank deposits
Frank E Ratts	7D	Buried Valley	4	Sand and Gravel	100	Glaciofluvial aquifer in alluvial valley region (99%)
G G Allen	8D	Regolith	1	Metamorphic and Igneous	100	Based on surficial geology
Gadsden	6Db	Alternating Sandstone, Limestone and Shale - Deep Regolith	2	Bedded Sedimentary Rock	30	Percentage assigned based on productive aquifer coverage
Gadsden	6Fa	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	70	Percentage assigned based on productive aquifer coverage; soils have significant fines (SCL+SLT > 25%)
Gallatin	6Fa	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	100	Setting based on surface geology; high (56%) SCL = overbank deposits

(continued)

Hydrogeologic Environment (continued)

Plant	Hydrogeologic Setting		Hydrogeologic Environment		Percentage	Comment
	Code	Description	Code	Description		
Gen J M Gavin	6Fa	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	100	Setting based on productive aquifers, surficial geology
Genoa	6Fa	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	50	Percentage based on SNL/SCL soils; setting based on surficial geology and productive aquifers
Genoa	6Fb	River Alluvium Without Overbank Deposits	7	River Valleys and Floodplains without Overbank Deposits	50	Percentage based on SNL/SCL soils; setting based on surficial geology and productive aquifers
Gibson	7Ea	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	100	Predominant alluvial setting; soils have significant fines (SCL+SLT = 99%)
Gorgas	6Da	Alternating Sandstone, Limestone and Shale - Thin Soil	2	Bedded Sedimentary Rock	30	Percentage based on surficial geology; alluvial setting with coarser soils (= no overbank deposits)
Gorgas	6Fb	River Alluvium Without Overbank Deposits	7	River Valleys and Floodplains without Overbank Deposits	70	Percentage based on surficial geology; alluvial setting with coarser soils (= no overbank deposits)
Green River	6Fa	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	100	Predominant alluvial setting (>85% alluvium); soils have significant fines (SNL = 0%)
Greene County	10Ab	Unconsolidated and Semi-Consolidated Shallow Surficial Aquifer	10	Unconsolidated and Semiconsolidated Shallow Aquifers	30	Percentage based on surficial geology; soils have significant fines (SCL+SLT > 90%)
Greene County	10Ba	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	70	Percentage based on surficial geology; soils have significant fines (SCL+SLT > 90%)
H B Robinson	10Ab	Unconsolidated and Semi-Consolidated Shallow Surficial Aquifer	10	Unconsolidated and Semiconsolidated Shallow Aquifers	100	Setting based on aquifers, surficial geology, soils, Heath (1985); Heath region coverage incorrect (Coastal Plain, not Piedmont)

(continued)

Hydrogeologic Environment (continued)

Plant	Hydrogeologic Setting		Hydrogeologic Environment		Percentage	Comment
	Code	Description	Code	Description		
Hammond	6Db	Alternating Sandstone, Limestone and Shale - Deep Regolith	2	Bedded Sedimentary Rock	100	Based on aquifers, surficial residuum (massive red clay)
Harllee Branch	8E	River Alluvium	6	River Valleys and Floodplains with Overbank Deposit	100	Assigned based on predominant surficial geology (99% floodplain and alluvium gravel terraces)
Harrison	6Da	Alternating Sandstone, Limestone and Shale - Thin Soil	2	Bedded Sedimentary Rock	20	Percentage based on surficial geology; thin soils inferred from surficial geology
Harrison	6Fa	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	80	Percentage based on surficial geology; 0%SNL = overbank deposits
Hatfield's Ferry	6Da	Alternating Sandstone, Limestone and Shale - Thin Soil	2	Bedded Sedimentary Rock	40	Percentage based on surficial geology; thin regolith inferred from colluvium
Hatfield's Ferry	6Fa	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	60	Percentage based on surficial geology; soils < 1% SNL
Hennepin	7Aa	Glacial Till Over Bedded Sedimentary Rock	3	Till Over Sedimentary Rock	30	Percentage to capture uncertainty in soils, surficial geology, principal aquifer
Hennepin	7Bb	Outwash Over Bedded Sedimentary Rock	2	Bedded Sedimentary Rock	30	Percentage to capture uncertainty in soils, surficial geology, principal aquifer
Hennepin	7Ea	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	40	Percentage to capture uncertainty in soils, surficial geology, principal aquifer
Heskett	7Ea	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	100	Predominant alluvium surficial geology(96%); mixed soils
Holcomb	5Gb	River Alluvium Without Overbank Deposits	7	River Valleys and Floodplains without Overbank Deposits	100	Alluvial valley with very coarse soils
Homer City	6Da	Alternating Sandstone, Limestone and Shale - Thin Soil	2	Bedded Sedimentary Rock	100	Setting based on aquifer coverages & Heath (1985); thin regolith inferred from colluvium

(continued)

Hydrogeologic Environment (continued)

Plant	Hydrogeologic Setting		Hydrogeologic Environment		Percentage	Comment
	Code	Description	Code	Description		
Hoot Lake	9E	Outwash	8	Outwash	100	Based on productive aquifer, soils, surficial geology
Hugo	6Da	Alternating Sandstone, Limestone and Shale - Thin Soil	2	Bedded Sedimentary Rock	40	Percentage based on surficial geology; soil/regolith thickness inferred from Heath (1985)
Hugo	6Fa	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	60	Percentage based on surficial geology; fine soils with about 10% SNL
Hunter	4B	Consolidated Sedimentary Rock	2	Bedded Sedimentary Rock	100	Setting based on aquifer and surficial geology coverages
Huntington	4B	Consolidated Sedimentary Rock	2	Bedded Sedimentary Rock	100	Setting based on aquifer and surficial geology coverages
Intermountain	2E	Playa Lakes	5	Alluvial Basins Valleys and Fans	100	Setting based on surficial geology coverage, Heath (1985)
J H Campbell	7F	Glacial Lake Deposits	4	Sand and Gravel	100	Based on surficial geology, soils
J M Stuart	6E	Solution Limestone	12	Solution Limestone	50	Percentage based on surficial geology; low (< 2%) SNL
J M Stuart	6Fa	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	50	Percentage based on surficial geology; low (< 2%) SNL
J R Whiting	7F	Glacial Lake Deposits	4	Sand and Gravel	100	Based on surficial geology
Jack McDonough	8C	Mountain Flanks	2	Bedded Sedimentary Rock	100	Assigned based on predominant surficial geology (94% stony colluvium on metamorphic rocks; less silt and clay than in colluvium over limestone)
Jack Watson	10Ab	Unconsolidated and Semi-Consolidated Shallow Surficial Aquifer	10	Unconsolidated and Semiconsolidated Shallow Aquifers	100	Based on all coverages
James H Miller Jr	6Da	Alternating Sandstone, Limestone and Shale - Thin Soil	2	Bedded Sedimentary Rock	20	Percentage based on surficial geology; soils have significant fines (SCL+SLT > 25%)

(continued)

Hydrogeologic Environment (continued)

Plant	Hydrogeologic Setting		Hydrogeologic Environment		Percentage	Comment
	Code	Description	Code	Description		
James H Miller Jr	6Fa	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	80	Percentage based on surficial geology; soils have significant fines (SCL+SLT > 25%)
Jim Bridger	4B	Consolidated Sedimentary Rock	2	Bedded Sedimentary Rock	100	Based on aquifer and surficial geology coverages, Heath (1985)
John E Amos	6da	Alternating Sandstone, Limestone and Shale - Thin Soil	2	Bedded Sedimentary Rock	40	Percentage based on surficial geology; thin soils inferred from surficial geology
John E Amos	6Fa	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	60	Percentage based on surficial geology; 0%SNL = overbank deposits
John Sevier	6E	Solution Limestone	12	Solution Limestone	50	Percentage based on surface geology; setting based on surface geology and aquifer type, with possibility of solution limestone from Heath (1985)
John Sevier	6Fa	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	50	Percentage, setting based on surface geology; low (<1%) SNL = overbank deposits
Johnsonville	6E	Solution Limestone	12	Solution Limestone	30	Percentage based on surface geology; setting based on aquifer coverages, Heath (1985); placed in Nonglaciated Central region based on aquifer coverages and Heath (1985)
Johnsonville	6Fa	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	70	Percentage, setting based on surface geology; low (3%) SNL = overbank deposits; placed in Nonglaciated Central region based on aquifer coverages and Heath (1985)
Joliet 29	7Aa	Glacial Till Over Bedded Sedimentary Rock	3	Till Over Sedimentary Rock	100	Based on aquifers, soils; soils don't suggest outwash like surficial geology does
Keystone	6Da	Alternating Sandstone, Limestone and Shale - Thin Soil	2	Bedded Sedimentary Rock	100	Setting based on aquifer coverages & Heath (1985); thin regolith inferred from colluvium
Killen Station	6E	Solution Limestone	12	Solution Limestone	30	Percentage based on surficial geology; low (< 2%) SNL

(continued)

Hydrogeologic Environment (continued)

Plant	Hydrogeologic Setting		Hydrogeologic Environment		Percentage	Comment
	Code	Description	Code	Description		
Killen Station	6Fa	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	70	Percentage based on surficial geology; low (< 2%) SNL
Kingston	6E	Solution Limestone	12	Solution Limestone	20	Percentage based on surface geology; setting based on surface geology and aquifer type, with possibility of solution limestone from Heath (1985)
Kingston	6Fa	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	80	Percentage, setting based on surface geology; high (67 %) SCL = overbank deposits
Kraft	11A	Solution Limestone and Shallow Surficial Aquifers	12	Solution Limestone	100	Only possible assignment; predominant alluvium (84%) not well represented
L V Sutton	10Ab	Unconsolidated and Semi-Consolidated Shallow Surficial Aquifer	10	Unconsolidated and Semiconsolidated Shallow Aquifers	20	Percentage based on surficial geology; sandy soils
L V Sutton	10Bb	River Alluvium Without Overbank Deposits	7	River Valleys and Floodplains without Overbank Deposits	80	Percentage based on surficial geology; sandy soils
Lansing	6Da	Alternating Sandstone, Limestone and Shale - Thin Soil	2	Bedded Sedimentary Rock	40	Percentage based on surficial geology, productive aquifers; loess = thin soils
Lansing	6Fb	River Alluvium Without Overbank Deposits	7	River Valleys and Floodplains without Overbank Deposits	60	Percentage based on surficial geology, productive aquifers; coarse-grained soils
Laramie R Station	6Fb	River Alluvium Without Overbank Deposits	7	River Valleys and Floodplains without Overbank Deposits	100	Based on aquifer and surficial geology coverages, Heath (1985)
Lawrence EC	7Ea	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	100	Alluvial valley with low coarse soils (<1% SNL)
Lee	10Ab	Unconsolidated and Semi-Consolidated Shallow Surficial Aquifer	10	Unconsolidated and Semiconsolidated Shallow Aquifers	30	Percentage based on surficial geology; sandy soils
Lee	10Bb	River Alluvium Without Overbank Deposits	7	River Valleys and Floodplains without Overbank Deposits	70	Percentage based on surficial geology; sandy soils

(continued)

Hydrogeologic Environment (continued)

Plant	Hydrogeologic Setting		Hydrogeologic Environment		Percentage	Comment
	Code	Description	Code	Description		
Leland Olds	7Eb	River Alluvium Without Overbank Deposits	7	River Valleys and Floodplains without Overbank Deposits	50	Percentage based on surficial geology; assumed coarse soils
Leland Olds	7G	Thin Till Over Bedded Sedimentary Rock	3	Till Over Sedimentary Rock	50	Percentage based on surficial geology; assumed coarse soils
Lon Wright	7Ea	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	30	Alluvial based on predominant Heath, productive aquifer; percentage based on soil textures
Lon Wright	7Eb	River Alluvium Without Overbank Deposits	7	River Valleys and Floodplains without Overbank Deposits	70	Alluvial based on predominant Heath, productive aquifer; percentage based on soil textures
Louisa	7Ea	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	50	Alluvial Valley; significant coarse-grained deposits
Louisa	7Eb	River Alluvium Without Overbank Deposits	7	River Valleys and Floodplains without Overbank Deposits	50	Alluvial Valley; significant coarse-grained deposits
Marion	7Aa	Glacial Till Over Bedded Sedimentary Rock	3	Till Over Sedimentary Rock	100	Assigned to Glaciated Central region based on surficial geology (pre-Wisconsin drift)
Marshall	8D	Regolith	1	Metamorphic and Igneous	100	Based on surficial geology
Martin Lake	10Ab	Unconsolidated and Semi-Consolidated Shallow Surficial Aquifer	10	Unconsolidated and Semiconsolidated Shallow Aquifers	100	Setting based on aquifer and surficial geology coverages
Mayo	8D	Regolith	1	Metamorphic and Igneous	100	Based on surficial geology
Meramec	7Ea	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	100	Based on surficial, predominant Heath
Merom	7D	Buried Valley	4	Sand and Gravel	100	Glaciofluvial aquifer overlaid by alluvial deposits
Miami Fort	7Ea	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	100	Assigned based on productive aquifers, surficial geology and soil (3% SNL)

(continued)

Hydrogeologic Environment (continued)

Plant	Hydrogeologic Setting		Hydrogeologic Environment		Percentage	Comment
	Code	Description	Code	Description		
Milton R Young	7G	Thin Till Over Bedded Sedimentary Rock	3	Till Over Sedimentary Rock	100	Based on principal aquifer and surficial geology coverages
Mitchell	6Da	Alternating Sandstone, Limestone and Shale - Thin Soil	2	Bedded Sedimentary Rock	60	Percentage based on surficial geology; thin regolith inferred from colluvium
Mitchell	6Fa	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	40	Percentage based on surficial geology; soils 0 % SNL
Mitchell	6Fa	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	100	Setting based on surficial geology; low SNL (< 1%) = overbank deposits
Mohave	2Ha	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	100	Setting based on predominant surficial geology, Heath (1985)
Monroe	7Ea	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	100	Based on Heath region, productive aquifers, soils
Morgantown	10Ab	Unconsolidated and Semi-Consolidated Shallow Surficial Aquifer	10	Unconsolidated and Semiconsolidated Shallow Aquifers	100	Assigned based on location and aquifer and surficial geology coverages; Heath region incorrect (it's Atlantic Coastal Plain, not Piedmont)
Mountaineer (1301)	6Fa	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	100	Setting based on surficial geology; low SNL (10%) = overbank deposits
Mt Storm	6Da	Alternating Sandstone, Limestone and Shale - Thin Soil	2	Bedded Sedimentary Rock	100	Setting based on surficial geology, aquifer coverages; thin soils inferred from surficial geology
Muscatine Plant #1	7Ea	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	50	Alluvial Valley; significant coarse-grained deposits
Muscatine Plant #1	7Eb	River Alluvium Without Overbank Deposits	7	River Valleys and Floodplains without Overbank Deposits	50	Alluvial Valley; significant coarse-grained deposits
Muskogee	6Fa	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	100	Surficial geology indicates alluvium/colluvium; Heath (1985) indicates fine soils over sands and gravels

(continued)

Hydrogeologic Environment (continued)

Plant	Hydrogeologic Setting		Hydrogeologic Environment		Percentage	Comment
	Code	Description	Code	Description		
Neal North	7Ea	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	100	Alluvial Valley setting
Neal South	7Ea	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	100	Alluvial Valley setting
Nebraska City	7Ea	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	100	Alluvial based on predominant Heath, productive aquifer, soil textures
New Castle	7Aa	Glacial Till Over Bedded Sedimentary Rock	3	Till Over Sedimentary Rock	20	Percentage and setting based on Heath region & surficial geology; thin regolith inferred from colluvium
New Castle	7D	Buried Valley	4	Sand and Gravel	80	Percentage and setting based on Heath region & book
Newton	7Aa	Glacial Till Over Bedded Sedimentary Rock	3	Till Over Sedimentary Rock	100	Based on soils, surficial geology, aquifer coverages
North Omaha	7Ea	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	100	Alluvial based on predominant Heath, productive aquifer; soil texture (28% SCL, 10% SNL) = overbank deposits
Northeastern	6Da	Alternating Sandstone, Limestone and Shale - Thin Soil	2	Bedded Sedimentary Rock	40	Percentage based on surficial geology, which indicates thin residual soils
Northeastern	6Fa	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	60	Percentage based on surficial geology; soils < 2% SNL
Nucla	4B	Consolidated Sedimentary Rock	2	Bedded Sedimentary Rock	100	Based on surficial geology, aquifer coverages
Oklaunion	6Da	Alternating Sandstone, Limestone and Shale - Thin Soil	2	Bedded Sedimentary Rock	100	Setting based on surficial geology; thin soil inferred
Paradise	6Fa	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	100	Predominant alluvial setting (93% alluvium); soils have significant fines (SNL = 0%)
Petersburg	7D	Buried Valley	4	Sand and Gravel	100	Glaciofluvial aquifer in alluvial valley region (similar to 1043)

(continued)

Hydrogeologic Environment (continued)

Plant	Hydrogeologic Setting		Hydrogeologic Environment		Percentage	Comment
	Code	Description	Code	Description		
Pleasant Prairie	7Ac	Glacial Till Over Solution Limestone	12	Solution Limestone	100	Setting based on aquifer and soil coverages (high SCL soils)
Port Washington	7Ac	Glacial Till Over Solution Limestone	12	Solution Limestone	100	Setting based on aquifer and soil coverages (high SCL soils)
Portland	7Ac	Glacial Till Over Solution Limestone	12	Solution Limestone	100	Setting based on aquifer and surficial geology coverage
Possum Point	10Ab	Unconsolidated and Semi-Consolidated Shallow Surficial Aquifer	10	Unconsolidated and Semiconsolidated Shallow Aquifers	100	Based on productive aquifer coverage; Heath region incorrect
Potomac River	10Ab	Unconsolidated and Semi-Consolidated Shallow Surficial Aquifer	10	Unconsolidated and Semiconsolidated Shallow Aquifers	50	Percentage based on surficial geology coverage; Heath region incorrect
Potomac River	10Bb	River Alluvium Without Overbank Deposits	7	River Valleys and Floodplains without Overbank Deposits	50	Percentage based on surficial geology coverage; sandy soils (51% SNL) = no overbank deposits; Heath region incorrect
Presque Isle	9F	Moraine	4	Sand and Gravel	100	Based on surficial geology, Heath region, soils
R Gallagher	6Fa	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	100	Predominant alluvial setting; soils have significant fines (SCL+SLT = 99%)
R M Schahfer	7D	Buried Valley	4	Sand and Gravel	100	Glaciofluvial aquifer in alluvial valley region
Reid Gardner	2C	Alluvial Fans	5	Alluvial Basins Valleys and Fans	100	Based on surficial geology; consistent with productive aquifers
Richard Gorsuch	6Fa	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	100	Assigned based on productive aquifers, surficial geology and soil (3% SNL)
Riverbend	8D	Regolith	1	Metamorphic and Igneous	100	Based on surficial geology
Rodemacher	10Ab	Unconsolidated and Semi-Consolidated Shallow Surficial Aquifer	10	Unconsolidated and Semiconsolidated Shallow Aquifers	50	Setting percentage determined from Heath, productive aquifer, and surficial geology coverages

(continued)

Hydrogeologic Environment (continued)

Plant	Hydrogeologic Setting		Hydrogeologic Environment		Percentage	Comment
	Code	Description	Code	Description		
Rodemacher	10Ba	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	50	Setting percentage determined from Heath, productive aquifer, and surficial geology coverages
Roxboro	8D	Regolith	1	Metamorphic and Igneous	100	Based on surficial geology, productive aquifers
Sandow	10Ab	Unconsolidated and Semi-Consolidated Shallow Surficial Aquifer	10	Unconsolidated and Semiconsolidated Shallow Aquifers	100	Setting based on aquifer and surficial geology coverages; Heath region coverage is incorrect (based on Heath [1985] and aquifer coverages)
Scherer	8D	Regolith	1	Metamorphic and Igneous	100	Most common Piedmont setting (residuum)
Shawnee	10Bb	River Alluvium Without Overbank Deposits	7	River Valleys and Floodplains without Overbank Deposits	100	Predominant alluvial setting (100% alluvium); soils have low fines (SCL = 9%)
Shawville	6Da	Alternating Sandstone, Limestone and Shale - Thin Soil	2	Bedded Sedimentary Rock	100	Setting based on aquifer coverages & Heath (1985); thin regolith inferred from colluvium
Sheldon	7Aa	Glacial Till Over Bedded Sedimentary Rock	3	Till Over Sedimentary Rock	30	Percentage based on productive aquifer coverage; buried valley indicated by Heath (1985)
Sheldon	7D	Buried Valley	4	Sand and Gravel	70	Percentage based on productive aquifer coverage; buried valley indicated by Heath (1985)
South Oak Creek	7Ac	Glacial Till Over Solution Limestone	12	Solution Limestone	100	Setting based on aquifer and soil coverages (high SCL soils)
Springerville	4B	Consolidated Sedimentary Rock	2	Bedded Sedimentary Rock	100	Assigned based on productive aquifers (consolidated sandstone)
St Johns River Power	11B	Coastal Deposits	4	Sand and Gravel	100	Based on sea island surficial geology
Stanton Energy Ctr	11A	Solution Limestone and Shallow Surficial Aquifers	12	Solution Limestone	100	Based on both aquifer coverages

(continued)

Hydrogeologic Environment (continued)

Plant	Hydrogeologic Setting		Hydrogeologic Environment		Percentage	Comment
	Code	Description	Code	Description		
Stockton Cogen Company	2C	Alluvial Fans	5	Alluvial Basins Valleys and Fans	50	Percentage based on surficial geology; Central Valley soils show significant fines
Stockton Cogen Company	2Ha	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	50	Percentage based on surficial geology; Central Valley soils show significant fines
Syl Laskin	9E	Outwash	8	Outwash	60	Percentage based on surficial geology
Syl Laskin	9Ga	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	40	Percentage based on surficial geology
Tecumseh EC	7Ea	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	100	Alluvial valley with low coarse soils (<3% SNL)
Texas-New Mexico	10Ab	Unconsolidated and Semi-Consolidated Shallow Surficial Aquifer	10	Unconsolidated and Semiconsolidated Shallow Aquifers	100	Based on productive aquifers, Heath (1985) (Heath region coverage is incorrect)
Titus	6Db	Alternating Sandstone, Limestone and Shale - Deep Regolith	2	Bedded Sedimentary Rock	100	Setting based on aquifer and surficial geology coverage; deep regolith inferred from red, massive clay
Trimble County	6E	Solution Limestone	12	Solution Limestone	40	Heath incorrect; Percentage based on surficial geology (56% alluvium, 44% clay); soils have significant fine-grained (1% SNL)
Trimble County	6Fa	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	60	Heath incorrect; Percentage based on surficial geology (56% alluvium, 44% clay); soils have significant fine-grained (1% SNL)
Tyrone	6E	Solution Limestone	12	Solution Limestone	100	Based on principal aquifer coverage
Valley	7Ac	Glacial Till Over Solution Limestone	12	Solution Limestone	100	Setting based on aquifer and soil coverages (high SCL soils)
Vermilion	7Aa	Glacial Till Over Bedded Sedimentary Rock	3	Till Over Sedimentary Rock	100	Based on aquifers, soils; soils don't suggest outwash like surficial geology does
Victor J Daniel Jr	10Ab	Unconsolidated and Semi-Consolidated Shallow Surficial Aquifer	10	Unconsolidated and Semiconsolidated Shallow Aquifers	60	Percentage based on surficial geology

(continued)

Hydrogeologic Environment (continued)

Plant	Hydrogeologic Setting		Hydrogeologic Environment		Percentage	Comment
	Code	Description	Code	Description		
Victor J Daniel Jr	10Ba	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	40	Percentage based on surficial geology, soils
W A Parish	10Ab	Unconsolidated and Semi-Consolidated Shallow Surficial Aquifer	10	Unconsolidated and Semiconsolidated Shallow Aquifers	30	Percentage based on surficial geology and productive aquifer coverages
W A Parish	10Ba	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	70	Percentage based on surficial geology and productive aquifer coverages; high SCL (96%) = overbank deposits
W H Weatherspoon	10Ab	Unconsolidated and Semi-Consolidated Shallow Surficial Aquifer	10	Unconsolidated and Semiconsolidated Shallow Aquifers	30	Percentage based on surficial geology; sandy soils
W H Weatherspoon	10Bb	River Alluvium Without Overbank Deposits	7	River Valleys and Floodplains without Overbank Deposits	70	Percentage based on surficial geology; sandy soils
W S Lee	8D	Regolith	1	Metamorphic and Igneous	100	Setting based on aquifers, surficial geology, soils, Heath (1985)
Wabash River	7D	Buried Valley	4	Sand and Gravel	100	Glaciofluvial aquifer in Alluvial Valley region
Walter C Beckjord	7Aa	Glacial Till Over Bedded Sedimentary Rock	3	Till Over Sedimentary Rock	60	Percentage based on surficial geology; placed in glaciated central based on Heath (1985); soils 2% SNL
Walter C Beckjord	7Ea	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	40	Percentage based on surficial geology; placed in glaciated central based on Heath (1985); soils 2% SNL
Wansley	8C	Mountain Flanks	2	Bedded Sedimentary Rock	30	Percentage based on surficial geology
Wansley	8E	River Alluvium	6	River Valleys and Floodplains with Overbank Deposit	70	Percentage based on surficial geology
Warrick	6Fa	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	100	Predominant alluvial setting; soils have significant fines (SCL+SLT = 94%)
Waukegan	7Bc	Outwash Over Solution Limestone	12	Solution Limestone	100	Based on soils, surficial geology, aquifer coverages

(continued)

Hydrogeologic Environment (continued)

Plant	Hydrogeologic Setting		Hydrogeologic Environment		Percentage	Comment
	Code	Description	Code	Description		
Weston	9E	Outwash	8	Outwash	100	Setting based on productive aquifer, surficial geology coverages
Widows Creek	6Da	Alternating Sandstone, Limestone and Shale - Thin Soil	2	Bedded Sedimentary Rock	20	Percentage based on surficial geology; thin soils inferred from colluvium
Widows Creek	6Fa	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	80	Percentage based on surficial geology; soils have significant fines (SCL+SLT > 25%)
Will County	7Aa	Glacial Till Over Bedded Sedimentary Rock	3	Till Over Sedimentary Rock	40	Percentage based on surficial geology (65% Floodplain and alluvium gravel terraces)
Will County	7Ea	River Alluvium With Overbank Deposits	6	River Valleys and Floodplains with Overbank Deposit	60	Percentage based on surficial geology (65% Floodplain and alluvium gravel terraces)
Wyodak	6Da	Alternating Sandstone, Limestone and Shale - Thin Soil	2	Bedded Sedimentary Rock	100	Based on aquifer and surficial geology coverages, Heath (1985)
Yates	8D	Regolith	1	Metamorphic and Igneous	40	Percentage assigned based on surficial geology (59% alluvium/colluvium, 42% residuum)
Yates	8E	River Alluvium	6	River Valleys and Floodplains with Overbank Deposit	60	Percentage assigned based on surficial geology (59% alluvium/colluvium, 42% residuum)

SCL = silty clay loam; SNL = sandy loam; SLT = silt loam.

Aller, L., T. Bennett, J.H. Lehr, R.J. Perry, and G. Hackett. 1987. *DRASTIC: A Standardized System for Evaluating Pollution Potential Using Hydrogeologic Settings.* EPA-600/2-87-035. U.S. Environmental Protection Agency. Office of Research and Development. Ada, OK. April.

Heath, R.C. 1985. *National Water Summary 1984*. *State Summaries of Groundwater Resources*. Water-Supply Paper 2275. U.S. Geological Survey, Washington, DC.

Attachment C-3: Climate Center Assignments

Plant	Climate Center	Explanation If Not Closest Climate Center
A B Brown	Indianapolis, IN	Closest Met Station (Nashville) receives much more precipitation (12.26" out of range) than the site location. Used second closest because only slightly below (1.3) expected precipitation range for plant.
A/C Power- Ace Operations	Las Vegas, NV	
Allen	Little Rock, AR	
Alma	Madison, WI	Closest Met Station (St. Cloud) receives less rain than plant location. Used second closest Met Station because 5-year averages fell within expected precipitation range for the plant.
Antelope Valley	Bismarck, ND	
Arkwright	Watkinsville, GA	Closest Met Station (Atlanta) receives 6.96" more precipitation than plant location. Used second closest Met Station because 5-year averages are only slightly above (0.2) expected precipitation range for the plant.
Asheville	Knoxville, TN	
Baldwin	East St. Louis, IL	
Barry	Tallahassee, FL	Closest Met Station (New Orleans) receives much more precipitation (5.06" out of range) than the site location. Used second closest because only slightly above (3.4) expected precipitation range for plant.
Bay Front	Madison, WI	
Bay Shore	Put-in-Bay, OH	
Belews Creek	Greensboro, NC	
Ben French	Rapid City, SD	
Big Cajun 2	Lake Charles, LA	Closest Met Station (New Orleans) receives much more precipitation (5.06" out of range) than the site location. Used second closest because only slightly below (2.77) expected precipitation range for plant.
Big Sandy	Cincinnati, OH	Closest Met Station (Lexington) receives much more precipitation (8.35" out of range) than plant location. Used second closest Met Station because 5-year averages fell within expected precipitation range for the plant.
Big Stone	St. Cloud, MN	

(continued)

Climate Center Assignments (continued)

Plant	Climate Center	Explanation If Not Closest Climate Center
Black Dog Steam Plant	Madison, WI	Closest Met Station (St Cloud) is dryer (<27.5") than the 28-33" that the site receives. Madison fits in precipitation range (32.5") and is second closest.
Blue Valley	Topeka, KS	
Bowen	Atlanta, GA	
Brandon Shores	Seabrook, NJ	
Buck	Greensboro, NC	
Bull Run	Knoxville, TN	
C D McIntosh Jr	Orlando, FL	Closest Met Station (Tampa) receives less precipitation (5.31" out of range) than site location. Used second closest Met Station because 5-year averages fell within expected precipitation range for the plant.
C P Crane	Seabrook, NJ	
Cape Fear	Greensboro, NC	
Carbon	Salt Lake City, UT	
Cardinal	Pittsburgh, PA	
Cayuga	Indianapolis, IN	
Chalk Point	Seabrook, NJ	
Cholla	Phoenix, AZ	Closest Met Station (Flagstaff) receives much more precipitation (13.92" out of range) than plant location. Used second closest Met Station because 5-year averages were close (.31 higher) than the expected precipitation range for the plant.
Cliffside	Greensboro, NC	
Clover	Lynchburg, VA	
Coal Creek	Bismarck, ND	
Coleto Creek	San Antonio, TX	
Colstrip	Glasgow, MT	
Conemaugh	Pittsburgh, PA	
Conesville	Columbus, OH	
Council Bluffs	North Omaha, NE	
Crawford	East St. Louis, IL	
Crist	Tallahassee, FL	

(continued)

Climate Center Assignments (continued)

Plant	Climate Center	Explanation If Not Closest Climate Center
Cross	Charleston, SC	
Cumberland	Nashville, TN	
Dale	Lexington, KY	
Dallman	East St. Louis, IL	
Dan E Karn	East Lansing, MI	
Dan River	Greensboro, NC	
Danskammer	Bridgeport, CT	
Dave Johnston	Cheyenne, WY	
Dickerson	Seabrook, NJ	
Dolet Hills	Shreveport, LA	
Duck Creek	East St. Louis, IL	
Dunkirk	Ithaca, NY	
E D Edwards	Chicago, IL	
E W Brown	Lexington, KY	
Eckert Station	East Lansing, MI	
Edgewater	Madison, WI	
Elmer W Stout	Indianapolis, IN	
F B Culley	Indianapolis, IN	Closest Met Station (Nashville) receives much more precipitation (12.26" out of range) than plant location. Used second closest Met Station because 5-year & 30-year averages fell within expected precipitation range for the plant.
Fayette Power Prj	San Antonio, TX	
Flint Creek	Columbia, MO	Used http://www.weather.com and Envirofacts to determine that avg. precipitation for site was ~47". The closest Met Station (Tulsa) receives much less (~17") precipitation per year. Used second closest station.
Fort Martin	Pittsburgh, PA	
Frank E Ratts	Indianapolis, IN	
G G Allen	Greensboro, NC	
Gadsden	Atlanta, GA	

(continued)

Climate Center Assignments (continued)

Plant	Climate Center	Explanation If Not Closest Climate Center
Gallatin	Nashville, TN	
Gen J M Gavin	Cincinnati, OH	Closest Met Station (Columbus) receives less rain than plant location. Used second closest Met Station because 5-year averages fell within expected precipitation range for the plant. Also average precipitation for the second closest Met Station was nearest to http://www.weather.com
Genoa	Madison, WI	
Gibson	Indianapolis, IN	
Gorgas	Atlanta, GA	
Green River	Indianapolis, IN	Closest Met Station (Nashville) receives much more precipitation (12.26" out of range) than plant location. Used third closest Met Station because 5-year averages fell within expected precipitation range for the plant.
Greene County	Atlanta, GA	
H B Robinson	Charleston, SC	
Hammond	Atlanta, GA	
Harllee Branch	Watkinsville, GA	
Harrison	Pittsburgh, PA	
Hatfield's Ferry	Pittsburgh, PA	
Hennepin	Chicago, IL	
Heskett	Bismarck, ND	
Holcomb	Dodge City, KS	
Homer City	Pittsburgh, PA	
Hoot Lake	St. Cloud, MN	
Hugo	Shreveport, LA	Closest Met Station (Dallas) receives less precipitation (6.45" out of range) than plant location. Used second closest because only slightly above (2.07) expected precipitation range for plant.
Hunter	Grand Junction, CO	Closest Met Station (Salt Lake City) receives 8.6" more precipitation than plant location. Used second closest Met Station because 5-year averages fell within expected precipitation range for the plant.
Huntington	Cedar City, UT	Two closest Met Stations are out of range. Used second closest Met Station because 5-year averages fell within expected precipitation range for the plant.
Intermountain	Ely, NV	Closest Met Station (Salt Lake City) receives 6.1" more precipitation than plant location. Used second closest Met Station because 5-year averages fell within expected precipitation range for the plant.
J H Campbell	East Lansing, MI	

(continued)

Climate Center Assignments (continued)

Plant	Climate Center	Explanation If Not Closest Climate Center
J M Stuart	Cincinnati, OH	
J R Whiting	Put-in-Bay, OH	
Jack McDonough	Atlanta, GA	
Jack Watson	Tallahassee, FL	Closest Met Station (New Orleans) receives much more precipitation (5.06" out of range) than the site location. http://www.weather.com predicted average precipitation at plant location to be 65.2. Used third closest because its average was closest.
James H Miller Jr	Atlanta, GA	
Jim Bridger	Lander, WY	
John E Amos	Cincinnati, OH	The two closest Met Stations are out of the site's precipitation range. Used third closest Met Station because 5-year averages fell within expected precipitation range for the plant. Also average precipitation for the second closest Met Station was nearest to http://www.weather.com average.
John Sevier	Knoxville, TN	
Johnsonville	Nashville, TN	
Joliet 29	Chicago, IL	
Keystone	Pittsburgh, PA	
Killen Station	Cincinnati, OH	
Kingston	Knoxville, TN	
Kraft	Charleston, SC	
L V Sutton	Charleston, SC	
Lansing	Madison, WI	
Laramie R Station	Cheyenne, WY	
Lawrence EC	Topeka, KS	
Lee	Greensboro, NC	
Leland Olds	Bismarck, ND	
Lon Wright	North Omaha, NE	
Louisa	Des Moines, IA	
Marion	East St. Louis, IL	
Marshall	Greensboro, NC	

(continued)

Climate Center Assignments (continued)

Plant	Climate Center	Explanation If Not Closest Climate Center
Martin Lake	Shreveport, LA	
Mayo	Lynchburg, VA	
Meramec	East St. Louis, IL	
Merom	Indianapolis, IN	
Miami Fort	Cincinnati, OH	
Milton R Young	Bismarck, ND	
Mitchell - PA	Pittsburgh, PA	
Mitchell - WV	Pittsburgh, PA	
Mohave	Las Vegas, NV	
Monroe	Put-in-Bay, OH	
Morgantown	Norfolk, VA	
Mountaineer (1301)	Cincinnati, OH	Closest Met Station (Columbus) receives more rain than plant location. Although second closest site also falls within range, used third closest Met Station because site geography was similar and the station's 5-year averages fell within expected precipitation range for the plant.
Mt Storm	Pittsburgh, PA	
Muscatine Plant #1	Des Moines, IA	
Muskogee	Tulsa, OK	
Neal North	North Omaha, NE	
Neal South	North Omaha, NE	
Nebraska City	North Omaha, NE	
New Castle	Pittsburgh, PA	
Newton	Indianapolis, IN	Closest Met Station (East St. Louis) receives less rain than plant location. Used second closest Met Station because 5-year averages fell within expected precipitation range for the plant. Also average precipitation for the second closest Met Station was nearest to http://www.weather.com
North Omaha	North Omaha, NE	
Northeastern	Tulsa, OK	
Nucla	Grand Junction, CO	

(continued)

Climate Center Assignments (continued)

Plant	Climate Center	Explanation If Not Closest Climate Center
Oklaunion	Oklahoma City, OK	
Paradise	Cincinnati, OH	Closest Met Station (Nashville) receives much more precipitation (12.26" out of range) than plant location. Used third closest Met Station because 5-year averages fell within expected precipitation range for the plant.
Petersburg	Indianapolis, IN	
Pleasant Prairie	Chicago, IL	
Port Washington	Madison, WI	
Portland	Philadelphia, PA	
Possum Point	Norfolk, VA	
Potomac River	Seabrook, NJ	
Presque Isle	Sault Ste. Marie, MI	
R Gallagher	Cincinnati, OH	Closest Met Station (Lexington) receives much more precipitation (8.35" out of range) than plant location. Used second closest Met Station because 5-year & 30-year averages fell within expected precipitation range for the plant.
R M Schahfer	Chicago, IL	
Reid Gardner	Las Vegas, NV	
Richard Gorsuch	Columbus, OH	
Riverbend	Greensboro, NC	
Rodemacher	Lake Charles, LA	
Roxboro	Greensboro, NC	
Sandow	San Antonio, TX	
Scherer	Watkinsville, GA	Closest Met Station (Atlanta) receives 6.96" more precipitation than plant location. Used second closest Met Station because 5-year averages fell within expected precipitation range for the plant.
Shawnee	East St. Louis, IL	
Shawville	Pittsburgh, PA	
Sheldon	North Omaha, NE	

(continued)

Climate Center Assignments (continued)

Plant	Climate Center	Explanation If Not Closest Climate Center
South Oak Creek	Chicago, IL	
Springerville	Albuquerque, NM	Closest Met Station (Flagstaff) receives much more precipitation (8.92" out of range) than plant location. Used second closest Met Station because 5-year averages were within the expected precipitation range for the plant.
St Johns River Power	Jacksonville, FL	
Stanton Energy Ctr	Orlando, FL	
Stockton Cogen Company	Sacramento, CA	
Syl Laskin	St. Cloud, MN	
Tecumseh EC	Topeka, KS	
Texas-New Mexico	San Antonio, TX	Closest Met Station (Dallas) received less precipitation than site location. Used second closest Met Station because 5-year averages fell within expected precipitation range for the plant. Also average precipitation for the second closest Met Station was nearest to http://www.weather.com
Titus	Philadelphia, PA	
Trimble County	Cincinnati, OH	
Tyrone	Lexington, KY	
Valley	Madison, WI	
Vermilion	Chicago, IL	Closest Met Station (Indianapolis) receives more precipitation than plant location. Used second closest Met Station because 5-year averages fell within expected precipitation range for the plant.
Victor J Daniel Jr	Tallahassee, FL	Closest Met Station (New Orleans) receives much more precipitation (5.06" out of range) than the site location. Used second closest because only slightly above (3.4) expected precipitation range for plant.
W A Parish	Shreveport, LA	2 Closest Met Stations (Lake Charles & San Antonio) are more than 4" out of range. Used third closest because only slightly above (1.65") expected precipitation range for plant.
W H Weatherspoon	Greensboro, NC	
W S Lee	Watkinsville, GA	
Wabash River	Indianapolis, IN	
Walter C Beckjord	Cincinnati, OH	
Wansley	Atlanta, GA	
Warrick	Indianapolis, IN	Closest Met Station (Nashville) receives 12.2" more precipitation than plant location. Used second closest Met Station because 5-year averages fell within expected precipitation range for the plant.

(continued)

Climate Center Assignments (continued)

Plant	Climate Center	Explanation If Not Closest Climate Center
Waukegan	Chicago, IL	
Weston	Madison, WI	
Widows Creek	Nashville, TN	
Will County	East St. Louis, IL	
Wyodak	Rapid City, SD	
Yates	Atlanta, GA	

[This page intentionally left blank.]

Attachment C-4: Waterbody Assignments and Flow

Plant	CUSEG	Nearest Reach	Reach Type	QLOW	QMEAN
A B Brown	05140202014	OHIO R	Regular Reach	9167.38965	150031.6875
A/C Power- Ace Operations	18090205005	SEARLES L	Lake Shoreline		
Allen	08010211007	HORN LAKE CUTOFF	Lake Shoreline		
Alma	07040003009	MISSISSIPPI R	Regular Reach	5683.02002	25397.4707
Antelope Valley	10130201005	ANTELOPE CR	Start Reach	0	96.87
Arkwright	03070103007	OCMULGEE R	Regular Reach	428.79999	2708.53003
Asheville	06010105026	FRENCH BROAD R	Regular Reach	412.04999	1722.34998
Baldwin	07140204004	KASKASKIA R	Regular Reach	351.72	3832.12012
Barry	03160204014	MOBILE R	Regular Reach	7561.14014	63275.23828
Bay Front	07070005036	L SUPERIOR	Great Lakes Shoreline		
Bay Shore	04100010003	L ERIE, U.S. SHORE	Great Lakes Shoreline	0	0
Belews Creek	03010103098	BELEWS L	Lake Shoreline		
Ben French	10120110010	CASTLE CR	Start Reach	2.96	18.62
Big Cajun 2	08070100005	MISSISSIPPI R	Regular Reach	100937.8125	466865.5625
Big Sandy	05070204008	BIG SANDY R	Regular Reach	152.02	5746.95996
Big Stone	07020001033	BIG STONE LAKE	Lake Shoreline		
Black Dog Steam Plant	07020012001	BLACK DOG LAKE	Lake Shoreline		
Blue Valley	10300101034	LITTLE BLUE R	Regular Reach	23.2	141.75
Bowen	03150104008	ETOWAH R	Regular Reach	413.13	2294.86011
Brandon Shores	02060003037	CURTIS BAY	Coastal Shoreline	0	0
Buck	03040103040	YADKIN R	Regular Reach	912.72998	4722.54004
Bull Run	06010207015	CLINCH R	Regular Reach	102.46	4732.3501
C D McIntosh Jr	03100205014	NO LAKE PARKER	Lake Shoreline		

(continued)

Waterbody Assignments and Flow (continued)

Plant	CUSEG	Nearest Reach	Reach Type	QLOW	QMEAN
C P Crane	02060003025	CURTIS BAY	Coastal Shoreline	0	0
Cape Fear	03030002001	HAW R	Regular Reach	58.98	1584.83997
Carbon	14060007018	PRICE R	Regular Reach	1.92	77
Cardinal	05030106033	OHIO R	Regular Reach	3391.62012	37533.17188
Cayuga	05120108001	WABASH R	Regular Reach	965.09003	10100.21973
Chalk Point	02060006009	PATUXENT R	Wide-River Shoreline	0	0
Cholla	15020008017	CHOLLA COOLING POND	Lake Shoreline		
Cliffside	03050105031	BROAD R	Regular Reach	332.17001	1510.08997
Clover	03010102027	ROANOKE R	Regular Reach	408.64001	2702.59009
Coal Creek	10130101018	UNKNOWN LAKE	Lake Shoreline		
Coleto Creek	12100303014	MARCELINAS CR	Start Reach	1.11	3.79
Colstrip	10100001108	ARMELLS CR, E FK	Start Reach	0	18.64
Conemaugh	05010007002	CONEMAUGH R	Regular Reach	194.53999	1553.52002
Conesville	05040004071	MUSKINGUM R	Regular Reach	447.98001	4707.08008
Council Bluffs	10230006004	MISSOURI R	Regular Reach	4402.58984	31444.83008
Crawford	07130011018	ILLINOIS R	Regular Reach	3444.66992	20788.71094
Crist	03140305001	ESCAMBIA R	Terminal Reach	845.46002	6772.5498
Cross	03050201022	DIVERS CANAL TO LAKE MOU	Lake Shoreline		
Cumberland	05130205017	CUMBERLAND R	Regular Reach	536.47998	25322.66016
Dale	05100205047	KENTUCKY R	Regular Reach	35.32	5213.06982
Dallman	07130007003	LAKE SPRINGFIELD	Lake Shoreline		
Dan E Karn	04080103005	L HURON U.S. SH SAGINAW BAY	Great Lakes Shoreline	0	0
Dan River	03010103014	DAN R	Regular Reach	358.12	1954.15002
Danskammer	02020008022	HUDSON R	Wide-River Shoreline	0	0
Dave Johnston	10180007005	N PLATTE R	Regular Reach	65.24	502.87
Dickerson	02070008013	POTOMAC R	Regular Reach	895.57001	10528.36035
Dolet Hills	11140206019	BAYOU PIERRE LAKE	Lake Shoreline		

(continued)

Waterbody Assignments and Flow (continued)

Plant	CUSEG	Nearest Reach	Reach Type	QLOW	QMEAN
Duck Creek	07130003010	L CHAUTAUQUA	Lake Shoreline		
Dunkirk	04120101003	L ERIE, U.S. SHORE	Great Lakes Shoreline	0	0
E D Edwards	07130003018	ILLINOIS R	Regular Reach	2998.32007	13899.62988
E W Brown	05100205015	HERRINGTON LAKE	Lake Shoreline		
Eckert Station	04050004003	GRAND R	Regular Reach	73.47	484.28
Edgewater	04030101002	L MICHIGAN	Great Lakes Shoreline	0	0
Elmer W Stout	05120201005	WHITE R	Regular Reach	70.17	1429.92004
F B Culley	05140201001	OHIO R	Regular Reach	8728.7002	131543.0625
Fayette Power Prj	12090301003	CEDAR CREEK RESERVOIR	Lake Shoreline		
Flint Creek	11110103031	SWEPCO RSRVR,LT FLINT CK	Lake Shoreline		
Fort Martin	05020003001	MONONGAHELA R	Regular Reach	293.66	4497.75
Frank E Ratts	05120202003	WHITE R	Regular Reach	343.59	11525.13965
G G Allen	03050101009	CATAWBA R	Regular Reach	462.92001	2958.09009
Gadsden	03150106041	COOSA R	Regular Reach	1096.10999	9468
Gallatin	05130201006	OLD HICKORY L	Lake Shoreline		
Gen J M Gavin	05030202005	OHIO R	Regular Reach	4258.12012	55143.35938
Genoa	07060001017	MISSISSIPPI R	Regular Reach	6434.18018	29379.25
Gibson	05120113013	WABASH R	Regular Reach	2247.6001	26799.73047
Gorgas	03160109002	BLACK WARRIOR R, MULBERRY F	Lake Shoreline		
Green River	05110003001	GREEN R	Regular Reach	320.06	9752
Greene County	03160113011	BLACK WARRIOR R	Regular Reach	304.73001	9820.04004
H B Robinson	03040201042	L ROBERTSON	Lake Shoreline		
Hammond	03150105025	COOSA R	Regular Reach	1196.82996	6569.95996
Harllee Branch	03070101006	L SINCLAIR	Lake Shoreline		
Harrison	05020002008	WEST FORK R	Regular Reach	33.03	1038.32996
Hatfield's Ferry	05020005026	MONONGAHELA R	Regular Reach	479.79999	8278.94043
Hennepin	07130001026	ILLINOIS R	Regular Reach	3233.23999	13146.83984
Heskett	10130101001	MISSOURI R	Regular Reach	3461.55005	22744.26953

(continued)

Waterbody Assignments and Flow (continued)

Plant	CUSEG	Nearest Reach	Reach Type	QLOW	QMEAN
Holcomb	11030001001	ARKANSAS R	Regular Reach	0	197.92999
Homer City	05010007015	TWO LICK CR	Regular Reach	4.53	295.22
Hoot Lake	09020103002	OTTER TAIL R	Regular Reach	12.45	271.35999
Hugo	11140105041	KIAMICHI CR, N FK	Start Reach	2.55	53.16
Hunter	14060009034	ROCK CANYON CR	Start Reach	0	0.1
Huntington	14060009020	HUNTINGTON CR	Regular Reach	10.75	91.1
Intermountain		none		0	0
J H Campbell	04050002001	L MICHIGAN	Great Lakes Shoreline	0	0
J M Stuart	05090201024	OHIO R	Regular Reach	6767.47021	92214.6875
J R Whiting	04100001002	L ERIE, U.S. SHORE	Great Lakes Shoreline	0	0
Jack McDonough	03130002044	CHATTAHOOCHEE R	Regular Reach	726.45001	2952.18994
Jack Watson	03170009034	BILOXI BAY	Coastal Shoreline	0	0
James H Miller Jr	03160111005	BLACK WARRIOR R, LOCUST FK	Lake Shoreline		
Jim Bridger	14040105011	UNKNOWN LAKE	Lake Shoreline		
John E Amos	05050008007	KANAWHA R	Regular Reach	1390.22998	14930.83984
John Sevier	06010104011	HOLSTON R	Regular Reach	633	4079.15991
Johnsonville	06040005007	KENTUCKY L	Lake Shoreline		
Joliet 29	07120004004	DES PLAINS R	Regular Reach	1029.93005	3809.69995
Keystone	05010006002	CROOKED CR	Regular Reach	30.72	422.14999
Killen Station	05090201024	OHIO R	Regular Reach	6767.47021	92214.6875
Kingston	06010207001	CLINCH R	Regular Reach	266.35999	7347.89014
Kraft	03060109007	SAVANNAH R	Regular Reach	3570.52002	12365
L V Sutton	03030005011	CAPE FEAR R	Regular Reach	619.95001	8594.57031
Lansing	07060001009	MISSISSIPPI R	Regular Reach	7684.02002	32253.15039
Laramie R Station	10180011002	LARAMIE R	Regular Reach	28.53	90.8
Lawrence EC	10270104021	KANSAS R	Regular Reach	403.81	6720.29004
Lee	03020201007	NEUSE R	Regular Reach	76.18	1657.39001
Leland Olds	10130101020	MISSOURI R	Regular Reach	4270.4502	21650.67969

(continued)

Waterbody Assignments and Flow (continued)

Plant	CUSEG	Nearest Reach	Reach Type	QLOW	QMEAN
Lon Wright	10220003048	RAWHIDE CR	Start Reach	0.94	11.59
Louisa	07080101003	MISSISSIPPI R	Regular Reach	15067.92969	54665.96094
Marion	05140204030	L OF EGYPT	Lake Shoreline		
Marshall	03050101015	L NORMAN	Lake Shoreline		
Martin Lake	12010002050	MARTIN LAKE	Lake Shoreline		
Mayo	03010104045	MAYO CR	Start Reach	5.99	61.03
Meramec	07140101014	MISSISSIPPI R	Regular Reach	33305	177021.1875
Merom	05120111011	TURTLE CR RESERVOIR	Lake Shoreline		
Miami Fort	05090203012	OHIO R	Regular Reach	6516.18994	98615.0625
Milton R Young	10130101024	NELSON LAKE AND MISSOURI RIVER	Lake Shoreline		
Mitchell - PA	05020005002	MONONGAHELA R	Regular Reach	848.58002	9284.13965
Mitchell - WV	05030106013	OHIO R	Regular Reach	3419.20996	38713.19922
Mohave	15030101011	COLORADO R	Regular Reach	1916.72998	12134.36035
Monroe	04100001002	L ERIE, U.S. SHORE	Great Lakes Shoreline	0	0
Morgantown	02070011051	POTOMAC R	Wide-River Shoreline	0	0
Mountaineer (1301)	05030202008	OHIO R	Regular Reach	4242.58984	54823.21094
Mt Storm	02070002027	STONY R RES	Lake Shoreline		
Muscatine Plant #1	07080101005	MISSISSIPPI R	Regular Reach	14573.71973	54469.48047
Muskogee	11110102012	ARKANSAS R	Regular Reach	227.57001	21258.39062
Neal North	10230001021	MISSOURI R	Regular Reach	4217.7998	29486.82031
Neal South	10230001021	MISSOURI R	Regular Reach	4217.7998	29486.82031
Nebraska City	10240001002	MISSOURI R	Regular Reach	5807.77002	36764.01172
New Castle	05030104002	BEAVER R	Regular Reach	268.48001	2425.32007
Newton	05120114006	NEWTON LAKE	Lake Shoreline		
North Omaha	10230006009	MISSOURI R	Regular Reach	4365.6499	31400.93945
Northeastern	11070105012	VERDIGRIS R	Regular Reach	3.85	2168.47998
Nucla	14030003012	SAN MIGUEL R	Regular Reach	8.1	307.64001
Oklaunion	11130302061	BOGGY CR	Start Reach	0.09	14.93

(continued)

Waterbody Assignments and Flow (continued)

Plant	CUSEG	Nearest Reach	Reach Type	QLOW	QMEAN
Paradise	05110003003	GREEN R	Regular Reach	316.59	9663.71973
Petersburg	05120202003	WHITE R	Regular Reach	343.59	11525.13965
Pleasant Prairie	07120004012	L MICHIGAN AND J	Lake Shoreline		
Port Washington	04030101002	L MICHIGAN	Great Lakes Shoreline	0	0
Portland	02040105012	DELAWARE R	Regular Reach	1995.12	9089.00977
Possum Point	02070011074	POTOMAC R	Wide-River Shoreline	0	0
Potomac River	02070010025	POTOMAC R	Artificial Open Water Reach	919.89001	11721.87988
Presque Isle	04020105002	L SUPERIOR, U.S. SHORE	Great Lakes Shoreline	0	0
R Gallagher	05140101001	OHIO R	Regular Reach	7634.39014	119152.1875
R M Schahfer	07120001012	KANAKEE R	Regular Reach	458.92001	1410.56006
Reid Gardner	15010012006	MUDDY R	Regular Reach	0.68	19.22
Richard Gorsuch	05030202039	OHIO R	Regular Reach	4079.81006	48956.14062
Riverbend	03050101012	CATAWBA R	Regular Reach	412.28	2623.09009
Rodemacher	11140207020	RODEMACHER LAKE	Lake Shoreline		
Roxboro	03010104034	HYCO L	Lake Shoreline		
Sandow	12070102012	ALCOA LAKE	Lake Shoreline		
Scherer	03070103012	OCMULGEE R	Start Reach	655.48999	2490.72998
Shawnee	05140206009	OHIO R	Regular Reach	21748.59961	288452.1875
Shawville	02050201002	SUSQUEHANNA R, W BR	Regular Reach	96.9	1947.33997
Sheldon	10240008030	UNKNOWN LAKE	Lake Shoreline		
South Oak Creek	04040002004	L MICHIGAN	Great Lakes Shoreline	0	0
Springerville	15020002025	*A	Start Reach	0	2.49
St Johns River Power	03080103003	ST JOHNS R	Wide-River Shoreline	0	0
Stanton Energy Ctr	03080101036	ECOHLOCKHATCHEE R	Start Reach	5.95	131.42999
Stockton Cogen Company	18040002005	LITTLEJOHNS CR	Start Reach	0.21	50.61
Syl Laskin	04010201034	COLBY L AND PARTRIDGE R	Lake Shoreline		
Tecumseh EC	10270102003	KANSAS R	Regular Reach	388.51999	5923.74023
Texas-New Mexico	12070101008	LITTLE BRAZOS R	Start Reach	0.55	139.05

(continued)

Waterbody Assignments and Flow (continued)

Plant	CUSEG	Nearest Reach	Reach Type	QLOW	QMEAN
Titus	02040203010	SCHUYLKILL R	Regular Reach	91.25	1880.77002
Trimble County	05140101007	OHIO R	Regular Reach	7524.29004	117896.3125
Tyrone	05100205013	KENTUCKY R	Regular Reach	154.36	7097.54004
Valley	04040003001	MILWAUKEE R	Terminal Reach	10.71	540.60999
Vermilion	05120109006	VERMILION R, M FK	Regular Reach	3.45	340.35999
Victor J Daniel Jr	03170006007	PASCAGOULA R	Regular Reach	1256.55005	12878.25
W A Parish	12070104021	SMITHERS L	Lake Shoreline		
W H Weatherspoon	03040203016	LUMBER R	Regular Reach	97.9	865.13
W S Lee	03050109066	SALADA R	Regular Reach	20.68	461.51001
Wabash River	05120111018	WABASH R	Regular Reach	985.53998	10551.67969
Walter C Beckjord	05090201001	OHIO R	Regular Reach	6416.77002	92084.0625
Wansley	03130002032	CHATTAHOOCHEE R	Regular Reach	702.71002	4400.72021
Warrick	05140201022	LITTLE PIGEON CR	Regular Reach	61.57	1149.60999
Waukegan	04040002002	L MICHIGAN	Great Lakes Shoreline	0	0
Weston	07070002023	WISCONSIN R	Regular Reach	1069.30005	3484.32007
Widows Creek	06030001049	TENNESSEE R	Regular Reach	7221.95996	38237.07031
Will County	07110009002	WOOD R	Start Reach	29	87.81
Wyodak	10120201038	DONKEY CR	Start Reach	0	4.4
Yates	03130002061	CHATTAHOOCHEE R	Regular Reach	702.21997	4063.29004

[This page intentionally left blank.]

Appendix D. MINTEQA2 Nonlinear Sorption Isotherms

D.1 Overview of MINTEQA2 Modeling

Chemicals in leachate can be subject to complex geochemical interactions in soil and groundwater, which can strongly affect their rate of transport in the subsurface. EPA's Composite Model for Leachate Migration with Transformation Products (EPACMTP) treats these interactions as equilibrium sorption processes. The equilibrium assumption means that the sorption process occurs instantaneously, or at least very quickly relative to the time scale of constituent transport. Although sorption—or the attachment of leachate constituents to solid soil or aquifer particles—may result from multiple chemical processes, EPACMTP lumps these processes together into an effective soil-water partition coefficient (K_d). The retardation factor (R) accounts for the effects of equilibrium sorption of dissolved constituents onto the solid phase, removing them from solution and reducing the available mass in the dissolved phase. R, a function of the constituent-specific K_d and the soil or aquifer properties, is calculated as:

$$R = 1 + \frac{\rho_b \times K_d}{\Phi} \qquad \text{(D-1)}$$

where

R = Retardation factor
ρ_b = Soil or aquifer bulk density (g/cm^3)
K_d = Solid-water partition coefficient (g/cm^3)
ϕ = Water content (in unsaturated zone) or porosity (in saturated zone).

An isotherm is an expression of the equilibrium relationship between the aqueous concentration and the sorbed concentration of a metal (or other constituent) at a constant temperature. For metals, EPACMTP accounts for more complex geochemical reactions by using effective sorption isotherms generated using EPA's geochemical equilibrium speciation model for dilute aqueous systems, MINTEQA2 (U.S. EPA, 1991).

The MINTEQA2 model was used to generate one set of isotherms for each metal reflecting the range in geochemical environments expected at waste sites across the nation. The variability in geochemical environments at CCW sites across the country was represented by five geochemical master variables (groundwater composition, pH, concentration of iron oxide adsorption sites, leachate ionic strength, and concentration of dissolved and particulate natural organic matter), and the MINTEQA2 modeling was repeated (separately for each metal) for numerous combinations of master variable settings. This procedure resulted in nonlinear K_d versus aqueous metal concentration curves for combinations of master variable settings spanning the range of reasonable values (U.S. EPA 2003a).

For each metal, the resulting set of isotherms was tabulated into a supplementary input data file for use by the EPACMTP model, hereafter referred to as an "empirical nonlinear isotherm." In the fate and transport modeling for a particular metal, EPACMTP was executed, and the national probability distributions for these five master variables formed the basis for the Monte Carlo selection of the appropriate adsorption isotherm.

In modeling metals transport in the unsaturated zone, EPACMTP uses a range of K_d values from the nonlinear sorption isotherms. However, in modeling metals transport in the saturated zone, EPACMTP selects the lowest from all available K_d values corresponding to concentrations less than or equal to the maximum water table concentration. For more details see the *EPACMTP Technical Background Document* (U.S. EPA, 2003b). This simplification in the saturated zone is required for all solution options and is based on the assumption that, after dilution of the leachate plume in groundwater, the concentrations of metals will typically be in a range where the isotherm is approximately linear. However, this assumption may not be valid when the metal concentrations in the leachate are exceedingly high. Although EPACMTP is able to account for the effect of the geochemical environment at a site on the mobility of metals, the model assumes that the geochemical environment at a site is constant and not affected by the presence of the leachate plume. In reality, the presence of a leachate plume may alter the ambient geochemical environment.

D.2 Previous CCW Metals Modeling Effort

In a previous risk assessment for fossil fuel combustion wastes (FFCWs) conducted in 1998 (U.S. EPA, 1998), sorption isotherms generated using MINTEQA2 were used in EPACMTP to account for metal partitioning. However, these isotherms were not calculated specifically for use in FFCW modeling—they had been computed using MINTEQA2 in 1995 for use in modeling support for the Hazardous Waste Identification Rule (HWIR).

The disposal scenario for HWIR was the industrial Resource Conservation and Recovery Act (RCRA) Subtitle D nonhazardous waste landfill. In fact, the MINTEQA2 modeling that produced the isotherms had originally been designed to represent municipal solid waste landfills, and leachate from those landfills had been sampled so that appropriate forms of leachate organic acids at various concentrations could be included in the modeling. For the HWIR analysis, the scenario was changed to industrial Subtitle D, and only the isotherms corresponding to low concentrations of the leachate organic acids were used for HWIR modeling. The same isotherms were used in the 1998 FFCW risk assessment. As in the HWIR modeling, only the isotherms corresponding to the lowest setting of leachate organic carbon were used.

In 1999, EPA received review comments concerning the use of the industrial Subtitle D metal partitioning isotherms in the 1998 risk assessment. The most comprehensive review was prepared by Charles Norris and Christina Hubbard on behalf of the Environmental Defense Fund and other environmental advocacy groups (Norris and Hubbard, 1999). The Norris and Hubbard report criticized the 1998 risk assessment for using MINTEQA2 isotherms designed for a different scenario (nonhazardous industrial landfills). Norris and Hubbard also offered 20 specific criticisms on the input parameters and other factors involved in the MINTEQA2 modeling. EPA responded by evaluating each of these criticisms through review and assessment of MINTEQA2 input values, model sensitivity tests, and consultations with experts. This review

is documented in U.S. EPA (2000, 2001a). The evaluation of the Norris and Hubbard comments resulted in suggested revisions in the MINTEQA2 modeling strategy, as described in U.S. EPA (2001b).

Based on a review of available information on CCW leachate composition and an analysis of the potential effects of this composition on metals mobility, EPA (U.S. EPA, 2001b) also determined that if MINTEQA2 is to be used at CCW sites, leachate from CCW facilities should be studied to look for trends in composition, especially with regard to the concentrations of constituents that may

- Contribute to elevated groundwater pH
- Compete with the contaminant metal for sorption sites and thus result in reduced metal sorption (e.g., Ca, Mg, SO_4, other metals)
- Complex with the contaminant metal so that the metal is less likely to be sorbed (e.g., SO_4, CO_3, organic ligands)
- Precipitate with the contaminant metal (e.g., SO_4, CO_3).

D.3 MINTEQA2 Modeling Revisions for CCW Risk Assessment

Many of the suggested revisions from U.S. EPA (2001b) were implemented in the MINTEQA2 modeling for the current CCW risk assessment. Some of the suggested revisions were not implemented, either because they were not applicable (e.g., organic carbon assumptions were not changed, because CCW leachate has negligible organic carbon) or because models or data were not adequate to carry forth the recommendation. These revisions are discussed in greater detail in U.S. EPA (2003c).

In addition to revising the MINTEQA2 model, EPA compiled leachate characteristics into the CCW constituent database (see **Appendix A**) and statistically analyzed these data to identify three chemically distinct CCW leachate types: conventional CCW (including ash and flue gas desulfurization [FGD] sludge), codisposed CCW and coal cleaning wastes, and fluidized bed combustion (FBC) waste. Leachate concentration ranges for major ions (e.g., Ca, SO_4, Mg, Na, Cl, etc.) and pH were developed for each of these waste types and were used to represent CCW leachate during MINTEQA2 modeling.

As needed, sorption reactions were included for those CCW constituents known to undergo significant sorption. Including elevated concentrations of leachate constituents and their corresponding sorption reactions in the MINTEQA2 model allowed for full competition with the contaminant metal for sorption sites. The metal solubilizing effect through complexation between the contaminant metal and dissolved ligands was also included, as was the potential for metal precipitation. Because precipitation of the metal can serve to attenuate the transportable concentration, the equilibrium fraction in all three phases (dissolved, sorbed, and precipitated) were stored and made available for use by EPACMTP. The precipitated fraction was used to develop a solubility limit that was used during EPACMTP modeling (U.S. EPA, 2003c).

D.4 MINTEQA2 Modeling for CCW Risk Assessment

The expected natural variability in K_d for a particular metal was represented during the MINTEQA2 modeling effort by varying the input parameters that most impact K_d: groundwater type (carbonate or noncarbonate), pH, concentration of aquifer sorbents, composition and concentration level of CCW leachate, and concentration of the contaminant metal. The natural pH range for the two groundwater types was sampled from a range of 7 to 8 for carbonate aquifers and 4 to 10 for noncarbonate aquifers (U.S. EPA, 2003c).

In addition, CCW leachate ranges from acidic (pH < 2) to highly alkaline (pH > 12), and it can impact unsaturated zone and groundwater pH. To account for this possibility, the CCW leachate/ groundwater system was equilibrated at a series of pH values that spanned the range of expected variability in mixed CCW leachate-groundwater systems (U.S. EPA, 2003c).

To account for the variability in the sorption capacity of soil and aquifer materials, the soil and groundwater systems were equilibrated with various concentrations of two commonly occurring natural sorbents: ferric (iron) oxyhydroxide (FeOx) and particulate organic matter (POM). CCW leachate can include elevated concentrations of inorganic constituents such as calcium, sulfate, sodium, potassium, and chloride, which may reduce sorption of metals due to competition for sorption sites or complexation with metals in solution. To account for this effect, these leachate components were added to the MINTEQA2 model inputs at concentrations representative of the three CCW waste types (conventional CCWs, codisposed CCW and coal cleaning wastes, and FBC wastes). This new MINTEQA2 master variable is termed leachate "richness" or ionic strength (U.S. EPA, 2003c).

The results of each MINTEQA2 model run were compiled as the equilibrium distribution of the contaminant metal among dissolved, sorbed, and precipitated fractions for each metal concentration, and were saved in a separate file indexed with the settings of all variables used to define the system. These files were produced for all possible values for the variables defining the system, and were compiled into a database of indexed K_d values for use in the EPACMTP fate and transport model (U.S. EPA, 2003c).

D.5 EPACMTP Modeling Revisions to Accommodate MINTEQA2 Updates

EPA updated EPACMTP to support the new system variable (leachate ionic strength) for isotherm selection, to address issues regarding the impacts of leachate pH on ambient soil and aquifer pH, and to address issues regarding solubility limits for metals in solution. A brief description of these model changes are discussed below, with more detail provided in U.S. EPA (2003d).

Ionic Strength. A new system or "master" variable was added to include ionic strength as a key for choosing the representative isotherm from the database for both the unsaturated and saturated zones.

Leachate Effects on Geochemical Environment. These effects were addressed in EPACMTP under the following constraints: (1) no significant impairment of the computational efficiency for probabilistic applications; (2) data requirements limited to readily available data;

and (3) a scientifically defensible approach, given significant uncertainties with respect to the true impacts of leachate pH on the subsurface. Two modifications to the EPACMTP were considered: (1) determine the governing pH in the soil column (either the pH of the leachate or the native soils); and (2) determine the pH of the saturated zone as a result of the infiltrating leachate.

The approach selected for determining the governing pH of the soil column (unsaturated zone) beneath the waste management unit (WMU) compares the operational life of the WMU (the duration of leaching) to an estimate of the first arrival time of the contaminant front at the water table (a surrogate for the residence time of the contaminant in the soil column). If the operational life of the WMU is *relatively* long compared to the time required for the contaminant to migrate to the water table, there is a high likelihood that the leachate permeates the soil column and that the pH environment is governed by the leachate. Conversely, a relatively short operational life and retarded contaminant migration would favor ambient soil pH conditions. An analysis of the relationship between operational life and travel time indicated that a ratio of approximately 5 (operational life over travel time) would, in many cases, result in a balanced selection of cases where leachate pH governs versus cases where soil pH governs over approximately 10,000 Monte Carlo iterations.

For each iteration of EPACMTP, the operational life was compared to a travel-time estimate based on a K_d averaged from isotherms selected based on the leachate pH and soil pH. If the ratio was greater than 5, the pH of the leachate was assumed to govern, and the pH of the leachate was used to select the isotherm for transport in the unsaturated zone. If the ratio was less than 5, the soil pH was used to select the isotherm.

In the saturated zone, the impacts of leachate pH were handled using a simple homogeneous mixing calculation. The volume of leachate released from the WMU was mixed with the volume of the aquifer that was likely to be impacted by a plume. The resulting mixed pH was used to select the isotherm for transport in the saturated zone with one limitation: in carbonate environments, the mixed pH in the aquifer was not allowed to drop below a pH of 6. Such acid conditions would likely result in significant dissolution of the soil matrix.

Metal Solubility Limits. As mentioned above, each sorption isotherm comprises equilibrium concentrations of the three contaminant phases (dissolved, sorbed, and precipitated) over a range of total concentration values. An examination of the change in the dissolved-phase concentrations relative to changes in the total concentration in any isotherm reveals solubility behavior for that contaminant: if the dissolved component does not change with increasing total concentration, a solubility limit has been achieved. If, however, the dissolved component increases along with the total concentration, then there is capacity for more dissolved mass in the groundwater or soil porewater.

EPACMTP uses this information (contained in each isotherm file) to determine if a solubility limit should be imposed in the saturated zone. Once an isotherm has been selected (after pH considerations have been addressed), the equilibrium states corresponding to the three highest total concentrations are examined. If the dissolved concentration changes more than one tenth of one percent over the last three points, then EPACMTP assumes there is no solubility limit. If the change in dissolved concentration is less than one tenth of one percent, EPACMTP

assumes a solubility limit has been reached and caps the concentration of the leachate entering the saturated zone at the water table to that limit.

D.6 Sampled K_ds from MINTEQA2 Isotherms by EPACMTP

As described above, a range of K_ds from an isotherm were used by EPACMTP in each unsaturated zone transport simulation. To simplify the presentation of K_d here, an *effective* K_d value was calculated and reported by EPACMTP. An effective K_d was determined by first estimating the value of the retardation factor, described in Equation D-1, as follows:

$$R = \frac{t_{WT} \times q}{d_{soil}}$$

where

t_{wt} = time for leachate to reach the water table (yr)
q = seepage velocity of leachate through the unsaturated soil column (m/yr)
d_{dsoil} = depth to the water table or length of the unsaturated soil column (m)

Substituting this value for R in Equation D-1 and solving for K_d yields an effective K_d that is based on the first arrival of the leachate front at the water table.

Table D-1 presents selected percentiles of K_d sampled from the MINTEQA2 isotherms for every waste management modeling scenario conducted in the CCW risk assessment for the groundwater pathway. Each scenario corresponds to a unique combination of waste type, metal species, waste management unit type, and subsurface domain (unsaturated zone or saturated zone). The values presented for the saturated zone are taken from the set of actual K_d values used in each modeling scenario.

Table D-1. Select Percentiles of K_d Sampled from MINTEQA2 Isotherms by Updated EPACMTP by Waste Type, Metal Species, WMU Type, and Subsurface Domain

Waste Stream	Metal	CASID	WMU	Zone	Percentiles of K_d							
					10%	25%	50%	75%	80%	85%	90%	95%
Ash	Aluminum	7429905	LF	Saturated	8.5E-20	3.0E-16	5.1E-10	5.0E-03	7.1E-02	9.2E-01	1.6E+00	2.8E+00
Ash	Aluminum	7429905	SI	Saturated	8.6E-20	2.5E-10	2.6E-04	9.4E-01	2.2E+00	2.7E+00	2.8E+00	6.2E+00
Ash	Aluminum	7429905	LF	Unsaturated	6.9E-03	1.0E-01	2.0E-01	1.8E+01	5.9E+01	1.0E+02	1.8E+02	3.7E+02
Ash	Aluminum	7429905	SI	Unsaturated	2.0E-11	3.7E-02	1.6E-01	1.1E+00	1.6E+00	2.8E+00	3.0E+00	6.6E+00
Ash	Antimony	7440360	LF	Saturated	4.3E-03	2.5E-02	7.5E-02	2.0E-01	2.1E-01	3.0E-01	3.4E-01	7.1E-01
Ash	Antimony	7440360	SI	Saturated	2.8E-03	1.9E-02	9.6E-02	2.0E-01	2.1E-01	3.3E-01	3.5E-01	5.9E-01
Ash	Antimony	7440360	LF	Unsaturated	9.1E-02	2.9E-01	9.6E-01	7.6E+00	1.0E+01	1.4E+01	2.2E+01	4.9E+01
Ash	Antimony	7440360	SI	Unsaturated	0.0E+00	1.1E-01	3.4E-01	6.8E-01	7.9E-01	9.5E-01	1.3E+00	1.6E+00
Ash	Arsenic3	22569728	LF	Saturated	5.2E-02	1.5E-01	4.1E-01	6.6E-01	8.0E-01	9.2E-01	1.0E+00	1.1E+00
Ash	Arsenic3	22569728	SI	Saturated	4.1E-02	1.4E-01	4.1E-01	6.8E-01	8.1E-01	9.2E-01	1.0E+00	1.1E+00
Ash	Arsenic3	22569728	LF	Unsaturated	1.5E-01	5.0E-01	1.2E+00	2.2E+00	2.7E+00	3.7E+00	5.8E+00	1.2E+01
Ash	Arsenic3	22569728	SI	Unsaturated	5.7E-05	3.1E-01	7.4E-01	1.3E+00	1.5E+00	1.6E+00	1.8E+00	2.1E+00
Ash	Arsenic5	15584040	LF	Saturated	1.1E+00	7.0E+00	3.4E+01	9.8E+01	1.2E+02	1.6E+02	2.1E+02	6.3E+02
Ash	Arsenic5	15584040	SI	Saturated	1.2E+00	6.7E+00	2.9E+01	8.9E+01	1.1E+02	1.5E+02	2.9E+02	5.9E+02
Ash	Arsenic5	15584040	LF	Unsaturated	0.0E+00	4.2E+00	3.6E+01	1.0E+02	1.3E+02	1.6E+02	2.1E+02	4.2E+02
Ash	Arsenic5	15584040	SI	Unsaturated	2.9E-01	3.2E+00	2.1E+01	7.6E+01	9.5E+01	1.3E+02	2.0E+02	4.7E+02
Ash	Barium	7440393	LF	Saturated	2.4E-01	4.2E-01	5.6E-01	9.2E-01	1.1E+00	1.2E+00	1.6E+00	2.3E+00
Ash	Barium	7440393	SI	Saturated	2.5E-01	4.4E-01	5.7E-01	9.3E-01	1.1E+00	1.2E+00	1.6E+00	2.4E+00
Ash	Barium	7440393	LF	Unsaturated	0.0E+00	7.1E+00	2.0E+02	6.6E+02	7.9E+02	1.0E+03	1.4E+03	2.2E+03
Ash	Barium	7440393	SI	Unsaturated	0.0E+00	8.7E-01	1.8E+00	5.9E+00	8.0E+00	1.2E+01	1.9E+01	5.3E+01
Ash	Boron	7440428	LF	Saturated	3.8E-11	4.3E-10	1.7E-07	2.8E-06	3.6E-06	5.4E-06	7.1E-06	1.0E-05
Ash	Boron	7440428	SI	Saturated	2.6E-10	3.2E-08	1.7E-06	6.5E-06	7.7E-06	8.9E-06	1.1E-05	1.3E-05
Ash	Boron	7440428	LF	Unsaturated	2.6E-03	1.3E-01	4.4E-01	1.8E+00	2.2E+00	2.8E+00	3.9E+00	6.2E+00
Ash	Boron	7440428	SI	Unsaturated	0.0E+00	2.5E-02	1.1E-01	2.2E-01	2.5E-01	3.4E-01	6.6E-01	1.6E+00
Ash	Cadmium	7440439	LF	Saturated	1.6E-01	4.0E-01	7.7E-01	1.8E+00	2.1E+00	3.4E+00	5.1E+00	7.0E+00
Ash	Cadmium	7440439	SI	Saturated	1.6E-01	4.0E-01	7.1E-01	1.7E+00	2.0E+00	3.4E+00	5.1E+00	7.3E+00
Ash	Cadmium	7440439	LF	Unsaturated	2.2E-01	9.8E-01	2.0E+00	4.3E+00	5.2E+00	7.1E+00	9.4E+00	1.3E+01
Ash	Cadmium	7440439	SI	Unsaturated	-4.1E-02	3.1E-01	7.8E-01	2.0E+00	2.7E+00	4.0E+00	6.3E+00	1.0E+01

(continued)

Select Percentiles of K_d Sampled from MINTEQA2 Isotherms by Updated EPACMTP by Waste Type, Metal Species, WMU Type, and Subsurface Domain (continued)

Waste Stream	Metal	CASID	WMU	Zone	Percentiles of K_d							
					10%	25%	50%	75%	80%	85%	90%	95%
Ash	Cobalt	7440484	LF	Saturated	5.2E-01	1.0E+00	4.0E+00	1.6E+01	1.8E+01	3.5E+01	6.1E+01	1.1E+02
Ash	Cobalt	7440484	SI	Saturated	6.5E-01	9.6E-01	2.7E+00	1.1E+01	1.6E+01	2.9E+01	6.7E+01	1.2E+02
Ash	Cobalt	7440484	LF	Unsaturated	2.4E-01	2.6E+00	8.7E+00	2.8E+01	3.5E+01	4.6E+01	7.1E+01	1.1E+02
Ash	Cobalt	7440484	SI	Unsaturated	-2.9E-02	1.3E+00	3.3E+00	1.0E+01	1.3E+01	1.9E+01	3.9E+01	8.7E+01
Ash	Lead	7439921	LF	Saturated	9.0E+00	1.6E+01	2.5E+01	3.9E+01	4.3E+01	5.0E+01	9.7E+01	1.7E+02
Ash	Lead	7439921	SI	Saturated	8.3E+00	1.6E+01	2.2E+01	3.5E+01	4.0E+01	4.5E+01	6.3E+01	1.8E+02
Ash	Lead	7439921	LF	Unsaturated	0.0E+00	2.0E+01	3.3E+01	5.2E+01	5.9E+01	7.0E+01	9.6E+01	1.6E+02
Ash	Lead	7439921	SI	Unsaturated	-1.1E-01	1.3E+00	1.9E+01	3.3E+01	3.6E+01	4.1E+01	4.9E+01	1.3E+02
Ash	Mercury	7439976	LF	Saturated	2.0E-05	1.0E-04	4.4E-04	2.1E-03	2.4E-03	4.6E-03	5.6E-03	9.5E-03
Ash	Mercury	7439976	LF	Unsaturated	8.9E-02	2.0E+00	5.4E+00	1.1E+01	1.3E+01	1.7E+01	2.3E+01	3.7E+01
Ash	Molybdenum	7439987	LF	Saturated	7.8E-07	3.0E-05	1.8E-03	6.0E-02	1.2E-01	2.3E-01	3.7E-01	5.9E-01
Ash	Molybdenum	7439987	SI	Saturated	3.2E-07	3.0E-05	4.3E-03	6.0E-02	1.1E-01	2.1E-01	2.4E-01	4.0E-01
Ash	Molybdenum	7439987	LF	Unsaturated	2.3E-02	1.6E-01	3.4E-01	9.1E-01	1.1E+00	1.5E+00	2.0E+00	3.1E+00
Ash	Molybdenum	7439987	SI	Unsaturated	6.0E-11	5.0E-02	1.6E-01	4.0E-01	5.3E-01	7.0E-01	1.3E+00	1.6E+00
Ash	Nitrate/Nitrite	14797558	LF	Saturated	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00
Ash	Nitrate/Nitrite	14797558	SI	Saturated	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00
Ash	Nitrate/Nitrite	14797558	LF	Unsaturated	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00
Ash	Nitrate/Nitrite	14797558	SI	Unsaturated	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00
Ash	Selenium4	10026036	LF	Saturated	3.3E-01	7.3E+00	1.5E+02	1.2E+03	2.0E+03	2.9E+03	3.6E+03	5.8E+03
Ash	Selenium4	10026036	SI	Saturated	4.5E-02	2.9E+00	2.4E+02	1.3E+03	2.0E+03	3.0E+03	3.6E+03	5.8E+03
Ash	Selenium4	10026036	LF	Unsaturated	1.8E-01	5.4E+00	1.3E+02	8.6E+02	1.5E+03	2.3E+03	3.2E+03	5.2E+03
Ash	Selenium4	10026036	SI	Unsaturated	0.0E+00	2.0E+00	1.6E+02	9.4E+02	1.7E+03	2.5E+03	3.4E+03	5.6E+03
Ash	Selenium6	7782492	LF	Saturated	5.1E-13	6.3E-11	3.0E-07	2.7E-04	6.7E-04	1.8E-03	2.7E-03	4.9E-03
Ash	Selenium6	7782492	SI	Saturated	7.4E-09	6.8E-07	8.4E-05	9.5E-04	1.4E-03	2.2E-03	3.2E-03	4.5E-03
Ash	Selenium6	7782492	LF	Unsaturated	3.3E-03	1.1E-01	2.0E-01	9.1E-01	1.3E+00	1.7E+00	2.5E+00	4.0E+00
Ash	Selenium6	7782492	SI	Unsaturated	0.0E+00	2.2E-02	1.1E-01	2.1E-01	2.3E-01	2.8E-01	5.0E-01	1.6E+00
Ash	Thallium	7440280	LF	Saturated	1.1E-02	1.7E-02	2.7E-02	4.3E-02	5.2E-02	6.3E-02	8.1E-02	1.1E-01
Ash	Thallium	7440280	SI	Saturated	1.2E-02	1.7E-02	2.5E-02	4.4E-02	5.2E-02	6.6E-02	8.5E-02	1.6E-01

(continued)

Select Percentiles of K_d Sampled from MINTEQA2 Isotherms by Updated EPACMTP by Waste Type, Metal Species, WMU Type, and Subsurface Domain (continued)

Waste Stream	Metal	CASID	WMU	Zone	Percentiles of K_d							
					10%	25%	50%	75%	80%	85%	90%	95%
Ash	Thallium	7440280	LF	Unsaturated	5.5E-02	2.0E-01	4.0E-01	1.5E+00	1.8E+00	2.4E+00	3.2E+00	4.8E+00
Ash	Thallium	7440280	SI	Unsaturated	0.0E+00	8.1E-02	1.7E-01	2.9E-01	3.5E-01	4.9E-01	9.3E-01	1.6E+00
Ash & Coal	Aluminum	7429905	LF	Saturated	1.8E-07	1.8E-07	4.0E-07	1.3E-01	1.3E-01	1.3E-01	3.5E-01	2.8E+00
Ash & Coal	Aluminum	7429905	SI	Saturated	1.8E-07	4.0E-07	7.3E-03	1.9E-01	2.0E-01	8.8E-01	5.8E+00	1.4E+01
Ash & Coal	Aluminum	7429905	LF	Unsaturated	0.0E+00	1.2E-01	1.1E+01	4.7E+01	5.9E+01	7.6E+01	9.8E+01	1.5E+02
Ash & Coal	Aluminum	7429905	SI	Unsaturated	1.4E-02	1.1E-01	2.6E-01	9.4E-01	1.5E+00	1.7E+00	2.9E+00	6.3E+00
Ash & Coal	Antimony	7440360	LF	Saturated	1.8E-03	5.4E-03	1.7E-02	7.3E-02	8.0E-02	1.5E-01	3.2E-01	5.5E-01
Ash & Coal	Antimony	7440360	LF	Unsaturated	0.0E+00	1.8E-01	1.4E+00	6.4E+00	8.2E+00	1.1E+01	1.4E+01	2.2E+01
Ash & Coal	Arsenic3	22569728	LF	Saturated	5.5E-03	1.7E-02	6.4E-02	1.9E-01	2.7E-01	4.6E-01	7.4E-01	9.9E-01
Ash & Coal	Arsenic3	22569728	SI	Saturated	4.6E-03	1.6E-02	5.1E-02	1.4E-01	2.6E-01	4.5E-01	7.1E-01	1.1E+00
Ash & Coal	Arsenic3	22569728	LF	Unsaturated	0.0E+00	1.6E-01	3.8E-01	1.9E+00	2.6E+00	3.6E+00	5.4E+00	9.8E+00
Ash & Coal	Arsenic3	22569728	SI	Unsaturated	3.8E-02	1.1E-01	2.3E-01	6.3E-01	8.8E-01	1.3E+00	1.6E+00	2.1E+00
Ash & Coal	Arsenic5	15584040	LF	Saturated	3.3E-02	7.2E-01	6.3E+00	3.3E+01	4.6E+01	5.4E+01	9.5E+01	3.2E+02
Ash & Coal	Arsenic5	15584040	SI	Saturated	3.3E-02	3.5E-01	2.3E+00	1.4E+01	2.1E+01	2.9E+01	4.8E+01	1.5E+02
Ash & Coal	Arsenic5	15584040	LF	Unsaturated	0.0E+00	4.8E-01	6.4E+00	3.4E+01	4.6E+01	6.3E+01	9.8E+01	2.2E+02
Ash & Coal	Arsenic5	15584040	SI	Unsaturated	1.9E-01	9.6E-01	3.5E+00	1.6E+01	2.2E+01	3.2E+01	5.1E+01	1.3E+02
Ash & Coal	Barium	7440393	LF	Saturated	3.7E-02	4.5E-02	2.3E-01	1.3E+00	1.3E+00	1.5E+00	1.9E+00	2.2E+00
Ash & Coal	Barium	7440393	SI	Saturated	1.3E-02	4.5E-02	1.8E-01	1.3E+00	1.3E+00	1.5E+00	2.1E+00	2.2E+00
Ash & Coal	Barium	7440393	LF	Unsaturated	0.0E+00	4.6E-01	1.5E+01	5.0E+01	6.2E+01	7.4E+01	1.0E+02	1.6E+02
Ash & Coal	Barium	7440393	SI	Unsaturated	5.2E-02	5.0E-01	2.4E+00	9.6E+00	1.4E+01	2.1E+01	3.7E+01	5.4E+01
Ash & Coal	Boron	7440428	LF	Saturated	7.1E-08	2.1E-07	8.1E-07	2.4E-06	3.5E-06	5.9E-06	9.5E-06	1.3E-05
Ash & Coal	Boron	7440428	SI	Saturated	6.0E-08	2.1E-07	6.3E-07	1.8E-06	3.6E-06	6.4E-06	1.0E-05	1.5E-05
Ash & Coal	Boron	7440428	LF	Unsaturated	0.0E+00	5.2E-02	1.5E-01	2.2E-01	2.5E-01	3.2E-01	4.4E-01	6.9E-01
Ash & Coal	Boron	7440428	SI	Unsaturated	7.9E-07	3.1E-02	1.1E-01	2.0E-01	2.3E-01	2.8E-01	5.6E-01	1.6E+00
Ash & Coal	Cadmium	7440439	LF	Saturated	1.7E-03	4.6E-02	1.5E-01	7.7E-01	1.0E+00	2.1E+00	2.9E+00	4.7E+00
Ash & Coal	Cadmium	7440439	SI	Saturated	1.7E-03	4.6E-02	8.5E-02	6.1E-01	1.0E+00	2.1E+00	3.2E+00	4.5E+00
Ash & Coal	Cadmium	7440439	LF	Unsaturated	0.0E+00	2.6E-01	7.9E-01	2.2E+00	2.7E+00	3.9E+00	5.9E+00	9.0E+00
Ash & Coal	Cadmium	7440439	SI	Unsaturated	6.4E-02	1.7E-01	4.0E-01	1.6E+00	2.2E+00	3.3E+00	4.4E+00	7.1E+00

(continued)

Select Percentiles of K_d Sampled from MINTEQA2 Isotherms by Updated EPACMTP by Waste Type, Metal Species, WMU Type, and Subsurface Domain (continued)

Waste Stream	Metal	CASID	WMU	Zone	Percentiles of K_d							
					10%	25%	50%	75%	80%	85%	90%	95%
Ash & Coal	Cobalt	7440484	LF	Saturated	7.6E-03	6.6E-02	5.8E-01	2.8E+00	3.2E+00	5.4E+00	8.5E+00	2.9E+01
Ash & Coal	Cobalt	7440484	SI	Saturated	7.6E-03	6.5E-02	2.1E-01	2.4E+00	2.9E+00	4.1E+00	6.1E+00	1.1E+01
Ash & Coal	Cobalt	7440484	LF	Unsaturated	0.0E+00	7.9E+00	3.1E+01	9.3E+01	1.1E+02	1.5E+02	2.9E+02	5.7E+02
Ash & Coal	Cobalt	7440484	SI	Unsaturated	2.3E-01	4.5E-01	1.7E+00	4.9E+00	5.9E+00	7.3E+00	1.1E+01	2.1E+01
Ash & Coal	Lead	7439921	LF	Saturated	1.1E-01	2.3E+00	6.6E+00	2.1E+01	2.9E+01	3.8E+01	4.1E+01	6.3E+01
Ash & Coal	Lead	7439921	SI	Saturated	1.1E-01	2.3E+00	3.7E+00	2.2E+01	2.9E+01	3.9E+01	4.4E+01	6.3E+01
Ash & Coal	Lead	7439921	LF	Unsaturated	0.0E+00	4.4E+00	1.4E+01	4.1E+01	4.9E+01	5.6E+01	7.1E+01	1.2E+02
Ash & Coal	Lead	7439921	SI	Unsaturated	-6.3E-03	9.5E-01	4.5E+00	2.0E+01	3.0E+01	3.9E+01	4.6E+01	6.4E+01
Ash & Coal	Mercury	7439976	LF	Saturated	9.0E-04	2.1E-03	1.3E-02	7.9E-02	3.3E-01	6.8E-01	2.9E+00	4.3E+00
Ash & Coal	Mercury	7439976	SI	Saturated	1.7E-03	3.3E-03	6.2E-02	8.8E-01	1.5E+00	2.8E+00	2.9E+00	4.3E+00
Ash & Coal	Mercury	7439976	LF	Unsaturated	0.0E+00	1.6E+00	6.3E+00	1.5E+01	1.8E+01	2.2E+01	3.0E+01	4.4E+01
Ash & Coal	Mercury	7439976	SI	Unsaturated	4.8E-02	1.5E-01	3.6E-01	1.6E+00	1.6E+00	2.1E+00	3.0E+00	4.4E+00
Ash & Coal	Molybdenum	7439987	LF	Saturated	3.4E-06	6.7E-05	2.5E-03	4.6E-02	7.5E-02	1.6E-01	2.7E-01	7.1E-01
Ash & Coal	Molybdenum	7439987	SI	Saturated	3.4E-06	3.1E-04	1.1E-02	7.5E-02	7.5E-02	1.6E-01	3.1E-01	9.4E-01
Ash & Coal	Molybdenum	7439987	LF	Unsaturated	0.0E+00	7.3E-02	2.1E-01	7.7E-01	1.0E+00	1.4E+00	2.0E+00	2.9E+00
Ash & Coal	Molybdenum	7439987	SI	Unsaturated	4.7E-03	8.3E-02	2.0E-01	5.0E-01	7.6E-01	1.3E+00	1.6E+00	2.2E+00
Ash & Coal	Nitrate/Nitrite	14797558	LF	Saturated	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00
Ash & Coal	Nitrate/Nitrite	14797558	SI	Saturated	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00
Ash & Coal	Nitrate/Nitrite	14797558	LF	Unsaturated	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00
Ash & Coal	Nitrate/Nitrite	14797558	SI	Unsaturated	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00
Ash & Coal	Selenium4	10026036	LF	Saturated	3.7E-01	6.1E+00	5.3E+01	5.3E+02	8.5E+02	1.1E+03	1.9E+03	6.5E+03
Ash & Coal	Selenium4	10026036	SI	Saturated	3.7E-01	1.6E+01	1.2E+02	7.7E+02	9.1E+02	1.4E+03	3.3E+03	7.3E+03
Ash & Coal	Selenium4	10026036	LF	Unsaturated	0.0E+00	2.9E+00	4.4E+01	3.2E+02	5.5E+02	8.7E+02	1.5E+03	5.1E+03
Ash & Coal	Selenium4	10026036	SI	Unsaturated	4.7E-01	1.0E+01	1.0E+02	6.1E+02	8.6E+02	1.1E+03	2.9E+03	7.0E+03
Ash & Coal	Selenium6	7782492	LF	Saturated	2.9E-08	7.7E-08	1.8E-05	6.4E-04	1.1E-03	1.4E-03	3.2E-03	8.8E-03
Ash & Coal	Selenium6	7782492	SI	Saturated	4.7E-08	3.2E-06	1.1E-04	9.6E-04	1.1E-03	1.6E-03	5.1E-03	1.0E-02
Ash & Coal	Selenium6	7782492	LF	Unsaturated	0.0E+00	5.9E-02	1.6E-01	7.0E-01	1.0E+00	1.5E+00	2.3E+00	3.7E+00
Ash & Coal	Selenium6	7782492	SI	Unsaturated	1.7E-03	5.2E-02	1.3E-01	2.6E-01	3.4E-01	5.6E-01	1.2E+00	1.6E+00

(continued)

Select Percentiles of K_d Sampled from MINTEQA2 Isotherms by Updated EPACMTP by Waste Type, Metal Species, WMU Type, and Subsurface Domain (continued)

Waste Stream	Metal	CASID	WMU	Zone	Percentiles of K_d							
					10%	25%	50%	75%	80%	85%	90%	95%
Ash & Coal	Thallium	7440280	LF	Saturated	2.9E-04	1.9E-03	9.8E-03	2.0E-02	2.4E-02	3.7E-02	5.0E-02	7.7E-02
Ash & Coal	Thallium	7440280	LF	Unsaturated	0.0E+00	1.2E-01	3.9E-01	1.3E+00	1.7E+00	2.1E+00	2.7E+00	3.9E+00
FBC	Aluminum	7429905	LF	Saturated	4.1E-25	4.6E-25	1.4E-17	5.2E-08	1.7E-07	1.7E-07	5.3E-07	7.1E-03
FBC	Aluminum	7429905	LF	Unsaturated	0.0E+00	1.5E-01	2.5E-01	4.6E+02	8.6E+02	1.9E+03	3.9E+03	8.7E+03
FBC	Antimony	7440360	LF	Saturated	3.0E-04	1.2E-02	2.0E-02	6.9E-02	8.7E-02	1.1E-01	1.4E-01	2.3E-01
FBC	Antimony	7440360	LF	Unsaturated	0.0E+00	2.1E-01	4.1E-01	2.2E+00	3.0E+00	4.3E+00	7.2E+00	1.8E+01
FBC	Arsenic3	22569728	LF	Saturated	4.5E-02	1.0E-01	3.9E-01	5.0E-01	6.1E-01	6.4E-01	6.6E-01	6.7E-01
FBC	Arsenic3	22569728	LF	Unsaturated	0.0E+00	4.6E-01	1.1E+00	2.1E+00	7.0E+00	1.6E+01	3.4E+01	1.5E+02
FBC	Arsenic5	15584040	LF	Saturated	3.3E+00	1.0E+01	3.4E+01	7.3E+01	9.3E+01	9.4E+01	1.6E+02	2.4E+02
FBC	Arsenic5	15584040	LF	Unsaturated	0.0E+00	1.1E+01	3.8E+01	8.2E+01	9.9E+01	1.1E+02	1.7E+02	2.5E+02
FBC	Barium	7440393	LF	Saturated	1.6E-01	2.1E-01	4.6E-01	1.4E+00	5.7E+00	7.8E+00	9.3E+00	1.1E+01
FBC	Barium	7440393	LF	Unsaturated	0.0E+00	7.7E+00	7.1E+01	2.7E+02	3.7E+02	4.7E+02	6.3E+02	1.0E+03
FBC	Boron	7440428	LF	Saturated	5.5E-07	1.3E-06	4.9E-06	6.3E-06	7.8E-06	8.1E-06	8.2E-06	8.7E-06
FBC	Boron	7440428	LF	Unsaturated	0.0E+00	1.6E-01	2.6E-01	2.0E+00	2.5E+00	3.2E+00	4.6E+00	8.2E+00
FBC	Cadmium	7440439	LF	Saturated	2.0E-01	2.5E-01	5.9E-01	2.7E+00	3.3E+00	3.4E+00	4.0E+00	5.3E+00
FBC	Cadmium	7440439	LF	Unsaturated	0.0E+00	9.5E-01	2.4E+00	6.3E+00	7.3E+00	8.5E+00	1.0E+01	2.1E+01
FBC	Cobalt	7440484	LF	Saturated	4.7E-01	1.1E+00	5.8E+00	4.4E+01	4.6E+01	7.0E+01	7.3E+01	9.5E+01
FBC	Cobalt	7440484	LF	Unsaturated	0.0E+00	2.2E+00	8.9E+00	4.1E+01	5.5E+01	7.3E+01	9.0E+01	1.1E+02
FBC	Lead	7439921	LF	Saturated	6.8E+00	9.9E+00	2.1E+01	6.9E+01	1.1E+02	1.2E+02	1.7E+02	1.8E+02
FBC	Lead	7439921	LF	Unsaturated	0.0E+00	1.3E+01	2.5E+01	8.3E+01	1.0E+02	1.3E+02	1.5E+02	2.0E+02
FBC	Mercury	7439976	LF	Saturated	1.6E-05	4.2E-05	2.2E-04	1.6E-03	3.1E-03	4.4E-03	5.8E-03	7.0E-03
FBC	Mercury	7439976	LF	Unsaturated	0.0E+00	1.1E+00	6.7E+00	1.9E+01	2.8E+01	4.5E+01	7.8E+01	2.1E+02
FBC	Molybdenum	7439987	LF	Saturated	7.5E-07	8.0E-06	1.3E-04	3.1E-03	7.8E-03	1.3E-02	2.7E-02	4.5E-02
FBC	Molybdenum	7439987	LF	Unsaturated	0.0E+00	1.5E-01	2.3E-01	6.7E-01	9.6E-01	1.3E+00	1.9E+00	3.7E+00
FBC	Nitrate/Nitrite	14797558	LF	Saturated	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00
FBC	Nitrate/Nitrite	14797558	LF	Unsaturated	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00
FBC	Selenium4	10026036	LF	Saturated	6.4E-02	1.2E+00	1.8E+01	1.2E+02	2.8E+02	4.8E+02	8.8E+02	1.4E+03
FBC	Selenium4	10026036	LF	Unsaturated	0.0E+00	2.1E+00	1.9E+01	1.5E+02	2.9E+02	4.8E+02	7.9E+02	1.3E+03

(continued)

Select Percentiles of K_d Sampled from MINTEQA2 Isotherms by Updated EPACMTP by Waste Type, Metal Species, WMU Type, and Subsurface Domain (continued)

Waste Stream	Metal	CASID	WMU	Zone	Percentiles of K_d							
					10%	25%	50%	75%	80%	85%	90%	95%
FBC	Selenium6	7782492	LF	Saturated	1.7E-08	1.8E-07	2.9E-06	5.9E-05	1.4E-04	2.3E-04	3.8E-04	6.9E-04
FBC	Selenium6	7782492	LF	Unsaturated	0.0E+00	1.4E-01	2.3E-01	1.6E+00	2.3E+00	3.5E+00	7.6E+00	2.1E+01
FBC	Thallium	7440280	LF	Saturated	9.5E-03	1.5E-02	2.3E-02	5.1E-02	5.3E-02	6.2E-02	1.1E-01	2.3E-01
FBC	Thallium	7440280	LF	Unsaturated	0.0E+00	2.3E-01	6.2E-01	1.9E+00	2.2E+00	2.7E+00	3.3E+00	5.0E+00

D.7 References

Norris, C.H., and C.E. Hubbard. 1999. *Use of MINTEQA2 and EPACMTP to Estimate Groundwater Pathway Risks from the Land Disposal of Metal-Bearing Wastes*. Prepared for The Environmental Defense Fund, Friends of the Earth, Hoosier Environmental Council, and Mineral Policy Center. Available in docket number EPA-HQ-RCRA-2001-0022 as document number EPA-HQ-RCRA-2001-0022-0610.

U.S. EPA (Environmental Protection Agency). 1991. *MINTEQ2A/PRODEF2A, A Geochemical Assessment Model for Environmental Systems: Version 3.0 User's Manual*. EPA/600/3-91/021. Office of Research and Development, Athens, GA. Available at http://nepis.epa.gov/EPA/html/Pubs/pubtitleORD.htm.

U.S. EPA (Environmental Protection Agency). 1998. *Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions. Update to Methodology for Assessing Health Risks Associated with Indirect Exposure to Combustor Emissions*. EPA-600/R-98/137. National Center for Environmental Assessment, Cincinnati, OH.

U.S. EPA (Environmental Protection Agency). 2000. Discussion and Recommendations Concerning Comments Received by the U.S. EPA on the EPACMTP and MINTEQA2 Models as Used in Modeling Support for FFCW Rulemaking. Prepared by Allison Geoscience Consultants, Inc. for the Office of Solid Waste, Washington, DC.

U.S. EPA (Environmental Protection Agency). 2001a. Modeling Sensitivity Analyses Concerning Comments Received by the U.S. EPA on the MINTEQA2 Model as Used in Modeling Support for FFCW Rulemaking. Prepared by Allison Geoscience Consultants, Inc., for the Office of Solid Waste, Washington, DC.

U.S. EPA (Environmental Protection Agency). 2001b. *Revisions in Input Data and Modeling Procedures for Using MINTEQA2 in Estimating Metal Partition Coefficients*. Prepared by Allison Geoscience Consultants, Inc., for the Office of Solid Waste, Washington, DC.

U.S. EPA (Environmental Protection Agency). 2003a. *EPACMTP Parameters/Data Document*. Office of Solid Waste, Washington, DC. Available at http://www.epa.gov/waste/nonhaz/industrial/tools/cmtp/index.htm.

U.S. EPA (Environmental Protection Agency). 2003b. *EPACMTP Technical Background Document*. Office of Solid Waste, Washington, DC. Available at http://www.epa.gov/waste/nonhaz/industrial/tools/cmtp/index.htm.

U.S. EPA (Environmental Protection Agency). 2003c. *Estimation of Metal Partition Coefficients for Fossil Fuel Combustion Waste Risk Assessment*. Prepared by Allison Geoscience Consultants, Inc. (under subcontract to RTI International) for U.S. EPA. June.

U.S. EPA (Environmental Protection Agency). 2003d. *Addendum to the EPACMTP Technical Background Document*. Office of Solid Waste, Washington, DC.

Appendix E. Surface Water, Fish Concentration, and Contaminant Intake Equations

This appendix presents the equations used to model surface water and fish concentrations and intake of drinking water and fish. These equations are presented in the following attachments:

- **Attachment E-1** provides the equations comprising the surface water equilibrium partitioning model, including equations that estimate steady state concentrations in the water column (dissolved and total) and sediments.
- **Attachment E-2** provides the equations that use bioconcentration factors (BCFs) to calculate fish tissue concentrations from total.
- **Attachment E-3** provides the equations used to calculate daily contaminant intake rates from drinking water and fish consumption.

E.1 Aluminum Surface Water Precipitation

Because the fate and transport of aluminum is controlled more by solubility than by sorption in surface water, the surface water model includes algorithms to estimate aluminum concentrations in the water column and sediments by accounting for precipitation and fallout of aluminum in the water column. These calculations proceed in a stepwise fashion, as follows.

Step 1. Initially, assume all influent aluminum is dissolved in the water column.

Fraction in water column (fwater) = 1
Fraction in sediment layer (fbenth) = 0
Fraction dissolved (fd) = 1

Total water column concentration (Cwctot) = dissolved water column concentration (Cwd).

Step 2. Compare the dissolved water column concentration (Cwd) to the maximum soluble concentration (Csol) calculated in MINTEQA2 for the waterbody pH (see **Section 3.5.4**, Table 3-6 for aluminum solubilities and **Section C.6.3**, Table C-11 for waterbody pH).

Step 3. If the dissolved water concentration (Cwd) is greater than the solubility limit (Csol), reset the dissolved water concentration to the solubility limit, and precipitate and settle out the excess aluminum to the benthic sediment layer.

If Cwctot > Csol, then

```
        Fwater  =  Csol / Cwctot
        Fbenth  =  (Cwctot - Csol) / Cwctot
          Cwbs  =  (Cwctot - Csol) * dwc / db
         Cwtot  =  Cwctot * dwc / dz
           Cdw  =  Csol
        Cwctot  =  Csol
Else
           Cdw  =  Cwctot
          Cwbs  =  0
         Cwtot  =  Cwctot * rsParam!dwc / rsParam!dz
End If
```

where:

```
           Cdw  =  issolved waterbody concentration
          Csol  =  maximum soluble concentration
          Cwbs  =  total concentration in bed sediment
         Cwtot  =  total waterbody concentration from loading
            db  =  depth of the upper benthic layer
           dwc  =  depth of the water column
            dz  =  depth of the waterbody
        fbenth  =  fraction in sediment layer
            fd  =  fraction dissolved
        fwater  =  fraction in water column.
```

Table E-1-1. Fraction of Contaminant in Water Column (Unitless)

f_{Water}

$$d_w = d_z - d_b$$

$$f_{Water} = \frac{\left[1 + \left(K_{dsw} \times TSS \times 0.000001\right)\right] \times \frac{d_w}{d_z}}{\left[\left[1 + \left(K_{dsw} \times TSS \times 0.000001\right)\right] \times \frac{d_w}{d_z}\right] + \left[\left(bsp + K_{dbs} \times bsc\right) \times \frac{d_b}{d_z}\right]}$$

Name	Description	Value
bsc	Bed sediment particle concentration (g/cm^3) or (kg/L)	1
bsp	Bed sediment porosity (cm^3/cm^3)	0.6
d_b	Depth of upper benthic layer (m)	0.03
d_w	Depth of water column (m)	Site Data; See Appendix C
d_z	Depth of the waterbody (m)	Calculated
K_{dbs}	Sediment-water partition coefficient (mL/g)	Chemical Data; See Section 3
K_{dsw}	Suspended sediment-water partition coefficient (mL/g)	Chemical Data; See Section 3
TSS	Total suspended solids (mg/L)	Site Data; See Appendix C
0.000001	Conversion factor (L/mL)(g/mg)	

Table E-1-2. Fraction of Contaminant in Benthic Sediments (Unitless)

f_{Benth}

$$f_{Benth} = \frac{(bsp + K_{dbs} \times bsc) \times \frac{d_b}{d_z}}{\left[(1 + K_{dsw} \times TSS \times 0.000001) \times \frac{d_W}{d_z}\right] + \left[(bsp + K_{dbs} \times bsc) \times \frac{d_b}{d_z}\right]}$$

Name	Description	Value
bsc	Bed sediment particle concentration (g/cm^3) or (kg/L)	1
bsp	Bed sediment porosity (cm^3/cm^3)	0.6
d_b	Depth of upper benthic layer (m)	0.03
d_w	Depth of water column (m)	Site Data; See Appendix C
d_z	Depth of the waterbody (m)	Calculated
K_{dbs}	Sediment-water partition coefficient (mL/g)	Chemical Data; See Section 3
K_{dsw}	Suspended sediment-water partition coefficient (mL/g)	Chemical Data; See Section 3
TSS	Total suspended solids (mg/L)	Site Data; See Appendix C
0.000001	Conversion factor (L/mL)(g/mg)	

Table E-1-3. Dissolved Fraction (Unitless)

f_d

$$f_d = \frac{1}{1 + K_{dsw} \times TSS \times 0.000001}$$

Name	Description	Value
K_{dsw}	Suspended sediment-water partition coefficient (mL/g)	Chemical Data; See Section 3
TSS	Total suspended solids (mg/L)	Site Data; See Appendix C
0.000001	Conversion factor (L/mL)(g/mg)	

Table E-1-4. Liquid-Phase Transfer Coefficient - Lakes (m/day) (Mercury Only)

KL_{lakes}

$$KL_{Lakes} = \sqrt{C_d} \times u_w \times \sqrt{\frac{R_a}{R_w}} \times \frac{k^{0.33}}{L_2} \times \left(\frac{{}_w}{R_w \times D_w} \right)^{-0.67} \times 86400$$

Name	Description	Value
C_d	Drag coefficient (unitless)	0.0011
D_w	Diffusivity in water (cm^2/s)	Chemical Data; See Section 3, Table 3-4
k	von Karman's constant (unitless)	0.4
L_2	Viscous sublayer thickness (unitless)	4
$_w$	Viscosity of water (g/cm-s)	0.0169
R_a	Density of air (g/cm^3)	0.0012
R_w	Density of water (g/cm^3)	1
u_w	Mean annual wind speed (m/sec)	Site Data; See Appendix C
86400	Conversion factor (sec/day)	

Table E-1-5. Liquid-Phase Transfer Coefficient - Rivers (m/day) (Mercury Only)

KL_{rivers}

$$KL_{Rivers} = \sqrt{\frac{D_w \times U \times 0.0001}{d_z}} \times 86400$$

Name	Description	Value
D_w	Diffusivity in water (cm^2/s)	Chemical Data; See Section 3, Table 3-4
d_z	Depth of the waterbody (m)	Calculated
U	Current velocity of the waterbody (m/s)	Site Data; See Appendix C
86400	Conversion factor (sec/day)	

Table E-1-6. Gas-Phase Transfer Coefficient - Lakes (m/day) (Mercury Only)

K_{gas}

$$K_{gas} = \sqrt{C_d} \times u_w \times \frac{k^{0.33}}{L_2} \times \left(\frac{a}{R_a \times D_a} \right)^{-0.67} \times 86400$$

Name	Description	Value
C_d	Drag coefficient (unitless)	0.0011
D_a	Diffusivity of chemical in air (cm^2/s)	Chemical Data; See Section 3, Table 3-4
k	von Karman's constant (unitless)	0.4
L_2	Viscous sublayer thickness (unitless)	4
A	Viscosity of air (g/cm-s)	0.000181
R_a	Density of air (g/cm^3)	0.0012
u_w	Mean annual wind speed (m/sec)	Site Data; See Appendix C
86400	Conversion factor (sec/day)	

Table E-1-7. Diffusion Transfer Rate (m/day) (Mercury Only)

K_v

$$Tempadjust = \theta_{water}^{(T_w - T_{hlc})}$$

$$K_v = \frac{1}{\frac{1}{K_L} + \frac{1}{K_g \times H'}} \times TempAdjust$$

$$H' = \frac{HLC}{R \times T_w}$$

Name	Description	Value
HLC	Henry's Law constant (atm-m^3/mole	Chemical Data; See Section 3, Table 3-4
H'	Dimensionless Henry's Law constant (unitless)	Calculated
K_g	Gas-Phase Transfer Coefficient (m/d)	Calculated
K_L	Liquid-Phase Transfer Coefficient (m/d)	Calculated
R	Ideal Gas Constant (atm-m^3/K-mole)	0.00008205
θ_{water}	Temperature correction (unitless)	1.026
T_{hlc}	Temperature of HLC (K)	298
T_w	Temperature of the waterbody (K)	Site Data; See Appendix C

Note: Drawn from U.S. EPA, 1998 (EPA-530-D-98-001A and EPA-600/R-98/137).

Table E-1-8. Water Concentration Dissipation Rate Constant (1/day)

K_{wt}

$$K_{wt} = \left(f_{Water} \times f_d \times k_{vol}\right) + \left(f_{benth} \times K_b\right) + \left(f_{Water} \times k_{sw}\right) + \left(f_{benth} \times k_{sed}\right) + k_h$$

$$K_b = \frac{WB}{d_b}$$

$$k_{vol} = \frac{K_v}{d_w}$$

Name	Description	Value
d_b	Depth of upper benthic layer (m)	0.03
d_w	Depth of water column (m)	Site Data; See Appendix C
F_{benth}	Fraction of contaminant in benthic sediments (unitless)	Calculated
f_d	Dissolved fraction (unitless)	Calculated
f_{Water}	Fraction of contaminant in water column (unitless)	Calculated
K_b	Benthic burial rate constant (1/day)	Calculated
k_h	Hydrolysis rate (1/day)	0
k_{sed}	Degradation rate for sediment (1/day)	0
k_{sw}	Degradation rate for water column (1/day)	0
K_v	Diffusion transfer rate (m/day)	Calculated (mercury only)
k_{vol}	Water column volatilization rate constant (1/day)	Calculated (mercury only)
WB	Rate of Burial (m/day)	0

Table E-1-7. Diffusion Transfer Rate (m/day) (Mercury Only)

K_v

$$Tempadjust = \theta_{water}^{(T_w - T_{hlc})}$$

$$K_v = \frac{1}{\frac{1}{K_L} + \frac{1}{K_g \times H'}} \times TempAdjust$$

$$H' = \frac{HLC}{R \times T_w}$$

Name	Description	Value
HLC	Henry's Law constant (atm-m^3/mole	Chemical Data; See Section 3, Table 3-4
H'	Dimensionless Henry's Law constant (unitless)	Calculated
K_g	Gas-Phase Transfer Coefficient (m/d)	Calculated
K_L	Liquid-Phase Transfer Coefficient (m/d)	Calculated
R	Ideal Gas Constant (atm-m^3/K-mole)	0.00008205
θ_{water}	Temperature correction (unitless)	1.026
T_{hlc}	Temperature of HLC (K)	298
T_w	Temperature of the waterbody (K)	Site Data; See Appendix C

Note: Drawn from U.S. EPA, 1998 (EPA-530-D-98-001A and EPA-600/R-98/137).

Table E-1-10. Total Water Column Concentration (g/m^3 or mg/L)

C_{wcTot}

$$d_w = d_z - d_b$$

$$C_{wcTot} = C_{wTot} \times f_{water} \times \frac{d_z}{d_w}$$

Name	Description	Value
Cw_{Tot}	Total Waterbody Concentration from Loading (g/m^3 or mg/L)	Calculated
d_b	Depth of upper benthic layer (m)	0.03
d_w	Depth of water column (m)	Site Data; See Appendix C
d_z	Depth of the waterbody (m)	Calculated
f_{Water}	Fraction of contaminant in water column (unitless)	Calculated

Table E-1-11. Dissolved Waterbody Concentration (mg/L)

C_{dw}

$$d_w = d_z - d_b$$

$$C_{dw} = Cw_{Tot} \times f_{Water} \times f_d \times \frac{d_Z}{d_w}$$

Name	Description	Value
Cw_{Tot}	Total Waterbody Concentration from Loading (g/m^3 or mg/L)	Calculated
d_b	Depth of upper benthic layer (m)	0.03
d_w	Depth of water column (m)	Site Data; See Appendix C
d_z	Depth of the waterbody (m)	Calculated
f_d	Dissolved fraction (unitless)	Calculated
f_{Water}	Fraction of contaminant in water column (unitless)	Calculated

Table E-1-12. Total Concentration in Bed Sediment (g/m^3 or mg/L)

C_{wbs}

$$d_z = d_w + d_b$$

$$C_{bs} = C_{wTot} \times f_{benth} \times \frac{d_z}{d_b}$$

Name	Description	Value
Cw_{Tot}	Total Waterbody Concentration from Loading (g/m^3 or mg/L)	Calculated
d_b	Depth of upper benthic layer (m)	0.03
d_w	Depth of water column (m)	Site Data; See Appendix C
d_z	Depth of the waterbody (m)	Calculated

Table E-2-1. Concentration in Fish at Different Trophic Levels (mg/kg)

C_{fish}

For Mercury:

$$C_{fish} = 0.15 * C_{dw} \times BCF$$

For Non-Volatile Metals:

$$C_{fish} = Cw_{tot} \times BCF$$

Name	Description	Value
BCF	Bioconcentration factor for specified trophic level (L/kg)	Chemical Data; See Section 3
C_{dw}	Dissolved waterbody concentration (mg/L)	Calculated
Cw_{Tot}	Total waterbody concentration from loading (g/m^3 or mg/L)	Calculated
0.15	Fraction of dissolved mercury assumed to be methyl mercury (unitless)	

Table E-2-2. Average Fish Fillet Concentration Ingested by Humans (mg/kg)

C_{fish_fillet}

$$C_{fish\_fillet} = F_{T3} \times C_{fishT3F} + F_{T4} \times C_{fishT4F}$$

Name	Description	Value
$C_{fishT3F}$	Concentration of contaminant in fish at different trophic levels (mg/kg)	Calculated
$C_{fishT4F}$	Concentration of contaminant in fish at different trophic levels (mg/kg)	Calculated
F_{T3}	Fraction of trophic level 3 intake (unitless)	0.36
F_{T4}	Fraction of trophic level 4 intake (unitless)	0.64

Table E-3-1. Contaminant Intake from Drinking Water (mg/kg-d)

Idw

$$I_{dw} = \frac{C_{dw} \times CR_{dw} \times F_{dw}}{BW * 1000}$$

Name	Description	Value
BW	Body weight (kg)	Exposure Data; See Appendix F
C_{dw}	Dissolved waterbody concentration (mg/L)	Calculated
CR_{dw}	Consumption rate of water (mL/day)	Exposure Data; See Appendix F
F_{dw}	Fraction of drinking water ingested that is contaminated (unitless)	1
1000	Conversion factor (mL/L)	

Table E-3-2. Daily Intake of Contaminant from Fish Ingestion (mg/kg BW/day)

I_{fish}

$$I_{fish} = \frac{C_{fish\_fillet} \times CR_{fish} \times F_{fish}}{1000 \times BW}$$

Name	**Description**	**Value**
BW	Body weight (kg)	Exposure Data; See Appendix F
C_{fish_fillet}	Average fish fillet concentration ingested by humans (mg/kg)	Calculated
CR_{fish}	Consumption rate of fish (g WW/day)	Exposure Data; See Appendix F
F_{fish}	Fraction of fish intake from contaminated source (unitless)	1
1000	Conversion factor (g/kg)	

Appendix F. Human Exposure Factors

Exposure factors are data that quantify human behavior patterns (e.g., ingestion rates of fish and drinking water) and characteristics (e.g., body weight) that affect a person's exposure to environmental contaminants. These data can be used to construct realistic assumptions concerning an individual's exposure to and subsequent intake of a contaminant in the environment. The exposure factors data also enable EPA to differentiate the exposures of individuals of different ages (e.g., a child vs. an adult). The derivation and values used for the human exposure factors in this risk assessment are described below, and the exposure factors selected for the probabilistic analyses are also presented.

F.1 Exposure Parameters Used in Probabilistic Analysis

F.1.1 Introduction

The general methodology for collecting human exposure data for the probabilistic analysis relied on the *Exposure Factors Handbook*, or EFH (U.S. EPA, 1997a-c), which was used in one of three ways:

1. When EFH percentile data were adequate (most input variables), maximum likelihood estimation was used to fit selected parametric models (gamma, lognormal, Weibull, and generalized gamma) to the EFH data. The chi-square measure of goodness of fit was then used to choose the best distribution. Parameter uncertainty information (e.g., for averages, standard deviations) also was derived using the asymptotic normality of the maximum likelihood estimate or a regression approach.

2. When EFH percentile data were not adequate for statistical model fitting (a few variables), models were selected on the basis of results for other age cohorts or, if no comparable information was available, by assuming lognormal as a default distribution and reasonable coefficients of variation (CVs).

3. When data were not adequate for either 1 or 2 above, variables were fixed at EFH-recommended mean values or according to established EPA policy.

Table F-1 lists all of the parameters used in the probabilistic analysis. Both fixed variables and the values used to define distributed data are provided.

Probabilistic risk analyses involve "sampling" values from probability distribution functions (PDFs) and using the values to estimate risk. In some cases, distributions are infinite, and there is a probability, although very small, that very large or very small values might be selected from the distributions. Because selecting extremely large or extremely small values is unrealistic (e.g., the range of adult body weights is not infinite), maximum and minimum values

were imposed on the distributions. The minimum and maximum values are included in Table F-1.

F.1.2 Exposure Parameter Distribution Methodology

This section describes how stochastic or distributed input data for each exposure factor were collected and processed. Exposure parameter distributions were developed for use in the Monte Carlo analysis. For most variables for which distributions were developed, exposure factor data from the EFH were analyzed to fit selected parametric models (i.e., gamma, lognormal, Weibull). Steps in the development of distributions included preparing data, fitting models, assessing fit, and preparing parameters to characterize distributional uncertainty in the model inputs.

For many exposure factors, EFH data include sample sizes and estimates of the following parameters for specific receptor types and age groups: mean, standard deviation, standard error, and percentiles corresponding to a subset of the following probabilities: 0.01, 0.02, 0.05, 0.10, 0.15, 0.25, 0.50, 0.75, 0.85, 0.90, 0.95, 0.98, and 0.99. These percentile data, where available, were used as a basis for fitting distributions. Although in no case were all of these percentiles actually provided for a single factor, seven or more are typically present in the EFH data. Therefore, using the percentiles was a fuller use of the available information than fitting distributions simply based on the method of moments (e.g., selecting models that agree with the data mean and standard deviation). For some factors, certain percentiles were not used in the fitting process because sample sizes were too small to justify their use. Percentiles were used only if at least one data point was in the tail of the distribution. If the EFH data repeated a value across several adjacent percentiles, only one value (the most central or closest to the median) was used in most cases (e.g., if both the 98th and 99th percentiles had the same value, only the 98th percentile value was used).

The EFH does not use standardized age cohorts across exposure factors. Data for different exposure factors are reported for different age categories. Therefore, to obtain the percentiles for fitting the four standardized age cohorts (i.e., ages 1 to 5, 6 to 11, 12 to 19, and more than 20), each EFH cohort-specific value for a given exposure factor was assigned to one of these four cohorts. When multiple EFH cohorts fitted into a single CCW cohort, the EFH percentiles were averaged within each CCW cohort (e.g., data on 1- to 2-year-olds and 3- to 5-year-olds from EFH were averaged for the CCW 1- to 5-year-old cohort). If sample sizes were available, weighted averages were used, with weights proportional to sample sizes. If sample sizes were not available, equal weights were assumed (i.e., the percentiles were simply averaged).

Table F-1. Summary of Exposure Parameters Used in Probabilistic Analysis

Parameter	Units	Variable Type	Constants	Mean (or shape)	Std Dev (or scale)	Minimum	Maximum	Reference
Averaging time for carcinogens	yr	Constant	7.00E+01					U.S. EPA (1989)
Body weight (adult)	kg	Lognormal		7.12E+01	1.33E+01	1.50E+01	3.00E+02	U.S. EPA (1997a); Tables 7-2, 7-4, 7-5
Body weight (child 1)	kg	Lognormal		1.55E+01	2.05E+00	4.00E+00	5.00E+01	U.S. EPA (1997a); Tables 7-3, 7-6, 7-7
Body weight (child 2)	kg	Lognormal		3.07E+01	5.96E+00	6.00E+00	2.00E+02	U.S. EPA (1997a); Tables 7-3, 7-6, 7-7
Body weight (child 3)	kg	Lognormal		5.82E+01	1.02E+01	1.30E+01	3.00E+02	U.S. EPA (1997a); Tables 7-3, 7-6, 7-7
Consumption rate: fish (adult, child)	g/d	Lognormal		6.48E+00	1.99E+01	0.00E+00	1.50E+03	U.S. EPA (1997b); Table 10-64
Exposure duration (adult resident)	yr	Weibull		1.34E+00	1.74E+01	1.00E+00	5.00E+01	U.S. EPA (1999) (ACS)
Exposure duration (child)	yr	Weibull		1.32E+00	7.06E+00	1.00E+00	5.00E+01	U.S. EPA (1999) (ACS)
Exposure frequency (adult resident)	d/yr	Constant	3.50E+02					U.S. EPA Policy
Fraction contaminated: drinking water	Fraction	Constant	1.00E+00					U.S. EPA Policy
Fraction contaminated: fish	Fraction	Constant	1.00E+00					U.S. EPA Policy
Fraction of fish consumed that is trophic level (T3) fish	Fraction	Constant	3.60E-01					U.S. EPA (1997b); Table 10-66
Fraction of fish consumed that is trophic level 4 (T4) fish	Fraction	Constant	6.40E-01					U.S. EPA (1997b); Table 10-66
Ingestion rate: drinking water (adult resident)	mL/d	Gamma		3.88E+00	3.57E+02	1.04E+02	1.10E+04	U.S. EPA (1997a); Table 3-6
Ingestion rate: drinking water (child 1 resident)	mL/d	Gamma		2.95E+00	2.37E+02	2.60E+01	3.84E+03	U.S. EPA (1997a); Table 3-6
Ingestion rate: drinking water (child 2 resident)	mL/d	Gamma		3.35E+00	2.35E+02	3.40E+01	4.20E+03	U.S. EPA (1997a); Table 3-6
Ingestion rate: drinking water (child 3 resident)	mL/d	Gamma		2.82E+00	3.42E+02	3.30E+01	5.40E+03	U.S. EPA (1997a); Table 3-6

Because the EFH data are always positive and are almost always skewed to the right (i.e., have a long right tail), three two-parameter probability models commonly used to characterize such data (gamma, lognormal, and Weibull) were selected. In addition, a three-parameter model (generalized gamma) was used that unifies them[1] and allows for a likelihood ratio test of the fit of the two-parameter models. However, only the two-parameter models were selected for use in the analysis because the three-parameter generalized gamma model did not significantly improve the goodness of fit over the two-parameter models. This simple setup constitutes a considerable improvement over the common practice of using a lognormal model in which adequate EFH data are available to support maximum likelihood estimation.

Lognormal, gamma, Weibull, and generalized gamma distributions were fit to each factor data set using maximum likelihood estimation (Burmaster and Thompson, 1998). When sample sizes were available, the goodness of fit was calculated for each of the four models using the chi-square test (Bickel and Doksum, 1977). When percentile data were available but sample sizes were unknown, a regression F-test for the goodness of fit against the generalized gamma model was used. For each of the two-parameter models, parameter uncertainty information (i.e., mean, standard deviation, scale, and shape) was provided as parameter estimates for a bivariate normal distribution that could be used for simulating parameter values (Burmaster and Thompson, 1998). The information necessary for such simulations includes estimates of the two model parameters, their standard errors, and their correlation. To obtain this parameter uncertainty information, the asymptotic normality of the maximum likelihood estimate (Burmaster and Thompson, 1998) was used when sample sizes were available, and a regression approach was used when sample sizes were not available (Jennrich and Moore, 1975; Jennrich and Ralston, 1979). In either case, uncertainty can be expressed as a bivariate normal distribution for the model parameters.

The parameter values selected are described in more detail in the following subsections. **Section F.1.3** discusses fixed parameters. **Section F.1.4** describes, for each exposure factor, the EFH data used to develop the distributions, along with the final distributional statistics.

F.1.3 Fixed Parameters

Certain parameters were fixed, based on central tendency values from the best available source (usually EFH recommendations), either because no variability was expected or because the available data were not adequate to generate distributions. Fixed (constant) parameters are shown in **Table F-2** along with the value selected for the risk analysis and the data source. These constants included variables for which limited or no percentile data were provided in the EFH: exposure frequency, fractions of T3 and T4 fish consumed, and fraction contaminated for the various media. Most of these values were extracted directly from the EFH. When evaluating carcinogens, total dose was averaged over the lifetime of the individual, assumed to be 70 years.

[1] Gamma, Weibull, and lognormal distributions are all special cases of the generalized gamma distribution.

Table F-2. Summary of Human Exposure Factor Data Used in Modeling: Constants

Description	Value	Units	Source
Fraction contaminated: drinking water	1	Fraction	EPA policy
Fraction contaminated: fish	1	Fraction	EPA policy
Fraction of T3 fish consumed	0.36	Fraction	U.S. EPA (1997b); Table 10-66
Fraction of T4 fish consumed	0.64	Fraction	U.S. EPA (1997b); Table 10-66
Exposure frequency (adult, child)	350	d/yr	EPA policy
Averaging time for carcinogens (adult, child)	70	yr	U.S. EPA (1989)

The fraction contaminated for drinking water was assumed to be 1 (i.e., all drinking water available for consumption at a site is potentially contaminated), with actual concentrations depending on fate and transport model results. Thus, households for which the drinking water pathway was analyzed were assumed to get 100 percent of their drinking water from groundwater. Exposure frequency was set to 350 days per year in accordance with EPA policy, assuming that residents take an average of 2 weeks' vacation time away from their homes each year.

F.1.4 Variable Parameters

F.1.4.1 Fish Consumption

Table F-3 presents fish consumption data and distributions. Fish consumption data were obtained from Table 10-64 of the EFH (U.S. EPA, 1997b). Data (in g/d) were available for adult freshwater anglers in Maine. The Maine fish consumption study was one of four recommended freshwater angler studies in the EFH (U.S. EPA, 1997b). The other recommended fish consumption studies (i.e., Michigan and New York) had large percentages of anglers who fished from Great Lakes, which is not consistent with the modeling scenarios used in this risk analysis. The anglers in the Maine study fished from streams, rivers, and ponds; these data were more consistent with the CCW modeling scenarios. Although the Maine data have a lower mean than the Michigan data, the Maine data compared better with a national U.S. Department of Agriculture (USDA) study. Also, the Maine study included percentile data, which were necessary to develop a distribution.

Percentile data were used to fit parametric models (gamma, lognormal, and Weibull), and measures of goodness of fit were used to select lognormal as the most appropriate model. The fraction of fish intake that is locally caught was assumed to be 1 (in accordance with EPA policy). The fraction of consumed T3 and T4 fish was 0.36 and 0.64, respectively (Table 10-66, U.S. EPA, 1997b).

Table F-3. Fish Consumption Data and Distribution

EFH Data (g/d)									Distribution		
Age Cohort	N	Data Mean	Data SD	P50	P66	P75	P90	P95	Distribution	Pop-Estd Mean	Pop-Estd SD
All ages	1,053	6.4		2	4	5.8	13	26	Lognormal	6.48	19.9

N = Number of samples; P50–P95 = Percentiles; Pop-Estd = Population-estimated; SD = Standard deviation.

F.1.4.2 Drinking Water Intake

Table F-4 presents drinking water intake data and distributions. Drinking water intake data were obtained from Table 3-6 of the EFH (U.S. EPA, 1997a). Data (in mL/d) were presented by age groups. Weighted averages of percentiles, means, and standard deviations were calculated for the three child age groups and adults. 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. The fraction of drinking water contaminated was assumed to be 1 (in accordance with EPA policy).

Table F-4. Drinking Water Intake Data and Distributions

EFH Data (mL/d)													Distributions		
Age Cohort	**N**	**Data Mean**	**Data SD**	**P01**	**P05**	**P10**	**P25**	**P50**	**P75**	**P90**	**P95**	**P99**	**Distribution**	**Pop-Estd Mean**	**Pop-Estd SD**
1–5	3,200	697.1	401.5	51.62	187.6	273.5	419.2	616.5	900.8	1,236	1,473	1,917	Gamma	698	406
6–11	2,405	787	417	68	241	318	484	731	1,016	1,338	1,556	1,998	Gamma	787	430
12–19	5,801	963.2	560.6	65.15	241.4	353.8	574.4	868.5	1,247	1,694	2,033	2,693	Gamma	965	574
20+	13,394	1,384	721.6	207.6	457.5	607.3	899.6	1,275	1,741	2,260	2,682	3,737	Gamma	1,383	703

N = Number of samples; P01–P99 = Percentiles; Pop-Estd = Population-estimated; SD = Standard deviation.

F.1.4.3 Body Weight

Table F-5 presents body weight data and distributions. Body weight data were obtained from Tables 7-2 through 7-7 of the EFH (U.S. EPA, 1997a). Data (in kg) were presented by age and gender. Weighted averages of percentiles, means, and standard deviations were calculated for 1- to 5-year-olds, 6- to 11-year-olds, 12- to 19-year olds, and adult age groups; male and female data were weighted and combined for each age group. These percentile data were used as the basis for fitting distributions. These data were analyzed 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.

Table F-5. Body Weight Data and Distributions

EFH Data (kg)													Distributions		
Age Cohort	**N**	**Data Mean**	**Data SD**	**P05**	**P10**	**P15**	**P25**	**P50**	**P75**	**P85**	**P90**	**P95**	**Distribution**	**Pop-Estd Mean**	**Pop-Estd SD**
1–5	3,762	15.52	3.719	12.5	13.1	13.45	14.03	15.26	16.67	17.58	18.32	19.45	Lognormal	15.5	2.05
6–11	1,725	30.84	9.561	22.79	24.05	25.07	26.44	29.58	33.44	36.82	39.66	43.5	Lognormal	30.7	5.96
12–19	2,615	58.45	13.64	43.84	46.52	48.31	50.94	56.77	63.57	68.09	71.98	79.52	Lognormal	58.2	10.2
20+	12,504	71.41	15.45	52.86	55.98	58.21	61.69	69.26	78.49	84.92	89.75	97.64	Lognormal	71.2	13.3

N = Number of samples; P05–P95 = Percentiles; Pop-Estd = Population-estimated; SD = Standard deviation.

F.1.4.4 Exposure Duration

Table F-6 presents exposure duration data and distributions. Exposure duration was assumed to be equivalent to the average residence time for each receptor. Exposure durations for adult and child residents were determined using data on residential occupancy from the EFH Table 15-168 (U.S. EPA, 1997c). The data represent the total time a person is expected to live at a single location, based on age. The table presents male and female data combined. Adult residents aged 21 to 90 were pooled. For child residents, the 3-year-old EFH age group was used for the 1- to 5-year-old CCW cohort. The 6- and 9-year-old EFH age groups were pooled for the 6- to 11-year-old CCW cohort.

Table F-6. Exposure Duration Data and Distributions

EFH Data		Distributions		
Age Cohort	Data Mean (yr)	Distribution	Pop-Estd Shape (yr)[a]	Pop-Estd Scale (yr)
1–5	6.5	Weibull	1.32	7.059
6–11	8.5	Weibull	1.69	9.467
Adult	16.0	Weibull	1.34	17.38

Pop-Estd = Population-estimated.
[a] Distributions used in risk assessment.

In an analysis of residential occupancy data, Myers et al. (U.S. EPA, 2000) found that the data, for most ages, were best fit by a Weibull distribution. The Weibull distribution as implemented in Crystal Ball is characterized by three parameters: location, shape, and scale. Location is the minimum value and, in this case, was presumed to be 0. Shape and scale were determined by fitting a Weibull distribution to the pooled data, as follows. To pool residential occupancy data for the age cohorts, an arithmetic mean of data means was calculated for each age group. Then, assuming a Weibull distribution, the variance within each age group (e.g., 6-year-olds) was calculated in the age cohort. These variances in turn were pooled over the age cohort using equal weights. This is not the usual type of pooled variance, which would exclude the variation in the group means. However, this way, the overall variance reflected the variance of means within the age groups (e.g., within the 6-year-old age group). The standard deviation was estimated as the square root of the variance. The coefficient of variation was calculated as the ratio of the standard deviation divided by the Weibull mean. For each cohort, the population-estimated parameter uncertainty information (e.g., shape and scale) was calculated based on a Weibull distribution, the calculated data mean for the age cohort, and the CV.

F.2 Exposure Parameters Used in Screening Analysis

The 50th percentile values used for the human exposure factors in the screening analysis are presented in **Table F-7**.

Table F-7. 50th Percentile Exposure Data Used in the Screening Analysis

Parameter	Value	Age Cohort				Units	Reference	Table
		1–5 yr	6–11 yr	12–19 yr	20+ yr			
Body weight		15.3	29.6	56.8	69.3	kg	U.S. EPA (1997a)	T7-2, 7-3, 7-4, 7-5, 7-6, 7-7
Consumption rate of fish		2	2	2	2	g WW/d	U.S. EPA (1997b)	T10-64
Exposure duration		5	7.5	8	10	yr	U.S. EPA (1997c)	T15-164, 15-168
Ingestion rate of drinking water		0.6165	0.731	0.8685	1.275	L/d	U.S. EPA (1997a)	T3-6

F.3 References

Bickel, P.J., and K.A. Doksum. 1977. *Mathematical Statistics*. San Francisco, CA: Holden Bay.

Burmaster, D.E, and K.M. Thompson. 1998. Fitting second-order parametric distributions to data using maximum likelihood estimation. *Human and Ecological Risk Assessment 4*(2):319–339.

Jennrich, R.I., and R.H. Moore. 1975. Maximum likelihood estimation by nonlinear least squares. Pp. 57–65 in *Statistical Computing Section Proceedings of American Statistical Association*. American Statistical Association, Alexandria, VA.

Jennrich, R.I., and M.L. Ralston. 1979. Fitting nonlinear models to data. *Annual Review Biophysical Bioengineering 8*:195–238.

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 Office of Emergency and Remedial Response, Washington, DC. December.

U.S. EPA (Environmental Protection Agency). 1997a. *Exposure Factors Handbook, Volume I, General Factors.* EPA/600/P-95/002Fa. Office of Research and Development, Washington, DC. August.

U.S. EPA (Environmental Protection Agency). 1997b. *Exposure Factors Handbook, Volume II, Food Ingestion Factors.* EPA/600/P-95/002Fa. Office of Research and Development, Washington, DC. August.

U.S. EPA (Environmental Protection Agency). 1997c. *Exposure Factors Handbook, Volume III, Activity Factors.* EPA/600/P-95/002Fa. Office of Research and Development, Washington, DC. August.

U.S. EPA (Environmental Protection Agency). 1999. *Revised Risk Assessment for the Air Characteristic Study. Volume II: Technical Background Document*. EPA/530/R-99/019b. Office of Solid Waste, Washington, DC. August.

U.S. EPA (Environmental Protection Agency). 2000. *Options for Development of Parametric Probability Distributions for Exposure Factors.* EPA/600/R-00/058. National Center for Environmental Assessment, Office of Research and Development, Washington, DC. July.

Appendix G. Human Health Benchmarks

The CCW screening analysis and risk assessment require human health benchmarks to assess potential risks from chronic oral exposures. EPA uses reference doses (RfDs) to evaluate noncancer risk from oral exposures. Oral cancer slope factors (CSFs) are used to evaluate risk for carcinogens. This appendix provides the human health benchmarks used in the CCW screening and risk assessment. **Section G.1** describes the data sources and general hierarchy used to collect these benchmarks. **Section G.2** provides the benchmarks along with discussions of individual human health benchmarks extracted from a variety of sources.

G.1 Methodology and Data Sources

Several sources of health benchmarks are available. The hierarchy used health benchmarks developed by EPA to the extent that they were available. The analysis used available benchmarks from non-EPA sources for chemicals for which EPA benchmarks were not available, and ranked human health benchmark sources in the following order of preference:

- Integrated Risk Information System (IRIS)
- Superfund Technical Support Center Provisional Benchmarks
- Health Effects Assessment Summary Tables (HEAST)
- EPA health assessment documents
- Various other EPA health benchmark sources
- Agency for Toxic Substances and Disease Registry (ATSDR) minimal risk levels (MRLs)
- California Environmental Protection Agency (CalEPA) chronic inhalation reference exposure levels (RELs) and cancer potency factors.

G.1.1 Integrated Risk Information System (IRIS)

Benchmarks in IRIS are prepared and maintained by EPA, and RTI used values from IRIS whenever available. IRIS is EPA's electronic database containing information on human health effects (U.S. EPA, 2009). Each chemical file contains descriptive and quantitative information on potential health effects. Health benchmarks for chronic noncarcinogenic health effects include RfDs and inhalation reference concentrations (RfCs). Cancer classification, oral CSFs, and inhalation unit risk factors (URFs) are included for carcinogenic effects. IRIS is the official repository of Agency-wide consensus of human health risk information.

G.1.2 Superfund Provisional Benchmarks

The Superfund Technical Support Center (EPA's National Center for Environmental Assessment [NCEA]) derives provisional RfCs, RfDs, and CSFs for certain chemicals. Some of the provisional values have been externally peer reviewed. These provisional values have not undergone EPA's formal review process for finalizing benchmarks and do not represent Agency-wide consensus information.

G.1.3 Health Effects Assessment Summary Tables

HEAST is a listing of provisional noncarcinogenic and carcinogenic health toxicity values (RfDs, RfCs, URFs, and CSFs) derived by EPA (U.S. EPA, 1997). Although the health toxicity values in HEAST have undergone review and have the concurrence of individual EPA program offices, either they have not been reviewed as extensively as those in IRIS or their data set is not complete enough to be listed in IRIS. HEAST benchmarks have not been updated in several years and do not represent Agency-wide consensus information.

G.1.4 Other EPA Health Benchmarks

EPA has also derived health benchmark values in other risk assessment documents, such as Health Assessment Documents (HADs), Health Effects Assessments (HEAs), Health and Environmental Effects Profiles (HEEPs), Health and Environmental Effects Documents (HEEDs), Drinking Water Criteria Documents, and Ambient Water Quality Criteria Documents. Evaluations of potential carcinogenicity of chemicals in support of reportable quantity adjustments were published by EPA's Carcinogen Assessment Group (CAG) and may include cancer potency factor estimates. Health benchmarks derived by EPA for listing determinations (e.g., solvents) or studies (e.g., Air Characteristic Study) are also available. Health toxicity values identified in these EPA documents are usually dated and are not recognized as Agency-wide consensus information or verified benchmarks.

G.1.5 ATSDR Minimal Risk Levels

The ATSDR MRLs are substance-specific health guidance levels for noncarcinogenic endpoints (ATSDR, 2009). An MRL is an estimate of the daily human exposure to a hazardous substance that is likely to be without appreciable risk of adverse noncancer health effects over a specified duration of exposure. MRLs are based on noncancer health effects only and are not based on a consideration of cancer effects. MRLs are derived for acute, intermediate, and chronic exposure durations for oral and inhalation routes of exposure. Inhalation and oral MRLs are derived in a manner similar to EPA's RfCs and RfDs, respectively (i.e., ATSDR uses the no observed adverse effect level/uncertainty factor [NOAEL/UF] approach); however, MRLs are intended to serve as screening levels and are exposure duration specific. Also, ATSDR uses EPA's (U.S. EPA, 1994) inhalation dosimetry methodology in the derivation of inhalation MRLs.

G.1.6 CalEPA Cancer Potency Factors and Reference Exposure Levels

CalEPA has developed cancer potency factors for chemicals regulated under California's Hot Spots Air Toxics Program (CalEPA, 1999a). The cancer potency factors are analogous to EPA's oral and inhalation CSFs. CalEPA has also developed chronic inhalation RELs, analogous

to EPA's RfC, for 120 substances (CalEPA, 1999b, 2000, 2008). CalEPA used EPA's inhalation dosimetry methodology (U.S. EPA, 1994) in the derivation of inhalation RELs. The cancer potency factors and inhalation RELs have undergone internal peer review by various California agencies and have been the subject of public comment.

G.1.7 Surrogate Health Benchmarks

If no human health benchmarks were available from EPA or alternative sources, we sought benchmarks for similar chemicals to use as surrogate data. For example, the health benchmark of a mixture could serve as the surrogate benchmark for its components or a benchmark of a metal salt could serve as the surrogate for an elemental metal.

G.2 Human Health Benchmarks

The chronic human health benchmarks used to calculate the health-based numbers (HBNs) in the CCW screening analysis and risk assessment are summarized in **Table G-1**, which provides the Chemical Abstract Service Registry Number (CASRN), constituent name, RfD (mg/kg-d), oral CSF (mg/kg-d^{-1}), and reference for each benchmark. A key to the references cited and abbreviations used is provided at the end of the table.

For a majority of constituents, human health benchmarks were available from IRIS (U.S. EPA, 2009), Superfund Provisional Benchmarks, or HEAST (U.S. EPA, 1997). Benchmarks also were obtained from ATSDR (2009) or CalEPA (1999a, 1999b, 2000, 2008). This section describes benchmarks obtained from other sources, along with the Superfund Provisional Benchmarks values and special uses of IRIS benchmarks.

Table G-1. Human Health Benchmarks Used in CCW Risk Assessment

Constituent Name	CASRN	RfD (mg/kg-d)	Ref	CSFo (per mg/kg-d)	Ref	MCL (mg/L)	Notes
Aluminum	7429-90-5	1.0E+00	P				
Antimony	7440-36-0	4.0E-04	I				
Arsenic, inorganic	7440-38-2	3.0E-04	I	1.5E+0	I		
Barium	7440-39-3	2.0E-01	I				
Beryllium	7440-41-7	2.0E-03	I				
Boron	7440-42-8	2.0E-01	I				
Cadmium	7440-43-9	5.0E-04	I				RfD for H_2O (food = 1E-3)
Chloride	16887-00-6					250	
Chromium (III), insoluble salts	16065-83-1	1.5E+00	I				
Chromium (VI)	18540-29-9	3.0E-03	I				
Cobalt (and compounds)	7440-48-4	3.0E-04	P				
Copper	7440-50-8	1.0E-02	A			1.3	RfD is the intermediate oral MRL
Cyanide (amenable)	57-12-5	2.0E-02	I				

(continued)

Human Health Benchmarks Used in CCW Risk Assessment (continued)

Constituent Name	CASRN	RfD (mg/kg-d)	Ref	CSFo (per mg/kg-d)	Ref	MCL (mg/L)	Notes
Divalent mercury		3.0E-04	H				RfD is for mercuric chloride; used for food, water, soil
		1.0E-04	I				RfD is for methyl mercury; used for fish only
Fluoride	16984-48-8	1.2E-01	I				RfD is for fluorine; the alternative IRIS value (for skeletal, rather than dental, fluorosis) was used
Iron	7439-89-6	7.0E-01	P				
Lead and compounds (inorganic)	7439-92-1					0.015	
Manganese	7439-96-5	1.4E-01	I				RfD for food; H_2O and soil = 4.7E-2 mkd
Molybdenum	7439-98-7	5.0E-03	I				
Nickel, soluble salts	7440-02-0	2.0E-02	I				
Nitrate	14797-55-8	1.6E+00	I			10	
Nitrite	14797-65-0	1.0E-01	I				
Selenium	7782-49-2	5.0E-03	I				
Silver	7440-22-4	5.0E-03	I				
Strontium	7440-24-6	6.0E-01	I				
Sulfate	14808-79-8					250	
Thallium, elemental	7440-28-0	8.0E-05	I				RfD is for thallium chloride
Vanadium	7440-62-2	7.0E-03	H				
Zinc	7440-66-6	3.0E-01	I				

Key:
CASRN = Chemical Abstract Service registry number.
CSFo = Oral cancer slope factor.
RfD = Reference dose.
MCL = Maximum Contaminant Level.
Sources:
A = ATSDR MRLs (ATSDR, 2009)
H = HEAST (U.S. EPA, 1997)
I = IRIS (U.S. EPA, 2009)
P = PPRTV (U.S. EPA, 2006a, 2006b, 2008)

The provisional RfD of 1 mg/kg-d developed by NCEA for the Superfund Technical Support Center (U.S. EPA, 2006a) was used for aluminum.

The provisional RfD of 0.0003 mg/kg-d developed by NCEA for the Superfund Technical Support Center (U.S. EPA, 2008) was used for cobalt.

The provisional RfD of 0.7 mg/kg-d developed by NCEA for the Superfund Technical Support Center (U.S. EPA, 2006b) was used for iron.

For several constituents, IRIS benchmarks for similar chemicals were used as surrogate data. The rationale for these recommendations is as follows:

- Fluoride was based on fluorine. The IRIS RfD for fluorine is based on soluble fluoride. The primary RfD cited in IRIS (6E-02 mg/kg-d) is for dental fluorosis, a cosmetic effect. In this analysis, an alternative IRIS value (1.2E-01 mg/kg-d) for skeletal fluorosis in adults was used instead.

- Thallium was based on thallium chloride. IRIS contains RfDs for several thallium salts. The lowest value among the thallium salts (8E-05 mg/kg-d) is routinely used to represent thallium in risk assessments.

G.3 References

ATSDR (Agency for Toxic Substances and Disease Registry). 2009. *Minimal Risk Levels (MRLs) for Hazardous Substances.* Available at http://www.atsdr.cdc.gov/mrls.html.

CalEPA (California Environmental Protection Agency). 1999a. *Air Toxics Hot Spots Program Risk Assessment Guidelines: Part II. Technical Support Document for Describing Available Cancer Potency Factors*. Office of Environmental Health Hazard Assessment, Berkeley, CA. Available at http://www.oehha.org/air/cancer_guide/hsca2.html.

CalEPA (California Environmental Protection Agency). 1999b. *Air Toxics Hot Spots Program Risk Assessment Guidelines: Part III. Technical Support Document for the Determination of Noncancer Chronic Reference Exposure Levels*. SRP Draft. Office of Environmental Health Hazard Assessment, Berkeley, CA. Available (in two sections) at http://www.oehha.org/air/chronic_rels/ragsii.htmlhttp://www.oehha.org/air/chronic_rels/RAGSp3draft.html.

CalEPA (California Environmental Protection Agency). 2000. *Air Toxics Hot Spots Program Risk Assessment Guidelines: Part III. Technical Support Document for the Determination of Noncancer Chronic Reference Exposure Levels*. Office of Environmental Health Hazard Assessment, Berkeley, CA. Available (in four sections) at http://www.oehha.org/air/chronic_rels/22RELS2k.html http://www.oehha.org/air/chronic_rels/42kChREL.html http://www.oehha.org/air/chronic_rels/Jan2001ChREL.html http://www.oehha.org/air/chronic_rels/1201Crels.html.

CalEPA (California Environmental Protection Agency). 2008. *Air Toxics Hot Spots Program Risk Assessment Guidelines. Technical Support Document for the Determination of Noncancer Chronic Reference Exposure Levels*. Office of Environmental Health Hazard Assessment, Berkeley, CA. Available at http://www.oehha.org/air/hot_spots/2008/NoncancerTSD_final.pdf.

U.S. EPA (Environmental Protection Agency). 1997. *Health Effects Assessment Summary Tables (HEAST)*. EPA-540-R-97-036. FY 1997 Update. Office of Solid Waste and Emergency Response, Washington, DC.

U.S. EPA (Environmental Protection Agency). 2006a. *Provisional Peer Reviewed Toxicity Values for Aluminum (CASRN 7429-90-5).* 10-23-2006. National Center for Environmental Assessment. Superfund Technical Support Center, Cincinnati, OH.

U.S. EPA (Environmental Protection Agency). 2006b. *Provisional Peer Reviewed Toxicity Values for Iron and Compounds (CASRN 7439-89-6). Derivation of Subchronic and Chronic Oral RfDs.* 9-11-2006. National Center for Environmental Assessment. Superfund Technical Support Center, Cincinnati, OH.

U.S. EPA (Environmental Protection Agency). 2008. *Provisional Peer Reviewed Toxicity Values for Cobalt (CASRN 7440-48-4).* 8-25-2008. National Center for Environmental Assessment. Superfund Technical Support Center, Cincinnati, OH.

U.S. EPA (Environmental Protection Agency). 2009. Integrated Risk Information System (IRIS). National Center for Environmental Assessment, Office of Research and Development, Washington, DC. Available at http://www.epa.gov/iris/.

Appendix H. Ecological Benchmarks

Both the screening and full-scale CCW assessments included an ecological risk assessment that paralleled the human health risk assessment. The ecological risk assessment addressed two routes of exposure for ecological receptors: direct contact with contaminated media and ingestion of contaminated food items. For each CCW chemical for which ecological effect data were available, hazard quotients (HQs) were calculated using chemical-specific media concentrations assumed to be protective of ecological receptors of concern.

This appendix provides the ecological benchmarks used in both the CCW screening and full-scale risk assessment. **Section H.1** describes the data sources and methods used to develop these benchmarks. Additional details can be found in U.S. EPA (1998). **Section H.2** provides the benchmarks.

H.1 Data Sources and Methodology

To calculate ecological HQs, the concentration-based ecological benchmarks (also known as chemical stressor concentration limits, or CSCLs) were divided by the estimated concentrations of constituents in environmental media contaminated by CCW. The CSCLs are environmental quality criteria intended to represent a protective threshold value for adverse effects to various ecological receptors in aquatic ecosystems (surface water and sediment). An HQ greater than target of 1 indicates that the predicted concentration will be above the CSCL and, therefore, the potential for adverse ecological effects exists. In this regard, the use of CSCLs to calculate an ecological HQ is analogous to the use of the reference concentration (RfC) for human health where the air concentration is compared to the health-based concentration (the RfC), and an HQ greater than the target value of 1 is considered to indicate the potential for adverse health effects. **Table H-1** shows the receptor types assessed for each exposure route in each environmental medium addressed by the CCW risk assessment.

Table H-1. Ecological Receptors Assessed by Medium Impacted by CCW

Receptor Type	Surface Water	Sediment
Direct Contact Exposure		
Aquatic Community	✔	
Sediment Community		✔
Amphibians	✔	
Aquatic Plants and Algae	✔	
Ingestion Exposure		
Mammals	✔	
Birds	✔	

Ecological benchmarks for the CCW risk assessment were taken directly from the 1998 fossil fuel combustion risk analysis, *Non-Groundwater Pathways, Human Health and Ecological*

Risk Analysis for Fossil Fuel Combustion Phase 2 (FFC2) (U.S. EPA, 1998). The receptors and endpoints selected for the 1998 analysis were evaluated and considered appropriate for the goals of this risk assessment. The benchmarks were derived for each chemical and receptor to the extent that supporting data were available.

As in 1998, the lowest (most sensitive) benchmark for each chemical in each medium was selected to calculate HQs in the CCW risk assessment. For example, several receptors (aquatic invertebrates, mammals, and birds) may be exposed to constituents in surface water. The surface water HQ for a given chemical was calculated using whichever benchmark was lowest and would thus give the highest (most protective) HQ.

H.1.1 Direct Contact Exposure

Ecological receptors that live in close contact with contaminated media are considered to be potentially at risk. These receptors are exposed through direct contact with contaminants in surface water and sediment. The receptors selected to assess the direct contact exposure route for each medium are summarized in Table H-1. The benchmarks for receptor communities are not truly *community-level* concentration limits in that they do not consider predator-prey interactions. Rather, they are based on the theory that protection of 95 percent of the species in the community will provide a sufficient level of protection for the community (see, for example, Stephan et al., 1985, for additional detail). The following sections summarize the benchmark derivation methods for each receptor assessed for the direct contact route of exposure.

Aquatic Community Benchmarks

The aquatic community receptor comprises fish and aquatic invertebrates exposed through direct contact with constituents in surface water. For the aquatic community, the final chronic value (FCV), developed either for the Great Lakes Water Quality Initiative (U.S. EPA, 1993) or the National Ambient Water Quality Criteria (NAWQC) (U.S. EPA, 1995a,b), was the preferred source for the benchmark. If an FCV was unavailable and could not be calculated from available data, a secondary chronic value (SCV) was estimated using methods developed for wildlife criteria for the Great Lakes Initiative (e.g., 58 FR 20802; U.S. EPA, 1993). The SCV methodology is based on the original species data set established for the NAWQC; however, it requires fewer data points and includes statistically derived adjustment factors. For benchmark derivation, the minimum data set required at least one data point.

Amphibian Benchmarks

For amphibian populations, data availability severely limited benchmark development. A review of several compendia presenting amphibian ecotoxicity data (e.g., U.S. EPA, 1996; Power et al., 1989), as well as primary literature sources, found a lack of standard methods on endpoints, species, and test durations necessary to derive a chronic benchmark for amphibians. Consequently, an acute benchmark was derived for aqueous exposures in amphibians by taking a geometric mean of LC_{50} (i.e., concentration lethal to 50 percent of test subjects) data identified in studies with exposure durations less than 8 days. Although the use of acute effects levels produced a benchmark that was not consistent with the other (chronic) ecological benchmarks, the sensitivity of these receptors warranted the use of acute effects levels in the absence of chronic concentration limits. Recent studies (Hopkins and Rowe, 2004; Hopkins et al., 2006)

have confirmed that amphibians are among the most sensitive taxa to metals found in CCW, and selenium appears to be a significant stressor in CCW disposal scenarios. The endpoints considered in these studies were related to population sustainability and, consequently, are highly relevant to ecological risk assessment. However, these field studies were confounded by the fact that wildlife were exposed to multiple chemical pollutants (including radionuclides) and, as a result, acute effects data on individual metals remain the most appropriate source for quantitative benchmarks to assess the potential for adverse effects in amphibians.

Sediment Community Benchmarks

For the sediment community, benchmarks were selected based on a complete assessment of several sources proposing sediment benchmark values. Primary sources evaluated for developing sediment community benchmarks are shown in **Table H-2**.

Table H-2. Primary Sources Evaluated for Developing Sediment Community Benchmarks

Long, E.R., and L.G. Morgan. 1991. *The Potential for Biological Effects of Sediment-Sorbed Contaminants Tested in the National Status and Trends Program*. Technical Memorandum NOS OMA 52. National Oceanic and Atmospheric Administration (NOAA), Washington, DC.
Jones, D.S., G.W. Suter, II, and R.N. Hull. 1997. *Toxicological Benchmarks for Screening Contaminants of Potential Concern for Effects on Sediment-Associated Biota: 1997 Revision*. Oak Ridge National Laboratory, Oak Ridge, TN.
U.S. EPA (Environmental Protection Agency). 1997. *Protocol for Screening Level Ecological Risk Assessment at Hazardous Waste Combustion Facilities*. Internal Review Draft, February 28. Office of Solid Waste, Washington, DC.
U.S. EPA (Environmental Protection Agency). 1995. *Technical Support Document for the Hazardous Waste Identification Rule: Risk Assessment for Human and Ecological Receptors*. Office of Solid Waste, Washington, DC.
MacDonald, D.D. 1994. *Approach to the Assessment of Sediment Quality in Florida Coastal Waters. Volume 1*. Florida Department of Environmental Protection, Tallahassee, FL.

Algae and Aquatic Plant Benchmarks

For algae and aquatic plants, adverse effects concentrations were identified in the open literature or from a data compilation presented in *Toxicological Benchmarks for Screening Potential Contaminants of Concern for Effects on Aquatic Biota: 1996 Revision* (Suter and Tsao, 1996). For most contaminants, studies were not available for aquatic vascular plants, and lowest effects concentrations were identified for algae. The benchmark for algae and aquatic plants was based on (1) an LOEC for vascular aquatic plants or (2) an effective concentration (EC_{xx}) for a species of freshwater algae, frequently a species of green algae (e.g., *Selenastrum capricornutum*). Because of the lack of data for this receptor group and the differences between vascular aquatic plants and algae sensitivity, the lowest value of those identified was usually chosen.

H.1.2 Ingestion Exposure

The ingestion route of exposure addresses the exposure of terrestrial mammals and birds through ingestion of aquatic plants and prey. Thus, the CCW ecological benchmarks for ingestion exposure express media concentrations that, based on certain assumptions about receptor diet and foraging behavior, were expected to be protective of populations of mammals and birds feeding and foraging in contaminated surface water bodies.

The derivation of ingestion benchmarks began with the selection of appropriate ecotoxicological data based on a hierarchy of data sources. The assessment endpoint for the CCW ecological risk assessment was population viability; therefore, ecological benchmarks were developed from measures of reproductive/developmental success or, if unavailable, from other effects that could conceivably impair population dynamics. Population-level benchmarks were preferred over benchmarks for individual organisms; however, very few population-level benchmarks have been developed. Therefore, the CCW risk assessment used benchmarks derived from individual organism studies, and protection was inferred at the population level.

Once an appropriate ingestion exposure study was identified, a benchmark was calculated using a three-step process. The remainder of this section outlines the basic technical approach used to convert avian or mammalian benchmarks (in daily doses) to the media concentration benchmarks (in units of concentration) used to assess ecological risks for surface water and sediment contaminated by CCW waste constituents. The methods reflect exposure through the ingestion of contaminated plants, prey, and media, and include parameters on accumulation (e.g., bioconcentration factors), uptake (e.g., consumption rates), and dietary preferences.

Step 1: Scale Benchmark

The benchmarks derived for test species can be extrapolated to wildlife receptor species within the same taxon using a cross-species scaling equation (Equation H-1) (Sample et al., 1996). This is the default methodology EPA proposed for carcinogenicity assessments and reportable quantity documents for adjusting animal data to an equivalent human dose (57 FR 24152).

$$Benchmark_w = LOAEL_t \times \left(\frac{bw_t}{bw_w} \right)^{1/4} \quad \text{(H-1)}$$

where

$Benchmark_w$ = scaled ecological benchmark for species *w* (mg/kg/d)
$LOAEL_t$ = lowest observed adverse effects level for test species (mg/kg/d)
bw_t = body weight of the surrogate test species (kg)
bw_w = body weight of the representative wildlife species (kg).

Step 2: Identify Bioconcentration Factors/Bioaccumulation Factors

For metal constituents, whole-body bioconcentration factors (BCFs) and bioaccumulation factors (BAFs) were identified for aquatic organisms that could be used as food sources (e.g., fish). The Oak Ridge National Laboratory has proposed methods and data that are useful in predicting bioaccumulation (Sample et al. 1998a,b). These values were typically identified in the open literature and EPA references.

Step 3: Calculate Benchmarks

The following equation provided the basis for calculating surface water benchmarks using a population-inference benchmark (e.g., endpoint on fecundity).

$$Benchmark = \frac{\left[I_{fish} \times \left(BAF \;\; C_w\right)\right] + \left(I_w \times C_w\right)}{bw} \quad \text{(H-2)}$$

where

I_{fish} = intake of contaminated fish (kg/d)
BAF = whole-body bioaccumulation factor (L/kg)
bw = weight of the representative species (kg)
I_w = intake of contaminated water (L/d)
C_w = total concentration in the water (mg/L).

For chemicals that bioaccumulate significantly in fish tissue, the ingestion of contaminated food tends to dominate the exposure (i.e., $[I_{fish} \times C_{fish}] >> [I_w C_w]$), and the water term (i.e., $[I_w \times C_w]$) can be dropped from Equation H-2, resulting in Equation H-3:

$$Benchmark \;\; = \;\; \frac{I_{fish} \;\times\; \left(BAF \;\times\; C_w\right)}{bw} \quad \text{(H-3)}$$

At the benchmark dose (mg/kg/d), the concentration in water is equivalent to the chemical stressor concentration limit for that receptor as a function of body weight, ingestion rate, and the bioaccumulation potential for the chemical of concern. Hence, Equation H-3 can be rewritten to solve for the surface water ($CSCL_{sw}$) as follows:

$$CSCL_{sw} \;\; = \;\; \frac{benchmark \;\times\; bw}{I_w \;+\; \left(I_{fish} \;\times\; BAF\right)} \quad \text{(H-4)}$$

H.2 Ecological Benchmarks

The ecological benchmarks used to calculate ecological HQs in the CCW risk assessment are summarized in **Table H-3**, which provides the constituent name; the criterion and receptor for sediment and aquatic receptors; and the source for each benchmark.

Table H-3. Ecological Benchmarks Used in the CCW Risk Assessment

Constituent	Sediment Criterion (mg/kg)	Sediment Receptor	Aquatic Criterion (mg/L)	Aquatic Receptor	Source
Aluminum	ID	--	0.09	Aquatic Biota	U.S. EPA (1998)
Antimony	2	Sediment biota	0.03	Aquatic Biota	U.S. EPA (1998)
Arsenic total	0.51	Spotted sandpiper	ID	--	U.S. EPA (1998)
Arsenic III	ID	--	0.15	Aquatic Biota	U.S. EPA (1998)
Arsenic IV	ID	--	8.10E-03	Aquatic Biota	U.S. EPA (1998)
Barium	190	Spotted sandpiper	4.00E-03	Aquatic Biota	U.S. EPA (1998)
Beryllium	ID	--	6.60E-04	Aquatic Biota	U.S. EPA (1998)
Boron	ID	--	1.60E-03	Aquatic Biota	U.S. EPA (1998)
Cadmium	0.68	Sediment biota	2.50E-03	Aquatic Biota	U.S. EPA (1998)
Chromium total	16.63	Spotted sandpiper	ID	--	U.S. EPA (1998)
Chromium IV	ID	--	0.09	Aquatic Biota	U.S. EPA (1998)
Chromium VI	ID	--	0.01	Aquatic Biota	U.S. EPA (1998)
Cobalt	ID	--	0.02	Aquatic Biota	U.S. EPA (1998)
Copper	18.7	Sediment biota	9.30E-03	Aquatic Biota	U.S. EPA (1998)
Lead	0.22	Spotted sandpiper	3.00E-04	River Otter	U.S. EPA (1998)
Mercury	0.11	Spotted sandpiper	1.90E-07	Kingfisher	U.S. EPA (1998)
Molybdenum	34	Spotted sandpiper	0.37	Aquatic Biota	U.S. EPA (1998)
Nickel	15.9	Sediment biota	0.05	Aquatic Biota	U.S. EPA (1998)
Selenium total	ID	--	5.00E-03	Aquatic Biota	U.S. EPA (1998)
Selenium IV	ID	--	0.03	Aquatic Biota	U.S. EPA (1998)
Selenium VI	ID	--	9.50E-03	Aquatic Biota	U.S. EPA (1998)
Silver	0.73	Sediment biota	3.60E-04	Aquatic Biota	U.S. EPA (1998)
Thallium	ID	--	0.01	Aquatic Biota	U.S. EPA (1998)
Vanadium	18	Spotted sandpiper	0.02	Aquatic Biota	U.S. EPA (1998)
Zinc	120	Sediment biota	0.12	Aquatic Biota	U.S. EPA (1998)

ID = insufficient data.

H.3 References

Hopkins, W.A., and C.L. Rowe. 2004. Impacts of aquatic disposal of coal combustion by-products on vertebrates on the Savannah River Site. p. 141–147, In *Proceedings of State Regulation of Coal Combustion By-Product Placement at Mine Sites*. May 4–6, 2004, Harrisburg, PA. U.S. Department of Interior, Office of Surface Mining, Alton, IL.

Hopkins, W.A., S.E. DuRant, B.P. Staub, C.L. Rowe, and B.P. Jackson. 2006. Reproduction, embryonic development, and maternal transfer of contaminants in the amphibian *Gastrophryne carolinensis*. *Environmental Health Perspectives 114*(5):661-666.

Power, T., K.L. Clark, A. Harfenist, and D.B. Peakall. 1989. *A Review and Evaluation of the Amphibian Toxicological Literature.* Technical Report Series No. 61. Canadian Wildlife Service, Environment Canada, Hull, Quebec.

Sample, B.E., D.M. Opresko, and G.W. Suter, II. 1996. *Toxicological Benchmarks for Wildlife: 1996 Revision*. ES/ER/TM-86/R3. Oak Ridge National Laboratory, Oak Ridge, TN.

Sample, B.E., J.J. Beauchamp, R.A. Efroymson, and G.W. Suter, II. 1998b. *Development and Validation of Bioaccumulation Models for Small Mammals.* ES/ER/TM-219. Oak Ridge National Laboratory, Oak Ridge, TN.

Sloof, W. 1992. *Ecotoxicological Effect Assessment: Deriving Maximum Tolerable Concentrations (MTC) from Single Species Toxicity Data*. Guidance Document. Report No. 719102.018. National Institute of Public Health and Environmental Protection (RIVM), Hilversum, the Netherlands.

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 Ambient Water Quality Criteria for the Protection of Aquatic Organisms and Their Use*. NTIS PB85-220749. Office of Research and Development, U.S. Environmental Protection Agency, Springfield, VA.

Suter, G.W., and C.L. Tsao. 1996. *Toxicological Benchmarks for Screening Potential Contaminants of Concern for Effects on Aquatic Biota: 1996 Revision*. U.S. Department of Energy, Oak Ridge National Laboratory, Oak Ridge, TN. June.

U.S. EPA (Environmental Protection Agency). 1993. *Water Quality Guidance for the Great Lakes System and Correction.* Proposed Rule. 58 FR 20802.

U.S. EPA (Environmental Protection Agency). 1995a. *Great Lakes Water Quality Initiative Criteria Documents for the Protection of Aquatic Life in Ambient Water*. EPA-820-B-95-004. Office of Water, Washington, DC.

U.S. EPA (Environmental Protection Agency). 1995b. *Great Lakes Water Quality Initiative Criteria Documents for the Protection of Wildlife: DDT, Mercury, 2,3,78-TCDD, and PCBs*. EPA-820-B-95-008. Office of Water, Washington, DC.

U.S. EPA (Environmental Protection Agency). 1996. *Water Quality Criteria Documents for the Protection of Aquatic Life in Ambient Water: 1995 Updates.* EPA-820-B-96-001. Office of Water, Washington, DC.

U.S. EPA (Environmental Protection Agency). 1997b. *Protocol for Screening Level Ecological Risk Assessment at Hazardous Waste Combustion Facilities (Volumes 1 and 2)*. Office of Solid Waste. Internal Review Draft. February 28, 1997.

U.S. EPA (Environmental Protection Agency). 1998. Non-Groundwater Pathways, Human Health and Ecological Risk Analysis for Fossil Fuel Combustion Phase 2 (FFC2). Office of Solid Waste, Washington, DC.

Appendix I. Calculation of Health-Based Numbers (HBNs) for CCW Constituent Screening

Management of CCW can result in contaminants moving from a waste management unit (WMU) and contaminating groundwater, and surface water via groundwater transport from a CCW WMU. Under these scenarios, individuals living near WMUs may then come into contact with chemicals via ingestion of contaminated drinking water or ingestion of fish contaminated via chemical uptake and accumulation.

Key Features of HBN Calculations

- HBNs calculated for groundwater (mg/L) and surface water (mg/L)
- HBNs based on a target cancer risk of 10^{-5} and a HQ of 1
- Groundwater HBNs based on a residential drinking water scenario
- Surface water HBNs based on a recreational fisher scenario
- Adult and child receptors first exposed at ages 3, 8, 15 and 20
- Exposure factors set at central tendency values
- Source size set at the 95th percentile of CCW landfills.

Health-based numbers (HBNs) for groundwater (as drinking water) and surface water were used in this analysis to consider risks and hazards to human receptors from chemicals that are released from CCW management units and move through the subsurface. HBNs represent concentrations in environmental media that will not cause an exceedance of a target cancer risk of 10^{-5} or a hazard quotient (HQ) of 1.

The pathways included in the HBN calculations are summarized in **Table I-1**. The HBN for groundwater was based on domestic use of groundwater as drinking water. Surface water HBNs were based on a recreational fisher scenario in which the receptor was assumed to live at a different off-site location and to be exposed only to fish caught recreationally.

Table I-1. Pathways Included in HBN Calculations

HBN Calculation	Drinking Water Ingestion	Fish Ingestion
Groundwater HBN	✓	
Surface water HBN		✓

I.1 Methodology

All HBNs considered human receptors exposed to contaminated media and/or food items at different ages to take into account changing exposure patterns with age. The specific receptors considered were individuals exposed starting at ages 3, 8, 15, and 20. Depending on the start age, an appropriate exposure duration was selected for each receptor based on residency data. Each receptor was exposed for the period of time determined by the exposure duration, and the model accounted for changes in exposure patterns as a person ages. All exposure parameters selected for this analysis were based on 50th percentile values. Once the cancer risks and HQs were calculated for each receptor, HBNs were calculated based on total cancer risk, noncancer inhalation, and noncancer ingestion. The most protective HBN (i.e., the lowest across all age

groups) was selected. In all cases, the HBN calculations used central tendency exposure factors (e.g., body weight, exposure duration, exposure frequency, consumption rates).

The equations used to calculate the HBNs are provided at the end of this appendix; **Table I-2** lists the tables of equations by exposure pathway (**Tables I-4** through **I-8**). Data used in these equations to calculate the CCW HBNs can be found in the other appendices to this report, as well as in **Table I-9**, which provides the age cohort–specific human exposure factors used in the HBN calculations.

Table I-2. Key to Tables of Equations Used to Calculate HBNs

Equation for Fish Concentrations	
I-4	Concentration in Fish at Different Trophic Levels (mg/kg)
Equations for Human Exposure	
I-5	Daily Intake of Contaminant from Consumption of Fish (mg/kg BW/day)
I-6	Daily Intake of Contaminant from Consumption of Drinking Water (mg/kg BW/day)
Equations for Unit Risk Calculations and Health-based Numbers	
I-7	Cancer Risk and Hazard Quotient Due to Ingestion (unitless)
I-8	Health-Based Concentration (mg/L)

Groundwater HBNs were based on standard residential exposure assumptions for drinking water consumption, using equations from (U.S. EPA, 1998a). The surface water HBNs were based on concentrations in fish estimated using an aquatic food chain model; that methodology is described in the rest of this section.

The methodology used for estimating concentrations in fish was based on EPA's *Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions* (U.S. EPA, 1998a). An aquatic food chain model was used to estimate the concentration of constituent that may accumulate in fish. It was assumed for this analysis that fish are a food source for a recreational fisher. Trophic level three (T3) and four (T4) fish were considered in this analysis. T3 fish are those that consume invertebrates and plankton. T4 fish are those that consume other fish. Most of the fish that humans eat are T4 fish (e.g., salmon, trout, walleye, bass) and medium to large T3 fish (e.g., carp, smelt, perch, catfish, sucker, bullhead, sauger). For metals other than mercury, the calculation of contaminants in fish was based on the total concentration of contaminants in the waterbody (i.e., dissolved and suspended solids). For mercury, the calculation of contaminants in fish was based on the dissolved concentration of methyl mercury in the waterbody.

Fish tissue concentrations are dependent on a bioconcentration factor (BCF), which is used to estimate the amount of constituent being transferred from the waterbody into the fish tissue. Specifically, BCFs reflect the ratio between the tissue concentration in fish and the appropriate waterbody concentration. BCFs were developed for each constituent to reflect accumulation in each trophic level considered. They were also developed to estimate the concentration in the fish filet versus the total fish. Human receptors consume only the filet portion of the fish, which has a lower lipid content. Because some constituents tend to accumulate in the fatty tissue, the concentration in the filet portion of the fish is sometimes lower than the concentration in the whole fish.

I.2 Health-Based Numbers

Table I-3 provides the HBNs for surface water and groundwater.

Table I-3. Groundwater and Surface Water HBNs

Chemical	Benchmark Type	Groundwater HBN (mg/L)[1]	Surface Water HBN[2] (mg/L)
Aluminum	Noncancer	29.4	NA
Antimony	Noncancer	0.012	NA
Arsenic	Cancer	0.0029	0.23
Arsenic	Noncancer	0.0088	0.71
Barium	Noncancer	5.89	NA
Beryllium	Noncancer	0.059	1.0
Boron	Noncancer	5.87	NA
Cadmium	Noncancer	0.015	0.035
Chromium (III)	Noncancer	44	23,700
Chromium (VI)	Noncancer	0.088	47
Cobalt	Noncancer	0.0088	NA
Copper[3]	Noncancer (GW)/AWQ (SW)	0.29	1.3
Cyanide	Noncancer	0.59	NA
Fluoride	Noncancer	3.52	NA
Lead	MCL	0.015	NA
Manganese	Noncancer	1.4	NA
Mercury	Noncancer	0.0088	3.85E-06
Molybdenum	Noncancer	0.147	12
Nickel	Noncancer	0.59	237
Nitrate	MCL	10	NA
Nitrate	Noncancer	47	NA
Nitrite	Noncancer	2.9	NA
Selenium	Noncancer	0.147	0.038
Silver	Noncancer	0.147	NA
Strontium	Noncancer	17.6	NA
Thallium	Noncancer	0.0024	0.008
Vanadium	Noncancer	0.21	NA
Zinc	Noncancer	8.8	8.13

[1] Based on domestic drinking water ingestion.
[2] Based on fish consumption by a recreational fisher.
[3] Fish bioconcentration factor values for copper are zero. HBN based on National Ambient Water Quality Criteria.
AWQ = National Ambient Water Quality Criteria
MCL = maximum contaminant level or drinking water action level (for lead and copper)
NA = not available

Table I-4. Concentration in Fish at Different Trophic Levels (mg/kg)

Cfish_T

For Mercury:

$$Cfish_T = Cdiss \times BCF_T$$

$$Cdiss = 0.05 \times Cwt$$

For NonvolatileMetals:

$$Cfish_T = Cwt \times BCF_T$$

Name	Description	Value
Cdiss	Concentration in surface water (dissolved) (mg/L)	Calculated
Cwt	Concentration in surface water (total) (mg/L)	Set equal to 1 for HBN calculation
0.05	Fraction of total mercury as dissolved methyl mercury	Derived from U.S. EPA, 1997a
BCF_T3F	Bioconcentration factor for trophic level 3, fish filet (L/kg)	Chemical-specific (see App. J)
BCF_T3W	Bioconcentration factor for trophic level 3, fish whole (L/kg)	Chemical-specific (see App. J)
BCF_T4F	Bioconcentration factor for trophic level 4, fish filet (L/kg)	Chemical-specific (see App. J)
BCF_T4W	Bioconcentration factor for trophic level 4, fish whole (L/kg)	Chemical-specific (see App. J)

Source: U.S. EPA, 1998a.

Table I-5. Daily Intake of Contaminant from Consumption of Fish (mg/kg BW/day)

Ifish

$$Cfish_T = F_{T3} \times C_{fishT3F} + F_{T4} \times C_{fishT4F}$$

$$Ifish = Cfish_T \times CR_{fish} \times \frac{F_{fish}}{1{,}000 \times BW}$$

Name	Description	Value
1000	Conversion factor (g/kg)	
C_fishT3F	Concentration of contaminant in fish at different trophic levels (mg/kg)	Calculated (Table I-4)
C_fishT4F	Concentration of contaminant in fish at different trophic levels (mg/kg)	Calculated (Table I-4)
$Cfish_T$	Concentration of contaminant in fish (mg/kg)	Calculated (Table I-4)
BW	Body weight (kg)	Age-cohort-specific (Table I-9)
CR_fish	Consumption rate of fish (g WW/day)	Age-cohort-specific (Table I-9)
F_fish	Fraction of fish intake from contaminated source (unitless)	1 (protective value)
F_T3	Fraction of trophic level 3 intake (unitless)	0.36 (U.S. EPA, 1997d)
F_T4	Fraction of trophic level 4 intake (unitless)	0.64 (U.S. EPA, 1997d)

Source: U.S. EPA, 1998a.

Table I-6. Daily Intake of Contaminant from Consumption of Drinking Water (mg/kg BW/day)

Idw

$$Idw = C_{dw} \times Cr_{dw} \times \frac{F_{dw}}{BW}$$

Name	Description	Value
1000	Conversion factor (mL/L)	
Cdw	Concentration of contaminant in drinking water (mg/L)	Set equal to 1 for HBN calculation
BW	Body weight (kg)	Age-cohort-specific (Table I-9)
CR_dw	Consumption rate of water (L/day)	Age-cohort-specific (Table I-9)
F_dw	Fraction of drinking water ingested that is contaminated (unitless)	1 (protective value)

Source: U.S. EPA, 1998a.

Table I-7. Cancer Risk and Hazard Quotient Due to Ingestion (unitless)

Risk_Oral

$$HQ_{Oral} = \frac{I}{RfD}$$

$$Risk_{Oral} = \frac{I \times ED \times EF \times CSF_{Oral}}{AT \times 365}$$

Name	Description	Value
365	Conversion factor (days/yr)	
I	Intake rate from fish or drinking water (mg/kg/day)	Calculated (Tables I-5 and I-6)
CSF_{Oral}	Oral cancer slope factor (mg/kg/day)-1	Chemical-specific (see Appendix G)
RfD	Noncancer reference dose (mg/kg/day)	Chemical-specific (see Appendix G)
AT	Averaging time (yr)	70 (U.S. EPA, 1991)
ED	Exposure duration for oral ingestion (yr)	Age-cohort-specific (Table I-9)
EF	Exposure frequency (days/yr)	350 (U.S. EPA, 1991)

Table I-8. Health-Based Concentration (ppm)

CalcHBN

$$HBN_{NC_{Oral}} = \frac{C}{HQ_{Oral}} \times THQ$$

$$HBN_{Risk} = \frac{C}{Risk} \times TR$$

Name	Description	Value
THQ	Target noncancer hazard quotient (unitless)	1
TR	Target cancer risk (unitless)	1.00E-5
HQ_Oral	Noncancer hazard quotient for ingestion (unitless)	Calculated (Table I-7)
Risk	Total cancer risk (unitless)	Calculated (Table I-7)
C	Constituent concentration in media (mg/L or mg/kg)	Value set to unit concentration of 1

Back calculation assuming linearity.

Table I-9. Age Cohort-Specific Human Exposure Factors

Parameter	Cohort_1	Cohort_2	Cohort_3	Cohort_4
Body weight (BW) (kg)	15.3	29.6	56.8	69.3
Start year (SY) (yr)	3	8	15	20
Fish consumption rate (CR_fish) (g WW/day)	2	2	2	2
Exposure duration (ED) (yr)	5	7.5	8	10
Drinking water consumption rate (CR_dw) (L/day)	0.6165	0.731	0.8685	1.275

I.3 References

Bidleman, T.F. 1988. Atmospheric Processes. *Environmental Science and Technology* 22(4).

EPRI (Electric Power Research Institute). 1997. *Coal Combustion and Low Volume Wastes Comanagement Survey.* EPRI. Palo Alto, CA. June.

U.S. EPA (Environmental Protection Agency). 1996. *Soil Screening Guidance Technical Background Document*. EPA/540/R/95/128. Office of Solid Waste and Emergency Response, Washington, DC. May.

U.S. EPA (Environmental Protection Agency). 1997a. *Exposure Factors Handbook, Volume I, General Factors.* EPA/600/P-95/002Fa. Office of Research and Development, Washington, DC. August.

U.S. EPA (Environmental Protection Agency). 1997b. *Exposure Factors Handbook, Volume II, Food Ingestion Factors.* EPA/600/P-95/002Fa. Office of Research and Development, Washington, DC. August.

U.S. EPA (Environmental Protection Agency). 1997c. *Exposure Factors Handbook, Volume III, Activity Factors.* EPA/600/P-95/002Fa. Office of Research and Development, Washington, DC. August.

U.S. EPA (Environmental Protection Agency). 1997d. *The Parameter Guidance Document*. A Companion Document to the Methodology for Assessing Health Risks Associated with Multiple Pathways Exposure to Combustor Emissions (Internal Draft). NCEA-0238. National Center for Environmental Assessment, Cincinnati, OH, pp. 1-281. March.

U.S. EPA (Environmental Protection Agency). 1998a. *Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions.* EPA 600/R-98/137. National Center for Environmental Assessment, Cincinnati, OH. December.

U.S. EPA (Environmental Protection Agency). 1998b. *Technical Background Document for the Supplemental Report to Congress on Remaining Fossil-Fuel Combustion Wastes—Ground-Water Pathway Human Health Risk Assessment. Revised Draft Final.* Office of Solid Waste, Washington, DC. June.

U.S. EPA (Environmental Protection Agency). 2001. *Supplemental Guidance for Developing Soil Screening Levels for Superfund Sites: Peer Review Draft*. OSWER 9355.4-24. Office of Solid Waste and Emergency Response. March.

U.S. EPA (Environmental Protection Agency). 2005. *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities.* Final. EPA530-R-05-006. Office of Solid Waste. September.

[This page intentionally left blank.]

Appendix J. Chemical-Specific Inputs Used in CCW Constituent Screening

Chemical-specific inputs used to develop the CCW HBNs include the bioconcentration factors needed to estimate exposure concentrations in fish. Values for these inputs are obtained from the best available literature source. **Table J-1** provides, for each chemical in the CCW screening analysis, the values used in the analysis along with the source of each value.

Table J-1. Fish Bioconcentration Factors

Parameter[a]	Value (L/kg)	Reference	Comment
Aluminum (7429905)			
No Data			
Antimony (7440360)			
BCF_T3F	0	Barrows et al. (1980)	BCF_T3W was used as a surrogate.
BCF_T3W	0	Barrows et al. (1980)	Species was sunfish.
BCF_T4F	0	Barrows et al. (1980)	BCF_T3W was used as a surrogate.
BCF_T4W	0	Barrows et al. (1980)	BCF_T3W was used as a surrogate.
Arsenic (7440382)			
BCF_T3F	4	Barrows et al. (1980)	BCF_T3W was used as a surrogate.
BCF_T3W	4	Barrows et al. (1980)	Species was sunfish.
BCF_T4F	4	Barrows et al. (1980)	BCF_T3W was used as a surrogate.
BCF_T4W	4	Barrows et al. (1980)	BCF_T3W was used as a surrogate.
Barium (7440393)			
No Data			
Beryllium (7440417)			
BCF_T3F	19	Barrows et al. (1980)	BCF_T3W was used as a surrogate.
BCF_T3W	19	Barrows et al. (1980)	Species was sunfish.
BCF_T4F	19	Barrows et al. (1980)	BCF_T3W was used as a surrogate.
BCF_T4W	19	Barrows et al. (1980)	BCF_T3W was used as a surrogate.
Boron (7440428)			
No Data			
Cadmium (7440439)			
BCF_T3F	270	Kumada et al. (1972)	BCF_T3W was used as a surrogate.
BCF_T3W	270	Kumada et al. (1972)	Geomean of 3 data points in Table 2.
BCF_T4F	270	Kumada et al. (1972)	BCF_T4W was used as a surrogate.
BCF_T4W	270	Kumada et al. (1972)	Geomean of 3 data points in Table 2. Species were doce and rainbow trout.

(continued)

Fish Bioconcentration Factors (continued)

Parameter[a]	Value (L/kg)	Reference	Comment
Chromium(III) (16065831)			
BCF_T3F	0.6	Stephan (1993)	BCF_T4F was used as a surrogate.
BCF_T3W	0.6	Stephan (1993)	BCF_T4F was used as a surrogate.
BCF_T4F	0.6	Stephan (1993)	Geomean (as cited in Stephan, 1993) based on Buhler et al. (1977) and Calamari et al. (1982). Used chromium as a surrogate.
BCF_T4W	0.6	Stephan (1993)	BCF_T4F was used as a surrogate.
Chromium(VI) (18540299)			
BCF_T3F	0.6	Stephan (1993)	BCF_T4F was used as a surrogate.
BCF_T3W	0.6	Stephan (1993)	BCF_T4F was used as a surrogate.
BCF_T4F	0.6	Stephan (1993)	Geomean (as cited in Stephan, 1993) based on Buhler et al. (1977) and Calamari et al. (1982). Used chromium as a surrogate.
BCF_T4W	0.6	Stephan (1993)	BCF_T4F was used as a surrogate.
Copper (7440508)			
BCF_T3F	0	Stephan (1993)	
BCF_T3W	0	Stephan (1993)	
BCF_T4F	0	Stephan (1993)	
BCF_T4W	0	Stephan (1993)	
Cobalt (7440484)			
No Data			
Cyanide (57125)			
No Data			
Fluoride (16984488)			
No Data			
Manganese (7439965)			
No Data			
Molybdenum (7439987)			
BCF_T3F	4	Eisler (1989)	BCF_T4F was used as a surrogate.
BCF_T3W	4	Eisler (1989)	BCF_T4F was used as a surrogate.
BCF_T4F	4	Eisler (1989)	Geomean of values found on pages 27 and 28. Species were rainbow trout and steelhead trout.
BCF_T4W	4	Eisler (1989)	BCF_T4F was used as a surrogate.
Nickel (7440020)			
BCF_T3F	0.8	Stephan (1993)	BCF_T4F was used as a surrogate.
BCF_T3W	0.8	Stephan (1993)	BCF_T4F was used as a surrogate.
BCF_T4F	0.8	Stephan (1993)	Derived from Calamari et al. (1982) (as cited in Stephan, 1993).
BCF_T4W	0.8	Stephan (1993)	BCF_T4F was used as a surrogate.

(continued)

Fish Bioconcentration Factors (continued)

Parameter[a]	Value (L/kg)	Reference	Comment
Selenium (7782492)			
BCF_T3F	490	Lemly (1985)	Based on threadfin shad and blueback herring. Units corrected.
BCF_T3W	490	Lemly (1985)	BCF_T3F was used as a surrogate.
BCF_T4F	1,700	Lemly (1985)	Based on threadfin shad and blueback herring. Units corrected.
BCF_T4W	1,700	Lemly (1985)	BCF_T4F was used as a surrogate.
Silver (7440224)			
BCF_T3F	0	Barrows et al. (1980)	BCF_T3W was used as a surrogate.
BCF_T3W	0	Barrows et al. (1980)	Species was sunfish.
BCF_T4F	0	Barrows et al. (1980)	BCF_T3W was used as a surrogate.
BCF_T4W	0	Barrows et al. (1980)	BCF_T3W was used as a surrogate.
Strontium (7440246)			
No Data			
Thallium (7440280)			
BCF_T3F	34	Barrows et al. (1980)	BCF_T3W was used as a surrogate.
BCF_T3W	34	Barrows et al. (1980)	Species was sunfish.
BCF_T4F	130	Stephan (1993)	Derived from Zitko et al. (1975) (as cited in Stephan, 1993).
BCF_T4W	130	Stephan (1993)	BCF_T4F was used as a surrogate.
Total Nitrate Nitrogen (14797558)			
No Data			
Vanadium (7440622)			
No Data			
Zinc (7440666)			
BCF_T3F	350	Murphy et al. (1978)	BCF_T3W was used as a surrogate.
BCF_T3W	350	Murphy et al. (1978)	Geomean of converted dry weight concentration in Table 1 of bluegills at Site A and B.
BCF_T4F	350	Murphy et al. (1978)	BCF_T3W was used as a surrogate.
BCF_T4W	350	Murphy et al. (1978)	BCF_T3W was used as a surrogate.

[a] BCF_T3F = Bioconcentration factor for trophic level 3 fish, filet
BCF_T3W= Bioconcentration factor for trophic level 3 fish, whole
BCF_T4F = Bioconcentration factor for trophic level 4 fish, filet
BCF_T4W= Bioconcentration factor for trophic level 4 fish, whole

References

Barrows, M.E., S.R. Petrocelli, K.J. Macek, and J.J. Carroll. 1980. Chapter 24: Bioconcentration and elimination of selected water pollutants by bluegill sunfish (*Lepomis macrochirus*). In: *Dynamics, Exposure and Hazard Assessment of Toxic Chemicals,* R. Haque (ed.), Ann Arbor Science Publishers Inc., Ann Arbor, MI. pp. 379-392.

Eisler, R. 1989. *Molybdenum Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review*. Contaminant Hazard Reviews, Report No. 19, Biological Report 85(1.19). Laurel, MD. August.

Kumada, H., S. Kimura, M. Yokote, and Y. Matida. 1972. Acute and chronic toxicity, uptake, and retention of cadmium in freshwater organisms. *Bulletin of Freshwater Fisheries Research Laboratory 22*(2):157-165. December 20.

Lemly, A.D. 1985. Toxicology of selenium in a freshwater reservoir: Implications for environmental hazard evaluation and safety. *Ecotoxicology and Environmental Safety 10*:314-338.

Murphy, B.R., G.J. Atchison, and A.W. McIntosh. 1978. Cadmium and zinc in muscle of bluegill (*Lepomis macrochirus*) and largemouth bass (*Micropterus salmoides*) from an industrially contaminated lake. *Environmental Pollution 17*:253-257.

Stephan, C.E. 1993. *Derivation of Proposed Human Health and Wildlife Bioaccumulation Factors for the Great Lakes Initiative* (Draft). PB93-154672. Environmental Research Laboratory, Office of Research and Development, Duluth, MN. March.

U.S. EPA (Environmental Protection Agency). 1997. *Mercury Study Report to Congress. Volume III - Fate and Transport of Mercury in the Environment*. EPA 452/R-97/005. Office of Air Quality Planning and Standards and Office of Research and Development, Washington, DC.

Appendix K. Screening Analysis Results

Table K-1. CCW Surface Impoundment (SI) Human Health Screening Results: Groundwater-to-Drinking-Water Pathway

Chemical	Benchmark Type	HBN (mg/L)	2002 SI Porewater	
			90th Percentile	HQ(Cancer Risk)
Analytes Exceeding Risk Criteria[1]				
Antimony	Noncancer	1.17E-02	6.40E-02	5.45E+00
Arsenic	Cancer	2.86E-03	5.18E+00	(1.81E-02)
Arsenic	Noncancer	8.81E-03	5.18E+00	5.88E+02
Boron	Noncancer	5.87E+00	7.52E+01	1.28E+01
Cadmium	Noncancer	1.47E-02	1.31E-01	8.91E+00
Chromium (VI)	Noncancer	8.81E-02	3.66E-01	4.15E+00
Cobalt	Noncancer	8.81E-03	6.27E+00	7.13E+02
Fluoride	Noncancer	3.52E+00	1.91E+01	5.42E+00
Lead	MCL	1.50E-02	1.77E-01	1.18E+01
Manganese	Noncancer	1.38E+00	7.67E+00	5.56E+00
Molybdenum	Noncancer	1.47E-01	1.00E+00	6.81E+00
Nickel	Noncancer	5.87E-01	7.49E-01	1.27E+00
Nitrate	MCL	1.00E+01	6.02E+02	6.02E+01
Nitrite	Noncancer	2.94E+00	5.22E+00	1.78E+00
Selenium	Noncancer	1.47E-01	3.56E-01	2.43E+00
Thallium	Noncancer	2.35E-03	4.52E-02	1.93E+01
Vanadium	Noncancer	2.06E-01	4.78E-01	2.33E+00
Analytes Below Risk Criteria[1]				
Aluminum	Noncancer	2.94E+01	2.30E+01	7.84E-01
Barium	Noncancer	5.89E+00	3.02E-01	5.15E-02
Beryllium	Noncancer	5.87E-02	5.68E-03	9.67E-02
Chromium (III)	Noncancer	4.40E+01	3.66E-01	8.31E-03
Copper	Noncancer	2.93E-01	2.84E-01	9.69E-01
Mercury	Noncancer	8.81E-03	2.50E-04	2.84E-02
Silver	Noncancer	1.47E-01	5.00E-03	3.41E-02
Strontium	Noncancer	1.76E+01	8.74E+00	4.96E-01
Zinc	Noncancer	8.81E+00	6.70E-01	7.60E-02

[1] Risk criteria are 1E-05 cancer risk or a hazard quotient (HQ) of 1E+00 for noncancer endpoints.
HBN = health-based number
90th percentile = 90th percentile SI porewater concentration
HQ = hazard quotient
MCL = maximum contaminant level

Table K-2. CCW Surface Impoundment (SI) Human Health Screening Results: Groundwater-to-Surface-Water (Fish Ingestion) Pathway

Chemical	Benchmark Type	HBN (mg/L)	2002 SI Porewater	
			90th Percentile	HQ (Cancer Risk)
Analytes Exceeding Risk Criteria[1]				
Arsenic	Cancer	0.23	5.18E+00	(2.24E-04)
Arsenic	Noncancer	0.71	5.18E+00	7.28E+00
Cadmium	Noncancer	0.035	1.31E-01	3.73E+00
Mercury	Noncancer	3.85E-06	2.50E-04	6.50E+01
Selenium	Noncancer	0.038	3.56E-01	9.50E+00
Thallium	Noncancer	0.008	4.52E-02	5.69E+00
Analytes Below Risk Criteria[1]				
Antimony	AWQ	4.3	6.40E-02	1.49E-02
Beryllium	Noncancer	1.00	5.68E-03	5.69E-03
Chromium (III)	Noncancer	23,700	3.66E-01	1.54E-05
Chromium (VI)	Noncancer	47	3.66E-01	7.72E-03
Copper[2]	AWQ	1.3	2.84E-01	2.18E-01
Molybdenum	Noncancer	12	1.00E+01	8.43E-02
Nickel	Noncancer	237	7.49E-01	3.16E-03
Zinc	Noncancer	8.13	6.70E-01	8.24E-02

[1] Risk criteria are 1E-05 cancer risk or a hazard quotient (HQ) of 1E+00 for noncancer endpoints.
[2] Fish bioconcentration factor values for copper are zero. HBN based on National Ambient Water Quality Criteria.
HBN = health-based number
90th percentile = 90th percentile SI porewater concentration
HQ = hazard quotient
AWQ = National Ambient Water Quality Criteria
MCL = maximum contaminant level

Table K-3. CCW Landfill Leachate Human Health Screening Results: Groundwater-to-Drinking-Water Pathway

Chemical	Benchmark Type	HBN (mg/L)	2002 Landfill Leachate	
			90th Percentile	HQ(Cancer Risk)
Analytes Exceeding Risk Criteria[1]				
Antimony	Noncancer	1.17E-02	2.61E-01	2.22E+01
Arsenic	Cancer	2.86E-03	3.94E-01	(1.38E-03)
Arsenic	Noncancer	8.81E-03	3.94E-01	4.48E+01
Boron	Noncancer	5.87E+00	1.06E+01	1.80E+00
Cadmium	Noncancer	1.47E-02	4.94E-02	3.37E+00
Chromium (VI)	Noncancer	8.81E-02	2.00E-01	2.27E+00
Cobalt	Noncancer	8.81E-03	8.25E-02	9.33E+00
Fluoride	Noncancer	3.52E+00	6.34E+00	1.80E+00
Lead	MCL	1.50E-02	2.39E-01	1.59E+01
Molybdenum	Noncancer	1.47E-01	6.16E-01	4.20E+00
Nitrite	Noncancer	2.94E+00	3.47E+00	1.18E+00
Selenium	Noncancer	1.47E-01	1.76E-01	1.20E+00
Thallium	Noncancer	2.35E-03	5.00E-02	2.13E+01
Vanadium	Noncancer	2.06E-01	4.50E-01	2.19E+00
Analytes Below Risk Criteria[1]				
Aluminum	Noncancer	2.94E+01	1.05E+01	3.58E-01
Barium	Noncancer	5.89E+00	1.60E+00	2.73E-01
Beryllium	Noncancer	5.87E-02	1.58E-02	2.70E-01
Chromium (III)	Noncancer	4.40E+01	2.00E-01	4.54E-03
Copper	Noncancer	2.93E-01	1.50E-01	5.12E-01
Cyanide	Noncancer	5.87E-01	6.32E-02	1.08E-01
Manganese	Noncancer	1.38E+00	1.37E+00	9.92E-01
Mercury	Noncancer	8.81E-03	2.69E-03	3.06E-01
Nickel	Noncancer	5.87E-01	3.09E-01	5.27E-01
Nitrate	MCL	1.00E+01	2.83E+00	2.83E-01
Silver	Noncancer	1.47E-01	3.95E-02	2.69E-01
Strontium	Noncancer	1.76E+01	9.70E+00	5.51E-01
Zinc	Noncancer	8.81E+00	1.94E+00	2.20E-01

[1] Risk criteria are 1E-05 cancer risk or a hazard quotient (HQ) of 1E+00 for noncancer endpoints, applied to 90th percentile concentrations.

HBN = health-based number
90th percentile = 90th percentile concentration
HQ = hazard quotient
MCL = maximum contaminant level

Table K-4. CCW Landfill Leachate Human Health Screening Results: Groundwater-to-Surface-Water Pathway

Chemical	Benchmark Type	HBN (mg/L)	2002 Landfill Leachate	
			90th Percentile	HQ (Cancer Risk)
Analytes Exceeding Risk Criteria[1]				
Arsenic	Cancer	0.23	3.94E-01	(1.71E-05)
Cadmium	Noncancer	0.035	4.94E-02	1.41E+00
Mercury	Noncancer	3.85E-06	2.69E-03	7.00E+02
Selenium	Noncancer	0.038	1.76E-01	4.69E+00
Thallium	Noncancer	0.008	5.00E-02	6.29E+00
Analytes Below Risk Criteria[1]				
Antimony	AWQ	4.3	2.61E-01	6.07E-02
Arsenic	Noncancer	0.71	3.94E-01	5.54E-01
Beryllium	Noncancer	1.00	1.58E-02	1.59E-02
Chromium (III)	Noncancer	23,700	2.00E-01	8.44E-06
Chromium (VI)	Noncancer	47	2.00E-01	4.22E-03
Copper[2]	AWQ	1.3	1.50E-01	1.15E-01
Cyanide	AWQ	222	6.32E-02	2.85E-04
Molybdenum	Noncancer	12	6.16E-01	5.20E-02
Nickel	Noncancer	237	3.09E-01	1.30E-03
Zinc	Noncancer	8.13	1.94E+00	2.38E-01

[1] Risk criteria are 1E-05 cancer risk or a hazard quotient (HQ) of 1E+00 for noncancer endpoints, applied to 90th percentile concentrations.

[2] Fish bioconcentration factor values for copper are zero. HBN based on National Ambient Water Quality Criteria.

HBN = health-based number90th percentile = 90th percentile concentration

HQ = hazard quotient

AWQ = National Ambient Water Quality Criteria

MCL = maximum contaminant level

Table K-5. Surface Impoundment Ecological Screening Results: Direct Surface Impoundment and Groundwater-to-Surface-Water Pathways

Chemical	CSCL (mg/L)	2002 SI Porewater 90th Percentile (mg/L)	2002 SI Porewater HQ	1998 SI Water 95th Percentile (mg/L)	1998 SI Water HQ
Analytes Exceeding Risk Criterion[1]					
Aluminum	8.70E-02	2.30E+01	2.65E+02	5.11E+00	5.87E+01
Arsenic III	1.50E-01	5.18E+00	3.45E+01	5.50E-01	3.67E+00
Arsenic IV	8.10E-03	5.18E+00	6.39E+02	5.50E-01	6.79E+01
Barium	4.00E-03	3.02E-01	7.54E+01	7.12E-01	1.78E+02
Boron	1.60E-03	7.52E+01	4.70E+04	4.60E+02	2.88E+05
Cadmium	2.50E-03	1.31E-01	5.23E+01	2.50E-01	1.00E+02
Chromium VI	1.10E-02	3.66E-01	3.33E+01	2.67E-02	2.43E+00
Cobalt	2.30E-02	6.27E+00	2.73E+02	1.00E-02	4.35E-01
Copper	9.30E-03	2.84E-01	3.05E+01	3.90E-01	4.19E+01
Lead	3.01E-04	1.77E-01	5.88E+02	2.50E-01	8.31E+02
Mercury	1.90E-07	2.50E-04	1.32E+03	1.50E-03	7.89E+03
Nickel	5.20E-02	7.49E-01	1.44E+01	6.00E-01	1.15E+01
Selenium IV	2.80E-02	3.56E-01	1.27E+01	7.80E+00	2.79E+02
Selenium total	5.00E-03	3.56E-01	7.13E+01	7.80E+00	1.56E+03
Selenium VI	9.50E-03	3.56E-01	3.75E+01	7.80E+00	8.21E+02
Silver	3.60E-04	5.00E-03	1.39E+01	5.00E-03	1.39E+01
Vanadium	2.00E-02	4.78E-01	2.39E+01	8.00E-01	4.00E+01
Analytes Not Exceeding Risk Criterion[1]					
Antimony	3.00E-02	6.40E-02	2.13E+00	1.37E-01	4.57E+00
Beryllium	6.60E-04	5.68E-03	8.61E+00	1.00E-03	1.52E+00
Chromium III	8.60E-02	3.66E-01	4.26E+00	4.00E-01	4.65E+00
Molybdenum	3.70E-01	1.00E+00	2.70E+00	5.00E-01	1.35E+00
Thallium	1.20E-02	4.52E-02	3.77E+00	5.00E-02	4.17E+00
Zinc	1.20E-01	6.70E-01	5.58E+00	6.70E-01	5.58E+00

[1] Risk criterion is a hazard quotient (HQ) of 10 (for direct exposure to impoundment waters).
SI = surface impoundment
CSCL = chemical stressor concentration level

Table K-6. Landfill Ecological Screening Results: Groundwater-to-Surface-Water Pathway

Chemical	CSCL (mg/L)	2002 - Landfill Leachate	
		90th Percentile (mg/L)	HQ
Analytes Exceeding Risk Criterion[1]			
Aluminum	8.70E-02	1.05E+01	1.21E+02
Arsenic IV	8.10E-03	3.94E-01	4.87E+01
Barium	4.00E-03	1.60E+00	4.01E+02
Beryllium	6.60E-04	1.58E-02	2.40E+01
Boron	1.60E-03	1.06E+01	6.61E+03
Cadmium	2.50E-03	4.94E-02	1.98E+01
Chromium VI	1.10E-02	2.00E-01	1.82E+01
Copper	9.30E-03	1.50E-01	1.61E+01
Lead	3.01E-04	2.39E-01	7.94E+02
Mercury	1.90E-07	2.69E-03	1.42E+04
Selenium total	5.00E-03	1.76E-01	3.52E+01
Selenium VI	9.50E-03	1.76E-01	1.85E+01
Silver	3.60E-04	3.95E-02	1.10E+02
Vanadium	2.00E-02	4.50E-01	2.25E+01
Zinc	1.20E-01	1.94E+00	1.61E+01
Analytes Not Exceeding Risk Criterion[1]			
Antimony	3.00E-02	2.61E-01	8.70E+00
Arsenic III	1.50E-01	3.94E-01	2.63E+00
Chromium III	8.60E-02	2.00E-01	2.33E+00
Cobalt	2.30E-02	8.25E-02	3.59E+00
Molybdenum	3.70E-01	6.16E-01	1.67E+00
Nickel	5.20E-02	3.09E-01	5.95E+00
Selenium IV	2.80E-02	1.76E-01	6.28E+00
Thallium	1.20E-02	5.00E-02	4.17E+00

[1] Risk criterion is a hazard quotient (HQ) of 10
CSCL = chemical stressor concentration level

Appendix L. Time to Peak Concentration at Receptor Well for Selected CCW Constituents

L.1 Introduction

This appendix presents plots of arrival times for the peak well concentrations used to calculate groundwater-to-drinking-water risks for selected CCW constituents (arsenic III and V, boron, cobalt, selenium IV and VI, and thallium[1]). The arrival times are plotted as cumulative distributions for surface impoundments and landfills**.** These constituents were selected to represent the chemicals with the highest estimated risks and to span the range of mobility in the subsurface.

Groundwater pathway modeling conducted in support of the CCW risk assessment consisted of probabilistic fate and transport simulations of mostly metal constituents present in three different waste types (ash, ash and coal, and fluidized bed combustion wastes) managed in landfills and surface impoundments. Three liner designs were also considered: no liner; a 3-foot clay liner; and a composite liner (a composite of geomembrane, geosynthetic clays, and/or compacted clays), assigned to each CCW waste management unit (WMU) based on liner type data in the EPRI database (see **Appendix B**). The predicted groundwater concentrations were used to estimate potential risks to humans and the environment exposed to the modeled CCW constituents.

Among the inputs to the model were distributions of infiltration rates of water through the landfills and surface impoundments corresponding to each of the three liner types. Among the outputs generated by the groundwater pathway fate and transport model were the peak concentration observed at the receptor well and the time at which the peak was observed. For each probabilistic simulation scenario (a constituent in a particular waste type managed in a particular type of WMU), approximately 10,000 sets of model inputs generated an equivalent number of groundwater observations. Some were non-zero concentrations, others were zero. For these zero-value observations, the model also assigned a value of zero to arrival time. Zero-value observations can be attributed to zero-value infiltration rates (which occur only for WMUs with composite liners); in that case, no mass leaves the WMU and there is no time of travel. Zero-value observations can also be attributed to fate and transport conditions that retard the movement of a constituent from the WMU through the subsurface to the extent that the dissolved component was not observed within the established maximum allowable timeframe (10,000 years). In this case, the time of travel is greater than 10,000 years.

To better understand the time frames in which risks associated with exposures to contaminated groundwater may occur, an analysis was performed to graphically represent distributions of arrival time of the peak groundwater concentrations at the nearby drinking water well. The analysis was performed across all waste types with respect to liner and WMU type.

[1] Thallium was not modeled in the surface impoundment scenario, and thus no arrival times were calculated here.

What follows is a description of how the peak concentrations and their arrival times were treated to create the plots presented in this appendix, including the treatment of zero-value observations.

L.2 Methodology

Given a constituent managed in a particular type of WMU (e.g., arsenic in landfills), all infiltration rates and their corresponding peak concentrations and arrival times predicted by the model were extracted from the input/output data for simulations across all waste types in which the selected constituent was found. The triplets of data needed to prepare the graphs—infiltration rate, peak concentration, and arrival time—were then filtered from the data and segregated by liner type. Zero-value observed concentrations were treated in the following manner:

- Zero-value observations corresponding to zero-value infiltration rates were assigned an arbitrary value of -1, effectively excluding those data from the graphs. This was appropriate, because when infiltration is zero, there is no plume and no contaminants enter or are transported in groundwater. Only the composite liner scenarios produced zero-value infiltration rates.

- Zero-value observations corresponding to non-zero infiltration rates were assigned an arbitrary arrival time greater than (>) 10,000 years, the maximum simulation timeframe. These data points are also not shown in the plots, as only times up to 10,000 years were visible.

Table L-1 shows the distribution of zero-value concentration observations by WMU and composite liner scenario. The total observations in this table include data points with a modeled arrival time of >10,000 years and those with zero infiltration rates. Note that for surface impoundments, there are fewer model runs (observations) for thallium because thallium results are not available for ash and coal waste streams because of very limited data in the CCW constituent database.

Table L-1. Distribution of Zero-Value Concentrations

WMU Type	Total Observations[a] (Number)	Zero Infiltration Rates (Number)	Zero Infiltration Rates[b] (%)	Composite Liner Observations (Number)	Composite Liner Zero Infiltration Rates (%)
Landfill	29,717	3,538	11.9%	4,847	73.0%
Surface Impoundment (As, B, Se)[c]	19,825	500	2.5%	1,406	35.6%
Surface Impoundment (Tl)[d]	9,905	389	3.9%	1,130	34.4%

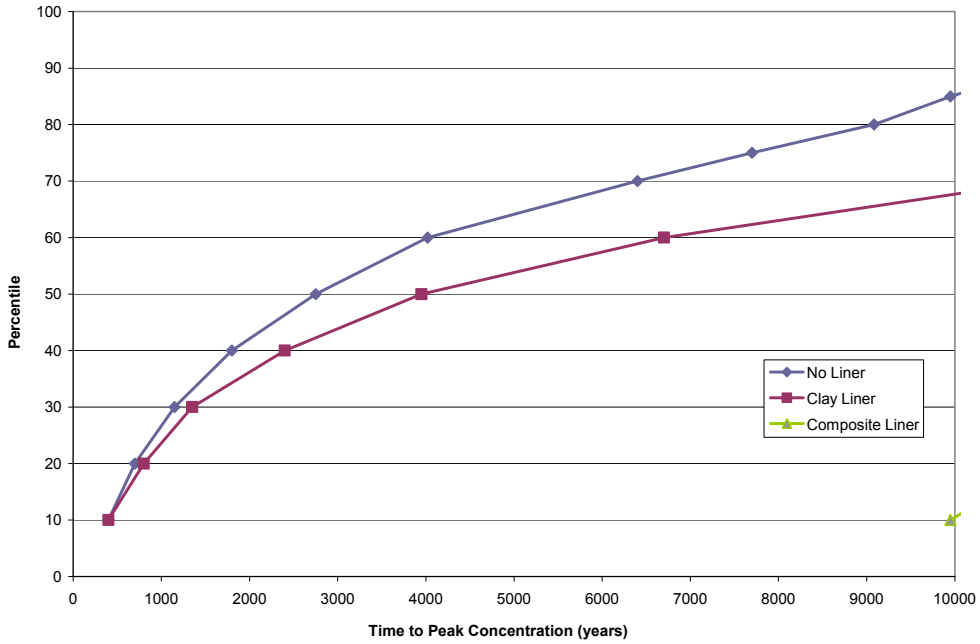
[a] Per constituent across all waste types and liners.
[b] Out of all observations.
[c] Observations for arsenic, boron, cobalt, and selenium; all were modeled in both ash and ash and coal waste streams managed in surface impoundments.
[d] Observations for thallium only, which was detected only in ash waste streams managed in surface impoundments.

After zero infiltration rate observations were filtered from each data set, percentiles of arrival time of the peak observed concentration were plotted on the y-axis by liner type and WMU (**Figures L-1** through **L-21**). The x-axis range for landfills is 0 to 10,000 years. For

surface impoundments, plots are provided on both the full 0 to 10,000-year time frame and a shorter time frame, so that the shape of the cumulative distribution can be seen for the lower time-of-travel range characteristic of these facilities.

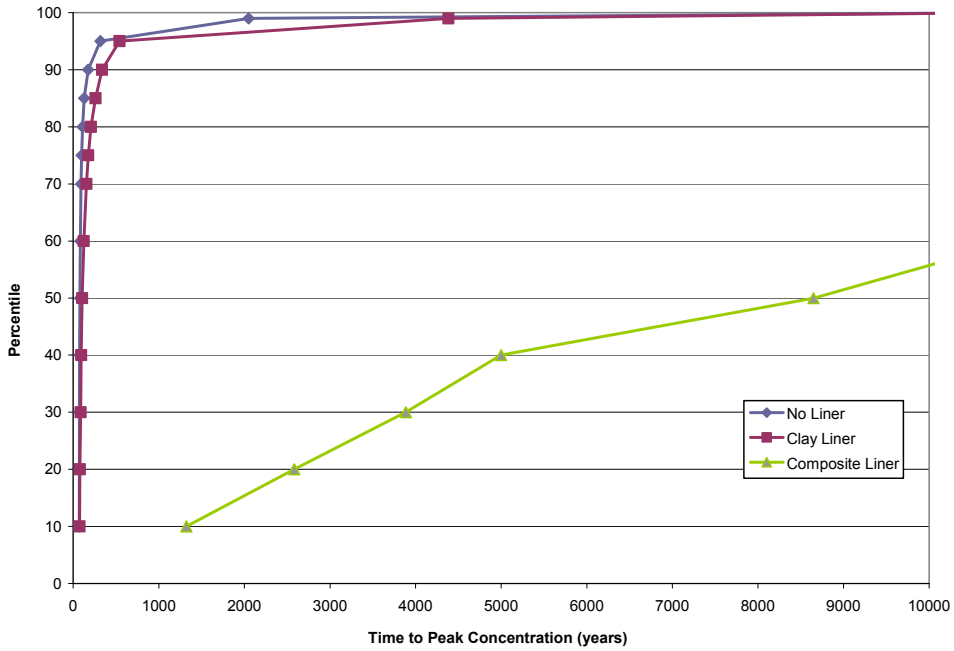
The figures are organized alphabetically by metal, and there are three figures for each metal: landfills, surface impoundments (0–10,000 years), and surface impoundments (shorter time frame).

The shorter arrival times for clay-lined landfills compared to unlined landfills are an artifact of the fact that liners were modeled at each landfill as reported in the EPRI survey, and each landfill location has a different subsurface geology. The shorter arrival times mainly reflect more transmissive soils and aquifer materials at the clay-lined facility locations.



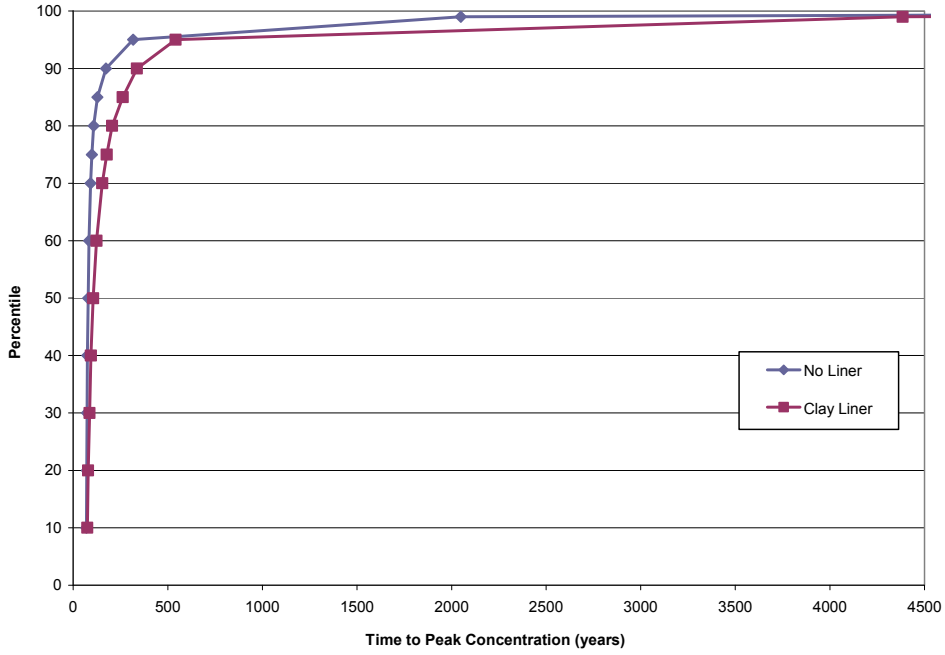
73% of composite liner results have no infiltration and are not plotted on this graph.

Figure L-1. Time to peak distribution for arsenic III: landfills, all waste types.



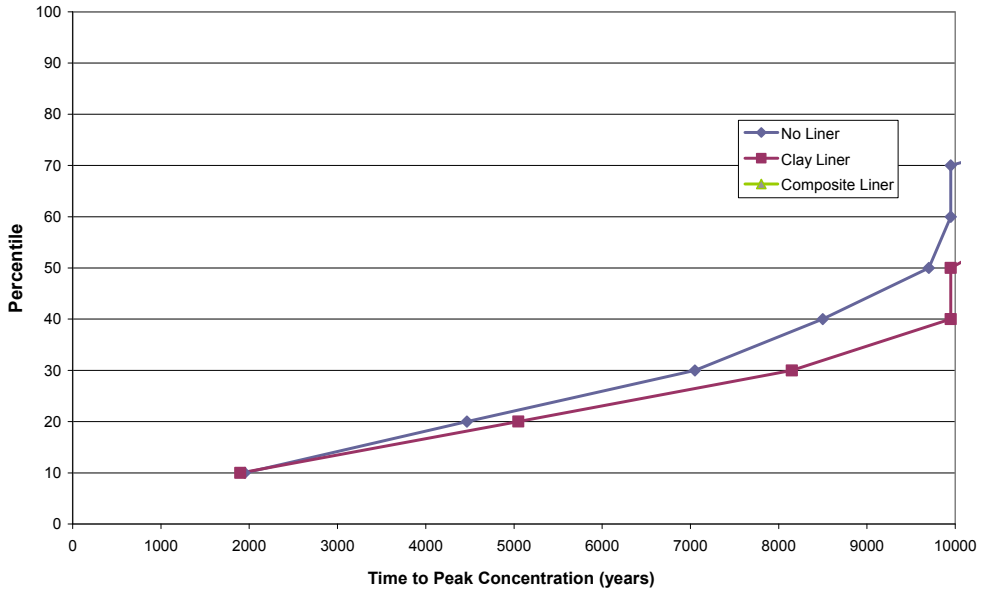
35.6% of composite liner results have no infiltration and are not plotted on this graph.

Figure L-2. Time to peak distribution for arsenic III: surface impoundments, all waste types, full 10,000 year time frame.



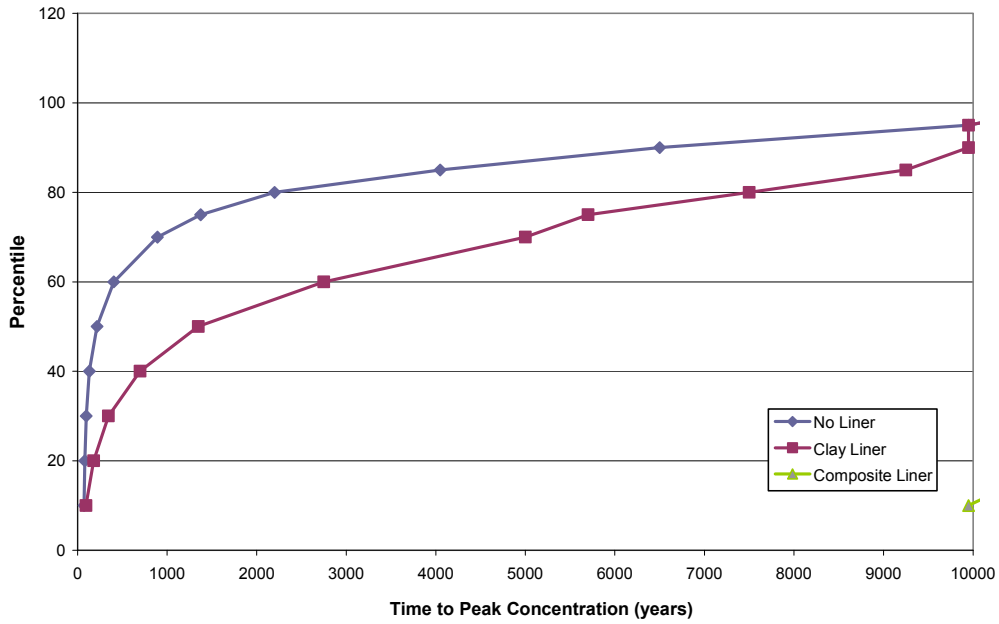
35.6% of composite liner results have no infiltration and are not plotted on this graph.

Figure L-3. Time to peak distribution for arsenic III: surface impoundments, all waste types, shorter time frame.



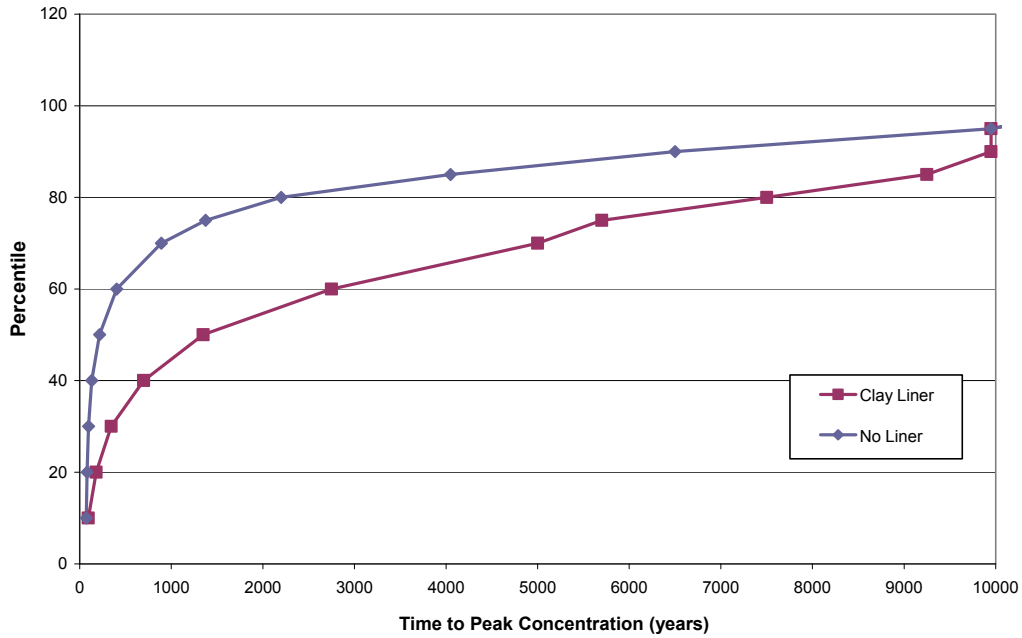
73% of composite liner results have no infiltration and are not plotted on this graph.

Figure L-4. Time to peak distribution for arsenic V: landfills, all waste types.



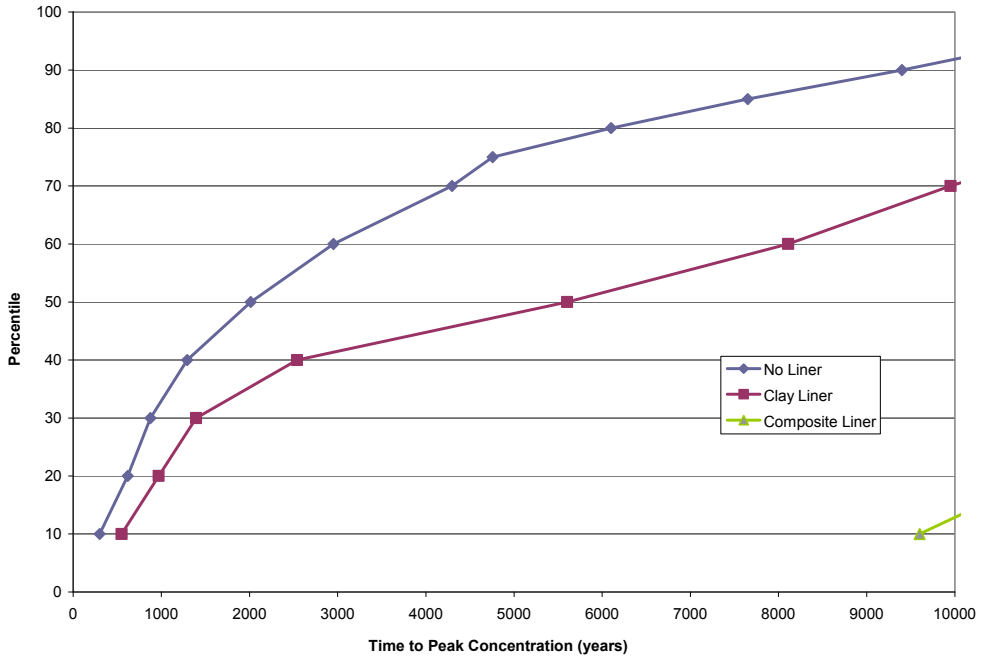
35.6% of composite liner results have no infiltration and are not plotted on this graph.

Figure L-5. Time to peak distribution for arsenic V: surface impoundments, all waste types, full 10,000 year time frame.



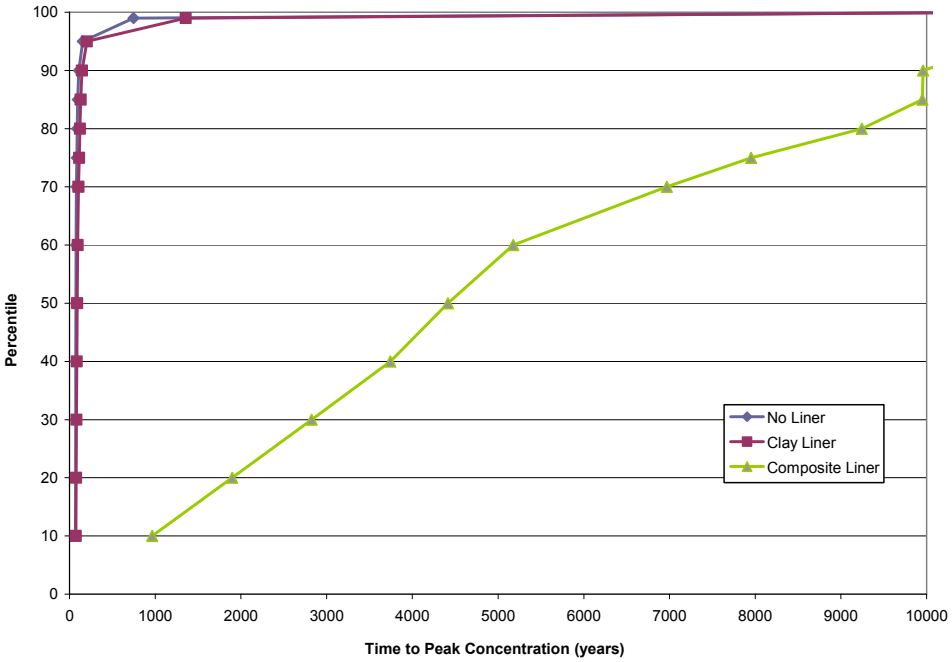
35.6% of composite liner results have no infiltration and are not plotted on this graph.

Figure L-6. Time to peak distribution for arsenic V: surface impoundments, all waste types, shorter time frame.



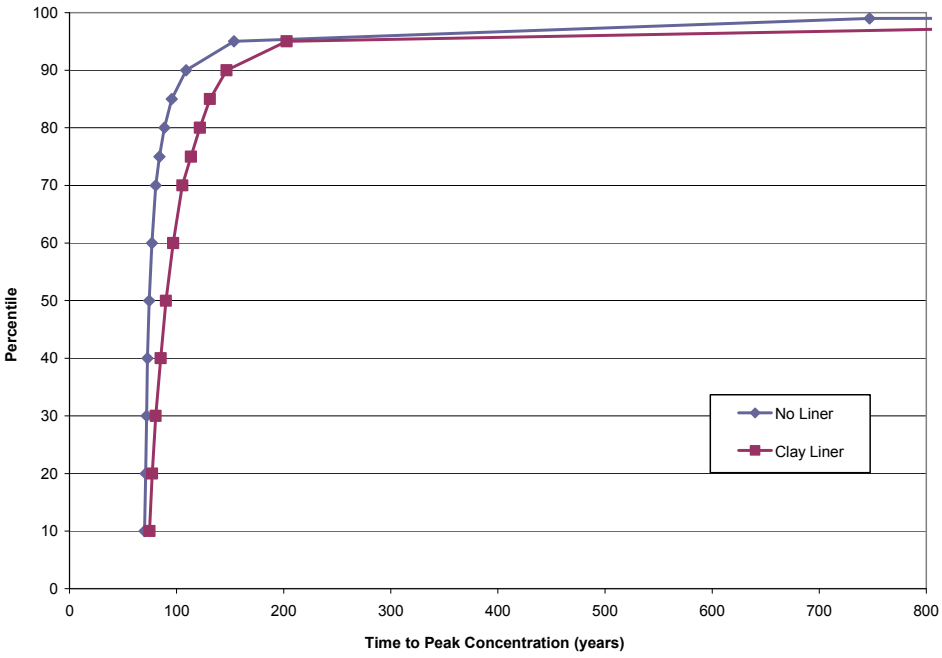
73% of composite liner results have no infiltration and are not plotted on this graph.

Figure L-7. Time to peak distribution for boron: landfills, all waste types.



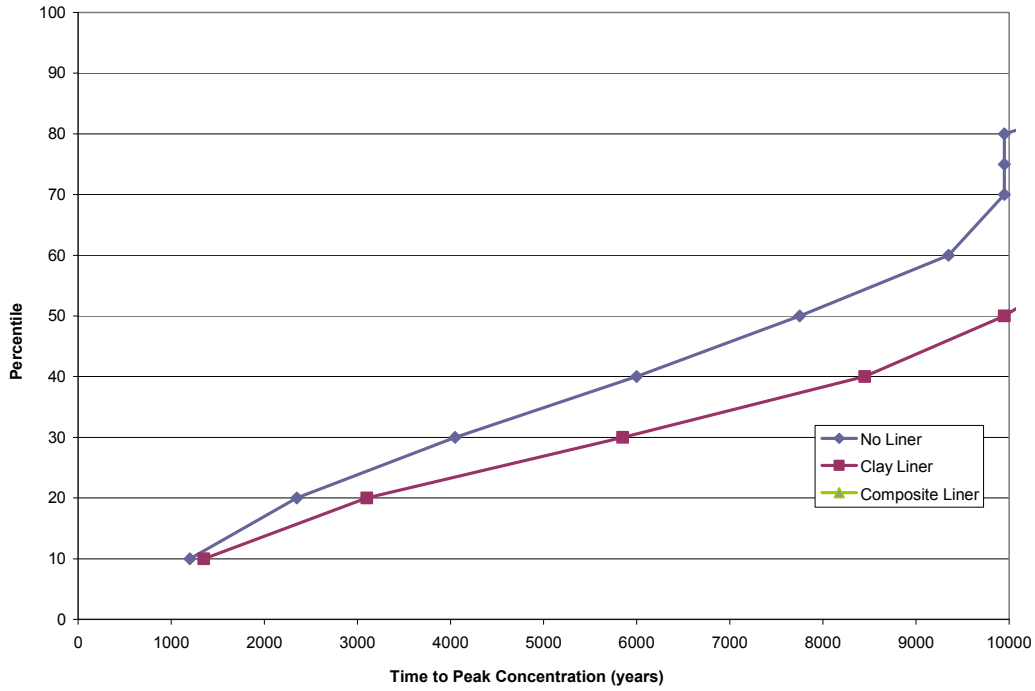
35.6% of composite liner results have no infiltration and are not plotted on this graph.

Figure L-8. Time to peak distribution for boron: surface impoundments, all waste types, full 10,000 year time frame.



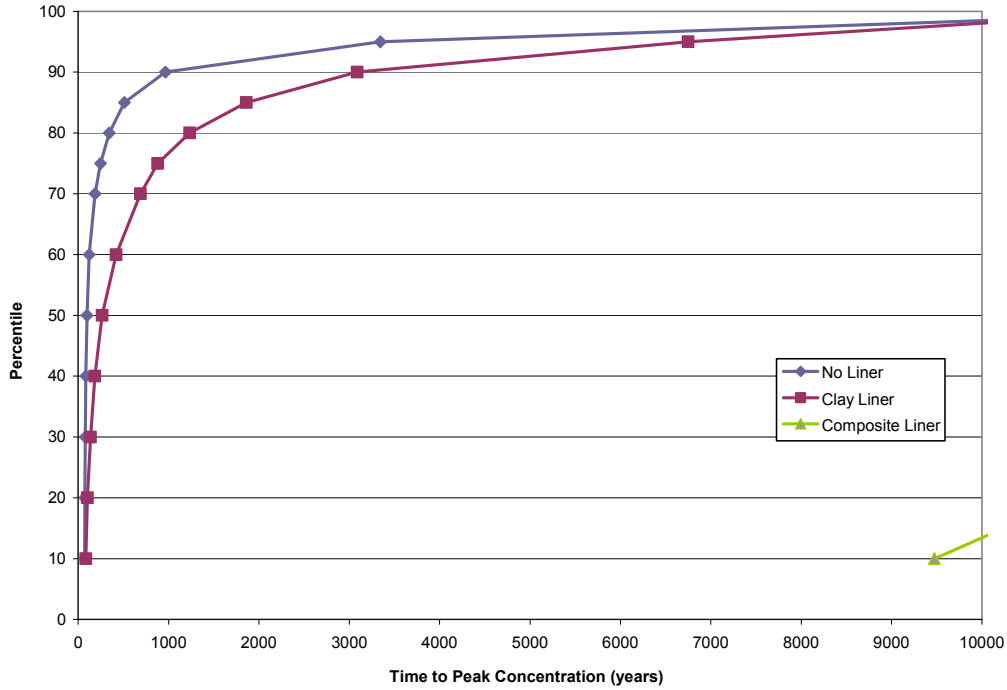
35.6% of composite liner results have no infiltration and are not plotted on this graph.

Figure L-9. Time to peak distribution for boron: surface impoundments, all waste types, shorter time frame.



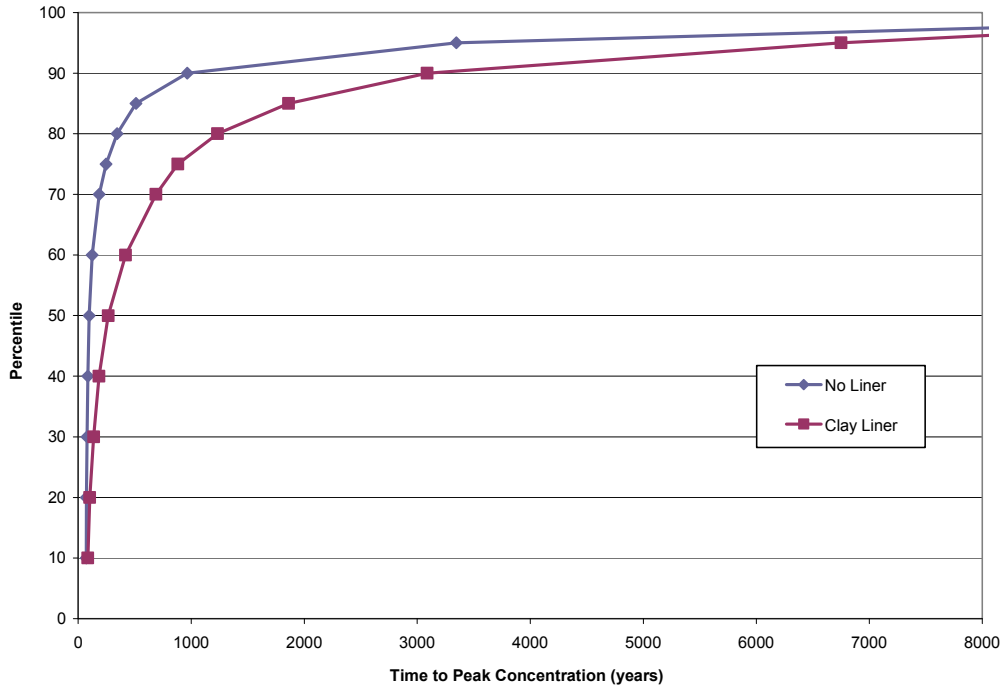
73% of composite liner results have no infiltration and are not plotted on this graph.

Figure L-10. Time to peak distribution for cobalt: landfills, all waste types.



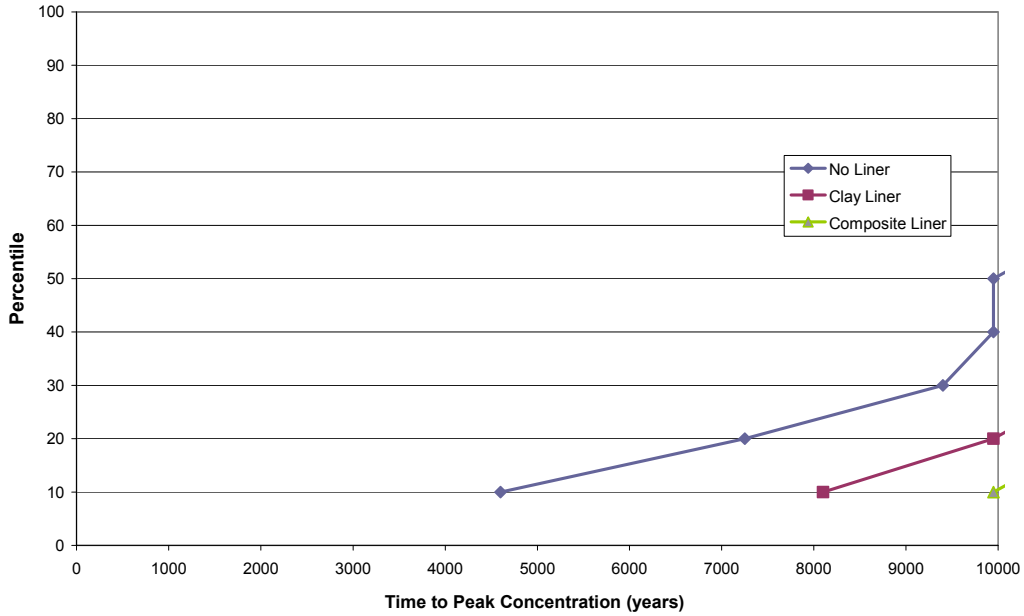
35.6% of composite liner results have no infiltration and are not plotted on this graph.

Figure L-11. Time to peak distribution for cobalt: surface impoundments, all waste types, full 10,000 year time frame.



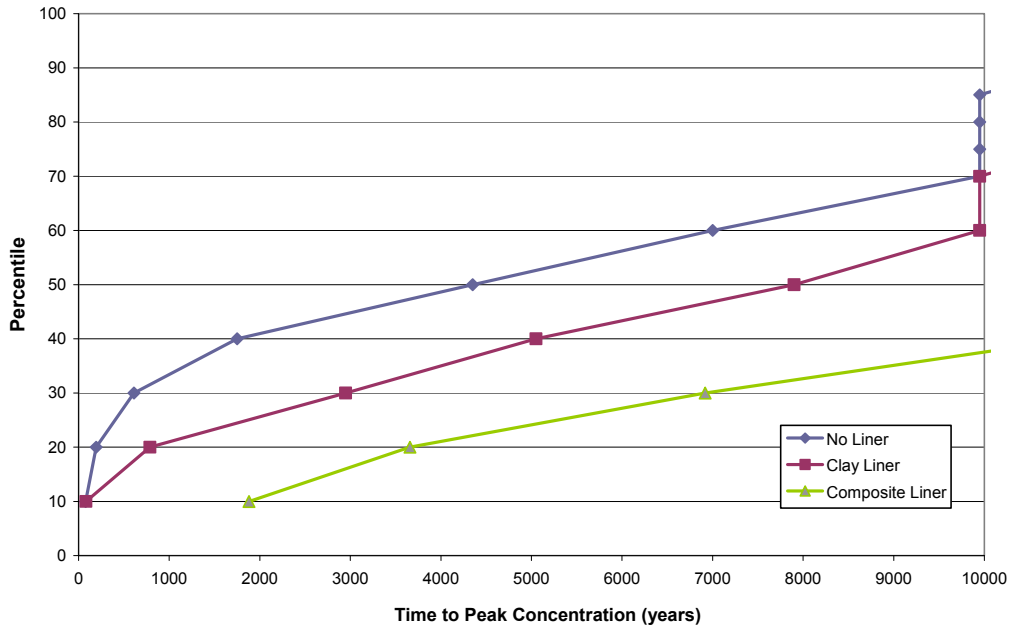
35.6% of composite liner results have no infiltration and are not plotted on this graph.

Figure L-12. Time to peak distribution for cobalt: surface impoundments, all waste types, shorter time frame.



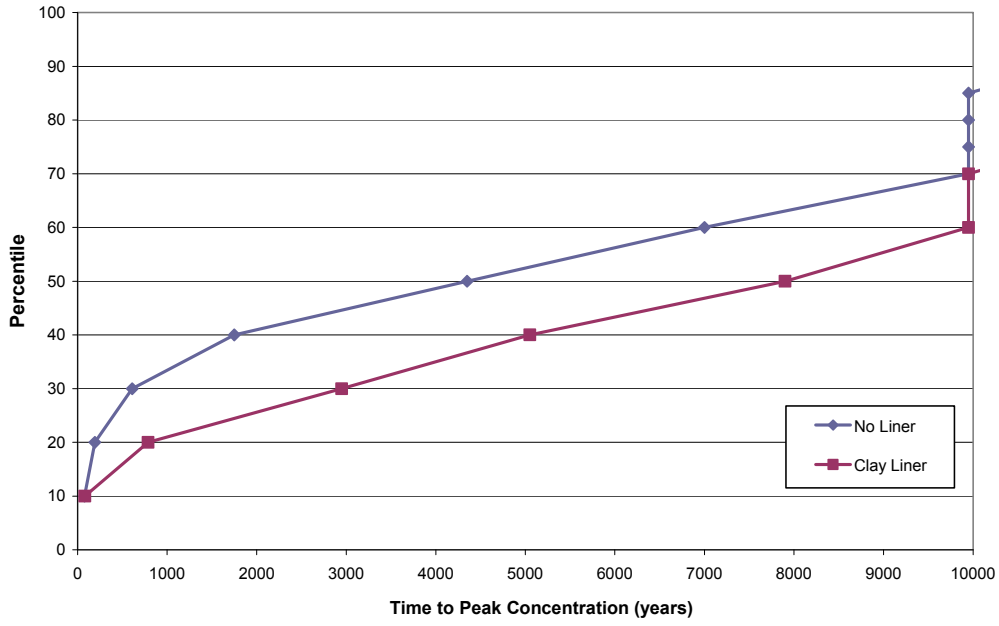
73% of composite liner results have no infiltration and are not plotted on this graph.

Figure L-13. Time to peak distribution for selenium IV: landfills, all waste types.



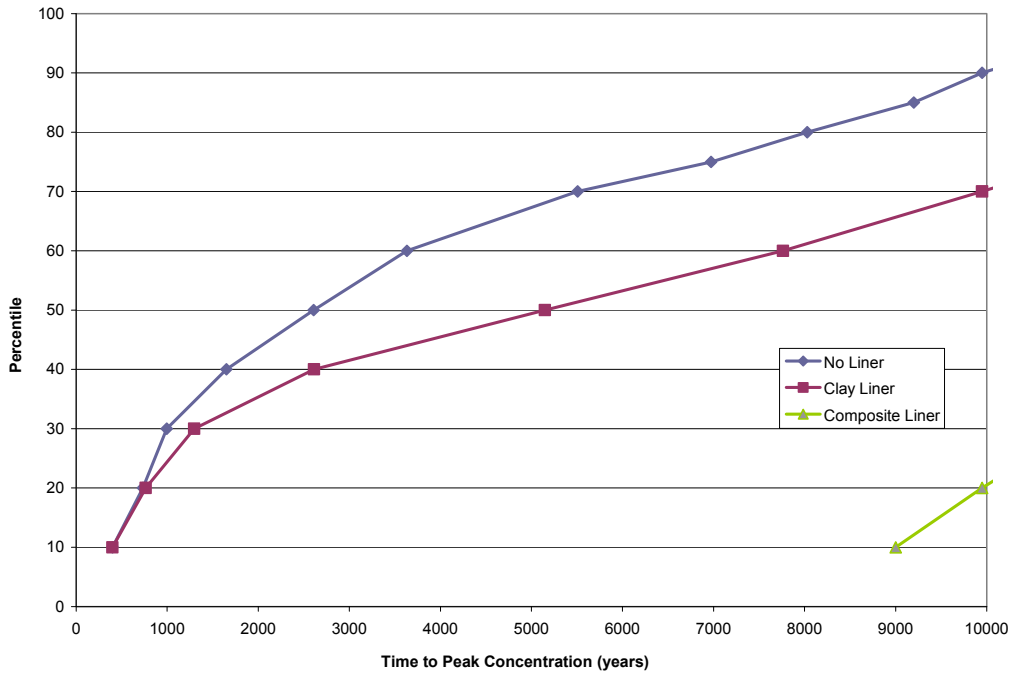
35.6% of composite liner results have no infiltration and are not plotted on this graph.

Figure L-14. Time to peak distribution for selenium IV: surface impoundments, all waste types, full 10,000 year time frame.



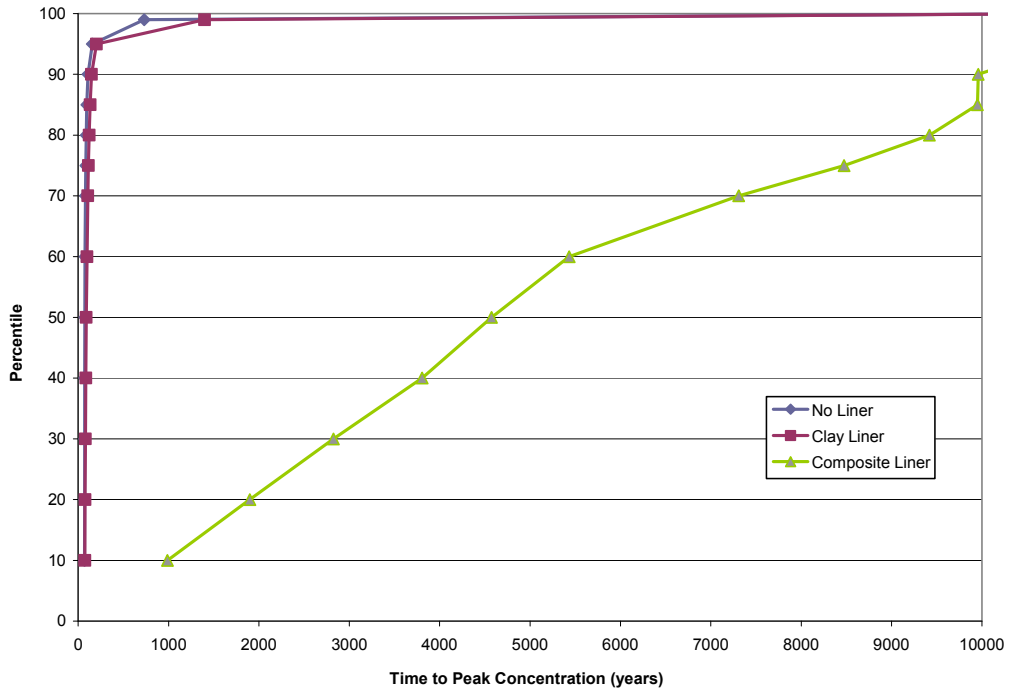
35.6% of composite liner results have no infiltration and are not plotted on this graph.

Figure L-15. Time to peak distribution for selenium IV: surface impoundments, all waste types, shorter time frame.



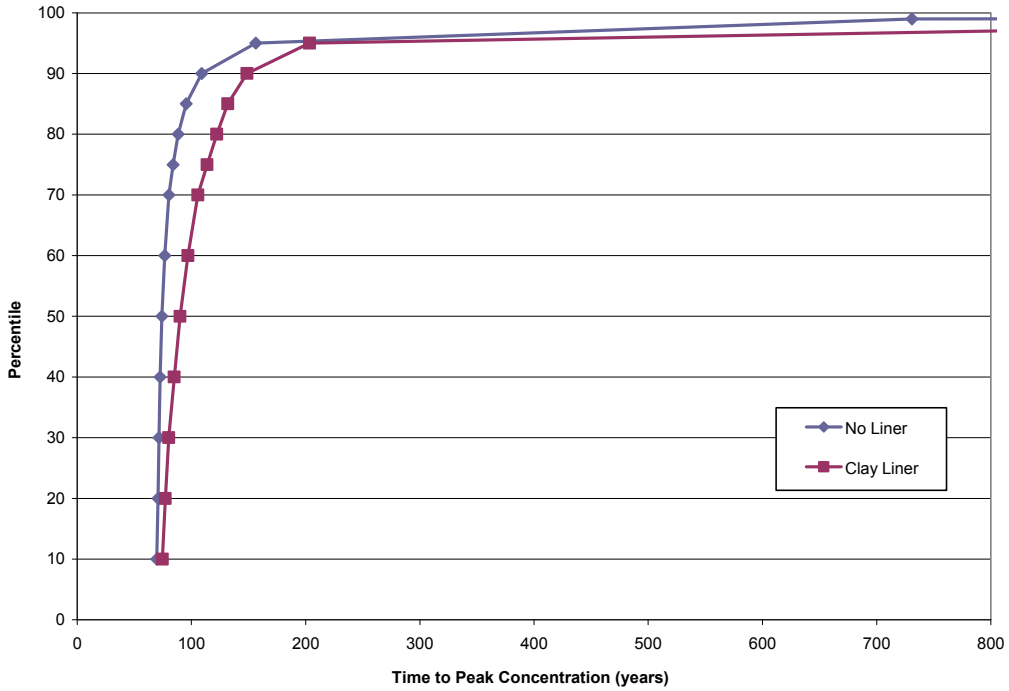
73% of composite liner results have no infiltration and are not plotted on this graph.

Figure L-16. Time to peak distribution for selenium VI: landfills, all waste types.



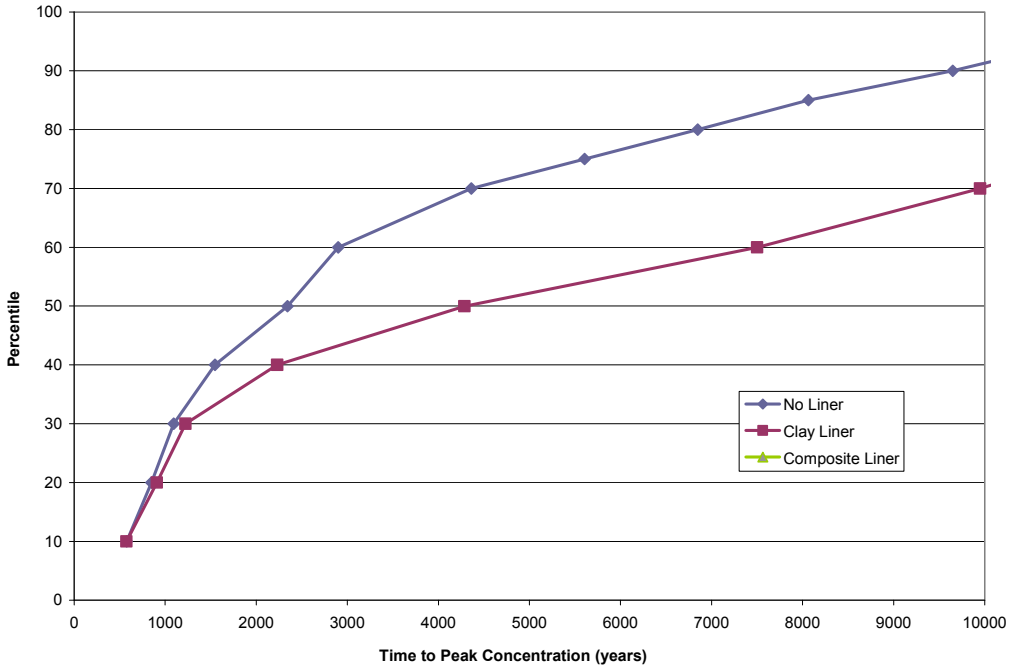
35.6% of composite liner results have no infiltration and are not plotted on this graph.

Figure L-17. Time to peak distribution for selenium VI: surface impoundments, all waste types, full 10,000 year time frame.



35.6% of composite liner results have no infiltration and are not plotted on this graph.

Figure L-18. Time to peak distribution for selenium VI: surface impoundments, all waste types, shorter time frame.



73% of composite liner results have no infiltration and are not plotted on this graph.

Figure L-19. Time to peak distribution for thallium: landfills, all waste types.