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Characterization of Field Leachates at Coal Combustion Product Management Sites

Arsenic, Selenium, Chromium, and Mercury Speciation

Technical Report

Characterization of Field Leachates at Coal Combustion Product Management Sites

Arsenic, Selenium, Chromium, and Mercury Speciation

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Final Report, November 2006

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REPORT SUMMARY

A large amount of laboratory-generated leachate data has been produced over the last two decades to estimatecoal combustion product (CCP) leachate concentrations, and a variety of leaching methods have been used. No one method, however, has been shown to accurately represent field leaching conditions. In fact, little work has been performed to systematically evaluate field-generated leachates representative of a range of coal types, combustion systems, and management methods, and only limited work has been conducted to determine the species of key constituents in CCP field leachates. For this project, field leachate samples were collected from a wide variety of CCP management sites distributed throughout the United States in order to provide a broad characterization of major and trace constituents in the leachate. Speciation of arsenic, selenium, chromium, and mercury in the leachates was also determined. This report presents an evaluation of analytical results as a function of CCP type, management method, and source coal.

Background

The leachability CCPs can vary widely based on factors such as coal type and combustion/collection processes. CCP leachates commonly have neutral to alkaline pH, and as a result, the mobility of heavy metal cations such as lead and cadmium is limited. However, other constituents typically occur as oxyanions, which are more mobile than metal cations under alkaline pH conditions. Arsenic, selenium, and chromium are of particular interest due to the multiple species that may be present in CCP leachate, and because the speciation of these elements affects both mobility and toxicity. Mercury is also of interest due to the expected increase in future concentrations as well as the toxicity of organic species at low concentrations. EPRI and the U.S. Department of Energy (DOE) cosponsored this project to characterize field leachates at CCP management sites.

Objectives

To broadly characterize CCP leachate samples, collected in the field from a wide variety of CCP management settings, including speciation of arsenic, selenium, chromium, and, in some cases, mercury.

Approach

Eighty-one field leachate samples were collected from 29 CCP management facilities. Samples were collected from leachate wells, leachate collection systems, drive-point piezometers, lysimeters, the ash/water interface at impoundments, impoundment outfalls and inlets, and seeps. All samples—collected using uniform sampling procedures and analyzed by a single laboratory for over 30 constituents—were intended to represent CCP leachate in actual management settings. Arsenic, chromium, and selenium speciation samples were collected at all sites, and mercury speciation samples were collected at 15 sites. Mercury samples were collected using

ultraclean methods. Total and monomethylmercury were preserved using HCl, while dimethylmercury was purged from the collected water samples with an argon stream in the field, and collected on Carbotrap™ adsorbent tubes. Laboratory analytical methods were selected to provide detection limits of less than one part per billion for most trace elements, and less than 1 part per trillion for mercury and its species.

Results

Results showed that

- Sulfate was the dominant anion in coal ash leachate samples, the only constituent in the leachate with a median concentration greater than 100 mg/L. Major cations in bituminous coal ash leachate were calcium and magnesium, while ash leachate derived from subbituminous/lignite coal was dominated by sodium.
- Silicon and boron had the highest median concentrations (greater than 1000 µg/L) in ash among the minor and trace constituents. Median concentrations of strontium, molybdenum, lithium, aluminum, and barium were greater than 100 µg/L. Conversely, median concentrations of chromium, beryllium, thallium, silver, lead, and mercury were lower than 1 µg/L; silver, beryllium, and lead were rarely detected.
- Most constituents (22 out of the 34 analyzed) had higher concentrations in ash landfill leachate samples than in ash impoundment leachate samples. Concentrations of most major constituents were higher in flue gas desulfurization (FGD) leachate than in ash leachate.
- Arsenic concentrations in ash leachate ranged from 1.4 to 1380 µg/L, with a median of 25 µg/L. The dominant arsenic species was As(V). As(III) was only dominant in four samples from impoundments where bituminous coal ash was managed.
- Selenium concentration in ash leachate ranged from 0.07 to 1760 µg/L, with a median of 19 µg/L. Se(IV) was the dominant species in ash ponds and for bituminous coal ash, while Se(VI) was predominant in landfill settings and for subbituminous/lignite coal ash.
- Mercury concentrations were very low, with a median concentration of 3.8 ng/L and maximum of 61 ng/L in coal ash leachate, and a median concentration of 8.3 ng/L and maximum of 79 ng/L in FGD leachate. The concentration of organic mercury species was almost always less than 1 ng/L.

EPRI Perspective

There has been a long running debate regarding the validity of the many lab leaching tests used in CCP studies. This research provides a broad leachate database that can be used to bracket expected leachate concentrations in actual field settings, and to evaluate differences among CCP types and management methods. In related research, this database will be used for improving leachate prediction models. Knowledge of leaching behavior is critical in accurately evaluating the long-term risks associated with CCP management sites.

Keywords

Coal Combustion Products; Leachate; Arsenic; Chromium; Mercury; Selenium

ABSTRACT

Field leachate samples were collected from 29 coal combustion product (CCP) management sites from several geographic locations in the United States to provide a broad characterization of major and trace constituents in the leachate. In addition, speciation of arsenic, selenium, chromium, and mercury in the leachates was determined. A total of 81 samples were collected representing a variety of CCP types, management approaches, and source coals. Samples were collected from leachate wells, leachate collection systems, drive-point piezometers, lysimeters, the ash/water interface at impoundments, impoundment outfalls and inlets, and seeps.

Results suggest distinct differences in the chemical composition of leachate from coal ash and flue gas desulfurization (FGD) sludge, landfills and impoundments, and from bituminous and subbituminous/lignite coals. Concentrations of many constituents were higher in landfill leachate than in impoundment leachate. Furthermore, aluminum, carbonates, chloride, chromium, copper, mercury, sodium, and sulfate concentrations were higher in leachates for ash from subbituminous/lignite coal; while antimony, calcium, cobalt, lithium, magnesium, manganese, nickel, thallium, and zinc concentrations were higher in leachate from bituminous coal ash.

FGD leachate had a different chemical signature than ash leachate. Concentrations of most major constituents in FGD leachate were higher than in ash leachate; this is particularly true for chloride and potassium. In addition, median concentrations of boron, strontium, and lithium were higher in FGD leachate than in ash leachate, while concentrations of selenium, vanadium, uranium, and thallium were lower.

Analysis of speciation samples indicated that ash leachate is usually dominated by As(V) and Cr(VI). Selenium was mostly in the form of Se(IV), although there were a significant number of samples dominated by Se(VI). Se(IV) dominated in impoundment settings when the source coal was bituminous or a mixture of bituminous and subbituminous, while Se(VI) was predominant in landfill settings and when the source coal was subbituminous/lignite. Mercury concentrations were very low in all samples, with a median of 3.8 ng/L in ash leachate and 8.3 ng/L in FGD leachate. The organic species of mercury always had low concentration, usually less than 5 percent of the total mercury concentration.

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INTRODUCTION

Background

Coal combustion products (CCPs)—fly ash, bottom ash, boiler slag, and flue gas desulfurization (FGD) solids—are derived primarily from incombustible mineral matter in coal and sorbents used to capture gaseous components from the flue gas, and as such contain a wide range of inorganic constituents. Concentrations of these constituents in CCPs and their leachability can vary widely by coal type and combustion/collection processes. Since CCP leachates commonly have neutral to alkaline pH, mobility of heavy metal cations such as lead and cadmium is limited. Other constituents, such as arsenic and selenium, typically occur as oxyanions, which are more mobile than metal cations under alkaline pH conditions. Knowledge of factors controlling the leachability and mobility in groundwater of the different constituents is critical to development of appropriate CCP management practices, including treatment of ash ponds and groundwater management at dry disposal sites and large scale land application uses.

There has been a large amount of laboratory-generated leachate data produced over the last two decades to estimate CCP leachate concentrations. A wide variety of leaching methodologies have been used, and it is difficult to compare results across test methods. There has been little work done to systematically evaluate field-generated leachates representative of a range of coal types, combustion systems, and management methods.

Arsenic, selenium, chromium, and mercury are of particular interest due to the multiple species that may be present in CCP leachate. The speciation affects both mobility and toxicity. Previous research has indicated that arsenic and selenium concentrations in laboratory-generated ash leachates generally range from less than 1 µg/L to about 800 µg/L (EPRI, 2003a). Arsenic concentrations higher than 1,000 µg/L in ash porewater have been associated with pyrite oxidation in areas where coal mill rejects are concentrated (EPRI, 2003b). Only limited work has been performed to determine the species of arsenic and selenium present in field leachates. The species of arsenic and selenium present in the leachate will have a significant effect on their release from the ash and mobility in groundwater (EPRI, 1994; EPRI, 2000a; EPRI, 2004).

Speciation of chromium and mercury are also important considerations with respect to mobility and toxicity. Hexavalent chromium (Cr(VI)) is more mobile and more toxic then trivalent chromium (Cr(III)), which has relatively low solubility. Mercury may be present in CCP leachates in very low concentrations, on the order of parts per trillion; there are few measurements of mercury species present in field leachates using ultra clean sampling methods.

Objectives

The objective of this research was to characterize CCP leachate samples collected in the field from a wide variety of CCP management settings. Characterization included speciation of arsenic, selenium, chromium, and, in some cases, mercury. This research provides field-scale data that can be compared to laboratory-generated data, and that can be used to model and predict the effects of CCP management methods on leachate quality and the long-term fate of inorganic constituents at CCP management sites.

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METHODS

Site Selection

Preliminary information on power plant configurations, emission controls, and CCP management methods was assembled for 274 power plants operated by 32 utilities. A subset of management sites was selected from this list, based on individual site considerations as well as development of a range of site types representative of the industry.

A distribution of sites was selected to encompass:

- a broad geographic distribution;
- a range of CCP types (fly ash, bottom ash, flue gas desulfurization solids);
- a representative distribution of CCP management methods (landfills and impoundments, active and inactive);
- coal types from various coal source regions;
- varying plant characteristics
  - boiler types;
  - particulate controls;
  - NOx controls;
  - SO_2 controls;
  - units with and without flue gas conditioning.

Individual sites were evaluated based on:

- availability of leachate sampling points;
- whether or not the site was believed to have leachate in sufficient quantities for sampling (i.e., wet CCP).
- utility interest in participation;

Based on these criteria, 33 CCP sites in 15 states were selected for sampling.

Sample Collection

Leachate samples were collected from several access points, including leachate wells, lysimeters, leachate collection systems, sluice lines, direct push drive-points, core samples, and ponds. The goal was to obtain undiluted samples representative of CCP leachate. Samples were collected by a variety of methods, depending on sample type and accessibility. In all cases, the samples were filtered in-line and collected directly into bottles containing appropriate preservatives. Sample collection is described below, and a comparison of analytical results for samples collected from different sample points is provided in Appendix B.

Direct Push Samples

Shallow porewater samples were collected from within the CCP using two direct-push methods: drive-point piezometers and t-handle probes. The drive-point sampler consisted of a ¾-inch stainless steel drive-point piezometer driven into the CCP to the desired sampling depth using a slide hammer (Figure 2-1). A ½-inch plastic tube was attached to the drive-point and threaded through ¾-inch steel riser pipe. The sample was extracted by sliding chemically-inert ¼-inch FEP tubing through the ½ -inch tubing down the riser pipe and into the screened portion of the stainless steel drive-point. The FEP tubing was then attached to a peristaltic pump via a short length of clean flexible silicone pump tubing.

Figure 2-1
Direct Push Sample Collection Using a Drive Point Piezometer

The t-handle probe is composed of a single, thin-diameter stainless steel tube that has small manufactured slots cut into the tip for sample collection (Figure 2-2). A short plastic netting was placed over the tip of the probe just prior to installation to reduce intake of fine-grained sediments. Each t-handle probe was hand-driven into the CCP to a depth of as much as six feet. The top of the t-handle was then connected to a plastic syringe to initiate water flow. Once water flow was established, a short piece of silicone tubing was used to connect ¼-inch FEP tubing to the top of the probe. The ¼-inch FEP tubing was then connected to a peristaltic pump via a short length of clean flexible silicone pump tubing.

Figure 2-2
Direct-Push Sample Collection Using a T-Handled Probe

Leachate Wells, Lysimeters, and Leachate Collection Systems

Leachate wells, lysimeters, and leachate collection systems collect deep porewater within or immediately beneath the CCP. The leachate wells sampled for this study were installed by the utilities for the purpose of monitoring leachate quality. These wells, which consist of small-diameter (2- to 4-inch) polyvinylchloride (PVC) or stainless steel pipe with slotted screens at the bottom, are installed vertically in the CCP. Lysimeters[1] were also installed to monitor leachate quality, and differ from leachate wells in that they collect porewater beneath the CCP. Lysimeters are large collection devices, usually lined with plastic and filled with sand or gravel. Leachate percolates through the CCP and into the lysimeter, where it is removed from the sand or gravel through piping that extends to land surface. Leachate collection systems are installed to drain leachate from a CCP management unit, thus preventing head build-up on the liner. These systems typically consist of large-diameter (at least 4 inch) slotted plastic pipe embedded in a sand or gravel layer above the liner. Samples may be collected at clean-out ports where the pipes emerge from beneath the fill deposit, or at the tanks where the collected leachate is stored prior to processing.

[1] In a typical installation, lysimeters are installed beneath liners to monitor liner performance. However, the lysimeters monitored for this study were installed immediately beneath the CCP.

Whenever possible, low-flow methods were employed while sampling leachate wells to minimize disturbances within the sampling zone. Low-flow sampling is accomplished by pumping water at a rate that is compatible with the rate of recovery for the well (or similar sample point) and the matrix being sampled, using methods that do not cause water surging within the well (Puls and Barcelona, 1995). Purging and sampling were performed with a peristaltic pump or, for deeper wells, a bladder pump. In a few cases with restricted access, a hand-operated Waterra™ pump or bailer was used to retrieve samples.

When low-flow sampling methods could not be performed, either "minimum purge" sampling or "maximum purge" sampling was used. Minimum purge sampling was used in a few instances where CCP surrounding the well had relatively low permeability and would not achieve a stable drawdown during low-flow pumping. This method was only used on wells that were constructed of PVC. Maximum purge sampling was used in the few instances where an existing well was constructed of stainless steel or any other metal, which may have influenced the water sample, if the well could not support low-flow sampling flow rates. In these instances, the well was completely purged the day before sampling.

Lysimeters and leachate collection systems were sampled by lowering the peristaltic pump FEP tubing to the water surface. However, in some cases, the depth to water was too great for sampling with a peristaltic pump, in which case the Waterra pump or a bladder pump connected to Teflon™ tubing was used to withdraw the sample.

Surface Water and Sluice Samples

Surface water samples were collected from ash or FGD ponds. Typically, the pond samples were accessed from structures that extended above the water, or by boat. In either case, ¼-inch FEP tubing was lowered into the water and connected to a peristaltic pump via a short length of clean flexible silicone tubing. Samples were collected from different depths by attaching the FEP tubing to a clean water level indicator and lowering the tubing to the desired depth. In most cases, samples were collected from as near the ash/water interface as possible. Seep, sluice, and outfall samples were collected directly from the sluice pipe or outfall structure in a clean plastic container or plastic dip cup sampler (Figure 2-3). FEP tubing connected to a peristaltic pump via a short length of clean flexible silicone tubing was lowered into the container and the sample was collected.

Figure 2-3
Seep Sampling

Core Samples

Core samples were collected at selected sites where porewater samples could not otherwise be obtained. A hollow-stem auger drill rig was used to advance a lined split-spoon sampler or core barrel sampler into the CCP deposit. Typically, a preliminary borehole was drilled in advance of the sample borehole in order to log the intervals where the wettest CCP was encountered, and the sampler was then advanced in a second, adjacent borehole to the selected depth. Porewater was then extracted from the core in the laboratory.

Sample Preservation

Core Samples

Core samples for leachate analyses were collected in clear, large-diameter, plastic or Teflon liners. After the liner tubes were recovered, the ends were cut so that no air volume or disturbed sample was included in the tube, and the ends of the tubes were sealed with Parafilm™, plastic end caps, and tape. Tubes were stored in coolers with dry ice for shipment to the laboratory via overnight delivery. Leachate was extracted from wet ash samples in the laboratory by centrifuge, then filtered and preserved as described below for liquid samples.

Liquid Samples

Liquid leachate samples were filtered in the field and then split for the individual analyses. A 0.45 μm filter was used for all liquid samples, and turbid samples were prefiltered using either a 1.0 or 5.0 μm filter.

There are two general approaches for preservation of speciation samples: acid preservation and cryofreezing, each with drawbacks. Acid preservation approaches have limited holding times, and require prior knowledge of redox conditions at the sample point for selection of the appropriate preservation fluid—reducing conditions are particularly problematic. Cryofreezing is not commonly used and there may be nuances to this method that have not been explored. Since prior data on redox conditions were typically not available for this sampling, the freezing approach was employed. Samples for arsenic, selenium, and chromium speciation were immediately cryofrozen in the field using liquid nitrogen (Figure 2-4), and then kept frozen on dry ice with minimal air contact until analysis to prevent changes in speciation by oxidation.

Figure 2-4
Cryofreezing a Leachate Sample in Liquid Nitrogen

Separate water samples were collected for the determination of dissolved mercury (Hg_{diss}), dissolved methyl mercury ($MeHg_{diss}$), and dimethyl mercury (DMM). New tubing, filter materials, and sampling containers were used to prevent sample contamination. Samples for Hg_{diss} and $MeHg_{diss}$ were collected using in-line filtration of a defined sample volume (40 mL for Hg_{diss} and 250 mL for $MeHg_{diss}$) and preserved immediately with HCl. The fresh filters used for each of these filtration steps were collected and stored in Petri dishes for the determination of particulate mercury (Hg_{part}) and particulate methyl mercury ($MeHg_{part}$). DMM was purged from the collected water samples with an argon stream (30 min at 1 L/min) in the field, and collected on Carbotrap™ adsorbent tubes (Figure 2-5). These tubes were dried with an argon stream opposite to the adsorption direction (10 min at 1 L/min), sealed, and kept cold and dark until analysis. All collected samples were double-bagged to prevent contamination, and clean sampling protocols (consistent with USEPA method 1631) were followed.

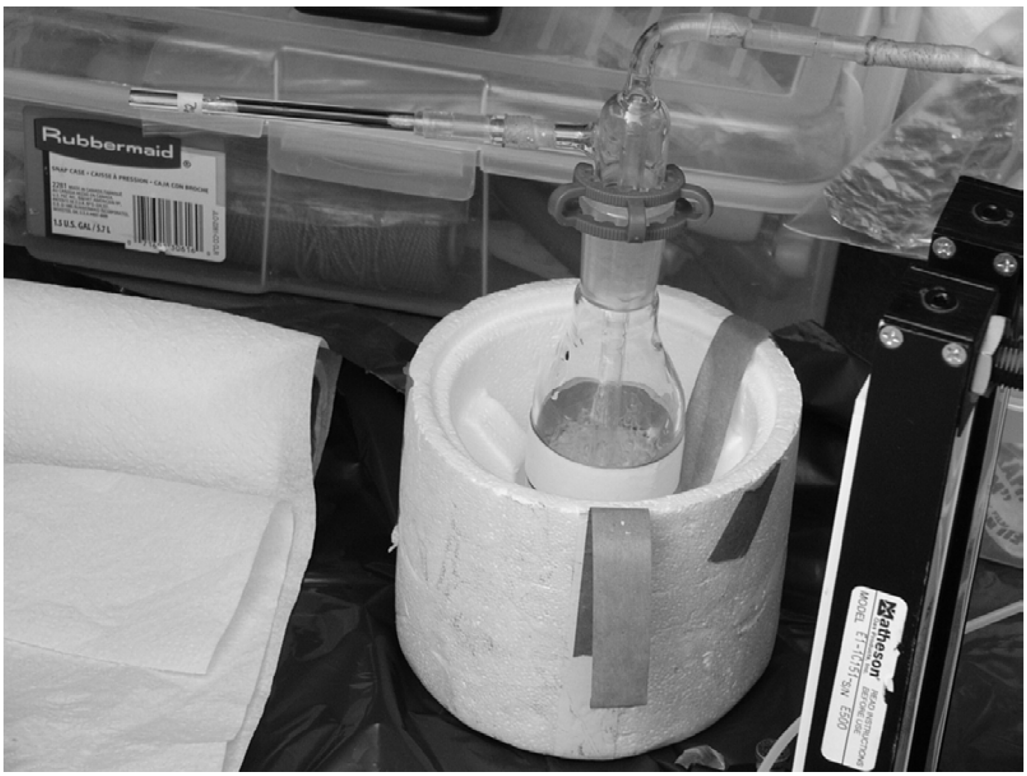


Figure 2-5
Argon Bubbling Through a Leachate Sample to Vaporize DMM

Field parameters including pH, conductivity, redox potential, and temperature were measured using an in-line flow cell and/or multi-probe sample collected during sampling.

Quality Control

A suite of quality control (QC) samples were analyzed for most sample trips, which consisted of sample and matrix spike duplicates, blanks, and reference materials as appropriate and available. Final data reported may be corrected to reflect the results of the QC samples to yield the most accurate and precise result possible.

Laboratory Preparation and Analysis

Determination of Dissolved Arsenic and Selenium by Dynamic Reaction Cell-ICP-MS (DRC-ICP-MS)

Dissolved arsenic and selenium were determined by a Perkin-Elmer DRC II ICP-MS in dynamic reaction cell (DRC) mode using ammonia as the reaction gas for the determination of arsenic, and a methane/ammonia mixture for selenium. Chromium was also determined together with selenium (under the same conditions), and the obtained results were in good agreement with the DF-ICP-MS results, which were reported in the final data set. Instrument settings and monitored isotopes are reported in Table 2-1, which also contains typical instrumental detection limits (IDLs) for each element. These IDLs represent the overall average of all analytical runs throughout the project, and are comprised of individual IDLs for each data set, which were calculated as three times the standard deviation of four instrument blanks (1 percent HNO_3) in each instrument run.

Table 2-1
Method Parameters for Total Arsenic, Selenium, and Chromium Determinations by DRC-ICP-MS

	As	Se + Cr
Measured masses	^{75}As	^{80}Se, ^{52} Cr
Monitor masses	^{77}Se, ^{78}Se, ^{82}Se	^{78}Se, ^{82}Se, ^{53}Cr
Dwell time	200 ms/isotope	200 ms/isotope
Reaction gas	NH_3 = 0.35 mL/min	NH_3 = 0.3 mL/min CH_4 = 0.45 mL/min
Bandpass	RPq = 0.6	RPq = 0.6
Typical IDL [ppb]	0.01	0.01(^{80}Se), 0.01 (^{52}Cr)

Arsenic is monoisotopic and therefore has no confirmation isotope; however, ^{77}Se was measured to compensate for the potential interference of $^{40}Ar^{35}Cl$ on ^{75}As. The major isotope ^{80}Se was used for quantification of selenium. In the absence of interferences, all isotopes of an element should yield the same result, and for most of the samples this was achieved with the selected instrument settings. However in the case of low selenium and high salt concentrations, the three measured selenium isotopes showed different results. In these cases, the result was flagged in the results table (Appendix A). ^{53}Cr was measured as a control isotope for ^{52}Cr, and the two chromium isotopes generally agreed very well. Rhodium and indium were used as internal standards. A certified reference material was analyzed with each analytical run to confirm accurate calibration, and a matrix duplicate, a matrix spike, and a matrix spike duplicate were analyzed with each batch.

Arsenic and Selenium Speciation by Ion-Chromatography Anion Self-Regenerating Suppressor ICP-MS (IC-ASRS-ICP-MS)

As(III), As(V), Se(IV), and Se(VI) were determined simultaneously by IC-ASRS-ICP-MS (Wallschläger and Roehl, 2001; Wallschläger et al., 2005) using a Dionex ion-chromatography system with anion self-regenerating suppressor (ASRS) coupled to a Perkin-Elmer DRC II (Figures 2-6 and 2-7). Method parameters are listed in Table 2-2. The ICP-MS was used in standard mode as the interfering anions are chromatographically separated in time from the analytes. Typical achieved MDLs were 0.1 ppb per species. In addition to the species mentioned above, any other unidentified anionic species such as soluble As-S compounds can be determined by this method.

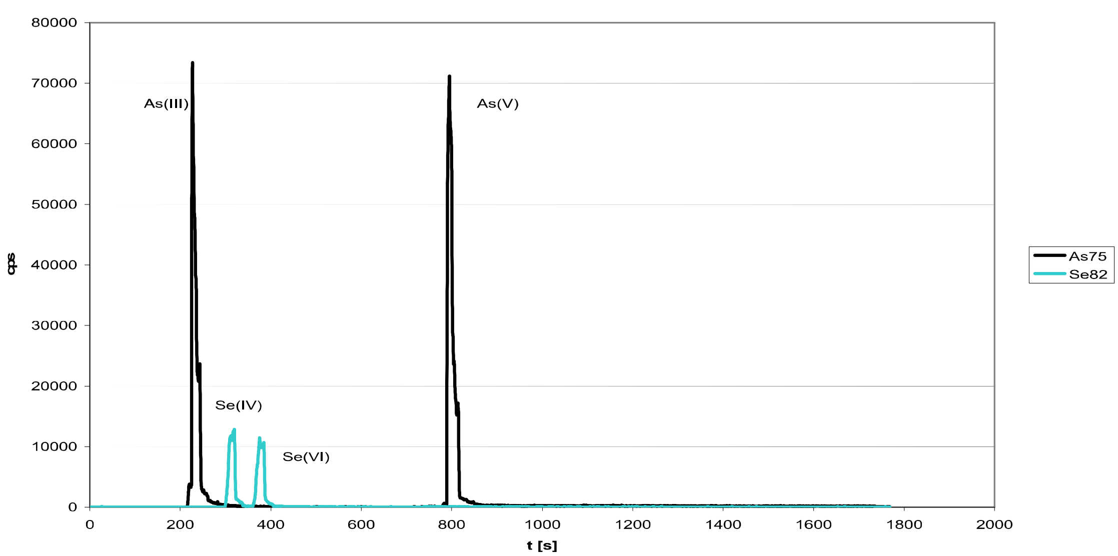


Figure 2-6
Chromatogram Showing 5 ppb Each for As(III), As(V), Se(IV), and Se(VI)

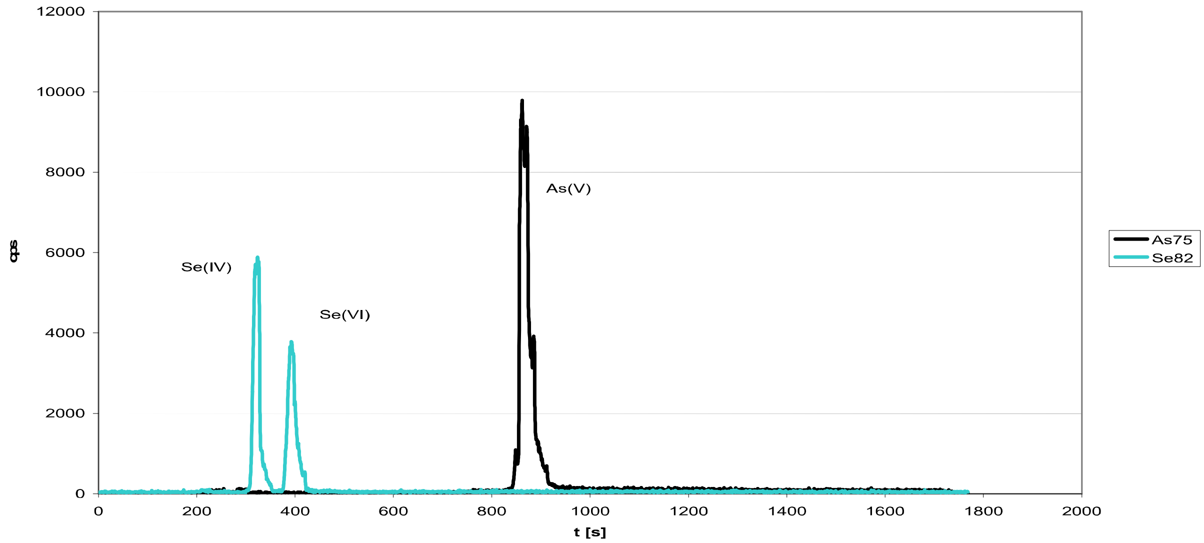


Figure 2-7
Chromatogram Showing Selenium and Arsenic Species for a Real Sample (10x dilution)

Table 2-2
Method Parameters for Arsenic, Selenium, and Chromium Speciation by IC-ASRS-DRC-ICP-MS

	Arsenic and Selenium Species	Chromium Species
Column	Dionex AS-16 4-mm + AG-16 4-mm	Dionex AS-16 4-mm + AG-16 4-mm
Eluent	sulfate in 3 mmol/L NaOH with 2 mmol/L oxalate 0→3 min: 1 mM SO_4^{2-} 3→4 min: 1→10 mM SO_4^{2-} 4→14 min: 10 mM SO_4^{2-} 14→16 min: 10→30 mM SO_4^{2-} 16→30 min: 30 mM SO_4^{2-} 30→35 min: 1 mM SO_4^{2-}	20 mM NaOH
Injection volume	1 mL	1 mL
Flow rate	1.2 mL/min	1.5 mL/min
Reaction gas	none	NH_3 = 0.3 mL/min
Bandpass	none	RPq = 0.3
Typical IDL [ppb]	0.1 As(III), 0.4 As(V), 0.05 Se(IV), 0.05 Se(VI)	0.01 Cr(III), 0.01 Cr(VI)

Determination of Dissolved Arsenic, Selenium, and Speciation in Sample Splits

A subset of the CCP leachate samples were split and forwarded to a separate laboratory for arsenic and selenium speciation analysis. These samples were field preserved using hydrochloric acid, rather than cryofreezing, and speciation analysis was performed within 48 hours of collection.

Total arsenic and selenium results were determined by Inductively Coupled Plasma Mass Spectrometry (ICP-MS) using scandium and niobium as internal standards. Due to the relatively high concentration of chloride present in the samples, an interference correction was employed for total arsenic during analysis.

Speciation for As(III), As(V), Se(IV), and Se(VI) was achieved by coupling a Hamilton PRP-X100 anion exchange column to the front end (sample introduction) of the ICP-MS instrument operated in a time domain mode. Lab Alliance pumps were used in conjunction with a gradient phosphate buffer mobile phase to elute and separate the compounds. Peak areas were used to quantitate species. Quality control measures performed during these analysis included reanalysis with greater elution times for samples where the sum of species was considerably different from the total concentration, review of chromatograms for unidentified species spikes, analytical sample duplicates, and analytical spike samples.

Chromium Speciation by Ion-Chromatography Anion Self-Regenerating Suppressor DRC-ICP-MS (IC-ASRS-DRC-ICP-MS)

Cr(III) and Cr(VI) were determined by IC-ASRS-DRC-ICP-MS using a Dionex ion-chromatography system with ASRS coupled to a Perkin-Elmer DRC II in DRC mode. This analysis was performed separately from the arsenic and selenium species determination, because Cr(III) must first be derivatized off-line to $(EDTA\text{-}Cr)^-$ before it can be determined together with Cr(VI) by anion-exchange chromatography prior to ICP-MS detection (Gürleyük and Wallschläger, 2001) (Figures 2-8 and 2-9). Modifications from the originally published method are listed in Table 2-2.

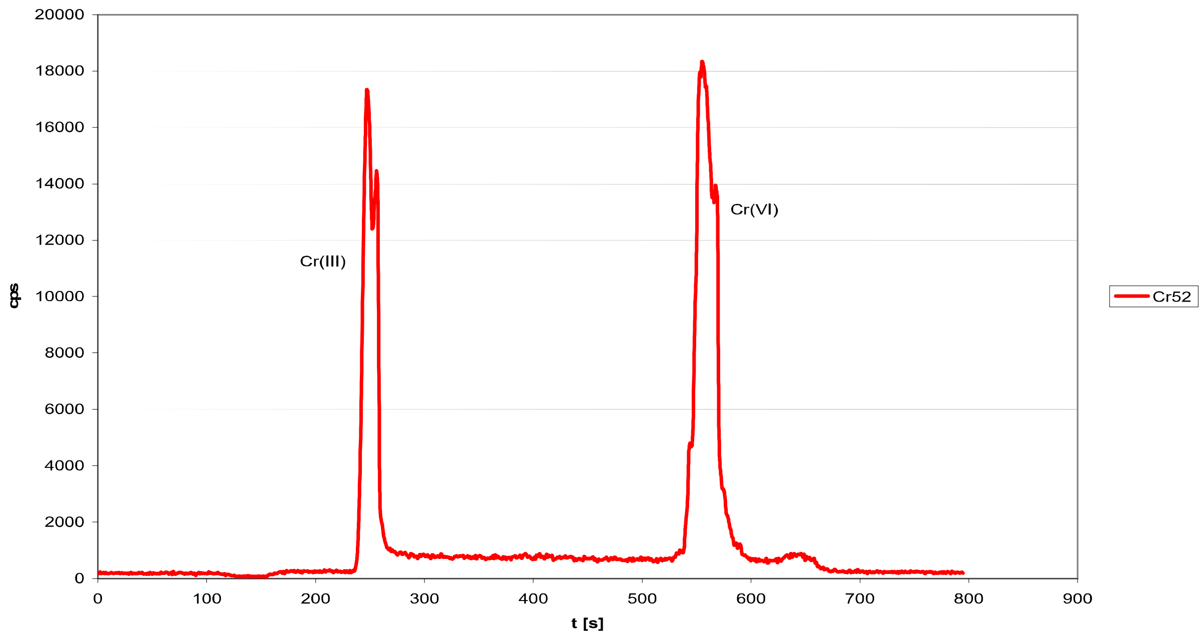


Figure 2-8
Chromatogram Showing 0.5 ppb Each for Cr(III) and Cr(VI)

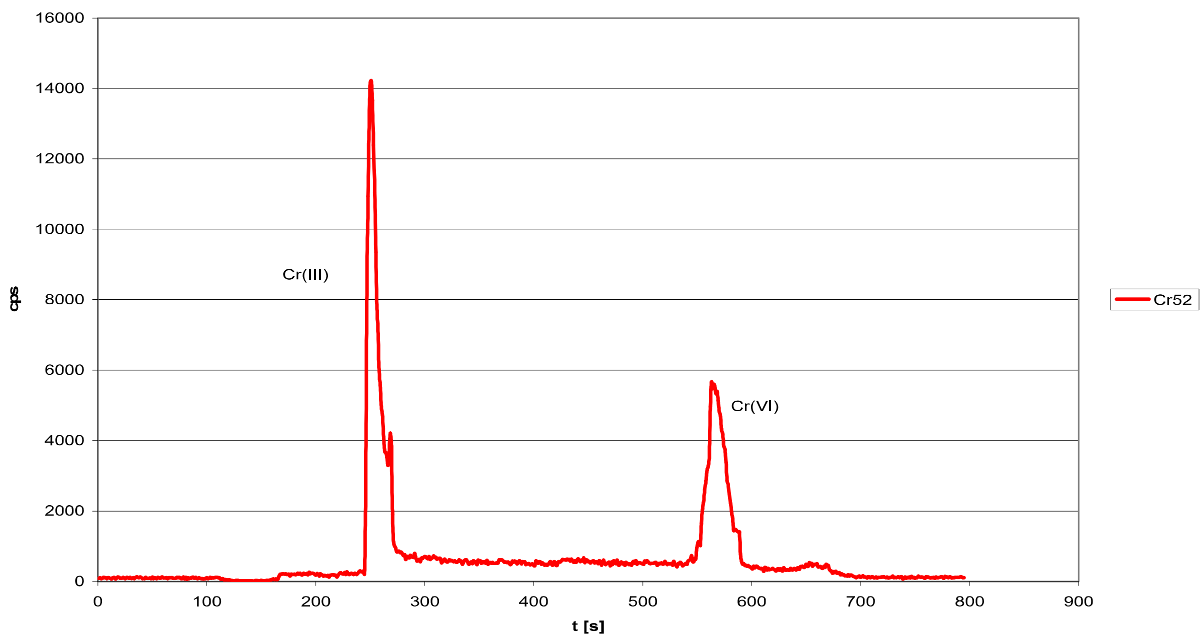


Figure 2-9
Chromatogram for Sample 034 Analyzed at a 2x Dilution

Mercury Speciation Methods

Dimethyl Mercury (DMM): DMM was purged from the collected water samples with an argon stream in the field, and collected on Carbotrap™ adsorbent tubes. These tubes were dried with an argon stream opposite to the adsorption direction, sealed, and kept cold and dark until analysis. DMM was desorbed thermally from the adsorbent trap onto an analytical trap, from which DMM was thermo-desorbed and analyzed by gas chromatography–ICP-MS (GC-ICP-MS) (similar to Lindberg et al., 2004). Figure 2-10 shows a typical chromatogram obtained by this technique: the first peak (around 70 s) is caused by elemental mercury (not quantified in this project), while the second peak (around 120 s) is DMM. The retention time of DMM is determined by analysis of DMM standards, and quantification is achieved by injecting gaseous Hg^0 standards (which is permissible, because the response of ICP-MS to mercury is species-independent).

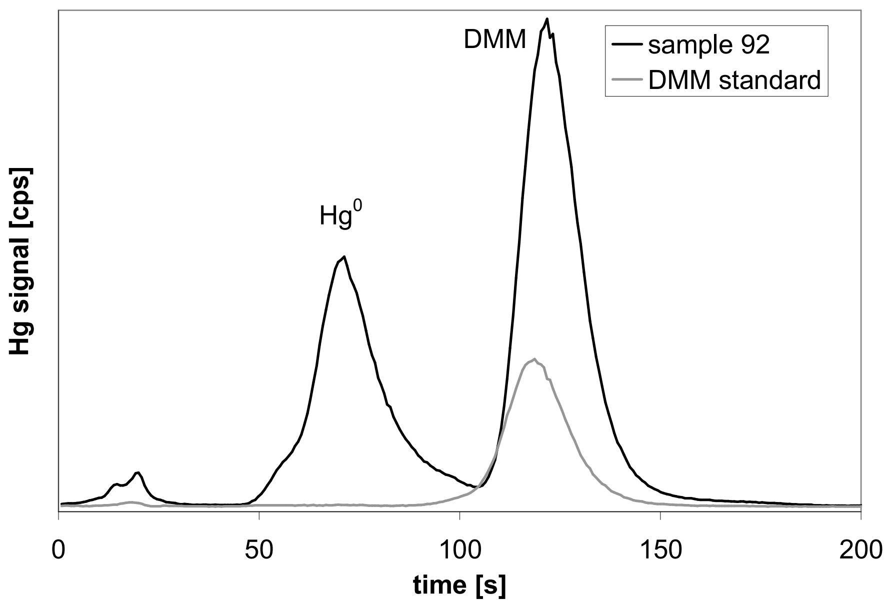


Figure 2-10
GC-ICP-MS Chromatogram for the Determination of DMM

Monomethyl Mercury (MeHg): MeHg was determined by GC-ICP-MS after derivatization to methylethyl mercury with sodium tetraethylborate. MeHg was isolated from filtered waters and particulate matter (yielding dissolved and particulate MeHg) by steam distillation as methyl mercury chloride (MeHgCl), and determined using isotope dilution with isotopically-enriched MeHg. For this purpose, each sample is spiked with a known amount of MeHg labeled with the isotope ^{201}Hg prior to the steam distillation process. The result is a GC-ICP-MS chromatogram (Figure 2-11) in which the MeHg signal (around 110 s) shows an altered isotope ratio (compared to the natural isotope abundance) reflecting the added spike. From the change in isotope ratio (in this case: $^{201}Hg/^{202}Hg$), the concentration of MeHg in the native sample is calculated. This isotope dilution technique is used routinely at Trent University for $MeHg_{diss}$ and Hg_{diss} determinations (see below), because it effectively corrects for variable procedural recoveries encountered when normal external calibration methods are used (Hintelmann & Ogrinc, 2003). Figure 2-11 shows a second peak (around 50 s), which represents some unspecific source of mercury in the instrumental setup; this signal has the "normal" mercury isotope ratio, proving that it's not MeHg.

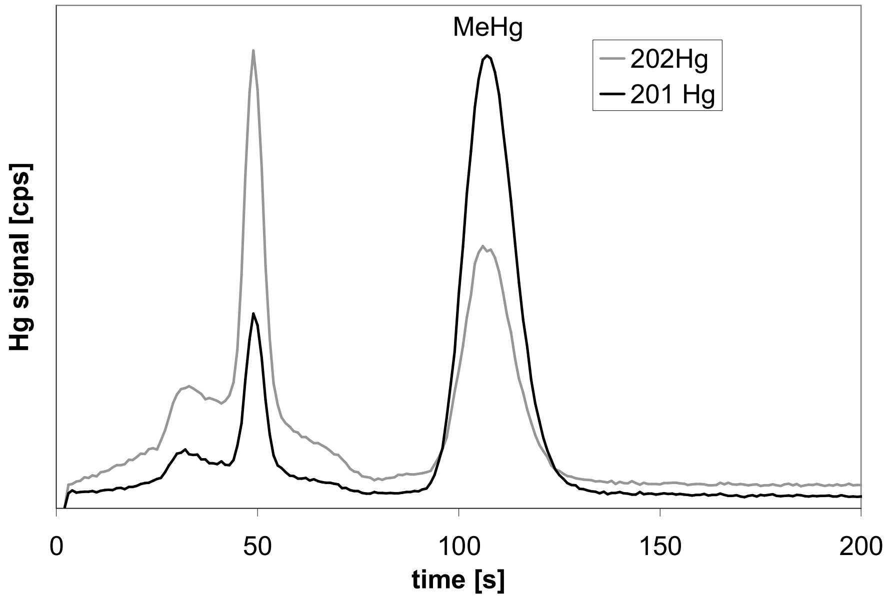


Figure 2-11
GC-ICP-MS Chromatogram for the Determination of MeHg by Isotope Dilution

<u>Mercury (Hg):</u> Total mercury in filtered waters and on filters with particulate matter (yielding dissolved and particulate mercury, Hg_{diss} and Hg_{part}) was determined by cold vapor-ICP-MS (CV-ICP-MS), also using an analog isotope dilution approach with ^{201}Hg for quantification. Samples for Hg_{diss} analysis were digested with BrCl and pre-reduced with $NH_2OH{\bullet}HCl$ prior to the CV-ICP-MS measurement (Hintelmann and Ogrinc, 2003). Table 2-3 summarizes the different analytical methods used to measure mercury speciation in the collected water samples and their typical performance characteristics. It is noteworthy that the blanks for Hg_{diss} and Hg_{part} are typically larger than many of the analyzed samples; however, since blanks are fairly constant, they can be subtracted.

Table 2-3
Mercury Speciation Methods

Parameter	Analyzed sample Volume (mL)	Typical Detection Limit (ng/L)	Typical Analytical Blank (ng/L)
DMM	105	0.005	none
$MeHg_{diss}$	50	0.02	0.02
$MeHg_{part}$	250	0.01	0.01
Hg_{diss}	n/a	0.2	1
Hg_{part}	40	1	5

Trace Element Determinations by Double-Focusing ICP-MS (DF-ICP-MS)

A Thermo Finnigan ELEMENT2 double-focusing inductively coupled plasma-mass spectrometer (DF-ICP-MS) was used in medium resolution mode to determine 22 elements of interest (Table 2-4). Each sample was analyzed at three different dilutions (500x, 100x, and 20x) to cover the different concentration ranges of the elements. Due to the high salt load of the samples, a dilution factor of less than 20x might lead to instrument damage and was therefore avoided; however, all field blanks and equipment blanks were analyzed undiluted because they did not contain salts. According to the typical concentrations encountered for different elements, the 500x diluted samples were analyzed for Li, B, Al, Si, Fe, Sr, and Mo; the 100x diluted samples for Li, Be, B, Al, V, Cr, Mn, Fe, Co, Ni, Cu, Zn, Sr, Mo, Ag, Cd, Sb, Ba, Tl, Pb, and U; and the 20x diluted samples for Li, Be, Al, V, Cr, Mn, Fe, Co, Ni, Cu, Zn, Mo, Ag, Cd, Sb, Ba, Tl, Pb, and U. If one element was analyzed at more than one dilution, the result obtained with the lowest dilution factor under consideration of the calibrated range was reported.

Table 2-4
Trace Metals by DF-ICP-MS

Element	Measured Isotope	Control Isotope	Isotopes Agree?	Typical IDL [ppb]
Aluminum	^{27}Al	monoisotopic		0.1
Antimony	^{121}Sb	^{123}Sb	Y	0.004
Barium	^{136}Ba	^{137}Ba	Y	0.06
Beryllium	^{9}Be	monoisotopic		0.01
Boron	^{10}B	^{11}B	Y	0.2
Cadmium	^{110}Cd	^{111}Cd, ^{114}Cd	N	0.004
Chromium	^{53}Cr	^{52}Cr	Y	0.01
Cobalt	^{59}Co	monoisotopic		0.002
Copper	^{65}Cu	^{63}Cu	Y	0.01
Iron	^{56}Fe	^{57}Fe	Y	0.1
Lead	^{208}Pb	^{206}Pb, ^{207}Pb	Y	0.003
Lithium	^{7}Li	not measurable		0.04
Manganese	^{55}Mn	monoisotopic		0.009
Molybdenum	^{98}Mo	^{95}Mo	Y	0.04
Nickel	^{60}Ni	^{58}Ni	Y (except in samples with high Fe concentrations)	0.03
Silica	^{28}Si	^{30}Si	Y	0.3
Silver	^{107}Ag	^{109}Ag	Y? (concentrations close to MDL)	0.005
Strontium	^{88}Sr	^{87}Sr	Y (after Rb correction of ^{87}Sr)	0.05
Thallium	^{205}Tl	^{203}Tl	Y? (concentrations close to MDL)	0.002
Uranium	^{238}U	not available	no interferences	0.001
Vanadium	^{51}V	^{50}V	N	0.004
Zinc	^{66}Zn	^{68}Zn	Y? (concentrations close to MDL)	0.09

At least two isotopes for each element were measured (if possible) to verify the absence of spectrometric interferences. Scandium, indium, rhodium, and germanium were used as internal standards to monitor and correct instrument drift and sample uptake effects. All measured and control isotopes are listed in Table 2-4. Typically, the results obtained for the measured and the control isotope were identical (within the analytical uncertainty); however, some exceptions are explained below. Average IDLs are also listed in Table 2-4. The method detection limit (MDL) was estimated as the IDL times the applicable dilution factor of the analyzed sample. The IDL/MDL was determined with each analytical run and varied slightly depending on the instrument performance on that day. All data reported were instrument-blank corrected. For quality control purposes, a certified reference material (CRM) was analyzed at two different dilutions per analytical run to confirm an accurate calibration. For each sample batch (usually one per sampling trip) one randomly selected sample was analyzed in duplicate and spiked and analyzed in duplicate to assess accuracy and reproducibility.

For some of the elements listed in Table 2-4, the results obtained for the measured and the control isotope did not match. Several elements (e.g., Ag, Zn, Tl) are present in most samples at concentrations of only 5-10 times the detection limit, so that analytical uncertainty and/or insufficient number of samples with detectable concentrations prevented a meaningful isotope comparison. In other cases, the control isotope had a very low abundance and although the sample concentration was very well detectable for the main isotope, the quantification by the minor isotope was impaired by low signal intensities (e.g., ^{50}V; natural abundance 0.25 percent). Also, in the used concentration range, ^{6}Li was not detected in medium resolution mode by the instrument; therefore, it was not used for confirming ^{7}Li.

In medium (or even high) resolution mode, some isobaric and polyatomic interferences could not be resolved: ^{58}Ni was not separated from ^{58}Fe in medium resolution mode (required resolution ~30,000; available resolution ~ 10,000). As the ^{58}Fe abundance is only 0.28 percent, the associated error is normally negligible; however, if the iron concentrations are extremely high, as in some of the analyzed samples, ^{58}Ni will be affected. Also, ^{87}Sr was also not separated from ^{87}Rb in medium resolution mode (required resolution ~300,000); however, the error in this case is not negligible as ^{87}Rb has an abundance of 27.8 percent. If ^{87}Sr is corrected for ^{87}Rb, both ^{87}Sr and ^{88}Sr yield identical results. For cadmium, both ^{111}Cd and ^{114}Cd were interfered with by MoO (required resolution ~100K and ~80K, respectively); in addition, ^{114}Cd was also affected by an isobaric interference of ^{114}Sn. Based on those considerations, ^{110}Cd was used for quantification. Generally, as spectroscopic interferences are normally positive, in the event that two isotopes yield a different result, the lower concentration will most likely be the uninterfered and therefore deliver the correct result.

Ancillary Parameters

Redox potential, pH, conductivity, dissolved oxygen, and temperature were determined in the field on the filtered samples with a YSI multiprobe (for wells, this measurement was made immediately after the low-flow conditions had stabilized; for all other types of water samples, this was done prior to collecting all other aliquots). Separate aliquots were used for these analyses and discarded afterwards.

Sodium, potassium, magnesium, and calcium were determined by cation-exchange chromatography with suppressed conductivity detection, and chloride and sulfate were determined by anion-exchange chromatography using the same detection principle, following standard methods. Total carbon (TC) and total inorganic carbon (TIC) were determined by flow injection-infrared spectrometry (Shimadzu Total Organic Carbon Analyzer) following standard methods, where TIC is liberated from the sample by addition of HCl, while TC is liberated by oxygen combustion; total organic carbon (TOC) is then determined by difference TC-TIC, which may lead to imprecise results in samples with low TOC content.

3
SAMPLE SUMMARY

Site and Sample Attributes

Location

The 33 sample sites are concentrated in the eastern United States where coal-fired power plants predominate (Figure 3-1). Attributes of sampled sites are listed in Table 3-1, and leachate sample attributes are listed in Table 3-2.

Symbols indicate number of sites within a state, but do not correspond to location of sampled sites.

	FA		FGD/SDA	
	LF	IMP	LF	IMP
Sites Completed Thru 2005	★	▼	✦	▾

Figure 3-1
Sample Site Locations by State

Facility Type

Samples were collected at 15 impoundments and 17 landfills (Table 3-1). One of the sites counted as an impoundment is the 14093 site. This site is a landfill that receives ash originally sluiced to an impoundment. Washing of ash during sluicing is believed to have an effect on ash leachate concentration; therefore, this site was counted as an impoundment.

The 27413 site is not classified as a landfill or impoundment. Ash was originally sluiced to this site, and later it was managed dry. There were no data to indicate whether the samples were collected in areas where ash was sluiced or managed dry; therefore, this site was not used in comparisons of landfill and impoundment ash.

Sample Methods

Landfill Samples

All of the 29 landfill leachate samples represent interstitial water. Three samples were collected from wells screened in the CCP, two samples were collected from lysimeters screened immediately beneath the CCP, one was collected from a surface seep, and 19 were collected from leachate collection systems (Table 3-3). The remaining four samples were core samples from soil borings; however, these samples did not yield sufficient water for analysis when centrifuged in the laboratory. As a result, 25 landfill leachate samples were analyzed.

The four dry cores were each collected from different sites, and, in each case, the dry core was the only sample collected at that site. These samples and sites are not included in the discussions that follow. As a result, for the remainder of this report, only 29 of the 33 sites will be referenced.

Impoundment Samples

Twenty-seven of the 53 impoundment samples represent interstitial water. These include eight samples collected from wells screened in the CCP, 13 samples collected from drive-point piezometers or push point samplers, three seep samples, and three core extracts (Table 3-3). The remaining 26 leachate samples include 12 collected from impoundments near the ash-water interface, and 14 samples collected from sluice lines or at impoundment outfalls.

Other Samples

The three leachate samples from site 27413 are interstitial water collected from temporary leachate wells.

Source Power Plant Attributes

Boiler Type

The majority of sites (24 of 29) sampled received CCP from pulverized coal (PC) plants with dry-bottom boilers (Table 3-1), representing 71 of the 81 leachate samples (Table 3-2). One site (one sample) received CCP from a wet-bottom PC boiler, and three sites (four samples) received CCP from cyclone boilers. The remaining site (five samples) received CCP from a plant that has both dry-bottom PC boilers and cyclones.

A variety of firing configurations are represented in the PC boilers including:

- Tangential: 10 sites, 34 samples
- Wall-fired (mostly opposed): 7 sites, 18 samples
- Multiple configurations: 9 sites, 25 samples

Source Coal

Most sites (11 sites, 48 samples) received CCP from power plants that burned bituminous coal (Tables 3-1 and 3-2). The power plant feeding one of these 11 sites (23214) also burns 5 percent petroleum coke.

Seven sites (13 samples) received CCP from plants that burn subbituminous coal, and four sites (five samples) received CCP from lignite-burning plants. The subbituminous and lignite samples will be grouped together in discussions that follow.

Four sites (seven samples) received CCP from plants that burn a blend of fuels:

- 22346: formerly bituminous, coal units burned a blend of 80 percent subbituminous and 20 percent bituminous coal at the time of sampling. This site also received oil ash.
- 22347: formerly bituminous, coal units burned a blend of 80 percent subbituminous and 20 percent bituminous coal at the time of sampling.
- 25410A and 25410B: an undetermined blend of subbituminous and bituminous coals, plus used tires and petroleum coke.

Three sites (eight samples) have CCP derived from a mixture of sources:

- 50183 received CCP from three different power plants burning bituminous and subbituminous coal.
- 27413 and 50210 received CCP from power plants that switched from bituminous to subbituminous coal.

Emission Controls

Six of the 29 sites received CCP from flue gas desulfurization (FGD) systems, the remaining sites received coal ash, either from plants without FGD systems or that was collected prior to the FGD system (Tables 3-1 and 3-2).

Fly Ash

Most fly ash samples came from plants (17 plants, 48 samples) with cold-side electrostatic precipitators (ESPs). Two sites (7 samples) received CCP from plants with hot-side ESPs and one site (1 sample) received CCP from a plant with a fabric filter. Three sites (11 samples) received CCP from multiple sources:

- 50183 received CCP from three plants, two have cold-side precipitators, and one has a hot-side ESP.
- 33104 received CCP from one plant with cold-side and hot-side ESPs on different units.
- 50213 received CCP from a plant with a cold-side ESP on two units, and a hot-side ESP and fabric filter on another unit.

Thirteen of the ash sample sites (41 samples) received CCP from units with flue gas conditioning to improve precipitator performance. NOx controls included low-NOx burners (12 samples), overfired air (5 samples), selective catalytic reduction (5 samples), and multiple types.

FGD

Five of the six FGD sites, representing 13 samples, received CCP from wet FGD systems. Four of these systems were coupled with cold-side ESPs; three of the four systems with ESPs systems used natural oxidation while the other used inhibited oxidation. The other wet FGD system was not coupled with an ESP or fabric filter, and used forced air oxidation. The FGD systems feeding three of these sites used magnesium-lime sorbent, one used lime, and one used limestone.

One site (1 sample) received CCP from a spray dryer system coupled with a fabric filter. The FGD sorbent used in this system was lime.

At one of the six FGD units, flue gas conditioning was used to improve precipitator performance. That unit also had a low-NOx burner.

Table 3-1
Attributes of Sample Sites and Source Power Plants

Site	Source Fuel Type	Source Plant Boiler Type	PC Boiler Firing	Source Plant Particulate Collection	Source Plant SO2 Control	Source Plant SO2 Sorbent	Source Plant Flue Gas Cond.	Source Plant NOx Control	Byproducts Managed	DUP	IMP	LF	QC
23214	Subbit	Cyclone		ESP cold-side	None	None	None	Combustion-OFA	FA Class C			1	
50183	Mix	Dry Bottom PC Boiler	multiple types	Multiple types	None	None	Yes	Multiple types	FA, BA			4	1
33106	Bit	Dry Bottom PC Boiler	tangential	ESP cold-side	None	None	Yes	Multiple types	FA, BA	1	7		3
20094A	Bit	Dry Bottom PC Boiler	wall-fired opposed	ESP multiple	None	None	None	Multiple types	FA, BA			1*	
20094B	Bit	Dry Bottom PC Boiler	wall-fired opposed	ESP multiple	None	None	None	Multiple types	FA, BA			1*	
34186A	Lig	Dry Bottom PC Boiler	tangential	ESP cold-side	Wet-natural	Mg-Lime	None	Multiple types	FA			1	
34186B	Lig	Dry Bottom PC Boiler	tangential	ESP cold-side	Wet-natural	Mg-Lime	None	Multiple types	FGD, BA		2		2
34186C	Lig	Dry Bottom PC Boiler	tangential	ESP cold-side	Wet-natural	Mg-Lime	None	Multiple types	FGD, FA, BA	1		1	
33104	Bit	Dry Bottom PC Boiler	tangential	Multiple types	None	None	None	Postcombustion SCR	FA, BA	1	5		1
50408	Bit	Dry Bottom PC Boiler	wall-fired	ESP cold-side	None	None	None	Combustion-none	FA, BA			1	
35015A	Bit	Dry Bottom PC Boiler	tangential	ESP cold-side	Wet-natural	Mg-Lime	Yes	Combustion-LNB	FGD, FA			6	
35015B	Bit	Multiple types	multiple types	ESP cold-side	None	None	None	Combustion-LNB	FA	1	5		1
31192	Subbit	Dry Bottom PC Boiler	tangential	Fabric filter	Wet-natural	Limestone	None	Other	FA, FGD, BA			1*	
13115A	Subbit	Dry Bottom PC Boiler	tangential	ESP cold-side	None	None	Yes	Multiple types	BA, FA		3		
13115B	Bit	Dry Bottom PC Boiler	tangential	ESP cold-side	None	None	Yes	Other	FA, BA		3		

Table 3-1
Attributes of Sample Sites and Source Power Plants (continued)

Site	Source Fuel Type	Source Plant Boiler Type	PC Boiler Firing	Source Plant Particulate Collection	Source Plant SO2 Control	Source Plant SO2 Sorbent	Source Plant Flue Gas Cond.	Source Plant NOx Control	Byproducts Managed	DUP	IMP	LF	QC
49003A	Bit	Dry Bottom PC Boiler	wall-fired opposed	ESP cold-side	None	None	Yes	Multiple types	FA		8		
49003B	Bit	Dry Bottom PC Boiler	wall-fired opposed	ESP cold-side	None	None	None	Combustion-LNB	FA			4	2
22346	Blend	Dry Bottom PC Boiler	multiple types	ESP cold-side	None	None	Yes	Multiple types	FA, OA	1	3		3
22347	Blend	Dry Bottom PC Boiler	tangential	ESP cold-side	None	None	Yes	Other	FA		1		
40109	Bit	Dry Bottom PC Boiler	tangential	ESP hot-side	None	None	None	Multiple types	FA, BA	1	5		1
27412	Subbit	Dry Bottom PC Boiler	wall-fired opposed	ESP cold-side	None	None	None	Combustion-OFA	FA, BA			1*	
27413	Mix	Dry Bottom PC Boiler	multiple types	ESP cold-side	None	None	Yes	Multiple types	FA				3
50210	Mix	Dry Bottom PC Boiler	multiple types	ESP cold-side	None	None	None	Multiple types	FA, BA			1	
43034	Lig	Wet Bottom PC Boiler	wall-fired	ESP cold-side	Wet-inhib	Limestone	None	Multiple types	FGD,FA			1	
50212	Subbit	Dry Bottom PC Boiler	wall-fired	ESP cold-side	None	None	Yes	Multiple types	FA	1		2	2[+]
23223A	Subbit	Dry Bottom PC Boiler	multiple types	Fabric filter	Spray Dryer	Lime	no data	Multiple types	SDA			1	
23223B	Subbit	Dry Bottom PC Boiler	multiple types		Wet-FO	Lime	no data	Multiple types	FGD		3		
25410A	Blend	Cyclone		ESP cold-side	None	None	Yes	Combustion-OFA	FA, BA		2		
25410B	Blend	Cyclone		ESP cold-side	None	None	Yes	Combustion-OFA	FA		1		
50211	Bit	Dry Bottom PC Boiler	wall-fired front	Fabric filter	None	None	no data	Combustion-LNB	FA			1	
14093	Bit	Dry Bottom PC Boiler	multiple types	ESP cold-side	None	None	Multiple	Multiple types	FA (sluiced)	1	3		2

Table 3-1
Attributes of Sample Sites and Source Power Plants (continued)

Site	Source Fuel Type	Source Plant Boiler Type	PC Boiler Firing	Source Plant Particulate Collection	Source Plant SO2 Control	Source Plant SO2 Sorbent	Source Plant Flue Gas Cond.	Source Plant NOx Control	Byproducts Managed	DUP	IMP	LF	QC
43035	Subbit	Dry Bottom PC Boiler	wall-fired opposed	ESP hot-side	None	None	None	Combustion-LNB	FA,BA,EA (sluiced)	1	2		1
50213	Subbit	Dry Bottom PC Boiler	multiple types	Multiple types	None	None	Multiple	Multiple types	FA			2	

Notes:

Ash at site 27413 was first sluiced, then managed dry.

* indicates that core sample collected at this site did not yield sufficient water for analysis.

[+] one of the two leachate samples collected at site 50212 was treated with CO_2

Abbreviations:

Bit = bituminous; Subbit = Subbituminous; Mix = CCP from different units burning different coals; Blend = CCP from a single unit burning two different fuels

PC = pulverized coal; ESP = electrostatic precipitator; OFA = overfired air; LNB = low-NOx burner

FA = fly ash; BA = bottom ash; EA = economizer ash; FGD = flue gas desulfurization sludge; OA = oil ash

LF = landfill; IMP = impoundment; DUP = duplicate sample; QC = quality control sample

Table 3-2
Leachate Sample Attributes

Sample ID	Source	Byproduct	Source Fuel Type	Site	Source Plant PC Boiler Type	PC Boiler Firing	Source Plant Particulate Collection	Source Plant SO2 Control	Source Plant SO2 Sorbent	Source Plant Flue Gas Cond.	Source Plant NOx Control
001	Landfill	FA,BA	Mix	50210	Dry Bottom PC Boiler	multiple types	ESP cold-side	None	None	None	Multiple types
002	Landfill	FA	Subbit	50213	Dry Bottom PC Boiler	multiple types	Multiple types	None	None	Multiple	Multiple types
003	Landfill	FA	Subbit	50213	Dry Bottom PC Boiler	multiple types	Multiple types	None	None	Multiple	Multiple types
004	Landfill	FA,BA	Mix	50183	Dry Bottom PC Boiler	multiple types	Multiple types	None	None	Yes	Multiple types
005	Landfill	FA,BA	Mix	50183	Dry Bottom PC Boiler	multiple types	Multiple types	None	None	Yes	Multiple types
006	Landfill	SDA	Subbit	23223A	Dry Bottom PC Boiler	multiple types	Fabric filter	Spray Dryer	Lime	no data	Multiple types
007	Impoundment	FGD	Subbit	23223B	Dry Bottom PC Boiler	multiple types		Wet-FO	Lime	no data	Multiple types
008	Impoundment	FGD	Subbit	23223B	Dry Bottom PC Boiler	multiple types		Wet-FO	Lime	no data	Multiple types
009	Impoundment	FGD	Subbit	23223B	Dry Bottom PC Boiler	multiple types		Wet-FO	Lime	no data	Multiple types
010	Landfill	FA	Subbit	23214	Cyclone		ESP cold-side	None	None	None	Combustion-OFA
012	Impoundment	FA	Bit	14093	Dry Bottom PC Boiler	multiple types	ESP cold-side	None	None	Multiple	Multiple types
013	Impoundment	FA	Bit	14093	Dry Bottom PC Boiler	multiple types	ESP cold-side	None	None	Multiple	Multiple types
014	Impoundment	FA	Bit	14093	Dry Bottom PC Boiler	multiple types	ESP cold-side	None	None	Multiple	Multiple types
015	Impoundment	FA,BA	Blend	25410A	Cyclone		ESP cold-side	None	None	Yes	Combustion-OFA
016	Impoundment	FA,BA	Blend	25410A	Cyclone		ESP cold-side	None	None	Yes	Combustion-OFA
017	Impoundment	FA,BA	Subbit	13115A	Dry Bottom PC Boiler	tangential	ESP cold-side	None	None	Yes	Multiple types
018	Impoundment	FA,BA	Bit	13115B	Dry Bottom PC Boiler	tangential	ESP cold-side	None	None	Yes	Other
019	Impoundment	FA	Subbit	13115A	Dry Bottom PC Boiler	tangential	ESP cold-side	None	None	Yes	Multiple types
020	Impoundment	FA,BA	Subbit	13115A	Dry Bottom PC Boiler	tangential	ESP cold-side	None	None	Yes	Multiple types
021	Impoundment	FA	Bit	49003A	Dry Bottom PC Boiler	wall-fired opposed	ESP cold-side	None	None	Yes	Multiple types
022	Impoundment	FA	Bit	49003A	Dry Bottom PC Boiler	wall-fired opposed	ESP cold-side	None	None	Yes	Multiple types
023	Impoundment	FA	Bit	49003A	Dry Bottom PC Boiler	wall-fired opposed	ESP cold-side	None	None	Yes	Multiple types
024	Landfill	FA	Bit	49003B	Dry Bottom PC Boiler	wall-fired opposed	ESP cold-side	None	None	None	Combustion-LNB
025	Landfill	FA	Bit	49003B	Dry Bottom PC Boiler	wall-fired opposed	ESP cold-side	None	None	None	Combustion-LNB
026	Impoundment	FA	Bit	49003A	Dry Bottom PC Boiler	wall-fired opposed	ESP cold-side	None	None	Yes	Multiple types
027	Landfill	FGD, FA	Bit	35015A	Dry Bottom PC Boiler	tangential	ESP cold-side	Wet-natural	Mg-Lime	Yes	Combustion-LNB
028	Landfill	FGD, FA	Bit	35015A	Dry Bottom PC Boiler	tangential	ESP cold-side	Wet-natural	Mg-Lime	Yes	Combustion-LNB
029	Landfill	FGD, FA	Bit	35015A	Dry Bottom PC Boiler	tangential	ESP cold-side	Wet-natural	Mg-Lime	Yes	Combustion-LNB

Table 3-2
Leachate Sample Attributes (continued)

Sample ID	Source	Byproduct	Source Fuel Type	Site	Source Plant PC Boiler Type	PC Boiler Firing	Source Plant Particulate Collection	Source Plant SO2 Control	Source Plant SO2 Sorbent	Source Plant Flue Gas Cond.	Source Plant NOx Control
030	Impoundment	FA	Bit	35015B	Multiple types	multiple types	ESP cold-side	None	None	None	Combustion-LNB
031	Impoundment	FA	Bit	35015B	Multiple types	multiple types	ESP cold-side	None	None	None	Combustion-LNB
032	Impoundment	FA,BA	Bit	35015B	Multiple types	multiple types	ESP cold-side	None	None	None	Combustion-LNB
037	Impoundment	FA	Bit	33106	Dry Bottom PC Boiler	tangential	ESP cold-side	None	None	Yes	Multiple types
038	Impoundment	FA	Bit	33106	Dry Bottom PC Boiler	tangential	ESP cold-side	None	None	Yes	Multiple types
039	Impoundment	FA	Bit	33106	Dry Bottom PC Boiler	tangential	ESP cold-side	None	None	Yes	Multiple types
042	Impoundment	FA	Bit	33106	Dry Bottom PC Boiler	tangential	ESP cold-side	None	None	Yes	Multiple types
043	Impoundment	FA	Bit	33106	Dry Bottom PC Boiler	tangential	ESP cold-side	None	None	Yes	Multiple types
044	Impoundment	FA	Bit	33106	Dry Bottom PC Boiler	tangential	ESP cold-side	None	None	Yes	Multiple types
049	Impoundment	FA,BA	Bit	33106	Dry Bottom PC Boiler	tangential	ESP cold-side	None	None	Yes	Multiple types
051	Impoundment	FA	Bit	40109	Dry Bottom PC Boiler	tangential	ESP hot-side	None	None	None	Multiple types
052	Impoundment	FA	Bit	40109	Dry Bottom PC Boiler	tangential	ESP hot-side	None	None	None	Multiple types
053	Impoundment	FA	Bit	40109	Dry Bottom PC Boiler	tangential	ESP hot-side	None	None	None	Multiple types
057	Impoundment	FA,BA	Bit	40109	Dry Bottom PC Boiler	tangential	ESP hot-side	None	None	None	Multiple types
059	Impoundment	FA,BA	Bit	40109	Dry Bottom PC Boiler	tangential	ESP hot-side	None	None	None	Multiple types
061	Impoundment	FA	Bit	33104	Dry Bottom PC Boiler	tangential	Multiple types	None	None	None	Postcombustion SCR
062	Impoundment	FA	Bit	33104	Dry Bottom PC Boiler	tangential	Multiple types	None	None	None	Postcombustion SCR
064	Impoundment	FA	Bit	33104	Dry Bottom PC Boiler	tangential	Multiple types	None	None	None	Postcombustion SCR
069	Impoundment	FA,BA	Bit	33104	Dry Bottom PC Boiler	tangential	Multiple types	None	None	None	Postcombustion SCR
070	Impoundment	FA,BA	Bit	33104	Dry Bottom PC Boiler	tangential	Multiple types	None	None	None	Postcombustion SCR
079	Impoundment	FA,OA	Blend	22346	Dry Bottom PC Boiler	multiple types	ESP cold-side	None	None	Yes	Multiple types
082	Impoundment	FA,OA	Blend	22346	Dry Bottom PC Boiler	multiple types	ESP cold-side	None	None	Yes	Multiple types
083	Impoundment	FA	Blend	22347	Dry Bottom PC Boiler	tangential	ESP cold-side	None	None	Yes	Other
084	Impoundment	FA,OA	Blend	22346	Dry Bottom PC Boiler	multiple types	ESP cold-side	None	None	Yes	Multiple types
090	See Notes	FA	Mix	27413	Dry Bottom PC Boiler	multiple types	ESP cold-side	None	None	Yes	Multiple types
091	See Notes	FA	Mix	27413	Dry Bottom PC Boiler	multiple types	ESP cold-side	None	None	Yes	Multiple types
092	See Notes	FA	Mix	27413	Dry Bottom PC Boiler	multiple types	ESP cold-side	None	None	Yes	Multiple types

Table 3-2
Leachate Sample Attributes (continued)

Sample ID	Source	Byproduct	Source Fuel Type	Site	Source Plant PC Boiler Type	PC Boiler Firing	Source Plant Particulate Collection	Source Plant SO2 Control	Source Plant SO2 Sorbent	Source Plant Flue Gas Cond.	Source Plant NOx Control
093	Landfill	FA,BA	Subbit	27412	Dry Bottom PC Boiler	wall-fired opposed	ESP cold-side	None	None	None	Combustion-OFA
097	Landfill	FA	Subbit	50212	Dry Bottom PC Boiler	wall-fired	ESP cold-side	None	None	Yes	Multiple types
098	Landfill	FA,BA	Mix	50183	Dry Bottom PC Boiler	multiple types	Multiple types	None	None	Yes	Multiple types
099	Landfill	FA,BA	Mix	50183	Dry Bottom PC Boiler	multiple types	Multiple types	None	None	Yes	Multiple types
101	Landfill	FA,BA	Bit	50408	Dry Bottom PC Boiler	wall-fired	ESP cold-side	None	None	None	Combustion-none
102	Landfill	FA	Bit	50211	Dry Bottom PC Boiler	wall-fired front	Fabric filter	None	None	no data	Combustion-LNB
105	Impoundment	FGD	Lig	34186B	Dry Bottom PC Boiler	tangential	ESP cold-side	Wet-natural	Mg-Lime	None	Multiple types
106	Landfill	FGD,FA,BA	Lig	34186C	Dry Bottom PC Boiler	tangential	ESP cold-side	Wet-natural	Mg-Lime	None	Multiple types
107	Impoundment	FGD	Lig	34186B	Dry Bottom PC Boiler	tangential	ESP cold-side	Wet-natural	Mg-Lime	None	Multiple types
108	Landfill	FA	Lig	34186A	Dry Bottom PC Boiler	tangential	ESP cold-side	Wet-natural	Mg-Lime	None	Multiple types
111	Landfill	FA	Bit	49003B	Dry Bottom PC Boiler	wall-fired opposed	ESP cold-side	None	None	None	Combustion-LNB
112	Landfill	FA	Bit	49003B	Dry Bottom PC Boiler	wall-fired opposed	ESP cold-side	None	None	None	Combustion-LNB
113	Impoundment	FA	Bit	49003A	Dry Bottom PC Boiler	wall-fired opposed	ESP cold-side	None	None	Yes	Multiple types
114	Impoundment	FA	Bit	49003A	Dry Bottom PC Boiler	wall-fired opposed	ESP cold-side	None	None	Yes	Multiple types
115	Impoundment	FA	Bit	49003A	Dry Bottom PC Boiler	wall-fired opposed	ESP cold-side	None	None	Yes	Multiple types
116	Impoundment	FA	Bit	49003A	Dry Bottom PC Boiler	wall-fired opposed	ESP cold-side	None	None	Yes	Multiple types
118	Impoundment	FA,BA	Bit	35015B	Multiple types	multiple types	ESP cold-side	None	None	None	Combustion-LNB
119	Impoundment	FA,BA	Bit	35015B	Multiple types	multiple types	ESP cold-side	None	None	None	Combustion-LNB
120	Landfill	FGD, FA	Bit	35015A	Dry Bottom PC Boiler	tangential	ESP cold-side	Wet-natural	Mg-Lime	Yes	Combustion-LNB
121	Landfill	FGD, FA	Bit	35015A	Dry Bottom PC Boiler	tangential	ESP cold-side	Wet-natural	Mg-Lime	Yes	Combustion-LNB
122	Landfill	FGD, FA	Bit	35015A	Dry Bottom PC Boiler	tangential	ESP cold-side	Wet-natural	Mg-Lime	Yes	Combustion-LNB
123	Landfill	FA	Bit	20094A	Dry Bottom PC Boiler	wall-fired opposed	ESP multiple	None	None	None	Multiple types
124	Landfill	FA,BA	Bit	20094B	Dry Bottom PC Boiler	wall-fired opposed	ESP multiple	None	None	None	Multiple types
126	Impoundment	FA,BA	Subbit	43035	Dry Bottom PC Boiler	wall-fired opposed	ESP hot-side	None	None	None	Combustion-LNB
127	Impoundment	FA,BA	Subbit	43035	Dry Bottom PC Boiler	wall-fired opposed	ESP hot-side	None	None	None	Combustion-LNB
128	Landfill	FGD,FA	Lig	43034	Wet Bottom PC Boiler	wall-fired	ESP cold-side	Wet-inhib	Limestone	None	Multiple types
ES-1	Landfill	FGD,FA	Subbit	31192	Dry Bottom PC Boiler	tangential	Fabric filter	Wet-natural	Limestone	None	Other

Table 3-2
Leachate Sample Attributes (continued)

Sample ID	Source	Byproduct	Source Fuel Type	Site	Source Plant PC Boiler Type	PC Boiler Firing	Source Plant Particulate Collection	Source Plant SO2 Control	Source Plant SO2 Sorbent	Source Plant Flue Gas Cond.	Source Plant NOx Control
HN-1	Impoundment	FA,BA	Bit	13115B	Dry Bottom PC Boiler	tangential	ESP cold-side	None	None	Yes	Other
HN-2	Impoundment	FA,BA	Bit	13115B	Dry Bottom PC Boiler	tangential	ESP cold-side	None	None	Yes	Other
SX-1	Impoundment	FA	Blend	25410B	Cyclone		ESP cold-side	None	None	Yes	Combustion-OFA

Notes:

Ash at site 27413 (samples 090, 091, 092) was first sluiced, then managed dry.

QC and duplicate samples not listed

Abbreviations:

Bit = bituminous; Subbit = Subbituminous; Mix = CCP from different units burning different coals; Blend = CCP from a single unit burning two different fuels

PC = pulverized coal; ESP = electrostatic precipitator; OFA = overfired air; LNB = low-NOx burner

FA = fly ash; BA = bottom ash; EA = economizer ash; FGD = flue gas desulfurization sludge;

Table 3-3
Sample Collection Methods

Sample ID	Site	Source	Byproduct	Point	Method
001	50210	Landfill	FA,BA	Leachate Well	Waterra Pump to Peristaltic
002	50213	Landfill	FA	Lysimeter	Bladder Pump
003	50213	Landfill	FA	Lysimeter	Bladder Pump
004	50183	Landfill	FA,BA	Leachate Collection System	Peristaltic Pump
005	50183	Landfill	FA,BA	Leachate Well	Waterra Pump to Peristaltic
006	23223A	Landfill	SDA	Leachate Collection System	Peristaltic Pump
007	23223B	Impoundment	FGD	Leachate Well	Bladder Pump
008	23223B	Impoundment	FGD	Leachate Well	Bladder Pump
009	23223B	Impoundment	FGD	Ash/Water Interface	Peristaltic Pump
010	23214	Landfill	FA	Leachate Collection System	Bailer to Peristaltic
012	14093	Impoundment	FA	Leachate Well	Waterra Pump to Peristaltic
013	14093	Impoundment	FA	Leachate Well	Peristaltic Pump
014	14093	Impoundment	FA	Leachate Well	Peristaltic Pump
015	25410A	Impoundment	FA,BA	Ash/Water Interface	Peristaltic Pump
016	25410A	Impoundment	FA,BA	Drive Point Piezometer	Peristaltic Pump
017	13115A	Impoundment	FA,BA	Ash/Water Interface	Peristaltic Pump
018	13115B	Impoundment	FA,BA	Leachate Well	Peristaltic Pump
019	13115A	Impoundment	FA	Sluice Line	Dip Sampler to Peristaltic Pump
020	13115A	Impoundment	FA,BA	Outfall	Peristaltic Pump
021	49003A	Impoundment	FA	Drive Point Piezometer	Peristaltic Pump
022	49003A	Impoundment	FA	Ash/Water Interface	Peristaltic Pump
023	49003A	Impoundment	FA	Drive Point Piezometer	Peristaltic Pump
024	49003B	Landfill	FA	Leachate Collection System	Dip Sampler to Peristaltic Pump
025	49003B	Landfill	FA	Leachate Collection System	Dip Sampler to Peristaltic Pump
026	49003A	Impoundment	FA	Outfall	Dip Sampler to Peristaltic Pump
027	35015A	Landfill	FGD, FA	Leachate Collection System	Dip Sampler to Peristaltic Pump
028	35015A	Landfill	FGD, FA	Leachate Collection System	Dip Sampler to Peristaltic Pump
029	35015A	Landfill	FGD, FA	Leachate Collection System	Dip Sampler to Peristaltic Pump
030	35015B	Impoundment	FA	Seep	Dip Sampler to Peristaltic Pump
031	35015B	Impoundment	FA	Drive Point Piezometer	Peristaltic Pump
032	35015B	Impoundment	FA,BA	Outfall	Peristaltic Pump
037	33106	Impoundment	FA	Drive Point Piezometer	Peristaltic Pump
038	33106	Impoundment	FA	T-Handle Probe	Peristaltic Pump
039	33106	Impoundment	FA	Drive Point Piezometer	Peristaltic Pump
042	33106	Impoundment	FA	Sluice Line	Peristaltic Pump
043	33106	Impoundment	FA	Sluice Line	Peristaltic Pump
044	33106	Impoundment	FA	Outfall	Peristaltic Pump
049	33106	Impoundment	FA,BA	Ash/Water Interface	Peristaltic Pump
051	40109	Impoundment	FA	Sluice Line	Peristaltic Pump
052	40109	Impoundment	FA	Drive Point Piezometer	Peristaltic Pump
053	40109	Impoundment	FA	T-Handle Probe	Peristaltic Pump
057	40109	Impoundment	FA,BA	Ash/Water Interface	Peristaltic Pump
059	40109	Impoundment	FA,BA	Outfall	Peristaltic Pump

Table 3-3
Sample Collection Methods (continued)

Sample ID	Site	Source	Byproduct	Point	Method
061	33104	Impoundment	FA	Drive Point Piezometer	Peristaltic Pump
062	33104	Impoundment	FA	Drive Point Piezometer	Peristaltic Pump
064	33104	Impoundment	FA	Sluice Line	Peristaltic Pump
069	33104	Impoundment	FA,BA	Ash/Water Interface	Peristaltic Pump
070	33104	Impoundment	FA,BA	Outfall	Peristaltic Pump
079	22346	Impoundment	FA,OA	Leachate Well	Peristaltic Pump
082	22346	Impoundment	FA,OA	Ash/Water Interface	Peristaltic Pump
083	22347	Impoundment	FA	Ash/Water Interface	Peristaltic Pump
084	22346	Impoundment	FA,OA	Leachate Well	Peristaltic Pump
090	27413	See Notes	FA	Leachate Well	Peristaltic Pump
091	27413	See Notes	FA	Leachate Well	Peristaltic Pump
092	27413	See Notes	FA	Leachate Well	Peristaltic Pump
093	27412	Landfill	FA,BA	Soil Boring	Core Extract
097	50212	Landfill	FA	Leachate Collection System	Peristaltic Pump
098	50183	Landfill	FA,BA	Leachate Collection System	Peristaltic Pump
099	50183	Landfill	FA,BA	Leachate Well	Waterra Pump to Peristaltic
101	50408	Landfill	FA,BA	Leachate Collection System	Peristaltic Pump
102	50211	Landfill	FA	Leachate Collection System	Peristaltic Pump
105	34186B	Impoundment	FGD	Ash/Water Interface	Peristaltic Pump
106	34186C	Landfill	FGD,FA,BA	Leachate Collection System	Dip Sampler to Peristaltic Pump
107	34186B	Impoundment	FGD	Sluice Line	Peristaltic Pump
108	34186A	Landfill	FA	Seep	Peristaltic Pump
111	49003B	Landfill	FA	Leachate Collection System	Dip Sampler to Peristaltic Pump
112	49003B	Landfill	FA	Leachate Collection System	Dip Sampler to Peristaltic Pump
113	49003A	Impoundment	FA	T-Handle Probe	Peristaltic Pump
114	49003A	Impoundment	FA	T-Handle Probe	Peristaltic Pump
115	49003A	Impoundment	FA	Ash/Water Interface	Peristaltic Pump
116	49003A	Impoundment	FA	Outfall	Dip Sampler to Peristaltic Pump
118	35015B	Impoundment	FA,BA	Ash/Water Interface	Peristaltic Pump
119	35015B	Impoundment	FA,BA	Outfall	Peristaltic Pump
120	35015A	Landfill	FGD, FA	Leachate Collection System	Dip Sampler to Peristaltic Pump
121	35015A	Landfill	FGD, FA	Leachate Collection System	Dip Sampler to Peristaltic Pump
122	35015A	Landfill	FGD, FA	Leachate Collection System	Dip Sampler to Peristaltic Pump
123	20094A	Landfill	FA	Soil Boring	Core Extract
124	20094B	Landfill	FA,BA	Soil Boring	Core Extract
126	43035	Impoundment	FA,BA	Seep	Dip Sampler to Peristaltic Pump
127	43035	Impoundment	FA,BA	Seep	Dip Sampler to Peristaltic Pump
128	43034	Landfill	FGD,FA	Leachate Collection System	Peristaltic Pump
ES-1	31192	Landfill	FGD,FA	Soil Boring	Core Extract

Table 3-3
Sample Collection Methods (continued)

Sample ID	Site	Source	Byproduct	Point	Method
HN-1	13115B	Impoundment	FA,BA	Soil Boring	Core Extract
HN-2	13115B	Impoundment	FA,BA	Soil Boring	Core Extract
SX-1	25410B	Impoundment	FA	Soil Boring	Core Extract

Notes:
Ash at site 27413 (samples 090, 091, 092) was first sluiced, then managed dry.
QC and duplicate samples not listed

Abbreviations:
FA = fly ash; BA = bottom ash; EA = economizer ash; FGD = flue gas desulfurization sludge; OA = oil ash

4

LEACHATE QUALITY AT CCP MANAGEMENT FACILITIES

Analytical data were entered in a database and reviewed for outliers; anomalous values were checked and corrected, if appropriate, by the Trent University laboratory. Data are summarized in this section; all results are listed in Appendix A.

Many of the data summaries that follow are based on box-whisker plots, which graphically show the distribution of concentrations for a given group of data (Figure 4-1). Non-detect values were plotted at their detection limit.

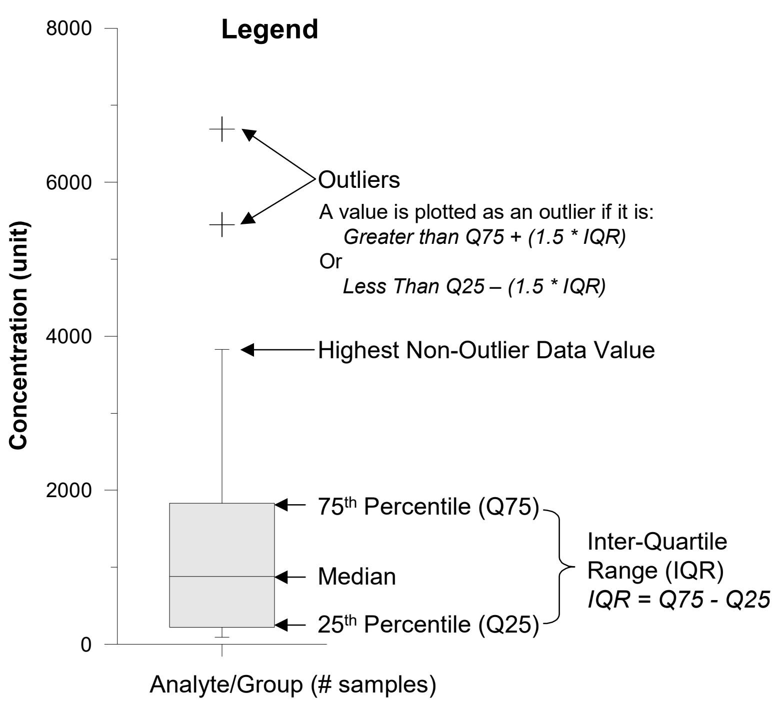


Figure 4-1
Legend for Box-Whisker Plots

Major Constituents

Ash Leachate

The collected leachate samples were generally moderately to strongly oxidizing (positive Eh compared to the standard hydrogen electrode) and moderately to strongly alkaline (Figure 4-2). The subbituminous/lignite ash samples had a slightly higher median pH than bituminous ash, and the highest pH values were from sites receiving subbituminous/lignite ash. The lowest Eh and lowest pH samples were from impoundments.

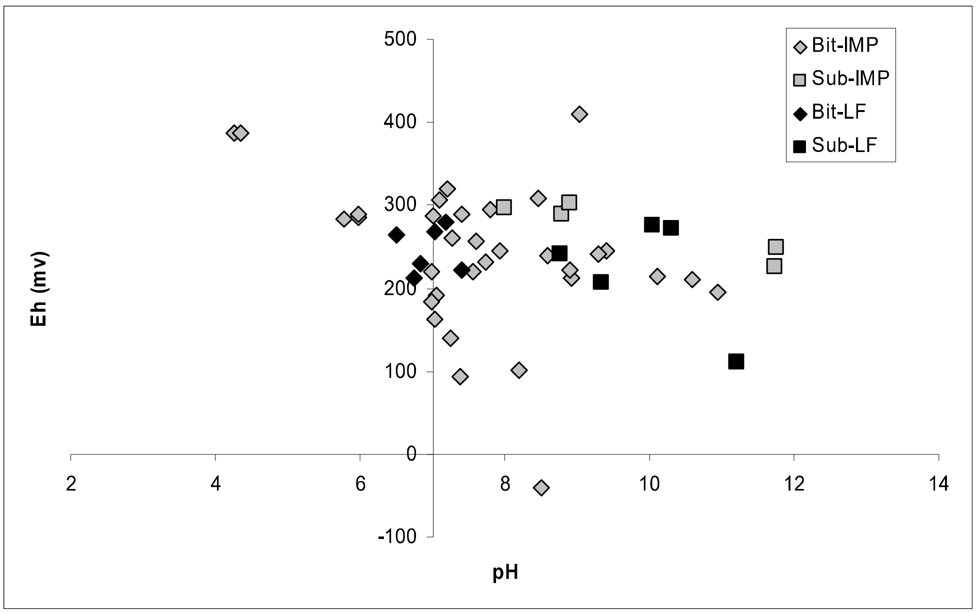


Figure 4-2
Eh-pH Diagram for Ash Samples

Sulfate was the only constituent in the ash leachate samples with a median concentration greater than 100 mg/L (339 mg/L; Figure 4-3, Table 4-1). Most samples had concentrations greater than 100 mg/L, and more than 25 percent of the samples had concentrations greater than 1,000 mg/L. The highest concentration for any constituent in ash leachate was for sulfate in sample 002 (6,690 mg/L; Table 4-1), a leachate sample collected from a landfill receiving subbituminous coal ash.

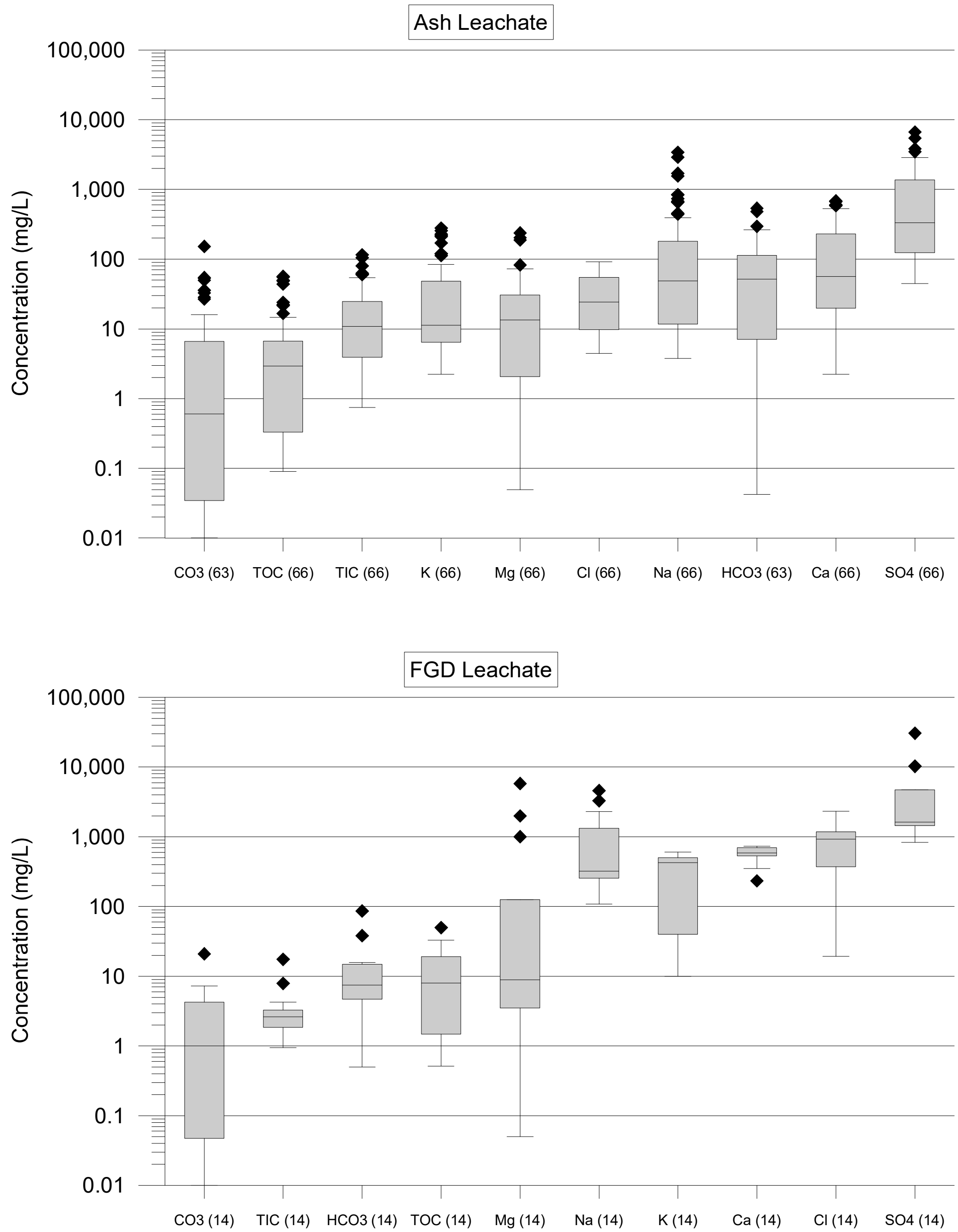


Figure 4-3
Ranges for Major Constituents in CCP Leachate

Table 4-1
Summary Statistics of CCP Leachate Analytical Results

	Ash Leachate Samples					FGD Leachate Samples				
	Count	Min	Median	Max	% BDL	Count	Min	Median	Max	% BDL
Ag (ug/L)	67	<0.2	<0.2	2.0	93%	14	<0.20	<0.20	<0.20	100%
Al (ug/L)	67	<2.0	114	44,400	16%	14	<24	179	890	14%
As (ug/L)	67	1.4	25	1,380	0%	14	11	28	230	0%
As(III)	67	<0.04	0.37	859	40%	14	<0.3	2.1	197	21%
As(V)	67	<0.08	18	534	8%	14	<0.5	5.4	63	21%
B (ug/L)	67	207	2,160	112,000	0%	14	1,450	9,605	98,500	0%
Ba (ug/L)	67	<18	108	657	4%	14	<30	73	158	7%
Be (ug/L)	67	<0.2	<0.4	8.6	94%	14	<0.20	<0.80	1.5	93%
Ca (mg/L)	66	<2.2	55	681	2%	14	234	589	730	0%
Cd (ug/L)	67	<0.2	1.5	65	12%	14	0.50	1.8	13	0%
Cl (mg/L)	66	4.5	25	92	0%	14	19	921	2,330	0%
Co (ug/L)	67	<0.04	1.0	133	31%	14	<0.028	1.0	78	36%
CO_3 (mg/L)	63	<0.01	0.60	152	13%	14	<0.010	1.0	21	21%
Cr (ug/L)	67	<0.2	0.60	5,100	45%	14	<0.20	<0.50	53	64%
Cr(III)	41	<0.01	0.16	340	34%	4	<0.1	0.082	1.3	50%
Cr(VI)	53	<0.006	0.7	5090	36%	5	<0.02	2.9	47	40%
Cu (ug/L)	67	<0.2	3.0	494	19%	14	<0.26	2.6	44	14%
Fe (ug/L)	67	<3	<50	25,600	52%	14	<4.6	<50	1,200	71%
H_2CO_3 (mg/L)	63	<0.01	<0.01	3.4	87%	14	<0.010	<0.010	0.041	93%
HCO_3 (mg/L)	63	0.042	53	535	0%	14	0.50	7.5	87	0%
Hg (ng/L)	22	0.25	3.8	61	0%	8	0.82	8.3	79	0%
K (mg/L)	66	<2.2	11	277	3%	14	10	425	609	0%
Li (ug/L)	67	<1.0	129	23,600	13%	14	<20	3,055	7,070	14%
Mg (mg/L)	66	<0.05	13	236	8%	14	<0.050	8.9	5,810	14%
Mn (ug/L)	67	<0.1	55	4,170	21%	14	<0.10	113	1,170	14%
Mo (ug/L)	67	<8.2	405	39,600	3%	14	164	341	60,800	0%
Na (mg/L)	66	3.8	52	3,410	0%	14	108	322	4,630	0%
Ni (ug/L)	67	<0.6	5.8	189	13%	14	<2.0	3.4	597	36%
Pb (ug/L)	67	<0.1	<0.20	8.0	73%	14	<0.14	<0.20	3.5	64%
Sb (ug/L)	67	<0.1	2.4	59	3%	14	<0.10	1.00	22	29%
Se (ug/L)	67	0.071	19	1,760	0%	14	1.1	6.2	2,360	0%
Se(IV)	67	<0.1	5.3	217	21%	14	<0.1	<2.0	79	64%
Se(VI)	67	<0.1	1.5	1300	34%	14	<0.3	2.2	1660	21%
Si (ug/L)	67	221	4,645	19,000	0%	14	400	2,480	45,400	0%
SO_4 (mg/L)	66	45	339	6,690	0%	14	836	1,615	30,500	0%
Sr (ug/L)	67	<30	829	12,000	1%	14	1,500	5,230	16,900	0%
TIC (mg/L)	66	0.75	11	115	0%	14	0.95	2.6	18	0%
Tl (ug/L)	67	<0.1	0.36	18	46%	14	<0.10	<0.22	2.9	86%
TOC (mg/L)	66	<0.09	3.3	57	24%	14	0.51	8.0	50	0%
U (ug/L)	67	<0.01	1.2	61	19%	14	<0.010	0.20	16	36%
V (ug/L)	67	<0.42	45	5,020	3%	14	<0.69	4.1	400	21%
Zn (ug/L)	67	<1.5	5.0	289	46%	14	<2.0	<5.0	68	57%
DO (%)	61	0.10	35	165	0%	14	0.20	14	95	0%
ORP (mV)	63	-41	241	411	2%	14	1.5	201	356	0%
pH (SU)	64	4.3	7.9	12	0%	14	6.2	9.0	12	0%
EC (µmho/cm)	64	174	990	12,760	0%	14	2,190	6,461	26,140	0%
Temp (ºC)	64	10	21	36	0%	14	9.9	17	27	0%

Notes:

Dissolved oxygen (DO) is percent saturation

More than 25 percent of the calcium, bicarbonate, and sodium concentrations in ash leachate were greater than 100 mg/L, and several sodium concentrations were greater than 1,000 mg/L, with the highest being 3,410 mg/L in sample 002.

Most of the ash leachate sample anion concentrations were dominated by sulfate (Figure 4-4). All of the exceptions were impoundment samples, three of which were porewater (samples 018, 061, and 084) while the other seven samples were pond, sluice, or outfall water. All except one of the exceptions had relatively low sulfate concentrations (two less than 200 mg/L and seven less than 100 mg/L), while sample 018 had a close to median sulfate concentration (339 mg/L) and a relatively high bicarbonate concentration (535 mg/L). All of the exceptions tended toward carbonate/bicarbonate type.

Cation concentrations in the leachate samples were usually dominated by calcium or calcium with varying percentages of sodium and magnesium when the source coal was bituminous, and by sodium when the source coal was subbituminous/lignite. Samples 017, 019, and 020 were exceptions to this relationship, having roughly equal percentages of the cations. The sodium-dominated subbituminous/lignite samples were collected from landfills, while samples 017, 019, and 020 were collected from an impoundment that receives more bottom ash than fly ash.

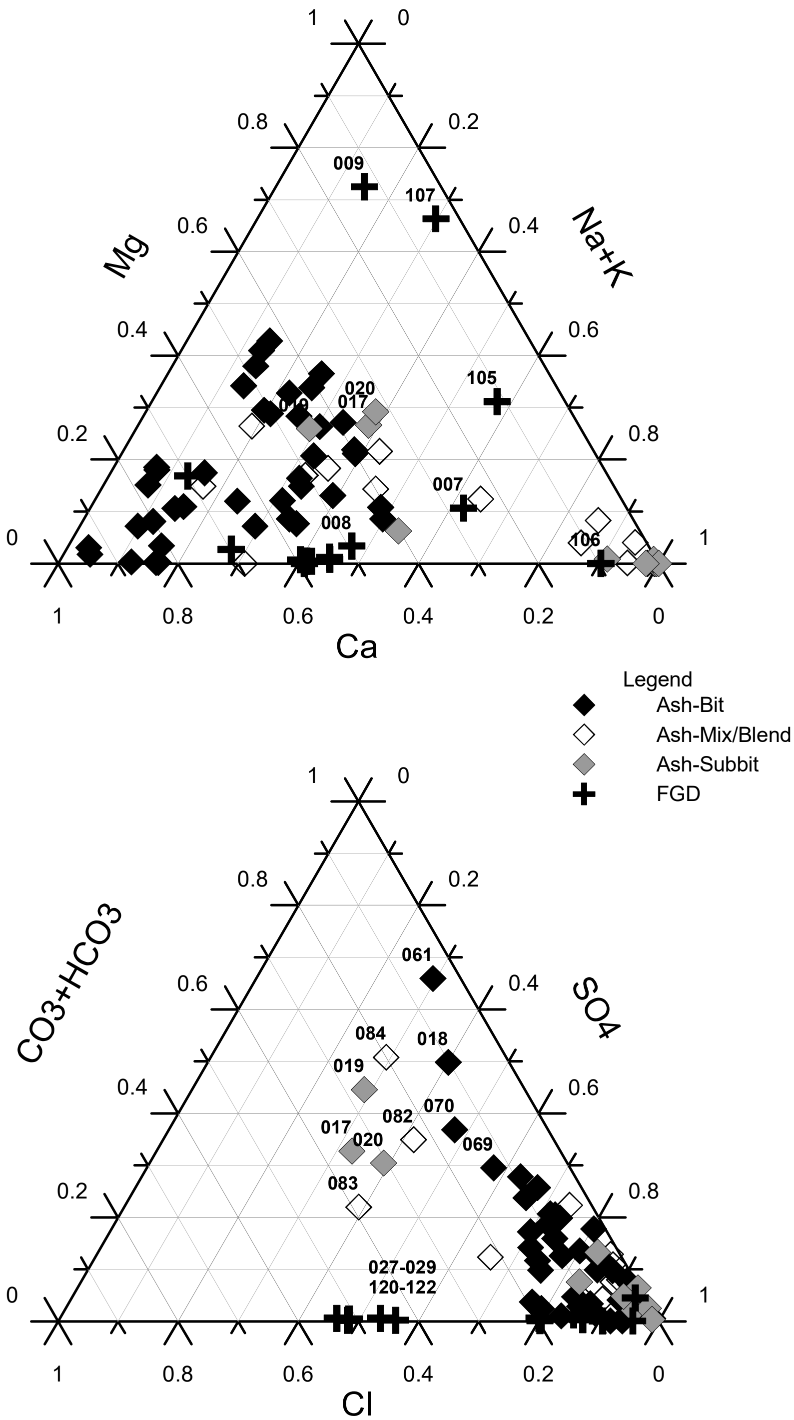


Figure 4-4
Ternary Plots Showing Relative Percentages of Major Constituents in Ash Leachate

FGD Leachate

Leachate samples collected from FGD product management sites (FGD leachates) were moderately to strongly oxidizing (positive Eh compared to the standard hydrogen electrode) and moderately to strongly alkaline (Figure 4-5). Landfill samples, as a group, were less oxic and more alkaline than impoundment samples, although the lowest Eh value was for an impoundment.

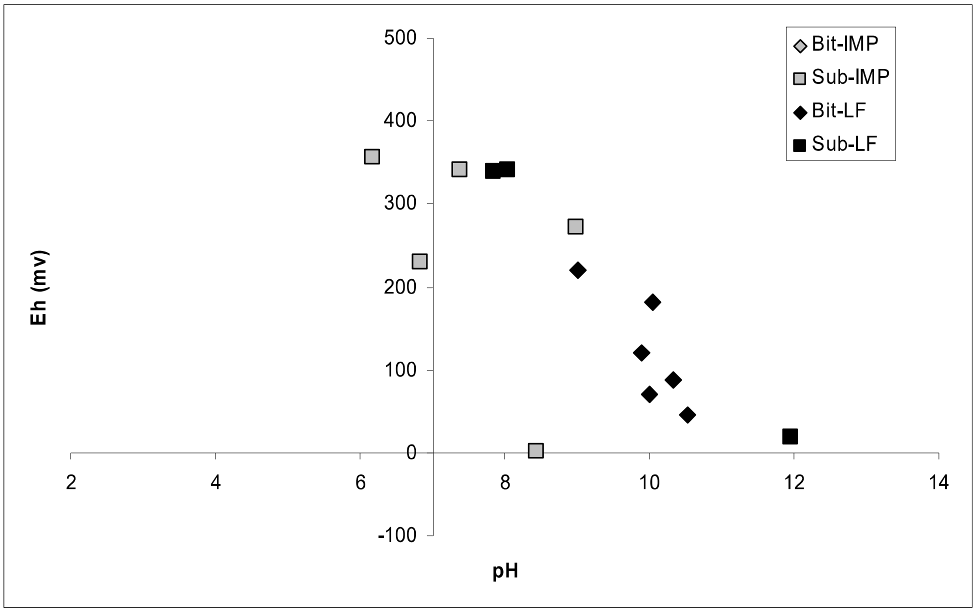


Figure 4-5
Eh-pH Diagram for FGD Leachate Samples

Concentrations of most major constituents (specifically, calcium, chloride, potassium, sodium, and sulfate) in FGD leachate were higher than in ash leachate (Figure 4-3). The median sulfate concentrations was 1,615 mg/L, and the maximum sulfate concentration was 30,500 mg/L, which was the highest single analytical result returned from the field leachate sampling. The high sulfate concentration was obtained from an impoundment where sluice water is recirculated.[2]

More than 25 percent of the chloride and sodium concentrations were greater than 1,000 mg/L, and median concentrations of chloride, calcium, potassium, and sodium were greater than 100 mg/L. Overall, the FGD leachate samples have higher concentrations of chloride and potassium, relative to the other major constituents, than ash leachate.

[2] Two of the 14 FGD leachate samples were from impoundments where sluice water is recirculated; however, the median concentrations from FGD sites without recirculation are also significantly higher than the ash leachate medians.

All of the FGD leachate samples from plants burning subbituminous/lignite coal were dominated by sulfate (Figure 4-4), while the six samples (027-029, 120-122) from a plant that burned bituminous coal had equal percentages of sulfate and chloride—sulfate concentrations were relatively low in these samples.[3] This plant (35015A) has a wet FGD system that uses magnesium-lime as sorbent, similar to some of the other FGD systems from which leachate samples were collected (Table 3-1).

Cation ratios in FGD leachate samples varied considerably, even among samples collected from the same site, largely due to varying magnesium concentrations. For example, samples 007, 008, and 009, all from the 23223B site, ranged from calcium-sodium to magnesium-sodium, primarily based on a variation in magnesium concentrations. Samples 105 and 107, both from the 34186B site, exhibited a similar range in cation ratios, which was also based on varying magnesium concentrations. However, there was no clear relationship between FGD sorbent, coal type, and cation chemistry in the FGD leachate samples.

Minor and Trace Elements

Box-whisker plots of minor and trace elements in ash and FGD leachate are sorted by median concentration, from highest concentration on the right to lowest concentration on the left.

Ash Leachate

Silica and boron had median concentrations higher than 1,000 µg/L in the ash leachate field samples (Figure 4-6). Median concentrations of strontium, molybdenum, lithium, aluminum, and barium were greater than 100 µg/L (Figure 4-6), while median concentrations of chromium, beryllium, thallium, silver, lead, and mercury were lower than 1 µg/L (Figure 4-7). Silver, beryllium, and lead were rarely detected (26 percent of the samples or less).

[3] Due to the low number of samples, the FGD leachate results were not differentiated by source coal in Figure 4-4.

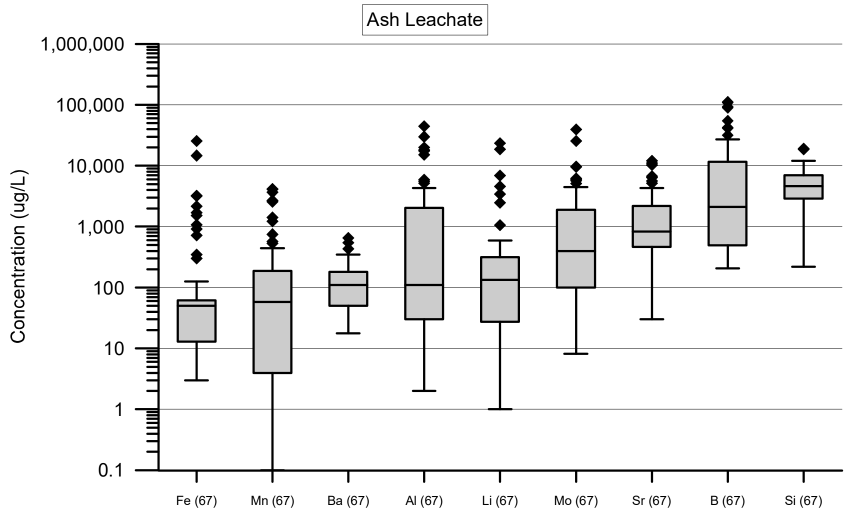


Figure 4-6
Ranges of Minor Constituents in Ash Leachate

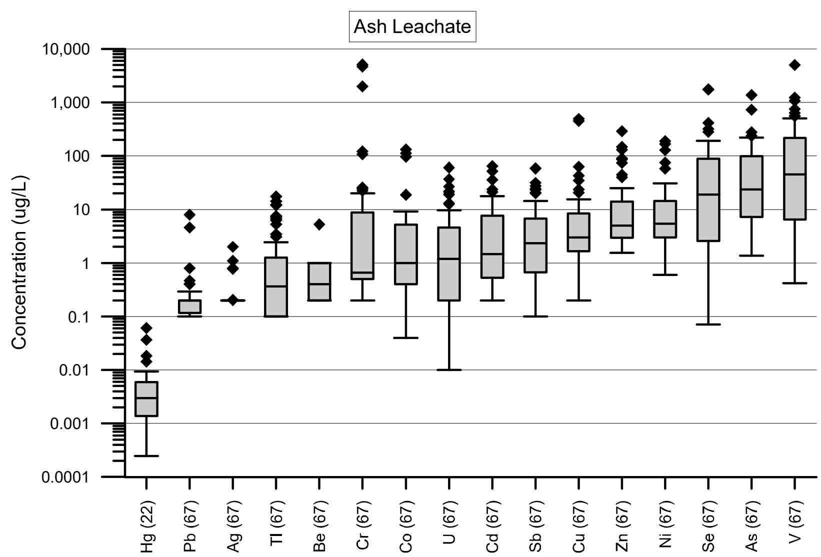


Figure 4-7
Ranges of Trace Constituents in Ash Leachate

FGD Leachate

Boron, strontium, lithium, and silica had median concentrations greater than 1,000 µg/L in the FGD field leachate samples (Figure 4-8). Median concentrations of molybdenum, aluminum, and manganese were greater than 100 µg/L (Figure 4-8), while median concentrations of chromium, beryllium, thallium, silver, lead, and mercury were lower than 1 µg/L (Figure 4-9).

Silver was not detected in the 14 FGD leachate samples, and beryllium, chromium, iron, lead, and thallium, were detected in less than 40 percent of the samples (Table 4-1).

The relative concentrations of minor and trace elements in FGD leachate were somewhat different than in ash leachate. Median concentrations of boron, strontium, and lithium in FGD leachate were a factor of 3 or more higher than in ash leachate, while concentrations of selenium and vanadium were a factor of 3 or more higher in ash leachate than in FGD leachate (Figure 4-10). Median concentrations of uranium and thallium were also a factor of 3 or more higher in the ash leachate, but the concentrations were very low (1 µg/L or less) in both leachates.

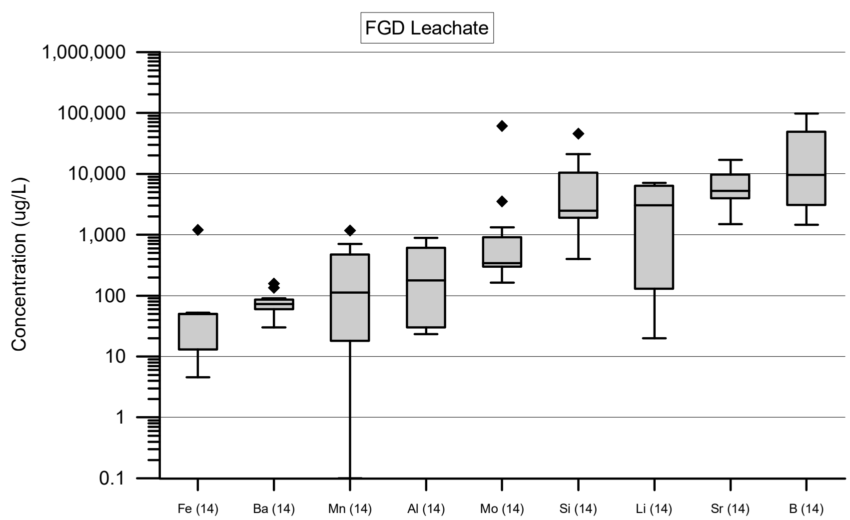


Figure 4-8
Ranges of Minor Constituents in FGD Leachate

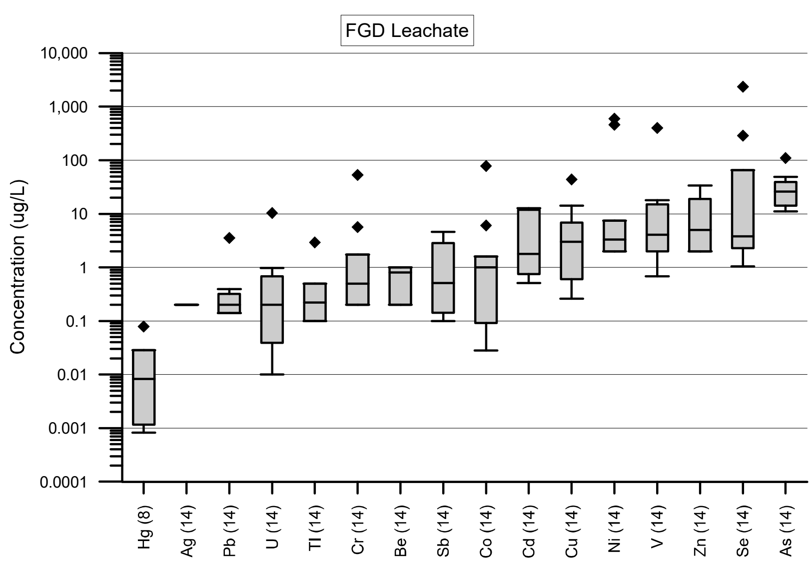


Figure 4-9
Ranges of Trace Constituents in FGD Leachate

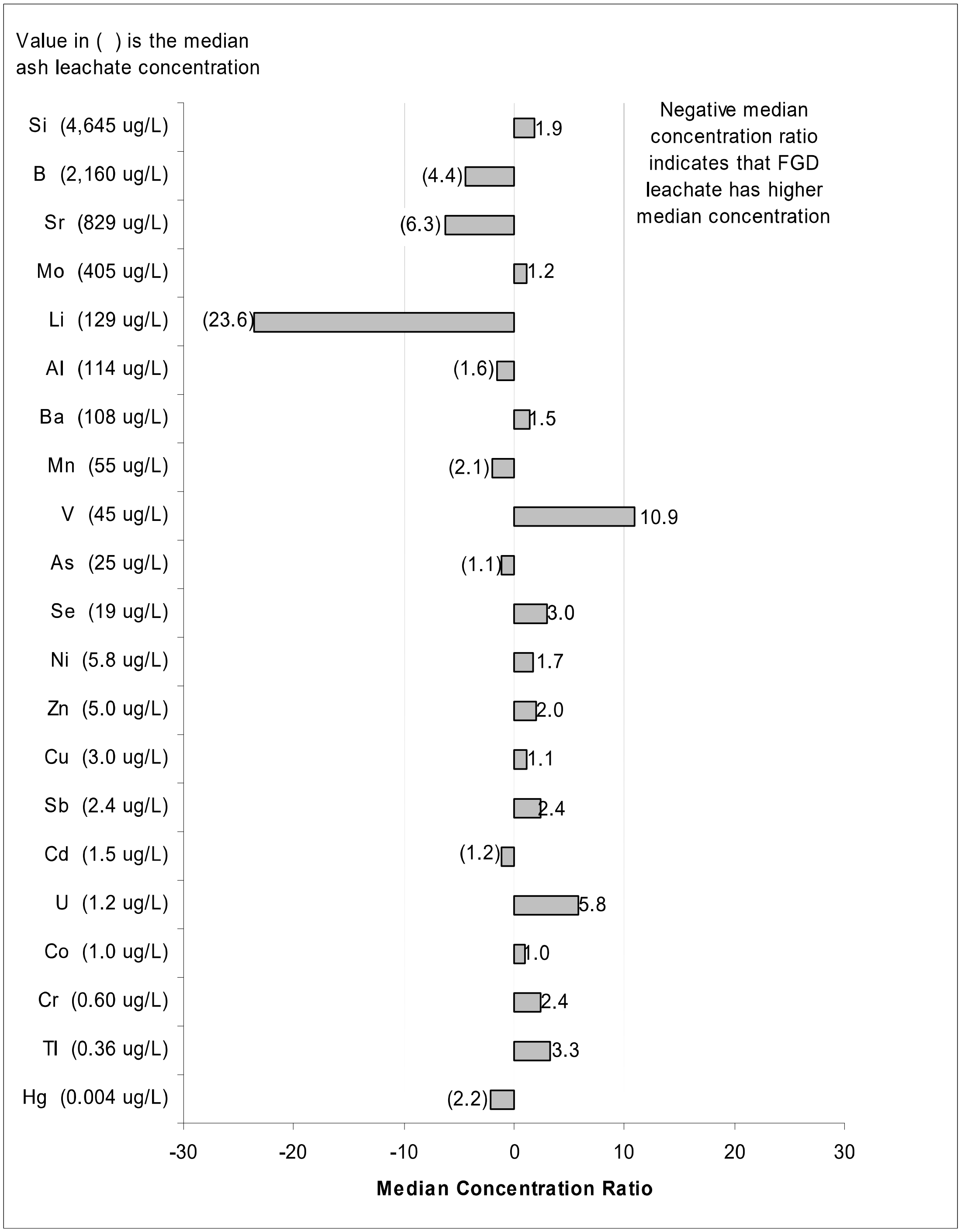


Figure 4-10
Comparison of Median Concentrations of Minor and Trace Elements in Ash and FGD Leachate

Comparison of Ash Leachate Concentrations to Site and Plant Attributes

Leachate concentrations were compared as a function of source coal type and management method in order to evaluate the differences in leachate quality. Samples from multiple sites are required for such a comparison to be meaningful. As a result, this comparison focused on ash samples because five or more samples from two or more sites were available for each comparison (Table 4-2). Summary statistics listing the count, minimum, median, and maximum concentration of each analyte by management type (landfill, impoundment), and source coal (bituminous, subbituminous/lignite) are listed in Table 4-3 for ash leachate and Table 4-4 for FGD leachate.

Table 4-2
Sample (A) and Site (B) Categories

A. Sample Count		**Source Coal**					
		Bit	**Blend**	**Lig**	**Mix**	**Subbit**	**total**
Ash	Impoundment	36	7	0	0	5	48
	Landfill	6	0	1	5	4	16
	Other	0	0	0	3	0	3
	total	42	7	1	8	9	67
FGD	Impoundment	0	0	2	0	3	5
	Landfill	6	0	2	0	1	9
	total	6	0	4	0	4	14
	All	**48**	**7**	**5**	**8**	**13**	**81**

B. Site Count		**Source Coal**					
		Bit	**Blend**	**Lig**	**Mix**	**Subbit**	**total**
Ash	Impoundment	7	4	0	0	2	13
	Landfill	3	0	1	2	3	9
	Other	0	0	0	1	0	1
	total	10	4	1	3	5	23
FGD	Impoundment	0	0	1	0	1	2
	Landfill	1	0	2	0	1	4
	total	1	0	3	0	2	6
	All	**11**	**4**	**4**	**3**	**7**	**29**

Table 4-3
Statistical Summary of Ash Leachate Samples by Management Method and Coal Type

	Landfill				Landfill				Impoundment				Impoundment			
	Bituminous				Subbituminous/Lignite				Bituminous				Subbituminous/Lignite			
	Count	min	med	max	Count	min	med	max	Count	min	med	max	Count	min	med	max
Ag (ug/L)	6	<0.2	<0.2	<0.2	5	<0.2	<0.2	0.78	36	<0.2	<0.2	2.0	5	<0.2	<0.2	<0.2
Al (ug/L)	6	<2	7.5	52	5	81	2,680	17,500	36	<5.9	62	15,100	5	730	4,190	5,920
As (ug/L)	6	1.4	6.2	11	5	4.1	45	84	36	5.1	58	1,380	5	4.1	5.1	6.4
B (ug/L)	6	11,100	23,050	89,500	5	6,080	18,400	41,500	36	207	1,085	112,000	5	470	860	3,890
Ba (ug/L)	6	23	45	50	5	<18	18	63	36	<30	141	545	5	36	140	350
Be (ug/L)	6	<0.2	<0.2	<0.8	5	<0.2	<1	<1	36	<0.2	<0.4	8.6	5	<0.2	<1	<1
Ca (mg/L)	5	235	405	431	5	6.3	19	596	36	12	51	681	5	<2.5	43	81
Cd (ug/L)	6	4.6	10	36	5	7.6	11	52	36	<0.2	1.2	21	5	<0.3	<0.3	2.1
Cl (mg/L)	5	15	29	73	5	11	28	92	36	4.5	15	87	5	31	72	85
Co (ug/L)	6	0.072	9.1	113	5	<0.42	3.3	133	36	<0.2	1.5	22	5	<0.04	<1	1.1
CO_3 (mg/L)	5	0.025	0.11	0.18	5	2.5	50	152	34	<0.01	0.13	16	5	1.1	4.4	36
Cr (ug/L)	6	<0.2	0.17	20	5	0.48	2,000	5,100	36	<0.2	<0.5	29	5	0.66	2.8	108
Cu (ug/L)	6	<0.91	1.1	2.8	5	1.6	43	494	36	<0.38	1.9	452	5	2.4	7.1	12
Fe (ug/L)	6	<8	34	90	5	<3.0	<50	46	36	<5	10	14,700	5	<25	<50	<50
H_2CO_3 (mg/L)	5	<0.01	<0.01	0.020	5	<0.01	<0.01	<0.01	34	<0.01	<0.01	3.4	5	<0.01	<0.01	<0.01
HCO_3 (mg/L)	5	100	229	265	5	1.0	108	481	34	0.042	28	535	5	1.1	110	241
Hg (ng/L)	2	2.1	3.0	3.8	3	14	18	37	7	0.38	1.4	5.2	2	5.4	7.4	9.4
K (mg/L)	5	23	170	219	5	73	80	120	36	<2.2	9.2	277	5	5.5	7.7	40
Li (ug/L)	6	431	5,740	23,600	5	<4.4	<20	27	36	30	213	1,060	5	<7.0	<20	16
Mg (mg/L)	5	69	188	236	5	0.53	6.7	57	36	0.080	6.8	72	5	<0.05	21	28
Mn (ug/L)	6	72	2,060	4,110	5	<1.5	1.5	7.7	36	<0.2	72	4,170	5	<0.2	<4	14
Mo (ug/L)	6	751	3,280	9,630	5	2,680	5,720	25,400	36	8.2	214	6,030	5	<30	80	524
Na (mg/L)	5	80	188	455	5	840	1,700	3,410	36	3.8	19	72	5	53	56	653
Ni (ug/L)	6	3.0	18	189	5	2.2	8.0	75	36	<0.6	7.1	72	5	<0.6	3.7	7.1
Pb (ug/L)	6	<0.12	<0.14	0.12	5	<0.2	0.29	0.29	36	<0.1	<0.15	8.0	5	<0.14	<0.2	0.21
Sb (ug/L)	6	0.14	2.5	9.1	5	0.67	0.90	5.2	36	0.29	6.1	59	5	0.24	0.48	0.62
Se (ug/L)	6	0.67	49	91	5	6.6	413	1,760	36	0.071	13	283	5	1.8	2.5	181
Si (ug/L)	6	2,300	6,075	9,400	5	221	1,540	9,900	36	700	4,715	18,500	5	2,200	3,400	10,300

Table 4-3
Statistical Summary of Ash Leachate Samples by Management Method and Coal Type (continued)

	Landfill				Landfill				Impoundment				Impoundment			
	Bituminous				Subbituminous/Lignite				Bituminous				Subbituminous/Lignite			
	Count	min	med	max	Count	min	med	max	Count	min	med	max	Count	min	med	max
SO_4 (mg/L)	5	845	2,350	2,440	5	2,870	3,830	6,690	36	45	171	1,830	5	91	131	1,120
Sr (ug/L)	6	1,320	4,600	10,300	5	<30	303	12,000	36	170	671	5,610	5	530	649	1,830
TIC (mg/L)	5	24	55	80	5	1.7	32	105	36	0.75	5.5	115	5	5.9	22	49
Tl (ug/L)	6	<0.1	0.47	5.3	5	<0.1	<0.1	<0.5	36	<0.1	0.68	18	5	<0.1	<0.1	<0.1
TOC (mg/L)	5	1.3	4.1	4.6	5	5.3	49	55	36	<0.09	0.64	22	5	0.40	6.0	7.9
U (ug/L)	6	7.4	19	37	5	0.22	5.7	21	36	<0.1	0.70	61	5	<0.02	1.1	1.2
V (ug/L)	6	<0.83	3.1	44	5	3.6	635	5,020	36	2.6	39	754	5	10	17	236
Zn (ug/L)	6	<2	45	289	5	<2	<5	12	36	<2	8.7	90	5	<2	8.4	11
DO (%)	6	16	53	95	5	0.20	14	87	34	2.9	40	165	5	1.6	4.5	35
ORP (mV)	6	213	247	280	5	111	240	276	33	41	240	409	5	225	289	303
pH (SU)	6	6.5	6.9	7.4	5	8.8	10	11	34	4.3	7.6	11	5	8.0	8.9	12
EC (umho/cm)	6	2,000	3,682	4,915	5	6,174	7,690	12,760	34	174	578	2,980	5	680	990	4,020
Temp (°C)	6	14	15	17	5	11	17	22	34	10	22	32	5	16	30	36

Table 4-4
Statistical Summary of FGD Leachate Samples by Management Method and Coal Type

	Landfill				Landfill				Impoundment*			
	Bituminous				Subbituminous/Lignite				Subbituminous/Lignite			
	Count	min	med	max	Count	min	med	max	Count	min	med	max
Ag (ug/L)	6	<0.2	<0.2	<0.2	3	<0.2	<0.2	<0.2	5	<0.2	<0.2	<1
Al (ug/L)	6	<24	149	229	3	<26	26	608	5	31	610	890
As (ug/L)	6	11	28	49	3	12	14	110	5	17	29	230
B (ug/L)	6	1,450	2,950	3,260	3	7,310	11,900	15,600	5	26,800	50,200	98,500
Ba (ug/L)	6	58	63	80	3	70	86	134	5	<30	75	158
Be (ug/L)	6	<0.2	<0.5	<0.8	3	<0.2	<0.2	<1	5	<0.2	<1	1.5
Ca (mg/L)	6	669	704	730	3	234	351	528	5	524	570	600
Cd (ug/L)	6	0.51	0.83	1.9	3	0.75	3.8	13	5	0.50	6.6	12
Cl (mg/L)	6	911	1,170	1,260	3	19	98	859	5	345	572	2,330
Co (ug/L)	6	<0.028	<0.55	0.093	3	<0.11	0.11	1.6	5	<0.092	6.1	78
CO_3 (mg/L)	6	0.73	2.9	7.3	3	0.047	0.44	21	5	<0.01	<0.01	1.7
Cr (ug/L)	6	<0.2	<0.35	<0.5	3	0.46	0.91	5.7	5	<0.4	<1.7	53
Cu (ug/L)	6	<0.26	0.34	3.5	3	0.60	1.5	3.6	5	0.41	6.9	44
Fe (ug/L)	6	<13	<31.5	<50	3	<4.6	<25	4.6	5	<4.7	4.7	1,200
H_2CO_3 (mg/L)	6	<0.01	<0.01	<0.01	3	<0.01	<0.01	<0.01	5	<0.01	<0.01	0.041
HCO_3 (mg/L)	6	3.4	5.9	16	3	0.50	15	87	5	4.9	7.9	38
Hg (ng/L)	3	1.2	12	21	2	0.82	40	79	3	1.9	4.2	28
K (mg/L)	6	470	500	609	3	10	30	350	5	20	80	500
Li (ug/L)	6	5,890	6,415	7,070	3	<20	33	130	5	<20	1,050	3,390
Mg (mg/L)	6	<2.5	4.3	9.6	3	<0.05	8.2	77	5	23	1,000	5,810
Mn (ug/L)	6	16	50	202	3	<0.1	<4	197	5	113	564	1,170
Mo (ug/L)	6	180	316	368	3	310	910	3,520	5	164	570	60,800
Na (mg/L)	6	247	291	341	3	108	141	2,310	5	606	1,330	4,630
Ni (ug/L)	6	<2	<3	3.5	3	<2	4.3	7.5	5	3.3	153	597
Pb (ug/L)	6	<0.14	<0.17	<0.2	3	<0.14	<0.2	0.39	5	<0.2	0.32	3.5
Sb (ug/L)	6	<0.1	<0.22	0.14	3	1.3	2.3	4.7	5	0.72	4.6	22
Se (ug/L)	6	1.1	2.4	3.9	3	17	51	65	5	3.7	159	2,360
Si (ug/L)	6	1,810	1,950	3,000	3	2,600	3,940	21,000	5	400	10,500	45,400

Table 4-4
Statistical Summary of FGD Leachate Samples by Management Method and Coal Type (continued)

	Landfill				Landfill				Impoundment*			
	Bituminous				Subbituminous/Lignite				Subbituminous/Lignite			
	Count	min	med	max	Count	min	med	max	Count	min	med	max
SO_4 (mg/L)	6	1,350	1,510	1,620	3	836	1,450	4,710	5	2,080	10,200	30,500
Sr (ug/L)	6	3,520	4,095	4,500	3	5,960	9,140	9,730	5	1,500	11,700	16,900
TIC (mg/L)	6	0.95	2.5	3.3	3	3.0	4.3	18	5	1.7	2.4	7.9
Tl (ug/L)	6	<0.1	<0.42	0.34	3	<0.1	<0.1	<0.1	5	<0.1	<0.5	2.9
TOC (mg/L)	6	0.51	1.4	2.4	3	7.9	8.1	19	5	9.9	21	50
U (ug/L)	6	<0.022	<0.15	0.097	3	<0.01	0.97	10	5	<0.2	0.68	16
V (ug/L)	6	<0.69	0.98	4.5	3	4.0	6.8	400	5	<1.8	15	103
Zn (ug/L)	6	<2	<3.5	12	3	<2	5.4	19	5	<2	23	68
DO (%)	6	11	23	81	3	0.40	65	95	5	0.20	0.30	36
ORP (mV)	6	46	104	220	3	18	339	341	5	1.5	271	356
pH (SU)	6	9.0	10.0	10.5	3	7.8	8.0	12.0	5	6.2	7.4	9.0
EC (umho/cm)	6	5,550	6,211	6,897	3	2,190	2,870	11,560	5	4,770	12,950	26,140
Temp (°C)	6	12	16	16	3	19	19	21	5	9.9	19	27

* Impoundment category includes two samples from impoundments where water is recirculated

Management in Impoundments Versus Landfills

Concentration ranges for ash leachate in impoundments and landfills are compared in Table 4-5, and selected constituents are graphically illustrated in Figure 4-11 for ash from bituminous coal, and Figure 4-12 for ash from subbituminous/lignite coal. Graphical comparisons for all analyzed constituents are presented in Appendix C, Figures C-1 and C-2.

Table 4-5
Comparison of Ash Leachate Concentrations From Landfills and Impoundments

	Landfill Concentration Higher			Impoundment Concentration Higher	
	Strongly	**Moderately**	**No Difference**	**Moderately**	**Strongly**
Ca (mg/L)	◆		○		
Cl (mg/L)		◆	○		
CO_3 (mg/L)		○	◆		
HCO_3 (mg/L)	◆	○			
K (mg/L)	◆○				
Mg (mg/L)	◆		○		
Na (mg/L)	◆○				
SO_4 (mg/L)	◆○				
Ag (ug/L)			◆○		
Al (ug/L)			○		◆
As (ug/L)	○				◆
B (ug/L)	◆○				
Ba (ug/L)					○◆
Be (ug/L)			◆○		
Cd (ug/L)	◆○				
Co (ug/L)	◆○				
Cr (ug/L)	○		◆		
Cu (ug/L)	○			◆	
Fe (ug/L)			○	◆	
Hg (ng/L)	◆○				
Li (ug/L)	◆		○		
Mn (ug/L)	◆		○		
Mo (ug/L)	◆○				
Ni (ug/L)		◆○			
Pb (ug/L)		○		◆	
Sb (ug/L)	○			◆	
Se (ug/L)	○	◆			
Si (ug/L)			◆	○	
Sr (ug/L)	◆		○		
Tl (ug/L)			○	◆	
U (ug/L)	◆	○			
V (ug/L)	○				◆
Zn (ug/L)		◆	○		

Notes:

◆ = bituminous source coal ○ = subbituminous/lignite source coal

Strongly indicates that interquartile range of one dataset is higher than the other dataset, or median is one order of magnitude higher in one dataset

Moderately indicates that a portion of the interquartile range, and the median, of one dataset is higher than the other dataset.

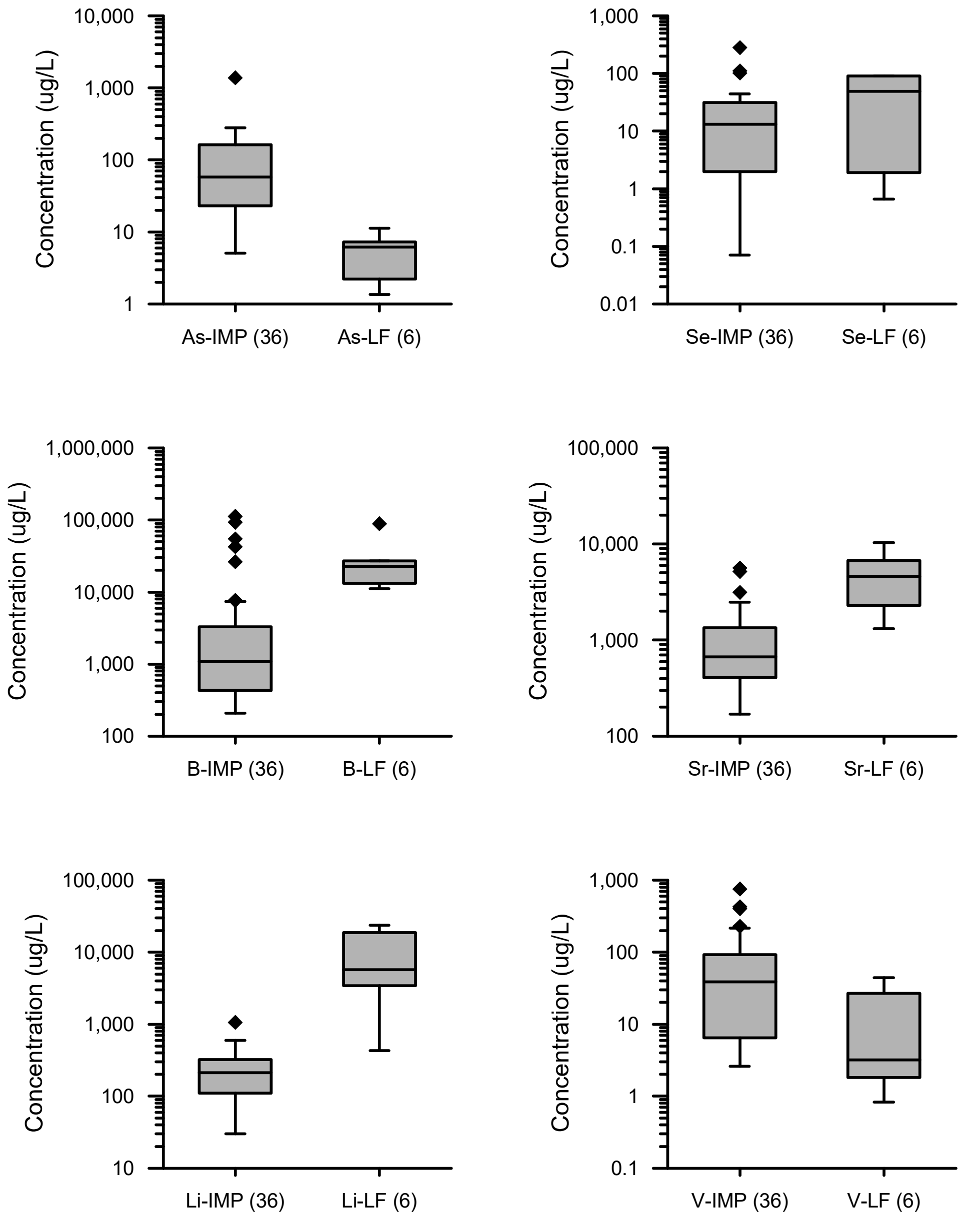


Figure 4-11
Comparison of Field Leachate Concentrations for Selected Constituents: Bituminous Coal Ash, Landfill versus Impoundment (See Appendix C for other parameters)

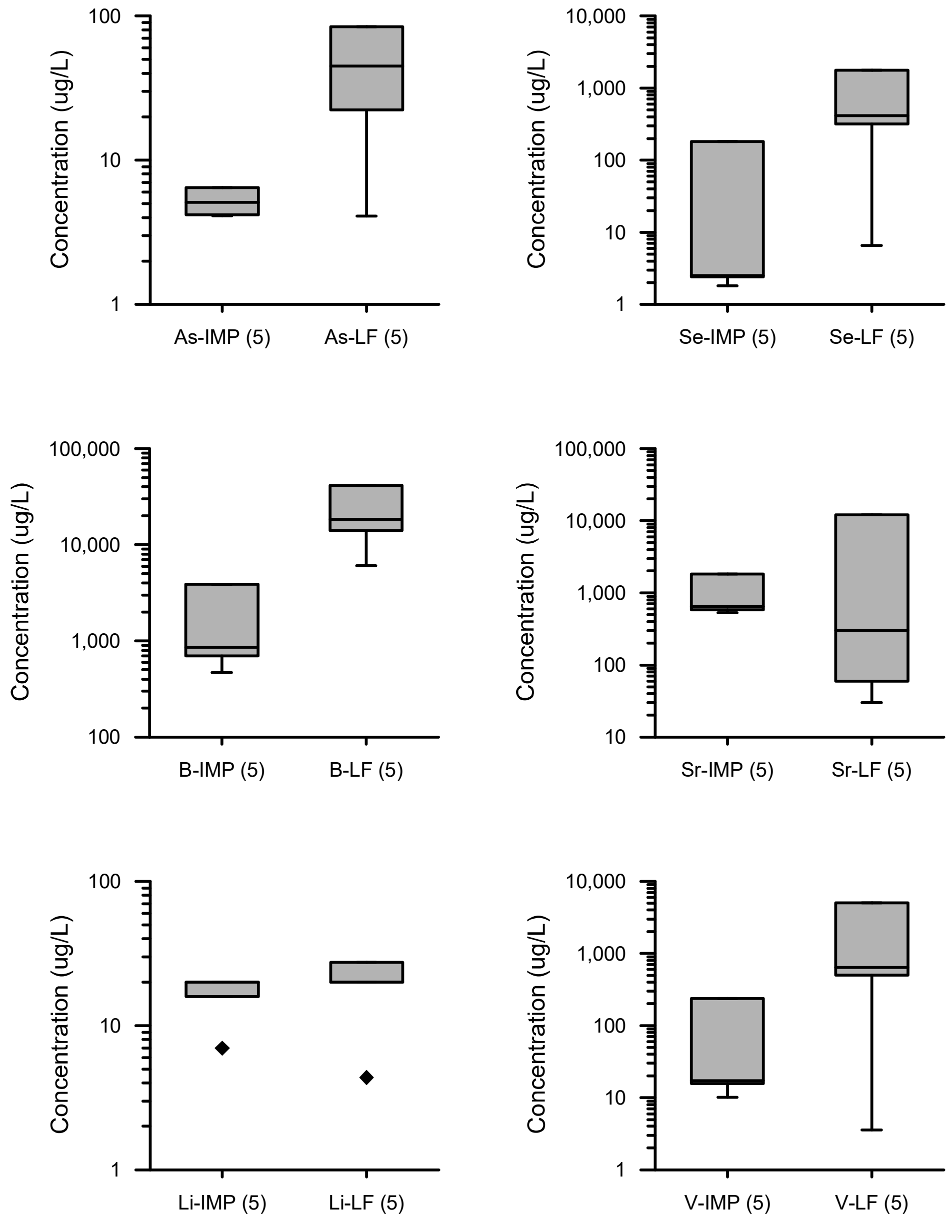


Figure 4-12
Comparison of Field Leachate Concentrations for Selected Constituents: Subbituminous/ Lignite Coal Ash, Landfill versus Impoundment (See Appendix C for other parameters)

Most constituents (22 out of the 34 analyzed) had higher concentration in the landfill leachate samples than in the impoundment leachate samples. The most significant factor contributing to this result is that the leachate in impoundments has a higher water to solid ratio than leachate in landfills, and is, in essence, more dilute. The pond water is more dilute due to the volume of water required to hydraulically transport ash, and the porewater in impoundments is often more dilute because constituents that are easily leached from the surface of the ash particles are washed off during sluicing.

Bituminous versus Subbituminous and Lignite Source Coal

Concentration ranges for ash leachate in impoundments and landfills are compared in Table 4-6, and selected constituents are graphically illustrated in Figure 4-13 for landfill leachate, and Figure 4-14 for impoundment leachate. All analyzed constituents are graphically illustrated in Appendix C, Figures C-3 and C-4.

The field leachate data demonstrate the dependence of several individual constituents on the source coal type. For major ions, leachate from bituminous coal ash had higher concentrations of calcium in both landfill and impoundment settings, while leachate from subbituminous/lignite coal had higher concentrations of carbonate and sodium in both management settings.

Minor and trace constituents for which concentrations in leachate from bituminous coal ash are higher than in leachate from subbituminous/lignite coal, regardless of management environment, are cobalt, lithium, manganese, nickel, antimony, thallium, and zinc (Table 4-6). The difference for lithium is particularly strong. This non-reactive element had a concentration range of 3,400 to 23,600 μg/L in landfill leachate from bituminous coal versus 5 to 27 μg/L in landfill leachate from subbituminous/lignite coal, and 30-1,060 μg/L (bituminous) versus 7 to 20 μg/L (subbituminous/lignite) in impoundment leachate (Figures 4-13 and 4-14). Manganese had similarly large concentration differences, particularly in the landfill environment. Thallium was only detected in leachate from bituminous coal ash (31 of 42 samples, 74 percent), and was not detected in leachate from subbituminous/lignite coal ash (0 of 10 samples).

Minor and trace constituents for which concentrations in leachate from subbituminous/lignite coal ash were higher than in leachate from bituminous coal, regardless of management environment, are aluminum, chromium, copper, and mercury (Table 4-6). The difference is most notable for aluminum and mercury, where the concentrations are an order of magnitude or more higher for both landfill and impoundment leachate.

Table 4-6
Comparison of Ash Leachate Concentrations for Bituminous and Lignite/Subbituminous Source Coal

	Bituminous Concentration Higher			Lig/Subbit Concentration Higher	
	Strongly	Moderately	No Difference	Moderately	Strongly
Ca (mg/L)	◆	○			
Cl (mg/L)			◆		○
CO_3 (mg/L)					○◆
HCO_3 (mg/L)			◆	○	
K (mg/L)		◆	○		
Mg (mg/L)	◆		○		
Na (mg/L)					○◆
SO_4 (mg/L)			○		◆
Ag (ug/L)			◆○		
Al (ug/L)					○◆
As (ug/L)	○				◆
B (ug/L)			○◆		
Ba (ug/L)			◆○		
Be (ug/L)			◆○		
Cd (ug/L)		○	◆		
Co (ug/L)		◆○			
Cr (ug/L)				○	◆
Cu (ug/L)				○	◆
Fe (ug/L)			◆○		
Hg (ng/L)					◆○
Li (ug/L)	◆○				
Mn (ug/L)	◆○				
Mo (ug/L)		○		◆	
Ni (ug/L)		◆○			
Pb (ug/L)			○	◆	
Sb (ug/L)	○	◆			
Se (ug/L)			○		◆
Si (ug/L)		◆	○		
Sr (ug/L)		◆	○		
Tl (ug/L)	○	◆			
U (ug/L)		◆	○		
V (ug/L)			○		◆
Zn (ug/L)	◆	○			

Notes:

◆ = Landfills ○ = Impoundments

Strongly indicates that interquartile range of one dataset is higher than the other dataset, or median is one order of magnitude higher in one dataset

Moderately indicates that a portion of the interquartile range, and the median, of one dataset is higher than the other dataset.

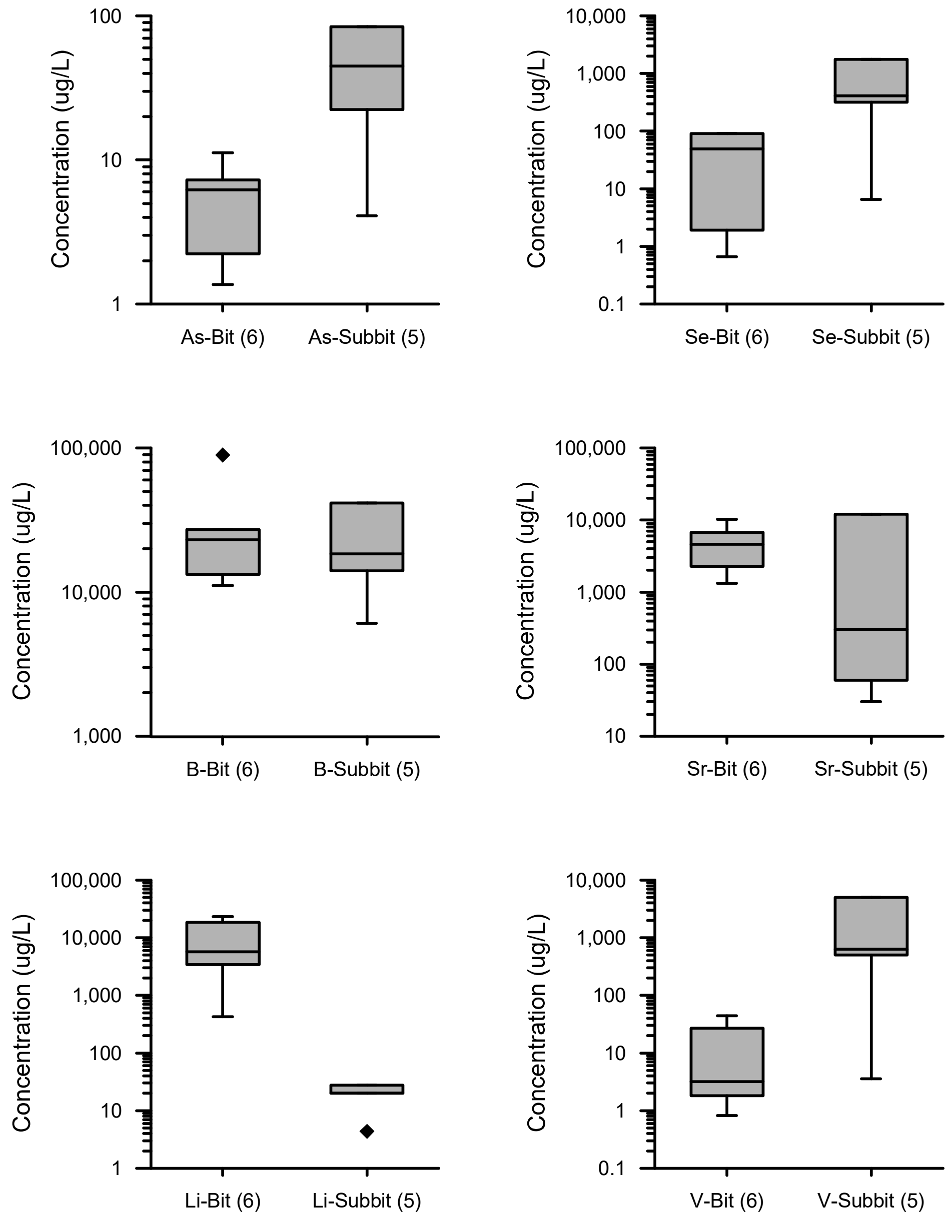


Figure 4-13
Comparison of Field Leachate Concentrations for Selected Constituents: Bituminous vs Subbituminous/Lignite Coal Ash, Landfills (See Appendix C for other parameters)

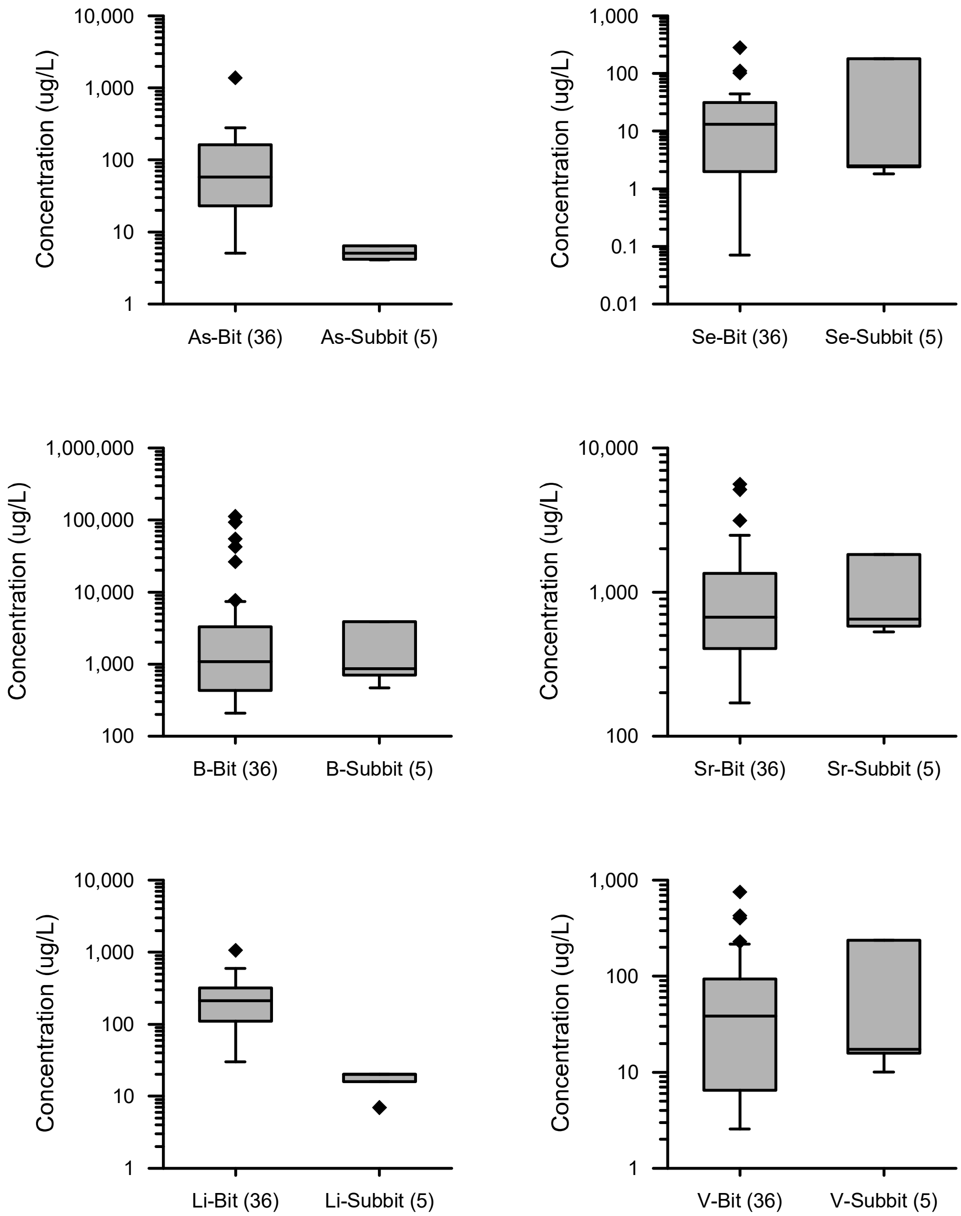


Figure 4-14
Comparison of Field Leachate Concentrations for Selected Constituents: Bituminous vs Subbituminous/Lignite Coal Ash, Impoundments (See Appendix C for other parameters)

Key constituents for which a consistent difference between bituminous and subbituminous/lignite leachate were not found included:

- Arsenic: Concentrations in impoundments were significantly higher when the source coal was subbituminous/lignite, and concentrations in landfills were significantly higher when the source coal was bituminous. Site-specific pH and redox conditions play a significant role in arsenic leaching.
- Boron: The highest boron concentrations (50,000 to 112,000 µg/L) were in leachate from bituminous coal ash, while the highest subbituminous/lignite concentration was 41,000 µg/L. However, there were numerous samples from bituminous ash leachate with considerably lower concentration, and as a result, the medians and interquartile ranges for boron were similar for the two coal types.
- Selenium and Vanadium: Concentrations of these two elements were, for the most part, higher in leachate from subbituminous/lignite coal ash than in leachate from bituminous coal ash. However, there were several relatively high concentrations in bituminous ash impoundments that increased the median sufficiently so that there were no significant differences in the interquartile ranges.
- Strontium and Uranium: For landfill leachate, these elements had significantly higher concentration when the source coal was bituminous than when the source coal was subbituminous/lignite. In impoundment leachate, the bituminous median values were lower than the subbituminous/lignite median values, although the maximum concentrations were significantly higher in the bituminous samples.

Evaluation of Unique Samples

Several samples stand out as unique either due to relatively high concentrations of selected constituents or power plant attributes. Table 4-7 and Table 4-8, respectively, list the maximum concentration of each constituent analyzed in ash and FGD leachate, and whether or not this concentration is significantly higher than the next highest concentration from another site. Table 4-8 excludes samples 106 and 107, which are from an FGD impoundment where concentrations of most constituents are very high because sluice water is recirculated.

For ash leachates, samples from three sites had four to seven constituents with the highest concentration: 50213 (7), 25410A (4), and 49003B (4). 50213 site had the highest concentrations of Co, CO_3, Cr, Cu, Na, Se, and SO_4. The 50213 site is a landfill with pH range from 10.0 to 10.3. The power plant units associated with the 50213 site are dry-bottom PC boilers that have burned subbituminous coal during the active life of the site. Two smaller units have cold-side electrostatic precipitators, while a larger unit utilized a hot-side precipitator for most of the active life of the 50213 site and a fabric filter for the last two years. The larger unit has a low-NOx burner. Leachate was collected in two lysimeters that directly underlie the ash. The leachate at this site was alkaline, with a pH higher than 10. Relatively high ORP values, low iron concentrations, and oxidized forms of arsenic, selenium, and chromium indicate that redox conditions at this site were oxidizing. The only uncommon attributes of this site are the lysimeters used to collect the leachate and the hot-side precipitator. Two other sites received ash from hot-side precipitators (40109 and 43035). These sites did not have similarly high leachate concentrations, however they are both impoundments that receive ash derived from bituminous coal.

Table 4-7
Ash Leachate Samples With Maximum Concentrations

	Count	Max	Sample	Site	Next*	Comment
Ag (ug/L)	67	2.0	HN-1	13115B	1.1	The three highest silver concentrations came from core samples.
Al (ug/L)	67	44,400	016	25410A	30,000	This sample also had relatively high concentrations of B, Cd, K, Mo, Pb, Si, V, and Zn.
As (ug/L)	67	1,380	061	33104	727	No consistent correlations to site/plant attributes.
B (ug/L)	67	112,000	013	14093	109,000	Concentration not significantly higher than other samples.
Ba (ug/L)	67	657	092	27413	545	Concentration not significantly higher than other samples.
Be (ug/L)	67	8.6	043	33106	5.2	Only four beryllium detects; these occurred in four of the five samples with pH lower than 6.0.
Ca (mg/L)	66	681	012	14093	665	Concentration not significantly higher than other samples.
Cd (ug/L)	67	65	016	25410A	52	Two highest concentrations in samples from plants with cyclone boilers, both burn petroleum coke, 25410A also burns used tires.
Cl (mg/L)	66	92	097	50212	87	Concentration not significantly higher than other samples.
Co (ug/L)	67	133	002	50213	113	No consistent correlations to site/plant attributes.
CO_3 (mg/L)	63	152	003	50213	53	No consistent correlations to site/plant attributes.
Cr (ug/L)	67	5,100	002	50213	2,000	May be partially due to erosion of balls (30% Cr) that are used when pulverizing the coal at 50213 plant.
Cu (ug/L)	67	494	002	50213	452	Second lysimeter (003) at this site had a concentration of 62 µg/L.
Fe (ug/L)	67	25,600	079	22346	14,700	No consistent correlations to site/plant attributes.
H_2CO_3 (mg/L)	63	3.4	043	33106	2.8	Highest at sites with low pH.
HCO_3 (mg/L)	63	535	097	50212	535	Concentration not significantly higher than other samples.
Hg (ng/L)	22	61	098	50183	37	Resample concentration at this point was 6 ng/L.
K (mg/L)	66	277	HN-1	13115B	255	Concentration not significantly higher than other samples.
Li (ug/L)	67	23,600	111	49003B	6,940	Two leachate collection system points were sampled twice at this site. For both sample events, one returned high lithium concentration and one returned lower, although still high lithium concentrations. Similar pH, ORP and DO values.
Mg (mg/L)	66	236	111	49003B	188	Concentration not significantly higher than other samples.
Mn (ug/L)	67	4,170	018	13115B	4,110	Concentration not significantly higher than other samples.
Mo (ug/L)	67	39,600	016	25410A	25,400	Two highest concentrations in samples from plants with Cyclone boilers, both burn petroleum coke, 25410A also burns used tires.
Na (mg/L)	66	3,410	002	50213	1,700	Two highest concentrations in samples from this site.
Ni (ug/L)	67	189	111	49003B	128	Two leachate collection system points were sampled twice at this site. For both sample events, one returned high nickel concentration and one returned low nickel concentrations. Similar pH, ORP and DO values.
Pb (ug/L)	67	8.0	051	40109	4.6	Two of three samples with lead higher than 1 µg/L were also the only two samples with pH < 5. Other sample (016) had pH of 11.5.
Sb (ug/L)	67	59	023	49003A	27	Antimony concentrations at this site are unusually high.
Se (ug/L)	67	1,760	003	50213	428	Two highest concentrations in samples from this site.

Table 4-7
Ash Leachate Samples With Maximum Concentrations (continued)

	Count	Max	Sample	Site	Next*	Comment
Si (ug/L)	67	19,000	016	25410A	18,500	Concentration not significantly higher than other samples.
SO_4 (mg/L)	66	6,690	002	50213	3,830	Two highest concentrations in samples from this site
Sr (ug/L)	67	12,000	108	34186A	11,100	Concentration not significantly higher than other samples.
TIC (mg/L)	66	115	18	13115B	105	Concentration not significantly higher than other samples.
Tl (ug/L)	67	18	032	35015B	12	Concentration not significantly higher than other samples.
TOC (mg/L)	66	57	098	50183	55	Concentration not significantly higher than other samples.
U (ug/L)	67	61	023	49003A	37	Several other elements relatively high in this sample.
V (ug/L)	67	5,020	010	23214	1,230	Two highest concentrations in samples from plants with Cyclone boilers, both burn petroleum coke.
Zn (ug/L)	67	289	111	49003B	130	Two leachate collection system points were sampled twice at this site twice. For both sample events, one returned high zinc concentration and one returned low zinc concentrations. Similar pH, ORP and DO values.

* next highest concentration from a different site.

The high chromium concentrations at 50213 were attributed by the utility to high chromium concentration in the flue gas as a result of erosion of the balls used to pulverize the coal. Chromium volatilized in the flue gas may condense on the ash particles and then readily leach from the particles in the landfill environment. High concentrations of other elements may be due to limited dilution. The ash is not saturated at this site; instead, the lysimeters collect porewater that was in tight contact with the ash particles.

The 49003B site is also a landfill and had the highest concentrations of Li, Mg, Ni, and Zn, and a pH range from 6.5 to 7.0. The 49003B source power plant has no unusual attributes, yet concentrations of most elements at one of the two leachate collection system sample points were higher than median concentrations for the whole sample set.

The 25410A site is an impoundment and had the highest concentrations of Al, Cd, Mo, and Si, and a pH of 11.7. The 25410A plant is different from most plants in the study in that it burns a blend of fuels including pet coke and tires in a cyclone boiler. The elevated concentrations at the 25410A site may to be associated with either the cyclone boiler or the fuel mixture, or both.

Table 4-8 lists maximum concentrations in FGD leachate samples. In general, there were too few samples to conclusively correlate high or low concentrations to plant and site attributes.

Table 4-8
FGD Leachate Samples With Maximum Concentrations

	Count	Max.	Sample	Site	Next*	Comment
Ag (ug/L)	12	50183L ND				All values below detection limits,
Al (ug/L)	12	890	008	23223B	608	No consistent correlations to site/plant attributes.
As (ug/L)	12	110	106	34186C	49	High DO (95%), low ORP (18 mV), pH 12.
B (ug/L)	12	98,500	009	23223B	15,600	No consistent correlations to site/plant attributes.
Ba (ug/L)	12	134	106	34186C	90	Concentration not significantly higher than other samples.
Be (ug/L)	12	50183L ND				All values below detection limits,
Ca (mg/L)	12	730	029	35015A	577	Concentration not significantly higher than other samples.
Cd (ug/L)	12	13	106	34186C	12	No consistent correlations to site/plant attributes.
Cl (mg/L)	12	1,260	028	35015A	859	No consistent correlations to site/plant attributes.
Co (ug/L)	12	78	009	23223B	1.6	No consistent correlations to site/plant attributes.
CO_3 (mg/L)	12	21	106	34186C	7.3	High value pH related.
Cr (ug/L)	12	53	009	23223B	5.7	No consistent correlations to site/plant attributes.
Cu (ug/L)	12	44	008	23223B	3.6	No consistent correlations to site/plant attributes.
Fe (ug/L)	12	1,200	007	23223B	4.6	Only sample with pH below 7 (6.2)
H_2CO_3 (mg/L)	12	0.041	007	23223B	<0.01	Only sample with pH below 7 (6.2)
HCO_3 (mg/L)	12	87	006	23223A	16	No consistent correlations to site/plant attributes.
Hg (ng/L)	8	79	128	43034	28	Most oxidized FGD sample collected.
K (mg/L)	12	609	121	35015A	350	No consistent correlations to site/plant attributes.
Li (ug/L)	12	7,070	122	35015A	2,720	No consistent correlations to site/plant attributes.
Mg (mg/L)	12	1,990	009	23223B	77	No consistent correlations to site/plant attributes.
Mn (ug/L)	12	704	007	23223B	202	No consistent correlations to site/plant attributes.
Mo (ug/L)	12	60,800	007	23223B	3,520	No consistent correlations to site/plant attributes.
Na (mg/L)	12	2,310	106	34186C	1,330	No consistent correlations to site/plant attributes.
Ni (ug/L)	12	597	007	23223B	7.5	No consistent correlations to site/plant attributes.
Pb (ug/L)	12	3.5	007	23223B	0.39	Detects only for with lignite/subbituminous ash.
Sb (ug/L)	12	4.7	006	23223A	4.6	No consistent correlations to site/plant attributes.
Se (ug/L)	12	2,360	009	23223B	65	No consistent correlations to site/plant attributes.
Si (ug/L)	12	21,000	106	34186C	12,700	No consistent correlations to site/plant attributes.
SO_4 (mg/L)	12	10,400	009	23223B	4,710	No consistent correlations to site/plant attributes.
Sr (ug/L)	12	16,900	007	23223B	9,730	No consistent correlations to site/plant attributes.
TIC (mg/L)	12	18	006	23223A	4.3	No consistent correlations to site/plant attributes.
Tl (ug/L)	12	2.9	009	23223B	0.34	No consistent correlations to site/plant attributes.
TOC (mg/L)	12	21	007	23223B	19	No consistent correlations to site/plant attributes.
U (ug/L)	12	10	006	23223A	0.97	No consistent correlations to site/plant attributes.
V (ug/L)	12	400	106	34186C	18	No consistent correlations to site/plant attributes.
Zn (ug/L)	12	34	009	23223B	23	No consistent correlations to site/plant attributes.

* next highest concentration from a different site.

Typical plant components in this study included wet-bottom coal-fired PC units, cold-side ESPs, and wet FGD systems. Less common were plants with cyclone boilers, non-coal fuel sources, hot-side ESPs, and dry FGD systems. Results for these less common configurations are discussed below:

- Cyclone Boilers: The power plants associated with 23214, 25410A, and 25410B use cyclone boilers. Cyclone boilers tend to burn hotter than PC boilers, and also burn a wider variety of fuels. These plants are the only ones sampled that burn petroleum coke, and the fuel burned at 25410A and 25410B also includes used tires. Leachate sampled at these sites had higher than median concentrations of most elements, and the highest concentrations of cadmium, molybdenum, and vanadium. Vanadium is often associated with petroleum coke. The relatively high concentrations from these samples may reflect the effect of the cyclone boiler, or the fuel. Concentrations at one of the sample locations from 25410A and 25410B were often higher than at 23214, but not sufficiently so to indicate any effects from the tires on ash leachate composition.
- Hot-Side ESPs: The plants associated with the 40109, 43035, and 50213 sites have hot-side ESP's, while the other plants with ESPs are cold-side. The 40109 and 43035 samples did not stand out in terms of high or low concentration. These sites are impoundments and receive bituminous coal ash. As previously discussed, the 50213 site is a landfill and received subbituminous ash, and had relatively high concentrations of several constituents, including selenium. The high selenium concentration is unusual in that less selenium capture in ash is expected from plants with hot-side ESPs, due to the higher temperatures at the collection point. Presence in the leachate may indicate that the selenium captured in the hot-side is present in a relatively soluble form for the subbituminous coal ash. Similarly, the relatively high concentrations at the 50213 site may indicate increased leachability for the subbituminous ash collected at the hotter temperatures. However, this is only one site and more data from plants burning subbituminous coal with hot-side ESPs are needed to confirm this observation. The relatively low concentrations seen at the 40109 and 43035 sites may suggest that the 50213 data are specific to the particular plant, fuel, or management setting.
- Oil Ash: 22346 is the only site sampled where oil ash was managed with coal ash. The leachate from the ash sampled at this site did not stand out in terms of low or high concentration. Since oil ash is generally high in vanadium and nickel, this result suggests that either the effect of the oil ash is not appreciable due to its volume relative to the coal ash, or that the coal ash geochemically mitigates releases from the oil ash.
- Wet-Bottom PC Boiler: 43034 is the only plant that has a wet-bottom PC boiler. The leachate from the FGD byproduct sampled at this site did not stand out in terms of low or high concentration.
- Dry FGD System: 23223A is associated with the only power plant that used a spray dryer system; all other FGD samples came from power plants with wet FGD systems. With a few exceptions, the leachate from this site tended to have relatively low concentrations. The most notable exception was uranium, which had a concentration of 10 μg/L at this site and less than 1 μg/L at the other FGD sites.

5

SPECIATION OF ARSENIC, SELENIUM, CHROMIUM, AND MERCURY AT CCP MANAGEMENT FACILITIES

The mobility and toxicity of inorganic constituents is sometimes strongly dependent on their aqueous speciation. This is particularly true for arsenic, selenium, and chromium, which can be present at elevated concentrations in CCP leachate. Important species in leachate and groundwater are As(III) and As(V), Se (IV) and Se(VI), and Cr(III) and Cr(VI). Organic species for the other constituents (e.g., methylarsenic acid) were not considered in this study. Generally speaking, As(III) and Cr(VI) are more toxic and more mobile than As(V) and Cr(III); and Se(IV) is more toxic to most terrestrial and aquatic wildlife than the more mobile Se(VI). It is important to know the species present in leachate in order to assess potential impacts associated with these constituents. Although mercury is generally present only at very low concentrations in ash leachate and is very immobile in groundwater, the organic mercury species (monomethyl mercury) can bioaccumulate to toxic levels in the surface water environment and is therefore of interest.

Evaluation of Speciation Sample Preservation Methods

Speciation of arsenic and selenium in field samples with widely varying matrix characteristics such as the CCP leachate is challenging because preservation techniques and analytical interferences can have a significant impact on the results. Several preservation methods (HCl, cryofreezing, EDTA, HNO_3, none) were compared on sample splits from one site, and a comparison of speciation results for 32 split samples from several sites using two preservation methods (HCl and cryofreezing) are presented in Appendix D.

Results varied by sample, and suggested that, regardless of preservation method, a critically important factor was minimizing hold times. Species recovery was poorest for the samples collected in 2003 (samples 001 through 032) due to longer holding times for the frozen samples. Importantly, the split sample data collected during this study indicated that, even when overall species recovery was low, the relative predominance of reduced or oxidized species of arsenic and selenium were similar regardless of preservation method or laboratory used. Speciation results presented in the following sections are for samples that were preserved by cryofreezing in the field with liquid nitrogen.

Arsenic

Overview of Results

Total arsenic was detected at concentrations well above the detection limit in all collected water samples (n = 81 after removing all QA samples)[4], and at least one species was detected in all except two samples. Review of duplicate samples indicated that analytical results were usually reproducible, particularly when concentrations were greater than 1 µg/L (Table 5-1).

Excluding duplicates, 51 of the 81 samples contained detectable concentrations of arsenite, 73 samples had detectable concentrations of arsenate, and 30 samples contained detectable concentrations of arsenic species other than arsenite or arsenate. These other species are either monomethyl arsenate or soluble arsenic-sulfur (As-S) compounds. Both types of other arsenic species are technically As(V) compounds (i.e., they contain arsenic in the +5 oxidation state); although they were not grouped with As(V) because they potentially have different chemical and environmental characteristics.

Monomethyl arsenate is either formed by microbial methylation of inorganic arsenic or used as a biocide. However, contrary to the case of mercury, the methylated (i.e., organic) forms of arsenic are less toxic than the inorganic forms, and are therefore generally not regarded as a source of concern. The soluble As-S compounds are formed by reaction of arsenite and free sulfide in reducing waters, and there are also some studies suggesting that these species are less toxic than arsenite and arsenate. In all except two samples (which had relatively low total arsenic concentration), the other arsenic species constituted the minority of all arsenic present (<20 percent).

The arsenic speciation mass balance (the sum of all individual species determined in a given sample divided by the independently-determined total arsenic concentration) varied strongly, and was not always satisfactory. Less than half (35 of 81 samples) had a recovery greater than 80 percent (Figure 5-1). Reasons for this somewhat disappointing performance likely originate from the complexity of the studied samples. Species recovery for the 2004/2005 samples was better than for the 2003 samples due to reduced holding times and other laboratory refinements (Appendices D and E).

[4] QA samples include blanks and duplicates.

Table 5-1
Arsenic Speciation Data

Site	Sample	Source	CCP	Coal	Total As (ug/L)	As(III) (ug/L)	As(V) (ug/L)	As, other species (ug/L)	Sum of Species	% Recovery	% As(III)	% As(V)	% As (other)
50210	001	LF	FA,BA	Mix	20	<0.3	9.5	2.1	11.6	57%			
50213	002	LF	FA	Subbit	48	<6	47	<6	47.2	**98%**	0.0%	100.0%	0.0%
50213	003	LF	FA	Subbit	84	<6	69	<6	68.8	**82%**	0.0%	100.0%	0.0%
50183	004	LF	FA,BA	Mix	19	8.4	5.2	<0.3	13.5	73%			
50183	005	LF	FA,BA	Mix	3.0	<0.2	1.3	<0.2	1.3	45%			
23223A	006	LF	SDA	Subbit	12	<0.3	0.94	<0.3	0.9	8%			
23223B	007	IMP	FGD	Subbit	20	<2	<2	<2	0.0	0%			
23223B	008	IMP	FGD	Subbit	17	0.75	<0.5	<0.3	0.7	4%			
23223B	009	IMP	FGD	Subbit	29	<6	<10	<6	0.0	0%			
23214	010	LF	FA	Subbit	22	1.5	10	<0.6	11.5	52%			
14093	012	IMP	FA	Bit	238	97	66	<0.6	163.3	69%			
14093	013	IMP	FA	Bit	22	3.7	<0.5	<0.3	3.7	17%			
14093	*013D*	*Dup*	*FA*	*Bit*	*22*	*1.9*	*<0.5*	*<0.3*	*1.9*	*9%*			
14093	014	IMP	FA	Bit	163	1.9	86	0.86	88.6	54%			
25410A	015	IMP	FA,BA	Blend	24	<0.6	24	<0.6	23.6	**99%**	0.0%	100.0%	0.0%
25410A	016	IMP	FA,BA	Blend	69	<0.6	25	<0.6	24.7	36%			
13115A	017	IMP	FA,BA	Subbit	4.1	0.88	<0.08	0.069	1.0	23%			
13115B	018	IMP	FA,BA	Bit	23	0.42	5.2	<0.06	5.6	24%			
13115A	019	IMP	FA	Subbit	5.1	0.57	<0.08	<0.06	0.6	11%			
13115A	020	IMP	FA,BA	Subbit	4.2	1.0	0.53	0.15	1.7	40%			
49003A	021	IMP	FA	Bit	194	2.1	208	<0.3	210.0	**108%**	1.0%	99.0%	0.0%
49003A	022	IMP	FA	Bit	11	13	0.49	<0.06	13.0	**118%**	96.3%	3.7%	0.0%
49003A	023	IMP	FA	Bit	218	0.79	189	<0.3	189.5	**87%**	0.4%	99.6%	0.0%
49003B	024	LF	FA	Bit	11	0.36	<0.2	<0.2	0.4	3%			
49003B	025	LF	FA	Bit	6.5	1.4	<0.08	<0.06	1.4	21%			
49003A	026	IMP	FA	Bit	11	11	0.40	<0.2	11.6	**107%**	96.5%	3.5%	0.0%
35015A	027	LF	FGD, FA	Bit	39	13	4.8	1.3	19.4	49%			
35015A	028	LF	FGD, FA	Bit	30	2.4	1.7	0.20	4.3	14%			
35015A	029	LF	FGD, FA	Bit	49	1.7	8.9	0.35	10.9	22%			

Table 5-1
Arsenic Speciation Data (continued)

Site	Sample	Source	CCP	Coal	Total As (ug/L)	As(III) (ug/L)	As(V) (ug/L)	As, other species (ug/L)	Sum of Species	% Recovery	% As(III)	% As(V)	% As (other)
35015B	030	IMP	FA	Bit	43	3.5	29	0.35	33.4	79%			
35015B	031	IMP	FA	Bit	221	201	24	0.69	225.5	**102%**	89.2%	10.5%	0.3%
35015B	032	IMP	FA,BA	Bit	25	17	17	0.074	34.5	136%	50.8%	49.0%	0.2%
33106	037	IMP	FA	Bit	56	0.30	34		34.3	61%			
33106	038	IMP	FA	Bit	123	2.6	53		56.0	46%			
33106	039	IMP	FA	Bit	42	1.4	53		54.2	128%	2.6%	97.4%	ND
33106	042	IMP	FA	Bit	24	<0.1	19		19.2	**81%**	0.0%	100.0%	ND
33106	043	IMP	FA	Bit	75	<0.05	28		27.6	37%			
33106	044	IMP	FA	Bit	5.1	0.39	2.5		2.9	57%			
33106	*044D*	*Dup*	*FA*	*Bit*	*4.9*	*<0.04*	*2.3*		*2.3*	*48%*			
33106	049	IMP	FA,BA	Bit	5.4	<0.04	2.3	<0.04	2.3	43%			
40109	051	IMP	FA	Bit	38	0.70	15		15.7	41%			
40109	052	IMP	FA	Bit	164	23	7.7		30.5	19%			
40109	053	IMP	FA	Bit	279	108	82	0.70	191.0	68%			
40109	057	IMP	FA,BA	Bit	99	<0.2	93		92.5	**94%**	0.0%	100.0%	ND
40109	059	IMP	FA,BA	Bit	124	<0.2	127		126.6	**102%**	0.0%	100.0%	ND
40109	*059D*	*Dup*	*FA,BA*	*Bit*	*125*	*<0.2*	*119*		*118.5*	**95%**	0.0%	100.0%	ND
33104	061	IMP	FA	Bit	1,380	859	519		1,377.4	**100%**	62.4%	37.6%	ND
33104	062	IMP	FA	Bit	62	<0.2	37		37.5	61%			
33104	064	IMP	FA	Bit	178	<0.4	150		150.2	**84%**	0.0%	100.0%	ND
33104	069	IMP	FA,BA	Bit	100	<0.2	94		93.6	**94%**	0.0%	100.0%	ND
33104	070	IMP	FA,BA	Bit	143	<0.2	136		135.7	**95%**	0.0%	100.0%	ND
33104	*070D*	*Dup*	*FA,BA*	*Bit*	*144*	*<0.2*	*137*	*0.53*	*137.6*	**96%**	0.0%	99.6%	0.4%
22346	079	IMP	FA,OA	Blend	99	9.5	104		113.8	**115%**	8.3%	91.7%	ND
22346	*079D*	*Dup*	*FA,OA*	*Blend*	*97*	*9.9*	*73*		*82.5*	**85%**	12.0%	88.0%	ND
22346	082	IMP	FA,OA	Blend	23	0.21	15		14.7	64%			
22347	083	IMP	FA	Blend	6.2	0.23	2.4		2.6	43%			
22346	084	IMP	FA,OA	Blend	727	71	535		606.0	**83%**	11.8%	88.2%	ND

Table 5-1
Arsenic Speciation Data (continued)

Site	Sample	Source	CCP	Coal	Total As (ug/L)	As(III) (ug/L)	As(V) (ug/L)	As, other species (ug/L)	Sum of Species	% Recovery	% As(III)	% As(V)	% As (other)
27413	090	See Notes	FA	Mix	23	0.28	18	0.67	18.9	**84%**	1.5%	95.0%	3.5%
27413	091	See Notes	FA	Mix	11	<0.05	9.4	0.15	9.6	**89%**	0.0%	98.4%	1.6%
27413	092	See Notes	FA	Mix	3.3	<0.05	0.49	0.10	0.6	18%			
50212	097	LF	FA	Subbit	45	<0.1	36	<0.1	36.3	**81%**	0.0%	100.0%	0.0%
50183	098	LF	FA,BA	Mix	77	0.66	60	0.29	60.5	79%			
50183	099	LF	FA,BA	Mix	4.8	0.10	3.7	0.19	4.0	**84%**	2.6%	92.7%	4.7%
50408	101	LF	FA,BA	Bit	2.2	<0.1	0.23	0.62	0.9	38%			
50211	102	LF	FA	Bit	7.2	<0.05	6.3	<0.05	6.3	**88%**	0.0%	100.0%	0.0%
34186B	105	IMP	FGD	Lig	230	197	50	3.8	250.6	**109%**	78.4%	20.1%	1.5%
34186C	106	LF	FGD,FA,BA	Lig	110	16	63	5.8	84.7	77%			
34186C	*106D*	*Dup*	*FGD,FA,BA*	*Lig*	*112*	*14*	*77*	*5.2*	*96.3*	**86%**	14.3%	80.2%	5.4%
34186B	107	IMP	FGD	Lig	31	0.95	15	<0.2	16.1	52%			
34186A	108	LF	FA	Lig	4.1	0.37	2.3	<0.05	2.7	65%			
49003B	111	LF	FA	Bit	5.9	<0.1	3.4	<0.1	3.4	58%			
49003B	112	LF	FA	Bit	1.4	0.68	0.95	0.20	1.8	133%	37.1%	52.1%	10.8%
49003A	113	IMP	FA	Bit	102	0.75	118	0.17	118.7	**116%**	0.6%	99.2%	0.1%
49003A	114	IMP	FA	Bit	24	<0.1	20	<0.1	20.5	**87%**	0.0%	100.0%	0.0%
49003A	115	IMP	FA	Bit	8.3	3.1	5.3	<0.05	8.3	**100%**	36.7%	63.3%	0.0%
49003A	116	IMP	FA	Bit	8.2	1.0	7.4	0.083	8.5	**103%**	11.9%	87.2%	1.0%
35015B	118	IMP	FA,BA	Bit	41	0.66	45	0.15	46.3	**114%**	1.4%	98.3%	0.3%
35015B	*118D*	*Dup*	*FA,BA*	*Bit*	*40*	*0.18*	*46*	*0.11*	*45.9*	**116%**	0.4%	99.4%	0.2%
35015B	119	IMP	FA,BA	Bit	30	<0.05	31	0.29	30.8	**102%**	0.0%	99.1%	0.9%
35015A	120	LF	FGD, FA	Bit	27	7.2	11	9.3	27.9	**104%**	25.7%	41.0%	33.2%
35015A	121	LF	FGD, FA	Bit	11	1.3	6.0	0.57	7.9	72%			
35015A	122	LF	FGD, FA	Bit	26	7.6	8.3	6.0	21.9	**86%**	34.8%	37.8%	27.4%
43035	126	IMP	FA,BA	Subbit	5.2	<0.1	3.6	<0.1	3.6	69%			
43035	*126D*	*Dup*	*FA,BA*	*Subbit*	*4.9*	*<0.1*	*3.2*	*<0.1*	*3.2*	*66%*			
43035	127	IMP	FA,BA	Subbit	6.4	<0.2	4.0	<0.2	4.0	63%			
43034	128	LF	FGD,FA	Lig	14	10	2.8	0.45	13.3	**94%**	75.4%	21.2%	3.4%

Table 5-1
Arsenic Speciation Data (continued)

Site	Sample	Source	CCP	Coal	Total As (ug/L)	As(III) (ug/L)	As(V) (ug/L)	As, other species (ug/L)	Sum of Species	% Recovery	% As(III)	% As(V)	% As (other)
13115B	HN-1	IMP	FA,BA	Bit	60	<0.1	34	0.23	33.8	57%			
13115B	HN-2	IMP	FA,BA	Bit	21	<0.1	6.9	0.14	7.1	34%			
25410B	SX-1	IMP	FA	Blend	72	0.88	47	<0.1	47.8	66%			

Notes:

Ash at site 27413 (samples 090, 091, 092) was first sluiced, then managed dry.

Abbreviations:

Bit = bituminous; Subbit = Subbituminous; Mix = CCP from different units burning different coals; Blend = CCP from a single unit burning two different fuels

FA = fly ash; BA = bottom ash; EA = economizer ash; FGD = flue gas desulfurization sludge; OA = oil ash

LF = landfill; IMP = impoundment; DUP = duplicate sample

ND = not determined

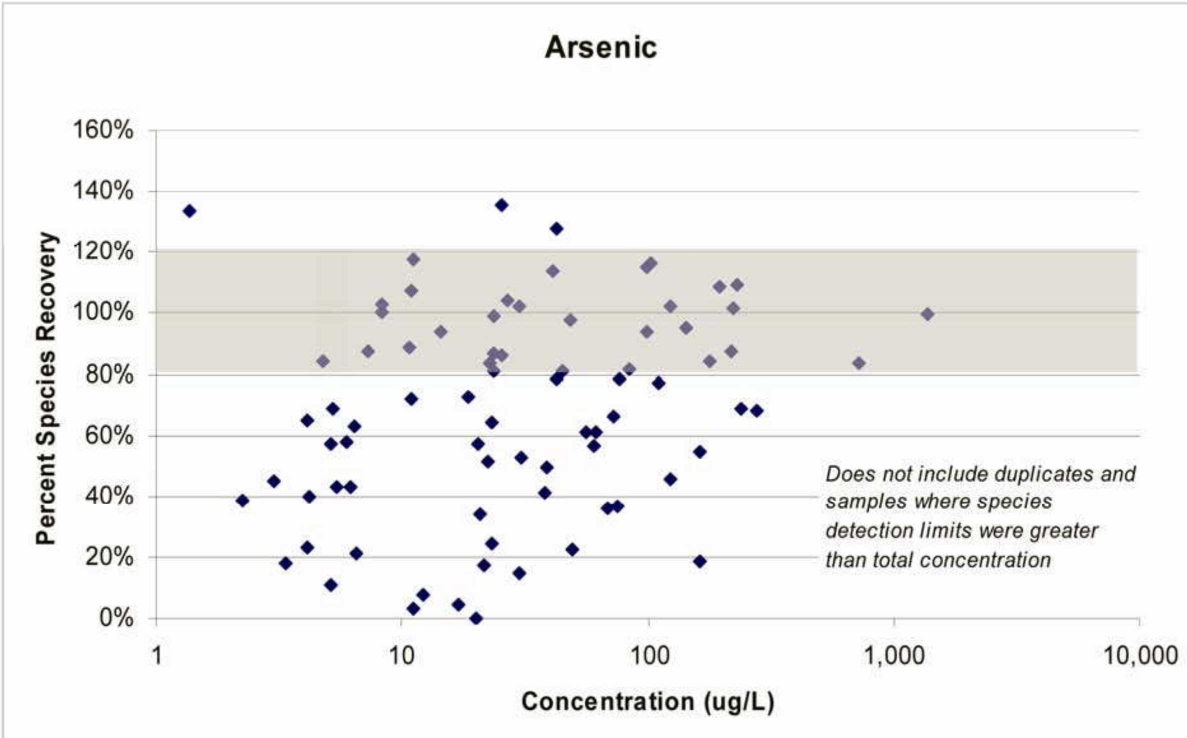


Figure 5-1
Arsenic Species Recovery

Comparison of Speciation to Site and Plant Attributes

Dominant species and relative percentages of the species were tabulated as a function of management method (landfill or impoundment) and source coal type. Relative species percentage was calculated for samples with greater than 80 percent recovery. The dominant species was determined based on the following criteria:

- For species recovery greater than 80 percent, a species was identified as dominant if its concentration was 60 percent or more of the sum of species.
- If species recovery was greater than 80 percent, and no species concentration was greater than 60 percent of the sum of species, then the sample was listed as "neutral".
- For species recovery less than 80 percent, a species was identified as dominant if its concentration was greater than 50 percent of the total concentration.[5]
- Samples with less than 80 percent species recovery in which no species concentration was greater than 50 percent of the total concentration were not tabulated.

[5] If the sum of species is 80 percent, and the species concentration is 50 percent of the total concentration, then that species accounts for at least 62.5 percent of the sum of species.

The relative percent of species recovery was tabulated for the 35 individual samples (not counting duplicates) in which the sum of species was greater than 80 percent of the total arsenic concentration (Table 5-1). For ash management sites (31 samples), the percentage of As(V) ranged from 3 to 100 percent with a median of 99 percent, the percentage of As(III) ranged from 0 to 96 percent with a median of 0.6 percent, and the percentage of other species ranged from 0 to 11 percent with a median of 0 percent. For FGD management sites (4 samples), the percentage of As(V) ranged from 20 to 41 percent with a median of 30 percent, the percentage of As(III) ranged from 26 to 78 percent with a median of 55 percent, and the percentage of other species ranged from 2 to 33 percent with a median of 15 percent. A more detailed tabulation by management method and source coal yields:

- For ash impoundments, the percentage of As(V) ranged from 3 to 100 percent for plants burning bituminous coal (20 samples), no samples from lignite/subbituminous plants had sufficient species recovery to calculate a ratio, and the percentage of As(V) ranged from 88 to 100 percent for sites receiving ash from units that burn a blend of bituminous and subbituminous coal (3 samples) (Figure 5-2).
- For ash landfills, the percentage of As(V) was 52 to 100 percent for plants burning bituminous coal (2 samples), 100 percent for plants burning lignite/subbituminous coal (3 samples), and 93 percent for a site that received ash from multiple units burning different coals (1 sample).
- One other ash management site (27413) where ash was originally sluiced, then landfilled, and where a mixture of coal sources were used, had 95 to 98 percent As(V) (2 samples).
- For FGD landfills, samples with greater than 80 percent species recovery had roughly equal percentages of As(III), As(V), and other arsenic species at sites receiving bituminous coal ash (2 samples), and a site receiving lignite ash had 72 percent As(III) (1 sample) (Figure 5-2).
- Similarly, an FGD impoundment/lignite sample had 72 percent As(III) (1 sample). There were no FGD impoundment/bituminous samples.

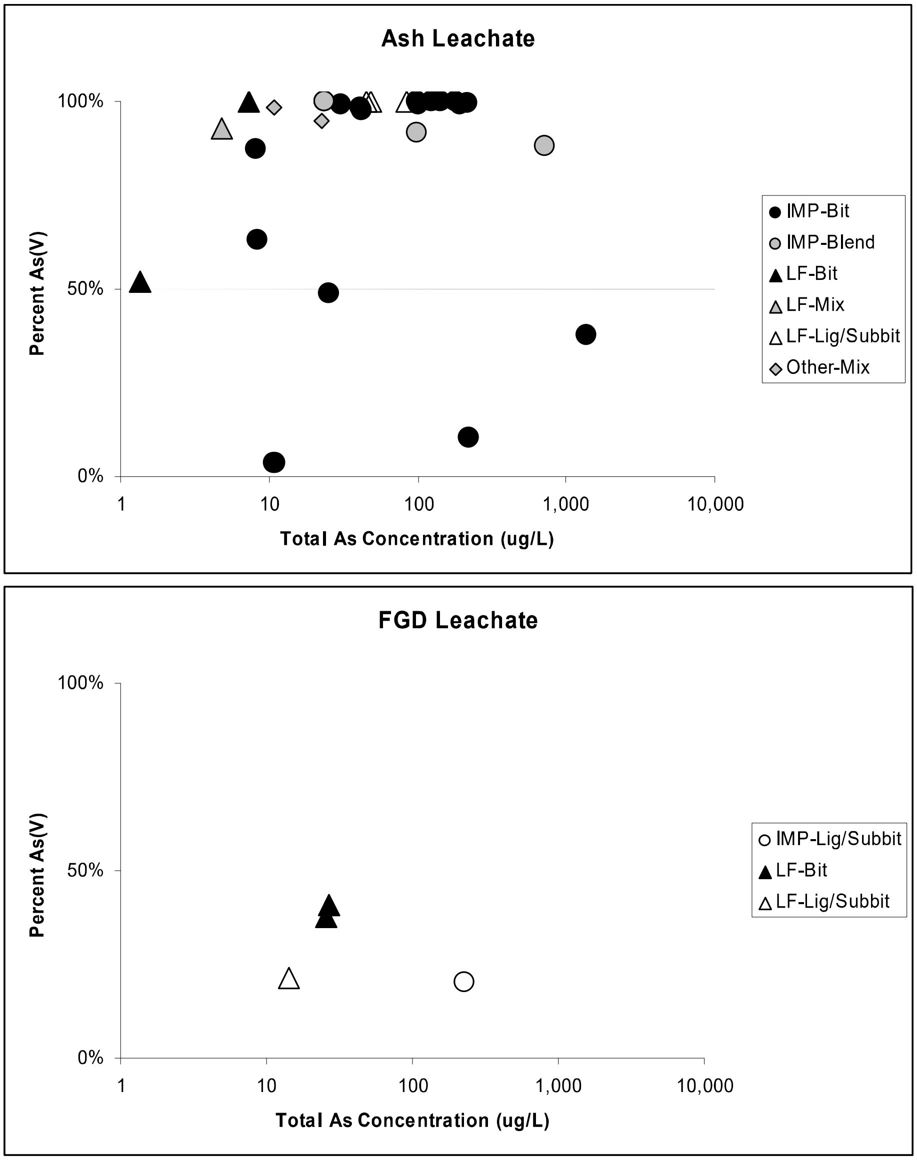


Figure 5-2
Relative Percent of As(V) vs Total As Concentration

Results of the dominant species analysis corroborates the results of the relative species analysis, and indicates that ash leachate is dominated by As(V) (Table 5-2). As(III) is only dominant in four samples from ash impoundment environments at sites where bituminous coal was burned, and in FGD leachate when bituminous coal was burned.

Table 5-2
Tabulation of Dominant Arsenic Species by Sample

Ash Samples	Impoundment	Landfill	*Total*
Ash – Bituminous	4 – 1 – 20 (36)	0 – 1 – 2 (6)	*4 – 2 – 22* *(42)*
Ash – Blend/Mix	0 – 0 – 5 (7)	0 – 0 – 2 (5)	*0 – 0 – 9** *(15*)*
Ash – Subbituminous/Lignite	0 – 0 – 2 (5)	0 – 0 – 4 (5)	*0 – 0 – 6* *(10)*
Total	*4 – 1 – 27* *(48)*	*0 – 1 – 8* *(16)*	*4 – 2 – 37** *(67*)*
FGD Samples	**Impoundment**	**Landfill**	***Total***
FGD – Bituminous		0 – 2 – 1 (6)	*0 – 2 – 1* *(6)*
FGD – Blend/Mix			
FGD – Subbituminous/Lignite	1 – 0 – 0 (5)	1 – 0 – 1 (3)	*2 – 0 – 1* *(8)*
Total	*1 – 0 – 0* *(5)*	*1 – 2 – 2* *(9)*	*2 – 2 – 2* *(14)*

Legend: number of samples in which → As(III) dominant - Neutral - As(V) dominant

(Total number of samples in group)

* Tabulation includes the samples from the 27413 site, which could not be characterized as landfill or impoundment.

The four ash leachate samples dominated by As(III) (022, 026, 031, and 061) came from three different sites (49003A, 35015B, and 33104), indicating that it is not a site-specific occurrence. Furthermore, other samples from each of the three sites were dominated by As(V), indicating that it is not a site-wide occurrence. Total arsenic concentration in the four samples dominated by As(III) ranged from 11 to 1,380 µg/L (Figure 5-3). The pH values of these samples were neutral to slightly alkaline (7.1 to 8.5 SU). Sample 031 had only 6 percent dissolved oxygen and a negative ORP value, indicative of reducing conditions. Most of the other samples with dissolved oxygen concentrations lower than 10 percent were not evaluated because species recovery was too low, and no other sample had a negative ORP value. Sample 061 had abundant dissolved oxygen (65 percent), although it also had a relatively low ORP value of 140 mV and a dissolved iron concentration of 2,170 µg/L, which may be indicative of reducing conditions. The total arsenic concentration for samples 031 and 061 were an order of magnitude or more higher than the other samples collected at these sites. Samples 022 and 026, both collected from the 49003A impoundment had field measurements indicative of oxic conditions, and total arsenic concentrations were at the low end of the range for samples collected at this site.

FGD leachate samples were evenly split between the reduced and oxidized species of arsenic. There was no correlation with pH, dissolved oxygen, or ORP. In fact, the two samples clearly dominated by As(V) (106 and 121) had lower ORP values than the two samples dominated by As(III) (105 and 128).

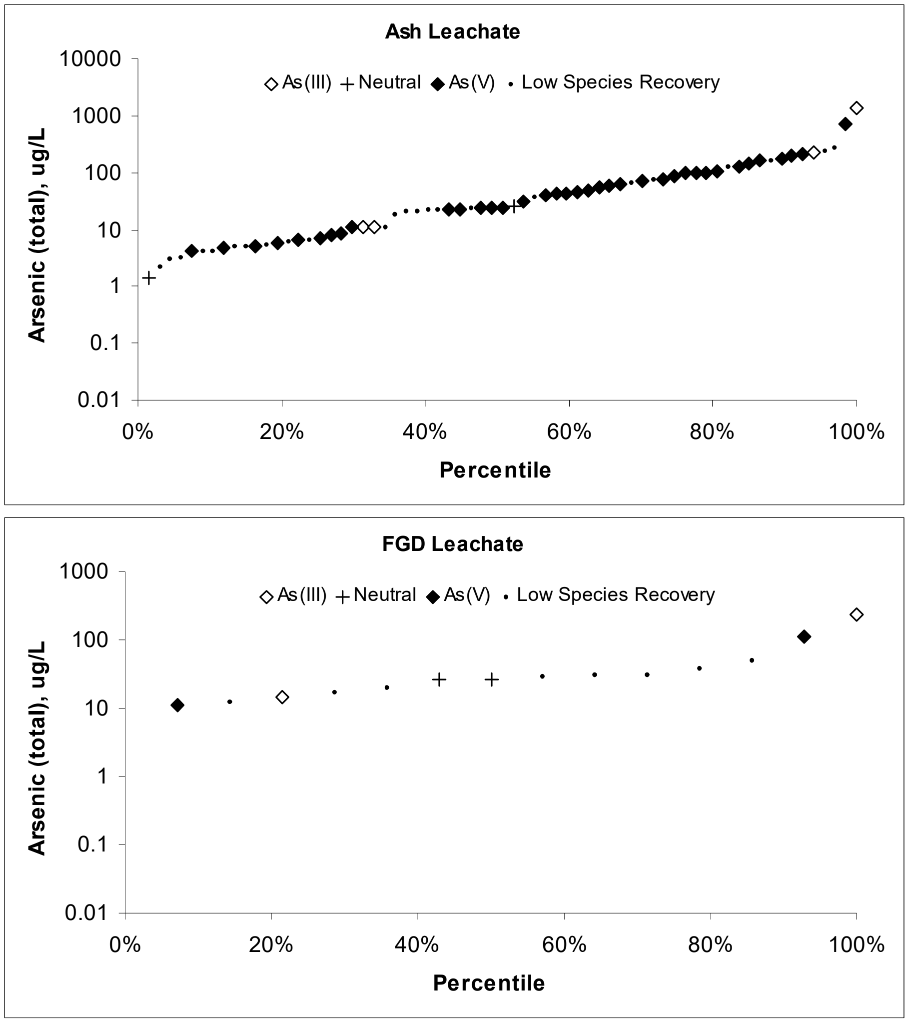


Figure 5-3
Species Predominance as a Function of Total Arsenic Concentration in Leachate.

Selenium

Overview of Results

Detectable concentrations of selenium were present in all 81 samples (Table 5-3). Review of duplicate sample results indicated that results were highly reproducible across the entire concentration range.

Selenite was detected in 58 of the 81 samples, and selenate was detected in 55 of the 81 samples. Two samples (107 and 128) contained other selenium species, which were theorized to be selenium-sulfur compounds.

Like arsenic, the selenium speciation mass balance varied strongly, and was not always satisfactory. Selenium had the same number of samples (35 of 81 samples) as arsenic with greater than 80 percent recovery (Figure 5-4); although the samples with poor species recovery were not always the same as arsenic.

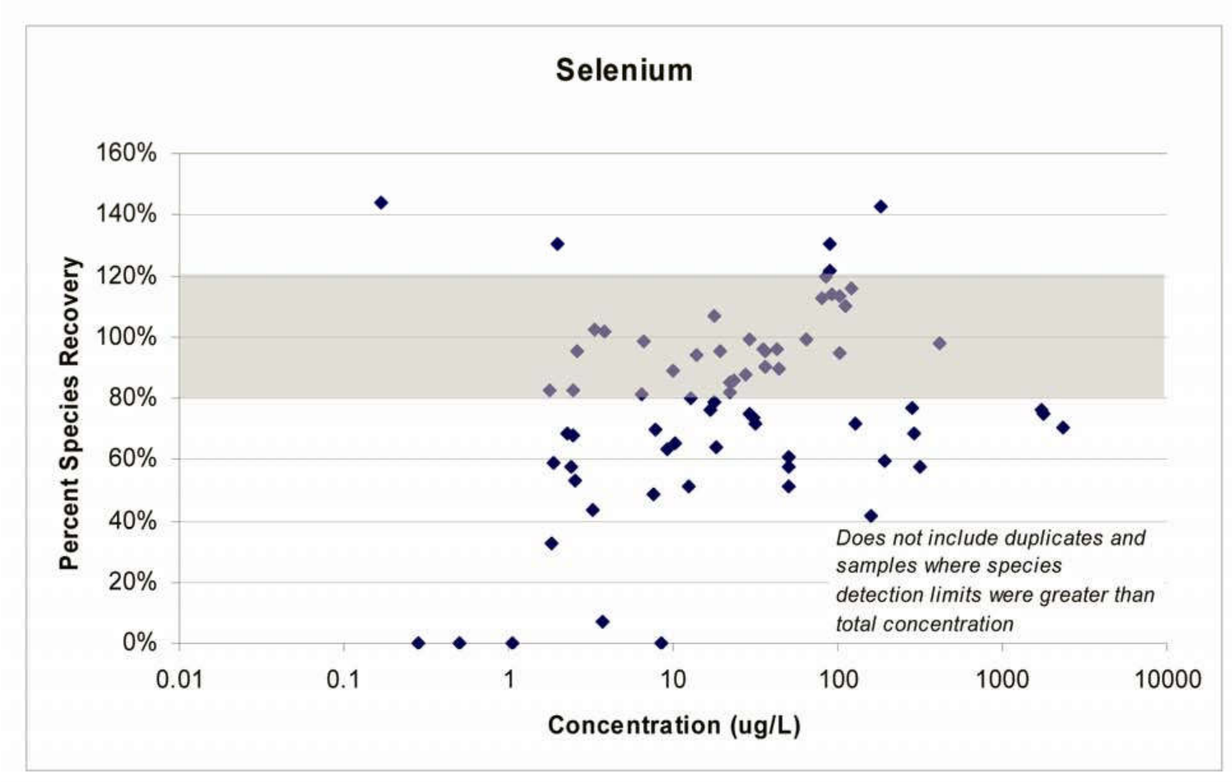


Figure 5-4
Selenium Species Recovery

Table 5-3
Selenium Speciation Data

Site	Sample	Source	CCP	Coal	Total Se (ug/L)	Se(IV) (ug/L)	Se(VI) (ug/L)	Se, other (ug/L)	Sum of Species	% Recovery	% Se(IV)	% Se(VI)	% Se (other)
50210	001	LF	FA,BA	Mix	127	8.3	83		91.3	72%			
50213	002	LF	FA	Subbit	1,730	19	1,300		1,318.6	76%			
50213	003	LF	FA	Subbit	1,760	76	1,240		1,315.9	75%			
50183	004	LF	FA,BA	Mix	50	8.1	22		30.3	61%			
50183	005	LF	FA,BA	Mix	7.6	3.1	0.57		3.7	49%			
23223A	006	LF	SDA	Subbit	17	1.6	11		12.8	76%			
23223B	007	IMP	FGD	Subbit	289	79	119		198.2	69%			
23223B	008	IMP	FGD	Subbit	3.7	<0.1	0.27		0.3	7%			
23223B	009	IMP	FGD	Subbit	2,360	<2	1,660		1,660.0	70%			
23214	010	LF	FA	Subbit	318	24	158		182.3	57%			
14093	012	IMP	FA	Bit	3.2	1.4	<0.2		1.4	43%			
14093	013	IMP	FA	Bit	0.28	<0.1	<0.1		0.0	0%			
14093	*013D*	*dup*	*FA*	*Bit*	*0.38*	*<0.1*	*<0.1*		*0.0*	*0%*			
14093	014	IMP	FA	Bit	1.8	0.59	<0.2		0.6	33%			
25410A	015	IMP	FA,BA	Blend	22	15	3.4		18.3	**82%**	81.2%	18.8%	ND
25410A	016	IMP	FA,BA	Blend	193	101	14		115.4	60%			
13115A	017	IMP	FA,BA	Subbit	2.4	0.26	1.1		1.4	57%			
13115B	018	IMP	FA,BA	Bit	0.50	<0.1	<0.2		0.0	0%			
13115A	019	IMP	FA	Subbit	1.8	0.14	1.3		1.5	**82%**	9.5%	90.5%	ND
13115A	020	IMP	FA,BA	Subbit	2.5	0.90	0.79		1.7	68%			
49003A	021	IMP	FA	Bit	6.5	5.3	<0.6		5.3	**81%**	100.0%	0.0%	ND
49003A	022	IMP	FA	Bit	31	20	2.2		22.7	74%			
49003A	023	IMP	FA	Bit	283	217	1.5		218.2	77%			
49003B	024	LF	FA	Bit	18	5.3	6.3		11.6	64%			
49003B	025	LF	FA	Bit	1.9	<0.1	1.1		1.1	59%			
49003A	026	IMP	FA	Bit	32	20	2.2		22.6	72%			
35015A	027	LF	FGD, FA	Bit	1.1	<0.3	<0.3		0.0	0%			
35015A	028	LF	FGD, FA	Bit	2.6	<0.3	1.4		1.4	53%			
35015A	029	LF	FGD, FA	Bit	2.3	<0.3	1.6		1.6	69%			

Table 5-3
Selenium Speciation Data (continued)

Site	Sample	Source	CCP	Coal	Total Se (ug/L)	Se(IV) (ug/L)	Se(VI) (ug/L)	Se, other (ug/L)	Sum of Species	% Recovery	% Se(IV)	% Se(VI)	% Se (other)
35015B	030	IMP	FA	Bit	44	27	12		39.5	**90%**	68.3%	31.7%	ND
35015B	031	IMP	FA	Bit	13	0.92	5.5		6.4	51%			
35015B	032	IMP	FA,BA	Bit	18	13	0.75		14.2	79%			
33106	037	IMP	FA	Bit	2.0	2.6	<1		2.6	131%	100.0%	0.0%	ND
33106	038	IMP	FA	Bit	0.13	<0.5	<1		0.0	0%			
33106	039	IMP	FA	Bit	0.17	0.24	<0.4		0.2	144%	100.0%	0.0%	ND
33106	042	IMP	FA	Bit	43	39	1.9		41.0	**96%**	95.3%	4.7%	ND
33106	043	IMP	FA	Bit	24	20	<1		20.2	**86%**	100.0%	0.0%	ND
33106	044	IMP	FA	Bit	14	11	1.7		13.1	**94%**	86.7%	13.3%	ND
33106	*044D*	*dup*	*FA*	*Bit*	*14*	*12*	*1.8*		*13.3*	***98%***	86.7%	13.3%	ND
33106	049	IMP	FA,BA	Bit	10	8.3	0.64		8.9	**89%**	92.8%	7.2%	ND
40109	051	IMP	FA	Bit	0.45	<0.5	<1		0.0	0%			
40109	052	IMP	FA	Bit	10	6.7	<4		6.7	65%			
40109	053	IMP	FA	Bit	1.2	<2	<4		0.0	0%			
40109	057	IMP	FA,BA	Bit	2.4	2.0	<1		2.0	**83%**	100.0%	0.0%	ND
40109	059	IMP	FA,BA	Bit	2.6	2.5	<1		2.5	**95%**	100.0%	0.0%	ND
40109	*059D*	*dup*	*FA,BA*	*Bit*	*2.6*	*2.2*	*<1*		*2.2*	***87%***	100.0%	0.0%	ND
33104	061	IMP	FA	Bit	4.3	<10	<20		0.0	0%			
33104	062	IMP	FA	Bit	112	90	32		122.5	**110%**	73.8%	26.2%	ND
33104	064	IMP	FA	Bit	103	97	<4		97.1	**95%**	100.0%	0.0%	ND
33104	069	IMP	FA,BA	Bit	36	33	1.7		34.8	**96%**	95.1%	4.9%	ND
33104	070	IMP	FA,BA	Bit	29	29	<4		28.8	**99%**	100.0%	0.0%	ND
33104	*070D*	*dup*	*FA,BA*	*Bit*	*29*	*28*	*<4*		*27.9*	***95%***	100.0%	0.0%	ND
22346	079	IMP	FA,OA	Blend	0.16	<0.2	<0.3		0.0	0%			
22346	*079D*	*dup*	*FA,OA*	*Blend*	*0.16*	*<0.2*	*<0.3*		*0.0*	*0%*			
22346	082	IMP	FA,OA	Blend	19	18	0.26		18.1	**95%**	98.6%	1.4%	ND
22347	083	IMP	FA	Blend	13	8.7	1.5		10.2	80%			
22346	084	IMP	FA,OA	Blend	0.57	<2	<3		0.0	0%			
27413	090	See Notes	FA	Mix	86	5.2	97		102.3	**120%**	5.1%	94.9%	ND
27413	091	See Notes	FA	Mix	122	3.6	138		141.9	**116%**	2.5%	97.5%	ND
27413	092	See Notes	FA	Mix	103	0.56	116		117.0	**113%**	0.5%	99.5%	ND

Table 5-3
Selenium Speciation Data (continued)

Site	Sample	Source	CCP	Coal	Total Se (ug/L)	Se(IV) (ug/L)	Se(VI) (ug/L)	Se, other (ug/L)	Sum of Species	% Recovery	% Se(IV)	% Se(VI)	% Se (other)
50212	097	LF	FA	Subbit	413	38	366		404.2	**98%**	9.4%	90.6%	ND
50183	098	LF	FA,BA	Mix	51	29	<2		29.3	58%			
50183	099	LF	FA,BA	Mix	2.0	<0.8	<2		0.0	0%			
50408	101	LF	FA,BA	Bit	91	<0.8	104		103.6	**114%**	0.0%	100.0%	ND
50211	102	LF	FA	Bit	80	5.3	85		90.8	**113%**	5.9%	94.1%	ND
34186B	105	IMP	FGD	Lig	8.5	<2	<4	<2	0.0	0%			
34186C	106	LF	FGD,FA,BA	Lig	65	<2	64	<2	64.4	**99%**	0.0%	100.0%	0.0%
34186C	*106D*	*dup*	*FGD,FA,BA*	*Lig*	*65*	*<2*	*65*	*<2*	*65.1*	***100%***	0.0%	100.0%	0.0%
34186B	107	IMP	FGD	Lig	159	<2	16	51	66.5	42%			
34186A	108	LF	FA	Lig	6.6	2.6	3.9	<0.5	6.5	**98%**	39.6%	60.4%	0.0%
49003B	111	LF	FA	Bit	91	39	72		110.3	122%	35.1%	64.9%	ND
49003B	112	LF	FA	Bit	0.67	<0.5	<1		0.0	0%			
49003A	113	IMP	FA	Bit	29	19	2.6		21.8	75%			
49003A	114	IMP	FA	Bit	0.071	<0.5	<1		0.0	0%			
49003A	115	IMP	FA	Bit	36	30	3.1		32.7	**90%**	90.7%	9.3%	ND
49003A	116	IMP	FA	Bit	35	31	3.3		34.0	**96%**	90.2%	9.8%	ND
35015B	118	IMP	FA,BA	Bit	18	18	1.3		18.9	**107%**	93.0%	7.0%	ND
35015B	*118D*	*dup*	*FA,BA*	*Bit*	*18*	*16*	*1.3*		*17.7*	***96%***	92.9%	7.1%	ND
35015B	119	IMP	FA,BA	Bit	28	23	1.7		24.4	**87%**	93.1%	6.9%	ND
35015A	120	LF	FGD, FA	Bit	3.3	1.8	1.5		3.4	**102%**	54.7%	45.3%	ND
35015A	121	LF	FGD, FA	Bit	3.9	1.1	2.8		3.9	**102%**	28.2%	71.8%	ND
35015A	122	LF	FGD, FA	Bit	1.1	<0.5	<1		0.0	0%			
43035	126	IMP	FA,BA	Subbit	89	13	103	<0.3	115.9	131%	10.8%	89.2%	0.0%
43035	*126D*	*dup*	*FA,BA*	*Subbit*	*88*	*13*	*104*	*<0.3*	*116.9*	*132%*	11.1%	88.9%	0.0%
43035	127	IMP	FA,BA	Subbit	181	12	245	<0.3	257.5	143%	4.8%	95.2%	0.0%
43034	128	LF	FGD,FA	Lig	51	17	6.7	1.8	25.9	51%			

Table 5-3
Selenium Speciation Data (continued)

Site	Sample	Source	CCP	Coal	Total Se (ug/L)	Se(IV) (ug/L)	Se(VI) (ug/L)	Se, other (ug/L)	Sum of Species	% Recovery	% Se(IV)	% Se(VI)	% Se (other)
13115B	HN-1	IMP	FA,BA	Bit	22	2.6	16		19.0	**85%**	13.9%	86.1%	ND
13115B	HN-2	IMP	FA,BA	Bit	9.2	<1	5.8		5.8	64%			
25410B	SX-1	IMP	FA	Blend	7.8	1.8	3.6		5.4	70%			

Notes:

Ash at site 27413 (samples 090, 091, 092) was first sluiced, then managed dry.

Abbreviations:

Bit = bituminous; Subbit = Subbituminous; Mix = CCP from different units burning different coals; Blend = CCP from a single unit burning two different fuels

FA = fly ash; BA = bottom ash; EA = economizer ash; FGD = flue gas desulfurization sludge; OA = oil ash

LF = landfill; IMP = impoundment; DUP = duplicate sample

ND = not determined

Comparison of Speciation to Site and Plant Attributes

Dominant species and relative percentages of the species were tabulated using the same procedure as for arsenic. For ash management sites (32 samples), the percentage of Se(IV) ranged from 0 to 100 percent with a median of 88 percent, the percentage of Se(VI) ranged from 0 to 100 percent with a median of 12 percent, and the percentage of other species was 0 percent for samples with greater than 80 percent species recovery. For FGD management sites (3 samples), the percentage of Se(IV) ranged from 0 to 55 percent with a median of 28 percent, the percentage of Se(VI) ranged from 45 to 100 percent with a median of 72 percent, and the percentage of other species was 0 percent. A more detailed tabulation by management method and source coal yields:

- For ash impoundments, the percentage of Se(VI) ranged from 0 to 86 percent for plants burning bituminous coal (19 samples), 89 to 95 percent for plants burning lignite/subbituminous coal (3 samples), and 1 to 19 percent for sites receiving ash from units that burn a blend of bituminous and subbituminous coal (2 samples) (Figure 5-5).
- For ash landfills, the percentage of Se(VI) was 65 to 100 percent for plants burning bituminous coal (3 samples), and 60 to 91 percent for plants burning lignite/subbituminous coal (2 samples).
- One other ash management site (27413) where ash was originally sluiced, then landfilled, and where a mixture of coal sources were used, had 95 to 99 percent Se(VI) (3 samples).
- For FGD landfills, the percentage of Se(VI) was 45 to 72 percent for plants burning bituminous coal (2 samples), and 100 percent for plants burning lignite/subbituminous coal (1 sample) (Figure 5-5).
- No FGD impoundment samples had greater than 80 percent species recovery.

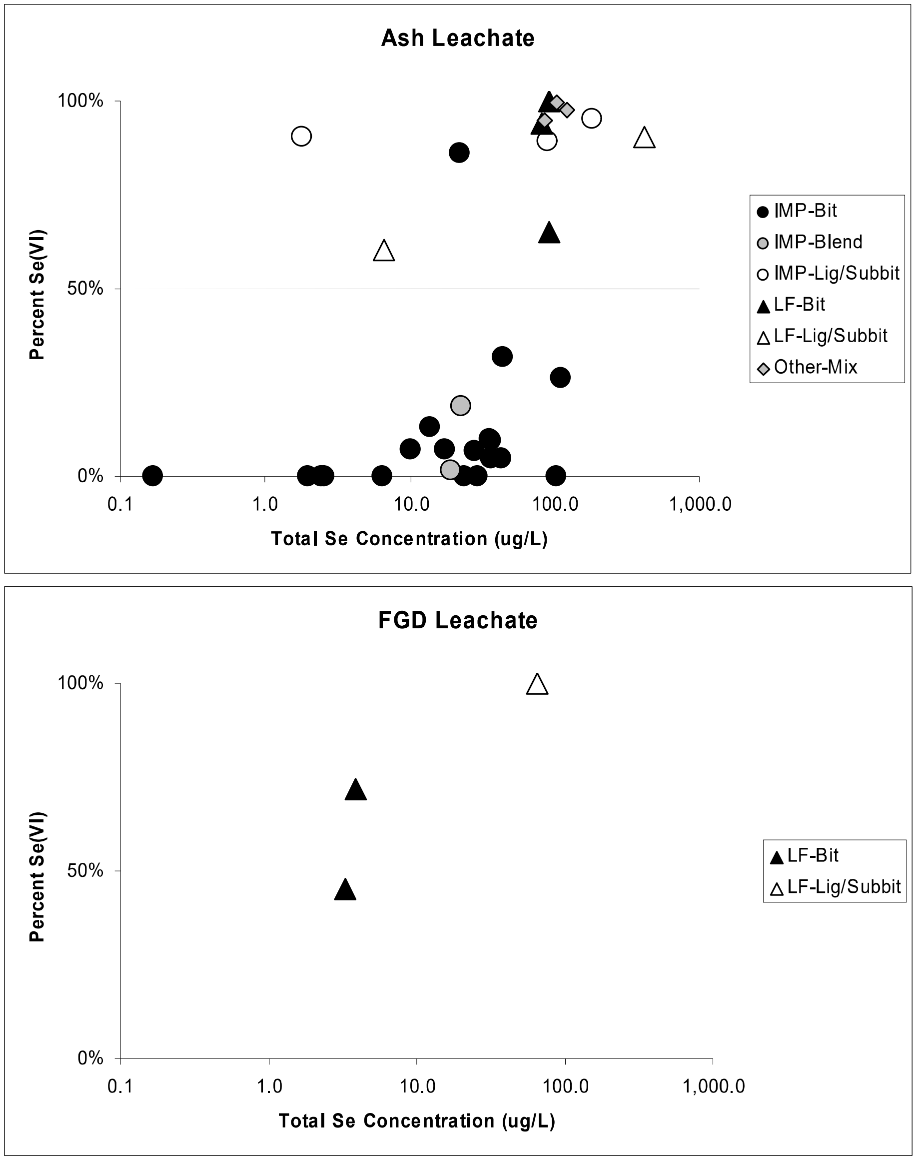


Figure 5-5
Relative Percent of Se(VI) versus Total Se Concentration

Results of the dominant species analysis corroborates the relative percentage analysis and indicates that ash leachate is dominated by Se(IV) in impoundment settings when the source coal is bituminous or a mixture of bituminous and subbituminous, while Se(VI) is predominant in landfill settings and when the source coal is subbituminous/lignite (Table 5-4). Most samples with relatively high concentration (>80 µg/L) were dominated by Se(VI) while samples with concentrations lower than 50 µg/L were mostly dominated by Se(IV) (Figure 5-6).

Table 5-4
Tabulation of Dominant Selenium Species by Sample

Ash Samples	Impoundment	Landfill	*Total*
Ash – Bituminous	24 – 0 – 2 (36)	0 – 0 – 4 (6)	*24 – 0 – 6* *(42)*
Ash – Blend/Mix	4 – 0 – 0 (7)	1 – 0 – 1 (5)	*5 – 0 – 4** *(15*)*
Ash – Subbituminous/Lignite	0 – 0 – 3 (5)	0 – 0 – 4 (5)	*0 – 0 – 7* *(10)*
Total	*28 – 0 – 5* *(48)*	*1 – 0 – 9* *(16)*	*29 – 0 – 17** *(67*)*
FGD Samples	**Impoundment**	**Landfill**	***Total***
FGD – Bituminous		0 – 1 – 3 (6)	*0 – 1 – 3* *(6)*
FGD – Blend/Mix			
FGD – Subbituminous/Lignite	0 – 0 – 1 (5)	0 – 0 – 2 (3)	*0 – 0 – 3* *(8)*
Total	*0 – 0 – 1* *(5)*	*0 – 1 – 5* *(9)*	*0 – 1 – 6* *(14)*

Legend: number of samples in which → Se(IV) dominant - Neutral - Se(VI) dominant
(Total number of samples in group)

* Tabulation includes the samples from the 27413 site, which could not be characterized as landfill or impoundment.

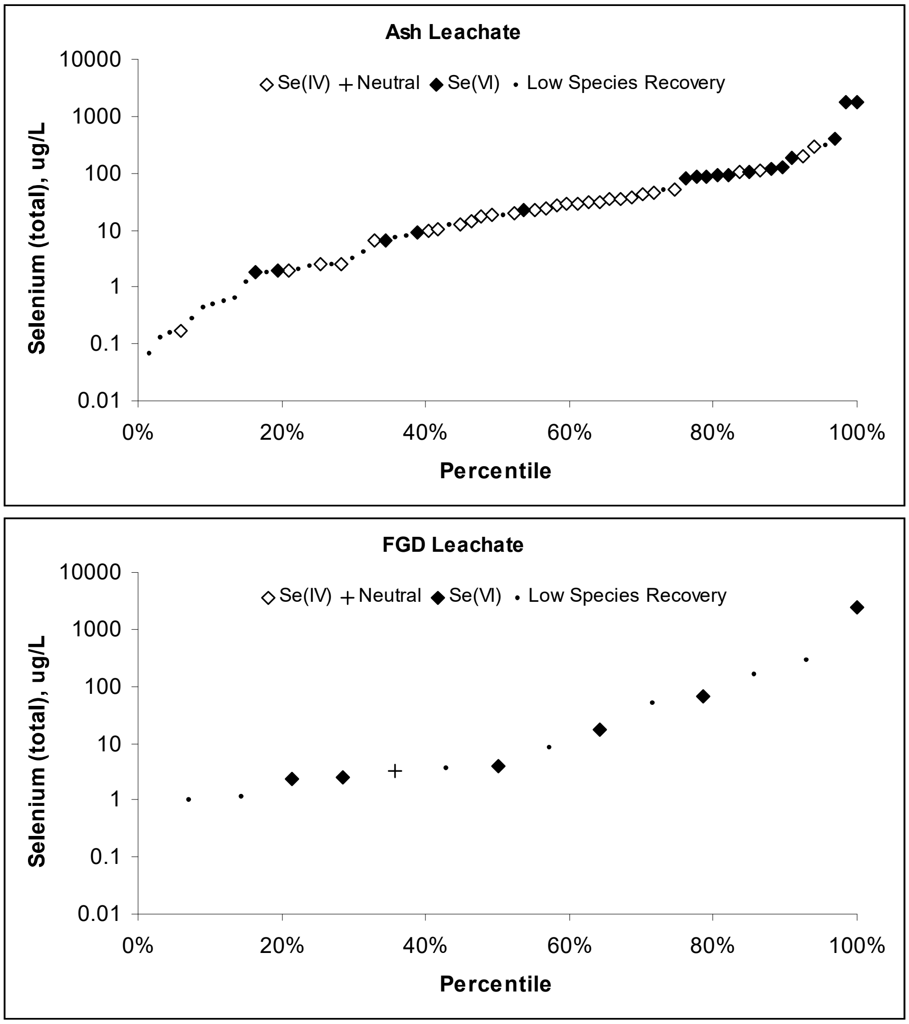


Figure 5-6
Species Predominance as a Function of Total Selenium Concentration in Leachate.

Chromium

Overview of Results

Chromium was detected in 42 of the 81 samples (Table 5-5). Chromium speciation was not always determined in samples for which total concentrations were non-detect or lower than 1 µg/L. Cr(III) analysis was performed for 45 samples, and 29 had detectable concentrations. Cr(VI) was analyzed in 58 samples and 37 had detectable concentrations. Review of duplicate samples indicated that chromium results were reproducible.

The speciation mass balance was good for total chromium concentrations greater than 5 µg/L: 16 of 19 samples with concentration greater than 5 µg/L had species recovery greater than 80 percent (Figure 5-7). The three other samples from this group had greater than 65 percent recovery.

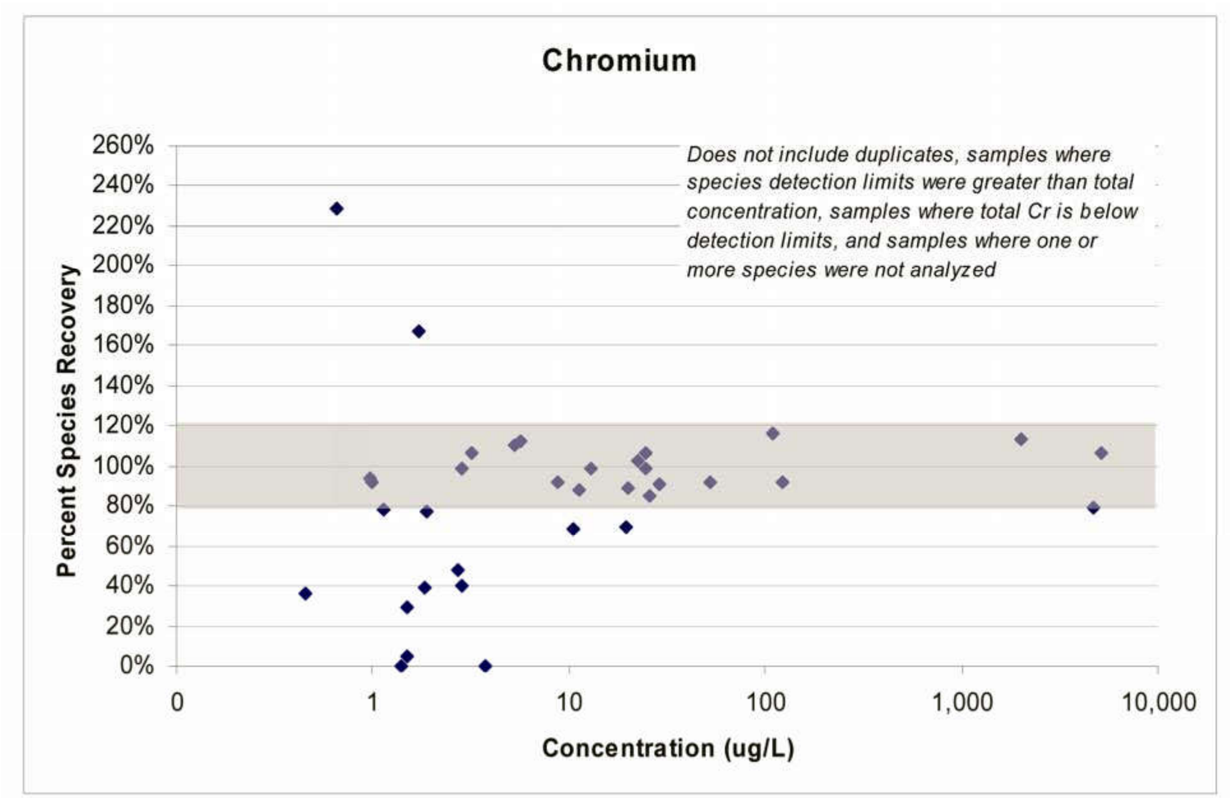


Figure 5-7
Chromium Species Recovery

Table 5-5
Chromium Speciation Data

Site	Sample	Source	Byproduct	Coal	Total Cr (ug/L)	Cr(III) (ug/L)	Cr(VI) (ug/L)	Sum of Species	% Recovery	% Cr(III)	% Cr(VI)
50210	001	LF	FA,BA	Mix	<0.5		2.2	2.20	*		
50213	002	LF	FA	Subbit	5,100	340	5,090	5,430.00	**106%**	6%	94%
50213	003	LF	FA	Subbit	4,670	190	3,530	3,720.00	80%		
50183	004	LF	FA,BA	Mix	8.8	<0.1	8.1	8.10	**92%**	0%	100%
50183	005	LF	FA,BA	Mix	0.66		1.5	1.50	229%	0%	100%
23223A	006	LF	SDA	Subbit	5.7	<0.1	6.4	6.40	**113%**	0%	100%
23223B	007	IMP	FGD	Subbit	1.7	<0.1	2.9	2.90	167%	0%	100%
23223B	008	IMP	FGD	Subbit	<0.5		<0.1	*	*		
23223B	009	IMP	FGD	Subbit	53	1.3	47	48.53	**92%**	3%	97%
23214	010	LF	FA	Subbit	26	<0.4	22	22.00	**85%**	0%	100%
14093	012	IMP	FA	Bit	<0.5		1.9	1.90	*		
14093	013	IMP	FA	Bit	<0.5		0.70	0.70	*		
14093	*013D*	*dup*	*FA*	*Bit*			*0.70*	*0.70*	*		
14093	014	IMP	FA	Bit	<0.5		0.50	0.50	*		
25410A	015	IMP	FA,BA	Blend	13	<0.4	13	12.80	**99%**	0%	100%
25410A	016	IMP	FA,BA	Blend	3.8	<0.1	<0.5	*	0%		
13115A	017	IMP	FA,BA	Subbit	2.8	<0.04	2.8	2.80	**98%**	0%	100%
13115B	018	IMP	FA,BA	Bit	<0.5		1.3	1.30	*		
13115A	019	IMP	FA	Subbit	0.96	<0.1	0.90	0.90	**94%**	0%	100%
13115A	020	IMP	FA,BA	Subbit	0.66		<0.05	*	0%		
49003A	021	IMP	FA	Bit	<0.5		<0.05	*	*		
49003A	022	IMP	FA	Bit	0.98	<0.04	0.90	0.90	**92%**	0%	100%
49003A	023	IMP	FA	Bit	<0.5		<0.5	*	*		
49003B	024	LF	FA	Bit	<0.5			*	*		
49003B	025	LF	FA	Bit	<0.5			*	*		
49003A	026	IMP	FA	Bit	1.1	<0.04	0.90	0.90	78%		
35015A	027	LF	FGD, FA	Bit	<0.5			*	*		
35015A	028	LF	FGD, FA	Bit	<0.5			*	*		
35015A	029	LF	FGD, FA	Bit	<0.5			*	*		

Table 5-5
Chromium Speciation Data (continued)

Site	Sample	Source	Byproduct	Coal	Total Cr (ug/L)	Cr(III) (ug/L)	Cr(VI) (ug/L)	Sum of Species	% Recovery	% Cr(III)	% Cr(VI)
35015B	030	IMP	FA	Bit	<0.5		<0.05	*	*		
35015B	031	IMP	FA	Bit	<0.5		<0.1	*	*		
35015B	032	IMP	FA,BA	Bit	1.4	<0.1	<0.05	*	0%		
33106	037	IMP	FA	Bit	<0.4	<0.01	<0.01	*	*		
33106	038	IMP	FA	Bit	<0.4	<0.01	<0.01	*	*		
33106	039	IMP	FA	Bit	<0.4	<0.01	<0.01	*	*		
33106	042	IMP	FA	Bit	<0.4	0.17	0.029	0.20	*		
33106	043	IMP	FA	Bit	29	26	<0.1	26.42	**91%**	100%	0%
33106	044	IMP	FA	Bit	<0.4	0.25	<0.01	0.25	*		
33106	*044D*	*dup*	*FA*	*Bit*	*<0.4*	*0.12*	*<0.01*	*0.12*	*		
33106	049	IMP	FA,BA	Bit	<0.4	0.074	<0.01	0.07	*		
40109	051	IMP	FA	Bit	11	9.9	<0.05	9.92	**88%**	100%	0%
40109	052	IMP	FA	Bit	<0.4	0.16	0.064	0.22	*		
40109	053	IMP	FA	Bit	<0.4	0.050	<0.01	0.05	*		
40109	057	IMP	FA,BA	Bit	1.9	1.1	0.41	1.47	77%		
40109	059	IMP	FA,BA	Bit	2.7	0.011	1.3	1.29	48%		
40109	*059D*	*dup*	*FA,BA*	*Bit*	*2.5*	*<0.01*	*1.2*	*1.23*	*49%*		
33104	061	IMP	FA	Bit	<0.4	0.27	<0.01	0.27	*		
33104	062	IMP	FA	Bit	10	0.95	6.2	7.19	69%		
33104	064	IMP	FA	Bit	22	0.044	23	23.02	**103%**	0%	100%
33104	069	IMP	FA,BA	Bit	3.2	0.46	3.0	3.44	**107%**	13%	87%
33104	070	IMP	FA,BA	Bit	5.3	0.63	5.3	5.91	**111%**	11%	89%
33104	*070D*	*dup*	*FA,BA*	*Bit*	*5.4*	*0.62*	*5.2*	*5.78*	***106%***	*11%*	*89%*
22346	079	IMP	FA,OA	Blend	<0.2	<0.02	<0.006	*	*		
22346	*079D*	*dup*	*FA,OA*	*Blend*	*<0.2*	*<0.02*	*<0.006*	*	*		
22346	082	IMP	FA,OA	Blend	25	1.2	23	24.19	**98%**	5%	95%
22347	083	IMP	FA	Blend	20	2.4	15	17.66	**89%**	14%	86%
22346	084	IMP	FA,OA	Blend	<0.2	0.039	<0.006	0.04	*		
27413	090	See Notes	FA	Mix	0.75			*	*		
27413	091	See Notes	FA	Mix	<0.2			*	*		
27413	092	See Notes	FA	Mix	122	2.8	109	111.61	**91%**	2%	98%

Table 5-5
Chromium Speciation Data (continued)

Site	Sample	Source	Byproduct	Coal	Total Cr (ug/L)	Cr(III) (ug/L)	Cr(VI) (ug/L)	Sum of Species	% Recovery	% Cr(III)	% Cr(VI)
50212	097	LF	FA	Subbit	2,000	40	2,230	2,270.00	**114%**	2%	98%
50183	098	LF	FA,BA	Mix	2.8	0.16	0.99	1.15	40%		
50183	099	LF	FA,BA	Mix	<0.2			*	*		
50408	101	LF	FA,BA	Bit	1.5	<0.08	0.075	0.07	5%		
50211	102	LF	FA	Bit	20	0.42	13	13.70	70%		
34186B	105	IMP	FGD	Lig	<0.4			*	*		
34186C	106	LF	FGD,FA,BA	Lig	0.91			*	*		
34186C	*106D*	*dup*	*FGD,FA,BA*	*Lig*	*0.88*			*	*		
34186B	107	IMP	FGD	Lig	<2			*	*		
34186A	108	LF	FA	Lig	0.48			*	*		
49003B	111	LF	FA	Bit	0.54			*	*		
49003B	112	LF	FA	Bit	<0.2			*	*		
49003A	113	IMP	FA	Bit	<0.2			*	*		
49003A	114	IMP	FA	Bit	0.31			*	*		
49003A	115	IMP	FA	Bit	1.5	0.34	0.092	0.43	29%		
49003A	116	IMP	FA	Bit	1.8	0.40	0.31	0.71	39%		
35015B	118	IMP	FA,BA	Bit	<0.2			*	*		
35015B	*118D*	*dup*	*FA,BA*	*Bit*	*<0.2*			*	*		
35015B	119	IMP	FA,BA	Bit	0.23			*	*		
35015A	120	LF	FGD, FA	Bit	<0.2			*	*		
35015A	121	LF	FGD, FA	Bit	<0.2			*	*		
35015A	122	LF	FGD, FA	Bit	<0.2			*	*		
43035	126	IMP	FA,BA	Subbit	108	4.1	121	125.04	**116%**	3%	97%
43035	*126D*	*dup*	*FA,BA*	*Subbit*	*109*	*2.1*	*122*	*124.39*	***114%***	*2%*	*98%*
43035	127	IMP	FA,BA	Subbit	24	0.53	26	26.03	**107%**	2%	98%
43034	128	LF	FGD,FA	Lig	0.46	0.16	<0.02	0.16	36%		

Table 5-5
Chromium Speciation Data (continued)

Site	Sample	Source	Byproduct	Coal	Total Cr (ug/L)	Cr(III) (ug/L)	Cr(VI) (ug/L)	Sum of Species	% Recovery	% Cr(III)	% Cr(VI)
13115B	HN-1	IMP	FA,BA	Bit	<0.5			*	*		
13115B	HN-2	IMP	FA,BA	Bit	<0.5			*	*		
25410B	SX-1	IMP	FA	Blend	<0.5		<0.1	*	*		

Notes:

Ash at site 27413 (samples 090, 091, 092) was first sluiced, then managed dry.

* indicates that sum of species was not calculated because individual species were not analyzed or not detected, **or** % recovery was not calculated because the total chromium concentration was below detection limits or individual species were not analyzed.

Abbreviations:

Bit = bituminous; Subbit = Subbituminous; Mix = CCP from different units burning different coals; Blend = CCP from a single unit burning two different fuels

FA = fly ash; BA = bottom ash; EA = economizer ash; FGD = flue gas desulfurization sludge; OA = oil ash

LF = landfill; IMP = impoundment; DUP = duplicate sample

ND = not determined

Comparison of Speciation to Site and Plant Attributes

For ash leachate samples with greater than 80 percent species recovery (20 samples), the percentage of Cr(III) ranged from 0 to 100 percent, with a median of 2 percent and the range of Cr(VI) was 0 to 100 percent with a median of 98 percent. For FGD leachate (3 samples), Cr(III) ranged from 0 to 3 percent with a median of 0 percent and Cr(VI) ranged from 97 to 100 percent with a median of 100 percent (Figure 5-8).

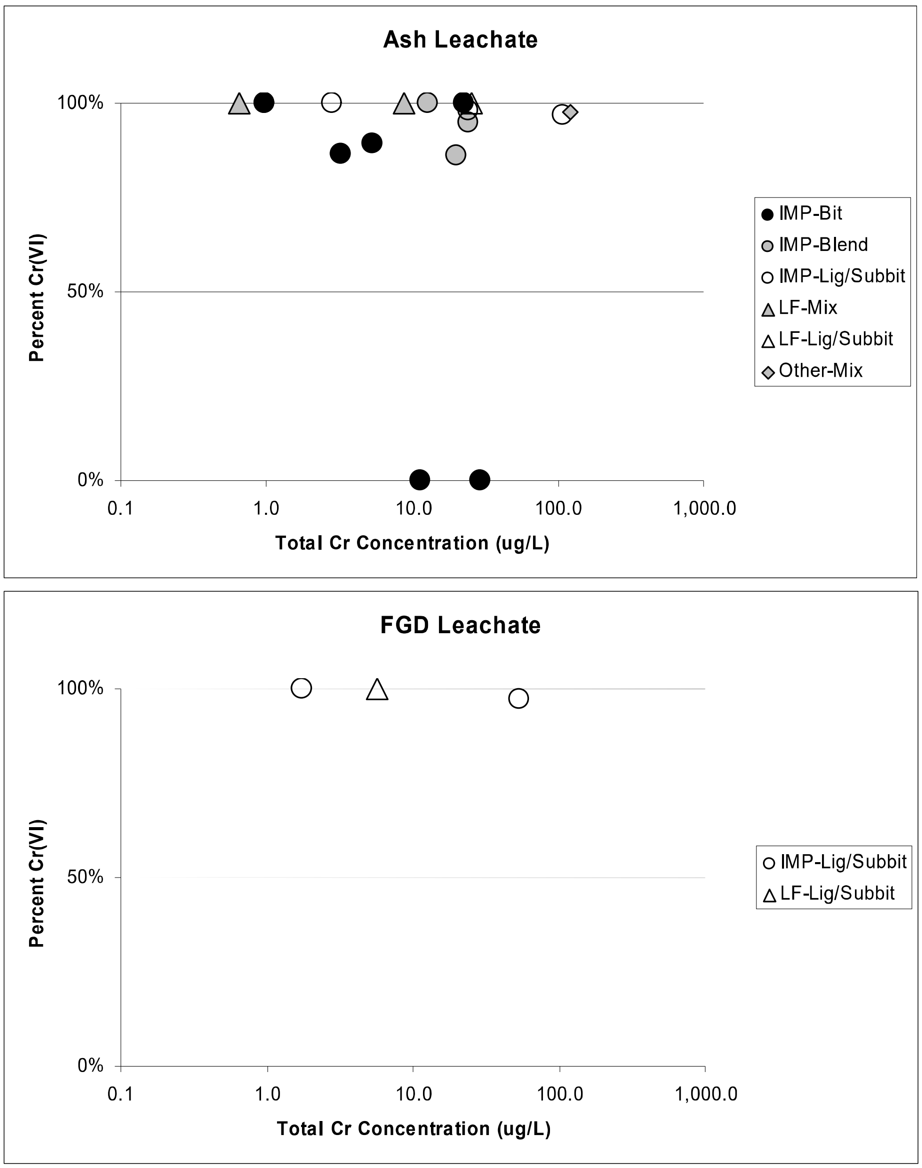


Figure 5-8
Percent Cr(VI) versus Total Cr Concentration

Using the same approach as for arsenic and selenium, the dominant chromium species was determined in 27 samples, and 24 of these were dominated by Cr(VI). The only samples dominated by Cr(III) were obtained from impoundments where the source coal was bituminous (Table 5-6). Two of these samples had very low pH (<4.5) and the other had relatively low concentration. There was no apparent relationship of between chromium speciation and total concentration (Figure 5-9).

The predominance of Cr(VI) matches geochemical expectations, because nearly all leachate samples are neutral to alkaline, and Cr(VI) is very soluble under such conditions, while Cr(III) would precipitate or bind strongly to mineral surfaces. The notable exceptions were samples 043 and 051, which only contained soluble Cr(III), and sample 057 which had a mixture of Cr(III) and Cr(V)), but also had a relatively low total concentration (1.9 µg/L). Samples 043 and 051 had the lowest pH values measured in the study (4.26 and 4.35, respectively; 1.5 pH units lower than the next lowest sample). Under the strongly acidic pH of these samples, the solubility of Cr(III) and Cr(VI) is reversed.

Five samples (002, 003, 092, 097, and 126) had Cr(VI) concentrations greater than 100 µg/L, and three of those samples (002, 003, and 097) had concentrations > 1,000 µg/L. All five samples were strongly alkaline (pH > 9.4) and oxidizing (Eh > 200 mV), and four are known to have had subbituminous coal as the CCP source (the coal source for sample 092 was uncertain).

Table 5-6
Tabulation of Dominant Selenium Species by Sample

Ash Samples	**Impoundment**	**Landfill**	***Total*****
Ash – Bituminous	3 – 0 – 6 (15)	0 – 0 – 1 (3)	*3 – 0 – 7* *(18)*
Ash – Blend/Mix	0 – 0 – 3 (4)	0 – 0 – 2 (3)	*0 – 0 – 6** *(9*)*
Ash – Subbituminous/Lignite	0 – 0 – 4 (5)	0 – 0 – 4 (5)	*0 – 0 – 8* *(10)*
Total	*3 – 0 – 13* *(24)*	*0 – 0 – 7* *(11)*	*3 – 0 – 21** *(37*)*
FGD Samples	**Impoundment**	**Landfill**	***Total*****
FGD – Bituminous			
FGD – Blend/Mix			
FGD – Subbituminous/Lignite	0 – 0 – 2 (2)	0 – 0 – 1 (3)	*0 – 0 – 3* *(5)*
Total	0 – 0 – 2 (2)	0 – 0 – 1 (3)	*0 – 0 – 3* *(5)*

Legend: number of samples in which → Cr(III) dominant - Neutral - Cr(VI) dominant

(Total number of samples in group)

* Tabulation includes two samples from the 27413 site, which could not be characterized as landfill or impoundment.

** Sum of total ash and FGD samples is less than 81 because only 42 samples had detectable chromium concentrations.

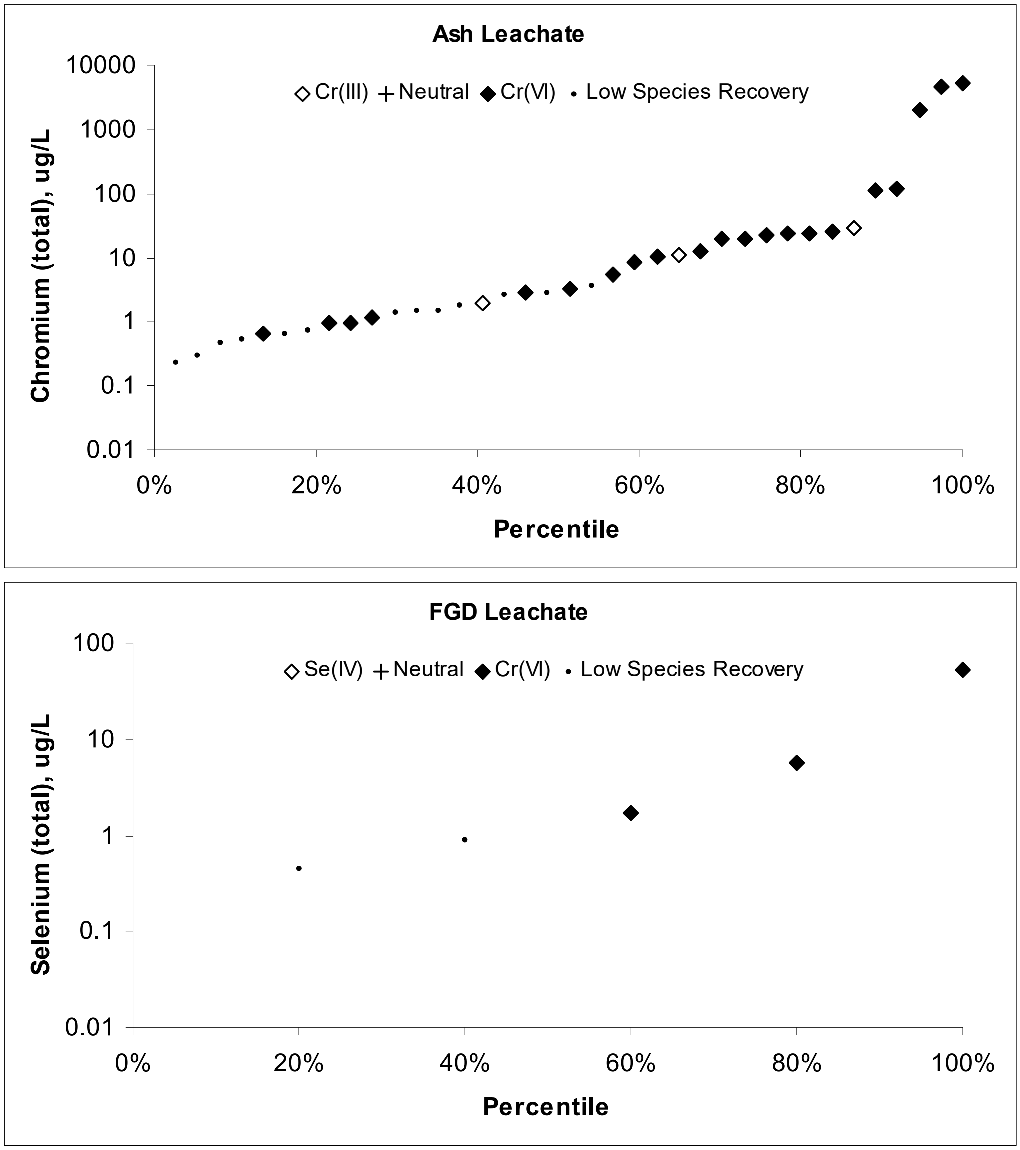


Figure 5-9
Species Predominance as a Function of Total Chromium Concentration in Leachate.

Mercury

Mercury speciation was determined on 31 samples, not counting duplicates (Table 5-7). Dimethyl mercury (DMM) was not determined on four of these samples, either because no sample was collected (due to logistic issues) or because the sample was lost during analysis (due to the fact that the employed analytical technique only allows one analysis attempt per sample). In addition, there was no particulate methyl mercury ($MeHg_{part}$) for one sample due to a field equipment problem; and dissolved methyl mercury and particulate mercury were not analyzed in another sample due to insufficient sample volume. The two duplicate samples showed poor reproducibility of results.

Table 5-7
Mercury Species Data

Site	Sample	Source	CCP	Coal	Hg_{diss} (ng/L)	DMM (ng/L)	$MeHg_{diss}$ (ng/L)	Hg_{part} (ng/L)	$MeHg_{part}$ (ng/L)
50210	001	LF	FA,BA	Mix		0.055			0.028
50213	002	LF	FA	Subbit	14	0.0051	0.11	254	0.032
50213	003	LF	FA	Subbit	18	<0.005	0.091	26	<0.01
50183	004	LF	FA,BA	Mix	5.9	<0.005	0.26	<1	0.036
50183	005	LF	FA,BA	Mix	2.1	0.0097	0.12	44	0.086
23223A	006	LF	SDA	Subbit	0.82	<0.005	0.54	25	0.092
23223B	007	IMP	FGD	Subbit	1.9	0.0074	<0.02	16	0.022
23223B	008	IMP	FGD	Subbit	4.2	<0.005	0.068	<1	0.013
23223B	009	IMP	FGD	Subbit	28		<0.02	121	0.015
49003A	021	IMP	FA	Bit	1.4	<0.005	0.034	155	0.020
49003A	022	IMP	FA	Bit	1.00	<0.005	0.027	53	0.027
49003A	023	IMP	FA	Bit	1.4	<0.005	<0.02	14	0.026
49003A	026	IMP	FA	Bit	0.38	<0.005	<0.02	17	<0.01
35015A	027	LF	FGD, FA	Bit	21	<0.005	1.6	4.3	<0.01
35015A	028	LF	FGD, FA	Bit	1.2	<0.005	0.18	13	<0.01
35015A	029	LF	FGD, FA	Bit	12	<0.005	0.70	59	0.011
35015B	030	IMP	FA	Bit	0.80	0.022	0.063	<1	0.11
35015B	031	IMP	FA	Bit	5.2	0.050	6.7	30	
35015B	032	IMP	FA,BA	Bit	1.4	0.032	0.047	186	0.055
22346	079	IMP	FA,OA	Blend	0.25	<0.005	<0.02	5.8	0.058
22346	*079D*	*dup*	*FA,OA*	*Blend*	*0.48*	*<0.005*	*0.053*	*3.0*	*0.052*
22346	082	IMP	FA,OA	Blend	5.9	<0.005	0.046	18	0.027
22347	083	IMP	FA	Blend	2.1	0.040	0.17	22	0.16
22346	084	IMP	FA,OA	Blend	0.58	<0.005	0.056	4.6	0.027
50212	097	LF	FA	Subbit	37	*	0.22	16	0.054
50183	098	LF	FA,BA	Mix	61	*	0.76	11	0.015
50183	099	LF	FA,BA	Mix	5.7	*	0.033	13	<0.01
50408	101	LF	FA,BA	Bit	2.1	*	<0.02	3.0	0.010
50211	102	LF	FA	Bit	3.8	*	0.12	52	<0.01
43035	126	IMP	FA,BA	Subbit	9.4		0.17	3.1	0.024
43035	*126D*	*dup*	*FA,BA*	*Subbit*	*2.0*		*0.21*	*6.1*	*0.024*
43035	127	IMP	FA,BA	Subbit	5.4		0.028	3.0	0.018
43034	128	LF	FGD,FA	Lig	79		6.4	100	0.059

Notes:

* Failed QC due to high concentration in the equipment blank sample.

Abbreviations:

Bit = bituminous; Subbit = Subbituminous; Mix = CCP from different units burning different coals; Blend = CCP from a single unit burning two different fuels

FA = fly ash; BA = bottom ash; EA = economizer ash; FGD = flue gas desulfurization sludge; OA = oil ash

LF = landfill; IMP = impoundment; DUP = duplicate sample

Total Hg_{diss} was detected in all 30 samples where collected, with concentrations ranging from 0.25 to 79 ng/L. Particulate mercury was detected in 27 of 30 samples.

DMM results were detectable in only 8 of the 22 samples that passed QC, and detected concentrations were lower than 0.06 ng/L. Samples 097 through 102 reported considerably higher DMM concentrations than the other samples; however, the second highest concentration was from equipment blank sample 084 (0.81 ng/L). As a result, DMM samples 097 through 102, which were collected on a single trip, failed to meet QC criteria, and were not reported here. There was no apparent difference in DMM concentration by coal type or management method.

$MeHg_{diss}$ was detected in 24 of 30 samples where analyzed, and concentrations ranged from non-detect to 6.7 ng/L. Only three samples had a $MeHg_{diss}$ concentration greater than 1 ng/L. The site with the highest concentration, 35015A, yielded two other samples with concentrations lower than 0.1 ng/L. There was no clear difference in $MeHg_{diss}$ concentrations by coal type, but there was a tendency for landfill leachate to yield higher concentrations than impoundment leachate.

Methylated vs. Inorganic Mercury

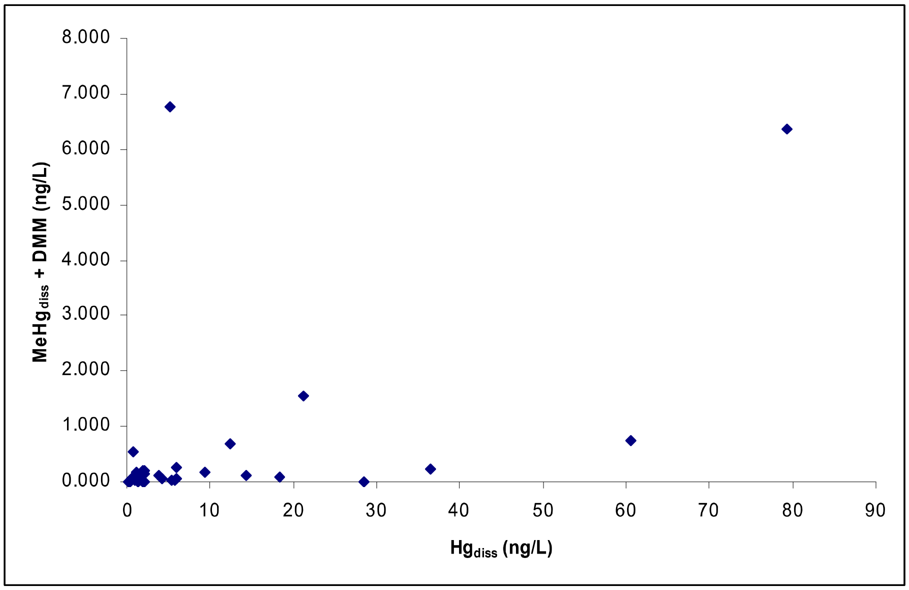
The relative methyl mercury fraction of the total mercury concentration was calculated as:

$$f(MeHg)\ [\%] = 100 \bullet [MeHg_{diss} + DMM)]/Hg_{diss}$$

DMM was added to the $MeHg_{diss}$ concentrations, because it is likely that any DMM present in the collected MeHg samples would have been volatilized by the time the samples were analyzed. There was no apparent correlation between the concentrations of total mercury and methylated mercury compounds (Figure 5-10). Furthermore, methylated mercury compounds constitute only a small fraction of the total mercury concentration in the studied waters, usually less than 5 percent (Figure 5-10). This is in agreement with most previous environmental mercury speciation studies. Only samples 006 and 031 had more than 15.2 percent $MeHg_{diss}$. Sample 006 had extremely low (<1 ng/L) Hg_{diss} and $MeHg_{diss}$ concentrations, while the $MeHg_{diss}$ concentration in sample 031 is suspect because: 1) it is higher than the total mercury (Hg_{diss}) concentration; and 2) it is two orders of magnitude higher than in two other samples (030 and 032) collected at that site on the same day (Table 5-7).

Dissolved vs. Particulate Mercury

Particulate mercury (Hg_{part} and $MeHg_{part}$) is a measure of the mercury on colloids in the water, which accumulate on the filter during sampling. As such, the particulate concentrations are dependent both on the mass of mercury on the particles and the mass of solids collected on the filters. It is of interest because mercury bound to colloids, which can move with groundwater, may be transported more quickly than mercury dissolved in water, which may sorb to the soil under the pH range typical of most groundwater.



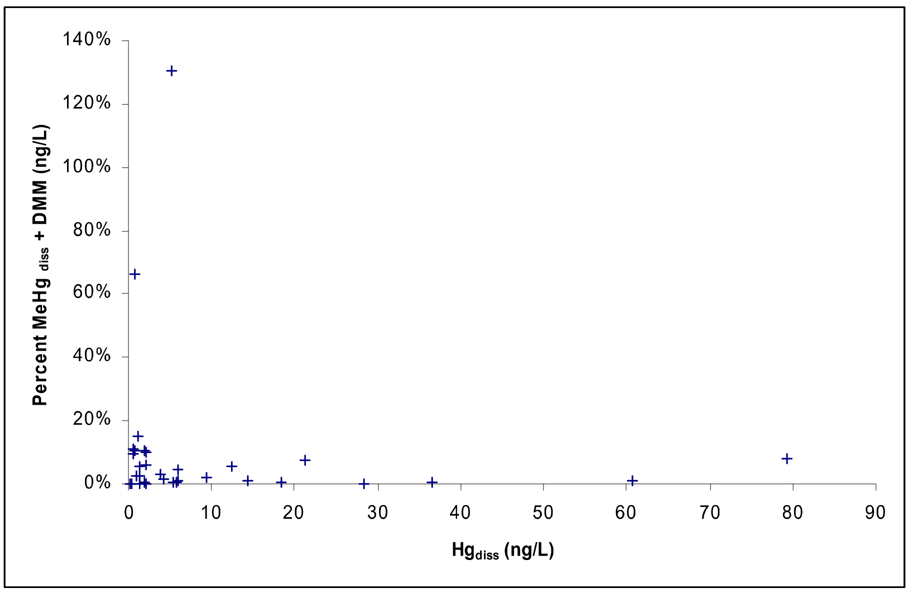


Figure 5-10
Comparison of Organic and Inorganic Mercury Concentrations

The Hg_{part} concentrations in the field leachate samples were low, ranging from <1 to 254 ng/L (Table 5-7). The highest concentration (sample 002) was obtained from a lysimeter at Site 50213, where subbituminous fly ash was managed. A second lysimeter at the same site had a particulate concentration of 26 ng/L. Conversely, the Hg_{diss} concentration associated with these two samples did not exhibit the variability of the particulate concentrations. There was no overall relationship between Hg_{part} and Hg_{diss} concentration (Figure 5-11), nor was there a relationship between $MeHg_{part}$ and $MeHg_{diss}$ (Figure 5-12).

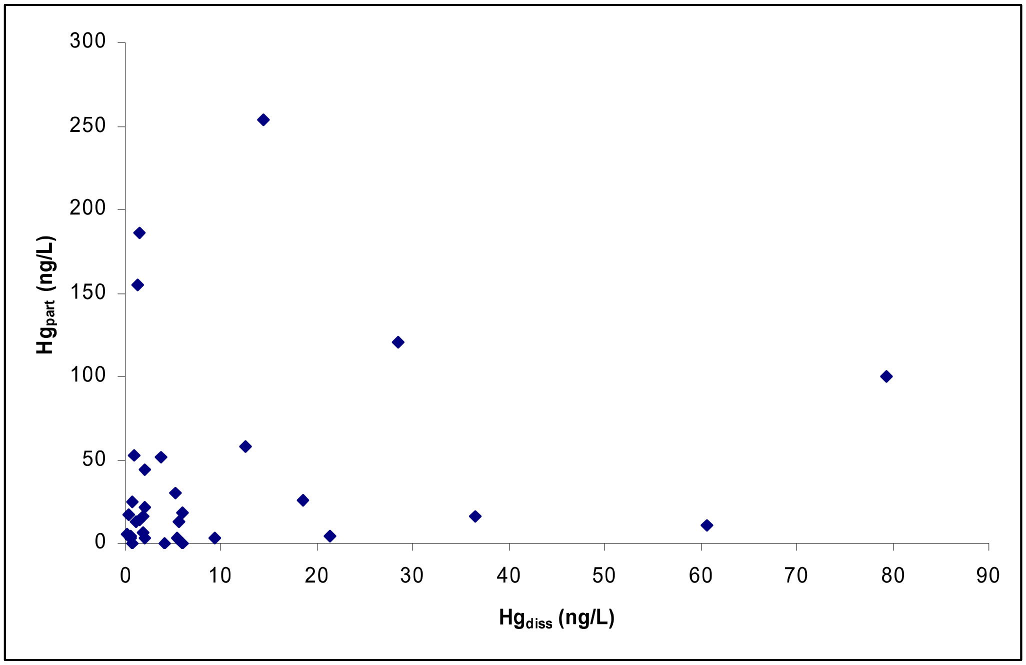


Figure 5-11
Dissolved versus Particulate Mercury Concentrations

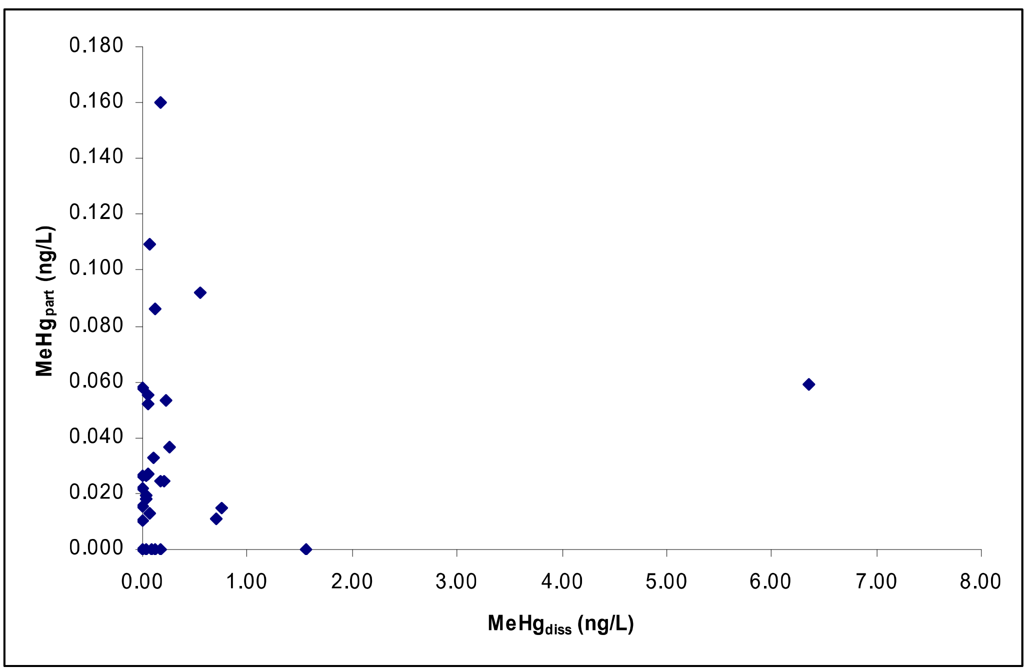


Figure 5-12
Dissolved versus Particulate Methyl Mercury Concentrations

6
CONCLUSIONS

The following conclusions are based on 81 field leachate samples collected at 29 CCP management sites. Due to their unique characteristics, coal ash leachate (67 samples) and FGD leachate (14 samples) were treated separately.

Chemical Composition of Coal Ash Field Leachate Samples

- Most leachate samples were moderately to strongly oxidizing and moderately to strongly alkaline. The subbituminous/lignite ash samples had higher median pH (10.0) than bituminous ash (6.9). Several samples with relatively low Eh and pH were collected from impoundments.
- The anion chemistry of coal ash leachate samples is dominated by sulfate. The median concentration of this constituent was 339 mg/L; this was the only constituent in the leachate with a median concentration greater than 100 mg/L.
- Major cation chemistry was strongly influenced by the type of coal burned at the power plant. Ash leachate derived from bituminous coal was dominated by calcium and magnesium, while ash leachate derived from subbituminous/lignite coal was dominated by sodium.
- Silica and boron had the highest median concentrations (4,645 and 2,160 µg/L, respectively) of the minor and trace constituents. Median concentrations of strontium, molybdenum, lithium, aluminum, and barium were greater than 100 µg/L. Conversely, median concentrations of chromium, beryllium, thallium, silver, lead, and mercury were lower than 1 µg/L; with silver, beryllium, and lead being rarely detected (detected in 7, 6, and 27 percent of the samples, respectively).
- Most constituents (22 out of the 34 analyzed) had higher concentrations in landfill leachate samples than in impoundment leachate samples.
- Leachate samples derived from bituminous coal ash had higher concentrations of calcium, magnesium, cobalt, lithium, manganese, nickel, antimony, thallium, and zinc than leachate from subbituminous coal ash. Lithium and manganese had concentrations an order of magnitude higher in the bituminous ash leachate samples, while thallium was only detected in leachate from bituminous ash.
- Leachate from subbituminous/lignite coal ash had higher concentrations of carbonates, chloride, sodium, sulfate, aluminum, chromium, copper, and mercury than leachate from bituminous coal. The difference was most notable for aluminum and mercury, where the concentrations were higher by an order of magnitude or more.

Chemical Composition of FGD Leachate Field Samples

- The FGD leachate samples were moderately to strongly oxidizing, and moderately to strongly alkaline. Landfill samples, as a group, were less oxic and more alkaline than impoundment samples, although the lowest Eh value was for an impoundment.
- Concentrations of most major constituents (specifically, calcium, chloride, potassium, sodium, and sulfate) in FGD leachate were higher than in ash leachate. The median sulfate concentration was 1,615 mg/L, and the maximum sulfate concentration was 30,500 mg/L, which was the highest single analytical result returned from the field leachate sampling. The high sulfate concentration was obtained from an impoundment where sluice water is recirculated.
- More than 25 percent of the chloride and sodium concentrations were greater than 1,000 mg/L, and median concentrations of chloride, calcium, potassium, and sodium were greater than 100 mg/L.
- The FGD leachate samples had higher percentages of chloride and potassium than the ash leachate samples.
- Anion concentrations were largely dominated by sulfate. Major cation concentrations (calcium, magnesium, potassium, sodium) were variable, with samples from the same site having different cation chemistry.
- The relative concentrations of minor and trace elements in FGD leachate were somewhat different than in ash leachate. Median concentrations of boron, strontium, and lithium in FGD leachate were a factor of 3 or more higher than in ash leachate, while concentrations of selenium, vanadium, uranium, and thallium in ash leachate were higher than in FGD leachate by a factor of 3 or more.
- Boron (9,605 µg/L), strontium (5,230 µg/L), lithium (3,055 µg/L), and silica (2,480 µg/L) had median concentrations greater than 1,000 µg/L in the FGD field leachate samples. Median concentrations of molybdenum, aluminum, and manganese were greater than 100 µg/L, while median concentrations of chromium, beryllium, thallium, silver, lead, and mercury were lower than 1 µg/L. Silver was not detected in the 14 FGD leachate samples, while beryllium (7 percent detects), chromium (36 percent), iron (29 percent), lead (36 percent), and thallium (14 percent), were usually not detected.

Speciation Analysis in Field Leachate Samples

Arsenic

- Arsenic concentrations in ash leachate ranged from 1.4 to 1,380 µg/L, with a median of 25 µg/L.
- The dominant arsenic species was determined in 43 samples. Most ash leachate samples (37) were dominated by As(V). As(III) was only dominant in four samples from impoundments where bituminous coal ash was managed. Two samples had equal amounts of arsenic species.

- Arsenic concentration in FGD leachate ranged from 11 to 230 μg/L, with a median of 28 μg/L.
- The dominant arsenic species was determined in 6 FGD leachate samples. Two were dominated by As(V), two were dominated by As(III), and two samples had equal amounts of the species.

Selenium

- Selenium concentration in ash leachate ranged from 0.07 to 1,760 μg/L, with a median of 19 μg/L.
- The dominant selenium species was determined in 46 leachate samples. Most ash leachate samples (29) were dominated by Se(IV). Se(VI) was dominant in 17 samples. Se(IV) dominated in impoundment settings when the source coal was bituminous or a mixture of bituminous and subbituminous, while Se(VI) was predominant in landfill settings and when the source coal was subbituminous/lignite. Most samples with relatively high concentration (>80 μg/L) were dominated by Se(VI) while samples with concentrations lower than 50 μg/L were mostly dominated by Se(IV).
- Selenium concentration in FGD leachate ranged from 1.1 to 2,360 μg/L, with a median of 6.2 μg/L.
- The dominant selenium species was determined in 7 FGD leachate samples. Six were dominated by Se(VI), one had similar percentages of both species, and none were dominated by Se(IV).

Chromium

- Chromium concentration in ash leachate ranged from <0.2 to 5,100 μg/L, with a median of 0.60 μg/L.
- The dominant chromium species was determined in 27 ash leachate samples. Most ash leachate samples (24) were dominated by Cr(VI). Cr(III) was dominant in three samples, two of which had acidic pH.
- Chromium concentration in FGD leachate ranged from <0.2 to 53 μg/L, with a median concentration below detection limits.
- The dominant chromium species was determined in three FGD leachate samples, and all three were dominated by Cr(VI).

Mercury

- Mercury concentrations in 22 ash leachate samples were very low, ranging from 0.25 to 61 ng/L, with a median concentration of 3.8 ng/L. Mercury concentrations in 8 FGD leachate samples were also very low, ranging from 0.82 to 79 ng/L, with a median concentration of 8.3 ng/L.
- The organic species of mercury always had low concentration, usually less than 5 percent of the total mercury concentration. Monomethyl mercury concentrations ranged from <0.02 to

6.7 ng/L, with a median concentration of 0.08 ng/L. Dimethyl mercury concentrations ranged from <0.02 to 0.06 ng/L, with a median concentration of <0.02 ng/L. There was no relationship between inorganic and organic mercury concentrations.

- There was no clear relationship between organic mercury concentrations and coal type, although there was a tendency for landfill leachate to yield slightly higher concentrations than impoundment leachate.

Effects of Power Plant Attributes on CCP Leachate Composition

- Power plants that have cyclone boilers and burn petroleum coke produced leachate samples with higher than median concentrations of most elements, and the highest concentrations of cadmium, molybdenum, and vanadium.
- There was no definitive relationship on leachate quality associated with hot-side and cold-side ESPs. Three sites receiving ash from hot-side ESPs were sampled. A landfill yielded the highest concentrations of Co, CO_3, Cr, Cu, Na, Se, and SO_4 of the sampled ash sites. However two impoundments did not show evidence of high concentrations.
- Oil ash was managed with coal ash at one site. The leachate from the ash sampled at this site did not show any evidence of low or high concentration for any elements.
- Most constituents in leachate from the single plant with a spray-dryer FGD system had lower concentration than leachate samples from the wet FGD systems used at other plants.

7
REFERENCES

EPRI, 1994. *Chemical Attenuation Reactions of Selenium*, Palo Alto, CA, Report TR-103535.

EPRI, 2000a. *Environmental Chemistry of Arsenic: A Literature Review*, Palo Alto, CA, Report 100585.

EPRI, 2000b. *Field Evaluation of the Comanagement of Utility Low-Volume Wastes with High-Volume Coal Combustion By-Products: HA Site*. Final Report 1000720, October 2000.

EPRI, 2003a. *Combustion By-Product Environmental Analysis System*, Palo Alto, CA, Report 1005263.

EPRI, 2003b. *Field Evaluation of Comanagement of Utility Low-Volume Wastes with High-Volume By-Products: MO Site*, Palo Alto, CA, Report 1005267.

EPRI, 2004. *Chemical Attenuation Coefficients for Arsenic Species Using Soil Samples from Selected Power Plant Sites*, Palo Alto, CA, Report 1005505.

Gürleyük, H. and D. Wallschläger, 2001. *Determination of Chromium Species Using Suppressed Ion Chromatography-Inductively-Coupled Plasma-Mass Spectrometry*, J. Anal. At. Spectrom. **16**, 926-930.

Hintelmann, H. and N. Ogrinc, 2003. *Determination of Stable Mercury Isotopes by ICP/MS and Their Application in Environmental Studies*, in: Cai, Y. & Braids, O.C. (eds.): Biogeochemistry of environmentally important trace elements, ACS Symposium Series 835, American Chemical Society, 321-338.

Lindberg, S.E., G. Southworth, E.M. Prestbo, D. Wallschläger, M.A. Bogle, and J. Price, 2004. *Gaseous Methyl- and Inorganic Mercury in Landfill Gas from Landfills in Florida, Minnesota, Delaware and California*, accepted for publication in Atmos. Environ.

Puls, R.W and M.J. Barcelona, 1995. *Low-Flow (Minimal Drawdown) Ground-Water Sampling Procedures*, U.S. Environmental Protection Agency, EPA Ground Water Issue, EPA540-S-95-504.

Wallschläger, D. and R. Roehl, 2001. *Determination of Inorganic Selenium Speciation in Waters by Ion Chromatography-Inductively-Coupled Plasma-Mass Spectrometry Using Eluant Elimination with a Membrane Suppressor*, J. Anal. At. Spectrom. **16**, 922-925.

Wallschläger, D., R.T. Wilkin, and R.G. Ford, 2005. *Soluble Arsenic-Thio Species in Sulfidic Waters*, submitted to Environ. Sci. Technol.

A
ANALYTICAL RESULTS

Table A-1
Hydrochemistry and Trace Elements

		001	002	003	004	005	006	007	008	009	010	012	QA-1	013
Chloride	mg/L	86.2	25	11	26	6.5	19	572	371	345	28	9	< 0.01	27.3
Sulfate	mg/L	909	6,690	5,450	1,960	350	1,450	3,150	2,080	10,400	3,830	1,650	0.47	1,700
Sodium	mg/L	443	3,410	2,910	672	93	108	1,330	606	743	1,700	30	0.4	55
Potassium	mg/L	255	80	80	20	< 5	10	80	20	40	118	< 20	< 0.2	75
Magnesium	mg/L	< 1	0.59	0.53	70	15	77	125	23	1,990	8	13	0.10	36
Calcium	mg/L	10	19	9	218	70	528	524	563	577	139	681	0.53	584
TOC	mg/L	13.9	55.1	49.8	43.9	4.5	8.1	20.5	16.2	9.9	5.3	1.9	0.4 (a)	6.3
TIC	mg/L	6.9	32.2	63.1	29.7	11.9	17.5	2.4	2.7	1.7	1.7	2.0	1.56	16.6
Temperature	°C	20.2	21.5	15.4	14.9	21.3	18.7	17.6	26.9	25.6	17.3	22.6	n/a	21.3
Spec. Cond.	mS/cm	3.5	12.8	11.2	3.8	0.8	2.9	8.3	4.8	13.0	7.7	2.7	n/a	2.9
Diss. Oxygen	% sat.	0.1	0.2	0.2	0.2	0.2	0.4	0.2	0.3	0.3	14	5	n/a	4
pH	pH	11.6	10.0	10.3	9.3	7.4	8.0	6.2	8.4	7.4	11.2	9.4	n/a	8.2
ORP (corr.)	mV	209	276	271	276	411	341	356	1	342	111	245	n/a	102
Lithium	ug/L	2,460	< 20	< 20	< 20	< 20	< 20	170	< 20	2,720	< 20	80	< 20	100
Beryllium	ug/L	< 1	< 1	< 1	< 1	< 1	< 1	< 1	< 1	< 1	< 1	< 1	< 4	< 1
Boron	ug/L	2,120	18,400	31,900	10,800	1,410	15,600	81,500	49,000	98,500	14,000	93,400	< 50	112,000
Aluminum	ug/L	18,100	2,680	17,500	< 30	< 30	< 30	610	890	190	980	530	< 30	< 30
Silicon	ug/L	6,900	5,800	1,200	6,100	6,400	2,600	10,500	400	12,700	9,900	1,500	< 100	18,500
Vanadium	ug/L	373	1,070	635	45	< 2	4	15	< 2	18	5,020	195	< 2	4
Manganese	ug/L	< 4	7	< 4	751	577	< 4	704	113	564	< 4	22	< 4	2,560
Iron	ug/L	< 50	< 50	< 50	< 50	< 50	< 50	1,200	< 50	< 50	< 50	< 50	< 50	14,700
Cobalt	ug/L	< 1	133	9	< 1	< 1	< 1	6	< 1	78	< 1	< 1	< 1	7
Nickel	ug/L	< 3	75	8	14	4	4	597	5	463	8	4	< 20	15
Copper	ug/L	11	494	62	6	3	4	14	44	7	15	< 3	< 3	< 3
Zinc	ug/L	< 5	< 5	< 5	< 5	6	19	23	< 5	34	12	12	< 30	45
Strontium	ug/L	800	60	< 30	930	80	9,140	16,900	14,900	11,700	3,900	2,250	< 30	1,260
Molybdenum	ug/L	9,740	5,720	6,200	1,200	440	310	60,800	570	320	25,400	740	< 30	100
Silver	ug/L	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 1	0.2
Cadmium	ug/L	17.7	8.8	7.6	1.9	0.8	0.7	12.3	11.8	4.2	51.9	1.5	< 2	0.4
Antimony	ug/L	0.9	0.8	0.7	0.6	< 0.3	4.7	2.8	0.7	4.6	1.0	6.7	< 3	0.7
Barium	ug/L	50	< 30	< 30	110	40	70	50	< 30	90	50	40	< 30	< 30
Thallium	ug/L	< 0.1	< 0.5	< 0.5	< 0.1	< 0.1	< 0.1	< 0.5	< 0.1	2.9	< 0.1	< 0.1	<0.01	0.6
Lead	ug/L	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	3.5	0.3	< 0.2	0.3	< 0.2	< 1	< 0.2
Uranium	ug/L	< 0.2	0.2	9.8	1.3	< 0.2	10.4	0.7	< 0.2	0.7	0.3	1.8	< 1	3.3

Table A-1
Hydrochemistry and Trace Elements (continued)

		013D	014	QA-2	015	016	SX-1	017	018	019	020	HN-1	HN-2	021
Chloride	mg/L	27.5	32.8	0.05	25.3	54.8	22.2	72.0	63.4	84.8	75.9	29.2	45.4	18.0
Sulfate	mg/L	1,610	1,370	0.40	782	910	1,530	91.4	339	124	131	1,260	810	193
Sodium	mg/L	56	17	0.9	60	731	52	53	57	56	54	72	53	31
Potassium	mg/L	74	26	< 0.2	20	229	38	8	9	6	6	277	48	11
Magnesium	mg/L	39	7	0.63	33	20	7	21	36	28	23	3	21	13
Calcium	mg/L	544	591	1.34	255	15	529	46	231	81	43	302	291	48
TOC	mg/L	6.2	3.9	0.6 (a)	5.3	24.0	16.6	6.7	14.2	6.0	0.4 (a)	21.5	22.5	1.2 (a)
TIC	mg/L	16.7	35.1	1.47	15.4	5.60	11.3	22.4	115.0	48.7	24.8	2.48	2.94	8.03
Temperature	°C	n/a	20.5	32	31.7	30.6	n/a	29.7	18.3	35.5	29.6	n/a	n/a	20.8
Spec. Cond.	mS/cm	n/a	2.6	0.0	1.6	5.1	n/a	0.7	1.6	1.0	0.7	n/a	n/a	0.6
Diss. Oxygen	% sat.	n/a	5.5	3	3.7	2.9	n/a	1.6	2.9	3.4	4.5	n/a	n/a	29.5
pH	pH	n/a	9.3	5.3	9.3	11.7	n/a	8.8	7.4	8.0	8.9	n/a	n/a	7.9
ORP (corr.)	mV	n/a	240	515	339	124	n/a	289	94	296	303	n/a	n/a	245
Lithium	ug/L	n/a	110	< 20	100	60	50	< 20	30	< 20	< 20	1,060	60	310
Beryllium	ug/L	n/a	< 1	< 4	< 1	< 1	< 0.8	< 1	< 1	< 1	< 1	< 0.8	< 0.8	< 0.8
Boron	ug/L	n/a	54,900	< 50	3,890	109,000	24,200	860	26,300	470	700	2,350	42,700	850
Aluminum	ug/L	n/a	300	< 30	100	44,400	< 150	1,920	80	4,190	730	< 150	< 150	80
Silicon	ug/L	n/a	1,500	< 100	8,800	19,000	2,400	3,000	10,300	3,400	2,200	3,400	3,300	5,400
Vanadium	ug/L	n/a	36	< 2	550	1,230	11	16	6	10	17	206	41	217
Manganese	ug/L	n/a	25	< 4	< 4	8	52	< 4	4,170	14	< 4	< 4	< 4	67
Iron	ug/L	n/a	< 50	< 50	< 50	1,530	< 50	< 50	3,190	< 50	< 50	< 50	< 50	300
Cobalt	ug/L	n/a	< 1	< 1	3	2	< 1	< 1	2	1	< 1	< 1	< 1	< 1
Nickel	ug/L	n/a	5	< 20	16	128	< 3	5	8	7	4	10	7	4
Copper	ug/L	n/a	< 3	< 3	< 3	21	< 3	12	35	8	7	7	5	6
Zinc	ug/L	n/a	40	< 30	< 5	130	25	8	7	9	11	16	< 5	6
Strontium	ug/L	n/a	3,140	< 30	4,300	1,200	2,690	530	640	580	720	930	680	730
Molybdenum	ug/L	n/a	6,030	< 30	420	39,600	3,010	80	100	< 30	< 30	1,910	500	710
Silver	ug/L	n/a	< 0.2	< 1	< 0.2	< 0.2	1.1	< 0.2	< 0.2	< 0.2	< 0.2	2.0	0.8	< 0.2
Cadmium	ug/L	n/a	21.2	< 2	1.0	64.7	14.0	< 0.3	0.4	< 0.3	< 0.3	8.5	1.5	1.2
Antimony	ug/L	n/a	2.0	< 3	1.4	2.4	1.3	0.5	0.6	0.6	0.5	3.4	1.6	31.4
Barium	ug/L	n/a	40	< 30	350	140	80	140	100	350	220	80	60	240
Thallium	ug/L	n/a	< 0.1	<0.01	2.5	0.3	3.1	< 0.1	0.1	< 0.1	< 0.1	< 0.5	< 0.5	1.5
Lead	ug/L	n/a	< 0.2	< 1	< 0.2	4.6	0.8	< 0.2	< 0.2	0.21	< 0.2	0.4	0.4	< 0.2
Uranium	ug/L	n/a	1.1	< 1	3.7	0.7	12.5	1.1	4.6	1.2	1.2	< 0.2	0.7	2.7

Table A-1
Hydrochemistry and Trace Elements (continued)

		022	023	024	025	026	027	028	029	030	031	032	034	035
Chloride	mg/L	17.8	28.4	23	15.3	17.9	932	1,260	1,200	33.8	87	55.9	< 0.01	< 0.01
Sulfate	mg/L	217	248	2,350	845	219	1,620	1,610	1,510	948	1,830	386	< 0.05	< 0.05
Sodium	mg/L	42	33	188	80	43	285	341	297	25	60	32	< 0.1	< 0.1
Potassium	mg/L	9	8	170	40	9	470	580	500	20	50	10	< 0.2	< 0.2
Magnesium	mg/L	14	28	203	82	14	3	10	4	39	35	50	< 0.05	< 0.05
Calcium	mg/L	43	79	405	235	43	671	722	730	332	665	124	< 0.05	< 0.05
TOC	mg/L	0.5 (a)	2.2	1.3 (a)	4.1	0.9 (a)	1.9	0.5 (a)	1.4 (a)	0.5 (a)	11.0	0.6 (a)	0.1 (a)	0.1 (a)
TIC	mg/L	2.49	27.3	54.5	79.9	1.04	1.00	3.25	0.95	10.4	1.53	12.9	0.43 (a)	0.46 (a)
Temperature	°C	21.6	17.4	15.6	15.2	22.2	16.3	16.1	15.5	15.4	15.6	13.9	23.0	23.6
Spec. Cond.	mS/cm	0.6	0.7	4.0	2.0	0.6	5.6	6.6	6.1	1.8	3.0	1.0	0.003	0.002
Diss. Oxygen	% sat.	39.1	17.6	16	15.8	22.4	11.8	10.6	17.1	29.6	6.1	14.5	84.7	71.1
pH	pH	7.1	7.0	7.0	6.5	7.2	10.0	9.0	9.9	8.5	8.5	7.8	5.67	5.40
ORP (corr.)	mV	307	287	268	264	319	71	220	121	308	-41	295	335	306
Lithium	ug/L	360	120	18,600	3,430	320	6,920	5,890	6,260	100	410	240	<0.1	<0.1
Beryllium	ug/L	< 0.8	< 0.8	< 0.8	< 0.8	< 0.8	< 0.8	< 0.8	< 0.8	< 0.8	< 0.8	< 0.8	<0.04	<0.04
Boron	ug/L	430	1,970	22,400	11,100	420	1,450	3,260	2,820	3,280	7,610	2,210	0.9	1.4
Aluminum	ug/L	40	90	< 30	< 30	40	190	< 30	130	190	140	< 30	0.4	0.8
Silicon	ug/L	3,600	3,400	9,400	5,400	3,300	3,000	1,900	2,000	700	3,700	5,400	6.7	18.4
Vanadium	ug/L	70	427	4	< 2	63	< 2	< 2	4	18	4	12	0.10	0.06
Manganese	ug/L	104	149	3,650	4,110	104	18	202	62	41	269	92	<0.02	0.05
Iron	ug/L	< 50	120	80	90	< 50	< 50	< 50	< 50	< 50	< 50	< 50	0.4	0.4
Cobalt	ug/L	8	2	96	8	8	< 1	< 1	< 1	< 1	1	3	<0.02	<0.02
Nickel	ug/L	19	9	167	6	21	3	< 3	< 3	3	8	17	0.08	0.09
Copper	ug/L	8	8	< 3	< 3	< 3	< 3	3	< 3	16	< 3	24	0.46	0.47
Zinc	ug/L	21	11	148	< 5	14	< 5	12	< 5	90	13	15	<0.3	0.7
Strontium	ug/L	430	1,990	6,460	2,290	400	3,520	3,980	4,300	990	2,480	360	<0.4	<0.4
Molybdenum	ug/L	410	500	3,870	2,420	400	180	350	300	140	210	120	<0.1	<0.1
Silver	ug/L	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	<0.02	<0.02
Cadmium	ug/L	1.1	1.0	9.1	5.1	1.6	0.5	0.8	0.6	< 0.3	0.5	1.2	<0.02	<0.02
Antimony	ug/L	24.3	59.1	4.9	0.5	23.5	< 0.3	< 0.3	< 0.3	5.0	2.7	3.8	<0.02	<0.02
Barium	ug/L	190	110	50	50	190	60	60	80	80	60	160	<0.2	<0.2
Thallium	ug/L	12.0	1.3	1.5	0.4	12.3	< 0.5	< 0.5	< 0.5	3.4	< 0.1	17.6	<0.02	<0.02
Lead	ug/L	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	0.03	<0.02
Uranium	ug/L	< 0.2	60.8	13.0	19.3	< 0.2	< 0.2	< 0.2	< 0.2	5.3	2.0	1.0	<0.01	<0.01

Table A-1
Hydrochemistry and Trace Elements (continued)

		036	037	038	039	042	043	044	044D	049	050	051	052	053
Chloride	mg/L	< 0.01	8.8	9.7	9.4	9.7	7.1	9.8	9.1	9.8	< 0.01	5.3	7.6	8.1
Sulfate	mg/L	< 0.05	123	121	101	57	111	70	70	53	< 0.05	111	128	176
Sodium	mg/L	< 0.1	3.8	3.9	4.7	8.6	8.5	8.3	8.3	7.0	0.1	11.8	6.8	5.6
Potassium	mg/L	< 0.2	2.2	2.3	5.3	5.2	7.0	5.0	5.0	4.0	< 0.2	13.6	11.1	9.2
Magnesium	mg/L	< 0.05	6.91	6.61	3.08	2.06	2.58	2.66	2.67	2.53	< 0.05	1.81	0.08	0.12
Calcium	mg/L	< 0.05	45.8	45.3	36.1	12.4	19.9	15.4	15.5	13.2	0.09	14.4	58.4	69.5
TOC	mg/L	0.1 (a)	< 0.09	< 0.09	< 0.09	< 0.09	< 0.09	< 0.09	< 0.09	< 0.09	0.1 (a)	< 0.09	0.8 (a)	0.7 (a)
TIC	mg/L	0.48 (a)	10.4	10.5	6.66	2.01	1.03	0.75 (a)	0.68 (a)	2.18	0.44 (a)	0.92	3.30	4.96
Temperature	°C	23.6	22	22.7	24.2	29.4	32	32	31.5	25.8	24.5	26.5	27.1	26.7
Spec. Cond.	mS/cm	0.001	0.379	0.381	0.317	0.178	0.293	0.209	0.210	0.174	0.009	0.287	0.588	0.468
Diss. Oxygen	% sat.	77	35	27.6	33.5	84.1	75.7	67.9	80.2	77.6	72	82.4	56	40.6
pH	pH	5.66	7.05	7.04	6.98	5.79	4.26	5.97	6.03	5.97	4.92	4.35	10.59	8.92
ORP (corr.)	mV	299	192	163	184	283	388	285	289	290	300	387	211	212
Lithium	ug/L	<0.1	82	81	125	179	239	146	145	99	<0.1	520	561	595
Beryllium	ug/L	<0.04	<0.4	<0.4	<0.4	1.6	8.6	0.8	1.3	<0.4	<0.04	5.2	<0.4	<0.4
Boron	ug/L	3.1	1390	1240	917	426	838	429	489	265	43.3	272	4620	7370
Aluminum	ug/L	1.0	15	14	6	148	3730	66	72	14	3.5	2150	15100	2010
Silicon	ug/L	21.5	7960	7660	7000	4700	5780	4730	5100	4670	15.3	5840	1890	1030
Vanadium	ug/L	0.10	13.8	6.9	2.6	70.8	35.6	9.6	9.5	5.6	0.21	4.7	754.4	62.4
Manganese	ug/L	0.67	248	244	261	42.7	77.5	86.1	88.6	79.4	23.4	113	0.4	5.9
Iron	ug/L	1.0	921	1700	1070	6	722	18	28	7	8.2	3240	16	30
Cobalt	ug/L	<0.02	1.7	0.7	<0.2	11.5	21.6	8.7	9.0	5.2	0.05	18.9	<0.2	0.2
Nickel	ug/L	0.45	7.2	4.2	2.4	37.8	71.9	26.7	27.5	13.6	2.98	58.2	<0.6	1.5
Copper	ug/L	0.55	0.5	1.0	<0.4	8.7	152	12.0	11.2	1.9	1.13	452	1.8	8.4
Zinc	ug/L	0.7	<3	<3	<3	58.1	80.4	35.6	32.9	18.3	5.6	74.6	<3	5.7
Strontium	ug/L	<0.4	1350	1360	1120	170	247	272	262	209	<0.4	806	5150	5610
Molybdenum	ug/L	<0.1	1110	1060	287	127	35	54	54	60	0.2	8	246	360
Silver	ug/L	<0.02	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.02	<0.2	<0.2	<0.2
Cadmium	ug/L	<0.02	4.6	4.1	1.2	1.3	1.9	0.7	0.8	0.5	<0.02	2.4	0.8	2.3
Antimony	ug/L	<0.02	4.6	2.4	0.3	13.9	17.8	8.7	8.8	7.1	<0.02	5.9	14.4	2.6
Barium	ug/L	<0.2	125	169	77	75	131	180	181	195	<0.2	545	250	87
Thallium	ug/L	<0.02	<0.2	<0.2	<0.2	0.7	4.2	1.6	1.5	0.7	<0.02	6.3	0.4	0.3
Lead	ug/L	<0.02	<0.1	<0.1	<0.1	0.2	1.9	0.2	<0.1	<0.1	<0.02	8.0	<0.1	0.5
Uranium	ug/L	<0.01	0.2	0.1	<0.1	<0.1	0.5	<0.1	<0.1	<0.1	<0.01	1.0	0.1	1.7

Table A-1
Hydrochemistry and Trace Elements (continued)

		057	059	059D	060	061	062	064	069	070	070D	077	078	079
Chloride	mg/L	5.6	4.5	4.6	< 0.01	7.1	15.8	5.0	7.3	12.1	9.7	< 0.01	< 0.01	77.2
Sulfate	mg/L	52	55	55	< 0.05	61	117	150	45	50	51	< 0.05	< 0.05	315
Sodium	mg/L	8.1	8.5	8.5	< 0.1	9.5	11.4	7.3	6.0	10.8	10.8	< 0.1	< 0.1	63
Potassium	mg/L	5.8	6.4	6.4	< 0.2	6.4	9.6	9.4	3.6	5.0	4.9	< 0.2	< 0.2	13
Magnesium	mg/L	1.53	1.37	1.43	< 0.05	4.97	0.11	1.49	2.16	1.78	1.81	< 0.05	< 0.05	19.5
Calcium	mg/L	16.8	16.8	16.5	0.20	55.1	76.5	58.1	19.0	26.0	26.3	< 0.05	< 0.05	95.3
TOC	mg/L	< 0.09	< 0.09	< 0.09	< 0.09	< 0.09	< 0.09	< 0.09	< 0.09	< 0.09	< 0.09	0.4 (a)	0.2 (a)	0.3 (a)
TIC	mg/L	6.02	5.07	4.99	0.43 (a)	38.3	3.98	3.92	6.04	9.44	9.55	0.34 (a)	0.28 (a)	20.6
Temperature	°C	28.5	31.2	n/a	25.7	27.6	29	30	27.7	29.4	28.9	28.6	27.0	19.5
Spec. Cond.	mS/cm	0.189	0.195	n/a	0.003	0.433	0.765	0.455	0.182	0.244	0.247	0.001	0.002	1.076
Diss. Oxygen	% sat.	89.2	165.1	n/a	90.2	65.3	37.9	67.7	63.5	67.9	68.3	64.3	74.3	28.0
pH	pH	7.66	9.04	n/a	5.4	7.25	10.95	10.12	7.57	8.91	9.1	5.07	5.58	6.75
ORP (corr.)	mV	n/a	409	na	277	140	196	214	220	223	220	263	236	114
Lithium	ug/L	267	293	288	<0.1	155	243	430	140	160	167	<0.05	<0.05	134
Beryllium	ug/L	<0.4	<0.4	<0.4	<0.04	<0.4	<0.4	<0.4	<0.4	<0.4	<0.4	<0.01	<0.01	<0.2
Boron	ug/L	300	351	309	1.2	2600	494	476	231	207	236	7.1	6.8	1110
Aluminum	ug/L	111	356	366	1.8	58	3900	2310	29	468	519	2.3	2.3	<2
Silicon	ug/L	5120	5010	5190	8.6	11100	6870	4760	7450	7190	6920	509	513	10100
Vanadium	ug/L	31.3	34.4	34.6	0.18	5.6	176.9	229.6	61.3	93.1	94.2	0.22	0.21	0.4
Manganese	ug/L	1.6	0.6	0.8	0.04	395	<0.2	<0.2	22.0	0.4	0.7	7.50	4.84	190
Iron	ug/L	6	26	25	0.7	2170	17	13	<5	27	46	12.9	2.28	25600
Cobalt	ug/L	<0.2	<0.2	<0.2	<0.02	0.3	<0.2	<0.2	1.3	<0.2	<0.2	0.007	0.003	0.18
Nickel	ug/L	2.3	1.7	1.4	<0.6	4.0	<0.6	0.9	5.4	0.6	<0.6	<0.03	<0.03	<0.6
Copper	ug/L	1.3	2.0	1.8	0.5	<0.4	1.2	0.5	0.7	1.8	2.1	0.30	0.27	<0.2
Zinc	ug/L	<3	4.0	<3	0.4	<3	<3	<3	<3	<3	<3	0.4	0.4	1.5
Strontium	ug/L	545	547	576	<0.4	1840	1010	478	340	258	263	3.37	3.39	2190
Molybdenum	ug/L	62	63	61	<0.1	95	173	217	78	61	63	<0.02	<0.02	135
Silver	ug/L	<0.2	<0.2	<0.2	<0.02	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.01	<0.01	<0.2
Cadmium	ug/L	<0.2	0.4	<0.2	<0.02	0.3	0.6	0.6	0.3	<0.2	<0.2	<0.01	<0.01	<0.2
Antimony	ug/L	6.2	5.6	5.5	<0.02	0.7	8.2	27.4	9.5	7.6	7.9	<0.005	0.005	<0.1
Barium	ug/L	182	171	166	<0.2	226	194	319	156	124	132	<0.1	<0.1	99.2
Thallium	ug/L	1.0	0.9	0.9	<0.02	0.5	<0.2	0.3	1.0	0.4	0.4	<0.005	<0.005	<0.1
Lead	ug/L	<0.1	<0.1	<0.1	<0.02	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	0.04	0.02	<0.1
Uranium	ug/L	0.5	1.2	1.3	<0.01	1.4	<0.1	0.7	0.3	2.2	2.2	<0.001	<0.001	1.91

Table A-1
Hydrochemistry and Trace Elements (continued)

		079D	082	083	084	088	089	090	091	092	TEB	094	095	096 (1)
Chloride	mg/L	77.9	72.0	68.4	67.9	< 0.01	0.37	11.8	5.35	4.67	0.22	0.06	0.04	92.4
Sulfate	mg/L	315	174	92.8	135	< 0.05	1.50	324	393	448	0.65	< 0.05	< 0.05	2,850
Sodium	mg/L	63	68	45	38	0.7	0.9	182	277	109	0.6	< 0.1	< 0.1	1,560
Potassium	mg/L	14	5	4	6	< 0.2	0.2	113	84	67	1.2	< 0.2	< 0.2	74
Magnesium	mg/L	19.4	19.1	12.6	30.8	< 0.05	0.35	0.15	< 0.05	< 0.05	< 0.05	< 0.05	< 0.05	9
Calcium	mg/L	98.0	79.1	34.4	105	< 0.05	0.72	11.9	2.22	287	1.30	< 0.05	< 0.05	9
TOC	mg/L	0.8 (a)	2.6	4.7	< 0.09	0.4 (a)	0.5 (a)	12.8	4.3	3.4	0.4 (a)	0.3 (a)	0.3 (a)	49.8
TIC	mg/L	19.7	35.9	11.9	60.5	0.28 (a)	1.20	13.8	7.62	0.85	1.37	0.21 (a)	0.23 (a)	128
Temperature	°C	18.0	30.2	25.9	19.2	n/a	n/a	17.2	16.8	15.9	n/a	12.4	13.7	16.1
Spec. Cond.	mS/cm	1.068	0.911	0.547	0.927	n/a	n/a	1.59	2.33	1.427	n/a	0.002	0.005	7.295
Diss. Oxygen	% sat.	21.0	65.1	100.0	40.7	n/a	n/a	n/a	n/a	n/a	n/a	84	73.3	67
pH	pH	6.84	8.64	9.36	7.78	n/a	n/a	10.86	11.52	11.17	n/a	6.2	5.44	7.29
ORP (corr.)	mV	87	241	217	198	n/a	n/a	246	288	346	n/a	227	261	223
Lithium	ug/L	134	60	27	139	<0.05	4	2	5	11	1.25	<0.05	<0.05	5
Beryllium	ug/L	<0.2	<0.2	<0.2	<0.2	<0.01	0.011	<0.2	<0.2	<0.2	<0.01	<0.01	<0.01	<0.2
Boron	ug/L	1200	442	1020	4310	89.6	215	1800	495	1080	240	1.1	0.7	5650
Aluminum	ug/L	<2	1080	2030	41	1.1	92.9	19900	30000	5140	38.6	7.8	1.3	1700
Silicon	ug/L	9970	4210	1050	2300	3780	6740	4200	4390	2460	7100	11.5	9.2	1400
Vanadium	ug/L	0.5	103	49.3	11.5	0.09	1.23	365	562	156	1.07	0.13	0.15	473
Manganese	ug/L	191	2.0	1.0	91.1	1.50	9.97	0.9	<0.1	<0.1	6.1	0.5	0.22	1.5
Iron	ug/L	25200	<3	<3	62.0	52.7	271	29.7	<8	<8	140	5.4	0.51	25.3
Cobalt	ug/L	0.18	0.80	0.53	0.81	<0.001	0.17	0.12	0.04	0.40	0.301	<0.001	<0.001	3.27
Nickel	ug/L	<0.6	3.6	4.4	4.6	<0.05	6.29	14	4	<1	12	0.06	0.08	7
Copper	ug/L	1.4	3.8	2.1	<0.2	0.33	2.02	1.4	0.5	1.4	1.2	1.4	2.5	30.0
Zinc	ug/L	2.5	<2	3.0	<2	<0.1	6.9	<2	<2	<2	2.9	3.3	4.3	<2
Strontium	ug/L	2140	828	1010	2520	9.48	82.6	830	1610	11100	135	0.31	0.11	311
Molybdenum	ug/L	132	21.9	27.7	283	0.04	0.83	1890	1390	658	0.75	0.02	<0.01	4510
Silver	ug/L	<0.2	<0.2	<0.2	<0.2	<0.01	<0.01	<0.2	<0.2	<0.2	<0.01	<0.01	<0.01	<0.2
Cadmium	ug/L	<0.2	<0.2	0.3	0.7	<0.005	0.037	6.1	4.8	2.8	0.04	0.02	0.01	15.0
Antimony	ug/L	<0.1	1.1	2.9	1.1	0.021	0.074	2.3	0.5	0.2	0.082	0.007	<0.005	0.8
Barium	ug/L	93.6	434	294	176	<0.2	10.7	89.3	259	657	29.7	0.6	<0.2	20
Thallium	ug/L	<0.1	0.5	<0.1	0.4	<0.005	<0.005	<0.1	<0.1	<0.1	0.021	<0.005	<0.005	<0.1
Lead	ug/L	<0.1	<0.1	0.2	<0.1	0.01	0.17	<0.1	<0.1	<0.1	0.03	0.09	0.06	<0.1
Uranium	ug/L	1.95	2.66	1.23	26.8	<0.0005	0.17	0.39	<0.01	0.01	0.02	<0.0005	<0.0005	5.53

Table A-1
Hydrochemistry and Trace Elements (continued)

		096D (1)	097	098	099	100	101	102	103	104	105	106	106D	107
Chloride	mg/L	92.5	91.7	38.7	27.3	0.07	37.2	73.0	0.01	0.02	1,080	859	715	2,330
Sulfate	mg/L	2,870	2,870	1,800	1,510	0.08	1,610	2,410	< 0.05	0.12	10,200	4,710	4,430	30,500
Sodium	mg/L	1,560	1,560	837	651	1.3	117	455	0.2	0.1	3,270	2,310	2,210	4,630
Potassium	mg/L	77	73	31	6	< 0.2	23	219	< 0.2	< 0.2	380	350	350	500
Magnesium	mg/L	10	7	44	16	< 0.05	188	69	< 0.05	< 0.05	1,000	< 0.05	< 0.05	5,810
Calcium	mg/L	11	6	52	73	0.17	392	431	< 0.05	< 0.05	600	234	228	570
TOC	mg/L	50.1	48.7	56.8	14.7	0.4 (a)	4.6	3.3	0.1 (a)	0.1 (a)	33.1	19.1	18.6	50.1
TIC	mg/L	128	105	39.7	14.1	0.28 (a)	27.8	24.3	0.16 (a)	0.27 (a)	7.88	4.27	4.36	1.85
Temperature	°C	16.5	17.4	12.9	15.1	13.4	16.9	15.8	n/a	6.6	9.94	19.0	19.0	19.18
Spec. Cond.	mS/cm	7.379	7.340	4.282	3.451	0.003	3.363	4.915	n/a	0.072	18.85	11.56	11.56	26.14
Diss. Oxygen	% sat.	61.1	69.4	27.5	37	81.1	86.1	94.7	n/a	64.5	36	95	95	2
pH	pH	7.71	9.35	8.58	7.91	5.94	6.74	7.41	n/a	9.54	8.99	11.96	11.96	6.83
ORP (corr.)	mV	224	206	39	103	238	213	222	n/a	288	271	18	18	230
Lithium	ug/L	5	4	63	<1	<0.05	431	6940	<0.05	<0.05	1050	130	132	3390
Beryllium	ug/L	<0.2	<0.2	<0.2	<0.2	<0.01	<0.2	<0.2	<0.01	<0.01	<0.2	<0.2	<0.2	1
Boron	ug/L	5950	6080	11700	2590	0.8	89500	23700	<0.1	0.2	26800	7310	7460	50200
Aluminum	ug/L	1700	4300	117	42	3	52	<2	2.3	2.7	31	608	618	708
Silicon	ug/L	1340	1540	4620	4410	25.7	6750	3940	5.9	17.9	2280	21000	22000	45400
Vanadium	ug/L	477	500	159	3.8	0.10	0.8	44.3	0.25	0.33	1.8	400	403	103
Manganese	ug/L	1.4	1.5	59.8	1230	0.39	1420	72.3	0.33	2.32	473	<0.1	0.1	1170
Iron	ug/L	20.1	46.3	<8	126	0.52	12.1	<8	2.05	1.36	4.7	4.6	6.6	52.4
Cobalt	ug/L	3.31	3.28	0.88	0.29	<0.001	9.19	0.07	<0.001	0.008	0.09	0.11	0.07	13.0
Nickel	ug/L	7	8	9	2	0.18	31	3	<0.03	0.25	3.3	7.5	8.0	153
Copper	ug/L	29.9	42.8	1.7	1.5	1.60	2.8	1.6	0.51	0.55	0.4	0.6	0.5	2
Zinc	ug/L	<2	<2	<2	<2	5	86	<2	0.2	0.7	<2	<2	<2	68
Strontium	ug/L	293	303	1700	93	0.72	1320	10300	0.67	3.88	6980	9730	10000	1500
Molybdenum	ug/L	4450	4480	2580	2070	0.05	751	9630	<0.04	<0.04	164	3520	3560	1320
Silver	ug/L	<0.2	<0.2	<0.2	<0.2	<0.01	<0.2	<0.2	<0.01	<0.01	<0.2	<0.2	<0.2	<1
Cadmium	ug/L	13.1	13.0	7.7	6.1	0.028	4.6	35.9	0.005	<0.005	0.5	12.8	11.8	6.6
Antimony	ug/L	0.8	0.9	0.7	0.2	0.013	0.1	4.4	0.013	<0.005	9.4	2.3	2.2	22.3
Barium	ug/L	16	18	34	66	0.7	23	48	<0.1	0.2	75	134	138	158
Thallium	ug/L	<0.1	<0.1	<0.1	<0.1	<0.005	<0.1	<0.1	<0.005	<0.005	<0.1	<0.1	<0.1	<0.5
Lead	ug/L	0.2	0.3	0.3	0.2	<0.1	0.1	0.1	0.017	<0.005	0.2	0.4	0.5	0.8
Uranium	ug/L	5.41	5.66	1.87	0.19	<0.0005	36.6	7.38	<0.0007	<0.0007	6.47	<0.01	0.04	16.0

Table A-1
Hydrochemistry and Trace Elements (continued)

		108	109	110	111	112	113	114	115	116	117	118	118D	119
Chloride	mg/L	84	0.29	0.17	28.5	n/a	13.4	19.6	16.9	16.8	< 0.01	66.2	66.3	64.8
Sulfate	mg/L	3,490	< 0.05	0.10	2,440	n/a	203	210	166	163	< 0.05	462	467	441
Sodium	mg/L	840	0.2	< 0.1	190	n/a	21	28	31	32	0.3	36	37	36
Potassium	mg/L	120	< 0.2	< 0.2	210	n/a	11	11	9	10	< 0.2	13	13	9
Magnesium	mg/L	57	< 0.05	< 0.05	236	n/a	22	20	17	16	< 0.05	72	74	67
Calcium	mg/L	596	< 0.05	< 0.05	405	n/a	49	53	45	38	< 0.05	121	123	123
TOC	mg/L	10.3	0.4 (a)	0.2 (a)	4.1	n/a	1.8	1.4 (a)	1.4 (a)	1.5	0.3 (a)	3.9	4.3	4.1
TIC	mg/L	18.8	0.86	0.73 (a)	59.9	n/a	14.2	16.7	1.57	2.48	0.75 (a)	19.2	19.4	21.6
Temperature	°C	10.6	n/a	n/a	15.05	14.2	20.98	22.03	16.0	15.5	n/a	14.65	14.4	10.48
Spec. Cond.	mS/cm	6.174	n/a	n/a	4.529	2.765	0.643	0.673	0.567	0.564	n/a	1.348	1.355	1.319
Diss. Oxygen	% sat.	87	n/a	n/a	58.7	46.7	28.4	15.1	87	98.4	n/a	80.7	120	122.8
pH	pH	8.76	n/a	n/a	7.18	6.83	7.74	6.99	7.28	7.41	n/a	7.6	7.49	8.6
ORP (corr.)	mV	240	n/a	n/a	280	229	231	220	261	289	n/a	257	244	240
Lithium	ug/L	27	<0.05	<0.05	23600	4540	347	187	318	312	<0.05	253	264	162
Beryllium	ug/L	<0.2	<0.01	<0.01	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.01	<0.2	<0.2	<0.2
Boron	ug/L	41500	0.9	2.0	27200	13300	1480	931	444	450	0.7	2200	2120	1700
Aluminum	ug/L	81	3.5	3.4	27	17	42	51	17	25	4.3	18	13	28
Silicon	ug/L	221	26.1	42.2	7440	2300	2840	12000	2890	2970	42.4	3710	3840	2870
Vanadium	ug/L	3.6	0.14	0.14	26.9	1.8	402	45.2	53.6	54.3	0.18	3.8	3.5	6.5
Manganese	ug/L	7.7	0.57	4.48	2700	531	147	445	59.3	58.1	0.40	155	167	59.6
Iron	ug/L	3.0	3.7	0.9	<13	55.4	<13	349	<13	<13	0.4	<13	<13	<13
Cobalt	ug/L	0.42	<0.001	0.039	113	8.91	1.76	5.36	7.15	7.05	0.039	3.76	3.53	1.58
Nickel	ug/L	2.2	<0.1	0.2	189	5	<2	6	14	14	<0.1	15	14	8
Copper	ug/L	1.6	0.64	0.96	1.3	0.9	0.4	1.6	9.8	8.8	0.53	2.5	3.0	1.9
Zinc	ug/L	<2	0.7	1.0	289	4	<2	6	16	13	0.8	11	9	<2
Strontium	ug/L	12000	0.59	40.5	6750	2740	662	771	405	411	0.61	507	513	465
Molybdenum	ug/L	2680	0.02	0.11	5100	2690	1280	264	340	336	0.02	131	128	88.7
Silver	ug/L	0.8	<0.01	<0.01	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.01	<0.2	<0.2	<0.2
Cadmium	ug/L	10.6	0.02	0.01	23.6	11.8	5.6	1.4	2.0	2.0	<0.005	1.4	1.0	0.6
Antimony	ug/L	5.2	<0.005	0.006	9.1	0.6	58.5	4.4	20.0	20.7	<0.005	3.1	2.8	2.5
Barium	ug/L	63	<0.1	0.6	40	43	105	62	182	177	0.6	150	153	118
Thallium	ug/L	<0.1	<0.005	<0.005	5.3	0.6	0.8	0.3	7.6	7.3	<0.005	14.2	11.0	6.8
Lead	ug/L	0.3	0.028	0.008	<0.14	<0.14	<0.14	<0.14	<0.14	<0.14	0.013	<0.14	<0.14	<0.14
Uranium	ug/L	21.1	<0.0008	0.001	18.9	21.8	7.91	0.20	0.15	0.17	<0.0008	1.75	1.73	2.02

Table A-1
Hydrochemistry and Trace Elements (continued)

		120	121	122	125	126	126D	127	128
Chloride	mg/L	1,150	1,190	911	0.09	42.5	42.7	31	98
Sulfate	mg/L	1,350	1,510	1,430	< 0.05	507	509	1,120	836
Sodium	mg/L	255	303	247	0.1	393	393	653	141
Potassium	mg/L	500	609	486	< 0.2	20	20	40	30
Magnesium	mg/L	5	6	< 2.5	< 0.05	< 0.05	< 0.05	< 0.05	8
Calcium	mg/L	710	698	669	< 0.05	< 2.5	< 2.5	13	351
TOC	mg/L	1.5	1.3 (a)	2.4	0.6 (a)	6.0	5.8	7.9	7.9
TIC	mg/L	2.81	2.53	2.37	0.39 (a)	5.90	5.89	7.40	3.03
Temperature	°C	16.16	13.65	12.02	12.08	16.75	17.02	16.4	20.5
Spec. Cond.	mS/cm	6.322	6.897	5.906	0.013	2.57	2.76	4.02	2.19
Diss. Oxygen	% sat.	81.3	29.8	77.8	46	35	35	13.1	65
pH	pH	10.33	10.04	10.53	6.04	11.75	11.75	11.74	7.84
ORP (corr.)	mV	87	181	46	373	249	241	225	339
Lithium	ug/L	6470	6360	7070	<0.05	7	8	16	33
Beryllium	ug/L	<0.2	<0.2	<0.2	<0.01	<0.4	<0.4	<0.2	<0.2
Boron	ug/L	3080	3160	1560	2.7	3070	2890	3890	11900
Aluminum	ug/L	167	24	229	4.2	5590	5620	5920	26
Silicon	ug/L	1890	1810	2360	1.1	9450	8860	10300	3940
Vanadium	ug/L	4.5	0.7	1.3	0.29	122	120	236	6.8
Manganese	ug/L	38.1	113	15.5	0.14	<0.4	<0.4	<0.2	197
Iron	ug/L	<13	<13	<13	0.3	<25	<25	<25	<25
Cobalt	ug/L	0.05	0.09	0.03	0.022	<0.04	<0.04	0.20	1.61
Nickel	ug/L	3	<2	<2	<0.1	<0.6	<0.6	<2	<2
Copper	ug/L	0.3	0.4	1.4	0.04	4.2	3.9	2.4	1.5
Zinc	ug/L	<2	<2	<2	0.2	<2	<2	<2	5
Strontium	ug/L	4500	4210	3860	0.63	649	648	1830	5960
Molybdenum	ug/L	333	368	223	0.02	220	223	524	910
Silver	ug/L	<0.2	<0.2	<0.2	<0.01	<0.2	<0.2	<0.2	<0.2
Cadmium	ug/L	1.9	1.6	0.8	<0.005	1.0	1.0	2.1	3.8
Antimony	ug/L	0.1	0.1	<0.1	<0.005	0.4	0.4	0.2	1.3
Barium	ug/L	78	65	58	0.4	36	34	64	86
Thallium	ug/L	0.3	<0.1	<0.1	<0.005	<0.1	<0.1	<0.1	<0.1
Lead	ug/L	<0.14	<0.14	<0.14	<0.007	<0.14	<0.14	<0.14	<0.14
Uranium	ug/L	0.02	0.10	0.04	<0.0008	<0.02	<0.02	<0.02	0.97

Table A-1
Hydrochemistry and Trace Elements (continued)

Footnotes:
(1) = Samples 096 and 096D are samples of leachate that were treated with CO2 prior to analysis.
(a) = sample concentration less than 5 times blank
n/a = not analyzed

Table A-2
Speciation

	Sample ID	001	002	003	004	005	006	007	008	009	010	012	QA-1
As, diss.	ug/L	20.4	48.4	84	18.6	3.0	12.2	20.1	16.9	28.9	22.3	238	0.11
As(III), diss.	ug/L	< 0.3	< 6	< 6	8.4	< 0.2	< 0.3	< 2	0.7 (a)	< 6	1.5 (a)	97.0	< 0.02
As(V), diss.	ug/L	9.5	47	69	5.2	1.3	0.9 (a)	< 2	< 0.5	< 10	10	66	< 0.03
As, other	ug/L	2.1	< 6	< 6	< 0.3	< 0.2	< 0.3	< 2	< 0.3	< 6	< 0.6	< 0.6	< 0.02
Cr, diss.	ug/L	< 0.5	5,100	4,670	8.8	0.7	5.7	2	< 0.5	52.9	25.8	< 0.5	< 3
Cr(III), diss.	ug/L	n/a	340	190	< 0.1	n/a	< 0.1	< 0.1	n/a	1	< 0.4	n/a	n/a
Cr(VI), diss.	ug/L	2.2	5,090	3,530	8.1	1.5	6.4	2.9	< 0.1	47	22	1.9	< 0.05
Se, diss.	ug/L	127	1,730	1,760	49.9	7.6	16.8 (b)	289	3.7 (b)	2,360	318	3.24	0.10 (a)
Se(IV), diss.	ug/L	8.3	19	76	8.1	3.15	1.6	79.5	< 0.1	< 2	24.4	1.4	< 0.02
Se(VI), diss.	ug/L	83.0	1,300	1,240	22.1	0.57	11.2	119	0.27 (a)	1,660	158	< 0.2	< 0.03
Se, other	ug/L	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
Hg_{diss}.	ng/L	n/a	14.4	18.4	5.9	2.1 (a)	0.8 (a)	1.9 (a)	4.2 (a)	28.4	n/a	n/a	n/a
Hg_{part}.	ng/L	n/a	254	26	< 1	44	25 (a)	16 (a)	< 1	121	n/a	n/a	n/a
$MeHg_{diss}$.	ng/L	n/a	0.11	0.09 (a)	0.26	0.12	0.54	< 0.02	0.07 (a)	< 0.02	n/a	n/a	n/a
$MeHg_{part}$.	ng/L	0.03 (a)	0.03 (a)	< 0.01	0.04 (a)	0.09	0.09	0.02 (a)	0.01 (a)	0.02 (a)	n/a	n/a	n/a
DMM	ng/L	0.055	0.005	< 0.005	< 0.005	0.010	< 0.005	0.007	< 0.005	n/a	n/a	n/a	n/a

Table A-2
Speciation (continued)

	Sample ID	013	013D	014	QA-2	015	016	SX-1	017	018	019	020	HN-1	HN-2
As, diss.	ug/L	21.6	22	163	0.12	23.8	68.6	72.0	4.11	23.1	5.11	4.19	59.8	20.6
As(III), diss.	ug/L	3.7	1.9	1.9	0.02 (a)	< 0.6	< 0.6	0.9	0.88	0.42	0.57	1.00	< 0.1	< 0.1
As(V), diss.	ug/L	< 0.5	< 0.5	86	< 0.03	24	25	46.9	<0.08	5.22	<0.08	0.53	33.6	6.9
As, other	ug/L	< 0.3	< 0.3	0.9 (a)	< 0.02	< 0.6	< 0.6	< 0.1	0.1	< 0.06	< 0.06	0.1	0.2	0.1
Cr, diss.	ug/L	< 0.5	n/a	< 0.5	< 3	12.9	3.8	< 0.5	3	< 0.5	1.0	0.7	< 0.5	< 0.5
Cr(III), diss.	ug/L	n/a	n/a	n/a	n/a	< 0.4	< 0.1	n/a	< 0.04	n/a	< 0.1	n/a	n/a	n/a
Cr(VI), diss.	ug/L	0.7	0.7	0.5	< 0.05	12.8	< 0.5	< 0.1	2.8	1.3	0.9	< 0.05	n/a	n/a
Se, diss.	ug/L	0.28 (b)	0.38 (b)	1.81 (b)	0.10 (a)	22.4	193	7.77	2.4	0.50 (b)	1.8	2.5	22.2	9.15
Se(IV), diss.	ug/L	< 0.1	< 0.1	0.6 (a)	< 0.02	14.9	101	2 (a)	0.3 (a)	< 0.1	0.1 (a)	0.9	3 (a)	< 1
Se(VI), diss.	ug/L	< 0.1	< 0.1	< 0.2	< 0.03	3.4	14.3	4 (a)	1.1	< 0.2	1.3	0.8	16	6
Se, other	ug/L	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
$Hg_{diss.}$	ng/L	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
$Hg_{part.}$	ng/L	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
$MeHg_{diss.}$	ng/L	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
$MeHg_{part.}$	ng/L	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
DMM	ng/L	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a

Table A-2
Speciation (continued)

	Sample ID	021	022	023	024	025	026	027	028	029	030	031	032	034
As, diss.	ug/L	194	11.1	218	11.2	6.47	10.8	39.1	30.0	48.9	42.5	221	25.4	< 0.02
As(III), diss.	ug/L	2.1	12.5	0.8 (a)	0.4 (a)	1.35	11.2	13.2	2.4	1.7	3.5	201	17.5	< 0.01
As(V), diss.	ug/L	208	0.49	189	<0.2	<0.08	0.4 (a)	4.8	1.7	8.9	29.5	23.6	16.9	< 0.8
As, other	ug/L	< 0.3	< 0.06	< 0.3	< 0.2	< 0.06	< 0.2	1.3	0.2	0.3	0.4	0.7	0.1	n/a
Cr, diss.	ug/L	< 0.5	1.0	< 0.5	< 0.5	< 0.5	1.1	< 0.5	< 0.5	< 0.5	< 0.5	< 0.5	1.4	0.08
Cr(III), diss.	ug/L	n/a	< 0.04	n/a	n/a	n/a	< 0.04	n/a	n/a	n/a	n/a	n/a	< 0.1	0.06
Cr(VI), diss.	ug/L	< 0.05	0.9	< 0.5	n/a	n/a	0.9	n/a	n/a	n/a	< 0.05	< 0.1	< 0.05	< 0.01
Se, diss.	ug/L	6.5	30.7	283	18.2	1.9 (b)	31.5	1.05 (b)	2.56 (b)	2.29	44.1	12.5	18.0	< 0.02
Se(IV), diss.	ug/L	5.3	20.5	217	5.3	< 0.1	20.4	< 0.3	< 0.3	< 0.3	27.0	0.9 (a)	13.5	< 0.1
Se(VI), diss.	ug/L	< 0.6	2.2	1.5	6.3	1.1	2.2	< 0.3	1.4	1.6	12.5	5.5	0.7	< 0.2
Se, other	ug/L	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
Hg_{diss}.	ng/L	1.4 (a)	1.0 (a)	1.4 (a)	n/a	n/a	0.4 (a)	21.3	1.2 (a)	12.4	0.8 (a)	5.2	1.4 (a)	n/a
Hg_{part}.	ng/L	155	53	14 (a)	n/a	n/a	17 (a)	4 (a)	13 (a)	59	< 1	30	186	n/a
$MeHg_{diss}$.	ng/L	0.03 (a)	0.03 (a)	< 0.02	n/a	n/a	< 0.02	1.56	0.18	0.70	0.06 (a)	6.71	0.05 (a)	n/a
$MeHg_{part}$.	ng/L	0.02 (a)	0.03 (a)	0.03 (a)	n/a	n/a	< 0.01	< 0.01	< 0.01	0.01 (a)	0.11	n/a	0.05	n/a
DMM	ng/L	< 0.005	< 0.005	< 0.005	n/a	n/a	< 0.005	< 0.005	< 0.005	< 0.005	0.022	0.050	0.032	n/a

Table A-2
Speciation (continued)

	Sample ID	035	036	037	038	039	042	043	044	044D	049	050	051	052
As, diss.	ug/L	< 0.02	0.03 (a)	56.0	123	42.3	23.7	75.2	5.1	4.9	5.4	0.12	38.1	164
As(III), diss.	ug/L	< 0.01	< 0.01	0.30	2.63	1.39	< 0.1	< 0.05	0.39	< 0.04	< 0.04	< 0.01	0.70 (a)	22.8
As(V), diss.	ug/L	< 0.8	< 0.8	34	53	53	19 (a)	28	3 (a)	2 (a)	2 (a)	< 0.8	15	8 (a)
As, other	ug/L	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	< 0.04	n/a	n/a	n/a
Cr, diss.	ug/L	0.07	0.14	< 0.4	< 0.4	< 0.4	< 0.4	29.2	< 0.4	< 0.4	< 0.4	0.80	11.3	< 0.4
Cr(III), diss.	ug/L	0.07	0.07	< 0.01	< 0.01	< 0.01	0.17	26.4	0.25	0.12	0.07	0.84	9.92	0.16
Cr(VI), diss.	ug/L	< 0.01	< 0.01	< 0.01	< 0.01	< 0.01	0.03 (a)	< 0.1	< 0.01	< 0.01	< 0.01	< 0.01	< 0.05	0.06
Se, diss.	ug/L	< 0.02	0.02 (a)	1.98 (b)	0.13 (a)	0.17 (a)	42.6	23.5 (b)	13.9	13.6	10.0	0.02 (a)	0.45 (b)	10.2
Se(IV), diss.	ug/L	< 0.1	< 0.1	2.6	< 0.5	0.2 (a)	39.1	20.2	11.4	11.5	8.3	< 0.1	< 0.5	7
Se(VI), diss.	ug/L	< 0.2	< 0.2	< 1	< 1	< 0.4	1.9	< 1	1.7	1.8	0.6 (a)	< 0.2	< 1	< 4
Se, other	ug/L	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
Hg_{diss}.	ng/L	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
Hg_{part}.	ng/L	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
$MeHg_{diss}$.	ng/L	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
$MeHg_{part}$.	ng/L	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
DMM	ng/L	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a

Table A-2
Speciation (continued)

	Sample ID	053	057	059	059D	060	061	062	064	069	070	070D	077	078
As, diss.	ug/L	279	98.6	124	125	< 0.02	1,380	61.5	178	99.5	143	144	< 0.008	0.017 (a)
As(III), diss.	ug/L	108	< 0.2	< 0.2	< 0.2	< 0.01	859	< 0.2	< 0.4	< 0.2	< 0.2	< 0.2	< 0.04	< 0.04
As(V), diss.	ug/L	82	93	127	119	< 0.8	519	37	150	94	136	137	< 0.8	< 0.8
As, other	ug/L	0.7	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	0.53	n/a	n/a
Cr, diss.	ug/L	< 0.4	1.9	2.7	2.5	0.10	< 0.4	10.5	22.4	3.2	5.3	5.4	0.02	0.02
Cr(III), diss.	ug/L	0.05	1.06	0.01 (a)	< 0.01	0.05	0.27	0.95	0.04 (a)	0.46	0.63	0.62	<0.02	0.02
Cr(VI), diss.	ug/L	< 0.01	0.41	1.28	1.23	< 0.01	< 0.01	6.24	23.0	2.98	5.28	5.17	<0.006	<0.006
Se, diss.	ug/L	1.24 (b)	2.44	2.58 (b)	2.55	< 0.02	4.31	112	103	36.4	29.1	29.4	< 0.008	< 0.008
Se(IV), diss.	ug/L	< 2	2.0	2.5	2.2	< 0.1	<10	90.4	97	33.1	29	28	< 0.04	< 0.04
Se(VI), diss.	ug/L	< 4	< 1	< 1	< 1	< 0.2	<20	32.1	< 4	1.7 (a)	< 4	< 4	< 0.06	< 0.06
Se, other	ug/L	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
Hg_{diss}.	ng/L	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	1.9 (a)	2.5 (a)
Hg_{part}.	ng/L	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	3 (a)	4 (a)
$MeHg_{diss}$.	ng/L	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	< 0.02	0.06 (a)
$MeHg_{part}$.	ng/L	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	0.15	0.09
DMM	ng/L	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	< 0.005	< 0.005

Table A-2
Speciation (continued)

	Sample ID	079	079D	082	083	084	088	089	090	091	092	TEB	094	095
As, diss.	ug/L	99.1	97.0	23.0	6.19	727	0.076 (a)	0.896	22.6	10.8	3.33	0.922	0.035 (a)	0.046 (a)
As(III), diss.	ug/L	9.5	9.9	0.2 (a)	0.23	71	n/a	n/a	0.28	< 0.05	< 0.05	0.01 (a)	< 0.01	< 0.01
As(V), diss.	ug/L	104	73	15	2.4 (a)	535	n/a	n/a	18.0	9.4	0.5	0.09	< 0.02	< 0.02
As, other	ug/L	n/a	n/a	n/a	n/a	n/a	n/a	n/a	0.67	0.15 (a)	0.10 (a)	n/a	n/a	n/a
Cr, diss.	ug/L	< 0.2	< 0.2	24.6	19.9	< 0.2	0.22	1.22	0.7	< 0.2	122	0.49	0.03	< 0.01
Cr(III), diss.	ug/L	<0.02	<0.02	1.25	2.43	0.04 (a)	n/a	n/a	n/a	n/a	3 (a)	n/a	n/a	n/a
Cr(VI), diss.	ug/L	<0.006	<0.006	22.9	15.2	<0.006	n/a	n/a	n/a	n/a	109	n/a	n/a	n/a
Se, diss.	ug/L	0.16 (a)	0.16 (a)	19.1	12.8	0.57 (b)	0.010 (a)	0.194 (b)	85.5	122	103	0.094 (a)	0.037 (a)	0.063 (a)
Se(IV), diss.	ug/L	< 0.2	< 0.2	17.9	8.72	< 2	n/a	n/a	5.2	3.6	0.6 (a)	< 0.05	< 0.05	< 0.05
Se(VI), diss.	ug/L	< 0.3	< 0.3	0.3 (a)	1.5 (a)	< 3	n/a	n/a	97	138	116	< 0.05	< 0.05	< 0.05
Se, other	ug/L	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
Hg_{diss}.	ng/L	0.2 (a)	0.5 (a)	5.9	2.1 (a)	0.6 (a)	n/a	n/a	n/a	n/a	n/a	n/a	1.9 (a)	0.9 (a)
Hg_{part}.	ng/L	6 (a)	3 (a)	18 (a)	22 (a)	5 (a)	n/a	n/a	n/a	n/a	n/a	n/a	15 (a)	6 (a)
$MeHg_{diss}$.	ng/L	< 0.02	0.05 (a)	0.05 (a)	0.17	0.06 (a)	n/a	n/a	n/a	n/a	n/a	n/a	0.03 (a)	0.06 (a)
$MeHg_{part}$.	ng/L	0.06	0.05	0.03 (a)	0.16	0.03 (a)	n/a	n/a	n/a	n/a	n/a	n/a	< 0.01	0.02 (a)
DMM	ng/L	< 0.005	< 0.005	< 0.005	0.040	< 0.005	n/a	n/a	n/a	n/a	n/a	n/a	n/a	0.808

Table A-2
Speciation (continued)

Sample ID		096 (1)	096D (1)	097	098	099	100	101	102	103	104	105	106	106D
As, diss.	ug/L	38.3	37.8	44.9	76.9	4.80	0.200	2.23	7.24	0.009 (a)	0.031 (a)	230	110	112
As(III), diss.	ug/L	< 0.1	< 0.1	< 0.1	0.66	0.10 (a)	< 0.01	< 0.1	< 0.05	< 0.01	< 0.01	197	15.9	13.8
As(V), diss.	ug/L	28.2	28.4	36.3	59.5	3.7	< 0.02	0.2 (a)	6.3	0.11	0.08	50.3	63.0	77.3
As, other	ug/L	< 0.1	< 0.1	< 0.1	0.29	0.19	n/a	0.62	< 0.05	n/a	n/a	3.83	5.78	5.22
Cr, diss.	ug/L	1,990	1,980	2,000	2.8	< 0.2	0.03	1.5	19.6	< 0.02	< 0.02	< 0.4	0.9	0.9
Cr(III), diss.	ug/L	120	140	40 (a)	0.2	n/a	n/a	< 0.08	0.4 (a)	n/a	n/a	n/a	n/a	n/a
Cr(VI), diss.	ug/L	2,050	2,030	2,230	0.99	n/a	n/a	0.07	13.3	n/a	n/a	n/a	n/a	n/a
Se, diss.	ug/L	428	427	413	50.7	2.04 (b)	0.047 (a)	91.0	80.5	0.008 (a)	0.008 (a)	8.5 (b)	64.8	65.1
Se(IV), diss.	ug/L	37.3	37.6	38.2	29.3	< 0.8	< 0.05	< 0.8	5.3	< 0.05	< 0.05	< 2	< 2	< 2
Se(VI), diss.	ug/L	363	367	366	< 2	< 2	< 0.05	104	85	< 0.05	< 0.05	< 4	64	65
Se, other	ug/L	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	< 0.05	< 0.05	< 2	< 2	< 2
Hg_{diss}.	ng/L	29.5	32.2	36.5	60.6	5.7	1.5 (a)	2.1 (a)	3.8 (a)	n/a	n/a	n/a	n/a	n/a
Hg_{part}.	ng/L	23 (a)	10 (a)	16 (a)	11 (a)	13 (a)	3 (a)	3 (a)	52	n/a	n/a	n/a	n/a	n/a
$MeHg_{diss}$.	ng/L	0.22	0.20	0.22	0.76	0.03 (a)	< 0.02	< 0.02	0.12	n/a	n/a	n/a	n/a	n/a
$MeHg_{part}$.	ng/L	0.03 (a)	0.03 (a)	0.05	0.01 (a)	< 0.01	0.01 (a)	0.01 (a)	< 0.01	n/a	n/a	n/a	n/a	n/a
DMM	ng/L	0.216	0.335	0.262	0.035	0.265	n/a	0.565	2.47	n/a	n/a	n/a	n/a	n/a

Table A-2
Speciation (continued)

	Sample ID	107	108	109	110	111	112	113	114	115	116	117	118	118D
As, diss.	ug/L	30.6	4.09	0.014 (a)	0.055 (a)	5.94	1.36	102	23.5	8.32	8.24	0.015 (a)	40.8	39.5
As(III), diss.	ug/L	1.0	0.37	< 0.01	< 0.01	< 0.1	0.7	0.8	< 0.1	3.05	1.01	< 0.01	0.66	0.18
As(V), diss.	ug/L	15.1	2.3	< 0.02	0.05 (a)	3.4	0.9	118	20.5	5.3	7.4	< 0.02	45.5	45.6
As, other	ug/L	< 0.2	< 0.05	n/a	n/a	< 0.1	0.2	0.2	< 0.1	< 0.05	0.08	n/a	0.15	0.11
Cr, diss.	ug/L	<2	0.5	0.02	0.03	0.5	< 0.2	< 0.2	0.3	1.5	1.8	0.03	< 0.2	< 0.2
Cr(III), diss.	ug/L	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	0.34	0.40	n/a	n/a	n/a
Cr(VI), diss.	ug/L	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	0.09	0.31	n/a	n/a	n/a
Se, diss.	ug/L	159	6.56 (b)	0.013 (a)	0.021 (a)	90.5	0.67 (b)	29.3	0.07 (a)	36.1	35.4	0.010 (a)	17.6	18.5
Se(IV), diss.	ug/L	< 2	2.6	< 0.05	< 0.05	38.7	< 0.5	19.2	< 0.5	29.6	30.7	< 0.05	17.5	16.5
Se(VI), diss.	ug/L	16	3.9	< 0.05	< 0.05	72	< 1	3 (a)	< 1	3	3	< 0.05	1.3 (a)	1.3 (a)
Se, other	ug/L	51	< 0.5	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
$Hg_{diss.}$	ng/L	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
$Hg_{part.}$	ng/L	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
$MeHg_{diss.}$	ng/L	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
$MeHg_{part.}$	ng/L	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
DMM	ng/L	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a

Table A-2
Speciation (continued)

	Sample ID	119	120	121	122	125	126	126D	127	128
As, diss.	ug/L	30.2	26.8	11.0	25.5	< 0.009	5.20	4.86	6.42	14.3
As(III), diss.	ug/L	< 0.05	7.2	1.3	7.6	< 0.02	< 0.1	< 0.1	< 0.2	10.1
As(V), diss.	ug/L	30.5	11.4	6.0	8.3	< 0.4	4 (a)	3 (a)	4 (a)	3 (a)
As, other	ug/L	0.29	9.3	0.6	6.0	< 0.02	< 0.1	< 0.1	< 0.2	0.4
Cr, diss.	ug/L	0.2	< 0.2	< 0.2	< 0.2	0.05	108	109	24.4	0.5
Cr(III), diss.	ug/L	n/a	n/a	n/a	n/a	0.04	4.15 (a)	2.13 (a)	0.5 (a)	0.16
Cr(VI), diss.	ug/L	n/a	n/a	n/a	n/a	0.02 (a)	121	122	25.5	< 0.02
Se, diss.	ug/L	27.9	3.30 (b)	3.86 (b)	1.13 (b)	< 0.005	88.7	88.3	181	50.9
Se(IV), diss.	ug/L	22.8	1.8	1.1 (a)	< 0.5	< 0.06	12.5	13.0	12.3	17.4
Se(VI), diss.	ug/L	1.7	2 (a)	3 (a)	< 1	< 0.3	103	104	245	7
Se, other	ug/L	n/a	n/a	n/a	n/a	< 0.3	< 0.3	< 0.3	< 0.3	1.8
Hg_{diss}.	ng/L	n/a	n/a	n/a	n/a	3.1 (a)	9.4	2.0 (a)	5.4	79.3
Hg_{part}.	ng/L	n/a	n/a	n/a	n/a	3 (a)	3 (a)	6 (a)	3 (a)	100
$MeHg_{diss}$.	ng/L	n/a	n/a	n/a	n/a	0.16	0.17	0.21	0.03 (a)	6.36
$MeHg_{part}$.	ng/L	n/a	n/a	n/a	n/a	0.02 (a)	0.02 (a)	0.02 (a)	0.02 (a)	0.06
DMM	ng/L	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a

Footnotes:
(1) = Samples 096 and 096D are samples of leachate that were treated with CO_2 prior to analysis.
(a) sample concentration less than 5 times blank
(b) isotope ratios do not match
n/a = not analyzed

B
LEACHATE VARIABILITY AS A FUNCTION OF SAMPLE POINT

Leachate samples were collected from a variety of sample points representing interstitial (porewater) and surface water matrices. Interstitial water from pores of the CCP was collected using leachate wells, lysimeters, drive-point piezometers, and t-handle probes. Seeps and leachate collection systems provided interstitial water that was potentially exposed to the atmosphere. Surface water samples were collected from the ash/water interface in impoundments and from impoundment outfalls. Ash handling waters were collected from sluice pipes.

To evaluate the significance of the type of sample point on the leachate quality results, different sampling points within the same site were compared. Nine sites had multiple sample points for the same CCP management unit. Seven of the sites were impoundments (Table B-1), one site was an impoundment with recirculated water (Table B-2), and one site was a landfill (Table B-3). Indicator parameters, concentrations of reactive constituents (arsenic, chromium, selenium), and non-reactive constituents (boron, sulfate) were compared.

For the seven impoundments, several different methods of sampling were available for comparison of interstitial water, surface water, and sluice water (Table B-1). Comparing different sampling points within a single site yielded the following general observations:

- Field-measured oxidation-reduction potential (ORP) was always higher in surface water samples than interstitial samples. Sluice water ORP was similar to the surface water.
- The pH of interstitial water tended to be higher than surface water samples. Sluice water pH was variable, and in one case was significantly lower than either the interstitial water or surface water.
- Total dissolved solids concentration in interstitial waters were higher than surface waters, suggesting either increased dilution in the pond or higher equilibrium concentrations in the ash sediments due to increased proximity or contact time. Sluice pipe inlet samples were collected at three of the impoundment sites, and in each case, the TDS concentration in the sluice sample was higher than the pond and outfall concentrations, but lower than the interstitial water samples, which suggests that both dilution in the pond and additional leaching in the sediments is occurring.

Table B-1
Comparison of Leachate Samples From Different Collection Points at Impoundments

Site	Analyte	Interstitial				Surface Water		Other
		Drive Point Piez.	Leachate Well	T-Handle Probe	Seep	Ash / Water Interface	Outfall	Sluice Line
33106	ORP (mV)	188		163		**290**	**285**	**335**
	pH (STD)	**7.0**		**7.0**		6.0	6.0	5.0
	TDS	**223**		**247**		99	119	136
	As (ug/L)	49		**123**		5.4	5.1	49
	Cr (ug/L)	<0.40		<0.40		<0.40	<0.40	**14**
	Se (ug/L)	1.1		0.13		**10**	**14**	**33**
	B (ug/L)	**1,154**		**1,240**		265	429	632
	SO4 (mg/L)	**112**		**121**		53	70	84
49003A	ORP (mV)	266		225		284	304	
	pH (STD)	7.5		7.4		7.2	7.3	
	TDS	**455**		**411**		328	325	
	As (ug/L)	**206**		**63**		9.7	9.5	
	Cr (ug/L)	<0.50		0.053		**1.2**	**1.5**	
	Se (ug/L)	145		15		33	33	
	B (ug/L)	**1,410**		**1,205**		437	435	
	SO4 (mg/L)	221		207		192	191	
33104	ORP (mV)	168				220	223	214
	pH (STD)	9.1				7.6	8.9	10
	TDS	**300**				120	163	260
	As (ug/L)	**721**				100	143	178
	Cr (ug/L)	5.0				3.2	5.3	**22**
	Se (ug/L)	58				36	29	**103**
	B (ug/L)	**1,547**				231	207	476
	SO4 (mg/L)	89				45	50	**150**
35015B	ORP (mV)	<41			**308**	**257**	**267**	
	pH (STD)	8.5			8.5	7.6	8.2	
	TDS	**2,750**			1,456	870	793	
	As (ug/L)	**221**			43	41	28	
	Cr (ug/L)	<0.50			<0.50	<0.20	0.82	
	Se (ug/L)	13			**44**	18	23	
	B (ug/L)	**7,610**			3,280	2,200	1,955	
	SO4 (mg/L)	**1,830**			948	462	414	
22346	ORP (mV)		156			**241**		
	pH (STD)		7.3			**8.6**		
	TDS		694			606		
	As (ug/L)		**413**			23		
	Cr (ug/L)		<0.20			**25**		
	Se (ug/L)		0.37			**19**		
	B (ug/L)		**2,710**			442		
	SO4 (mg/L)		**225**			174		

Table B-1
Comparison of Leachate Samples From Different Collection Points at Impoundments (continued)

Site	Analyte	Interstitial				Surface Water		Other
		Drive Point Piez.	Leachate Well	T-Handle Probe	Seep	Ash / Water Interface	Outfall	Sluice Line
40109	ORP (mV)	211		212			**409**	**387**
	pH (STD)	**11**		**8.9**		7.7	9.0	4.4
	TDS	**258**		**311**		126	125	172
	As (µg/L)	**164**		**279**		99	124	38
	Cr (µg/L)	<0.40		<0.40		**1.9**	**2.7**	**11**
	Se (µg/L)	**10**		1.2		2.4	2.6	0.45
	B (µg/L)	**4,620**		**7,370**		300	351	272
	SO4 (mg/L)	**128**		**176**		52	55	111
25410A	ORP (mV)	124				**339**		
	pH (STD)	**12**				9.3		
	TDS	**2,205**				1,273		
	As (µg/L)	**69**				24		
	Cr (µg/L)	3.8				**13**		
	Se (µg/L)	**193**				22		
	B (µg/L)	**109,000**				3,890		
	SO4 (mg/L)	**910**				782		

In some cases, multiple samples were taken from a sample point; these results were averaged.
Bold indicates that these concentrations are significantly higher than concentrations observed in samples from the other matrix.

- Arsenic concentrations were always significantly higher in interstitial waters than in surface waters. Sluice water arsenic showed no consistent trend relative to the interstitial water and surface water.
- Chromium concentrations were always highest in the sluice water samples, variable in the surface water samples, and always low in the interstitial water. This may suggest that chromium initially leached from fly ash in the sluice line was later removed from solution at these sites (all fly ash from bituminous coal).
- Selenium concentrations were variable, sometimes highest in the interstitial water, sometimes highest in the surface water, and sometimes highest in the sluice water.
- Boron and sulfate are highly soluble constituents. Boron concentrations were always significantly higher in the interstitial water than the surface water or the sluice water, suggesting either dilution by transport water and pond water, or increased leaching in the interstitial waters, or both. Sulfate was similar to boron, although the relative difference between the sampling points was not as great.

One impoundment site (23223B) utilized recirculated pond water. At this site, surface water concentrations of all constituents were much higher than the interstitial water, reflecting the concentration build-up due to surface water reuse (Table B-2).

Table B-2
Comparison of Leachate Samples From Different Collection Points at an Impoundment With Recirculated Water

Site	Analyte	Interstitial Leachate Well	Surface Water Ash / Water Interface
23223B	ORP (mV)	179	**342**
	pH (STD)	7.3	7.4
	TDS	4,851	**14,233**
	As (µg/L)	18	**29**
	Cr (µg/L)	0.62	**53**
	Se (µg/L)	146	**2,360**
	B (µg/L)	65,250	**98,500**
	SO4 (mg/L)	2,615	**10,400**

In some cases, multiple samples were taken from a sample point; these results were averaged.
Bold indicates that these concentrations are significantly higher than concentrations observed in samples from the other matrix.

One landfill site (50183) had samples collected from a leachate collection system and a leachate well (Table B-3). Both provide samples of interstitial water, the difference being that the leachate collection system provides an opportunity for exposure to atmospheric conditions that does not exist in a leachate well when properly sampled. In this case, the sample from the leachate collection system had a lower ORP, and had much higher concentrations of all constituents than the leachate well sample. The large difference in water quality at this site may reflect heterogeneity at the site rather than a systematic difference in sampling location. The landfill receives fly ash from three different plants, and the plants burn different coal types.

Exposure to atmospheric conditions, particularly oxygen, may be particularly important when measuring species concentrations in the leachate. Speciation by sample point was compared for the nine sites with multiple sample points. These data indicated wide variability in some cases, but no clear pattern of speciation change was associated with sample points (see Tables 5-1, 5-3, and 5-5).

Table B-3
Comparison of Leachate Samples From Different Collection Points at a Landfill

Site	Analyte	Interstitial	
		Leachate Well	Leachate Collection System
50183	ORP (mV)	**257**	158
	pH (STD)	7.7	**9.0**
	TDS	1,479	**3,080**
	As (µg/L)	3.9	**48**
	Cr (µg/L)	0.23	**5.8**
	Se (µg/L)	4.8	**50**
	B (µg/L)	2,000	**11,250**
	SO4 (mg/L)	930	**1,880**

In some cases, multiple samples were taken from a sample point; these results were averaged.
Bold indicates that these concentrations are significantly higher than concentrations observed in samples from the other matrix.

In summary, this analysis suggests that there were some systematic patterns to variation among sampling points at impoundment sites. Concentrations of non-reactive elements, sulfate and particularly boron, were significantly higher in interstitial leachate than in surface water leachate. Concentrations of arsenic were also consistently higher in interstitial water. Conversely, Chromium concentration tended to be slightly higher in sluice water and surface water samples.

C

BOX PLOTS COMPARING ASH LEACHATE CONCENTRATIONS BY SITE AND PLANT ATTRIBUTES

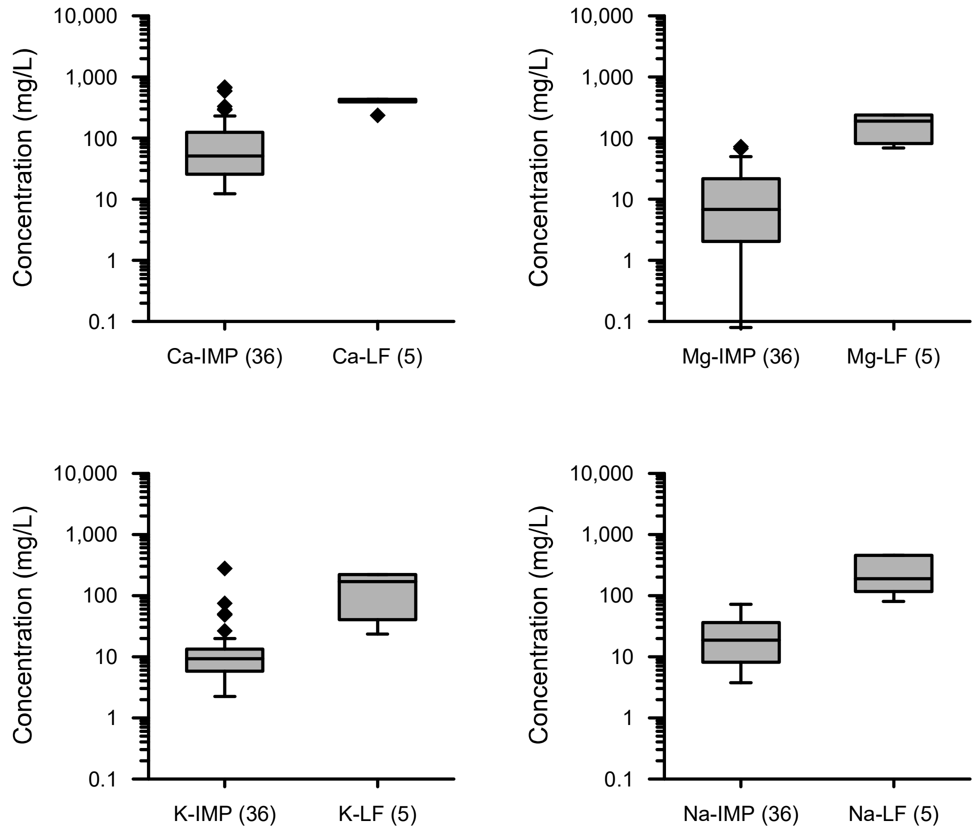


Figure C-1
Comparison of Field Leachate Concentrations: Bituminous Coal Ash, Landfill versus Impoundment

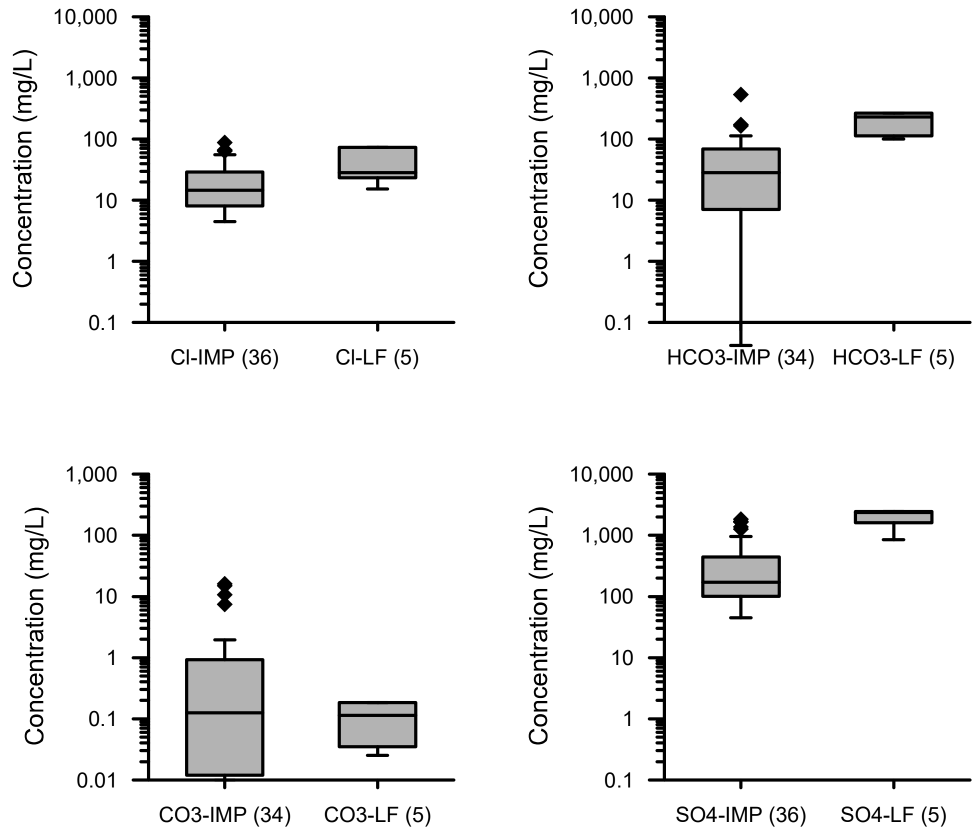


Figure C-1
Comparison of Field Leachate Concentrations: Bituminous Coal Ash, Landfill versus Impoundment (continued)

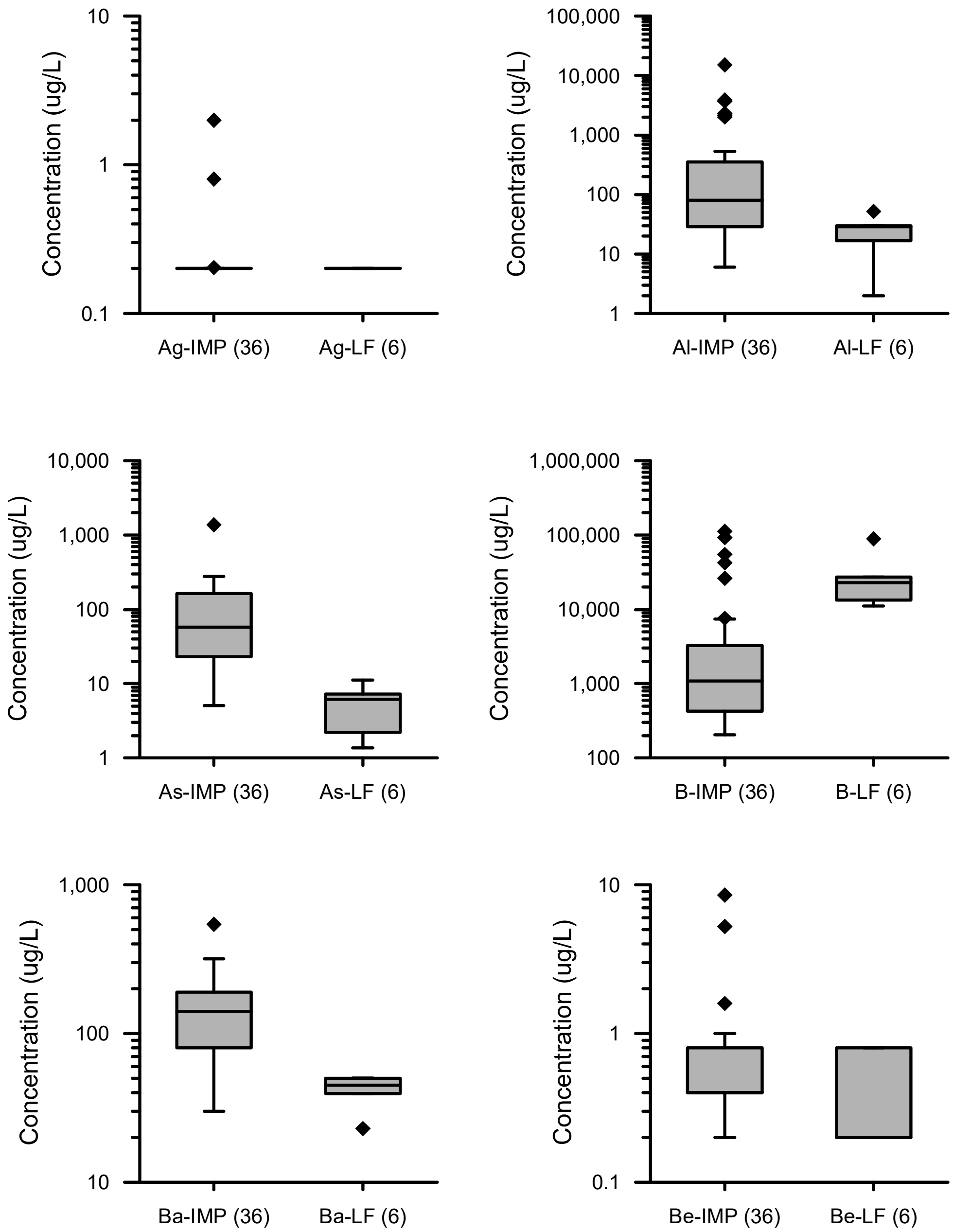


Figure C-1
Comparison of field Leachate Concentrations: Bituminous Coal Ash, Landfill versus Impoundment (continued)

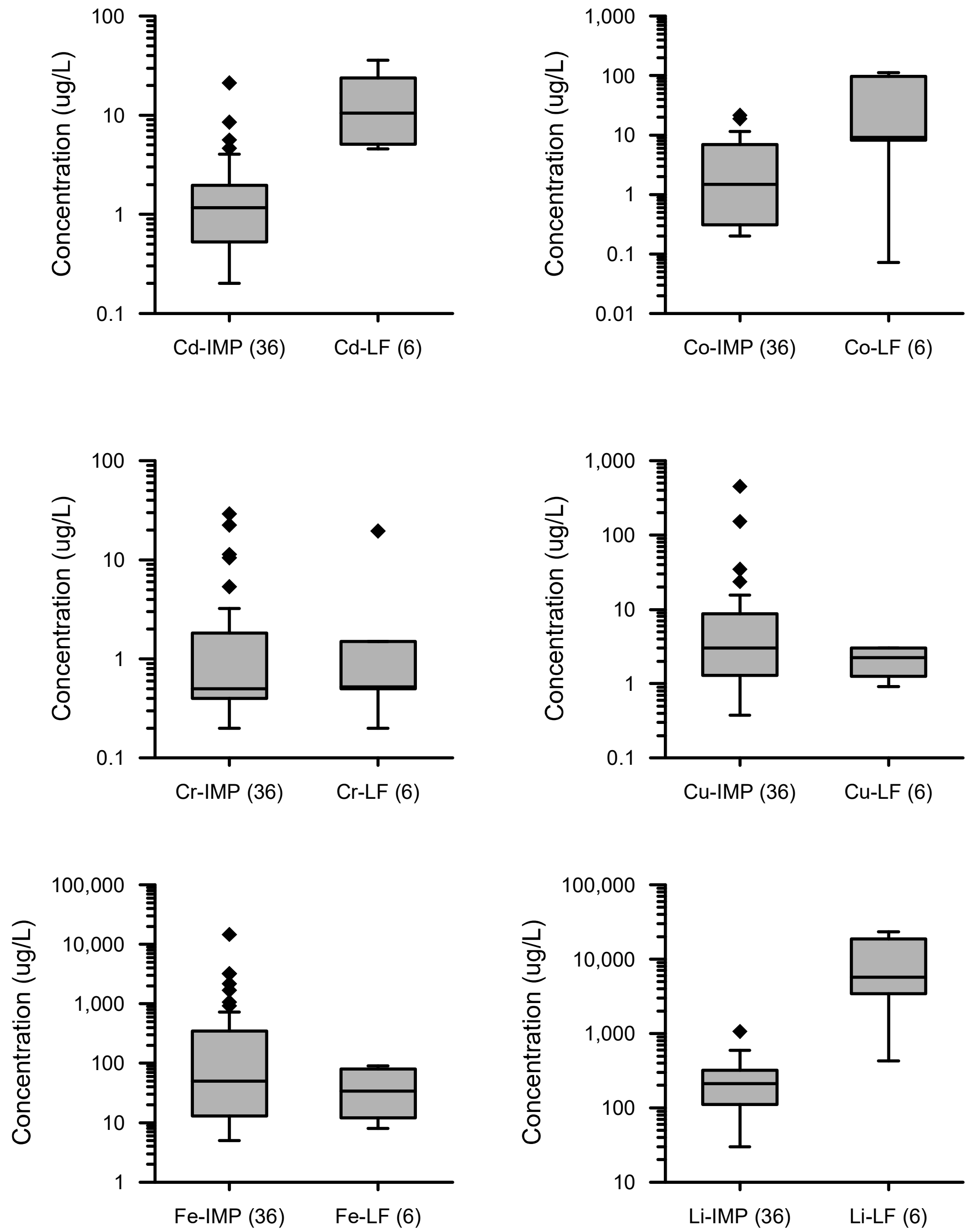


Figure C-1
Comparison of field Leachate Concentrations: Bituminous Coal Ash, Landfill versus Impoundment (continued)

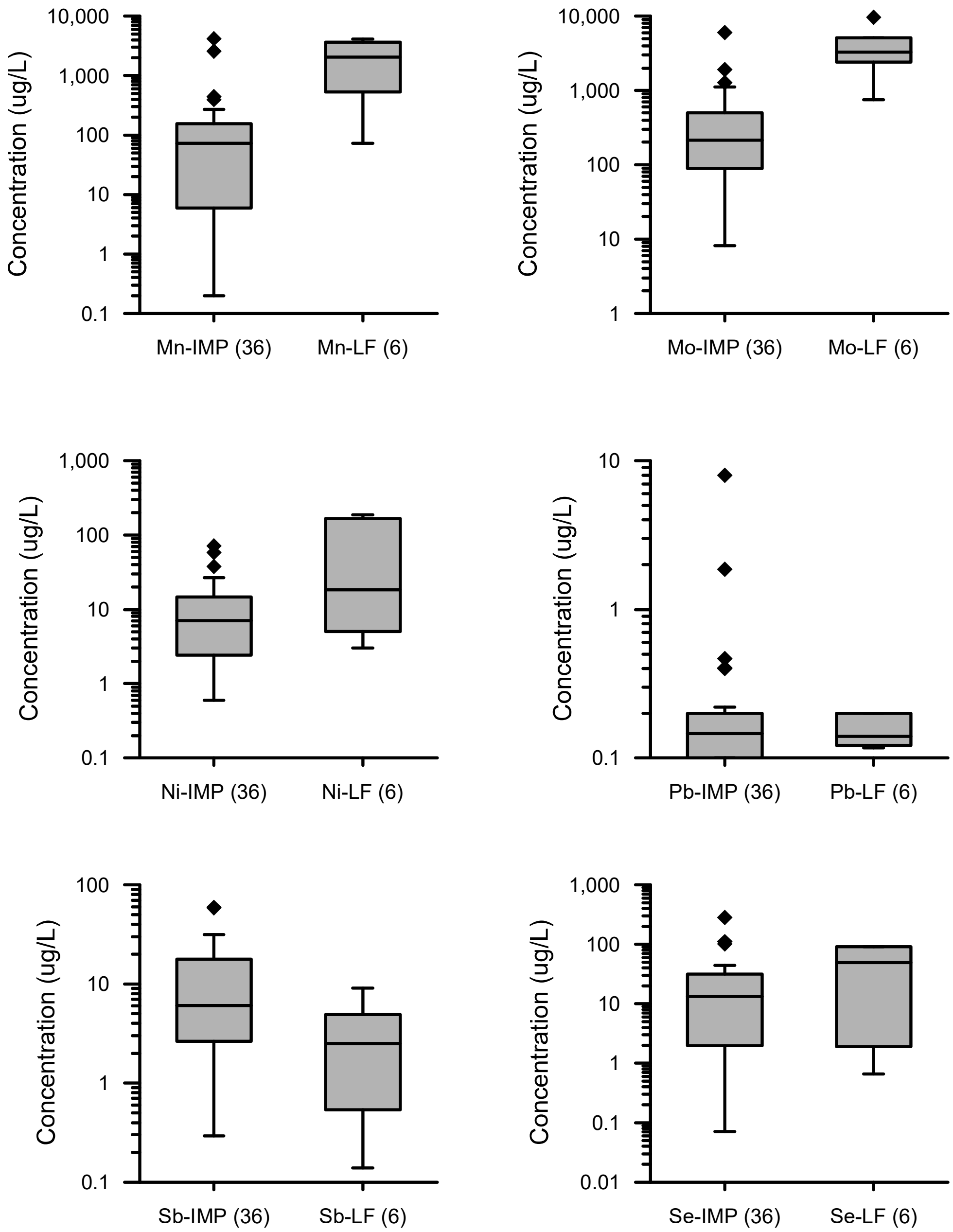


Figure C-1
Comparison of Field Leachate Concentrations: Bituminous Coal Ash, Landfill versus Impoundment (continued)

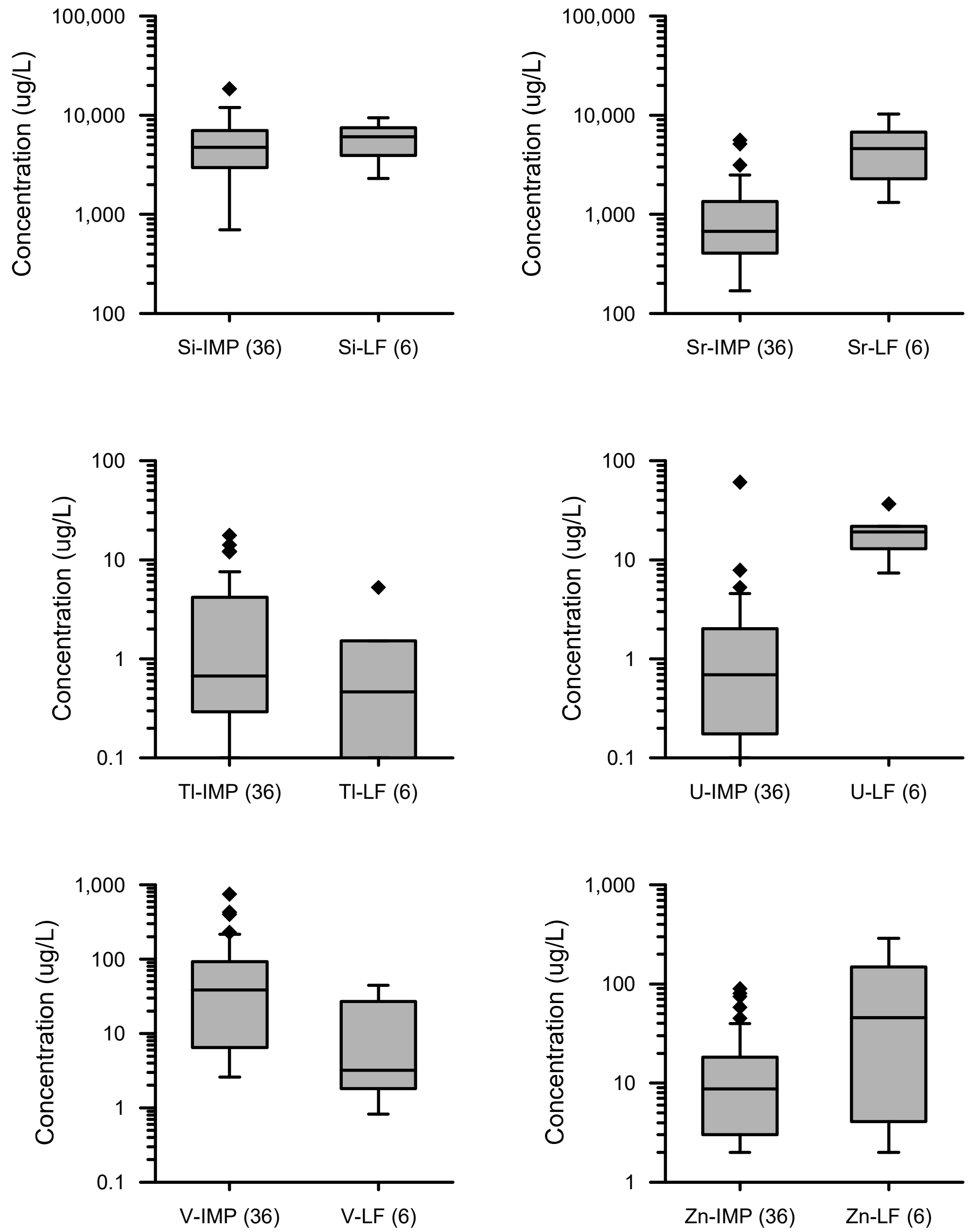


Figure C-1
Comparison of Field Leachate Concentrations: Bituminous Coal Ash, Landfill versus Impoundment (continued)

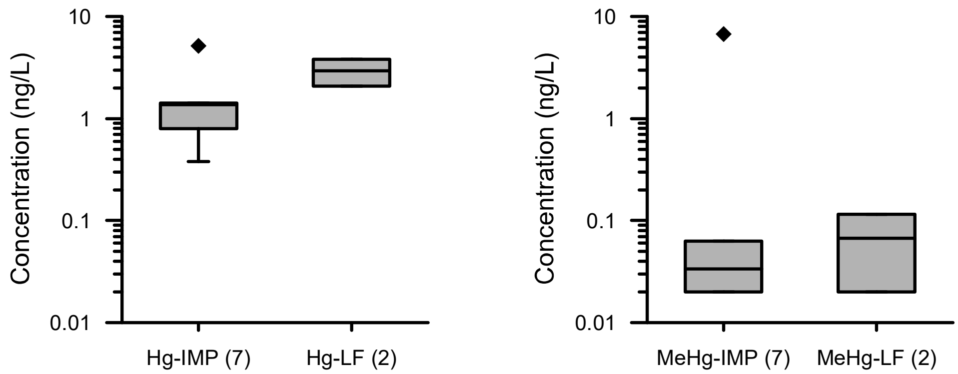


Figure C-1
Comparison of Field Leachate Concentrations: Bituminous Coal Ash, Landfill versus Impoundment (continued)

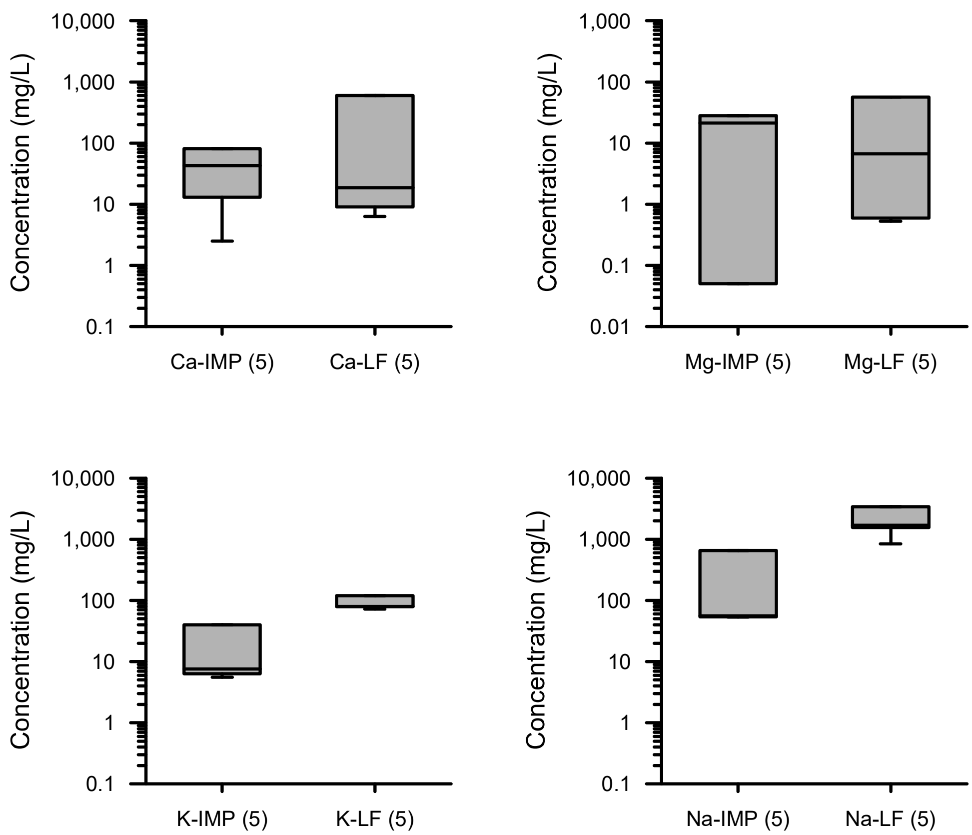


Figure C-2
Comparison of Field Leachate Concentrations: Subbituminous/Lignite Coal Ash, Landfill versus Impoundment

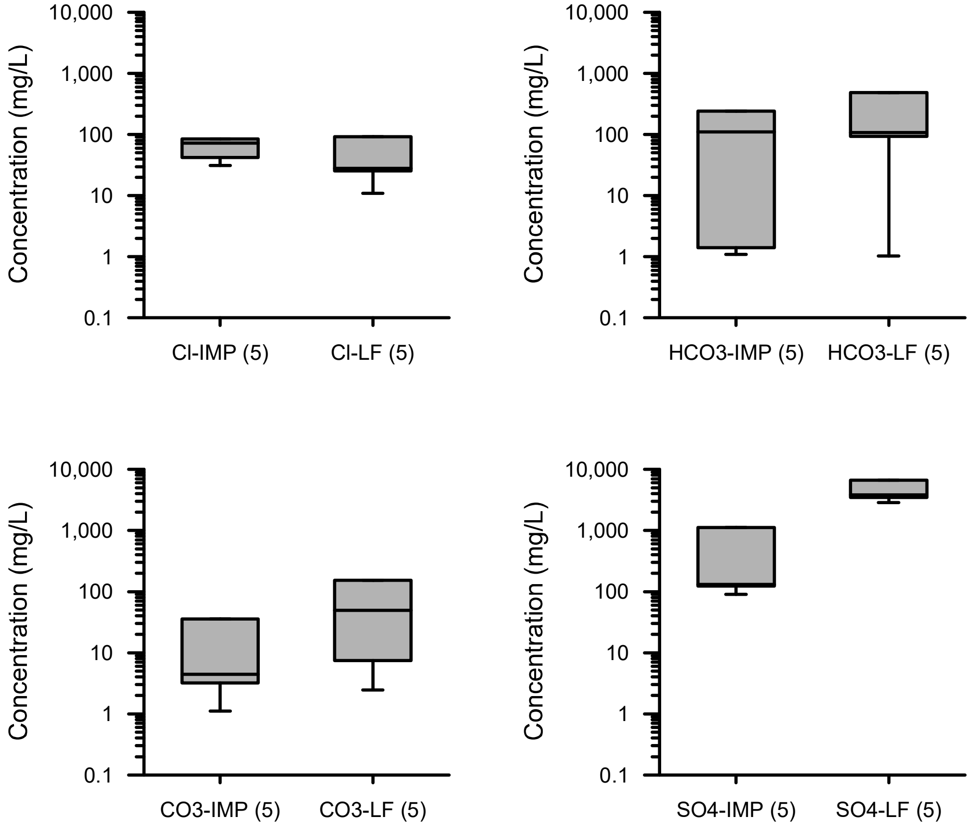


Figure C-2
Comparison of Field Leachate Concentrations: Subbituminous/Lignite Coal Ash, Landfill versus Impoundment (continued)

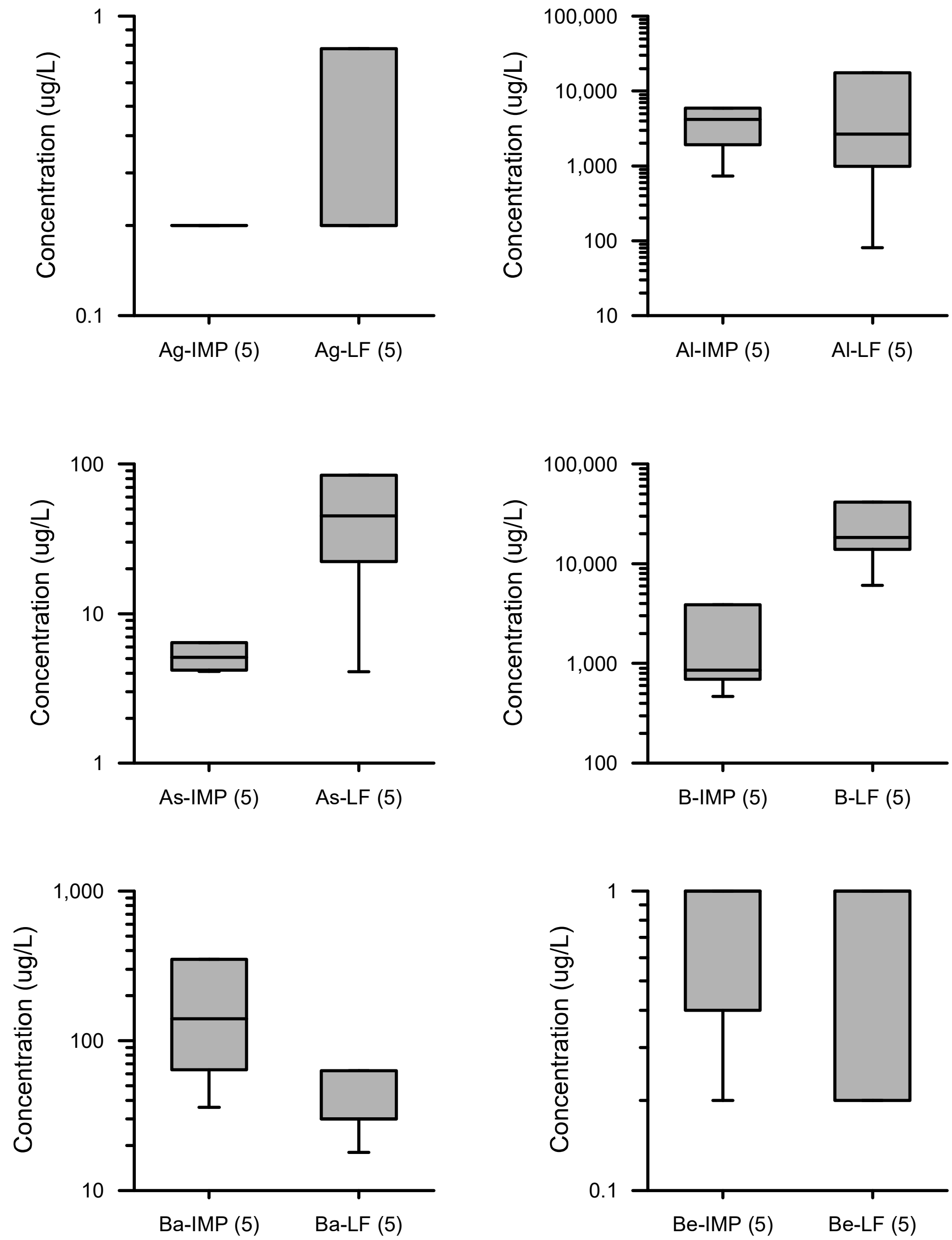


Figure C-2
Comparison of Field Leachate Concentrations: Subbituminous/Lignite Coal Ash, Landfill versus Impoundment (continued)

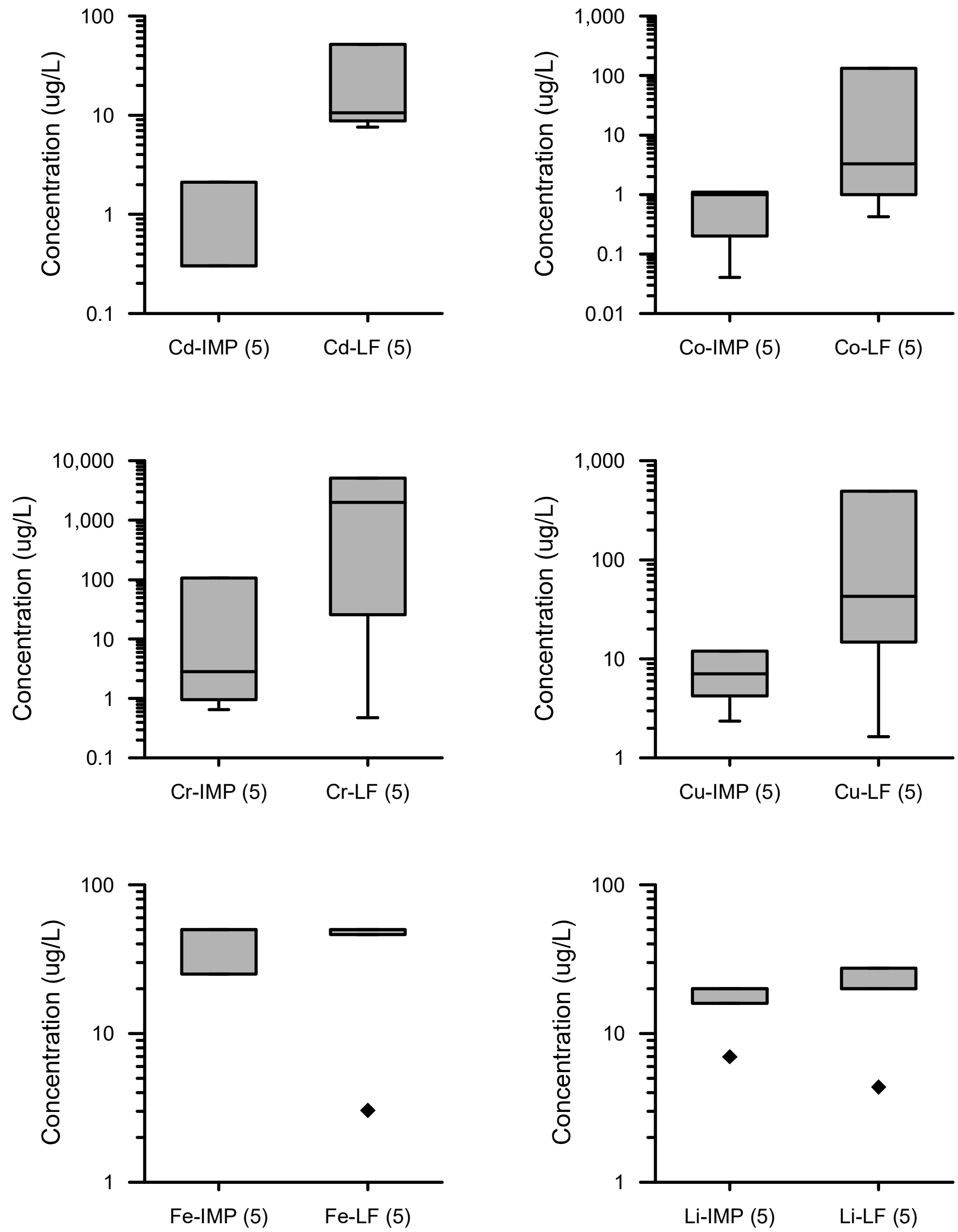


Figure C-2
Comparison of Field Leachate Concentrations: Subbituminous/Lignite Coal Ash, Landfill versus Impoundment (continued)

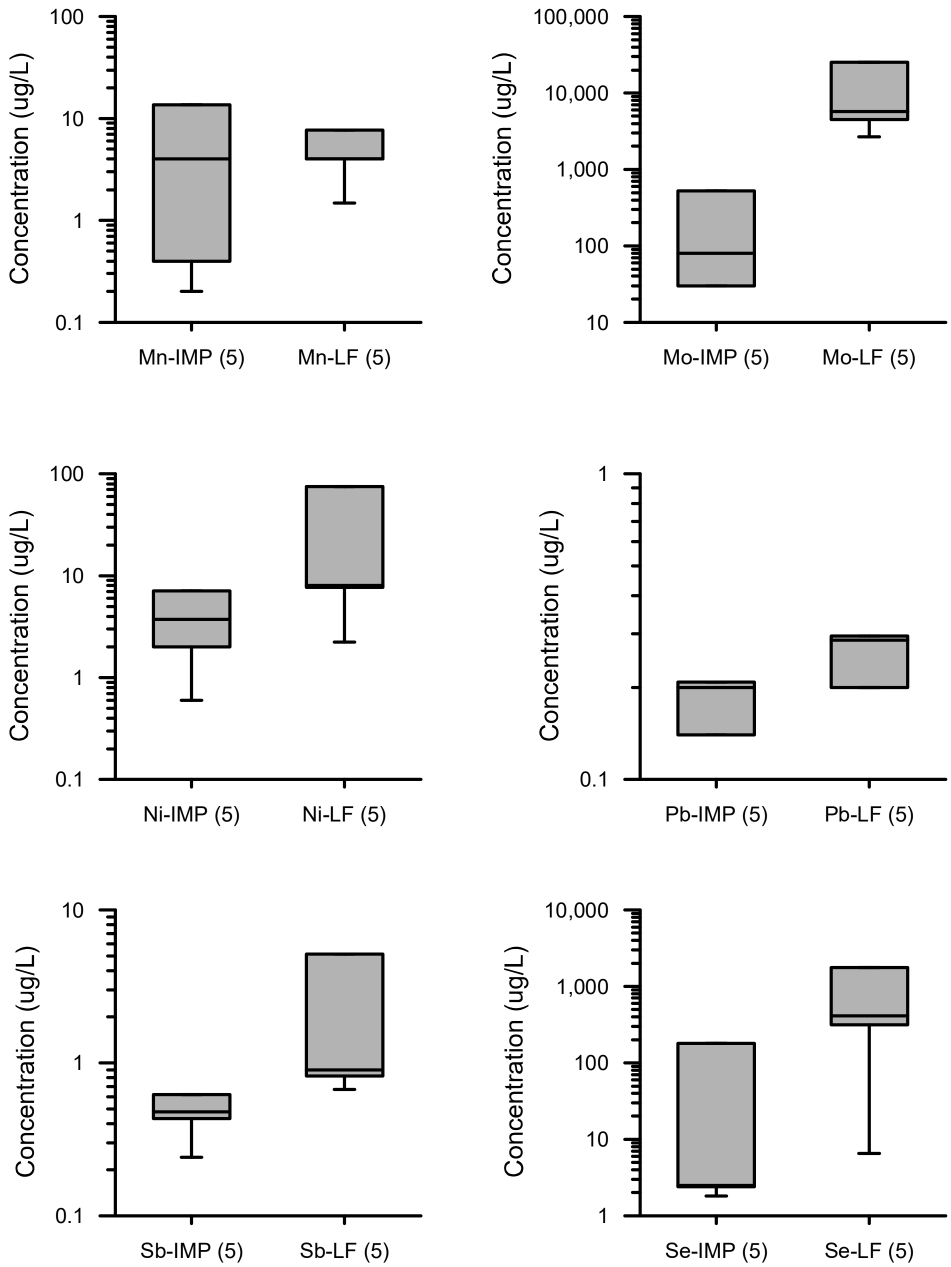


Figure C-2
Comparison of Field Leachate Concentrations: Subbituminous/Lignite Coal Ash, Landfill versus Impoundment (continued)

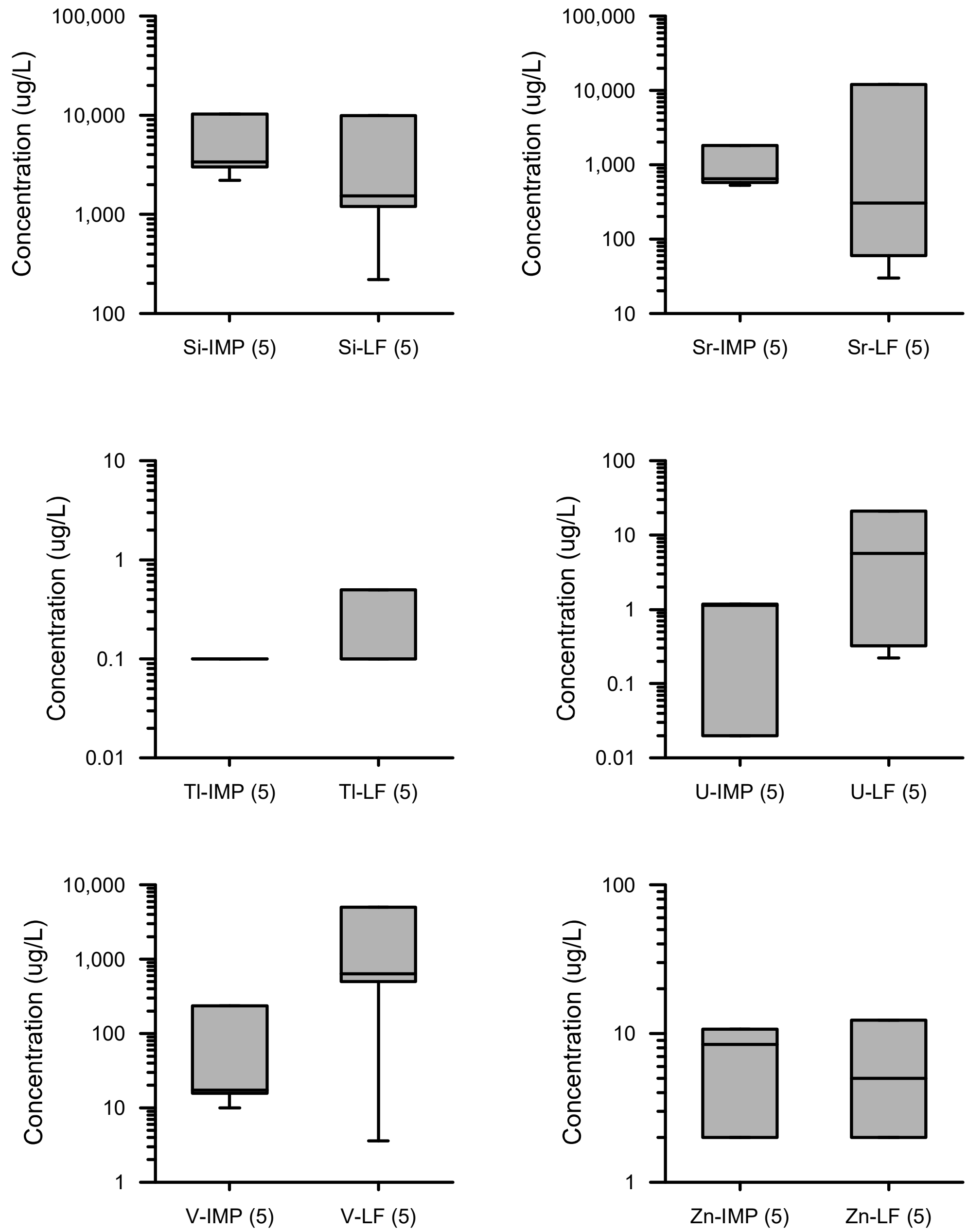


Figure C-2
Comparison of Field Leachate Concentrations: Subbituminous/Lignite Coal Ash, Landfill versus Impoundment (continued)

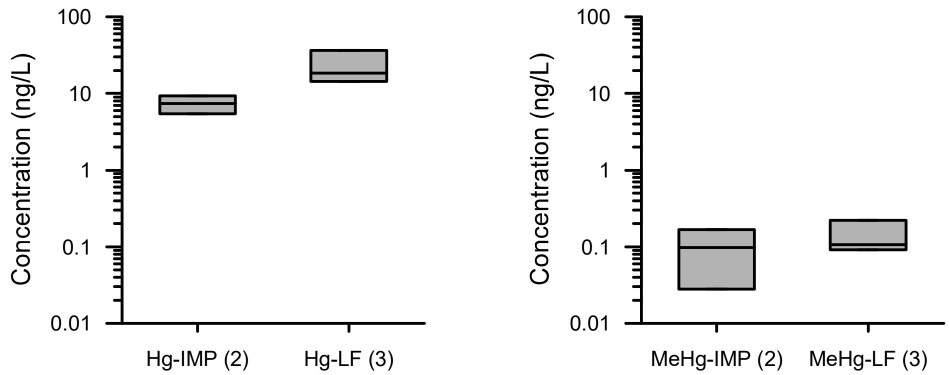


Figure C-2
Comparison of field Leachate Concentrations: Subbituminous/Lignite Coal Ash, Landfill versus Impoundment (continued)

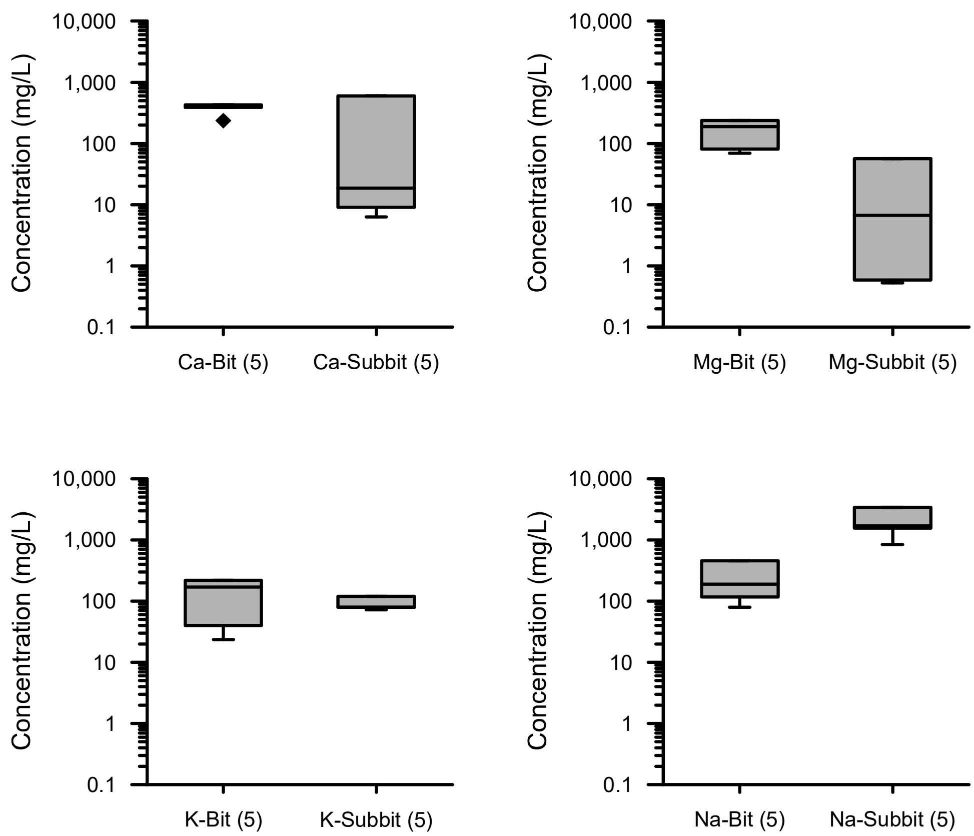


Figure C-3
Comparison of Field Leachate Concentrations: Bituminous vs. Subbituminous/Lignite Coal Ash, Landfills

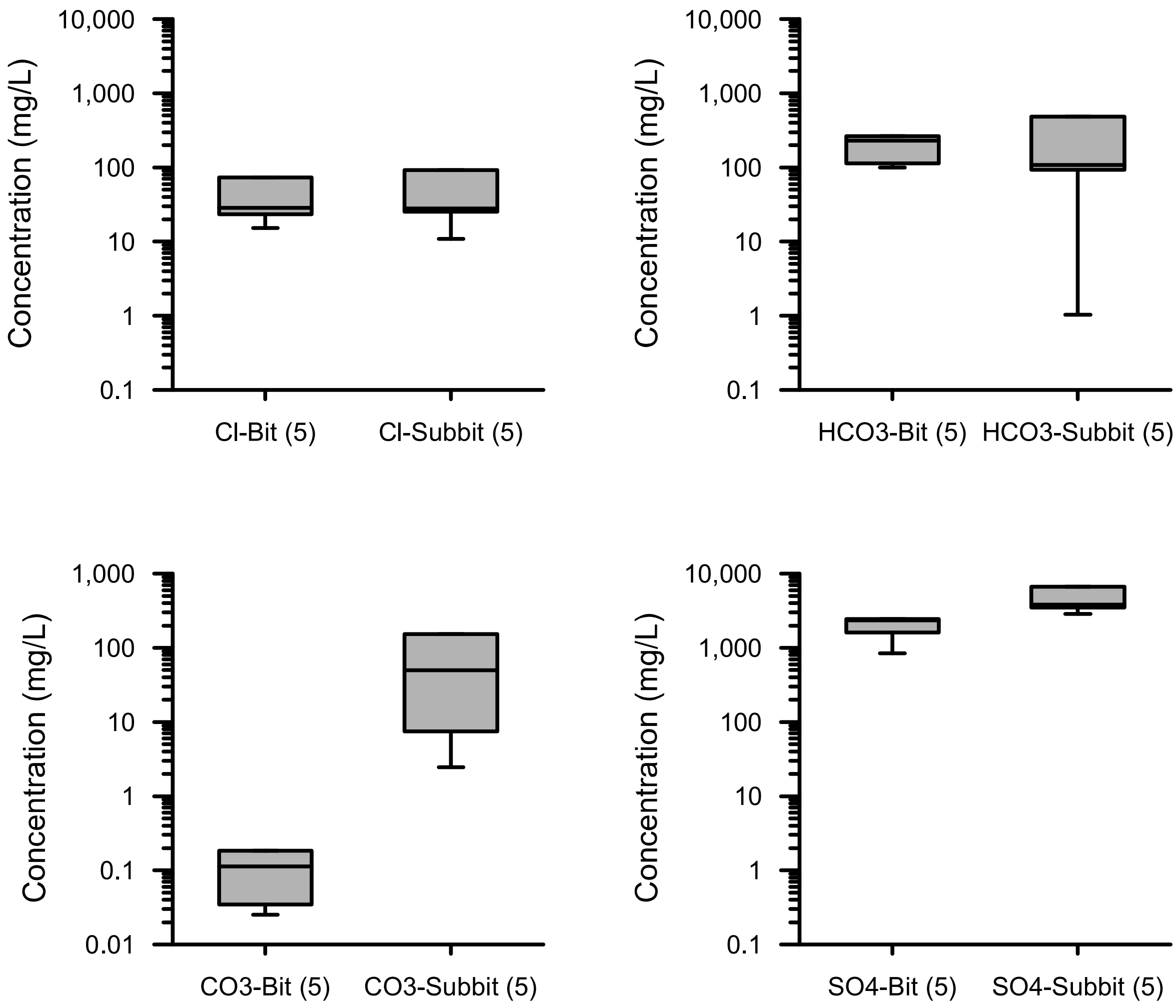


Figure C-3
Comparison of Field Leachate Concentrations: Bituminous vs. Subbituminous/Lignite Coal Ash, Landfills (continued)

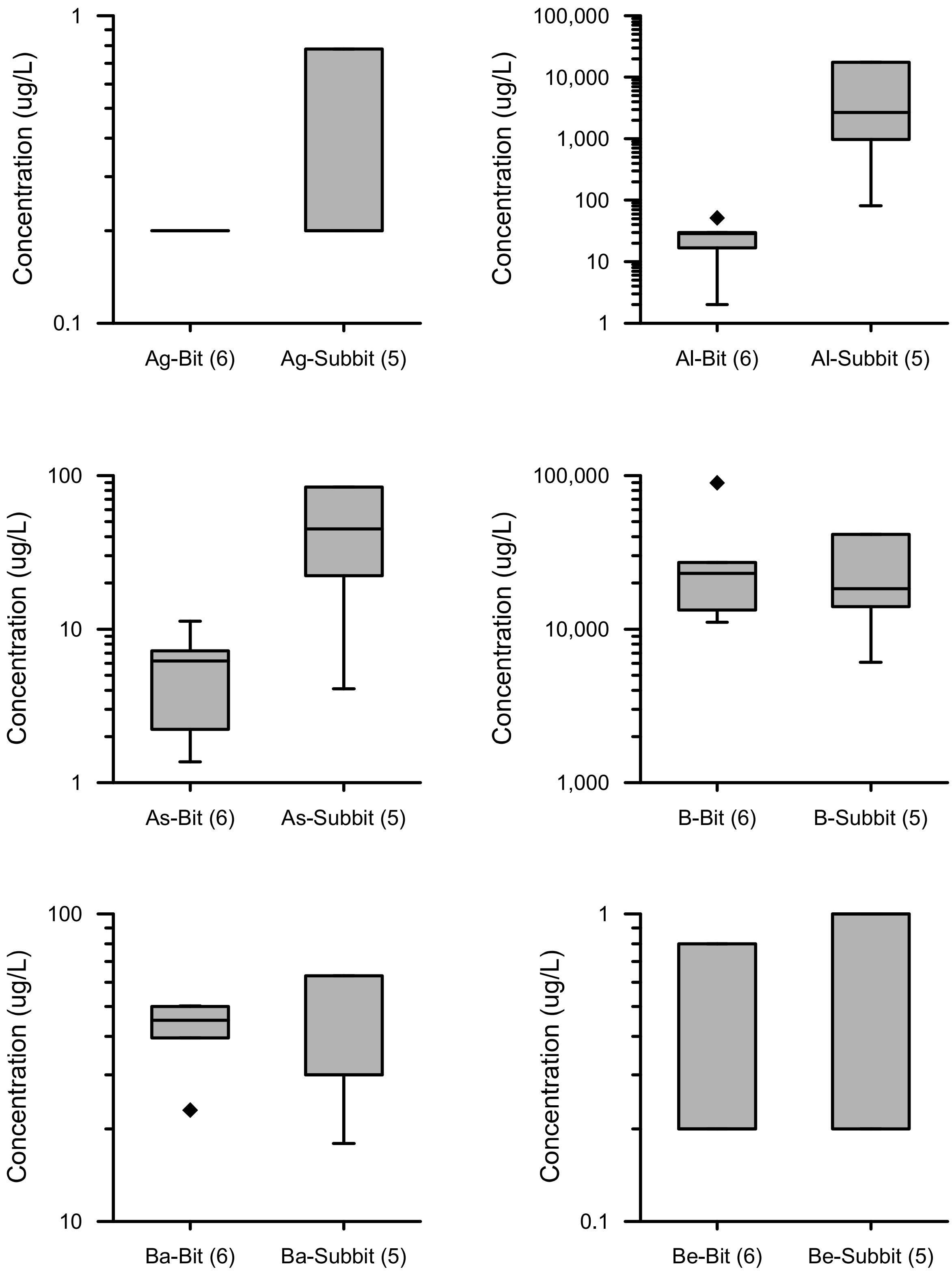


Figure C-3
Comparison of Field Leachate Concentrations: Bituminous vs. Subbituminous/Lignite Coal Ash, Landfills (continued)

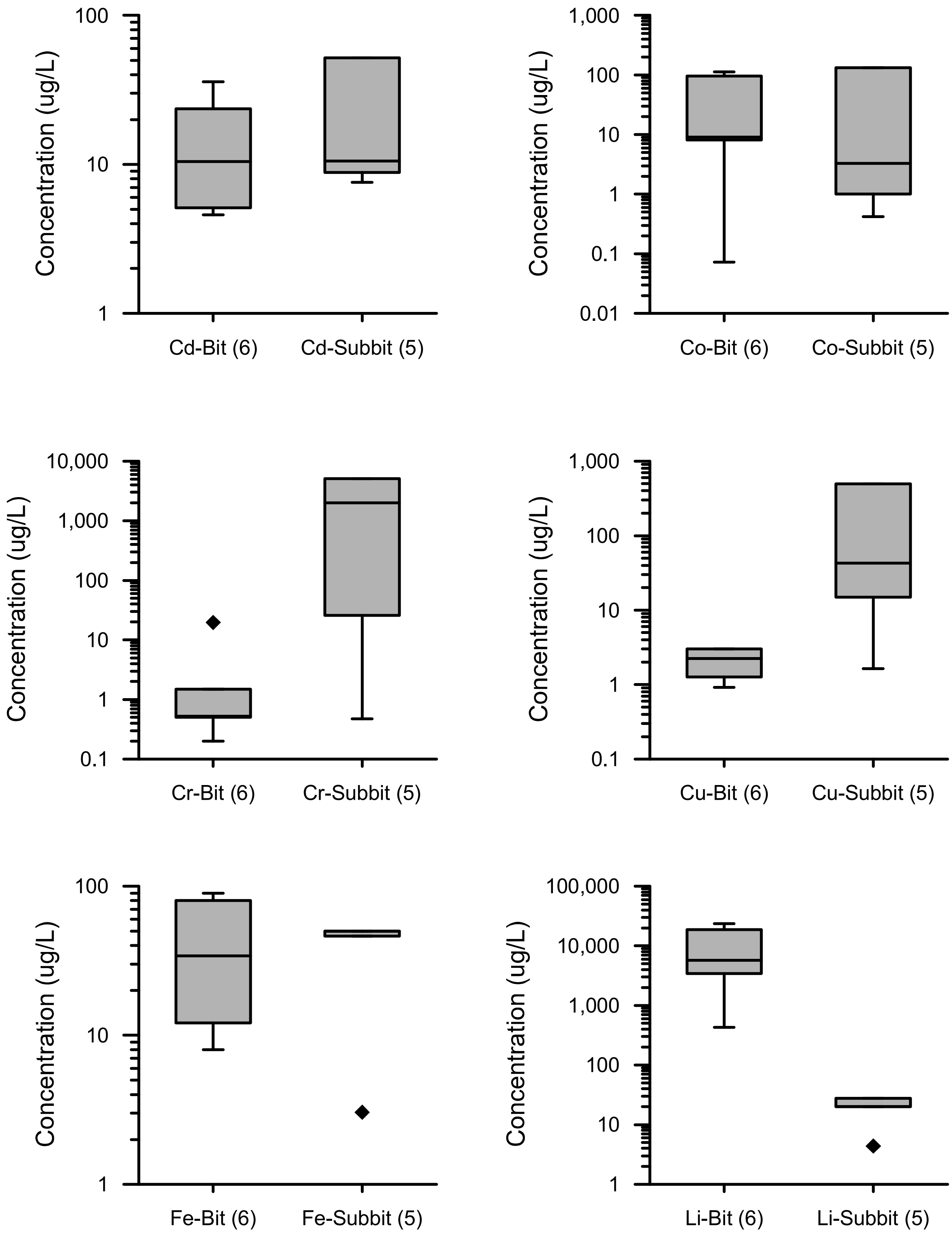


Figure C-3
Comparison of Field Leachate Concentrations: Bituminous vs. Subbituminous/Lignite Coal Ash, Landfills (continued)

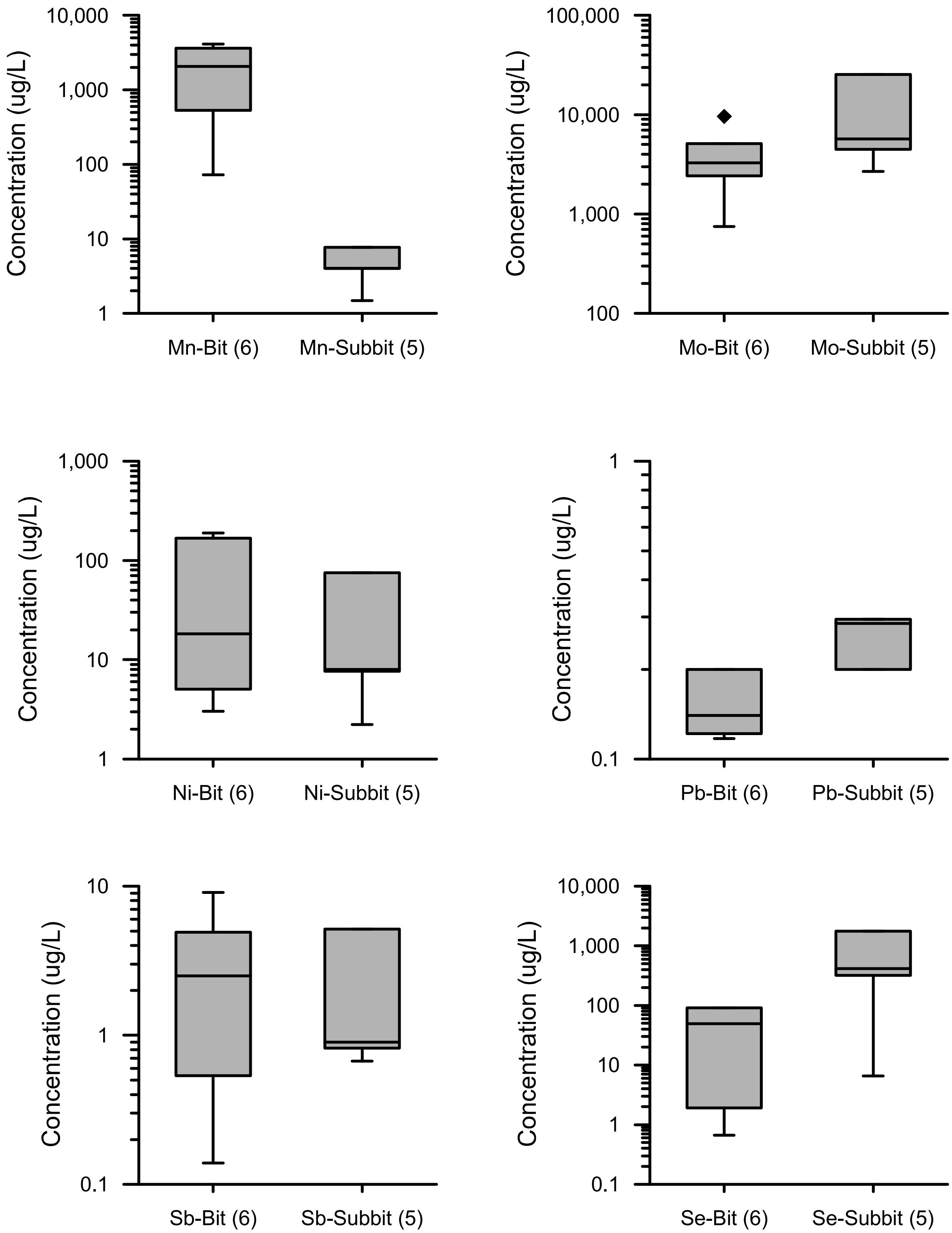


Figure C-3
Comparison of Field Leachate Concentrations: Bituminous vs. Subbituminous/Lignite Coal Ash, Landfills (continued)

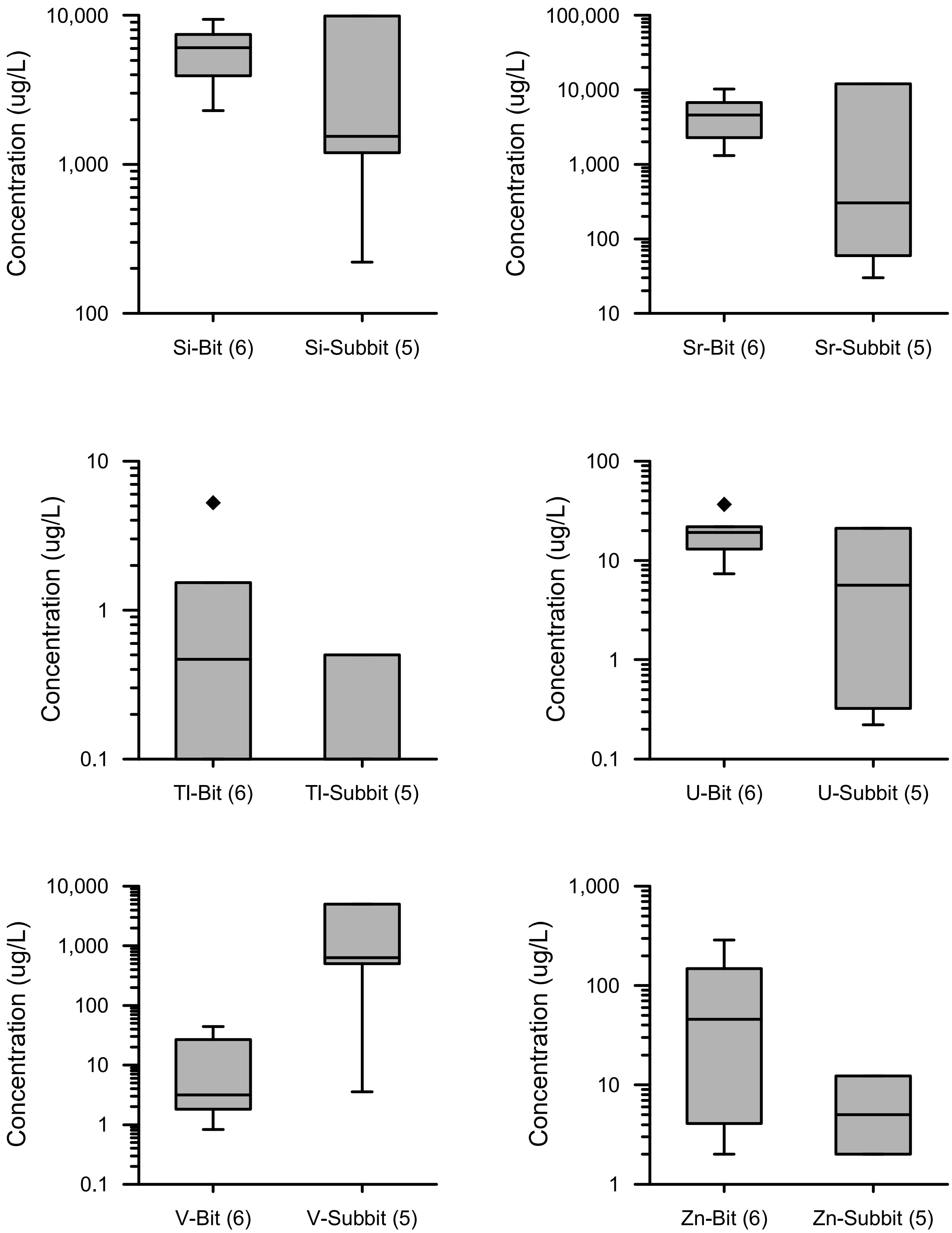


Figure C-3
Comparison of Field Leachate Concentrations: Bituminous vs. Subbituminous/Lignite Coal Ash, Landfills (continued)

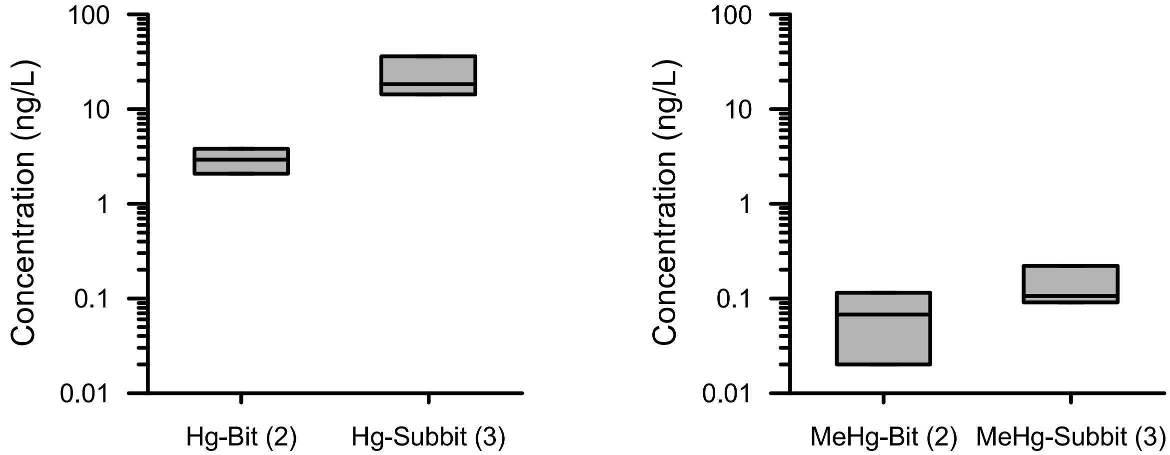


Figure C-3
Comparison of Field Leachate Concentrations: Bituminous vs. Subbituminous/Lignite Coal Ash, Landfills (continued)

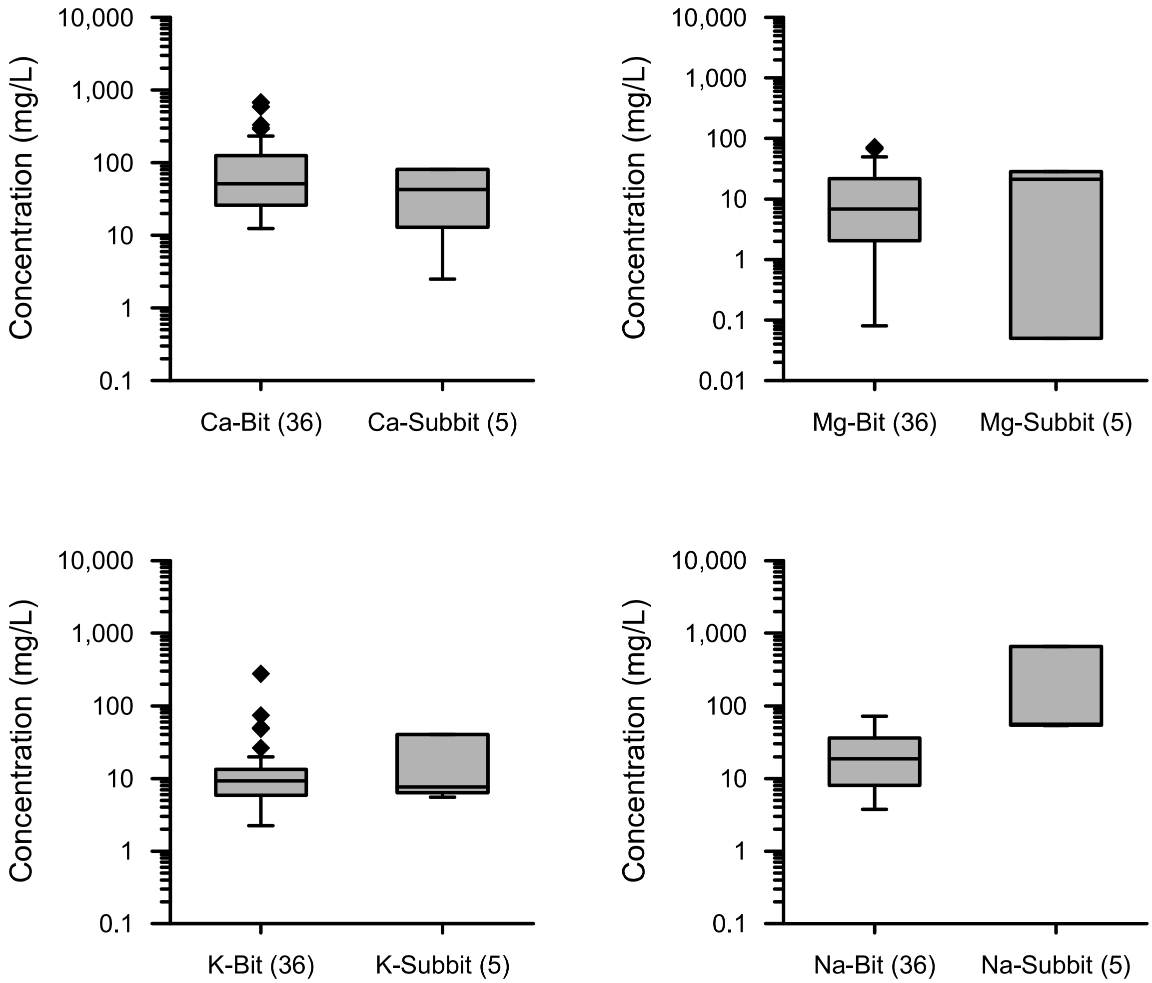


Figure C-4
Comparison of Field Leachate Concentrations: Bituminous vs. Subbituminous/Lignite Coal Ash, Impoundments

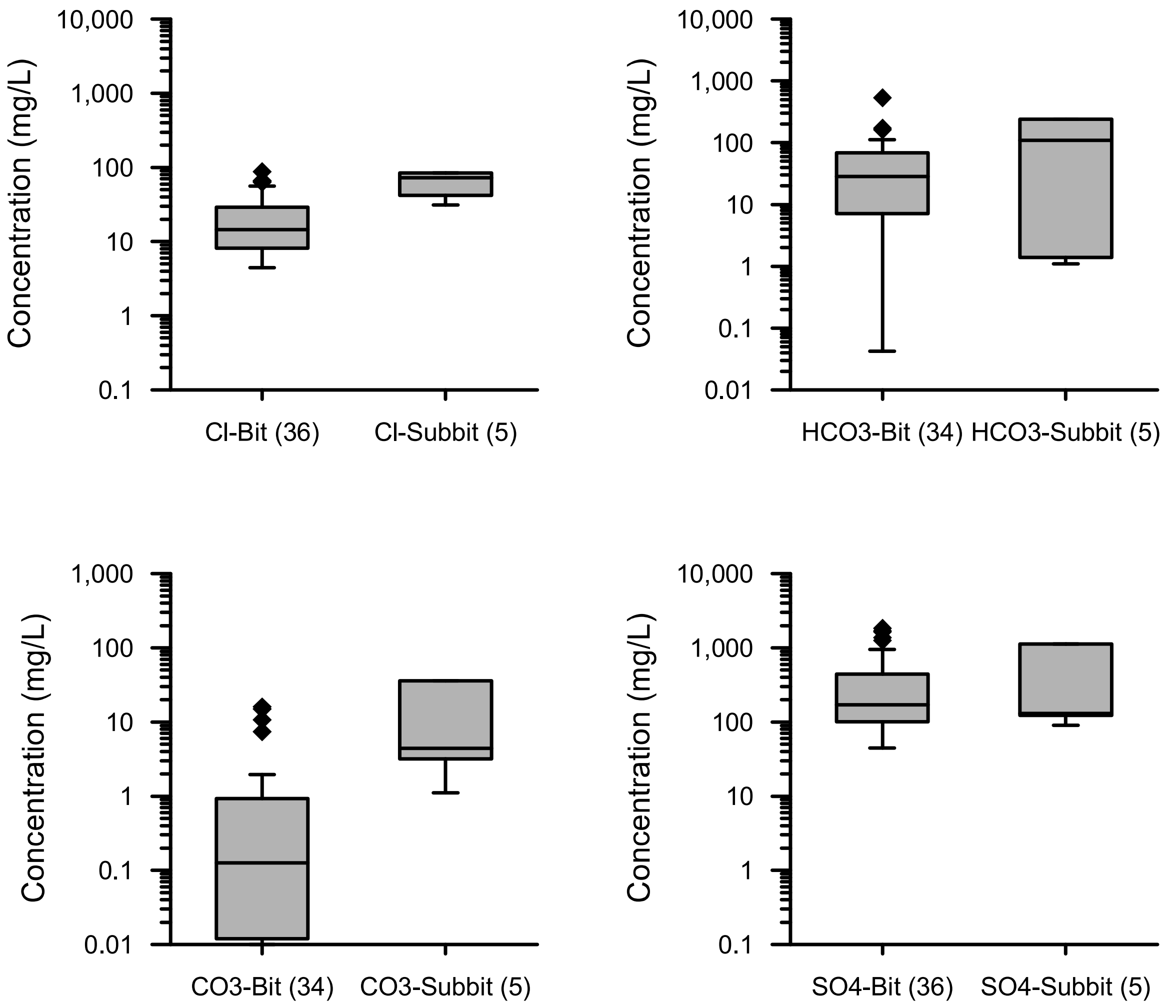


Figure C-4
Comparison of Field Leachate Concentrations: Bituminous vs. Subbituminous/Lignite Coal Ash, Impoundments (continued)

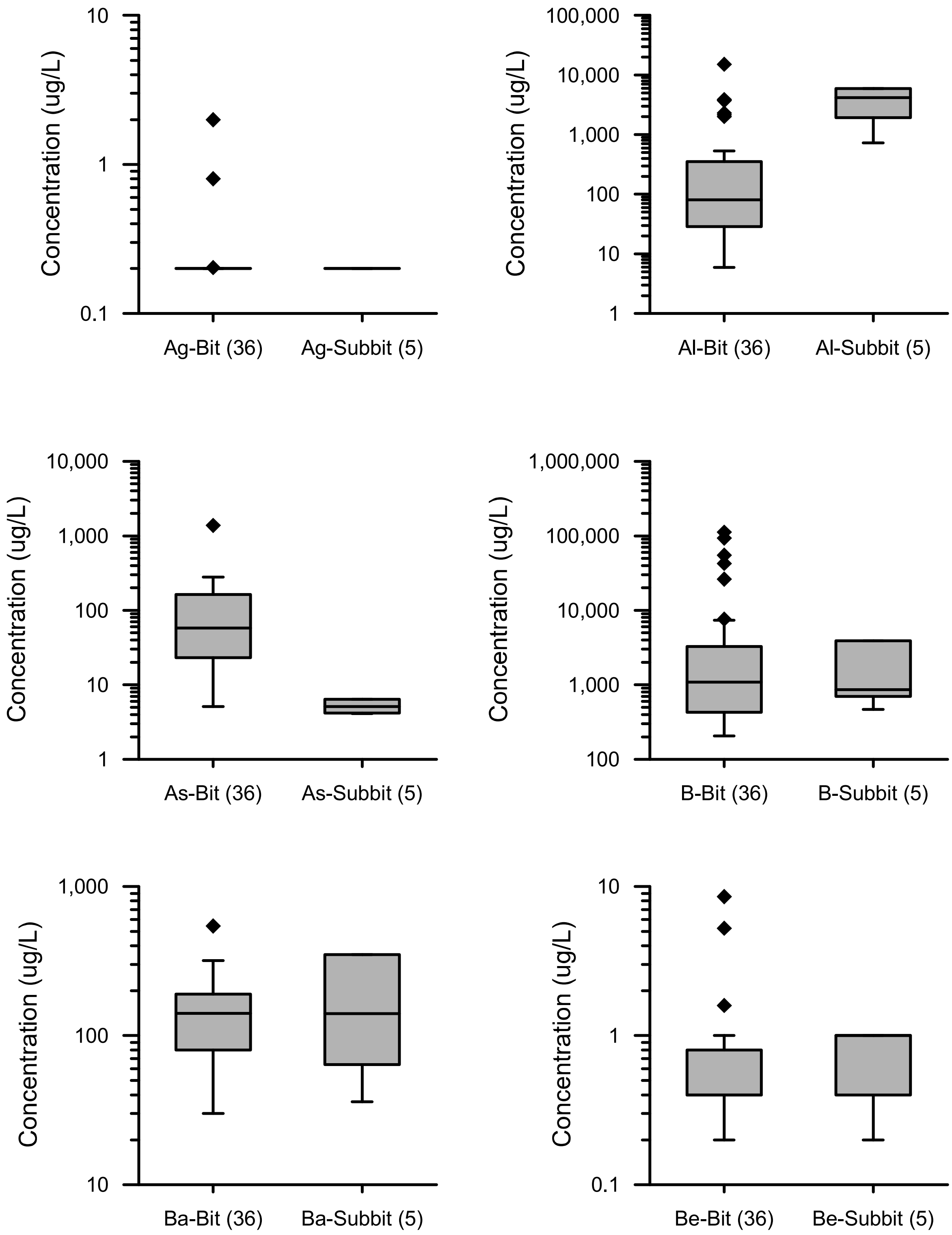


Figure C-4
Comparison of Field Leachate Concentrations: Bituminous vs. Subbituminous/Lignite Coal Ash, Impoundments (continued)

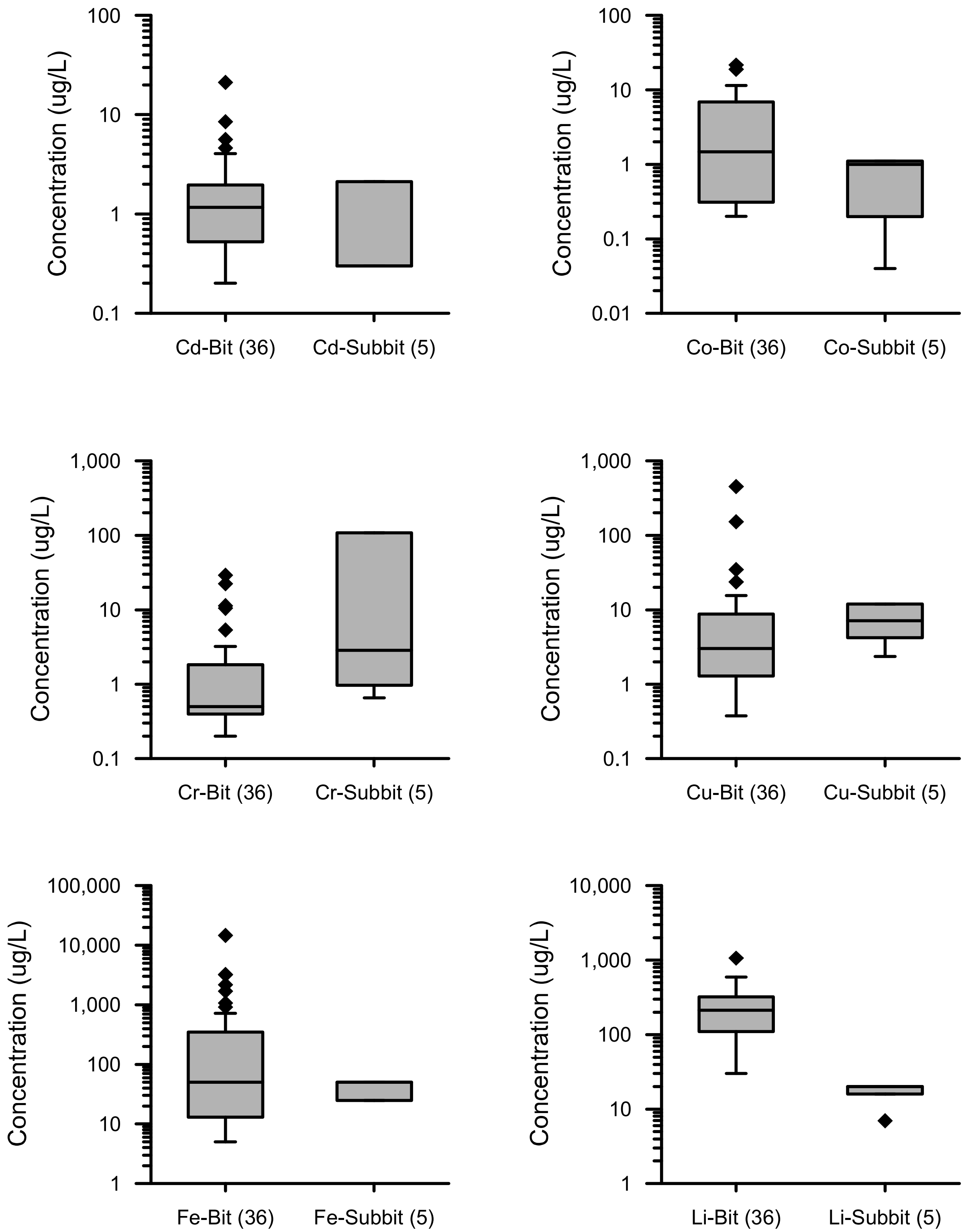


Figure C-4
Comparison of Field Leachate Concentrations: Bituminous vs. Subbituminous/Lignite Coal Ash, Impoundments (continued)

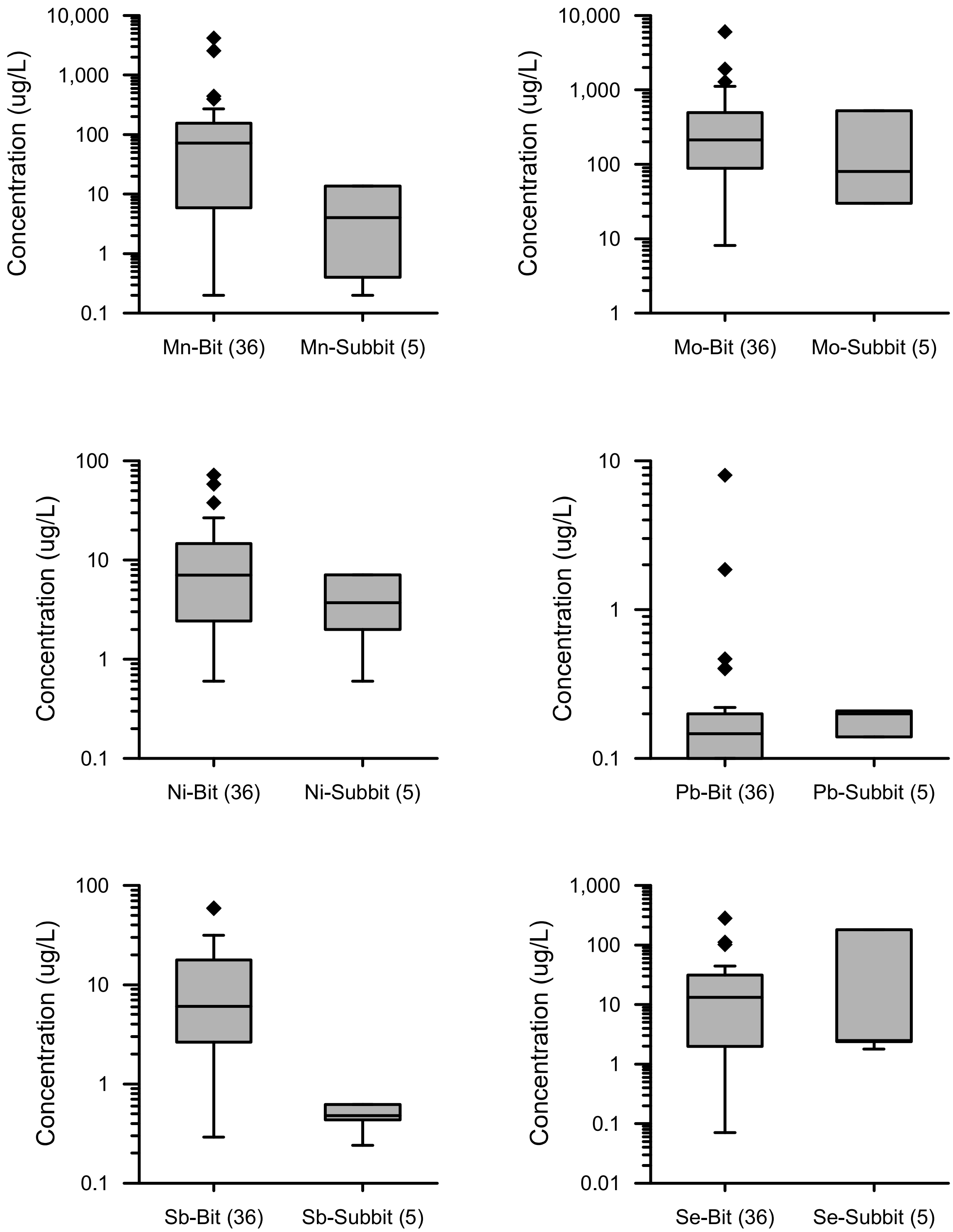


Figure C-4
Comparison of Field Leachate Concentrations: Bituminous vs. Subbituminous/Lignite Coal Ash, Impoundments (continued)

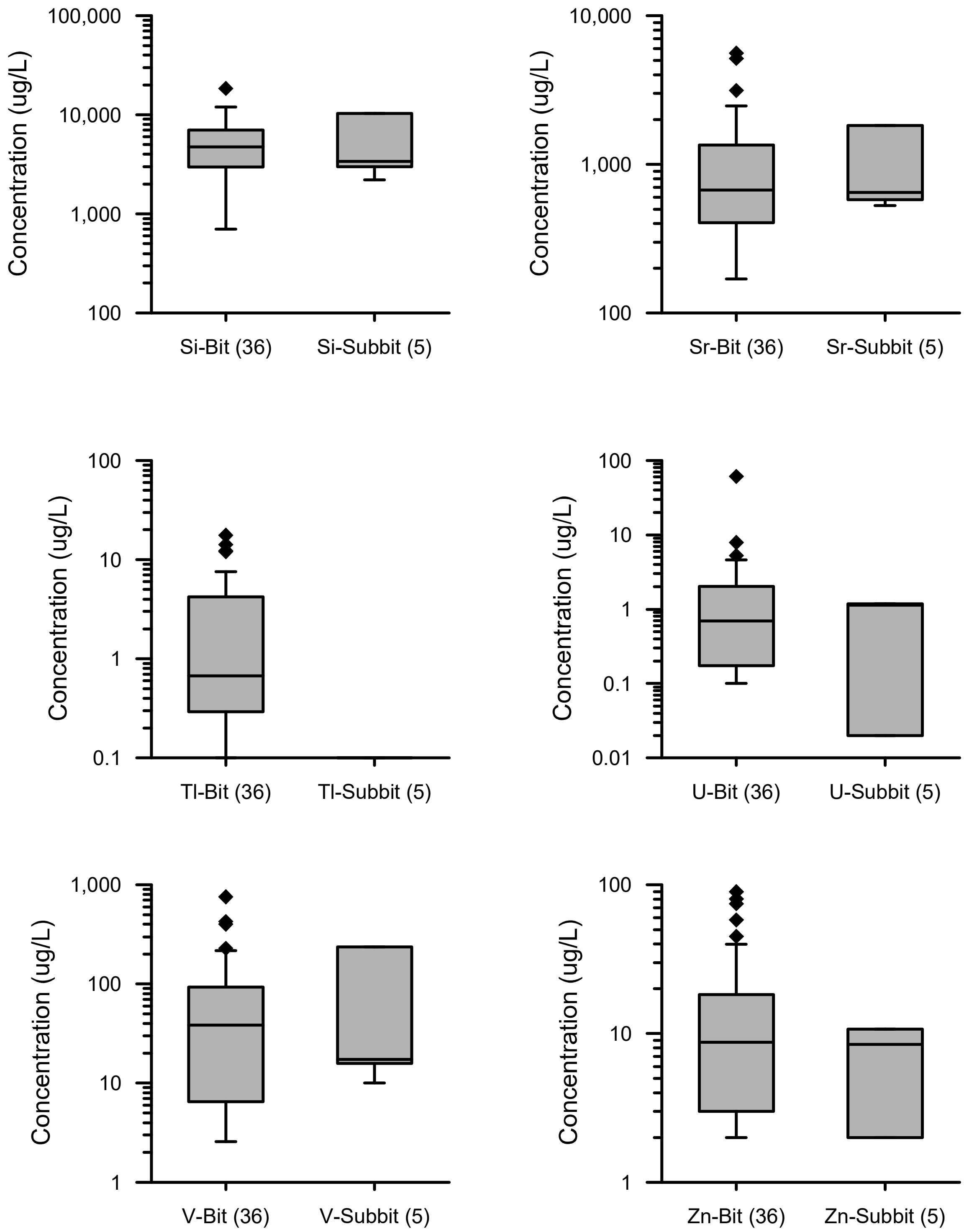


Figure C-4
Comparison of Field Leachate Concentrations: Bituminous vs. Subbituminous/Lignite Coal Ash, Impoundments (continued)

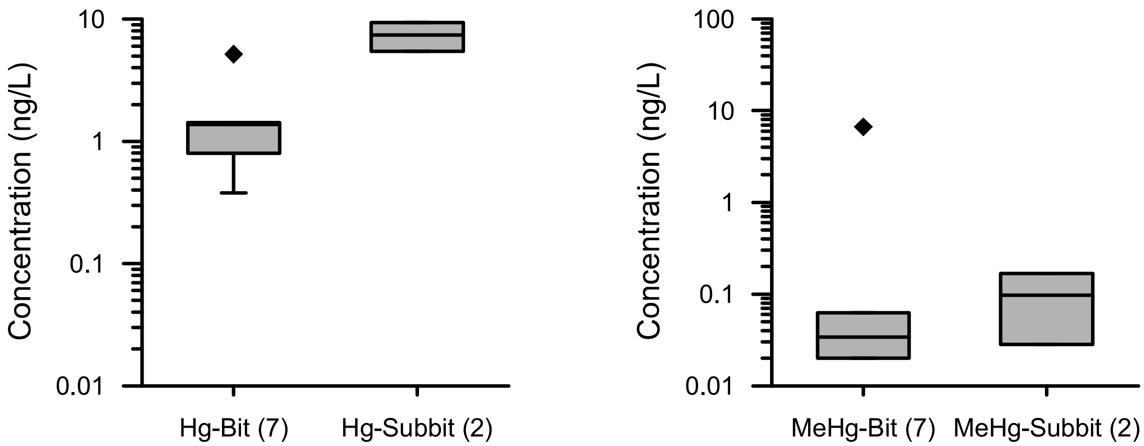


Figure C-4
Comparison of Field Leachate Concentrations: Bituminous vs. Subbituminous/Lignite Coal Ash, Impoundments (continued)

D

EVALUATION OF ARSENIC, SELENIUM, AND CHROMIUM SAMPLE PRESERVATION AND ANALYSIS METHODS

Cryofreezing Overview

Cryofreezing was used as the default sample preservation strategy for the speciation samples in this project for two reasons:

- Recent research has shown that both arsenic and selenium form soluble sulfur species in sulfidic waters, which are decomposed and precipitated under acidic conditions, thereby completely altering the original speciation information. This would have affected all samples that contain detectable concentrations of "other" arsenic or selenium species, although in most cases, these "other" species constituted less than 10 percent of the total concentration of the element, and so the associated error would have been relatively small. However, six samples (five arsenic and one selenium) contained "other" species at fractions > 10 percent of the corresponding total arsenic or selenium concentration. Since it wasn't known in advance how strongly sulfidic the sampled waters would be, and field observations confirmed (via smell) that some samples had significant concentrations of free reduced sulfur compounds, cryofreezing was used instead of acidification to prevent decomposition of soluble arsenic- and selenium-sulfur compounds.
- It is well established that Cr(VI) gets reduced by dissolved organic matter in acidified samples during storage. Since nearly all samples containing elevated chromium concentrations had Cr(VI) as their major species, this could have led to significantly altered chromium speciation results. Again, cryofreezing circumvents the issue of pH change during storage. This was confirmed in a test of preservation methods performed in 2004 (after analytical issues had been observed in 2003); while the cryofrozen split yielded almost exclusively Cr(VI), acidified splits yielded lower Cr(VI) concentrations (see Table D-2) and increasing Cr(III) concentrations over time. This already led to an altered chromium speciation pattern immediately after sample receipt, but yielded a completely reversed speciation result after several weeks of storage. For this reason, Cr(VI) is typically preserved under strongly alkaline conditions, but for the present project, this would have created other analytical issues related to the precipitation of Cr(III) and major trace elements (e.g. iron and manganese), and was thus avoided.

Unfortunately, during the analysis of samples collected in 2003, it was observed that the cryofreezing approach created another, unanticipated problem, during storage. When the cryofrozen samples were thawed prior to analysis, varying degrees of white-yellowish precipitates were observed in many samples, which did not re-dissolve at room temperature (over a time frame of weeks). When speciation analyses of these samples were conducted, a significant gap in the mass balance (= total element concentration – sum of its individual species) of arsenic and/or selenium was observed; chromium was not significantly affected by this issue. It was theorized that these precipitates were calcium sulfate or carbonate, and geochemical model calculations confirmed that the solubility of these minerals was exceeded in many samples.

To test if the precipitates contained the "missing" fractions of arsenic (for which the mass balance discrepancies were worse than for selenium), the precipitates were digested in nitric acid, and the resulting solutions analyzed for arsenic released from the precipitates. Table D-1 shows that for some samples, the "missing" fraction of arsenic was apparently indeed bound to the observed precipitates, but there are more samples than that for which this did not confirm the postulated loss mechanism. Additionally, significant mass balance discrepancies were also observed in samples containing no visible precipitates. Therefore, while this storage artifact was certainly responsible for incomplete arsenic or selenium speciation mass balance in some samples, it was definitely not the only process involved, and possibly not even the major one. Dissolution of the precipitates in nitric acid changes arsenic speciation, so it remains unclear if any one species of arsenic was selectively or preferentially removed from solution during the formation of the precipitates.

Formation of these precipitates was only observed in samples collected in 2003, because those samples were stored for a long period (up to 6 months) prior to analysis. By comparison, samples collected in 2004 and 2005 were typically analyzed for their arsenic and selenium speciation within four weeks after collection, and the sum of species in these samples was closer to the total concentration than in the 2003 samples. Consequently, it seems likely that the formation of precipitates resulted from excessively long cryofrozen storage, and can be avoided by keeping storage time to one month or less. Attempts to "recreate" the precipitates were unsuccessful (on a time scale of weeks), so no further attempts were made to resolve the issue and correct the speciation mass balance for samples with precipitates.

Table D-1
Arsenic Speciation Mass Balance, Including Losses to Precipitates Formed During Cryofrozen Storage, for Leachate Samples Collected In 2003

Sample ID	Lab ID	Total As	As(III)	As(V)	other As species	precipitated As	mass balance without precipitated As [%]	mass balance including precipitated As [%]
001	1	20.4	< 0.3	9.5	2.1	7.04	57	91
002	2	48.4	< 6	47	< 6	1.10	98	100
003	3	84	< 6	69	< 6	7.50	82	91
004	4	18.6	8.4	5.2	< 0.3	0.59	73	76
005	5	3.0	< 0.2	1.3	< 0.2	0.08(a)	45	47
006	6	12.2	< 0.3	0.9(a)	< 0.3	<0.05	8	8
007	7	20.1	< 2	< 2	< 2	0.07(a)	0	0
008	8	16.9	0.7(a)	< 0.5	< 0.3	0.07(a)	4	5
009	9	28.9	< 6	< 10	< 6	0.09(a)	0	0
010	10	22.3	1.5(a)	10	< 0.6	0.46	52	54
011	11	4.8	< 0.2	0.6	< 0.2	0.26	12	17
012	12	238	97.0	66	< 0.6	38.1	69	85
013	13	21.6	3.7	< 0.5	< 0.3	11.8	17	72
013D	13A	22	1.9	< 0.5	< 0.3	NA	9	9
014	14	163	1.9	86	0.9(a)	25.1	54	70
015	15	23.8	< 0.6	24	< 0.6	1.72	99	106
016	16	68.6	< 0.6	25	< 0.6	23.4	36	70
SX-1	core 3	72.0	0.9	46.9	< 0.1	1.16	66	68
017	17	4.11	0.88	<0.08	0.1	0.26	23	30
018	18	23.1	0.42	5.22	< 0.06	17.8	24	101
019	19	5.11	0.57	<0.08	< 0.06	0.36	11	18
020	20	4.19	1.00	0.53	0.1	0.14(a)	40	43
HN-1	core 1	59.8	< 0.1	33.6	0.2	5.65	57	66
HN-2	core 2	20.6	< 0.1	6.9	0.1	1.64	34	42
021	21	194	2.1	208	< 0.3	2.38	108	110
022	22	11.1	12.5	0.49	< 0.06	0.11(a)	118	119
023	23	218	0.8(a)	189	< 0.3	12.4	87	93
024	24	11.2	0.4(a)	<0.2	< 0.2	1.47	3	16
025	25	6.47	1.35	<0.08	< 0.06	1.04	21	37
026	26	10.8	11.2	0.4(a)	< 0.2	0.11(a)	107	108
027	27	39.1	13.2	4.8	1.3	2.31	49	55
028	28	30.0	2.4	1.7	0.2	0.17(a)	14	15
029	29	48.9	1.7	8.9	0.3	4.01	22	31
030	30	42.5	3.5	29.5	0.4	0.58	79	80
031	31	221	201	23.6	0.7	3.65	102	103
032	32	25.4	17.5	16.9	0.1	0.43	136	137

(a) = sample concentration less than 5 times blank
Concentrations in μg/L

Due to the large heterogeneity of the collected sample set, additional issues related to speciation preservation were observed in individual samples. Some samples showed obvious loss of total arsenic, selenium, and/or chromium upon acidification, which was verified by analyzing total arsenic, total selenium, and total chromium in the cryofrozen speciation samples (and finding significantly higher concentrations). For those samples, the formation of a brownish flocculate was usually observed in the acidified splits, which is probably due to precipitation of humic acids (which are soluble under the original alkaline conditions present in most samples, but insoluble at acidic pH). Evidently, the precipitates removed a fraction of total arsenic, selenium, or chromium from solution, which would have led to a speciation mass balance > 100 percent (barring other analytical issues). In such cases, the corresponding total element concentration measured in the cryofrozen split was used instead of the one in the acidified sample. By contrast, there were also a number of samples in which the formation of brownish precipitates was observed in the non-acidified splits taken for major anion and cation analysis. This reflects the precipitation of iron (oxy)hydroxide minerals caused by oxidation of high Fe(II) concentrations present in reducing waters. This problem was avoided by acidification, unless the process was so rapid that it began as the sample was being pumped and filtered.

In conclusion, the preservation for arsenic and selenium speciation by acidification does not appear suitable for the whole collected sample set, and must certainly be avoided for chromium speciation. Cryofreezing appears to be suitable in principle, but the sample storage time must be minimized to avoid irreversible formation of precipitates. Finally, it appears that the collected sample set is too heterogeneous for any one procedure that will preserve arsenic, selenium, and chromium speciation in all samples reliably; therefore, it might be necessary to collect multiple splits in parallel that are preserved differently.

Evaluation of Preservation Arsenic, Chromium, and Selenium Speciation by Preservation Method

The field team returned to the location of sample 002 and collected replicate samples for analysis of preservatives and differences associated with analytical laboratories. Five preservation techniques were used: no preservation, hydrochloric acid (HCl) in opaque bottles, hydrochloric acid in foil-wrapped (dark) bottles, ethylenediaminetetraacetic acid (EDTA), and nitric acid (HNO_3). Sample 002 is geochemically characterized by alkaline pH (>10), ORP of > 200, low dissolved oxygen (0.2%), low iron (<50 µg/L), and high sulfate (> 6,000 mg/L) concentration.

Results varied by analyte, preservation method, and laboratory (Table D-2). Chromium was most strongly effected. Concentrations of Cr(VI) in the acid-preserved samples were less than one-half of the concentration determined in the cryofrozen and unpreserved samples. This analysis clearly suggests that acid-preservation is not an appropriate technique for Cr(IV) in this geochemical environment.

Selenium concentrations were least affected by preservation technique. The poorest result was for the cryofrozen sample (sample 002), in which the sum of species was 76 percent of the total selenium concentration. This sample was collected in 2003 and subject to the issues described above associated with long hold times. The only apparent laboratory related relationship was for Se(IV); which was below detection limits in all samples other than the cryofrozen sample analyzed by laboratory 1, and detected at concentrations ranging from 76 to 94 µg/L by laboratory 2.

Table D-2
Arsenic, Selenium, and Chromium Speciation Using Different Preservatives

	As (III)	As (V)	As (other)	As species	Total Arsenic	% Recovery
Field blank	<5	0.02	NA	NA	0.24	NA
Unpreserved, Lab 1	<5	27.1	6.4	33.5	58.1	58
Unpreserved, Lab 2	4.1	63	NA	67	73	92
Cryofrozen, Lab 1	<6	47	<6	47	48.4	97
0.5% HCl preserved, Lab 1	<5	30.8	9.7	40.5	54.7	74
0.5% HCl preserved, Lab 2	4.9	95	NA	100	82	122
0.5% HCl+ dark preserved, Lab 1	<5	32.2	4.6	36.8	54.9	67
0.5% HCl+ dark preserved, Lab 2	NA	NA	NA	NA	NA	NA
EDTA preserved, Lab 2	4.0	72	NA	76	71	107
0.5% HNO_3 preserved, Lab 1	<5	5.1	2.4	7.5	51.7	15
0.5% HNO_3 preserved, Lab 2	3.7	65	NA	69	82	84
	Cr (III)	**Cr(VI)**	**Cr (other)**	**Cr species**	**Total Chromium**	**% Recovery**
Field blank	NA	<0.1	NA	NA	0.11	NA
Unpreserved, Lab 1	NA	4138	NA	NA	5204	NA
Unpreserved, Lab 2	NA	NA	NA	NA	NA	NA
Cryofrozen, Lab 1	340	5090	NA	5430	5100	106
0.5% HCl preserved, Lab 1	NA	2161	NA	NA	5217	NA
0.5% HCl preserved, Lab 2	NA	NA	NA	NA	NA	NA
0.5% HCl+ dark preserved, Lab 1	NA	1314	NA	NA	5242	NA
0.5% HCl+ dark preserved, Lab 2	NA	NA	NA	NA	NA	NA
EDTA preserved, Lab 2	NA	NA	NA	NA	NA	NA
0.5% HNO_3 preserved, Lab 1	NA	1760	NA	NA	5161	NA
0.5% HNO_3 preserved, Lab 2	NA	NA	NA	NA	NA	NA
	Se(IV)	**Se(VI)**	**Se (Other)**	**Se species**	**Total Selenium**	**% Recovery**
Field blank	<0.05	<0.05	NA	<0.05	0.14	--
Unpreserved, Lab 1	<25	1432	16	1448	1312	110
Unpreserved, Lab 2	94	1270	NA	1364	1400	97
Cryofrozen, Lab 1	19	1300	NA	1319	1730	76
0.5% HCl preserved, Lab 1	<25	1348	27	1375	1426	96
0.5% HCl preserved, Lab 2	91	1423	NA	1514	1500	101
0.5% HCl+ dark preserved, Lab 1	<25	1349	14	1363	1424	96
0.5% HCl+ dark preserved, Lab 2	NA	NA	NA	NA	NA	NA
EDTA preserved, Lab 2	87	1478	NA	1565	1400	112
0.5% HNO_3 preserved, Lab 1	<25	1307	NA	1307	1392	94
0.5% HNO_3 preserved, Lab 2	76	1416	NA	1492	1400	107

Samples collected 4/6/04 except Cryofrozen sample collected 8/5/03
Lab 2 did not analyze chromium
NA=not analyzed

Arsenic concentrations were most variable. First, there was a significant difference by laboratory. Laboratory 1 returned total arsenic concentrations between 52 and 58 mg/L (excluding the cryofrozen sample, which was collected on a different date), while laboratory 2 returned total arsenic concentrations between 71 and 82 mg/L. Laboratory 2 also achieved greater species recovery (84 to 122%) than laboratory 1 (15 to 97 percent). For laboratory 2, all preservation methods proved acceptable for preservation of arsenic species. For laboratory 1, only the cryofrozen sample yielded better than 80 percent species recovery. Significantly, all preservation methods identified As(V) as the species with highest concentration.

This test was performed on samples from a geochemical environment where the oxidized species would be expected in leachate samples, and results cannot be extrapolated to other environments, particularly those where the reduced species may be expected. However, the results show that several different preservation methods are capable of identifying the predominant species of arsenic and selenium in water samples from a high pH, high ORP, low oxygen, low iron, high sulfate environment. However, only cryofreezing adequately preserved chromium species.

Comparison of Cryofrozen and Hydrochloric Acid-Preserved Replicate Samples

Splits of 32 field leachate samples[6] were preserved in the field with HCl and forwarded to a separate laboratory (laboratory 2) for analysis of arsenic and selenium species. Analyses were performed as described in Section 2.

Arsenic

For arsenic, the cryofrozen sample sets[7] typically had lower total concentration than the acid-preserved samples (Figure D-1); however, since the total concentration analyses by both labs were performed on acid-preserved samples, this difference is laboratory related, rather than preservative-related. The percentage difference in total concentration was greatest when values were lower than 10 µg/L; the average difference for samples with concentration greater than 10 µg/L was 27 percent. The difference may be due to a correction applied by laboratory 2 to account for chloride interference.

The sum of arsenic species was compared to the independently measured total arsenic to determine the species recovery. For both sets of samples, the species recovery was typically closer to 100 percent when the total concentration was greater than 10 µg/L. In most cases, the cryofrozen sample had a higher species recovery, and was closer to 100 percent species recovery, than the acid-preserved sample (Figure D-1).

[6] The split sample comparison included one sample (085) that was taken at one of the field sites for another study, and is not otherwise included in this evaluation. The acid-preserved splits of samples 084 and 085 were not analyzed for selenium species.
[7] The cryofrozen sample sets included acid-preserved samples for total analysis and frozen samples for species analysis.

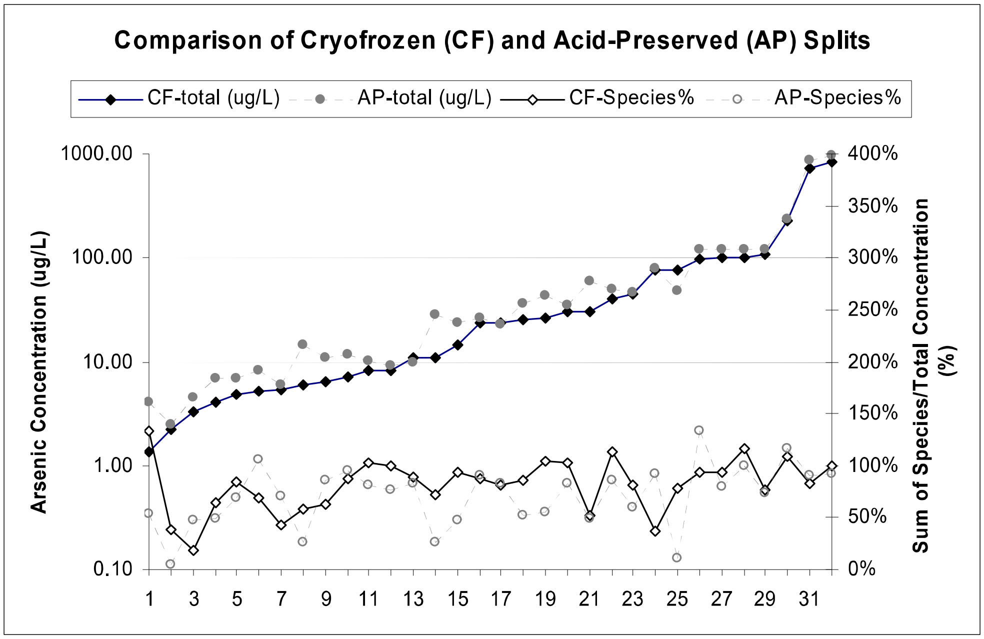


Figure D-1
Comparison of Total Arsenic Concentration and of Percent Species Recovery for Cryofrozen and Acid-Preserved Sample Splits

The dominant species in each sample split was determined based on the following criteria:

- For species recovery greater than 80 percent, a species was identified as dominant if its concentration was 60 percent or more of the sum of species.
- If species recovery was greater than 80 percent, and no species concentration was greater than 60 percent of the sum of species, then the sample was listed as "neutral".
- For species recovery less than 80 percent, a species was identified as dominant if its concentration was greater than 50 percent of the total concentration.[8]
- Samples with less than 80 percent species recovery in which no species concentration was greater than 50 percent of the total concentration were not tabulated.

Based on this approach, 27 of the 32 cryofrozen samples, and 22 of the 32 acid-preserved samples can be classified as dominated by As(III), dominated by As(V), or neutral (Table D-3). In 17 of the 20 common splits (where the dominant species could be determined in both samples), the two preservation techniques yielded similar results. In the three splits with different results, As(V) was dominant in the cryofrozen sample and As(III) in the acid-preserved sample. Two of these three samples had total arsenic concentration lower than 5 µg/L; the other was sample 106, which had an arsenic concentration of 110 µg/L.

[8] If the sum of species is 80 percent, and the species concentration is 50 percent of the total concentration, then that species accounts for at least 62.5 percent of the sum of species.

Table D-3
Dominant Arsenic Species in Split Samples

Cryofrozen								Acid-Preserved					
Split	% As(III)	% As(V)	% other	% recov.	DS	Total As		Split	% As(III)	% As(V)	% recov.	DS	Total As
T112	50%	70%	14%	133%	V	1.36		W112	54%	0%	54%	(III)	4.04
T101	0%	10%	28%	38%		2.23		W101	0%	0%	4%		2.50
T92	0%	15%	3%	18%		3.34		W92	0%	47%	47%		4.52
T108	9%	56%	0%	65%	(V)	4.09		W108	0%	48%	48%		6.91
T99	2%	78%	4%	84%	V	4.80		W99	69%	0%	69%	(III)	6.79
T126	0%	69%	0%	69%	(V)	5.20		W126	0%	106%	106%	V	8.32
T49	0%	43%	0%	43%		5.40		W49	20%	51%	71%	(V)	5.94
T111	0%	58%	0%	58%	(V)	5.94		W111	0%	27%	27%		14.32
T127	0%	63%	0%	63%	(V)	6.42		W127	0%	86%	86%	V	10.77
T102	0%	88%	0%	88%	V	7.24		W102	0%	94%	94%	V	11.74
T116	12%	90%	1%	103%	V	8.24		W116	10%	71%	81%	V	10.26
T115	37%	63%	0%	100%	V	8.32		W115	0%	77%	77%	(V)	9.08
T91	0%	88%	1%	89%	V	10.76		W91	0%	83%	83%	V	9.98
T121	12%	54%	5%	72%	(V)	11.00		W121	0%	26%	26%		28.36
T128	71%	20%	3%	94%	III	14.27		W128	44%	4%	48%		24.00
T114	0%	87%	0%	87%	V	23.53		W114	9%	81%	90%	V	26.50
T42	0%	81%	0%	81%	V	23.70		W42	8%	75%	83%	V	23.26
T122	30%	32%	24%	86%	neutral	25.54		W122	44%	8%	52%		36.28
T120	27%	43%	35%	104%	neutral	26.79		W120	44%	12%	56%		43.46
T119	0%	101%	1%	102%	V	30.20		W119	3%	79%	82%	V	34.74
T107	3%	49%	0%	52%		30.64		W107	2%	47%	48%		60.00
T118	2%	112%	0%	114%	V	40.78		W118	18%	67%	85%	V	48.94
T97	0%	81%	0%	81%	V	44.89		W97	0%	60%	60%	(V)	46.96
T43	0%	37%	0%	37%		75.20		W43	59%	32%	92%	neutral	77.76
T98	1%	77%	0%	79%	(V)	76.85		W98	10%	0%	10%		47.96
T57	0%	94%	0%	94%	V	98.60		W57	0%	133%	133%	V	120.00
T69	0%	94%	0%	94%	V	99.50		W69	0%	80%	80%	(V)	120.00
T113	1%	115%	0%	116%	V	101.98		W113	23%	76%	99%	V	120.00
T106	14%	57%	5%	77%	(V)	109.83		W106	71%	2%	73%	(III)	122.32
T105	85%	22%	2%	109%	III	229.95		W105	112%	5%	116%	III	233.00
T84	10%	74%	0%	83%	V	726.90		W84	8%	83%	90%	V	870.00
T85	59%	38%	3%	99%	neutral	829.10		W85	52%	41%	93%	neutral	950.00

DS indicates the dominant species in the sample, ( ) indicates that total species recovery was less than 80%, but one species was greater than 50%
Shading indicates samples where the dominant species could be determined in both splits.

Sample 106 was recirculated FGD system water, presenting a highly alkaline (pH near 12) and more concentrated matrix that may have confounded the analyses. Other complicating factors with sample 106 included high dissolved oxygen (95%) yet low ORP (18 mV), and low dissolved iron (4.6 µg/L).

Selenium

For selenium, the cryofrozen sample sets[9] typically had lower total concentration than the acid-preserved samples (Figure D-2). This difference, which, like arsenic, is laboratory related, was greatest when total concentration was lower than 10 µg/L; the average difference for samples with concentration greater than 10 µg/L was 25 percent.

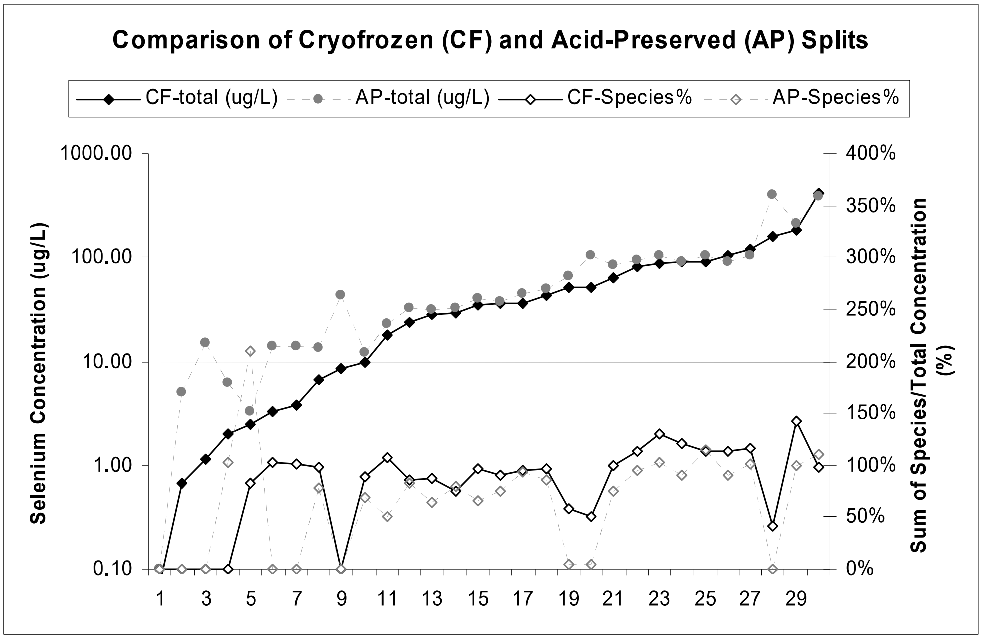


Figure D-2
Comparison of Total Selenium Concentration and of Percent Species Recovery for Cryofrozen and Acid-Preserved Sample Splits

The sum of species for both sets of samples was closer to 100 percent when the total concentration was greater than 10 µg/L. The cryofrozen split typically had higher species recovery than the acid-preserved split; although in some cases, particularly at concentrations near and greater than 100 µg/L, the cryofrozen split recovery was greater than 100 percent and the acid-preserved split recovery was closer to 100 percent. For concentrations greater than 10 µg/L, species recovery correlated well between the two preservation methods (Figure D-2).

The dominant selenium species was determined using the same approach as for arsenic. Based on this approach, 23 of the 30 cryofrozen sample splits, and 20 of the 30 acid-preserved sample splits can be classified as dominated by Se(IV), dominated by Se(VI), or neutral (Table D-4).

[9] The cryofrozen sample sets included acid-preserved samples for total analysis and frozen samples for species analysis.

Table D-4
Dominant Selenium Species in Split Samples

Cryofrozen								Acid-Preserved					
Split	% Se(IV)	% Se(VI)	% other	% recov.	DS	Total As		Split	% Se(IV)	% Se(VI)	% recov.	DS	Total As
T114	0%	0%	0%	0%		0.07		W114	0%	0%	0%		0.10
T112	0%	0%	0%	0%		0.67		W112	0%	0%	0%		5.00
T122	0%	0%	0%	0%		1.13		W122	0%	0%	0%		15.00
T99	0%	0%	0%	0%		2.04		W99	103%	0%	103%	IV	6.12
T57	83%	0%	0%	83%	IV	2.44		W57	210%	0%	210%	IV	3.23
T120	56%	46%	0%	102%	neutral	3.30		W120	0%	0%	0%		14.00
T121	29%	73%	0%	102%	VI	3.86		W121	0%	0%	0%		14.00
T108	39%	59%	0%	98%	neutral	6.56		W108	38%	39%	77%		13.32
T105	0%	0%	0%	0%		8.47		W105	0%	0%	0%		43.00
T49	83%	6%	0%	89%	IV	10.00		W49	70%	0%	70%	(IV)	12.01
T118	100%	7%	0%	107%	IV	17.62		W118	51%	0%	51%	(IV)	23.00
T43	86%	0%	0%	86%	IV	23.50		W43	83%	0%	83%	IV	32.54
T119	81%	6%	0%	87%	IV	27.95		W119	65%	0%	65%	(IV)	32.00
T113	66%	9%	0%	75%	(IV)	29.27		W113	79%	0%	79%	(IV)	33.00
T116	87%	9%	0%	96%	IV	35.35		W116	66%	0%	66%	(IV)	40.00
T115	82%	8%	0%	90%	IV	36.10		W115	75%	0%	75%	(IV)	37.00
T69	91%	5%	0%	96%	IV	36.40		W69	87%	7%	93%	IV	44.54
T42	92%	5%	0%	96%	IV	42.60		W42	80%	6%	86%	IV	49.94
T98	58%	0%	0%	58%	(IV)	50.74		W98	5%	0%	5%		65.98
T128	34%	13%	3%	51%		50.90		W128	0%	5%	5%		106.36
T106	0%	99%	0%	99%	VI	64.79		W106	3%	73%	76%	(VI)	85.44
T102	7%	106%	0%	113%	VI	80.48		W102	5%	89%	94%	VI	95.40
T126	14%	117%	0%	131%	VI	88.70		W126	14%	88%	102%	VI	104.34
T111	43%	79%	0%	122%	VI	90.54		W111	38%	53%	91%	neutral	91.00
T101	0%	114%	0%	114%	VI	91.00		W101	0%	115%	115%	VI	104.48
T92	1%	113%	0%	113%	VI	103.36		W92	0%	90%	90%	VI	90.86
T91	3%	113%	0%	116%	VI	122.22		W91	0%	102%	102%	VI	102.84
T107	0%	10%	32%	42%		159.00		W107	0%	0%	0%		400.00
T127	7%	136%	0%	143%	VI	180.60		W127	5%	95%	100%	VI	210.00
T97	9%	89%	0%	98%	VI	412.50		W97	16%	95%	111%	VI	380.00

DS indicates the dominant species in the sample, ( ) indicates that total species recovery was less than 80%, but one species was greater than 50%
Shading indicates samples where the dominant species could be determined in both splits.

In 18 of the 19 common splits (where the dominant species could be determined in both samples), the two preservation techniques yielded similar results. The only exception was sample 111, which was dominated by Se(VI) in the cryofrozen split and was neutral in the acid split. However, both samples had more Se(VI) than Se(IV). The species breakdown for sample 111 was 43 percent Se(IV) and 79 percent Se(VI) in the cryofrozen sample, and 38 percent Se(IV) and 53 percent Se(VI) in the acid-preserved sample. Sample 111 had neutral pH (7.2), was oxic (280 mV ORP and 59 percent dissolved oxygen), and did not exhibit a sulfur odor; as a result, the acid-preserved sample would not be expected to undergo precipitation of soluble sulfur species.

Summary

In summary, there are conditions under which one of the preservation methods may be more appropriate than the other. However, the split sample data collected during this study indicate that the preservation method does not affect results sufficiently to alter interpretation of the dominant species present in the sample.

E

LABORATORY ANALYTICAL ISSUES PERTAINING TO SPECIATION ANALYSIS

Determination of Total Arsenic, Selenium, and Chromium Concentrations

The determination of total chromium (TCr) by ICP-MS worked very well. Good agreement was obtained between the two isotopes ^{52}Cr and ^{53}Cr, as well as between the two instruments used (ICP-DRC-MS and ICP-DF-MS). Therefore, there is a high degree of confidence in the reported total chromium results, and they are not a reason if the speciation mass balance for chromium did not work out in any sample, which usually only happened in samples with low total chromium concentrations. Unfortunately, the determination of total arsenic and selenium by ICP-MS is more complicated than that of total chromium, and consequently, the quality of these data is somewhat impaired in certain samples, as discussed below. The problems associated with the determination of total arsenic and selenium by ICP-MS stem mostly from molecular interferences that overlap with the mass of the measured arsenic or selenium isotopes, and thus yield artificially-increased results. These interferences are caused either by constituents of the measured water samples or by molecules formed in the argon plasma used in ICP-MS analyses. To illustrate this problem, the method used for total selenium determination in the collected water samples is explained below.

In ICP-MS analyses, it is desirable to use the major isotope of the trace element of interest for its quantification, because it yields the highest signal, which usually translates into the lowest detection limit. Additionally, at least one other isotope of the same element should be measured, and if the concentrations determined in the sample by using two (or more) different isotopes agree well, then there is a high degree of confidence that this result is correct and not impaired by any significant molecular interferences. For selenium, the main isotope is ^{80}Se, but this isotope is impossible to measure by conventional ICP-MS instruments, because the argon plasma generates a large amount of the dimeric ion $^{40}Ar_2^+$, which has the same nominal mass as the ^{80}Se isotope, and the two signals cannot be separated. Although some publications suggest that ICP-DF-MS can resolve the overlap between analyte and interference for this example when it's used in the high resolution mode, the particular ICP-DF-MS instrument used by laboratory 1 did not achieve this separation consistently, and an ICP-DRC-MS instrument was used to address this issue, which was successful. The ICP-DRC-MS approach uses a cell with a reactive gas (here methane, CH_4) to break up the interference (by collision yielding two Ar atoms of mass 40) between the plasma and the mass spectrometer, while the analyte ^{80}Se remains unaffected, and can thus be determined free of the inference. However, in the collected water samples, there are additional interferences that complicate this approach. High bromide concentrations in the samples lead to the formation of the molecule $^{1}H^{79}Br^+$, which also has the nominal mass 80, but cannot be eliminated effectively by the reaction gas methane. Therefore, a second reaction gas (ammonia, NH_3) was added, which undergoes a chemical reaction with HBr, and thus forms reaction products that have masses other than 80, so ^{80}Se can be measured in waters containing bromide.

The minor isotopes used for confirmation of results obtained using the main isotope usually have different interferences than the main isotope, so if the results obtained for different isotopes agree, it is generally accepted that all known interferences have been removed efficiently, as intended during the method development. In the case of selenium, the control isotopes used were ^{78}Se and ^{82}Se, and it turns out that ^{78}Se has an interference from the plasma ($^{40}Ar^{38}Ar^+$), but not from bromide, while ^{82}Se has an interference from bromide ($^{1}H^{81}Br^+$), but not form the plasma, so the control strategy for these two interferences works very well. Unfortunately, due to the fact that the studied waters were often very complex and generally very different from site to site, there were additional interferences in some samples that could not be resolved by the described approach. While some additional interferences were identified, and their influence on the measured total selenium results was compensated for as much as possible (for example, it was found that copper formed ammonia clusters $Cu(NH_3)^+$ in the DRC, which interfered with the measurement of ^{80}Se and ^{82}Se), there remained some samples that either contained interferences that were not identifiable, or where known interferences exceeded the compensation capacity of the developed analytical method. In those cases, the total selenium concentrations determined using the three different selenium isotopes disagreed beyond the normal range of analytical error, and such results were flagged[10] in the results table (Appendix A). For such samples, the lowest total selenium concentration obtained with any selenium isotope was usually reported, because the molecular interferences are by nature positive (i.e. they mimic selenium), so the lowest result should be the least (or not) interfered.

Figure E-1 shows the agreement between the results obtained for the three measured selenium isotopes as a function of the total selenium concentration: With the exception of three samples, the total selenium concentrations determined using each of the three individual isotopes agree within the analytical uncertainty (± 10 percent) for samples containing total selenium greater than 5 µg/L. Generally, the agreement between the three selenium isotopes is good when total selenium concentrations are higher, and gets worse towards lower concentrations, because a certain amount of an interference caused by the sample matrix would have a bigger impact if the actual selenium signal is small, and because the analytical uncertainty itself increases with decreasing concentration. For those three samples with higher total selenium concentrations where the isotope agreement is not good, the reason probably lies in a combination of complex matrix (high salinity and trace element concentrations) and comparably low total selenium concentration (i.e. too low to resolve the interferences by dilution), although the actual reasons for these discrepancies likely vary from sample to sample, and were not explored further in this project. To eliminate this problem in future similar studies, it would be necessary to either add hydride generation (HG) as a sample introduction technique, which selectively volatilizes the selenium into the plasma while most of the other sample constituents stay behind in the liquid phase and are not introduced into the plasma (so they cannot produce interferences), or switch to a different detection technique altogether (e.g. atomic fluorescence spectrometry, AFS). There are also other potential analytical issues associated with HG and AFS, and there is no guarantee that these approaches would have resolved all problems for the present sample set.

[10] Identified in Table A-2 using flag (b), "isotope ratios do not match"

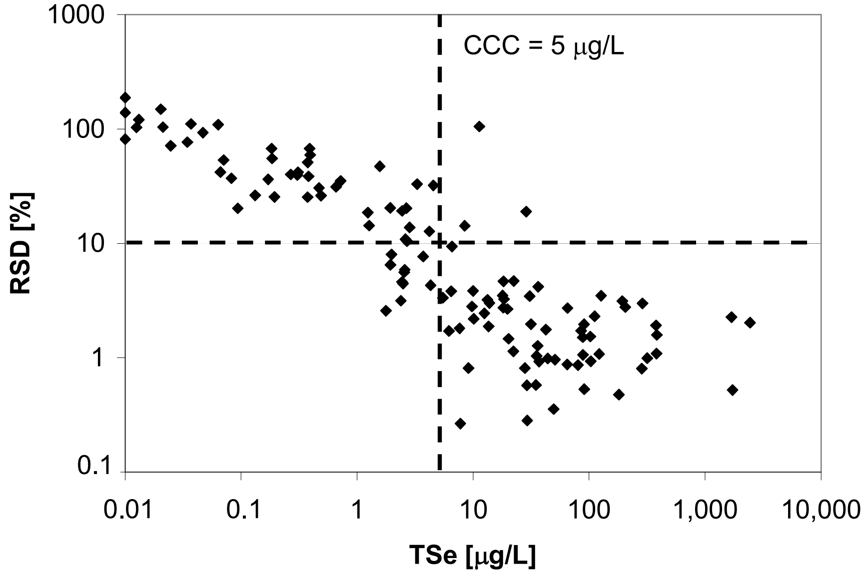


Figure E-1
Agreement Between Total Selenium Concentrations Determined Using the Isotopes ^{78}Se, ^{80}Se and ^{82}Se in All Collected Water Samples (expressed as percent relative standard deviation between the three individual results)

Besides interferences that affect individual selenium isotopes during the ICP-MS measurement, there are also matrix effects that affect all selenium isotopes at once, which relate to processes such as the sample introduction into the ICP-MS and the ionization of selenium in the plasma. The sample flow rate in ICP-MS measurements of bulk samples is regulated by the (constant) rotation speed and tubing diameter of a peristaltic pump, but the uptake of the sample into the plasma depends on its nebulization in the spray chamber; this process is assumed to be constant, and the fraction of the pumped sample nebulized is typically around 3 percent (so 97 percent of the sample goes to waste and is not measured). Parameters like the sample's viscosity or salinity can alter the nebulization process, and thus lead to higher or lower nebulization efficiency, thereby affecting the selenium signal obtained, which is proportional to the total amount of sample introduced into the plasma. To recognize and correct for such interferences, one or more internal standards (IS) are used, which are other trace elements spiked to the samples at a known concentration before analysis. The idea behind this is that a change in the sample introduction efficiency would affect the IS to the same degree as the analytes, and could thereby be compensated for mathematically.

The only condition that the IS needs to fulfill to be used for this correction approach is that it cannot be present in the samples in a measurable/significant concentration (so that the IS signal should always be constant if there were no sample uptake variations); for this reason, "exotic" elements like platinum group metals are commonly used for this purpose. In this project, rhodium was routinely used as the primary IS for total selenium measurements, and indium was used as a secondary IS to identify if there were problems associated with the rhodium measurement in any given sample. Several other commonly used IS elements were tried as well, but yielded less satisfactory results, usually because they occurred in the analyzed water samples in significant concentrations. The same was true to a lesser degree for indium, so it was not always usable as an IS, whereas rhodium generally fulfilled the absence condition. However, two additional problems were encountered related to the IS approach, which have not been reported in the literature before, and therefore were unanticipated and had to be recognized and dealt with during this project.

First, it was observed that certain matrix elements present in the studied waters produced interferences in the DRC process that mimicked one of the IS elements (for example, the strontium isotope ^{86}Sr forms an ammonia cluster $Sr(NH_3)^+$ in the DRC, which has the same nominal mass as the only rhodium isotope ^{103}Rh). This increases the apparent IS signal and suggests increased sample introduction efficiency for the particular sample, and since the analyte signal is normalized to the IS signal, leads to artificially decreased total selenium concentrations. This interference was recognized by the fact that the secondary IS was not elevated, and compensated for as much as possible by varying instrument parameter like the DRC gas flow rates and Rpa and Rpq (two DRC settings), but could not be eliminated altogether without compromising the efficiency with which the DRC removes the main interferences on the analytes (as discussed above). No alternate IS was found that fulfilled the absence condition and was not affected by this phenomenon, so more research is needed in this respect to find a way to compensate for this problem. One way to address the issue is the method of standard addition, where an interfered sample is measured repeatedly with varying amounts of the analyte added prior to analysis, but this procedure is impractical in routine operation, because every sample would need to be analyzed multiple times.

Secondly, it was noticed that the signal for either IS element increased unspecifically when high concentrations of a matrix element with similar or higher mass were present in the sample, e.g. barium (mass 137) increasing the IS signal for rhodium (mass 103) and indium (mass 115). This effect is the opposite of a well-known process in mass spectrometry called "space-charge effect", and could thus be referred to as "inverse space-charge effect". It was beyond the scope of this project to investigate the reasons for this observation, and the effect could not be eliminated by changing instrumental parameters, although it was moderated by increasing the acceleration voltage for the ions through the DRC. Like the previous interference, this issue causes an artificially-increased IS signal and thus leads to reduced total selenium concentrations. Contrary to interferences that lead to decreased sample introduction efficiency (and thereby to elevated apparent total selenium concentrations), these two effects would result in a positive speciation mass balance discrepancy (i.e. recovery > 100 percent), so since most samples showed a negative deviation in their selenium speciation mass balance, these two types of interferences did apparently not affect many of the measured samples; they may, however, explain why the sum of selenium species in some samples was significantly > 100 percent.

The second type of interference that is commonly compensated for by using internal standards relates to the ionization efficiency of the analyte in the plasma. This is a particular problem for selenium and arsenic, which have very high first ionization energies, and are ionized incompletely (25-50 percent) in the ICP. Major constituents of the matrix can alter the properties of the plasma, and thereby change the degree of ionization for these elements (and consequently their signal intensity); typical examples include major cations like sodium, which are easily ionized and thereby decrease the "energy" of the plasma, leading to reduced arsenic and selenium ionization, and organic carbon, which appears to enhance the ionization of arsenic and selenium by unknown mechanisms. Again, the IS could be used to compensate for these effects, but only if it shows a similar response to such interferences as the analytes of interest. This "similarity condition" is much harder to fulfill than the absence condition, and it's nearly impossible to fulfill them both perfectly for a large and inhomogeneous sample set, such as the present one. Of all tested IS elements, rhodium yielded that best results, but it has a significantly lower ionization energy than both arsenic and selenium, so that the analyte signals may have been suppressed in some samples without an effect on the IS. Again the result would be an artificially reduced total selenium or total arsenic concentration.

The preceding discussion makes it clear that the determination of total selenium in such complex samples as the studied waters is complicated, and that not all interferences can be compensated for, leading to possibly "wrong" total selenium concentrations, which in turn would impact the selenium speciation mass balance. This is probably one of the main reasons of why this mass balance did not work well in samples with low total selenium and high concentrations of certain matrix elements. Besides the mentioned HG sample introduction, an elegant way to eliminate many of the discussed interferences would be isotope dilution, which involves spiking a known amount of a particular selenium isotope to the sample prior to analysis. This is, however, expensive, because pure selenium isotopes would need to be obtained, and was consequently not available and could not be developed during this project. Given the (eco) toxicological importance of measuring relatively low total selenium concentrations in complex aqueous samples, this is an area which should be explored in future research, so that a much improved and reliable method for total selenium determinations by ICP-MS becomes available.

All analytical issues discussed above hold true for arsenic as well, but contrary to selenium, arsenic is monoisotopic, and consequently does not offer the possibility of compensating for (or even recognizing) certain interferences by "switching" to another isotope, which suggests that the total arsenic data quality should be poorer than for total selenium (which of course cannot be proven directly). The suggested improvements like HG sample introduction would also remedy many of the raised problems, and even isotope dilution with a long-lived arsenic radionuclide could be used for internal standardization. However, similar to selenium, these aspects were not explored during this project, and the fact that the arsenic speciation mass balance did not work well in some samples can certainly be partially attributed to problems associated with the total arsenic determination.

Determination of Arsenic, Selenium, and Chromium Speciation

The determination of Cr(III) and Cr(VI) by AEC-ICP-MS worked quite well, as supported by the reasonable chromium speciation mass balance. The only issue that was addressed during this project was the relatively high background caused by the presence of inorganic carbon in the used chromatographic eluant: this leads to the formation of $^{40}Ar^{12}C^{+}$, which interferes with the determination of the main chromium isotope ^{52}Cr, but this background was easily eliminated by using NH_3 as the reaction gas in the DRC.

For arsenic and selenium, the measurement of their speciation in the collected water samples was more complicated, and a number of significant interferences were encountered. These interferences are generally not related to the presence of spectral interferences, as discussed for the total arsenic and total selenium determinations above, because typically the interfering sample constituent is separated chromatographically in time from the analyte species. As an example, bromide in the samples will still produce a signal on mass 82, but this does not interfere with the measurement of Se(IV) or Se(VI), because the bromide signal either elutes before the Se(IV) peak, or–if the interfering peak is too large–Se(IV) at mass 77 can be used for quantification. Rather, besides the preservation/stability issues discussed above for the cryofrozen sample, the main problems encountered are caused by high salinity in some of the collected water samples, and by the presence of major trace elements that are incompatible with the chosen chromatographic conditions, so both are chromatographic issues occurring in the AEC, and not spectroscopic issues arising in the ICP-MS.

The salinity-based interference is caused by the fact that major anions, especially sulfate in the studied waters, are present in very high concentrations (up to 300 mmol/L), whereas the arsenic and selenium species are present in much lower concentrations (up to 9 µmol/L for selenium and 7 µmol/L for As), so the major anions are present in 30,000-fold excess. During the AEC analysis, the major anion competes with the trace element anions for binding sites on the chromatographic column, and if this competition becomes too strong, then the analytes are "flushed" out of the column without interacting properly with the stationary phase, which results in bad peak shapes that makes quantification inaccurate to impossible, and in the change of retention times, which makes identification uncertain or eliminates separation of different species altogether. The best way to eliminate this problem is by diluting the sample prior to analysis, but this approach is limited by the absolute concentration of the analytes in the same, so if the ratio of major anions to analytes is too large, the samples would have to be diluted to the point where the analytes fall below the detection limits to overcome the chromatographic problems.

This issue was encountered for a large number of the studied samples, and was addressed by modifying the AEC separation. Sulfate (instead of hydroxide) was used as the eluant anion, and this increases the tolerance of the separation for elevated sulfate concentrations in the sample (this approach is called "matrix matching"). However, even this remedy is limited by the absolute binding capacity of the column, so if the total amount of matrix anions injected exceeds this capacity, then proper separation of the analytes is no longer possible. Matrix matching yielded a significant improvement for the speciation mass balance of arsenic and selenium in many samples collected in 2004 and 2005, and for those samples where the mass balance still remained poor, there appeared to be a general correlation with the ratio of sample salinity to analyte concentration.

The second chromatographic issue was caused by high iron and especially manganese concentrations in some of the studied waters. Since the AEC separation is conducted under alkaline conditions (even after modification) to prevent the loss of acid-labile arsenic and selenium species, major sample constituents that precipitate under strongly alkaline conditions may cause problems. Although many of the collected samples were alkaline to begin with, the separation conditions were even more alkaline; this pH change during analysis particularly affected those samples that were acidic or circumneutral in the field. Under such conditions, manganese (and iron) can precipitate in the form of (oxy)hydroxide minerals within the AEC, and these precipitates bind the species As(V) and Se(IV) very strongly, which could lead to artificially low results for these two species. This issue was addressed by raising the pH of the eluant by about one unit, and by adding some oxalate into the eluent, which keeps manganese in solution. As for the salinity issue, though, there are limits to this approach, and the problems could not be eliminated in all samples, which is probably the main reason for the very low speciation mass balances encountered in some samples.

As the constitution of real world samples is highly variable and unpredictable, the best way to resolve this problem is by using more sensitive detection principles, because then the problematic samples can be diluted even more. At this point, though, ICP-MS is the most sensitive detection approach, even if certain ICP-MS instruments not available during this project may possibly yield lower detection limits for the AEC-ICP-MS determination of arsenic and selenium species than the used ICP-DRC-MS (in the standard mode for arsenic and selenium speciation). Further increases in detection sensitivity for arsenic and selenium can be achieved by using high-efficiency sample introduction systems, such as HG or membrane desolvation, between the AEC separation and the ICP-MS detection. This, however, is complicated and more expensive for use on a routine basis, and the required equipment was either not available permanently at Trent, or was incompatible with the relatively high chromatographic flow rates (and would thus have necessitated some modifications), so these options were not incorporated into the used methods. It should be noted, though, that AEC-HG-ICP-DRC-MS has been used successfully to measure selenium speciation at ng/L-levels in sea water, so this approach could be used in future studies, because it works in principle for the species As(III)/As(V) and Se(IV)/Se(VI), while its suitability for any other arsenic or selenium species is untested, which constitutes another reason why this technique was not routinely used in this project.

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Human and Ecological Risk Assessment of Coal Combustion Wastes

Draft

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August 6, 2007

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

The U.S. Environmental Protection Agency (EPA) is evaluating management options for solid wastes from coal combustion (e.g., fly ash, bottom ash, slag). As part of this effort, EPA is evaluating whether current management practices for coal combustion waste (CCW) pose risks to human health or ecological receptors. To inform this objective, EPA has conducted a nationwide assessment of the risks posed by CCW disposal practices across the country.

This report describes the results of the tiered, site-based, probabilistic (Monte Carlo) risk assessment of onsite CCW disposal practices at coal-fired power plants across the United States. These landfills and surface impoundments represent disposal practices for CCW reported in 1995. Although EPA acknowledges that management practices for CCW have improved since 1995, as documented in U.S. Department of Energy (DOE) (2006), EPA believes that characterizing risks from facilities observed in 1995 provides a snapshot of the potential risks from CCW disposal and can provide useful information as EPA evaluates CCW management options. In addition, the data available on these facilities' locations, environmental characteristics, and waste management units (WMUs) allow EPA to apply a site-based risk assessment approach that the agency believes characterizes the risks to human health and the environment from disposing CCW in landfills and surface impoundments.

In summary, this CCW risk assessment evaluates potential risk results at the 50th and 90th percentile exposure level, adopting a risk criteria of 10^{-5} for excess cancer risks. Potential noncancer and ecological risks are also evaluated at the 50th and 90th percentile levels, adopting a hazard quotient (HQ) risk criteria greater than 1 for noncancer effects to both human and ecological receptors. Overall, when all types of landfills and surface impoundments (as observed in 1995) are evaluated in aggregate, the risk at the 90th percentile exceeds the risk criteria for cancer and noncancer risks for certain constituents. There is no potential risk above the risk criteria (cancer and noncancer) found at the 50th percentile. The risk assessment also suggests that one of the most sensitive parameters in the risk assessment is infiltration rate. Infiltration rate is greatly influenced by whether and how a WMU is lined.

For humans exposed via the groundwater-to-drinking-water pathway, arsenic in CCW landfills poses a 90th percentile cancer risk of $5x10^{-4}$ for unlined units and $2x10^{-4}$ for clay-lined units. The 50th percentile risks are $1x10^{-5}$ (unlined units) and $3x10^{-6}$(clay-lined units). Risks are higher for surface impoundments, with an arsenic cancer risk of $9x10^{-3}$ for unlined units and $3x10^{-3}$ for clay-lined units at the 90th percentile. At the 50th percentile, risks for unlined surface impoundments are $3x10^{-4}$, and clay-lined units show a risk of $9x10^{-5}$. Five additional constituents have 90th percentile noncancer risks above the criteria (HQs ranging from greater than 1 to 4) for unlined surface impoundments, including boron and cadmium, which have been cited in CCW damage cases, referenced above. Boron and molybdenum show HQs of 2 and 3 for clay-lined surface impoundments. None of these noncarcinogens show HQs above 1 at the 50th percentile for any unit type.

Composite liners, which are used in the majority of new facilities constructed after 1995, effectively reduce risks from all pathways and constituents below the risk criteria (cancer and noncancer) for both landfills and surface impoundments[1].

Risks from clay-lined units, as modeled, are about one-third to one-half the risks of unlined

[1] These results suggest that with the 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).

units, but are still above the risk criteria used for this analysis.

Arrival times of the peak concentrations at a receptor well are much longer for landfills (hundreds to thousands of years) than for surface impoundments (most less than 100 years).

For humans exposed via the groundwater-to-surface-water (fish consumption) pathway, selenium (HQ = 2) and arsenic (cancer risk = $2x10^{-5}$) pose risks slightly above the risk criteria for unlined surface impoundments at the 90th percentile. For both constituents, lined 90th percentile risks and all 50th percentile risks are below the risk criteria. No constituents pose risks above the risk criteria for landfills at the 90th or 50th percentile.

Waste type has little effect on landfill risk results, but in surface impoundments, risks are up to 1 order of magnitude higher for codisposed CCW and coal refuse than for conventional CCW.

The higher risks for surface impoundments than landfills are likely due to higher waste leachate concentrations, a lower proportion of lined units, 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 the risk criteria for boron and lead at the 90th percentile, but 50th percentile risks are well below the risk criteria. For surface impoundments, 90th percentile risks for several constituents exceed the risk criteria, with boron showing the highest risks (HQ = 2,000). Only boron exceeds the risk criteria at the 50th percentile (HQ = 4). 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 are generally an order of magnitude or more below the risk criteria.

Background

EPA has conducted risk assessments to evaluate the environmental risks from CCW management practices,[2] including CCW disposal in landfills and surface impoundments. Although EPA determined (in April 2000) that certain CCWs were not subject to hazardous waste regulations and therefore would be subject to regulation as nonhazardous wastes, EPA did not specify regulatory options at that time. This risk assessment was conducted to identify and quantify human health and ecological risks that may be associated with current disposal practices for high-volume CCW, including fly ash, bottom ash, boiler slag, flue gas desulfurization (FGD) sludge, coal refuse waste, and wastes from fluidized-bed combustion (FBC) units. These risk estimates will help inform EPA's decisions about how to treat CCWs under Subtitle D of the Resource Conservation and Recovery Act.

Purpose and Scope of the Risk Assessment

The purpose of this risk assessment is to identify potential risks associated with CCW constituents, waste types, receptors, and exposure pathways, and to provide information about those scenarios that EPA can use to develop CCW management options.

The scope of this risk assessment is 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 a broad variety of climate, geologic, and land use settings. The large volumes of waste generated by these plants

[2] Details on EPA's previous CCW work can be found at http://www.epa.gov/epaoswer/other/fossil/ index.htm.

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.

Risk Assessment Methodology

To estimate the risks posed by the onsite management of CCW, this risk assessment determined the release of CCW constituents from landfills and surface impoundments, estimated the concentrations of these constituents in environmental media surrounding coal-fired utility power plants, and estimated the risks that these concentrations pose to human and ecological receptors. To evaluate the significance of these risks, the risk criteria adopted for this assessment are:

- An estimate of the excess lifetime cancer risk for individuals exposed to carcinogenic (cancer-causing) contaminants of 1 chance in 100,000 (10^{-5} excess cancer risk)
- An HQ (the ratio of predicted intake levels to safe intake levels) greater than 1 for constituents that can produce noncancer human health effects
- An HQ greater than 1 for constituents with adverse effects to ecological receptors.

In support of this risk assessment, EPA assembled a constituent database that includes leachate and total waste analyses 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 tiered risk assessment methodology (Figure ES-1) that 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 41 CCW constituents to identify the 26 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 do not require further analysis
- **Full-Scale Analysis**, which used a site-based Monte Carlo analysis to characterize at a national level the risks to human health and ecological receptors from onsite disposal (in landfills and surface impoundments) of CCW constituents that were not eliminated in the screening analysis.

The screening analysis looked at all probable exposure pathways from CCW management in landfills and surface impoundments and identified 21 CCW constituents and 3 exposure scenarios to evaluate in the full-scale analysis (Table ES-1). Exposure scenarios evaluated for people include contaminated groundwater being transported to drinking water wells from a CCW landfill or surface impoundment, and contaminated groundwater discharging into surface water and contaminating a nearby stream or lake where people catch and eat fish. The full-scale analysis also addressed ecological risk in the same waterbodies.

Constituents addressed in the full-scale analysis are listed in Table ES-2 along with the potential exposure pathways identified for full-scale modeling in the screening analysis.

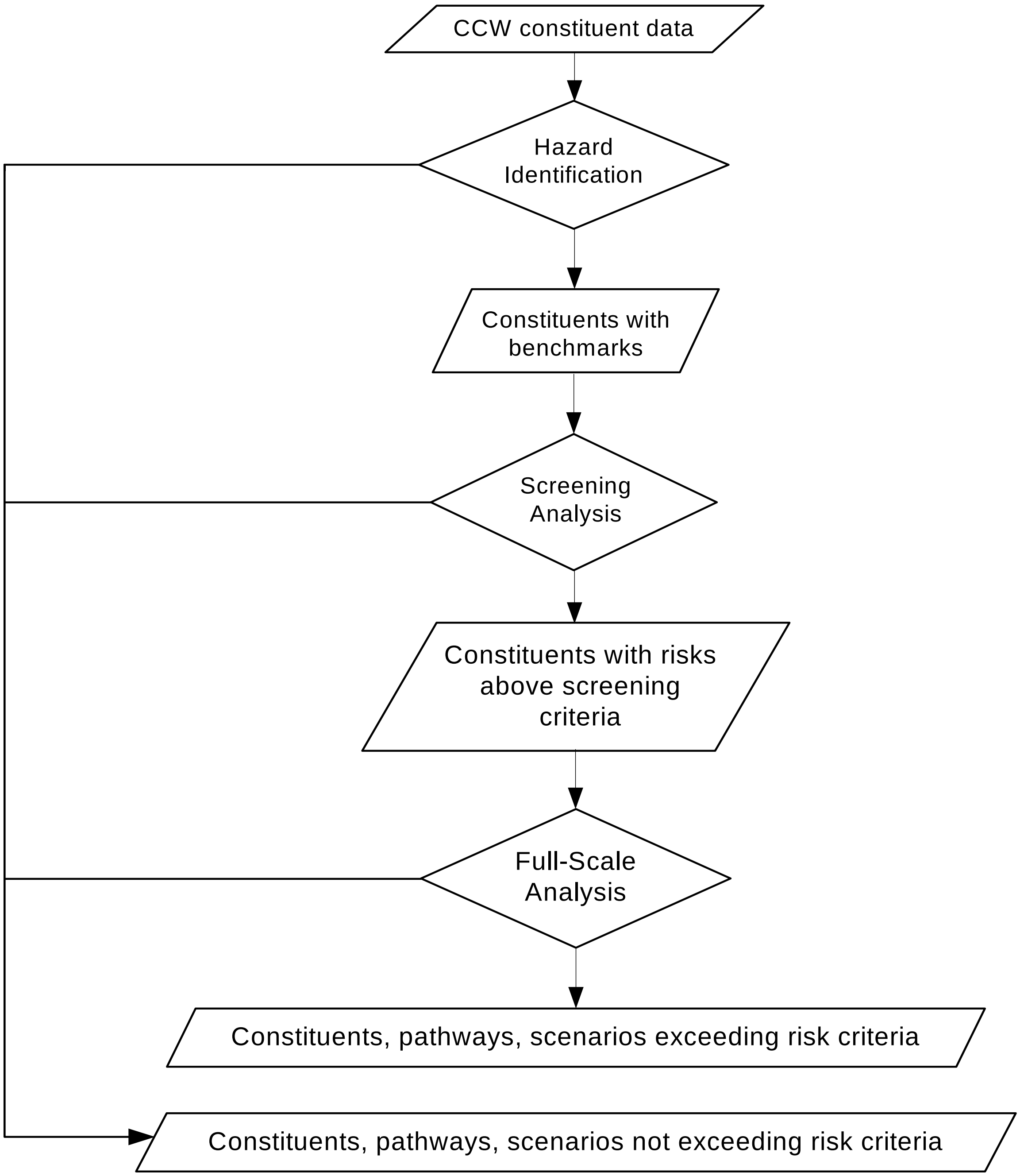


Figure ES-1. Overview of CCW risk assessment.

Table ES-1. Sources, Releases, Exposure Pathways, and Receptors Evaluated in the CCW Risk Assessment

Release Mechanism	Exposure Pathway	Exposure Mechanism	Receptor Type[a]	Screening Result
Landfills				
Leaching	Groundwater-to-drinking-water	Residential well	Resident	Full-scale analysis
	Groundwater-to-surface-water	Stream or lake, uptake by fish; contact with water, sediments	Recreational fisher; aquatic ecosystems	Full-scale analysis
Water erosion	Overland transport to surface water	Stream or lake, uptake by fish; contact with water, sediments	Recreational fisher; aquatic ecosystems	Below screening criteria
	Overland transport to soil	Soil ingestion; uptake from soil by plants, beef, dairy	Subsistence farmer; terrestrial ecosystems	Below screening criteria[b]
Wind erosion	Soil deposition	Soil ingestion; uptake from soil by plants, beef, dairy	Subsistence farmer; terrestrial ecosystems	Below screening criteria
	Fugitive dust	Inhalation	Resident	Below screening criteria
Surface impoundments				
Leaching	Groundwater-to-drinking water	Residential well	Resident	Full-scale analysis
	Groundwater-to-surface water	Stream or lake, uptake by fish; contact with water, sediments	Recreational fisher; ecological receptors	Full-scale analysis

[a] Human receptor types include adults and children.
[b] Except boron for plant toxicity. Also, damage cases indicate soil risks from selenium to terrestrial amphibians (Carlson and Adriano, 1993; Hopkins et al., 2006).

Table ES-2. Screening Analysis Results:
CCW Constituents Selected for Full-Scale Analysis[a]

Constituent	Human Health - Drinking Water		Human Health - Surface Water[b]		Ecological Risk - Surface Water	
	LF	SI	LF	SI	LF	SI
Arsenic	•	•	•	•	•	•
Boron	•	•			•	•
Cadmium	•	•	•	•	•	•
Lead	•	•			•	•
Selenium	•	•	•	•	•	•
Thallium	•	nd	•	nd	•	nd
Aluminum					•	•
Antimony	•	nd		nd		nd
Barium					•	•
Cobalt	na	•	na		na	•
Molybdenum	•	•				
Nitrate/Nitrite	•	•				
Chromium	•	•			•	•
Fluoride	•	•				
Manganese		•				
Vanadium	•	•			•	•
Beryllium					•	
Copper					•	•
Nickel		•				•
Silver					•	•
Zinc					•	

LF = landfill.
SI = surface impoundment
nd = nondetect—results are inconclusive because all analyses are nondetects.
na = not available—data were not available for cobalt in CCW landfill leachate.

[a] A mark in a cell indicates that the constituent was above the screening criteria for the indicated pathway and WMU type. Blank cells indicate that the constituent was below the screening criteria for a particular pathway/WMU combination. Risk screening was based on 90th percentile risk concentrations and no attenuation.

[b] Fish consumption pathway.

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). The risk assessment was also used to characterize waste management scenarios based on three liner types (unlined units, clay-lined units, and composite-lined units) and three waste types, as follows:

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

These three waste types and the two waste management options provide a good representation of CCW disposal practices and waste chemistry conditions that affect the release of CCW constituents from WMUs.[4],[5]

The full-scale analysis was implemented using a site-based 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 many CCW facilities across the United States, and a site-based approach 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 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 liner 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 (i.e., leaching) and transported to human and ecological receptors (i.e., via 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, we characterized the potential risks associated with the waste disposal scenarios and exposure pathways, including the uncertainties associated with the analysis results.

Results and Conclusions

Risks from clay-lined units are lower than those from unlined units, but 90th percentile risks are still well above the risk criteria for arsenic and thallium for landfills and arsenic, boron, and molybdenum for surface impoundments. Composite liners, as modeled in this assessment, effectively reduce risks from all constituents to below the risk criteria for both landfills and surface impoundments. Although it is likely that today, most new landfills have some type of liner (based on more recent data that were not incorporated into this assessment),

[3] Coal refuse is the waste coal produced from coal handling and preparation operations.

[4] Conventional CCW and codisposed CCW and coal refuse were modeled in landfills and surface impoundments and are the focus of the overall analysis. FBC wastes were treated separately because of limited data on FBC waste management units.

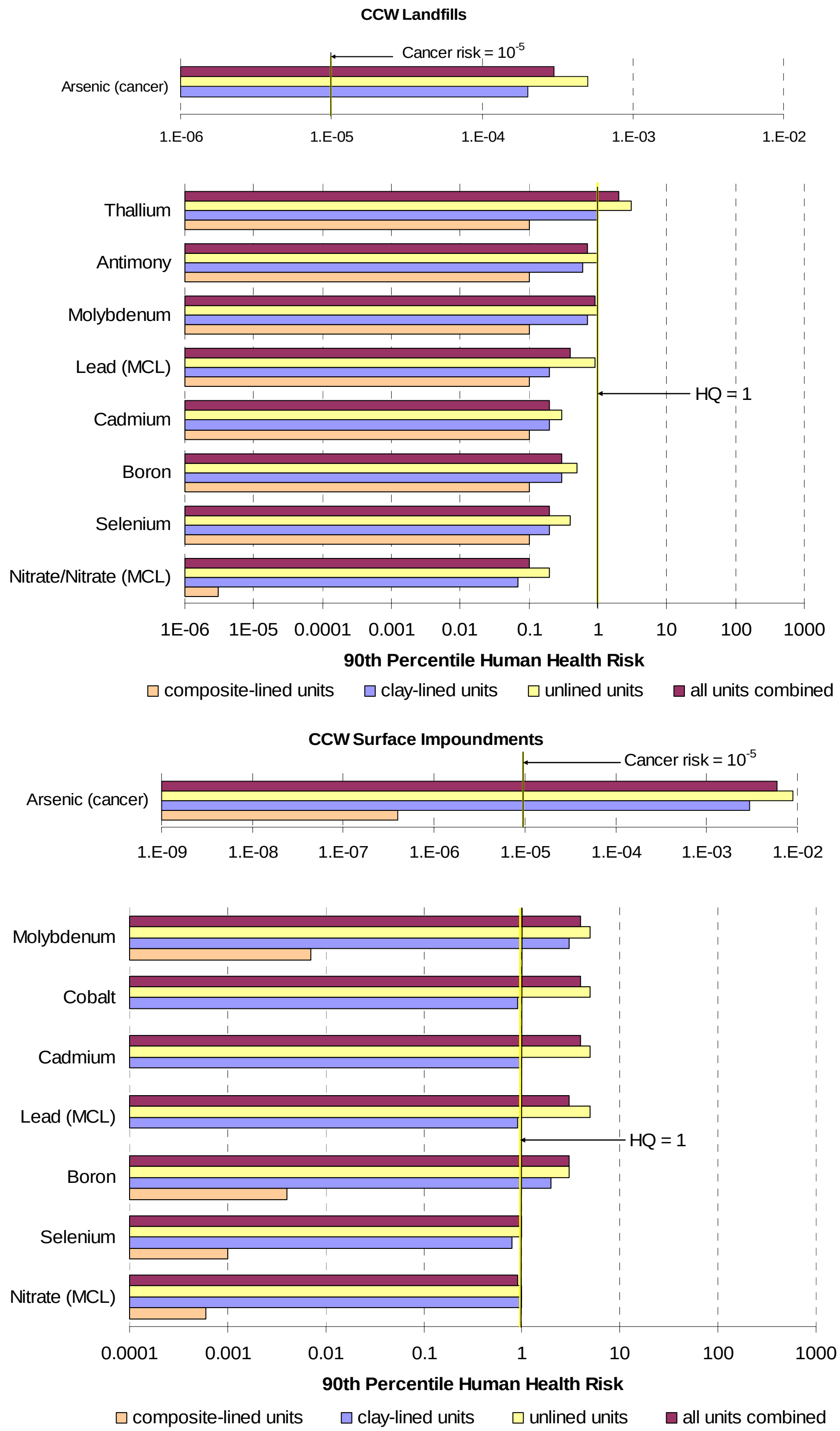
[5] Although different waste chemistries required the separate modeling of conventional CCW and CCW codisposed with coal refuse, the results were combined in this analysis to give an overall picture of the risks from CCW management,

it is not known how many unlined units continue to operate in the United States.

Recent data from a joint DOE/EPA survey suggests that more facilities are lined today than were in the 1995 data set on which this risk assessment is based. This suggests that the risks from CCW may be lower than the results presented in this report, although the older, unlined WMUs represented in this risk assessment may continue to pose potential risks to human health and the environment if they are closed with wastes in place.

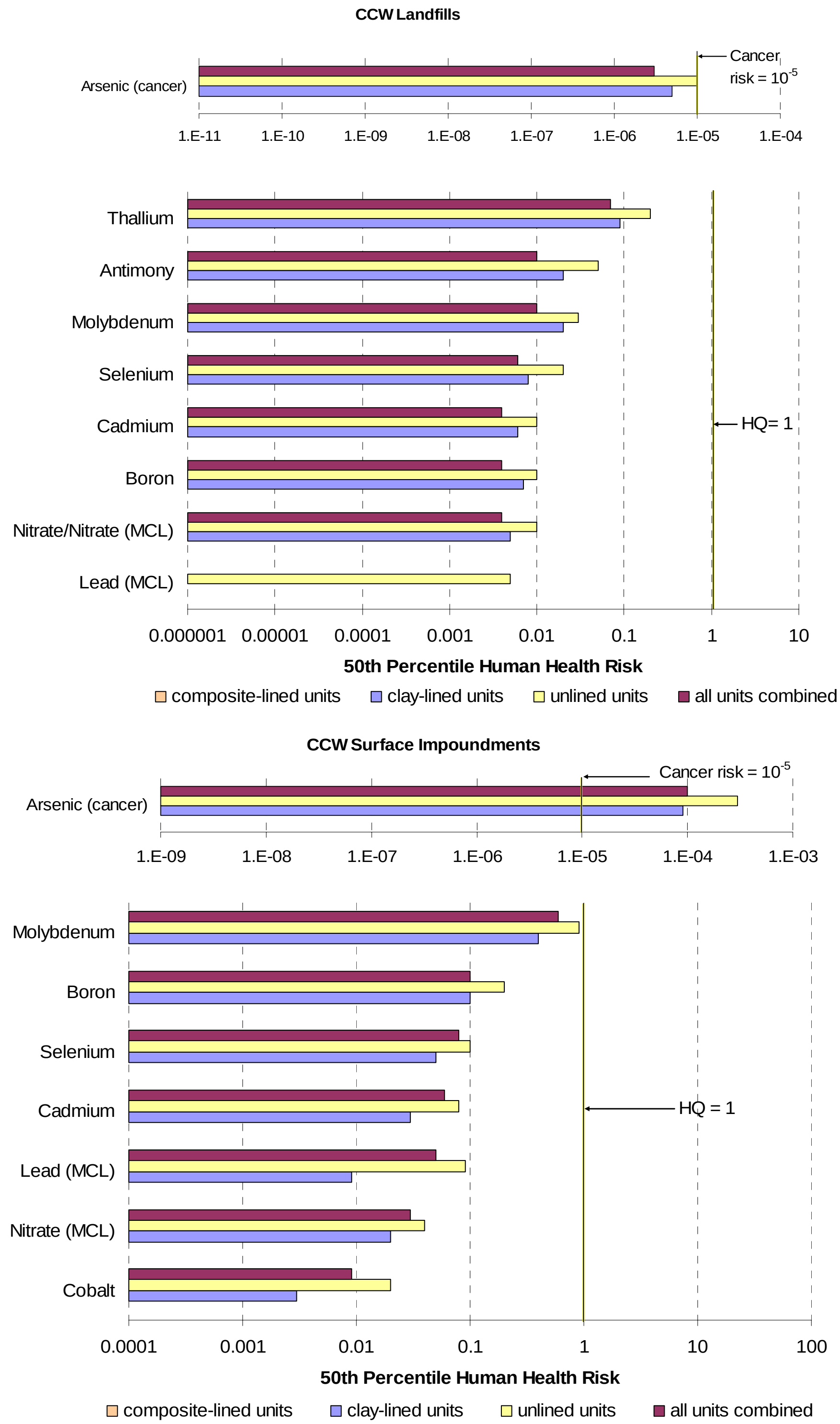
The CCW risk assessment results at the 90th percentile suggest that the management of CCW in landfills and surface impoundments as observed in 1995 for unlined and clay-lined units results in risks greater than the risk criteria of 10^{-5} for excess cancer risk to humans or an HQ greater than 1 for noncancer effects to both human and ecological receptors. Key risk findings include the following:

- 90th and 50th percentile risks for composite-lined units were consistently well below a cancer risk of 10^{-5} and an HQ of 1 for all constituents, waste management scenarios, and exposure pathways modeled in the CCW risk assessment.
- For humans exposed via the groundwater-to-drinking-water pathway (see Figures ES-2 and ES-3), arsenic and thallium show risks to human health above the risk criteria for unlined and clay-lined CCW landfills. Arsenic poses a 90th percentile cancer risk of 5×10^{-4} for unlined units and 2×10^{-4} for clay-lined units; thallium shows a 90th percentile HQ above 1 for unlined units only. As shown in Figure ES-3, 50th percentile results are at or below risk criteria for all constituents.
- Risks are higher for surface impoundments for the groundwater-to-drinking-water pathway, with a 90th-percentile arsenic cancer risk of 9×10^{-3} for unlined units and 3×10^{-3} for clay-lined units. For unlined units, 5 additional constituents have noncancer HQs ranging from 3 to 4 for the 90th percentile, including boron, lead, cadmium, cobalt, and molybdenum. Two constituents (boron and molybdenum) have 90th percentile HQs greater than 1 (2 and 3, respectively) for clay-lined surface impoundments. The 50th percentile results are approximately 10-fold greater than the 10^{-5} cancer risk level for arsenic in unlined (3×10^{-4}) and clay-lined (9×10^{-5}) surface impoundments.
- For humans exposed via the groundwater-to-surface-water (fish consumption) pathway, selenium (HQ = 2) and arsenic (cancer risk = 2×10^{-5}) show 90th percentile risks for unlined surface impoundments slightly above the risk criteria. All other waste management scenarios and all 50th percentile results show risks at or below the risk criteria for the fish consumption pathway.
- Waste type has little effect on landfill risk results, but surface impoundment risks are higher for codisposed CCW and coal refuse than for conventional CCW.
- Higher risks for surface impoundments than landfills are likely due to a combination of higher waste leachate concentrations, a higher proportion of unlined units, and a higher hydraulic head from 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, the 90th percentile risks for landfills exceed an HQ of 1 for boron and lead. For surface impoundments, risks for the 90th percentile for 6 constituents (boron, lead, arsenic, selenium, cobalt, and barium) exceed an HQ of 1, with boron showing the highest risks (HQ over 2,000). The exceedances for boron and selenium are consistent with reported ecological damage cases, which include impacts to waterbodies through the groundwater-to-surface-water pathway (e.g., Carlson and Adriano, 1993; U.S. EPA, 2007). Only boron exceeds the ecological risk criterion for surface water at the 50th percentile, with an HQ of 4.



A cancer risk of 10^{-5} or an HQ of 1 are the risk criteria for this analysis.
Results for "all units combined" are results across all liner types (unlined, clay-lined, composite-lined).
Note: When the composite liner bar does not appear on the chart, the 90th percentile risk index is below the minimum shown on the x-axis.

Figure ES-2. Full-scale 90th percentile risk results for the groundwater-to-drinking-water pathway.



A cancer risk of 10^{-5} or an HQ of 1 are the risk criteria for this analysis.
Results for "all units combined" are results across all liner types (unlined, clay-lined, composite-lined).
Note: When the composite liner bar does not appear on the chart, the 50th percentile risk index is below the minimum shown on the x-axis.

Figure ES-3. Full-scale 50th percentile risk results for the groundwater-to-drinking-water pathway.

- For ecological receptors exposed via sediment, 90th percentile risks for lead, arsenic, and cadmium exceeded a HQ of 1 for both landfills (HQs from 2 to 20) and surface impoundments (HQs from 20 to200) probably because these constituents strongly sorb to sediments. No constituents exceed the ecological risk criterion for sediments at the 50th percentile.

Sensitivity analysis results indicate that for more than 75 percent of the scenarios evaluated, the risk assessment model was most sensitive to parameters related to 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 flowrate 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 (adsorption coefficient, depth to groundwater, receptor well distance) are also important.

The multiple uncertainties associated with the CCW risk assessment include scenario uncertainty (i.e., uncertainty about the environmental setting around the plant), uncertainty in human exposure factors (such as exposure duration, body weight, and intake rates), uncertainty in human and ecological toxicity factors and potential cumulative risks, and uncertainty in estimates of fate and transport of waste constituents in the environment. Scenario uncertainty has been minimized by basing the risk assessment on conditions around existing coal-fired power plants around the United States, as observed in 1995. 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 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, as described below:

- **Appropriateness of CCW leachate data.** Although porewater data were available and used for CCW surface impoundments, available data for landfills were mainly Toxicity Characteristic Leaching Procedure (TCLP) analyses, which may not be representative of actual CCW leachate. Comparisons with recent (2006) 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 data, some of the higher concentrations in the 2006 data are not represented by the TCLP data. This suggests that selenium risks may be underestimated, which is consistent with selenium as a cause for CCW damage cases.
- **Limited CCW leachate data.** Because of a high proportion of nondetect values[6] and a limited number of measurements, mercury could not be addressed in the CCW risk assessment for landfills or surface impoundments, and antimony and thallium could not be assessed in surface impoundments. Mercury levels in leachate were measured in EPA's 2006 leaching study and suggest a limited concern for mercury for the CCW leachates investigated, but additional work is needed to extend these results to all CCW disposal facilities.
- **Clean Air Interstate Rule (CAIR) and Clean Air Mercury Rule (CAMR) impacts.** While CAIR and CAMR will reduce air 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 is conducting research on how much total mercury will increase in CCW from the use of mercury controls, as well as how the leachability of mercury and other metals will be impacted. Preliminary results suggest that the impacts on mercury leaching will depend on the mercury control process.
- **Arsenic speciation.** The current model does not speciate metals during subsurface

[6] Nondetect values are measurements where the concentration of a constituent is below the level that the analytical instrument can detect. The actual level could range from zero to just below the detection limit. Nondetects for constituents other than mercury were modeled at one-half the detection limit for this risk analysis.

transport. Damage cases and other studies suggest that arsenic readily converts from arsenic III in CCW leachate to the less mobile arsenic V in soil and groundwater. However, model runs conducted for both species suggest that the difference in risk between the two species is only about a factor of 2 at the 90th percentile risk level, which is not enough to bring arsenic risks below the risk criteria.

Uncertainties that EPA does not have enough data at this time to evaluate with respect to CCW risk results include the following:

- **Well distance.** Nearest well distances were taken from a survey of municipal solid waste (MSW) landfills because 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 performance.** Liner design and performance for CCW WMUs were based on data and assumptions EPA developed to be appropriate for municipal and nonhazardous industrial waste landfills. EPA believes that CCW landfills should have similar performance characteristics, but does not have quantitative data on CCW WMU liners to verify that.
- **Data gaps for ecological receptors.** Data were insufficient to develop screening levels and quantitative risk estimates for terrestrial amphibians, but EPA acknowledges that damage cases indicate risk to terrestrial amphibians through exposure to selenium (e.g., Hopkins et al., 2006).
- **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. These would be addressed through more site-specific studies on the proximity of these areas and species to CCW disposal units.
- **Synergistic and additive risk.** The impact of exposures to multiple contaminants on 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. The risk assessment also does not add risks across pathways (i.e., risks from drinking water and fish consumption), but EPA does not think that doing so would change the results markedly because the constituents of concern differ between pathways.

EPA recognizes that uncertainties in mercury levels in CCW leachate, both with and without the CAIR/CAMR mercury controls, represent a potentially significant gap in our knowledge of CCW risks.

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 to date can be found at http://www.epa.gov/epaoswer/other/fossil/index.htm.

Note that the full-scale risk assessment described in this report was mostly 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, 2006), 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 address the results and findings of these studies in future efforts to address CCW management practices.

The Agency is making the risk analysis document available in the Docket[1] to allow interested parties to submit comments on the analytical methodology, data, and assumptions used in the analysis and to submit additional information for the Agency to consider. In addition, the risk assessment will undergo independent scientific peer review by experts outside EPA following closure of the public comment period. Public comments will be made available to the peer reviewers for their consideration during the review process. The peer review will focus 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. As appropriate, EPA will update this analysis based on both public and peer-review comments.

[1] Available at http:www.regulations.gov; docket number EPA-HQ-RCRA-2006-0796.

1.2 Purpose and Scope of the Risk Assessment

The purpose of this risk assessment is 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 can use to develop management options for CCW management.

The scope of this risk assessment is 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 environmental media surrounding coal-fired utility power plants, and the risks that these concentrations pose to human and ecological receptors.

The size, design, and locations of the onsite CCW landfills and surface impoundments modeled in this risk assessment are 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.

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)

The releases, and hence media concentrations and risk estimates, are based on leaching to groundwater, wind and water erosion, and overland transport. This analysis does 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.

To evaluate the significance of the estimated risks, the risk criteria adopted for this assessment are

- An estimate of the excess lifetime cancer risk for individuals exposed to carcinogenic (cancer-causing) contaminants of 1 chance in 100,000 (10^{-5} excess cancer risk)[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.

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. 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 move on to the next step, constituent screening.
- **Constituent Screening**, which compared very conservative estimates of exposure concentrations (e.g., whole waste concentrations, leaching concentrations) to health-based concentration benchmarks to quickly and simply identify constituents and exposure pathways 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 Monte Carlo risk analysis.

This document focuses on the full-scale Monte Carlo analysis. Constituent screening results are also presented as part of the problem formulation discussion, along with a summary of the screening methodology.[3]

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[2] The typical cancer risk range used by the Office of Solid Waste and Emergency Response is 10^{-4} to 10^{-6}. In hazardous waste listings, the point of departure for listing a waste is 10^{-5}.

[3] Details on the CCW constituent screening analysis can be found in U.S. EPA (2002a).

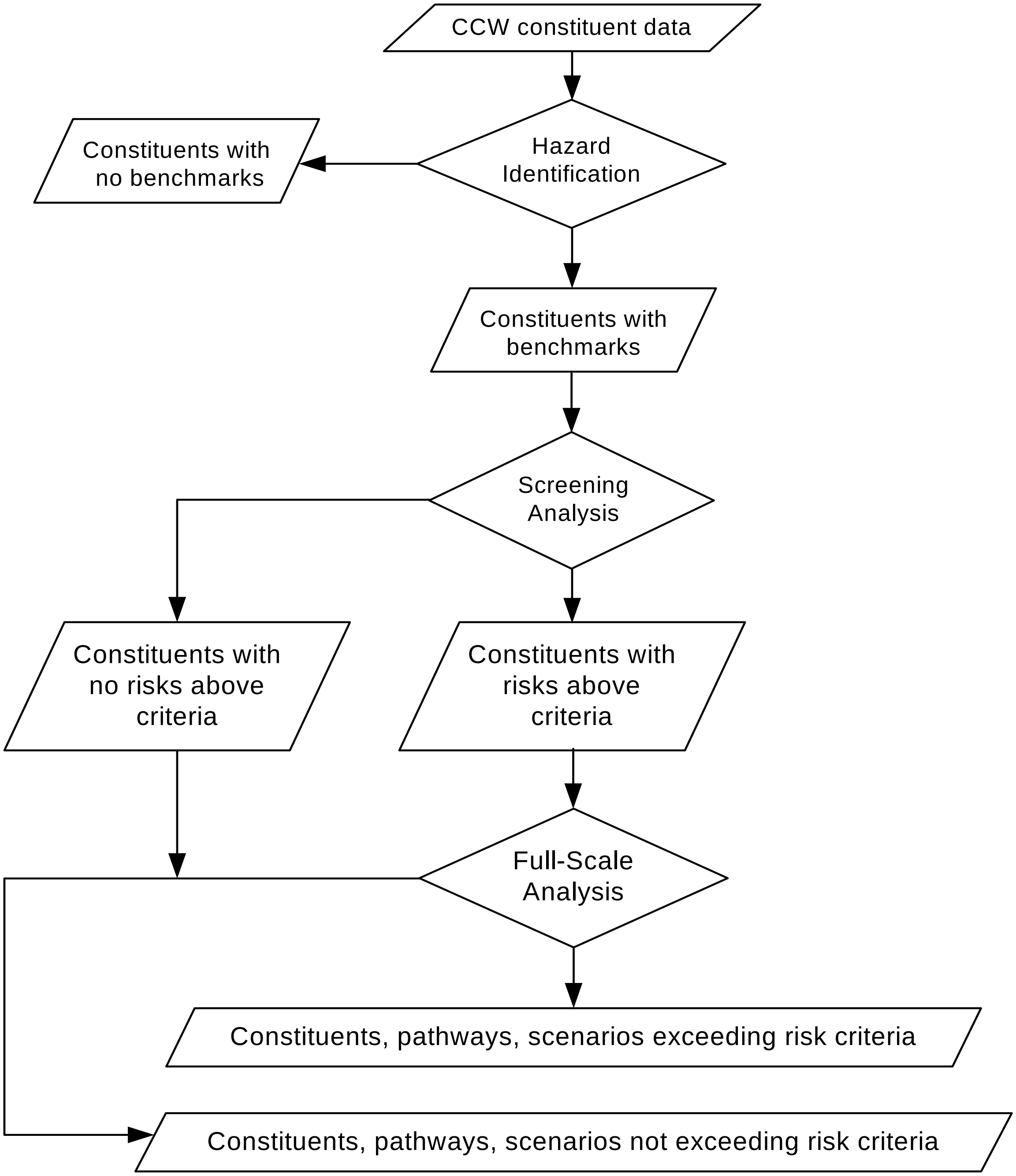

Figure 1-1. Overview of coal combustion waste risk assessment.

1.3.1 Waste Management Scenarios

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,**[4] 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.

1.3.2 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. This approach is ideal for this risk assessment because there are many CCW facilities across the United States, and a site-based approach 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

[4] 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.”

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

Analysis

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

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

Risk Characterization

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

7. Identify the waste disposal scenarios and environmental conditions that pose risks to human health or the environment that are above the risk criteria. 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) estimate constituent releases from CCW landfills and surface impoundments (source models), (2) model constituent concentrations in the environmental media of concern (groundwater and surface water), (3) calculate exposure, and (4) 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 that result in risks above the risk criteria. Results are presented as national estimates for CCW landfills and CCW surface impoundments, as well as by waste type and liner status. For risk exceedances, this section identifies the CCW constituents and pathways that exceed the risk criteria, along with any factors (such as liners or facility environmental setting) that might 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 the wastes disposed in the WMUs, including the CCW leachate concentration distributions used. Appendix B describes how EPA characterized the WMUs (landfills and surface impoundments), including surface area, capacities, geometry, and liner status. Appendix C presents the methodologies and data used to characterize the environmental setting at each CCW site, including delineating the site layout and determining the environmental setting (e.g., meteorology, climate, soils, aquifers, and waterbodies).

The remaining 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), and benchmarks for human health (Appendix G) and ecological risk (Appendix H).

2.0 Problem Formulation

The full-scale CCW risk assessment is 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 and screening 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 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 (Section 2.3), including data collection, fate and transport modeling to estimate exposure point concentrations, exposure assessment, and the calculation of 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 are based on the best data on industry operations and waste management practices that were available at that time. These data sources include 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). 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 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. With the addition of the supplemental data, the 2003 CCW constituent database covers approximately 140 waste disposal sites.[1] The 2003 database also has broader coverage of the major ion concentrations of CCW leachate (e.g., calcium, sulfate, pH).

2.1.1.1 Waste Types

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 (ash and FGD sludge) (moderate to high pH); codisposed CCW and coal refuse (low pH); and FBC waste (high pH). As shown in Table 2-1, each of these waste types includes several waste streams that are usually codisposed in landfills or surface impoundments.

Along with the type of WMU (landfill or surface impoundment), the three waste types in Table 2-1 define 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).

[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 Streams	Number of Sites by Waste Type[a] Landfill Leachate	Surface Impoundment Porewater	Total Waste[b]
Conventional CCW	*97*	*13*	*62*
Ash (not otherwise specified)	43	0	30
Fly ash	61	2	33
Bottom ash and slag	24	3	23
Combined fly and bottom ash	7	4	4
FGD sludge	4	6	5
Codisposed Ash & Coal Refuse	9	5	1
FBC 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] Number of sites by waste type from leachate, porewater, and whole waste data tables in the 2003 CCW constituent database.
[b] Whole waste concentration data.

2.1.1.2 Constituents and Pathways

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 U.S. EPA, 2002a, for a full list of sources). Table 2-2 shows the results of that search for each constituent. Benchmarks were found for 26 chemicals in the constituent database. Constituents without human health or ecological benchmarks were not addressed further in the risk analysis.[2]

To further narrow down the list of constituents, a screening analysis (U.S. EPA, 2002a) was conducted that compared very conservative estimates of exposure concentrations (e.g., whole waste concentrations, leaching concentrations) to health-based concentration benchmarks to quickly, simply, and safely identify constituents and exposure pathways 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. The technical background document for the CCW screening analysis (U.S. EPA, 2002a) provides further detail on the

[2] The CCW constituents without benchmarks are limited to common elements, ions, and compounds (e.g., iron, magnesium, phosphate, silicon, sulfate, sulfide, calcium, pH, potassium, sodium, carbon, sulfur) that 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]	Constituent	CAS ID	HHB[a]	EcoB[b]
Metals				***Inorganic Anions***			
Aluminum	7429-90-5	✔	✔	Chloride	16887-00-6		
Antimony	7440-36-0	✔	✔	Cyanide	57-12-5	✔	
Arsenic	7440-38-2	✔[c]	✔	Fluoride	16984-48-8	✔	
Barium	7440-39-3	✔	✔	Nitrate	14797-55-8	✔	
Beryllium	7440-41-7	✔[d]	✔	Nitrite	14797-65-0	✔	
Boron	7440-42-8	✔	✔	Phosphate	14265-44-2		
Cadmium	7440-43-9	✔[d]	✔	Silicon	7631-86-9		
Chromium	7440-47-3	✔[c]	✔	Sulfate	14808-79-8		
Cobalt	7440-48-4	✔	✔	Sulfide	18496-25-8		
Copper	7440-50-8	✔[e]	✔	***Inorganic Cations***			
Iron	7439-89-6			Ammonia	7664-41-7	✔	
Lead	7439-92-1	✔[e]	✔	Calcium	7440-70-2		
Magnesium	7439-95-4			pH	12408-02-5		
Manganese	7439-96-5	✔		Potassium	7440-09-7		
Mercury	7439-97-6	✔	✔	Sodium	7440-23-5		
Molybdenum	7439-98-7	✔	✔	***Nonmetallic Elements***			
Nickel	7440-02-0	✔	✔	Carbon	7440-44-0		
Selenium	7782-49-2	✔	✔	Sulfur	7704-34-9		
Silver	7440-22-4	✔	✔	***Measurements***			
Strontium	7440-24-6	✔		Total Dissolved Solids	none		
Thallium	7440-28-0	✔	✔	Total Organic Carbon	none		
Vanadium	7440-62-2	✔	✔	Dissolved Organic Carbon	none		
Zinc	7440-66-6	✔	✔				

[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 we used the more protective cancer benchmark for human health throughout this assessment.
[d] Probable carcinogen
[e] Safe Drinking Water Act Action Level only

screening analysis. As detailed there, the risks for all above-ground pathways analyzed (soil ingestion, inhalation, gardening, beef and dairy, and erosion and overland transport) for human receptors did not exceed the screening criteria for any constituent, so they were not considered any further in the risk assessment. The above-ground pathway risks for ecological receptors also did not exceed the screening criteria except for boron and selenium. Boron, which showed risks above the risk criterion in above-ground pathways due to plant toxicity in the CCW screening analysis, has been shown to be toxic to plants (Carlson and Adriano, 1993). Selenium has shown toxicity to terrestrial amphibians via above-ground pathways (Carlson and Adriano, 1993; Hopkins et al., 2006). Because the risks posed by these CCW constituents to ecological communities via above-ground pathways is well documented in damage cases and field studies (see above references and U.S. EPA, 2007), we did not believe that a full-scale above-ground pathway analysis was necessary to confirm this conclusion for two constituents. Thus, the full-scale risk assessment did not include any above-ground pathways, only groundwater pathways.

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 2-3 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.[3] Note that mercury was not addressed in the screening or full-scale analysis because of a very high proportion of nondetects in the CCW constituent database. Similarly, a high number of nondetects (or a very low number of measurements) prevented screening or full-scale analysis for antimony, thallium, and cobalt in surface impoundments. The uncertainties associated with these limited analytical results are discussed in Section 4.4.3.1.

Resources did not allow full-scale modeling to be conducted for all 21 constituents that had 90th percentile risks above the screening criteria. To reduce the number of constituents to be modeled, those 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 2-3 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. Nine constituents did not exceed 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.

2.1.2 Waste Management Scenarios

The full-scale CCW risk assessment models 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.

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[3] 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 2-3. 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 we used the more protective cancer benchmark for human health 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).

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.[4]

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[5] and CCW codisposed with coal refuse.

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 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 waste 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 2-4 shows how the plants are 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). EPA does not believe that the number and locations of onsite CCW landfills and surface impoundments has changed significantly since 1995, although liners are more prevalent in the newer facilities (see further discussion in Section 4.4.1). The DOE/EPA report (U.S. DOE, 2006) supports this conclusion.

[4] 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.

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

Table 2-4. 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

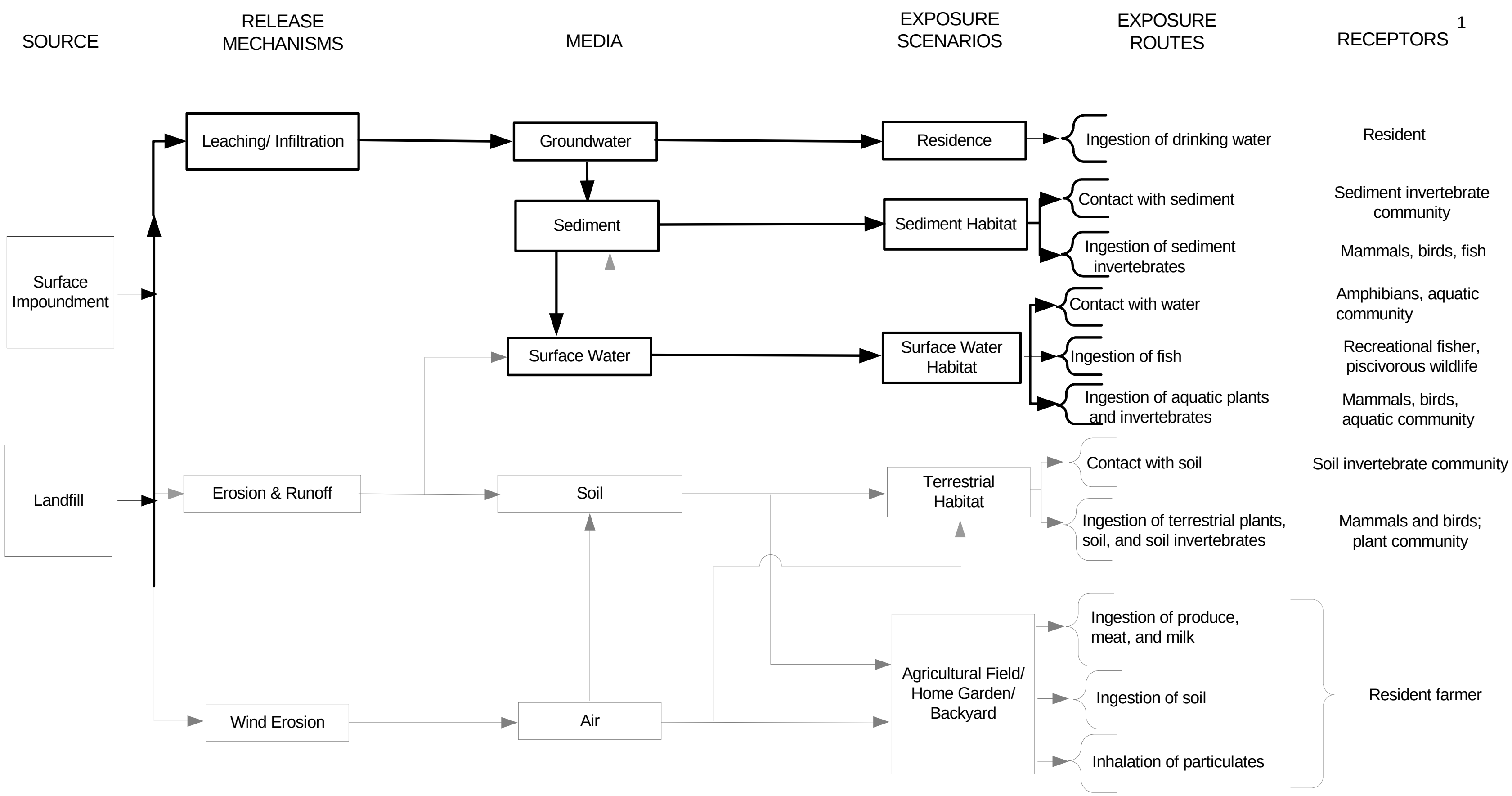
The waste stream/WMU combinations discussed above provide the waste management scenarios to be evaluated in the risk assessment. The full-scale assessment used the EPRI survey data to place these scenarios at actual onsite CCW disposal sites across the country. These sites were used as the basis for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

2.2.1 Conceptual Site Model

Figure 2-1 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)
- Above-ground pathways, including soil ingestion, inhalation, and consumption of produce, beef, and milk.



Note:

1. Resident, recreational fisher, and resident farmer include adult and four child age groups

Pathways addressed in full scale risk modeling

Pathways that screened out of bounding risk assessment

Figure 2-1. Conceptual site model of CCW risk assessment.

Ecological Risk

- Groundwater to surface water
- Above-ground soil
- Above-ground contamination of surface water and sediment.

As described in Section 2.1.1, the CCW screening analysis addressed all of these exposure pathways and receptors. Through that screening analysis, risks for all above-ground pathways (shown in gray instead of black in Figure 2-1) fell below the screening criteria and were not considered further in the full-scale risk assessment.[6] This enabled EPA to focus full-scale modeling on groundwater-to-drinking-water and groundwater-to-surface-water exposure pathways (shown in black in Figure 2-1). This groundwater pathway analysis evaluates exposures through drinking water ingestion and surface water contamination from groundwater discharge. For the groundwater-to-surface-water pathway, the analysis assumes 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 we 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, we 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 to model the land use scenarios of most concern for CCW disposal facilities:

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

Figures 2-2 and 2-3 show these two conceptual site layouts, 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 is 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 is assumed to be located at the property line of the facility to which it belongs.

[6] Although the risks from the aboveground screening analysis did exceed the risk criteria for boron and selenium in soil, to streamline the assessment, these compounds were not included in the full ecological assessment.

Adjacent to the WMU is a buffer area within which there is 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.

Residential Groundwater-to-Drinking-Water Scenario

The residential groundwater-to-drinking-water scenario calculates exposure through residential use of well water as drinking water. In the Monte Carlo analysis, the receptor well is randomly placed up to 1 mile downgradient from the edge of the WMU (this radial well distance is labeled R_{rw} in Figure 2-2), 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 is 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 in Section 4.4.3.3).

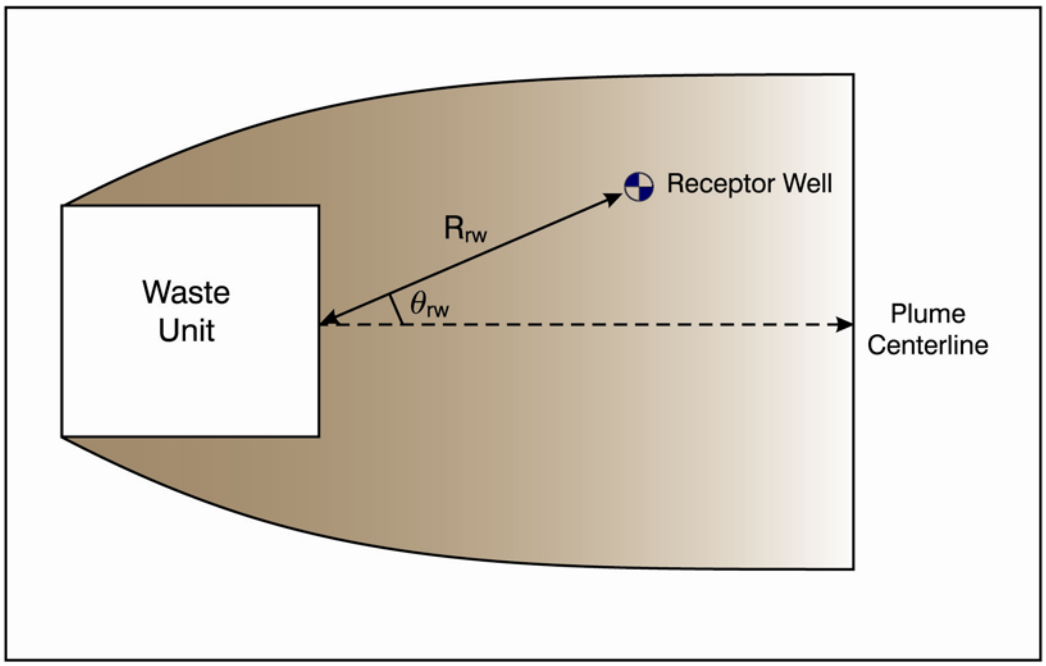


Figure 2-2. Conceptual site layout for residential groundwater ingestion scenario.

The angle off the contaminant plume centerline (θ_{rw} in Figure 2-2) 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 3.4.3. 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 collected using a site-based approach, as described in Appendix C.

Recreational Fisher and Ecological Risk Scenario

The recreational fisher[7] scenario 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. Note that the fisher's residence is not the same residence where the residential well is located, and therefore risks are not added across the drinking water and fish consumption pathways.

The waterbody was assumed to be a stream or lake located downgradient from the WMU, beginning where the buffer area ends (see Figure 2-3), and was also used as the most impacted aquatic system for the ecological risk assessment. Waterbody characteristics were determined based on site-specific, regional, or national data (as described in Appendix C), except for stream length, which was determined by the width of the plume as it intersects the waterbody.

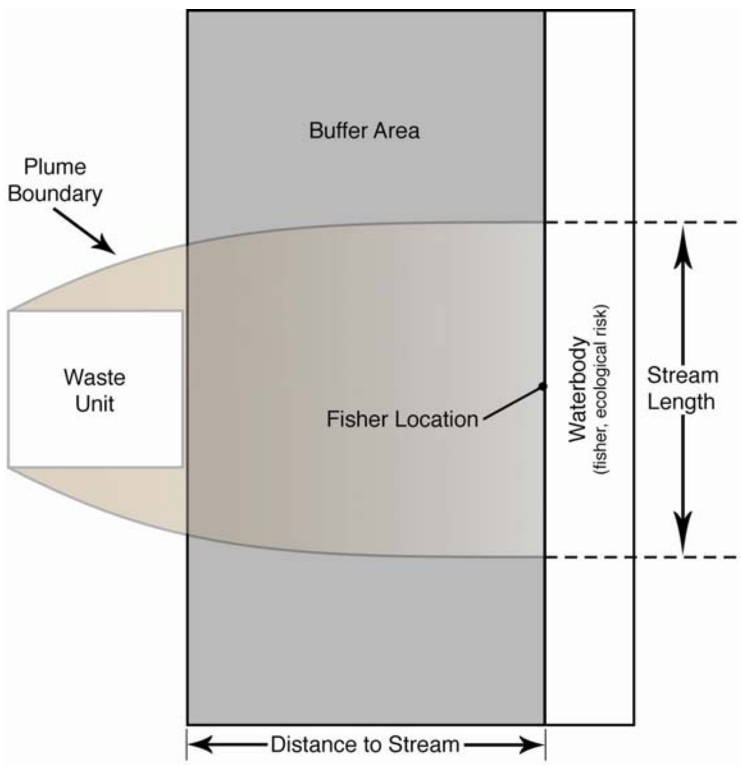


Figure 2-3. Conceptual site layout for residential fisher and aquatic ecological risk scenario.

[7] Only recreational fishers were considered because they represent the reasonable maximum exposed individuals and because the streams, lakes, or rivers that are near CCW plants are not likely to be used by commercial fishing operations.

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.

2.3 Analysis Scope and Design

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. To gain a better understanding of the risks that may be posed by these constituents, 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 site-based approach used data about waste management practices and environmental conditions at 181 utility CCW disposal sites across the United States.[8] 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. As described in Appendices B and C, site-specific data for the following model inputs were collected for these sites and used in the risk assessment:

- WMU dimensions
- WMU liner status (unlined, clay liner, composite liner)
- Waste type (conventional CCW, CCW codisposed with coal refuse, and FBC wastes)
- Geology (aquifer type)
- Soil texture
- Climate (precipitation, infiltration)
- Surface water type and flow conditions.

One question related to this risk assessment is how CCW facilities may have changed in the decade since the 1995 EPRI survey. Although the DOE/EPA survey does not include all of

[8] 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.

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 one can assess how liner conditions changed as CCW facilities were built or expanded over the past decade. 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 between 61 to 63 percent of the total population of the newly commissioned WMUs.[9] 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 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 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.

Because site-specific data were not readily available for depth to groundwater or receptor location (i.e., distances to nearest drinking water well and surface waterbody), national distributions for those inputs taken from a national hydrogeologic database (Newell et al., 1989 and 1990) developed to support EPA's national groundwater risk assessments were used in the Monte Carlo analysis to characterize the national variability of receptor distances (see Appendix C). This enabled EPA to assess the importance of those variables for the national risk distribution for individuals with reasonable maximum exposure to CCW.

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.

2.3.1 Data Collection

For the sites representing each WMU and waste type combination selected for analysis, the Monte Carlo analysis begins with input files that contain, for each Monte Carlo realization, the following site-based variables:

- WMU area, depth, and capacity
- WMU liner status

[9] 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.

- 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 flows.

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).

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.

2.3.2 Model Implementation

As a first step in the modeling process, the groundwater model (EPACMTP) reads 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 is run until contaminant concentrations at the receptor point return to zero after the concentration peak or for the maximum simulation time of 10,000 years, whichever comes first.

Groundwater model results are 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 include

- 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 calculates risk from drinking water ingestion and fish consumption for each realization. For ecological risk, the model uses surface water and sediment concentrations along with ecological benchmarks to estimate the risks to ecological receptors.

2.3.3 Exposure Assessment

Table 2-5 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 2-5. 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 estimates 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 conservatively 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 are incorporated into the development of ecological benchmarks (see Appendix H), which are surface water and sediment concentrations corresponding to an HQ of 1.

The time period for the exposure assessment is 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, we consider 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 use the highest (peak) annual average concentration to estimate ecological exposure and risk.

2.3.4 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 3.8; in addition, uncertainty related to these endpoints is discussed in Section 4.4.3.4.

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 the risk criteria of 1 in 100,000 excess cancers and an HQ greater than 1 for noncarcinogens. An HQ greater than 1 for was also used for the ecological risk criterion in the full-scale risk assessment.

3.0 Analysis

The full-scale analysis evaluates 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 various mechanisms. Releases to the atmosphere and by erosion and overland transport did not pose risks above the screening criteria in the screening analysis; therefore, these were not assessed in the full-scale analysis. Instead, the full-scale analysis focused on groundwater pathways, which exceeded the risk criteria for some constituents in the screening analysis. 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). These are the groundwater pathways evaluated in the full-scale CCW risk assessment.

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 include

- **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 groundwater and surface water
- **Exposure** models that estimate daily contaminant doses for humans and ecological receptors exposed to CCW constituents in groundwater and surface water
- **Risk** models that calculate risks to humans and ecological receptors.

This section describes the models and equations used to calculate exposure point concentrations and risk. Section 3.1 provides the overall structure for the analysis, including the spatial and temporal framework and the probabilistic (Monte Carlo) framework for the model runs. Sections 3.2 and 3.3 describe the landfill and surface impoundment source models used to predict environmental releases of constituents from CCW. Sections 3.4 and 3.5 describe the fate and transport modeling used to predict contaminant concentrations in groundwater and surface

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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 is 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.

water. Section 3.6 describes the human exposure calculations. Section 3.7 describes the health benchmarks used to develop human and ecological risk estimates, and Section 3.8 describes how these risks were calculated for human and ecological receptors.

Supporting detail can be found in the following appendices:

- **Appendix A, Constituent Data**, provides the CCW constituent concentrations used and describes how they were collected and processed

- **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

- **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

- **Appendix G, Human Health Benchmarks**, describes how the human toxicity benchmarks were selected and developed

- **Appendix H, Ecological Benchmarks**, describes how the ecological toxicity benchmarks were selected and developed.

3.1 General Modeling Approach

This section describes the framework, general assumptions, and constraints for the full-scale probabilistic analysis. Section 3.1.1 describes the temporal and spatial framework. Section 3.1.2 describes the probabilistic framework, and Section 3.1.3 describes how the assessment was implemented within the probabilistic framework.

3.1.1 Temporal and Spatial 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 are 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) is 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)[4] 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 produces 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.

[4] Distributions of exposure duration and other exposure variables were obtained from the *Exposure Factors Handbook* (U.S. EPA, 1997c,d,e)

The exposure duration for sensitive ecological receptors is 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.

3.1.2 Probabilistic Approach

The full-scale analysis evaluates risk in a probabilistic manner and is based on a Monte Carlo simulation that produces a distribution of exposures and risks. The general Monte Carlo approach is shown in Figure 3-1. The foundation of the Monte Carlo simulation is 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.

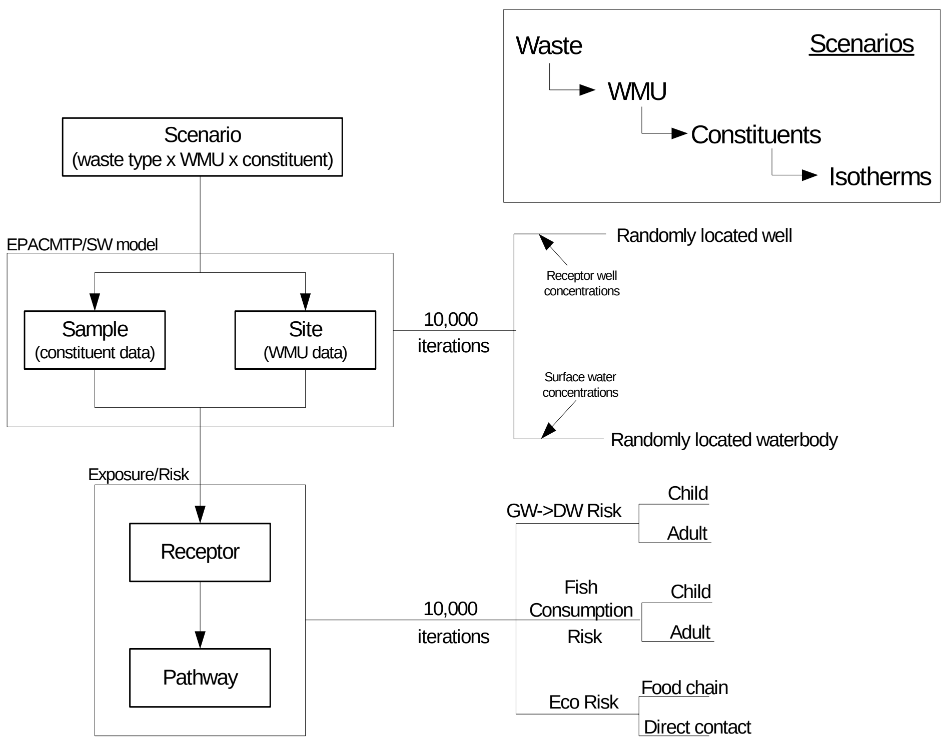


Figure 3-1. Overview of the Monte Carlo approach.

The detailed looping structure for the Monte Carlo analysis is shown in Figure 3-2. For each waste type/WMU combination, two separate loops are run. The first loop (shown with dashed lines in Figure 3-2) prepares a set of input files containing 10,000 sets of WMU and site data (as described in Section 3.1.3). The second loop (shown with solid lines in Figure 3-2) uses those input files to run 10,000 iterations of the source, fate and transport, exposure, and risk models for each constituent.

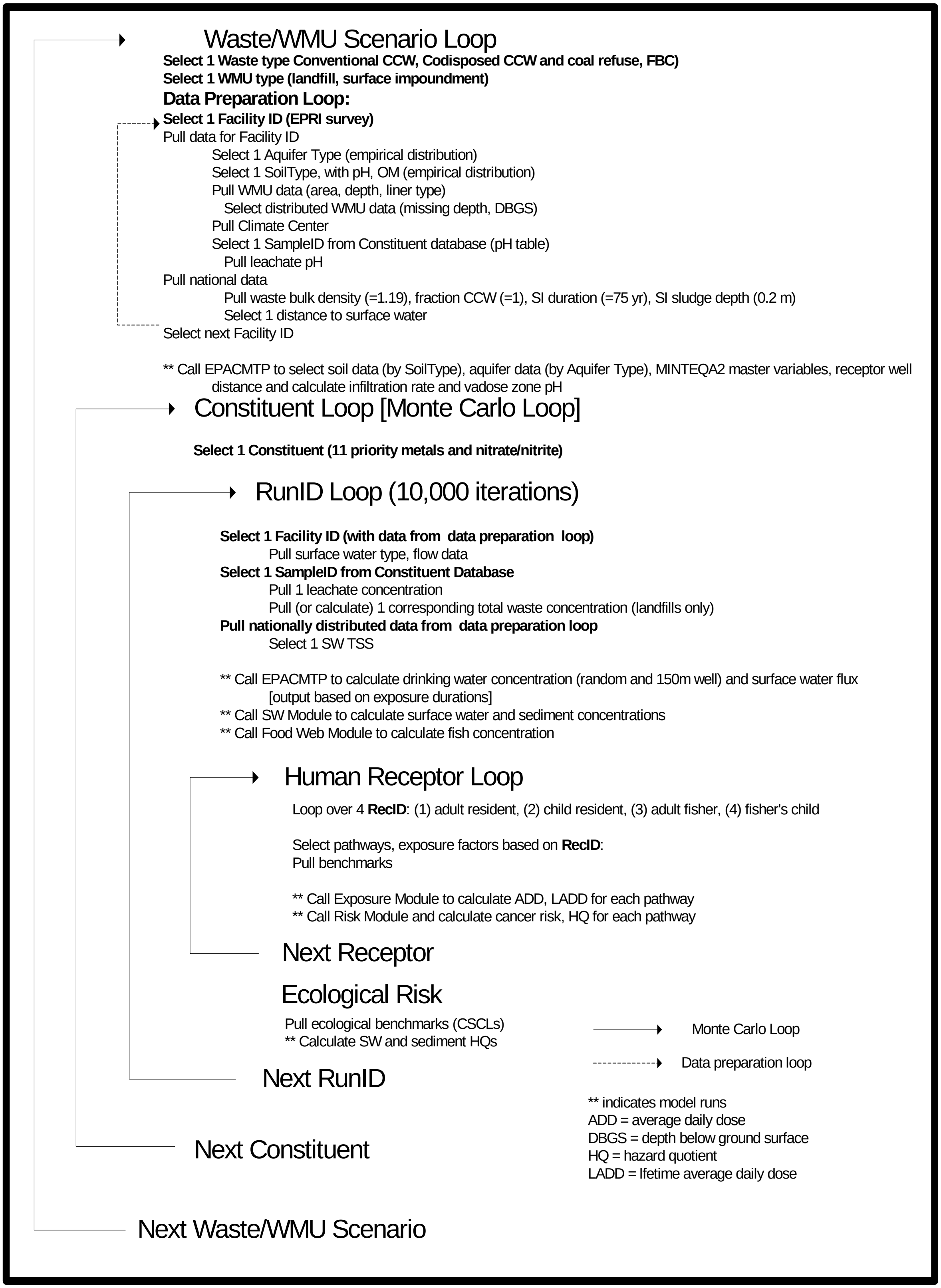


Figure 3-2. Monte Carlo looping structure.

3.1.3 Implementation of Probabilistic Approach

Table 3-1 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 3-1. 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, bottom ash, 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 the landfills and surface impoundments from the EPRI survey data 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):

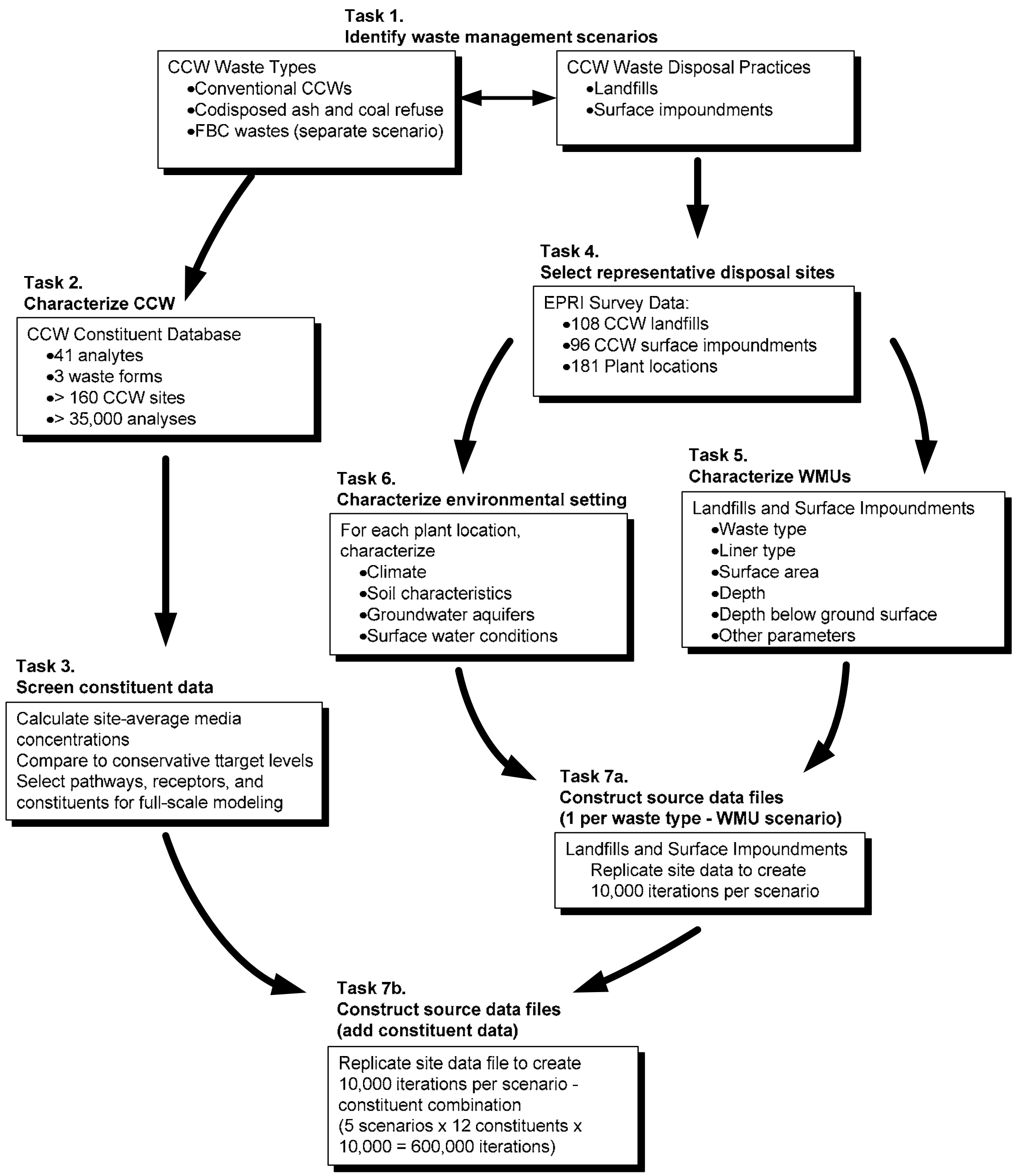


Figure 3-3. Process used to construct the Monte Carlo input database.

- Climatic data, including annual precipitation, temperature, and windspeed, were collected by assigning each site to a nearby meteorologic station.
- Soil and aquifer type were collected within a 5-km radius of each site to account for locational uncertainty for the WMUs.
- 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.

3.2 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.

3.2.1 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.

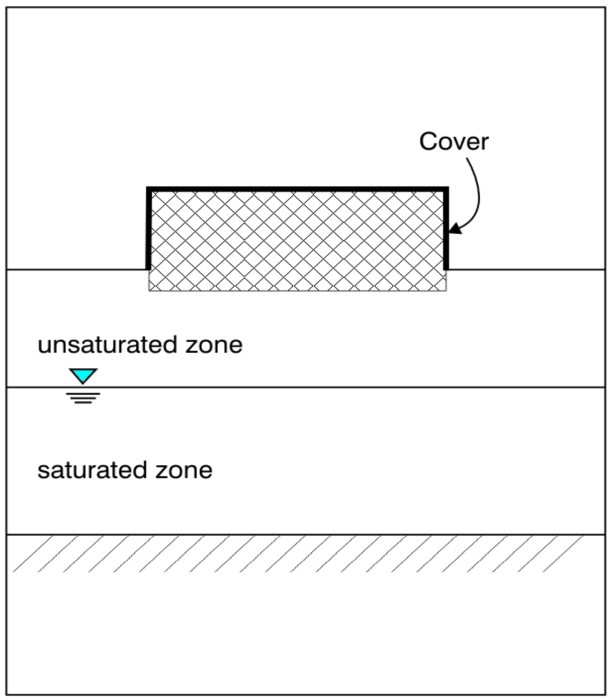


Figure 3-4. Conceptualization of a landfill in the landfill source-term model.

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 3.2.3 (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).

3.2.2 Modeling Approach and Assumptions

The starting point for the landfill source-term model simulation is the time when the landfill is closed (i.e., when the unit is filled with CCW).[5] 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 continue after closure until the contaminant is depleted from the waste mass in the landfill. This is a conservative assumption, as some metal will not leach from the waste mass.

Infiltration and Leaching

The average rate at which water percolates through the 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 used to determine infiltration rates for each CCW liner scenario are 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*

[5] 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. 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.

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, 1994b). 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 3.2.1, with the only variables that change 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 is based on a typical engineered compacted clay liner that is 3 feet thick with a design hydraulic conductivity of 1×10^{-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 are 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 3.2.1 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 3-2, these LDS flow rates include 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.

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).

Table 3-2. 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).

Source Depletion and Mass Balance

For this assessment**,** the landfill source-term model represents 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 is 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 are 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

$$TSOURC = \frac{CWASTE \times DEPTH \times FRACT \times CTDENS}{CZERO \times SINFIL} \tag{3-1}$$

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.

3.2.3 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 3-3 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 significant uncertainty in these liner assumptions is how representative the EPRI survey data are of current conditions at coal combustion facilities.

Table 3-3. 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

3.2.4 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 is used as an input for the EPACMTP unsaturated zone model.

3.3 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 is 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.

3.3.1 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.

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. Typically, the leachate concentration is equal to the concentration in the wastewater entering the impoundment.

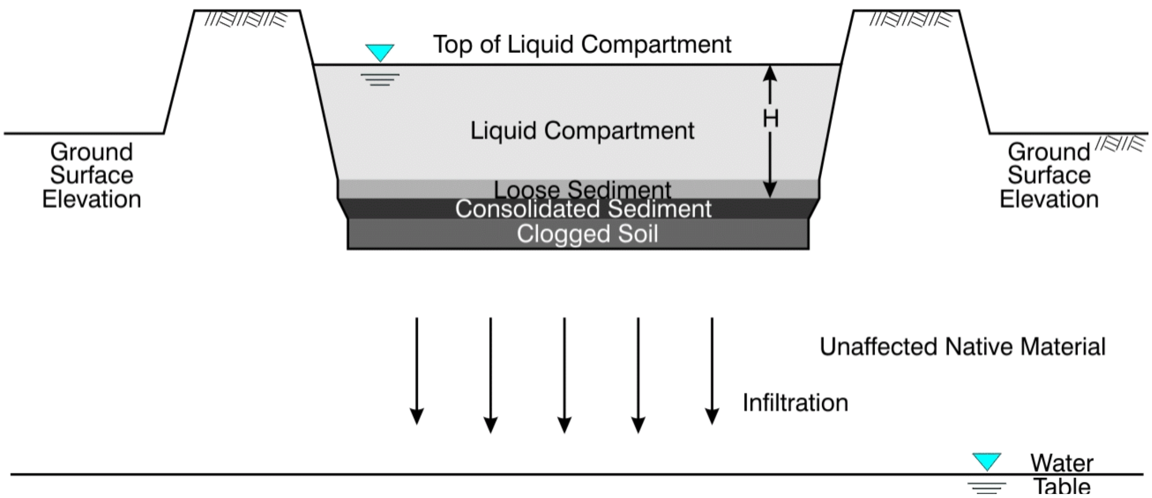


Figure 3-5. Schematic cross-section view of surface impoundment.

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 1×10^{-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 1×10^{-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 (CQA) 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; we assume 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.

3.3.2 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.

3.3.3 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 3.3.2.

- **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, we 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), we used the same crosswalk as we used for landfills (see Table 3-2). 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).

3.3.4 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 is used as an input for the EPACMTP unsaturated zone model.

3.4 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 3.5.

3.4.1 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 three 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 is used for the residential drinking water pathway

- A nearby river, stream, or lake, which is modeled as a fully penetrating surface waterbody and is 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.

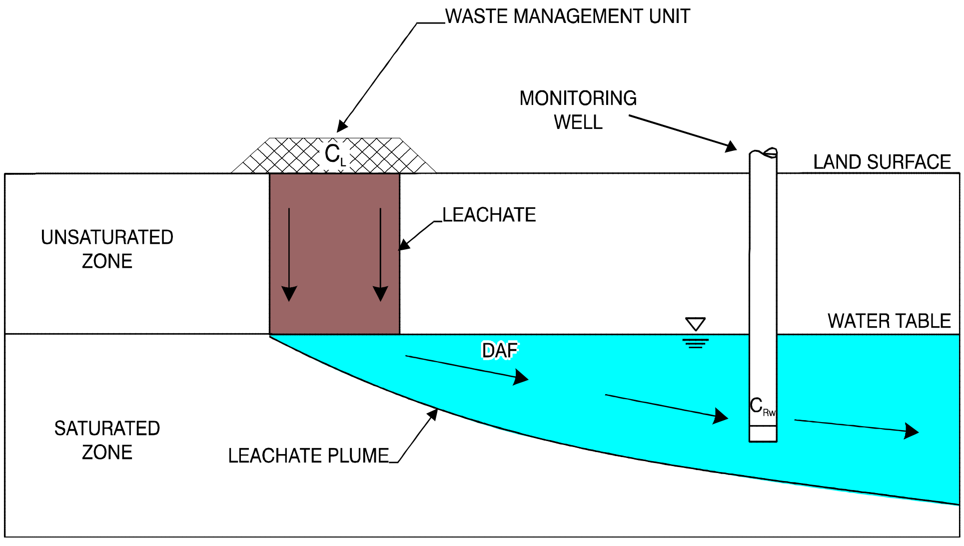


Figure 3-6. Conceptual model of the groundwater modeling scenario.

3.4.2 Modeling Approach and Assumptions

The transport of leachate from the WMU through the unsaturated and saturated zones was modeled using EPACMTP (U.S. EPA, 1996, 1997a, 2003a, 2003d, 2003d). 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.

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.

3.4.3 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, 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.

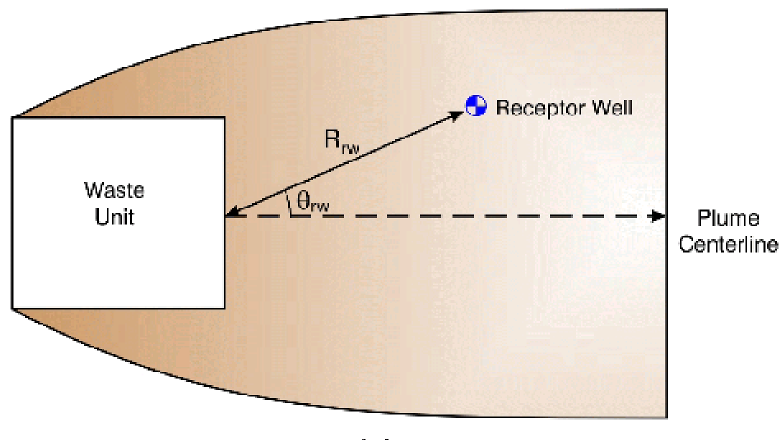


Figure 3-7. Schematic plan view showing contaminant plume and receptor well location.

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, but because information on the actual distance of drinking water wells from CCW facilities is very limited, EPA is seeking comments and additional data that are relevant to this issue.

- The angle off the contaminant plume centerline (θ_{rw} in Figure 3-7) was based on a uniform distribution ranging from zero to ninety degrees.

- Wells were placed within the lateral extent of the contaminant plume (shaded portion in Figure 3-7).

- 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 its depth was varied uniformly throughout the aquifer thickness or throughout the upper 10 m of the aquifer thickness, whichever was less.

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.

3.4.4 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.

3.5 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. EPACMTP assumes a fully penetrating stream; therefore, the entire groundwater and contaminant flux is passed to the surface water model. To ensure that an unrealistic flux of contaminated groundwater does not occur, the groundwater flow into the waterbody is compared to the stream flow. If the groundwater flux exceeds the stream flow, it is capped at the stream flow and the contaminant flux is reduced using the ratio of the stream flow to the incoming groundwater flow (i.e., 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 are 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 is 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 3.5.1), and aquatic organisms (see Section 3.5.2). Special modeling provisions for aluminum are described in Section 3.5.3.

3.5.1 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 3.5.3). 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 3-4 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 3-4. 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 3.5.2. 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.

3.5.2 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 assumes that fish are a food source for a recreational fisher. Trophic level three (TL3) and four (TL4) fish[6] 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/ epaoswer/hazwaste/id/hwirwste/peer03/aquatic/aqtfooda.pdf).

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 bioconcentration factors (BCFs), which are presented in Table 3-5. The equations used to model fish tissue concentrations are provided in Attachment E-2 to Appendix E.

Table 3-5. Bioconcentration Factors for Fish

CAS	Chemical	T3 Value	T4 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.

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

3.5.3 Aluminum Precipitation Model

A simple precipitation model was used for aluminum in lieu of the equilibrium-partitioning model, because aluminum is generally solubility limited in natural waters. 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 3-6 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 3-6. 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.

3.6 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 3.6.1 presents an overview of the receptors and selected exposure pathways considered for this assessment, including a discussion of how childhood exposure is considered in the analysis. Section 3.6.2 presents exposure factors (i.e., values needed to calculate human exposure) used in the analysis. Section 3.6.3 describes the methods used to estimate dose, including average daily dose (ADD) and lifetime average daily dose (LADD).

3.6.1 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. Although conservative, EPA considers this assumption to be reasonable and protective for fishers relying on locally caught fish as a food source.

Table 3-7 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 3-7. Receptors and Exposure Pathways

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

Childhood Exposure

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 is aged through the age cohorts as appropriate until the age corresponding to the selected exposure duration is 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).

3.6.2 Exposure Factors

The exposure factors used are listed in Table 3-8, 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 3-8. 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.

- **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 EFH. Because the EFH does not have fish ingestion rates for children, adult ingestion rates were used (as a conservative assumption).

- **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.

- **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 is based on the conservative 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.

3.6.3 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.

3.7 Toxicity Assessment

A chemical's ability to cause an adverse human health effect 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 then 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 as part of the screening assessment. The same benchmarks were used in the full-scale risk assessment, with a few updates. The data sources and collection methodology for these benchmarks are described briefly in Sections 3.7.1 (human health benchmarks) and 3.7.2 (ecological benchmarks), and in more detail in Appendix G (human health benchmarks) and Appendix H (ecological benchmarks). The discussion here is limited to the 12 constituents assessed in the full-scale risk assessment and for humans, covers only oral benchmarks (because all inhalation pathway risks fell below the screening criteria in the screening assessment). Appendices G and H cover all constituents and routes.

3.7.1 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.

- 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 increases. 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. 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 are described in more detail in Appendix G.

The chronic human health benchmarks used in the full-scale analysis are summarized in Table 3-9. For most constituents, human health benchmarks were available from IRIS. Benchmarks for a few constituents were obtained from ATSDR and Superfund Provisional

Benchmarks U.S. EPA (2001c,d). For chemicals for which purely health-based benchmarks were not available (lead), a drinking water action level was used (U.S. EPA, 2002d).

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.

Table 3-9. Human Health Benchmarks Used in the Full-Scale Analysis

Constituent	Type of Benchmark	Value	Units	Source[a]
Cancer Benchmark				
Arsenic	CSF	1.5E+00	$(mg/kg\text{-}d)^{-1}$	IRIS
Noncancer Benchmarks				
Aluminum	RfD	2.0E+00	mg/kg-d	ATSDR
Antimony	RfD	4.0E-04	mg/kg-d	IRIS
Barium	RfD	2.0E-01	mg/kg-d	IRIS
Boron	RfD	2.0E-01	mg/kg-d	IRIS
Cadmium	RfD (water)[b]	5.0E-04	mg/kg-d	IRIS
	RfD (food)[c]	1.0E-03	mg/kg-d	IRIS
Cobalt	RfD	2.0E-02	mg/kg-d	Superfund
Lead	MCL	0.015	mg/L	DWAL
Molybdenum	RfD	5.0E-03	mg/kg-d	IRIS
Nitrate/Nitrite	MCL[d]	10	mg/L	DWAL
Selenium	RfD	5.0E-03	mg/kg-d	IRIS
Thallium	RfD	8.0E-05	mg/kg-d	IRIS

[a] References:
ATSDR: Minimal Risk Levels, ATSDR (2002)
DWAL: Drinking Water Action Level, U.S. EPA (2002d)
Superfund: Superfund Risk Issue Paper, U.S. EPA (2001c,d)
IRIS: U.S. EPA (2002c)
HEAST: U.S. EPA (1997f)
[b] Used for drinking water ingestion.
[c] Used for fish ingestion.
[d] For nitrate.

3.7.2 Ecological Benchmarks

The ecological risk assessment addresses 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 3-10 shows the receptor types assessed for each exposure route (direct contact and ingestion) in each environmental medium addressed by the CCW risk assessment.

Table 3-10. 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. 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.

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 a significant 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 does not evaluate the effects on population dynamics in the sense that a reduction in the population is predicted over time in response to exposure to constituents released from CCW. Population-level modeling was well 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 3-11. Additional details on the CCW ecological benchmarks and CSCLs and their development can be found in Appendix H.

Table 3-11. Ecological Risk Criteria Used in the Full-Scale Analysis

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
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 (1998)
[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.

Ecological benchmarks for both the screening and full-scale CCW risk assessment were taken directly from the 1998 fossil fuel combustion risk assessment, *Non-Groundwater*

Pathways, Human Health and Ecological Risk Analysis for Fossil Fuel Combustion Phase 2 (FFC2) (U.S. EPA, 1998a). 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.

3.8 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 uses estimates of dose and toxicity to calculate individual excess lifetime carcinogenic risk estimates and noncancer HQs (Section 3.8.1). 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 uses estimates of exposure (media) concentrations and toxicity (media-specific concentration limits) to calculate an ecological HQ (Section 3.8.2).

3.8.1 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 3-12. 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 3-12. 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 3.6.3) 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.

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

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 is characterized through the use of HQs, which are generated by dividing an ADD (see Section 3.6.3) for ingestion pathways by the corresponding RfD.[7] 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 effect level (NOAEL), from a low observed adverse effect level (LOAEL), or from a 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).

$$HQ_i = \frac{ADD_i}{RfD} \qquad (3\text{-}5)$$

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.[8] 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.

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

[7] 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.

[8] Stream distance and well distance were sampled independently in the Monte Carlo analysis.

3.8.2 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 3.7.2, these ecological benchmarks were developed for representative organisms and communities in each environmental medium of concern.

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:

$$HQ_{surface\ water} = C_{sw} / CSCL_{sw} \tag{3-6}$$

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

$$HQ_{sediment} = C_{sediment} / CSCL_{sediment} \tag{3-7}$$

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 by receptor, pathway, and WMU type.

An overview of the assessment on which these results are 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.

Probabilistic results are 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), location (e.g., meteorological region), receptor (e.g., child), and health endpoint (e.g., cancer), the 90th percentile risk represents the high-end estimate that is compared to the appropriate risk criteria (for cancer or noncancer) to help determine whether CCW disposal practices are protective of public health. The risk criteria used are defined in terms of estimated lifetime cancer risk and noncancer hazard attributable to CCW disposal. The risk criteria adopted for this assessment are

- For chemical constituents that cause cancer (carcinogens), the criterion is an estimated excess lifetime cancer risk for exposed individuals of 1 case in 100,000 (i.e., 1×10^{-5})
- For constituents that cause adverse, noncancer health effects (noncarcinogens), the criterion is a 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.

In general, the full-scale analysis showed lower risks than the screening analysis, but still showed risks above risk criteria for certain CCW constituents, WMU types, pathways, and receptors at the 90th percentile. At the 50th percentile, risks are still above the risk criteria 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 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). The human health risk criteria for the analysis were a 10^{-5} excess cancer risk for arsenic and an HQ greater than 1 for the other constituents, each of which exhibits noncarcinogenic effects.

4.1.1 Groundwater-to-Drinking-Water Pathway

Tables 4-1 and 4-2 present the 90th and 50th percentile risk results, respectively, for the groundwater-to-drinking water pathway for landfills and surface impoundments. Results are shown across all units combined (i.e., across all liner types), as well as for each of the three unit types modeled in the analysis (unlined, clay-lined, and composite-lined). Except for arsenic, the results presented are for a child resident, because those risks for noncarcinogens were consistently higher than the risks for the adult resident. For arsenic, a carcinogen, adult risks are presented because the longer exposure duration and higher intake rates cause risks to be slightly higher for adults than for children. Results for arsenic and selenium are based on the arsenic III and selenium VI species, which are more mobile in soil and groundwater (causing higher receptor well concentrations). Results for other arsenic and selenium species for comparison can be found in the model uncertainty discussion in Section 4.4.2.

Figures 4-1 and 4-2 show the 90th and 50th percentile risk results. For each constituent, the graphs plot the 90th percentile (Figure 4-1) or 50th percentile (Figure 4-2) HQ or cancer risk level against the risk criteria (10^{-5} cancer risk or an HQ greater than 1) by the liner types reported in the EPRI survey. As in the table, the constituents are shown in order from highest risk in the full-scale analysis to lowest; the risk criteria are shown by the solid vertical line. Composite liners are not plotted in these figures when risks are below the x-axis minimum.

Note that not all 12 chemicals modeled in the full-scale assessment are presented for each pathway/WMU scenario. Only the chemicals for which the risks in the screening assessment exceeded the screening criteria for the scenario and 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 because of a high proportion of nondetects in the surface impoundment data for these CCW constituents. Similarly, adequate cobalt data were available only for surface impoundments. Screening-level human health risks for barium were below the screening criteria; therefore, barium is shown only in the ecological risk tables and figures. The screening analysis results in Section 2.1 and Table 2-3 show which CCW constituents were modeled for each pathway/WMU scenario.

Table 4-1. Summary of 90th Percentile Full-Scale CCW Human Risk Results: Groundwater-to-Drinking-Water Pathway

Chemical[b]	90th Percentile HQ or Cancer Risk Value[a]			
	All Units Combined[c]	Unlined Units	Clay–Lined Units	Composite–Lined Units
Landfills				
Arsenic (cancer)	3E-04	5E-04	2E-04	0
Thallium	2	3	1	0
Antimony	0.7	1	0.6	0
Molybdenum	0.9	1	0.7	0
Lead (MCL)[d]	0.4	0.9	0.2	0
Cadmium	0.2	0.3	0.2	0
Boron	0.3	0.5	0.3	0
Selenium	0.2	0.4	0.2	0
Nitrate/Nitrite (MCL)[d]	0.1	0.2	0.07	3E-06
Surface Impoundments				
Arsenic (cancer)	6E-03	9E-03	3E-03	4E-07
Molybdenum	4	5	3	7E-03
Cobalt	4	5	0.9	0
Cadmium	4	5	1	2E-09
Lead (MCL)[d]	3	5	0.9	1E-20
Boron	3	3	2	4E-03
Selenium	1	1	0.8	1E-03
Nitrate/Nitrite (MCL)[d]	0.9	1	1	6E-04

[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 not every chemical that was selected for full-scale modeling was modeled in every pathway/WMU scenario: only chemicals with adequate data and that were identified in the screening analysis as needing further assessment (see Section 2.1) were modeled for each scenario.

[c] Results across all unit types combined (unlined, clay-lined, and composite-lined).

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

Table 4-2. Summary of 50th Percentile Full-Scale CCW Human Risk Results: Groundwater-to-Drinking-Water Pathway

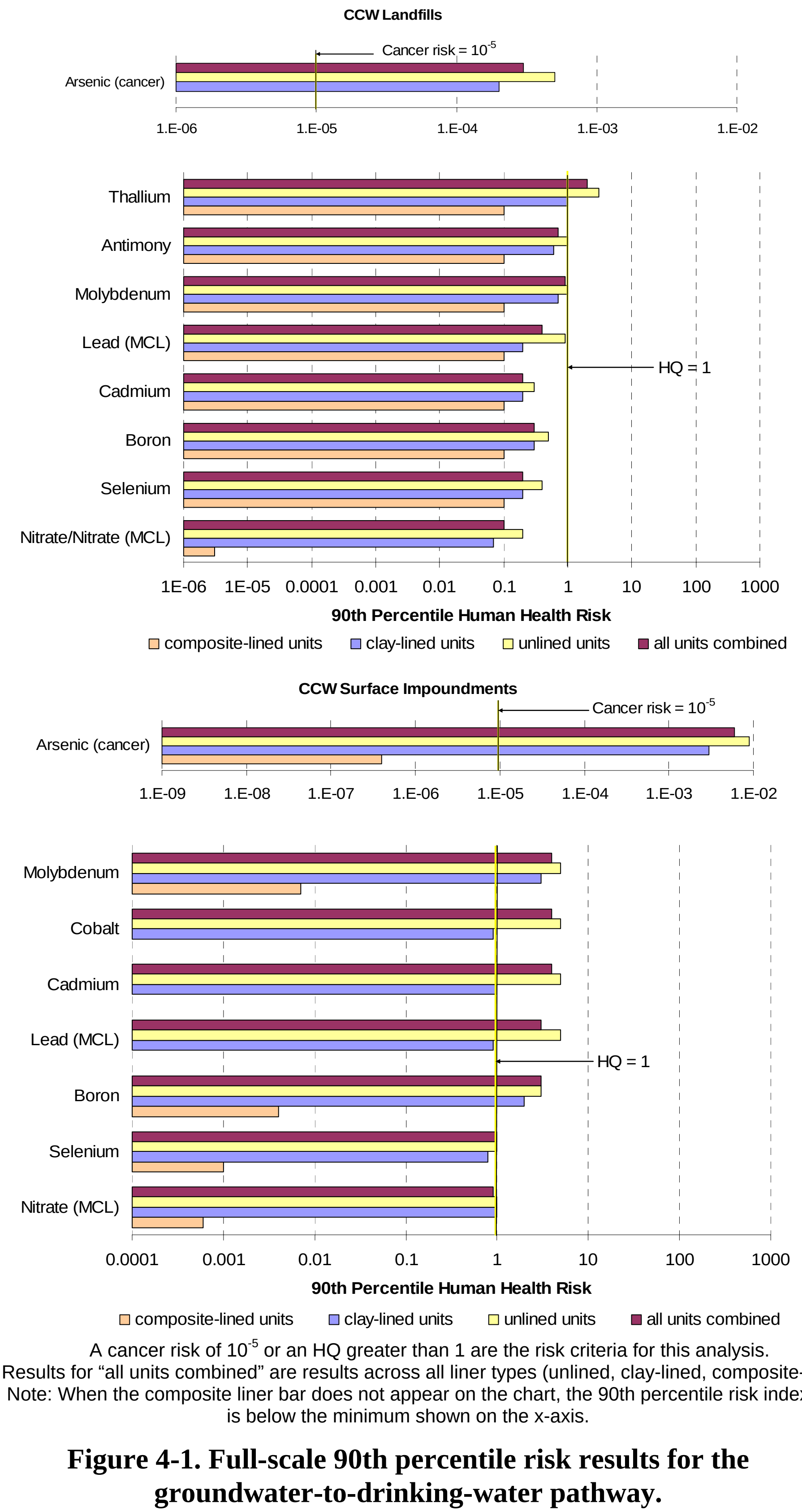
Chemical[b]	50th Percentile HQ or Cancer Risk Value[a]			
	All Units Combined[c]	Unlined Units	Clay–Lined Units	Composite–Lined Units
Landfills				
Arsenic (cancer)	3E-06	1E-05	5E-06	0
Thallium	0.07	0.2	0.09	0
Antimony	0.01	0.05	0.02	0
Molybdenum	0.01	0.03	0.02	0
Lead (MCL)[d]	2E-07	5E-03	6E-08	0
Cadmium	4E-03	0.01	6E-03	0
Boron	4E-03	0.01	7E-03	0
Selenium	6E-03	0.02	8E-03	0
Nitrate/Nitrite (MCL)[d]	4E-03	0.01	5E-03	0
Surface Impoundments				
Arsenic (cancer)	1E-04	3E-04	9E-05	0
Molybdenum	0.6	0.9	0.4	5E-12
Cobalt	9E-03	0.02	3E-03	0
Cadmium	0.06	0.08	0.03	0
Lead (MCL)[d]	0.05	0.09	9E-03	0
Boron	0.1	0.2	0.1	6E-12
Selenium	0.08	0.1	0.05	5E-12
Nitrate/Nitrite (MCL)[d]	0.03	0.04	0.02	7E-08

[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 not every chemical that was selected for full-scale modeling was modeled in every pathway/WMU scenario: only chemicals with adequate data and that were identified in the screening analysis as needing further assessment (see Section 2.1) were modeled for each scenario.

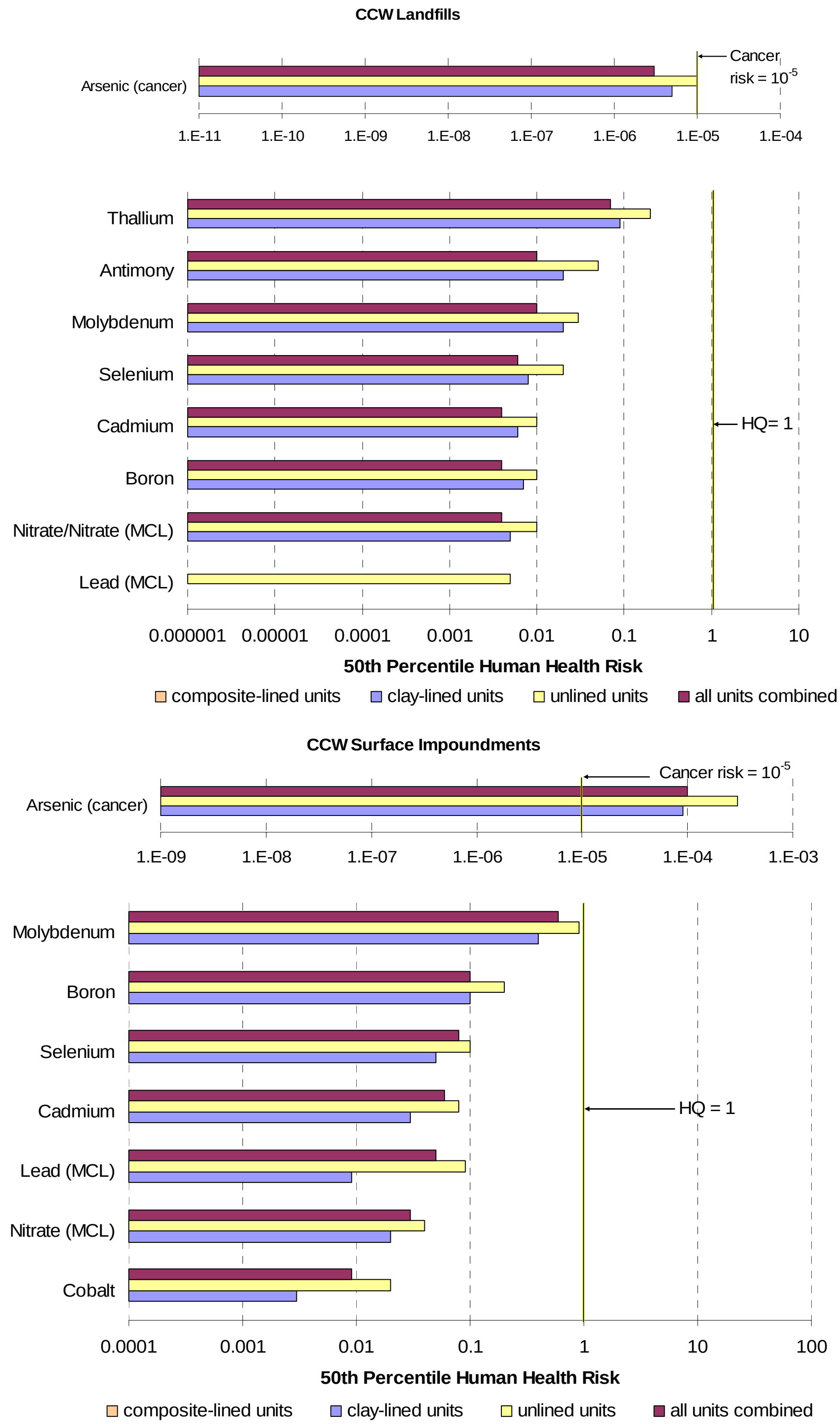
[c] Results across all unit types combined (unlined, clay-lined, and composite-lined).

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



A cancer risk of 10^{-5} or an HQ greater than 1 are the risk criteria for this analysis.
Results for "all units combined" are results across all liner types (unlined, clay-lined, composite-lined).
Note: When the composite liner bar does not appear on the chart, the 90th percentile risk index is below the minimum shown on the x-axis.

Figure 4-1. Full-scale 90th percentile risk results for the groundwater-to-drinking-water pathway.



A cancer risk of 10^{-5} or an HQ greater than 1 are the risk criteria for this analysis.
Results for “all units combined” are results across all liner types (unlined, clay-lined, composite-lined).
Note: When the composite liner bar does not appear on the chart, the 50th percentile risk index is below the minimum shown on the x-axis.

Figure 4-2. Full-scale 50th percentile risk results for the groundwater-to-drinking-water pathway.

As can be seen in Figure 4-1, the full-scale analysis produced lower risks for landfills than surface impoundments. For landfills, at the 90th percentile, arsenic shows risks above a cancer risk of $1x10^{-5}$ for both unlined units ($5x10^{-4}$) and clay-lined units ($2x10^{-4}$) and thallium shows a noncancer risk (3) above an HQ of 1 only for unlined units. Figure 4-2 shows that at the 50th percentile, all risks were at or below the risk criteria. Composite-lined units show zero or negligible risks (well below the risk criteria) for all constituents and percentiles examined.

For surface impoundments, the full-scale analysis produced arsenic risk estimates at the 90th percentile above a cancer risk of $1x10^{-5}$ for both unlined units ($9x10^{-3}$) and clay-lined units ($3x10^{-3}$) and a noncancer HQ above the criteria for boron (3), lead (5), cadmium (5), cobalt (5), and molybdenum (5) for unlined units, and for boron (2) and molybdenum (3) for clay-lined units. At the 50th percentile, only arsenic has risks above the 10^{-5} risk criterion for unlined ($3x10^{-4}$) and clay-lined ($9x10^{-5}$) surface impoundments. And as with landfills, the risks from composite-lined surface impoundments are well below the risk criteria.

The higher risks for surface impoundments as compared to landfills reflect higher constituent concentrations in the surface impoundment wastes, a higher proportion of unlined units (see Section 4.1.4), 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.

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. These 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 3.2.2), 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 and therefore represent an uncertainty in this analysis (see Sections 3.2.2 and 4.4.3.2).

Composite liners also significantly 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 risk criteria for this analysis. Infiltration rates for composite-lined surface impoundments are largely controlled by leak density (see Section 3.3), which is an empirical distribution from the same source as the landfill infiltration rates (U.S. EPA, 2002b), and are subject to similar uncertainties.

Arrival times for the peak arsenic concentration used to calculate risks are plotted as cumulative distributions for surface impoundments and landfills in Figure 4-3. As can be seen in the figure, the peak arrival time for surface impoundments is usually less than 100 years (i.e., peak concentration occurs shortly after closure); the 50th percentile is 78 years, and the 75th percentile is 105 years.[2] Arrival times for landfills are much longer, ranging from hundreds to thousands of years; the 50th percentile is 618 years and the 75th percentile is 3,343 years. The shorter arrival times for surface impoundments are primarily due to the hydraulic head of the waste liquids in the unit and the lower prevalence of liners in surface impoundments; by contrast, landfill leaching is driven by infiltration of precipitation through the cap and liner of the unit.

The arrival time of the peak concentration corresponds to the arrival of the maximum risk; however, for runs where the risk exceeds the risk criteria, the concentration that results in risk at the risk criteria will arrive somewhat before the peak concentration. Overall, however, the time to reach the risk criteria should be similar to the peak arrival times shown in Figure 4-3.

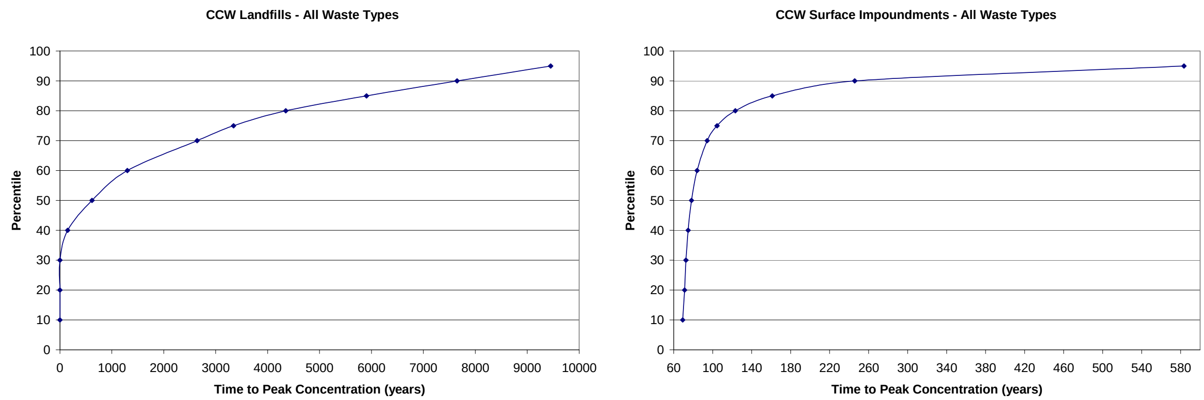


Figure 4-3. Comparison of peak arrival times for arsenic for CCW landfills and surface impoundments.

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

Tables 4-3 and 4-4 present the 90th and 50th percentile risk results, respectively, for the fish consumption pathway, where fish are contaminated by groundwater seeping into a waterbody downgradient from the WMU. The results presented are for a fisher's child because those risks were consistently higher than the risks for the adult fisher. Results for arsenic are based on arsenic III, which is more mobile in soil and groundwater (and so had higher receptor concentrations). The selenium results are based on selenium VI, which also represents the highest receptor concentrations. The uncertainty resulting from the model's inability to speciate metals during transport is discussed in Section 4.4.2.

For surface impoundments, 90th percentile selenium and arsenic risks for unlined units are slightly above a cancer risk of $1x10^{-5}$ ($2x10^{-5}$, arsenic) and slightly above a noncancer HQ of 1 (2 for selenium). Risks are below the risk criteria for clay-lined and composite-lined surface impoundments. Again, risks are higher for surface impoundments than for landfills (where risks are below risk criteria for all constituents) because of the higher waste concentrations, higher hydraulic head in these units, and a lower prevalence of liners, as discussed previously for the drinking water pathway. Fish consumption pathway 50th percentile results are well below the risk criteria for all constituents, waste management scenarios, and liner types.

[2] In other words, 50 percent of the arrival times are less than 78 years and 75 percent are less than 105 years.

As with the groundwater-to-drinking-water pathway analysis, the absence of risk from composite-lined units suggests that the composite liners modeled in this analysis are effective at preventing contaminants from reaching the surface waterbodies of interest.

Table 4-3. Summary of 90th Percentile Full-Scale CCW Human Risk Results: Groundwater-to-Surface-Water (Fish Consumption) Pathway

Chemical[b]	90th Percentile HQ or Cancer Risk Value[a]			
	All Units Combined[c]	Unlined Units	Clay-Lined Units	Composite–Lined Units
Landfills				
Arsenic (cancer)	6E-07	1E-06	3E-07	0
Selenium	0.3	0.7	0.1	0
Thallium	0.2	0.4	0.07	0
Cadmium	0.02	0.06	9E-03	0
Surface Impoundments				
Arsenic (cancer)	1E-05	2E-05	7E-06	6E-13
Selenium	2	2	1	2E-06
Cadmium	0.1	0.2	0.09	3E-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] Note that not every chemical that was selected for full-scale modeling was modeled in every pathway/WMU scenario: only chemicals with adequate data and that were identified in the screening analysis as needing further assessment (see Section 2.1) were modeled for each scenario.

[c] Results across all unit types combined (unlined, clay-lined, and composite-lined).

Table 4-4. Summary of 50th Percentile Full-Scale CCW Human Risk Results: Groundwater-to-Surface-Water (Fish Consumption) Pathway

Chemical[b]	50th Percentile HQ or Cancer Risk Value[a]			
	All Units Combined[c]	Unlined Units	Clay–Lined Units	Composite–Lined Units
Landfills				
Arsenic (cancer)	6E-11	1E-09	3E-10	0
Selenium	5E-05	7E-04	2E-04	0
Thallium	3E-05	5E-04	2E-04	0
Cadmium	2E-06	5E-05	8E-06	0
Surface Impoundments				
Arsenic (cancer)	2E-08	5E-08	3E-09	0
Selenium	3E-03	7E-03	4E-04	0
Cadmium	3E-04	9E-04	3E-05	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 not every chemical that was selected for full-scale modeling was modeled in every pathway/WMU scenario: only chemicals with adequate data and that were identified in the screening analysis as needing further assessment (see Section 2.1) were modeled for each scenario.

[c] Results across all unit types combined (unlined, clay-lined, and composite-lined).

4.1.3 Results by Waste Type/WMU Scenario

As described in Section 3.1, 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.[3] Section 4.1.3.1 presents these results by waste type. FBC wastes were also modeled in this assessment, but because of the small number of FBC waste disposal sites (7) in the EPRI/EPA database, the results are treated separately in Section 4.1.3.2.

4.1.3.1 Conventional CCW and CCW Codisposed with Coal Refuse

Tables 4-5 and 4-6 show 90^{th}- and 50^{th}-percentile risk results, respectively, by waste type and unit type for CCW landfills for the groundwater-to-drinking-water pathway. There was little difference in results between waste types for landfills, which showed very similar risks for conventional CCW and codisposed CCW and coal refuse. Risks are a factor of 2 or 3 greater for unlined landfills than for clay-lined landfills. For conventional CCW in landfills, arsenic cancer risks are 4×10^{-4} for unlined units, 2×10^{-4} for clay-lined units, and 0 for composite-lined units at

[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. 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.”

the 90^{th} percentile. Noncancer risks at the 90^{th} percentile exceeded 1 for only thallium in unlined units (3) and clay-lined units (2) and antimony in unlined units (2). For codisposed CCW and coal refuse in landfills, arsenic cancer risks are $5x10^{-4}$ for unlined units, $2x10^{-4}$ for clay-lined units, and 0 for composite-lined units at the 90^{th} percentile. Noncancer hazard quotients at the 90^{th} percentile exceeded 1 for only thallium in unlined units (2) and molybdenum in unlined units (2). 50^{th} percentile risks for the groundwater-to-drinking-water pathway were below the risk criteria for all waste types in all types of landfills. Landfills with composite liners show zero risks as modeled in this assessment (see Section 4.1.4 for a further discussion of risks by liner type).

The difference in risks between waste types is greater for surface impoundments. Tables 4-7 and 4-8 show 90th and 50th percentile risk results, respectively, by waste type and liner type for CCW surface impoundments (for the drinking water pathway). For conventional CCW in surface impoundments, arsenic cancer risks are $2x10^{-3}$ for unlined units, $9x10^{-4}$ for clay-lined units, and below the risk criteria for composite-lined units at the 90^{th} percentile. Noncancer hazard quotients at the 90^{th} percentile exceeded 1 for nitrate/nitrite (20), molybdenum (8), boron (7), selenium (2), and lead (3) in unlined units, and nitrate/nitrite (10), molybdenum (5) and boron (4) in clay-lined units. None of the risk criteria were exceeded at the 90^{th} percentile in composite-lined units. For codisposed CCW and coal refuse in surface impoundments, arsenic cancer risks are $2x10^{-2}$ for unlined units, $7x10^{-3}$ for clay-lined units, and below the risk criteria for composite-lined units at the 90^{th} percentile. Noncancer hazard quotients at the 90^{th} percentile exceeded 1 for cadmium (9), cobalt (8), lead (9), and molybdenum (3) in unlined units, and cadmium (3), cobalt (3), and molybdenum (2) in clay-lined units. None of the risk criteria were exceeded at the 90^{th} percentile in composite-lined units. As noted above, codisposal of CCW and coal refuse in surface impoundments results in risks up to 10-fold greater than those seen for conventional CCW managed in surface impoundments. This is likely due to the higher metal concentrations and the acidity of coal refuse leachate[4] for surface impoundments in the CCW database. As with landfills, clay-lined units show lower risks by a factor of 2 or 3 than unlined units, and composite liners show negligible or zero risks for either waste type.

[4] Metals tend to show greater solubility and mobility in acidic leachate.

Table 4-5. 90th Percentile Risk Results by CCW Type: Landfills, Groundwater-to-Drinking-Water Pathway

Chemical	90th Percentile HQ or Cancer Risk Value[a]			
	All Units Combined[b]	Unlined Units	Clay-Lined Units	Composite-Lined Units
Conventional CCW – 79 landfills				
Arsenic (cancer)	3E-04	4E-04	2E-04	0
Thallium	2	3	2	0
Antimony	1	2	0.8	0
Molybdenum	0.9	1	0.8	0
Lead (MCL)[b]	0.5	1	0.3	0
Cadmium	0.4	0.7	0.4	0
Boron	0.4	0.7	0.4	0
Selenium	0.1	0.2	0.1	0
Nitrate/nitrite (MCL)[c]	0.07	0.1	0.06	2E-06
Codisposed CCW and Coal Refuse – 41 landfills				
Arsenic (cancer)	3E-04	5E-04	2E-04	0
Thallium	1	2	1	0
Molybdenum	0.8	2	0.6	0
Antimony	0.5	0.8	0.3	0
Selenium	0.4	0.7	0.3	0
Lead (MCL)[c]	0.3	0.7	0.09	0
Boron	0.2	0.3	0.1	0
Nitrate/nitrite (MCL)[c]	0.2	0.2	0.1	3E-06
Cadmium	0.1	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] HQ or risk across all unit types combined (unlined, clay-lined, and composite-lined).
[c] Values are ratios of exposure concentration to MCL.

Table 4-6. 50th Percentile Risk Results by CCW Type: Landfills, Groundwater-to-Drinking-Water Pathway

Chemical	50th Percentile HQ or Cancer Risk Value[a]			
	All Units Combined[b]	Unlined Units	Clay-Lined Units	Composite-Lined Units
Conventional CCW – 79 landfills				
Arsenic (cancer)	2E-06	6E-06	4E-06	0
Thallium	0.08	0.2	0.1	0
Antimony	0.02	0.04	0.02	0
Molybdenum	0.03	0.05	0.04	0
Lead (MCL)[b]	3E-08	4E-04	2E-08	0
Cadmium	0.005	0.01	0.008	0
Boron	0.007	0.01	0.01	0
Selenium	0.004	0.009	0.006	0
Nitrate/nitrite (MCL)[c]	0.002	0.004	0.003	0
Codisposed CCW and Coal Refuse – 41 landfills				
Arsenic (cancer)	4E-06	2E-05	6E-06	0
Thallium	0.06	0.2	0.07	0
Molybdenum	0.006	0.02	0.006	0
Antimony	0.01	0.05	0.02	0
Selenium	0.008	0.03	0.01	0
Lead (MCL)[c]	6E-07	0.01	2E-07	0
Boron	0.002	0.008	0.003	0
Nitrate/nitrite (MCL)[c]	0.01	0.04	0.009	0
Cadmium	0.003	0.02	0.004	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] HQ or risk across all unit types combined (unlined, clay-lined, and composite-lined).
[c] Values are ratios of exposure concentration to MCL.

Table 4-7. 90th Percentile Risk Results by CCW Type: Surface Impoundments, Groundwater-to-Drinking-Water Pathway

Chemical	90th Percentile HQ or Cancer Risk Value[a]			
	All Units Combined[b]	**Unlined Units**	**Clay-Lined Units**	**Composite-Lined Units**
Conventional CCW – 44 surface impoundments				
Arsenic (cancer)	1E-03	2E-03	9E-04	2E-07
Nitrate/nitrite (MCL)[c]	10	20	10	9E-04
Molybdenum	6	8	5	7E-03
Boron	5	7	4	5E-03
Selenium	2	2	1	1E-03
Lead (MCL)[c]	1	3	0.7	1E-21
Cadmium	0.4	0.5	0.3	4E-11
Cobalt	0.01	0.01	6E-03	0
Codisposed CCW and Coal Refuse – 72 surface impoundments				
Arsenic (cancer)	2E-02	2E-02	7E-03	4E-06
Cadmium	8	9	3	5E-05
Cobalt	7	8	3	4E-08
Lead (MCL)[c]	6	9	1	1E-19
Molybdenum	3	3	2	4E-03
Boron	1	1	0.5	2E-03
Selenium	0.8	0.8	0.4	1E-03
Nitrate/nitrite (MCL)[c]	0.3	0.4	0.2	1E-04

[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] HQ or risk across all unit types combined (unlined, clay-lined, and composite-lined).
[c] Values are ratios of exposure concentration to MCL.

Table 4-8. 50th Percentile Risk Results by CCW Type: Surface Impoundments, Groundwater-to-Drinking-Water Pathway

Chemical	50th Percentile HQ or Cancer Risk Value[a]			
	All Units Combined[b]	Unlined Units	Clay-Lined Units	Composite–Lined Units
Conventional CCW – 44 surface impoundments				
Arsenic (cancer)	7E-05	1E-04	6E-05	0
Nitrate/nitrite (MCL)[c]	0.05	0.1	0.05	7E-08
Molybdenum	0.6	1.1	0.5	2E-11
Boron	0.2	0.4	0.2	3E-11
Selenium	0.07	0.1	0.07	2E-11
Lead (MCL)[c]	0.02	0.05	0.007	0
Cadmium	0.03	0.05	0.02	0
Cobalt	0.001	0.003	8E-04	0
Codisposed CCW and Coal Refuse – 72 surface impoundments				
Arsenic (cancer)	4E-04	6E-04	2E-04	0
Cadmium	0.1	0.1	0.05	0
Cobalt	0.3	0.4	0.09	0
Lead (MCL)[c]	0.09	0.1	0.01	0
Molybdenum	0.6	0.8	0.3	3E-18
Boron	0.1	0.1	0.06	5E-15
Selenium	0.08	0.1	0.03	5E-15
Nitrate/nitrite (MCL)[c]	0.02	0.03	0.01	4E-08

[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] HQ or risk across all unit types combined (unlined, clay-lined, and composite-lined).
[c] Values are ratios of exposure concentration to MCL.

4.1.3.2 FBC Wastes

Tables 4-9 and 4-10 show the 90^{th}- and 50^{th}-percentile risk results for FBC landfills by unit type. At the 90^{th} percentile in landfills, arsenic cancer risks are $3x10^{-5}$ for unlined units, $6x10^{-5}$ for clay-lined units, and 0 for composite-lined units. Noncancer hazard quotients exceed 1 for only thallium (4) and antimony (3) in clay-lined units. No risks exceeded the risk criteria at the 50^{th} percentile. 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] and illustrates how the probabilistic analysis design and availability of facility data can impact risk results (and why FBC results are treated separately in the risk characterization). As presented in Section 3.1.2 and in Figure 3-2, the Monte Carlo analysis was designed to evaluate risks posed by current waste management practices for a given WMU type, waste type, and waste constituent. This approach limits the effects of data availability for the different liner configurations when the risks are aggregated over all units (lined and unlined) combined. However, when the risk results of an exposure pathway are viewed at a resolution finer than the analysis design, a small sample size for a particular waste and WMU type scenario (as occurs for FBC waste), along with the interactions of liner type with other site-based inputs (notably infiltration rate and the size of the WMU), can produce unexpected results. In the case of FBC wastes, the characteristics (primarily infiltration rate and areas) of the three unlined landfills were such that their risks were lower than the clay-lined FBC landfills.

Table 4-9. 90th Percentile Risk Results for FBC Wastes: Landfills, Groundwater-to-Drinking-Water Pathway

Chemical	90th Percentile HQ or Cancer Risk Value[a]			
	All Units Combined[b]	Unlined Units	Clay-Lined Units	Composite-Lined Units
FBC Waste – 7 landfills				
Arsenic (Cancer)	4E-05	3E-05	6E-05	0
Thallium	2	1	4	0
Antimony	1	0.8	3	0
Lead (MCL)[c]	0.4	0.4	0.6	0
Molybdenum	0.3	0.2	0.5	0
Cadmium	0.2	0.1	0.3	0
Selenium	0.1	0.08	0.1	0
Nitrate/nitrite (MCL)[c]	0.05	0.03	0.07	5E-08
Boron	0.04	0.02	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] HQ or risk across all unit types combined (unlined, clay-lined, and composite-lined).
[c] Values are ratios of exposure concentration to MCL.

[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.

Table 4-10. 50th Percentile Risk Results for FBC Wastes: Landfills, Groundwater-to-Drinking-Water Pathway

Chemical	50th Percentile HQ or Cancer Risk Value[a]			
	All Units Combined[b]	Unlined Units	Clay-Lined Units	Composite-Lined Units
FBC Waste – 7 landfills				
Arsenic (Cancer)	0	0	4E-07	0
Thallium	0.008	0	0.2	0
Antimony	0.002	0	0.09	0
Lead (MCL)[c]	0	0	2E-04	0
Molybdenum	0.003	0	0.04	0
Cadmium	4E-07	0	0.01	0
Selenium	3E-04	0	0.01	0
Nitrate/nitrite (MCL)[c]	1E-04	3E-08	0.004	0
Boron	2E-04	0	0.003	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] HQ or risk across all unit types combined (unlined, clay-lined, and composite-lined).
[c] Values are ratios of exposure concentration to MCL.

4.1.4 Results by Unit Type

The effect of unit type on human health risk for the groundwater-to-drinking-water pathway can be seen in Tables 4-1 and 4-2, which compare 90th and 50th percentile risks, respectively, 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 4-11 shows the frequency of each of the unit types in the 1995 EPRI survey data modeled in this analysis, and it compares these data with the unit 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 between 61 and 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. As Table 4-11 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 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.

Table 4-11. Unit 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 described in Sections 3.2.1 and 3.3.1, 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 x 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 the risk criteria (arsenic and thallium). For surface impoundments, clay liners did reduce the risk to just below the risk criteria for cobalt, lead, and selenium.

Composite (clay and synthetic) liners, as modeled in this risk assessment (see Sections 3.2 and 3.3), were much more effective at reducing risk for all constituents; 90th (and

[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..

50th) percentile risks with composite liners for landfills were zero[7] for arsenic and metals and very low or zero for nitrate/nitrite, and were well below the risk criteria for all constituents for surface impoundments. The analysis used data collected for composite liner performance at industrial waste management 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 are not available, there is some uncertainty in applying these Industrial D liner performance data to CCW disposal units. Still, these rates do reflect actual performance data from liners under real WMUs, and they demonstrate that composite liners can be effective in reducing leaching from CCW WMUs and suggest that there will be a significant decrease in risk from CCW disposal if more facilities line their WMUs with composite liners. Information from the more recent DOE/EPA study (U.S. DOE, 2006) indicates that composite liners are much more prevalent in newly constructed facilities, so the risks from CCW disposal should be lower for newer CCW landfills and surface impoundments.

4.1.5 Constituents Not Modeled in the Full-Scale Assessment

As described in Section 2.1.1.2, resources did not allow full-scale modeling to be conducted for all 21 constituents that were above the screening criteria in the initial screening analysis; nine constituents that were judged to likely have generally lower risks to human health and ecological risks were not modeled in the full-scale risk assessment.[8] Five of these 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 constituents, we developed surrogate risk attenuation factors by dividing the screening risk results by the full-scale risk results, across all unit 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 4-12 shows the risk attenuation factor statistics for the modeled constituents, and Table 4-13 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 reflects differences in contaminant sorption and mobility. The 10th percentile attenuation factor was selected as a conservative value representing the more mobile constituents, such as arsenic, selenium, and molybdenum. The 50th percentile (or median) risk represents a central tendency value.

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.

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[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.

[8] These constituents of marginal concern had no human health HQs greater than 6 and only one or no ecological HQs greater than 100.

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 the risk criteria, with HQs ranging from 1.4 to 3.5. This is consistent with the general trend in this analysis of surface impoundments showing higher risks and more risks exceeding the risk criteria than CCW landfills.

Table 4-12. 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
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 4-13. Summary of Risk Results for 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

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 is, the comparison of modeled exposure concentrations to CSCLs to estimate risk has a binary outcome: either the constituent concentration is above the environmental quality criteria (HQ greater than 1) or the concentration is below the criteria (HQ less than or equal to 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 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. The ecological risk results are presented for all unit types combined and were not broken out separately for the different unit types.

The ecological risk results suggest the potential for adverse ecological effects to 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 Surface Water Receptors

Table 4-14 presents the 90th and 50th percentile results for the groundwater-to-surface-water pathway for surface water receptors for landfills and surface impoundments. For landfills, only boron (200) and lead (4) show HQs above the risk criteria at the 90th percentile. For surface impoundments, boron (2000), lead (20), arsenic (10), selenium (10), cobalt (5), and barium (2) showed 90th percentile risks above the risk criteria. The 50th percentile results are well below an HQ of 1 for landfills and only exceed an HQ of 1 for boron (4) in surface impoundments.

The difference in the number and magnitude of HQs that exceed the risk criterion between landfills and surface impoundments is likely the result of higher CCW constituent concentrations in surface impoundment porewater and 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.

Table 4-14. Summary of Full-Scale CCW Ecological Risk Results: Groundwater-to-Surface-Water Pathway, Aquatic Receptors[a]

Chemical	90th Percentile HQ	50th Percentile HQ	Pathway	Receptor
Landfills				
Boron	200	0.04	direct contact	aquatic biota
Lead	4	2E-08	ingestion	river otter
Selenium	1	3E-04	direct contact	aquatic biota
Arsenic	0.7	9E-10	direct contact	aquatic biota
Barium	0.8	3E-18	direct contact	aquatic biota
Cadmium	0.3	3E-05	direct contact	aquatic biota
Aluminum	0.008	1E-09	direct contact	aquatic biota
Surface Impoundments				
Boron	2000	4	direct contact	aquatic biota
Lead	20	0.02	ingestion	river otter
Arsenic	10	0.01	direct contact	aquatic biota
Selenium	10	0.02	direct contact	aquatic biota
Cobalt	5	0.007	direct contact	aquatic biota
Barium	2	0.003	direct contact	aquatic biota
Cadmium	1	0.004	direct contact	aquatic biota
Aluminum	0.02	0.0003	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.

4.2.2 Sediment Receptors

Table 4-15 presents the 90th and 50th percentile results of the ground-water-to-surface-water pathway for sediment receptors for landfills and surface impoundments. For landfills, lead, (20), arsenic (6), and cadmium (2) show 90th percentile risks above the risk criteria. For surface impoundments, lead (200), arsenic (100), and cadmium (20) showed 90th percentile risks above the risk criteria. Although cadmium was not above the risk criterion in surface water, it did have an HQ of 20 in sediments at the 90th percentile. None of the constituents modeled showed sediment risks at or above the risk criteria at the 50th percentile.

Table 4-15. Summary of Full-Scale CCW Ecological Risk Results: Groundwater-to-Surface-Water Pathway, Sediment Receptors[a]

Chemical	90th Percentile HQ	50th Percentile HQ	Pathway	Receptor
Landfills				
Lead	20	3E-08	ingestion	spotted sandpiper
Arsenic	6	7E-04	ingestion	spotted sandpiper
Cadmium	2	6E-05	direct contact	sediment biota
Antimony	0.9	4E-05	direct contact	sediment biota
Molybdenum	0.05	1E-05	ingestion	spotted sandpiper
Barium	0.002	6E-21	ingestion	spotted sandpiper
Surface Impoundments				
Lead	200	0.05	ingestion	spotted sandpiper
Arsenic	100	0.2	ingestion	spotted sandpiper
Cadmium	20	0.009	direct contact	sediment biota
Molybdenum	0.7	0.002	ingestion	spotted sandpiper
Barium	0.007	8E-06	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.

4.2.3 Constituents Not Modeled in the Full-Scale Assessment

As described in Section 2.1.1.2, resources did not allow full-scale modeling to be 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 modeled in the full-scale assessment. Tables 4-16 and 4-17 show the results of this comparison for the surface water ecological risk exposure pathway. Table 4-16 shows the risk attenuation factors for the modeled constituents, and Table 4-17 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

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

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 to 64, reflecting higher contaminant mobility from the higher hydraulic head in the surface impoundments and a lower prevalence of liners (compared to landfills). 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 risks only slightly above the risk criteria with the10th percentile attenuation (HQs ranging from 1.4 to 1.9). It is unlikely that these results represent true risks above the risk criteria: vanadium and copper are likely less mobile than the 10th percentile attenuation factor reflects (thus the true risk is likely lower), and the risks for chromium are based on the highly conservative assumption of 100 percent hexavalent chromium.

Table 4-16. Risk Attenuation Factor[a] Statistics for Modeled Constituents—Ecological Risk, Surface Water Pathway

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.

Table 4-17. Summary of Risk Results 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	-	-	-

4.3 Sensitivity Analysis

EPA conducted a sensitivity analysis on the probabilistic risk assessment to determine which model inputs were most important to risk, which in turn will help focus additional analyses or data collection efforts on the most important drivers of risk, and help identify the important factors to consider when evaluating regulatory and management options for CCW. The sensitivity analysis also can help identify parameters that are both sensitive and highly uncertain, which affects the confidence in the results. This 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 is not particularly useful for the purposes of the sensitivity analysis: to help direct additional analyses in support of developing CCW regulatory options, to help focus any future data collection efforts on the most sensitive variables, or to better understand sources of uncertainty in the CCW risk results.

The outputs from the sensitivity analysis are the 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 a larger number of input parameters that were significant (seven) than the fish consumption pathway (three). The most sensitive parameters for most (over 75 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 gradient (i.e., groundwater velocity).

For strongly sorbing contaminants (i.e., metals with high soil/water partition coefficients), sorption and travel time parameters become more important, including

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

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

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 (2005). In terms of the model inputs, the sensitivity analysis found that the most consistent drivers of the risk results are 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.

4.4 Variability and Uncertainty

> **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.

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.

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, we 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 is 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 resources regions throughout the contiguous 48 states.

In summary, a distribution of exposures was developed that includes 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 also was 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 would be sampled based on the soil and aquifer types that are likely to be present at the site.

The treatment of variability and uncertainty in model parameters using a Monte Carlo simulation forms the basis for the national exposure distributions used in this analysis to estimate risk. Previous sections of this document describe how we 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.

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 attempts to characterize current conditions by estimating risks from actual CCW disposal sites across the country, it is 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 is based.

Receptor Populations Evaluated. The human receptors evaluated for the CCW risk analysis are 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 is assumed that adults and children are exposed daily and that the private well is 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 conservative, 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.

Additive Risks Across Pathways. The human receptors evaluated in the CCW risk assessment are assumed not to consume both contaminated fish and drinking water. 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 significantly change the conclusions of this risk assessment in terms of the constituents and exceeding the risk criteria.

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 of the most intractable problems in conducting a predictive ecological risk assessment intended to reflect risks at a national scale is evaluating *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 ecosystem degradation is proceeding at an unprecedented rate, and 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., 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-based approach in which locations, habitats, and species of concern are identified and characterized with respect to the spatial scale of constituent releases.

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 is a probabilistic assessment that predicts 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.

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 employed in this assessment have been developed over many years to support regulatory applications. As a result they have been designed to be protective towards 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.

Arsenic Speciation. Because the models used in this assessment do not speciate metals during soil or groundwater transport, arsenic 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. Because the soil and groundwater models assume one form for each model run, the risk results presented for arsenic are based on arsenic III, which is a conservative, protective assumption (i.e., arsenic III has higher risks than arsenic V). 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 rapidly converted to arsenic V during subsurface transport, with the result that drinking water standards are rarely exceeded in offsite groundwater in spite of high landfill leachate concentrations (see, for example, U.S. EPA, 2000; U.S. EPA, 2003e; Lang and Schlictmann, 2004; Zillmer and Fauble, 2004). To address this uncertainty (i.e., how much an arsenic III assumption might overpredict offsite well concentrations) the models were run assuming arsenic V as the arsenic species in soil and groundwater. Figure 4-4 compares the risk results for arsenic III and arsenic V. Arsenic V has lower risks than arsenic III by about a factor of two, but the 90th percentile risks are still above risk criteria.

Bioavailability of Constituents to Ecological Receptors. For the purposes of this analysis, the model assumes that all forms of a constituent are equally bioavailable to ecological receptors, and therefore, the actual exposures that may occur in the field tend to be overestimated, thus making this a protective 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.

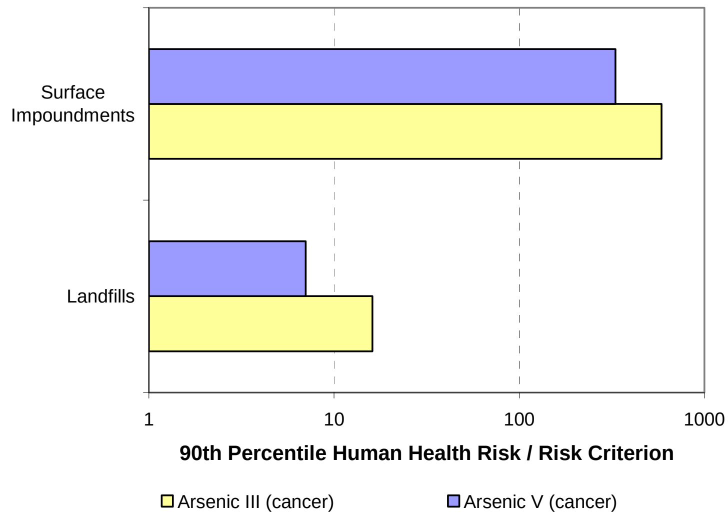


Figure 4-4. Comparison of risk results for arsenic III and arsenic V (based on results for all units combined).

Multiple Constituent Exposures to Receptors. The risk from each constituent was considered separately in this analysis. However, the waste concentration data on CCWs (as well as recent field studies such as U.S. EPA, 2006) suggest that exposure to multiple constituents is highly likely. The synergism or antagonism between different constituent combinations may elicit unexpected adverse impacts to humans and ecosystems. Hence, a single-constituent analysis may underestimate risks associated with multiple chemical stressors.

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 will 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

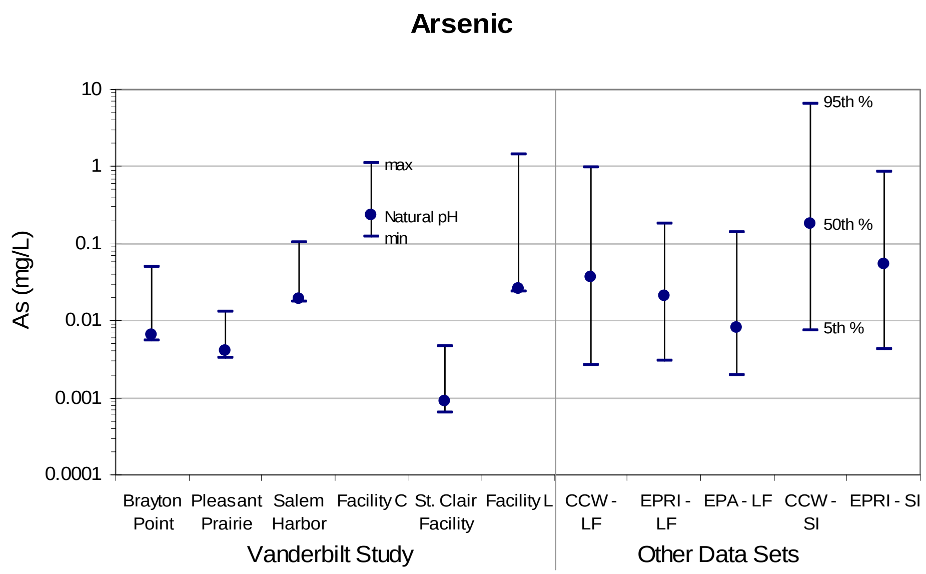
The CCW constituent database used to represent CCW total waste and waste leachate concentrations is arguably the most important data set in terms of driving the risk assessment results. The constituent data are subject to two primary uncertainties beyond the normal sampling and analysis uncertainty associated 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.

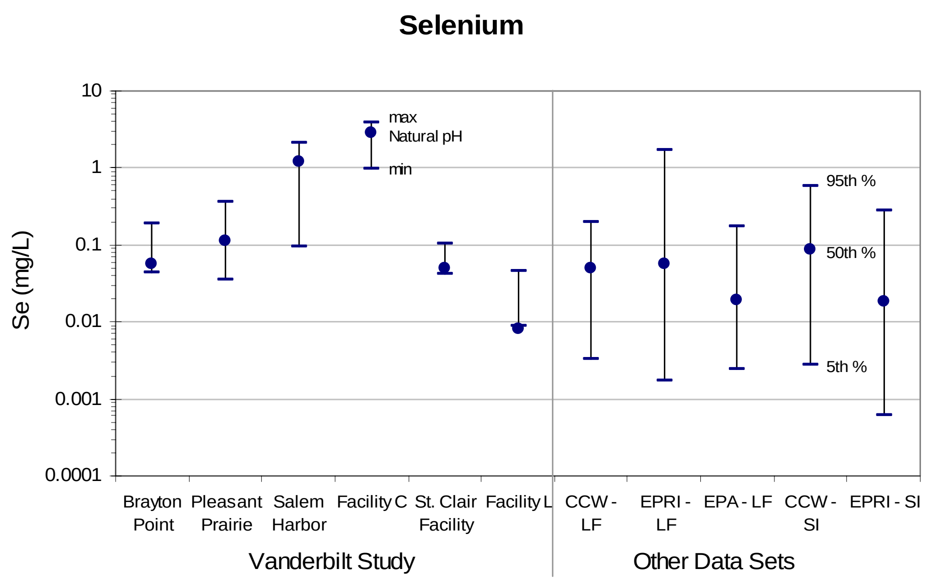
Appropriateness of Leachate Data. The CCW leachate data were collected from a varying number of sites using a variety of methods. The available landfill data are largely derived from the TCLP, a laboratory test designed to estimate leachate concentrations in municipal solid waste (MSW) landfills. The 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, this does not appear to be a significant source of uncertainty for this analysis.

Surface impoundment leachate is represented by porewater data taken beneath actual impoundments, but although these 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. Antimony, cobalt, mercury, and thallium are represented by one to only a few sites and only a few measurements, and results associated with these metals should be interpreted with caution. 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 been 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, 2006). Figure 4-5 plots the results from this study for arsenic and selenium, along with data from EPA's Leach2000 database and EPRI (as provided in U.S. EPA, 2006), against the data used for landfills and surface impoundments used in the CCW analysis. For the Vanderbilt leaching study, 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 is 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 be somewhat underestimated for landfills in this analysis. This is significant even though selenium risks from landfills were not above the risk criteria 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; U.S. EPA, 2003e; Lang and Schlictmann, 2004; Zillmer and Fauble, 2004).





Key to data sets:

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

Figure 4-5. Comparison of CCW leachate data with other leachate data.

Although the Vanderbilt Study does not cover all of the metals addressed in the CCW analysis, its 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 4-18 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. Constituents that could not be addressed in this analysis because of a high number of nondetects include mercury (for landfills and surface impoundments) and thallium, antimony, and cobalt (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).

Recent work by Vanderbilt University (U.S. EPA, 2006) sheds some light on mercury concentrations in leachate from some CCWs. Figure 4-6 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 (U.S. EPA, 2006). 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, which corresponds to a 90th percentile HQ of 20, 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, like the ones in the CCW database, could be biased high because of cross-contamination issues.

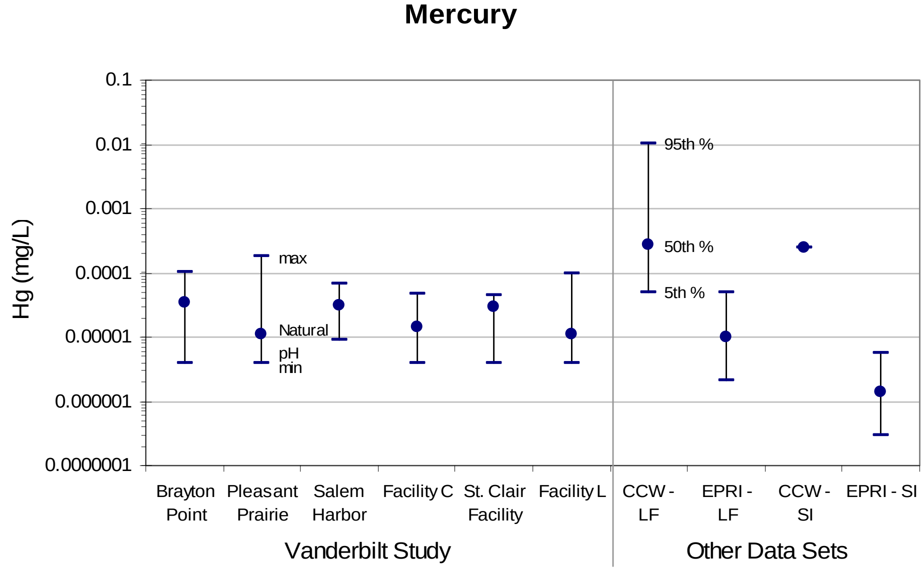
Finally, U.S. EPA (2006) and preliminary results of ongoing EPA studies suggest that both mercury levels and mercury leachability in CCW can vary depending on the flue gas mercury controls used at a power plant. Additional work is underway in this area.

[11] As explained in Appendix A, the CCW risk assessment uses site-averaged constituent concentrations. That is, an average value was used when there were multiple measurements for a chemical at a particular site.

Table 4-18. Proportion of Nondetect Analyses for Modeled CCW Constituents

Chemical	Measurements		Sites	
	Number	% nondetects	Number	% with all nondetects
Landfills				
Aluminum	397	18%	61	5%
Antimony	496	50%	66	41%
Arsenic	1182	49%	128	20%
Barium	1225	11%	126	5%
Boron	930	8%	83	2%
Cadmium	1237	50%	124	31%
Cobalt	559	56%	52	19%
Lead	1109	60%	125	30%
Mercury	***974***	***91%***	***101***	***58%***
Molybdenum	373	24%	58	10%
Nitrate/Nitrite	141	48%	20	15%
Selenium	1227	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%
Thallium	***11***	***100%***	***2***	***100%***

Results for constituents shown in ***bold italics*** were not presented in this report because of high detection limits or limited data.



Key to data sets:

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

Figure 4-6. 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, like 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.

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 one of the larger uncertainties in this analysis. EPA believes that the MSW well distance distribution used is protective for CCW landfills and surface impoundments.

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 location of 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.

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 are selected from a national database of aquifer properties.

Fish Bioconcentration and Bioaccumulation Factors. For fish consumption, exposure dose is 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. The aquatic BCFs were developed based on total surface water concentrations and concentrations in aquatic biota.

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 are not available for children; thus the adult 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 is 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 it was used throughout this assessment to establish statistical distributions of values for each exposure parameter for each receptor. There are some uncertainties, however, in the data that were used. Although it is possible to study various populations to determine various exposure parameters (e.g., age-specific soil ingestion rates or intake rates for food) or to assess past exposures (epidemiological studies) or current exposures, risk assessment is about prediction. Therefore, long-term exposure monitoring in this context is infeasible.

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 as well as 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 conservative estimate of potential risks.

Human Health Benchmarks. EPA routinely accounts for uncertainty in its development of RfDs and other human health benchmarks. For example, if certain toxicological data are missing from the overall toxicological database (e.g., reproductive data), EPA accounts for this by applying an uncertainty factor. In general, EPA human health benchmarks are derived using a health-protective approach.

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 can not 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 lowest observed adverse effect level (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

One of the most sensitive parameters in the risk assessment is infiltration rate. Infiltration rate is greatly influenced by whether and how a WMU is lined. The 1994 to 2004 DOE/EPA survey results (U.S. DOE, 2006) do not include 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 is based. This suggests that the risks from future CCW disposal facilities are likely to be lower than the results presented in this report. EPA will continue to work to integrate the DOE/EPA survey data into the CCW risk assessment and is seeking comments on how to address data gaps, in particular: (1) how to estimate the overall prevalence of liners in the CCW disposal facilities today, (2) how to determine the area and capacity of newer CCW landfills and surface impoundments, and (3) how the liners currently in CCW WMUs perform when compared to the industrial liner conditions assumed in this risk assessment.

Composite liners, as modeled in this risk assessment, effectively reduce risks from all pathways and constituents below the risk criteria for both landfills and surface impoundments.[12] 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. Risks from clay-lined units, as modeled, are about one-third to one-half the risks of unlined units, but are still above the risk criteria used for this analysis. 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). Key risk findings include the following:

- For humans exposed via the groundwater-to-drinking-water pathway, arsenic in CCW landfills poses a 90th percentile cancer risk of 5×10^{-4} for unlined units and 2×10^{-4} for clay-lined units. The 50th percentile risks are 1×10^{-5} (unlined units) and 3×10^{-6}(clay-lined

[12] These results suggest that with the 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).

units). Risks are higher for surface impoundments, with an arsenic cancer risk of $9x10^{-3}$ for unlined units and $3x10^{-3}$ for clay-lined units at the 90th percentile. At the 50th percentile, risks for unlined surface impoundments are $3x10^{-4}$, and clay-lined units show a risk of $9x10^{-5}$. Five additional constituents have 90th percentile noncancer risks above the criteria (HQs ranging from greater than 1 to 4) for unlined surface impoundments, including boron and cadmium, which have been cited in CCW damage cases referenced above. Boron and molybdenum show HQs of 2 and 3 for clay-lined surface impoundments. None of these noncarcinogens show HQs above 1 at the 50th percentile for any unit type.

- Arrival times of the peak concentrations at a receptor well are much longer for landfills (hundreds to thousands of years) than for surface impoundments (most less than 100 years).
- For humans exposed via the groundwater-to-surface-water (fish consumption) pathway, selenium (HQ = 2) and arsenic (cancer risk = $2x10^{-5}$) pose risks slightly above the risk criteria for unlined surface impoundments at the 90th percentile. For both constituents, lined 90th percentile risks and all 50th percentile risks are below the risk criteria. No constituents pose risks above the risk criteria for landfills at the 90th or 50th percentile.
- Waste type has little effect on landfill risk results, but in surface impoundments, risks are up to 1 order of magnitude higher for codisposed CCW and coal refuse than for conventional CCW.
- The higher risks for surface impoundments than landfills are likely due to higher waste leachate concentrations, a lower proportion of lined units, 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 the risk criteria for boron and lead at the 90th percentile, but 50th percentile risks are well below the risk criteria. For surface impoundments, 90th percentile risks for several constituents exceed the risk criteria, with boron showing the highest risks (HQ = 2,000). Only boron exceeds the risk criteria at the 50th percentile (HQ = 4). 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 are generally an order of magnitude or more below the risk criteria.

Sensitivity analysis results indicate that for most of the scenarios evaluated (over 75 percent), the risk assessment model was most sensitive to parameters related to 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 most important.

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) 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 data, some of the higher concentrations in the 2006 data are not represented by the TCLP data. This suggests that selenium risks 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, 2006) suggest that although total mercury will increase in CCW from the use of sorbents as mercury controls, the leachability of mercury may be reduced, but this work is ongoing and should be regarded as preliminary and limited at this time. For example, wet scrubbers have yet to be addressed, and initial data from both EPA and industry studies suggest that mercury may not be as stable as found from fly ash in the first report. As these data become available, EPA will consider how best to use them to update the existing risk assessment.
- **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, but additional data and analyses would be required to estimate the risks from these levels.
- **Arsenic speciation.** The current model does not speciate metals in the subsurface, which is of particular concern for arsenic. Damage cases and other studies suggest that arsenic readily converts from arsenic III in CCW leachate to the less mobile arsenic V in soil and groundwater. However, model runs conducted for both species suggest that the difference in risk between the two species is only about a factor of 2, which is not enough to reduce the 90th percentile cancer risks to below the risk criteria.

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. These would be addressed through more site-specific studies on the proximity of these areas and species to CCW disposal units.
- **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.

5.0 References

ATSDR (Agency for Toxic Substances and Disease Registry). 2002. *Minimal Risk Levels (MRLs) for Hazardous Substances*. http://www.atsdr.cdc.gov/mrls.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. Cross. 1989. Rates of leakage through landfill liners. *Geosynthetics 1989 Conference*. San Diego, CA.

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.

EPRI (Electric Power Research Institute). 1997. *Coal Combustion By-Products and Low-Volume Wastes Comanagement Survey*. Palo Alto, CA. June.

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.

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.

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.

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.

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.

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.

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.

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.

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.

U.S. EPA (Environmental Protection Agency). 1988b. *Wastes from the Combusion 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.

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). 1996. *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). 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.

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.

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). 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.

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.

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. http://www.epa.gov/ epaoswer/hazwaste/id/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/epaoswer/hazwaste/id/ hwirwste/peer03/vadoze/vz-aquif.pdf#search=%22EPACMTP%20peer%20review%22.

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/epaoswer/hazwaste/id/hwirwste/peer03/minteqa2/minteqa2.pdf.

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.

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. http://www.epa.gov/ttn/chief/software/water

U.S. EPA (Environmental Protection Agency). 2001c. *Risk Assessment Paper for Derivation of a Provisional RfD for Cobalt and Compounds (CASRN 7440-48-4)*. 00-122/3-16-01. National Center for Environmental Assessment, Superfund Technical Support Center, Cincinnati, OH.

U.S. EPA (Environmental Protection Agency). 2001d. *Risk Assessment Paper for Derivation of a Provisional Carcinogenicity Assessment for Cobalt and Compounds (CASRN 7740-48-4).* 00-122/3-16-01. National Center for Environmental Assessment. Superfund

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. September.

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.

U.S. EPA (Environmental Protection Agency). 2002c. *Integrated Risk Information System (IRIS)*. National Center for Environmental Assessment, Office of Research and Development, Washington, DC. http://www.epa.gov/iris/.

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. 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. http://www.epa.gov/epaoswer/non-hw/industd/tools/cmtp/epacmtp.htm#data.

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.

U.S. EPA (Environmental Protection Agency). 2003d. *Addendum to the EPACMTP Technical Background Document*. Office of Solid Waste, Washington, DC.

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.

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. *Sensitivity Analysis for the Coal Combustion Waste Risk Assessment.* Technical Report prepared by RTI International for U.S. EPA Office of Solid Waste under EPA contract 68-W-03-042, WA No. 1-16. Washington, DC. March 5.

U.S. EPA (Environmental Protection Agency). 2006. *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.

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.

Appendix A. Constituent Data

The coal combustion waste (CCW) risk assessment addresses metals and inorganic constituents identified by EPA as potential constituents of concern in CCW (Table A-1). U.S. Environmental Protection Agency (EPA) derived waste concentrations for these constituents from its 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 Screening Analysis

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 and 2003. The 2003 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 2003 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 2003 CCW constituent database now covers more than 160 sites. The 1999 data set included approximately 10,000 individual samples of CCW. The 2003 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 2003 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 2003 data set adds 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 2003 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 screening 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.

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 2003 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 is 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 are available in the same rank (which often occurs for SPLP and TCLP analyses), the screening analysis uses the method with the highest analyte concentrations. This ensures that the data used in the risk assessment are the best that are 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 Constituent Screening and Selection

The CCW risk assessment employed two steps to narrow the list of CCW constituents for the full-scale Monte Carlo risk assessment. Two steps were conducted to focus the full-scale analysis on the CCW constituents of most concern:

1. **Hazard Identification**, which involved collection of existing human health and ecological benchmarks for the constituents of concern. Only chemicals with benchmarks moved on to risk screening.
2. **Constituent Screening**, which compared health-based concentration benchmarks against very conservative estimates of exposure concentrations (e.g., whole waste concentrations, leaching concentrations) to quickly and simply "screen out" constituents and exposure pathways of no significant concern.

During the hazard identification step of the CCW risk assessment, constituents of potential concern were first identified by searching, from EPA and other reputable sources, for human health and ecological benchmarks for each chemical in the CCW constituent database. Table A-5 shows the result of that search; of the 41 chemicals in the database, 26 chemicals were found to have benchmarks.

Table A-5. Toxicity Assessment of CCW Constituents

Constituent	CAS ID	HHB	EcoB	Constituent	CAS ID	HHB	EcoB
Metals				***Inorganic Anions***			
Aluminum	7429-90-5	✔	✔	Chloride	16887-00-6		
Antimony	7440-36-0	✔	✔	Cyanide	57-12-5	✔	
Arsenic	7440-38-2	✔[a]	✔	Fluoride	16984-48-8	✔	
Barium	7440-39-3	✔	✔	Nitrate	14797-55-8	✔	
Beryllium	7440-41-7	✔[a]	✔	Nitrite	14797-65-0	✔	
Boron	7440-42-8	✔	✔	Phosphate	14265-44-2		
Cadmium	7440-43-9	✔[a]	✔	Silicon	7631-86-9		
Chromium	7440-47-3	✔[a]	✔	Sulfate	14808-79-8		
Cobalt	7440-48-4	✔[a]	✔	Sulfide	18496-25-8		
Copper	7440-50-8	✔[b]	✔	***Inorganic Cations***			
Iron	7439-89-6			Ammonia	7664-41-7	✔	
Lead	7439-92-1	✔[b]	✔	Calcium	7440-70-2		
Magnesium	7439-95-4			pH	12408-02-5		
Manganese	7439-96-5	✔		Potassium	7440-09-7		
Mercury	7439-97-6	✔	✔	Sodium	7440-23-5		
Molybdenum	7439-98-7	✔	✔	***Nonmetallic Elements***			
Nickel	7440-02-0	✔	✔	Carbon	7440-44-0		
Selenium	7782-49-2	✔	✔	Sulfur	7704-34-9		
Silver	7440-22-4	✔	✔	***Measurements***			
Strontium	7440-24-6	✔		Total Dissolved Solids	none		

Constituent	CAS ID	HHB	EcoB	Constituent	CAS ID	HHB	EcoB
Thallium	7440-28-0	✔	✔	Total Organic Carbon	none		
Vanadium	7440-62-2	✔	✔	Dissolved Organic Carbon	none		
Zinc	7440-66-6	✔	✔				

HHB = human health effect benchmark; EcoB = ecological benchmark.
[a] Carcinogen.
[b] Safe Drinking Water Act Action Level only.

To further narrow the list of constituents, a screening analysis (RTI, 2002) was conducted that compared health-based concentration benchmarks against very conservative estimates of exposure concentrations (e.g., 95th percentile whole waste and leachate concentrations) to quickly and simply "screen out" constituents and exposure pathways posing no significant risk to human health or the environment. Based on the number of pathways with screening failures and how much each chemical exceeded a benchmark, the constituents failing this screen were divided into two groups: (1) those of marginal concern and (2) those of greater concern. Table A-6 shows each of these groups. Constituents of greater concern were subjected to the full-scale probabilistic risk assessment described in this document.

Table A-6. Screening Analysis Results: Selection and Prioritization of CCW Chemicals for Further Analysis

Analyte	Human Health – Drinking Water		Human Health – Surface Water[a]		Ecological Risk – Surface Water		Modeling Priority
	LF Rank [maxHQ]	SI Rank [maxHQ]	LF Rank [maxHQ]	SI Rank [maxHQ]	LF Rank [maxHQ]	SI Rank [maxHQ]	
Constituents of Greater Concern (Full-Scale Analysis)							
Arsenic[b]	1 [140]	1 [1,800]	2 [22]	5 [1.7]	7 [4.9]	3 [64]	1
Boron	6 [4.0]	3 [28]	-	-	2 [660]	1 [4,700]	1
Cadmium	7 [3.4]	7 [8.9]	5 [1.4]	4 [3.7]	11 [2.0]	9 [5.2]	1
Lead	4 [16]	5 [12]	-	-	3 [79]	4 [59]	1
Mercury	-	-	1 [700]	1 [65]	1 [1,400]	2 [132]	1
Selenium	11 [1.2]	13 [2.4]	4 [4.7]	3 [9.5]	8 [3.5]	8 [7.1]	1
Thallium	3 [21]	4 [19]	3 [6.3]	2 [5.7]	-	-	1
Aluminum	-	-	-	-	5 [12]	6 [27]	2
Antimony	2 [22]	10 [5.5]	-	-	-	-	2
Barium	-	-	-	-	4 [40]	7 [7.5]	2
Cobalt		6 [11]	-	-	-	5 [27]	2
Molybdenum	5 [4.2]	8 [6.8]	-	-	-	-	2
Nitrate/Nitrite	- / 12 [1.2]	2 [60]/ 15 [1.2]	-	-	-	-	2
Constituents of Marginal Concern							
Chromium VI	8 [2.3]	12 [4.2]	-	-	12 [1.8]	10 [3.3]	3
Fluoride	10 [1.8]	11 [5.2]	-	-	-	-	3
Manganese	13 [1]	9 [5.6]	-	-	-	-	3
Vanadium	9 [2.2]	14 [2.3]	-	-	10 [2.3]	12 [2.4]	3
Beryllium	-	-	-	-	9 [2.4]	-	4

Analyte	Human Health – Drinking Water		Human Health – Surface Water[a]		Ecological Risk – Surface Water		Modeling Priority
	LF Rank [maxHQ]	SI Rank [maxHQ]	LF Rank [maxHQ]	SI Rank [maxHQ]	LF Rank [maxHQ]	SI Rank [maxHQ]	
Constituents of Greater Concern (Full-Scale Analysis)							
Copper	-	-	-	-	14 [1.6]	11 [3.1]	4
Nickel	-	16 [1.3]	-	-	-	13 [1.4]	4
Silver	-	-	-	-	6 [11]	14 [1.4]	4
Zinc	-	-	-	-	13 [1.6]	-	4

LF = landfill; maxHQ = maximum hazard quotient; SI = surface impoundment.
[a] Fish consumption pathway.
[b] Arsenic values for human health are [excess cancer risk / target risk (1E-05)].

A.4 Results

Attachment A-2 provides the site-averaged constituent data used in the full-scale CCW risk assessment by waste type/WMU scenario.

A.5 References

RTI (RTI International). 2002. *Constituent Screening for Coal Combustion Wastes*. Draft Report prepared for U.S. Environmental Protection Agency, Office of Solid Waste, Washington, DC. September.

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.

Anthracite Region Independent Power Producers Association (ARIPPA). 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.

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: http://www.acaa-usa.org/store-books.htm

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: http://www.acaa-usa.org/store-books.htm

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.

Dairyland Power Cooperative. 1991. Letter to Wisconsin Department of Natural Resources. May 22, 1991.

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.

Electric Power Research Institute (EPRI). 1995. *Laboratory Characterization of Atmospheric Fluidized-Bed Combustion By-Products, Final Report*. EPRI-TR-105527, EPRI, Palo Alto, CA.

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.

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.

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.

Electric Power Research Institute (EPRI). 1998. *Identification of Arsenic Species in Coal Ash Particles*. EPRI-TR-109002, EPRI, Palo Alto, CA.

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.

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.

Electric Power Research Institute (EPRI). 2001. *Environmental Evaluation for Use of Ash in Soil Stabilization Applications*. WO9227-01, EPRI, Palo Alto, CA.

Electric Power Research Institute (EPRI). n.d. Guidance for Comanagement of Mill Rejects at Coal-Fired Power Plants, Pre-Publication. EPRI-TR-108994, EPRI, Palo Alto, CA.

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.

Freeman United Coal Mining Company. 1991–2001. Quarterly Ash Analyses.

Haefner, R.J. n.d. *Water Quality at an Abandoned Mine Site Treated with Coal Combustion By-Products*. U.S. Geological Survey, Water Resources Division, Columbus, OH. Available: http://www.mcrcc.osmre.gov/PDF/Forums/CCB2/4d.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.

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: http://www.acaa-usa.org/store-books.htm

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.

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.

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: http://www.acaa-usa.org/store-books.htm

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.

Longley, R.D. Jr. 1997. Examination of Fly Ash Grouts to Be Used in Acid Mine Drainage Remediation.

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. M.T. Damian, and R. Baker. n.d. Injection of FGD grout to abate acid mine drainage in underground coal mines.

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: 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: 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*. Prepared by the Energy & Environmental Research Center, University of North Dakota, for the U.S. Environmental Protection Agency, Office of Solid Waste.

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.

PPL Generation, LLC. 1994–2001. Coal Ash Quality Results.

Radian Corporation. 1994. *Field Study for Disposal of Solid Wastes from Advanced Coal Processes: Ohio LIMB Site Assessment, Volume I: Text, Final Report*.

Radian Corporation. 1994. *Field Study of Circulating Fluidized-Bed Ash, Final Report*.

Radian Corporation. 1997. *Field Monitoring of Advanced Coal By-Products: Illinois Site, Volume II: Appendices, Final Report*.

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.

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.

Turris Coal Company. 1993–2001. Quarterly Ash Analyses.

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.

Pore Water:

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.

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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
18 – FBC	LF	Selenium	0.108333333	3	1	0.5

(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	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	
22 - FBC	LF	Molybdenum	0.0125	2	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)
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	
33 - FBC	LF	Cadmium	0.00125	1	1	
33 - FBC	LF	Cobalt	0.0025	1	1	
33 - FBC	LF	Mercury	0.00005	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)
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
4 - FBC	LF	Boron	0.693	4	0	13.026
4 - FBC	LF	Cadmium	0.0115	5	2	0.646539286
4 - FBC	LF	Lead	0.1834	5	1	18.35671429
4 - FBC	LF	Mercury	0.00005	5	5	0.087192857

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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	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
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

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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	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	
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	

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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)
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	
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	

(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 – 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
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	

(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)
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	
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	

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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)
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	
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	

(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)
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	
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	

(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 - 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	
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	

(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)
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	
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	

(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)
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	
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	

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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 - 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	
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	

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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	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
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	

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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)
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
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	

(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)
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	
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	

(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 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	
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	

(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)
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	
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	

(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)
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
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

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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)
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	
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	

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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)
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	
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	

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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)
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
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

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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)
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	
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	

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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)
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	
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	

(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)
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
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	

(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 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	
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	

(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)
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	
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	

(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)
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	
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	

(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)
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	
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	

(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)
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	
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	

(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)
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	

Appendix B. Waste Management Units

The source models supporting the coal combustion waste (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.

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 the U.S. Environmental Protection Agency (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 is nearly 62 million tons. This quantity is 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 includes 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 include 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 above) 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 are 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 can 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.

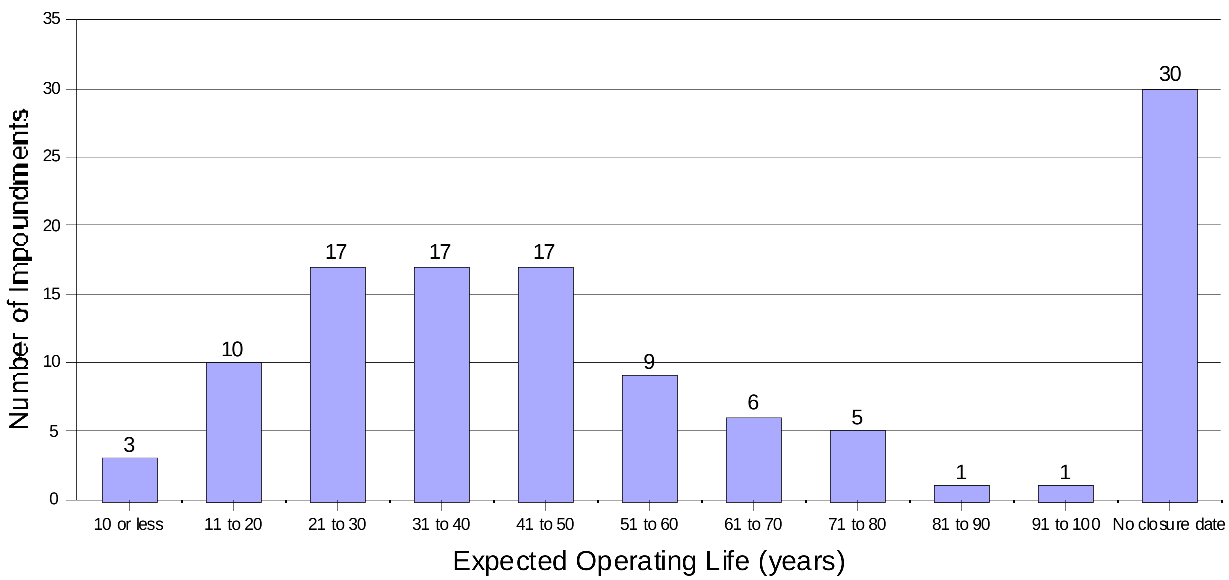

Figure B-1. Operating life of impoundments in the EPRI survey.

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.

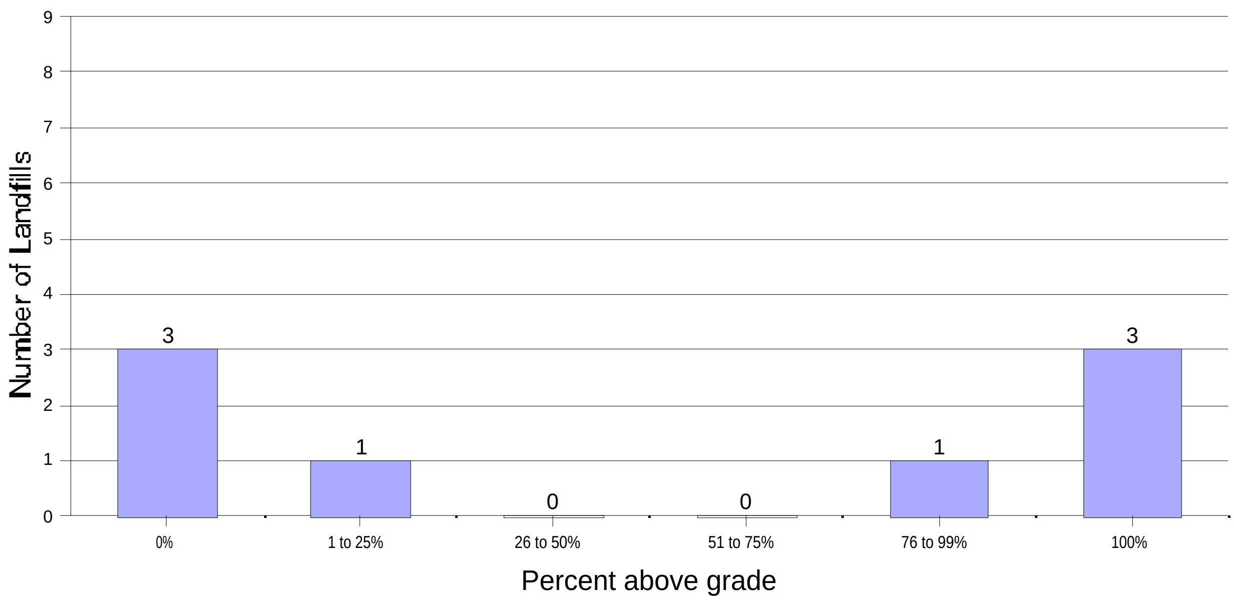

Figure B-2. Above- and below-grade geometry for landfills.

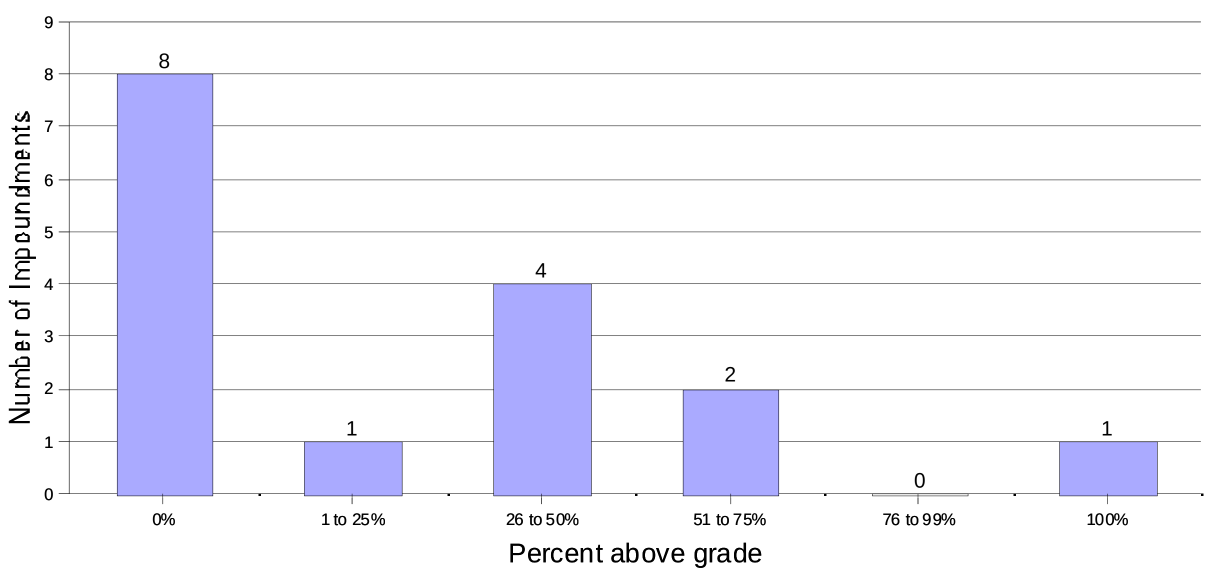

Figure B-3. Above- and below-grade geometry for impoundments.

For landfills, because the data were limited (8 sites), the model runs assume that the percent below grade ranges from 1 to 100 and is uniformly distributed. For each landfill iteration, a random value for percent below grade is picked and applied to the landfill depth to determine depth below ground surface. This value is constrained to be no deeper than the water table and is checked to see that EPACMTP groundwater mounding constraints are 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 is 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.

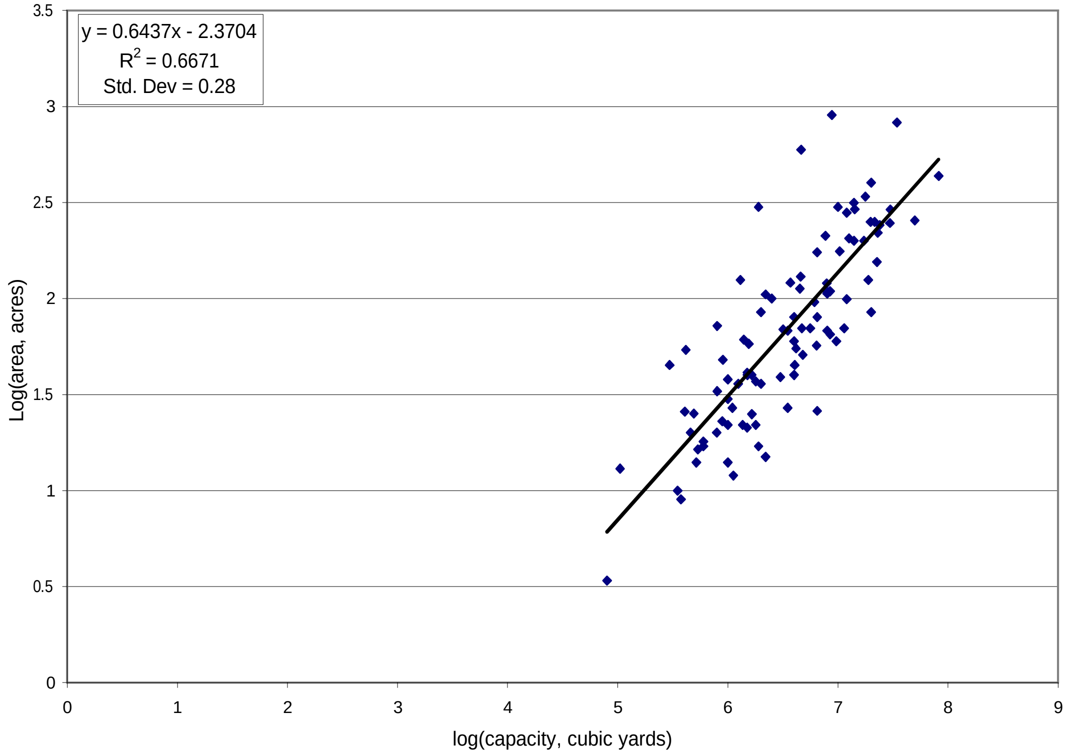

Figure B-4. Linear regression to impute landfill area from capacity.

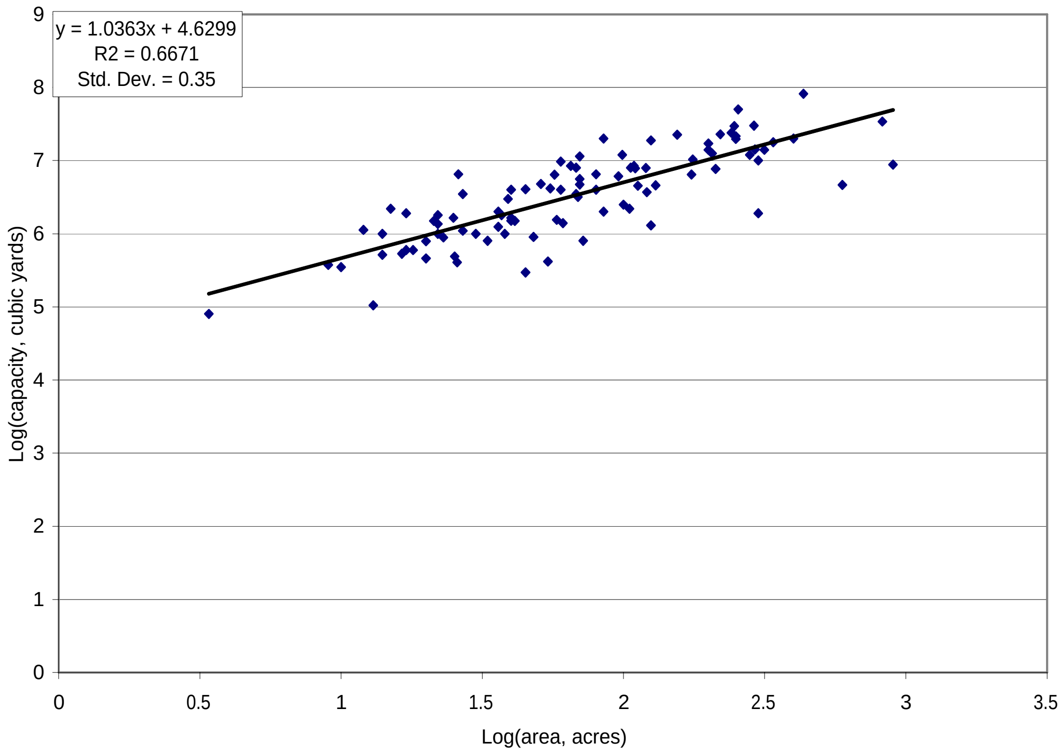

Figure B-5. Linear regression to impute landfill capacity from area.

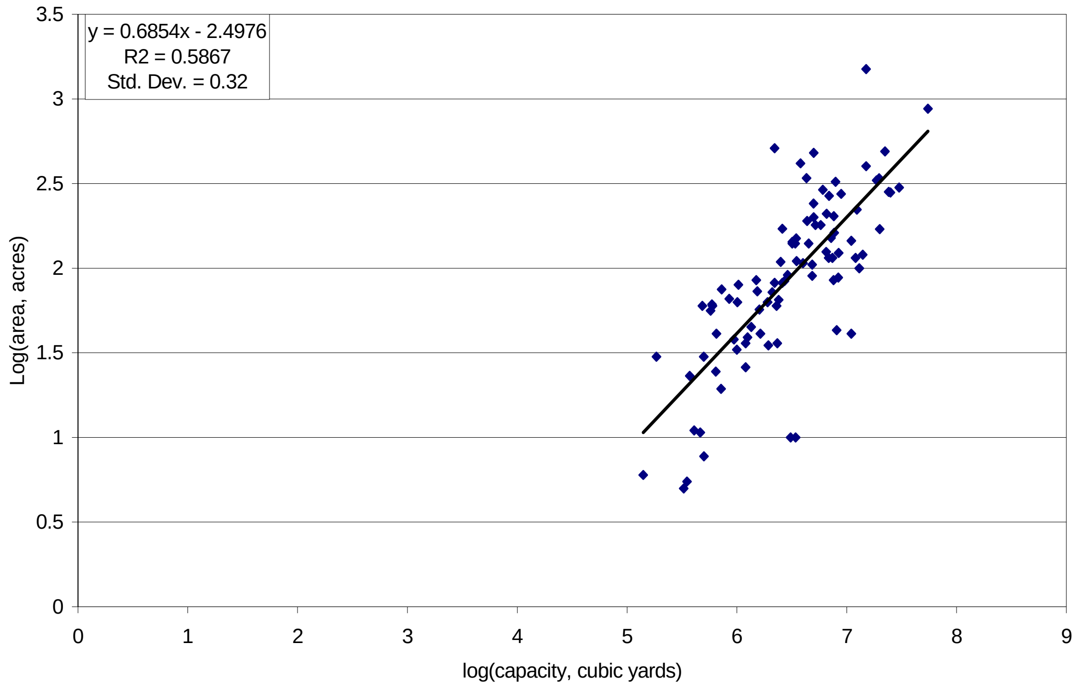

Figure B-6. Linear regression to impute surface impoundment area from capacity.

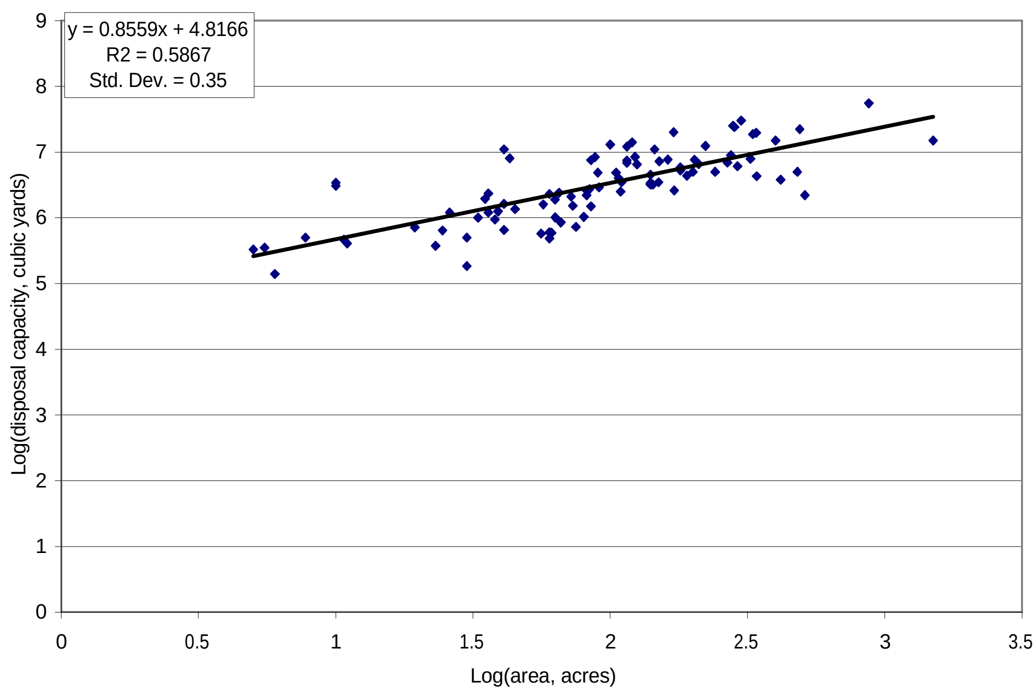

Figure B-7. Linear regression to impute surface impoundment capacity from area.

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.

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

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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

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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 are 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 a site-based data collection method. This method used the 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. Depending on the availability of information, data were collected on either a site-specific, regional, or national scale. Where appropriate, distributions were used in the Monte Carlo analysis to capture site-to-site and within-site variability in the parameters collected.

Site-based data were collected 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 meteorological 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 WMU and the waterbody at a random sample of the CCW sites. Similarly, a national distribution

[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.

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 is 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 a significant 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 conservative representation of actual well distances at CCW disposal sites. However, data on residential well distances from CCW landfills or surface impoundments will be needed to verify this hypothesis.

As discussed in Section 3.4.3, the groundwater model used in the CCW risk assessment places 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 remains within the plume and at a depth appropriate for surficial aquifers across the United States. These limits are consistent with other recent national risk assessments conducted by EPA OSW and provide 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

Memphis, Tennessee, United States 01 Jul 1993 USGS

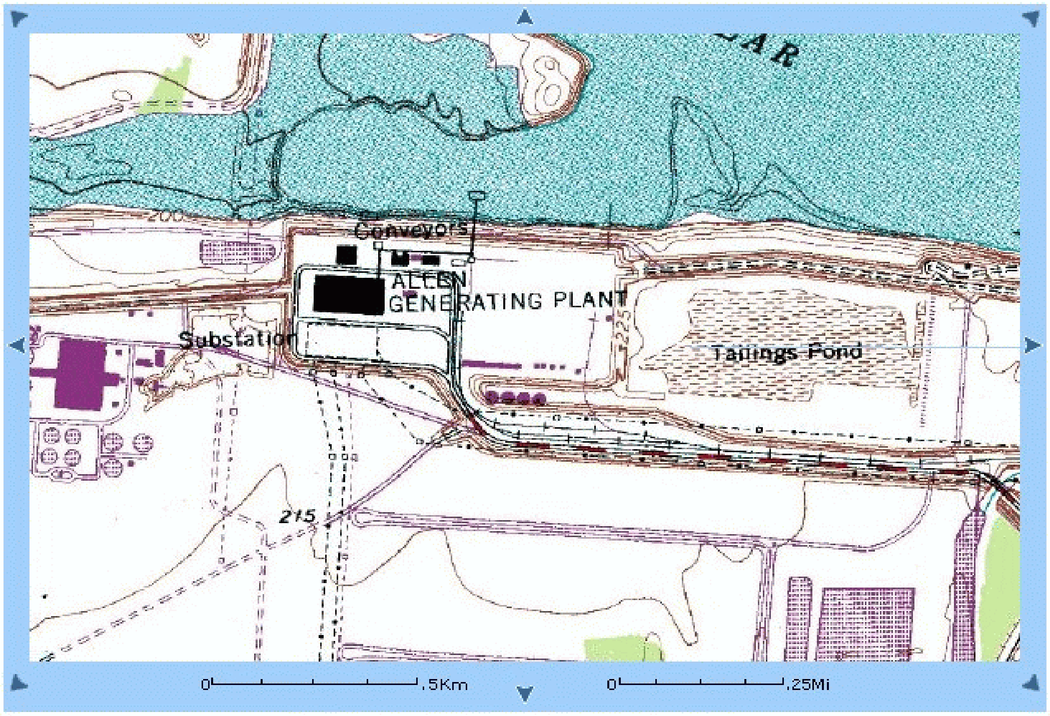

Image courtesy of the U.S. Geological Survey

Memphis, Tennessee, United States 10 Apr 1996 USGS

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 begins with GIS programs (in Arc Macro Language [AML]). These programs overlay 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 are 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 are 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 relies 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.

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[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 utilizes 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 will 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 include:

- 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 this 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 would be 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)

Table C-4. (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- (Hydrologic Evaluation of Landfill Performance-) 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 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/prismmt.html) 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

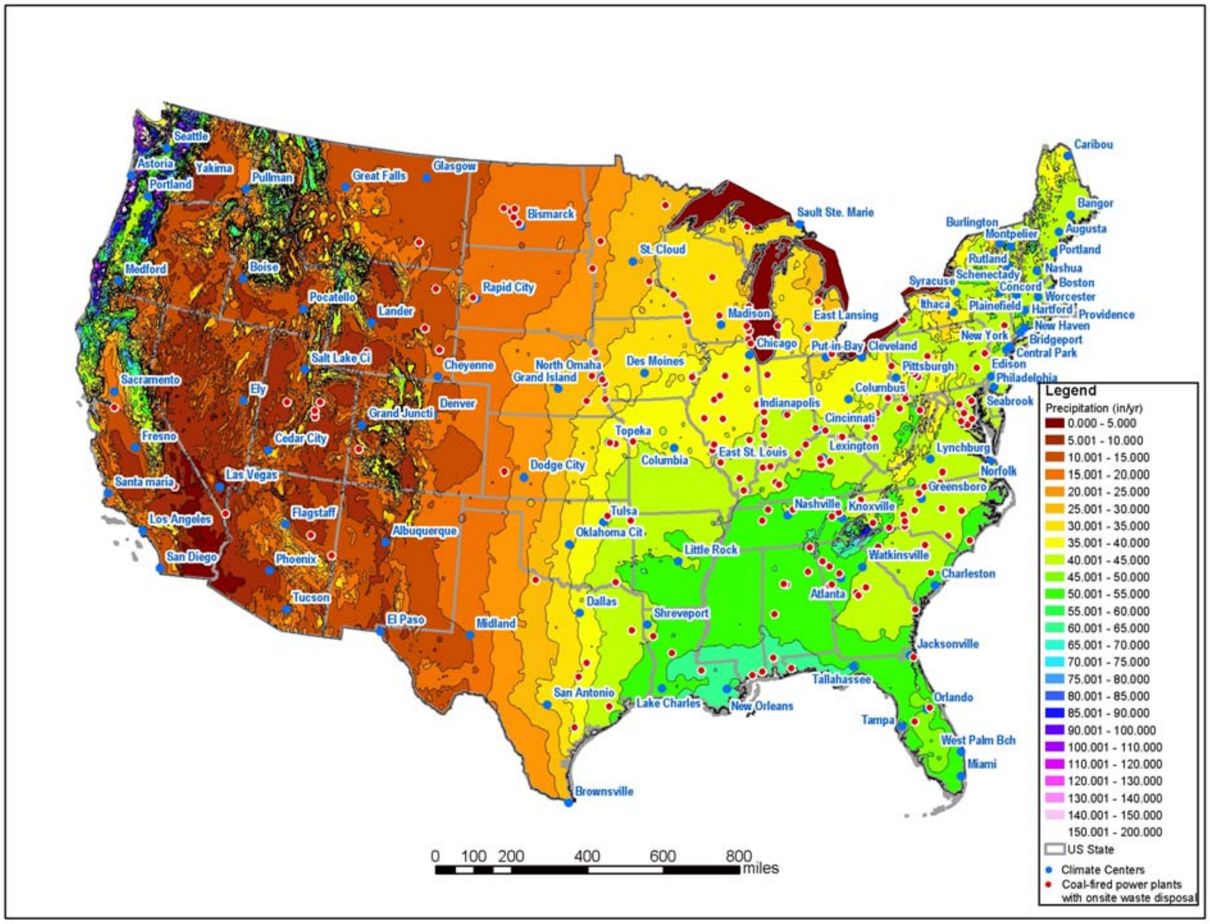

Figure C-2. EPACMTP climate centers, precipitation ranges, and CCW disposal sites.

- 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)

Table C-6. (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 are 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., 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

[5] The minimum 7-day average flow expected to occur within a 10-year return period (i.e., at least once in 10 years).

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://terraserver.usa.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 a considerable uncertainty in the CCW risk assessment and risk results for these waterbodies should be regarded as preliminary until a more sophisticated surface water model can be parameterized for these special cases. 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)

Table C-7. (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):

$$Width \;=\; 5.1867Q^{0.4559} \tag{C-1}$$

Water column depth (dwc) was derived from width, velocity (V), and flow using the continuity equation:

$$dwc \;=\; \frac{Q}{v \times Width} \tag{C-2}$$

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 are 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 are much larger than 13 acres, this will produce conservative risk results. However, given that many of the plants are located on very large waterbodies, this necessary simplification is one of the largest uncertainties in defining the environmental settings for the CCW risk assessment. Options can be developed to more accurately parameterize and model such large nonflowing waterbodies.

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/OWOW/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)

Table C-10. (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)

Table C-11. (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 Appendix C, 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://terraserver.usa.com/geographic.aspx) 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: http://water.usgs.gov/pubs/ofr/ofr99-77/index.html.

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. U.S. Geological Survey, Rapid City, SD.
Data Retrieval: http://water.usgs.gov/lookup/getspatial?ofr99-77_gwreguw
Metadata: http://water.usgs.gov/GIS/metadata/usgswrd/ofr99-77_gwreguw.html

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. U.S. Geological Survey, Rapid City, SD.

Data Retrieval: http://water.usgs.gov/lookup/getspatial?ofr99-77_aquif75m
Metadata: http://water.usgs.gov/GIS/metadata/usgswrd/ofr99-77_aquif75m.html

Clawges, R.M., and C.V. Price. 1999d. *Surficial geology of the conterminous United States. Version 1.0*. In Open-File Report 99-77. U.S. Geological Survey, Rapid City, SD. Data Retrieval: http://water.usgs.gov/lookup/getspatial?ofr99-77_geol75m
Metadata: http://water.usgs.gov/GIS/metadata/usgswrd/ofr99-77_geol75m.html

Heath, R.C. 1984. *Ground-Water Regions of the United States*. Water-Supply Paper 2242, U.S. Geological Survey, Washington, DC.

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. http://www.epa.gov/oscpmont/sap/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. Metadata: http://www.atlas.usgs.gov/aquifersm.html#Identification_Information

Miller, D.A., and R.A. White. 1998. *A Conterminous United States Multi-Layer Soil Characteristics Data Set for Regional Climate and Hydrology Modeling*. Available: http://www.essc.psu.edu/soil_info/index.cgi?soil_data&index.html

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: http://water.usgs.gov/GIS/metadata/usgswrd/ussoils.html

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: http://www.ftw.nrcs.usda.gov/stat_data.html

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.

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.

van der Leeden, F., F.L. Troise, and D.K. Todd. 1990. *The Water Encyclopedia.* 2nd Ed. Chelsea, MI: Lewis Publishers. 176 pp.

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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

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Hydrogeologic Environment (continued)

Plant	Hydrogeologic Setting Code	Hydrogeologic Setting Description	Hydrogeologic Environment Code	Hydrogeologic Environment Description	Percentage	Comment
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

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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

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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

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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

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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

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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)

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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

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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%)

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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

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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

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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)

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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

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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

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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

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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

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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	

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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	

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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	

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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	

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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	

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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	

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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	

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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

(continued)

Waterbody Assignments and Flow (continued)

Plant	CUSEG	Nearest Reach	Reach_Type	QLOW	QMEAN
Heskett	10130101001	MISSOURI R	Regular Reach	3461.55005	22744.26953
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

(continued)

Waterbody Assignments and Flow (continued)

Plant	CUSEG	Nearest Reach	Reach_Type	QLOW	QMEAN
Lee	03020201007	NEUSE R	Regular Reach	76.18	1657.39001
Leland Olds	10130101020	MISSOURI R	Regular Reach	4270.4502	21650.67969
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

(continued)

Waterbody Assignments and Flow (continued)

Plant	CUSEG	Nearest Reach	Reach_Type	QLOW	QMEAN
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
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

(continued)

Waterbody Assignments and Flow (continued)

Plant	CUSEG	Nearest Reach	Reach_Type	QLOW	QMEAN
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
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

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. The U.S. Environmental Protection Agency'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. R, a function of the constituent-specific K_d and the soil or aquifer properties, is calculated as:

$$R = 1 + \frac{\rho_b \times K_d}{\Phi} \tag{D-1}$$

where

R	=	Retardation factor
ρ_b	=	Soil or aquifer bulk density (mg)
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 is 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 coal combustion waste (CCW) sites across the country is 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 is repeated (separately for each metal) for numerous combinations of master variable settings. This procedure results 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 is 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 is executed and the national probability distributions for these five master variables form 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 are not applicable (e.g., organic carbon assumptions should not be 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 allows for full competition with the contaminant metal for sorption sites. The metal solubilizing effect through complexation between the contaminant metal and dissolved ligands is also included, as is 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 can impact vadose zone and groundwater pH. To account for this possibility, the CCW leachate/ groundwater system was equilibrated at a series of pH values that span 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 discuss 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 (vadose 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.5 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.

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.

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.

U.S. EPA (Environmental Protection Agency). 2003b. *EPACMTP Technical Background Document*. Office of Solid Waste, Washington, DC.

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 Appendix C, 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}	Sedment-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}	Sedment-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. Water Concentration Dissipation Rate Constant (1/d)

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_b	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-1-5. Total Waterbody Concentration from Loading (g/m^3 or mg/L)

Cw_{Tot}

$$V = Area_{WB} \times d_z$$

$$Cw_{Tot} = \frac{L_{total}}{V_{fx} \times f_{Water} \times \frac{d_z}{d_w} + K_{wt} \times V}$$

Name	Description	Value
$Area_{WB}$	Area of the waterbody (m^2)	Site Data; See Appendix C
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
K_{wt}	Water Concentration Dissipation Rate Constant (1/day)	Calculated
L_{Total}	Total waterbody load (g/day)	Calculated By EPACMTP
V	Flow independent mixing volume (m^3)	Calculated
V_{fx}	Waterbody annual flow mixing volume (m3/day)	Site Data; See Appendix C

Table E-1-6. 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-1-7. 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-1-8. 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 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 the U. S. Environmental Protection Agency (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 is 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 value was used).

The EFH does not use standardized age cohorts across exposure factors. Different exposure factors have data 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 fit into a single cohort, the EFH percentiles were averaged within each cohort (e.g., data on 1- to 2-year-olds and 3- to 5-year-olds were averaged for the 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 (1997b); 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.

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 include 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 is 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 are more consistent with our 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 age group was used for the 1- to 5-year-olds. The 6- and 9-year-old age groups were pooled for the 6- to 11-year-old 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 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 coal combustion waste (CCW) risk assessment will require human health benchmarks to assess potential risks from chronic oral and inhalation exposures. The U.S. Environmental Protection Agency (EPA) uses reference doses (RfDs) and reference concentrations (RfCs) to evaluate noncancer risk from oral and inhalation exposures, respectively. Oral cancer slope factors (CSFs), inhalation unit risk factors (URFs), and inhalation 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, 2002). Each chemical file contains descriptive and quantitative information on potential health effects. Health benchmarks for chronic noncarcinogenic health effects include RfDs and RfCs. Cancer classification, oral CSFs, and inhalation URFs are included for carcinogenic effects. IRIS is the official repository of Agency-wide consensus of human health risk information.

Inhalation CSFs are not available from IRIS, so we calculated them from inhalation URFs (which are available from IRIS) using the following equation:

$$inh\ CSF = inh\ URF \times 70\,kg \div 20\,m^3/d \times 1000\,\mu g/mg$$

In this equation, 70 kg represents average body weight; 20 m^3/d represents average inhalation rate; and 1000 µg/mg is a units conversion factor (U.S. EPA, 1997). EPA uses these standard estimates of body weight and inhalation rate in the calculation of the URF; therefore, we used these values to calculate inhalation CSFs.

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. These provisional health benchmarks can be found in Risk Assessment Issue Papers. 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, 2002). 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). CalEPA used EPA's (U.S. EPA, 1994) inhalation dosimetry methodology 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 risk assessment are summarized in Table G-1, which provides the Chemical Abstract Service Registry Number (CASRN), constituent name, RfD (mg/kg-d), RfC (mg/m^3), oral CSF (mg/kg-d^{-1}), inhalation URF [(µg/m^3)$^{-1}$], inhalation 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, 2002), Superfund Provisional Benchmarks, or HEAST (U.S. EPA, 1997). Benchmarks also were obtained from ATSDR (2002) or CalEPA (1999a, 1999b, 2000). This section describes benchmarks obtained from other sources, along with the Superfund Provisional Benchmarks values and special uses of IRIS benchmarks.

Provisional inhalation health benchmarks were developed in the Air Characteristic Study (U.S. EPA, 1999) for several constituents lacking IRIS, HEAST, alternative EPA, or ATSDR values. For vanadium, the study on which the ATSDR acute inhalation MRL is based was used but was adjusted for chronic exposure. Additional details on the derivation of this inhalation benchmark can be found in the *Revised Risk Assessment for the Air Characteristic Study* (U.S. EPA, 1999).

The provisional RfD of 0.02 mg/kg-d developed by NCEA for the Superfund Technical Support Center (U.S. EPA, 2001a) was used for cobalt.

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

Constituent Name	CASRN	RfD (mg/kg-d)	Ref	RfC (mg/m^3)	Ref	CSFo (per mg/kg-d)	Ref	URF (per µg/m3)	Ref	CSFi (per mg/kg-d)	Ref	MCL (mg/L)	Notes
Aluminum	7429-90-5	2.0E+00	A										RfD is for intermediate duration
Ammonia	7664-41-7	9.7E-01	H	1.0E-01	I								RfD= 34 mg/L
Antimony	7440-36-0	4.0E-04	I	2.0E-04	I								RfC is for antimony trioxide
Arsenic, inorganic	7440-38-2	3.0E-04	I	3.0E-05	Cal00	1.5E+0	I	4.3E-3	I	1.5E+1	calc		
Barium	7440-39-3	7.0E-02	I	5.0E-04	H								
Beryllium	7440-41-7	2.0E-03	I	2.0E-05	I			2.4E-3	I	8.4E+0	calc		
Boron	7440-42-8	9.0E-02	I	2.0E-02	H								
Cadmium	7440-43-9	5.0E-04	I	2.0E-05	Cal00			1.8E-3	I	6.3E+0	calc		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	1.0E-04	I			1.2E-2	I	4.2E+1	calc		
Cobalt (and compounds)	7440-48-4	2.0E-02	SF	1.0E-04	A			2.8E-3	SF	9.8E+0	calc		
Copper	7440-50-8											1.3	
Cyanide (amenable)	57-12-5	2.0E-02	I										
Divalent mercury		3.0E-04	H										RfD is for mercuric chloride; used for food, water, soil
Divalent mercury		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											0.3	
Lead and compounds (inorganic)	7439-92-1											0.015	
Manganese	7439-96-5	1.4E-01	I	5.0E-05	I								RfD for food; H_2O and soil = 4.7E-2 mkd
Molybdenum	7439-98-7	5.0E-03	I										

(continued)

Table G-1. (continued)

Constituent Name	CASRN	RfD (mg/kg-d)	Ref	RfC (mg/m^3)	Ref	CSFo (per mg/kg-d)	Ref	URF (per µg/m3)	Ref	CSFi (per mg/kg-d)	Ref	MCL (mg/L)	Notes
Nickel, soluble salts	7440-02-0	2.0E-02	I	2.0E-04	A								
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	2.0E-02	Cal00								
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
Total dissolved solids												500	
Vanadium	7440-62-2	7.0E-03	H	7.0E-05	AC								
Zinc	7440-66-6	3.0E-01	I										

Key: CASRN = Chemical Abstract Service registry number. CSFo = Oral cancer slope factor.
RfD = Reference dose. CSFi = Inhalation cancer slope factor.
RfC = Reference concentration. URF = Unit risk factor.
MCL = Maximum contaminant level.

[a] Sources:
A = ATSDR MRLs (ATSDR, 2002)
AC = Developed for the Air Characteristic Study (U.S. EPA, 1999)
calc = Calculated
Cal00 = CalEPA chronic REL (CalEPA, 2000)
H = HEAST (U.S. EPA, 1997)
I = IRIS (U.S. EPA, 2002)
SF = Superfund Risk Issue Paper (U.S. EPA, 2001a,b)

For several constituents, IRIS benchmarks for similar chemicals were used as surrogate data. The rationale for these recommendations is as follows:

- The RfC for antimony trioxide (2E-04 mg/m^3) was used as a surrogate for antimony.
- 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.
- The RfC for mercuric chloride (9E-05 mg/m^3) was used as a surrogate for elemental mercury. The RfDs for mercuric chloride (3E-04 mg/kg-d) and methyl mercury (1E-04 mg/kg-d) were used as surrogates for elemental mercury for assessing potential risks from food, soil, and water ingestion, and fish ingestion, respectively.
- Thallium was based on thallium chloride. There are several thallium salts that have RfDs in IRIS. 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). 2002. *Minimal Risk Levels (MRLs) for Hazardous Substances.* Available: 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: 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):
http://www.oehha.org/air/chronic_rels/ragsii.html,
http://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):
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

U.S. EPA (Environmental Protection Agency). 1994. *Methods for Derivation of Inhalation Reference Concentrations and Application of Inhalation Dosimetry*. EPA/600/8-90-066F.

Environmental Criteria and Assessment Office, Office of Health and Environmental Assessment, Office of Research and Development, Research Triangle Park, NC.

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). 1999. *Revised Risk Assessment for the Air Characteristic Study*. EPA-530-R-99-019a. Volume 2. Office of Solid Waste, Washington, DC.

U.S. EPA (Environmental Protection Agency). 2001a. *Risk Assessment Paper for Derivation of a Provisional RfD for Cobalt and Compounds (CASRN 7440-48-4).* 00-122/3-16-01. National Center for Environmental Assessment. Superfund Technical Support Center, Cincinnati, OH.

U.S. EPA (Environmental Protection Agency). 2001b. *Risk Assessment Paper for Derivation of a Provisional Carcinogenicity Assessment for Cobalt and Compounds (CASRN 7740-48-4).* 00-122/3-16-01. National Center for Environmental Assessment. Superfund Technical Support Center, Cincinnati, OH.

U.S. EPA (Environmental Protection Agency). 2002. Integrated Risk Information System (IRIS). National Center for Environmental Assessment, Office of Research and Development, Washington, DC. Available: http://www.epa.gov/iris/

Appendix H. Ecological Benchmarks

Both the screening and full-scale CCW assessments include an ecological risk assessment that parallels the human health risk assessment. The ecological risk assessment addresses 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 terrestrial (soil) and 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	Soil
Direct Contact Exposure			
Aquatic Community	✔		
Sediment Community		✔	
Soil Community			✔
Amphibians	✔		
Aquatic Plants and Algae	✔		
Terrestrial Plants			✔
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 (soil invertebrates, terrestrial plants, mammals, and birds) are exposed to constituents in soils. The soil HQ for a given chemical was calculated using whichever soil benchmark was lowest and would thus give the highest (most conservative) 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, sediment, and soil. The receptors selected to assess the direct contact exposure route for each medium were previously 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 is not consistent with other benchmarks, the sensitivity of these receptors warrants their use in lieu 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 are 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.

Soil Community Benchmarks

For the soil community, the preferred methods for deriving benchmarks are analogous to those used in deriving the NAWQC. Benchmark values for soil fauna were estimated to protect 95 percent of the species found in a typical soil community, including earthworms, insects, and various other soil fauna. The methodology presumes that protecting 95 percent of the soil species with a 50th percentile level of confidence will ensure long-term sustainability of a functioning soil community. The toxicity data on soil fauna were taken from several major compendia and supplemented with additional studies identified in the open literature.

The approach to calculating benchmarks for the soil community is based on efforts by Dutch scientists (i.e., the Netherlands' National Institute of Public Health and Environmental Protection [RIVM] methodology) to develop hazardous concentrations (HCs) at specified levels of protection (primarily 95 percent) at both a 95th percentile and a 50th percentile level of confidence (Sloof, 1992). For the soil fauna benchmarks, the 50th percentile level of confidence was selected because the 95th percentile appeared to be overly conservative for a "no effects" approach. The RIVM methodology follows two steps: (1) fitting a distribution to the log of the selected endpoints, and (2) extrapolating to a benchmark concentration based on the mean and standard deviation of a set of endpoints. The key assumptions in the Dutch methodology are that (1) lowest observed effects concentration (LOEC) data are distributed logistically, and (2) the 95 percent level of protection is ecologically significant. The following formula was used to calculate soil fauna benchmarks:

$$HC_{5\%} = \left[x_m - k_1 s_m\right] \tag{H-1}$$

where

- $HC_{5\%}$ = soil concentration protecting 95 percent of the soil species
- x_m = sample mean of the log LOEC data
- k_l = extrapolation constant for calculating the one-sided leftmost confidence limit for a 95 percent protection level
- s_m = sample standard deviation of the log LOEC data.

Sufficient data were available to develop benchmarks using this methodology for four of the metals of concern: cadmium, copper, lead, and zinc. For the remaining constituents, benchmark studies identifying effects to earthworms and other soil biota proposed by Oak Ridge National Laboratory (Efroymson et al., 1997a) or criteria developed by the Canadian Council of Ministers of the Environment (CCME, 1997) were used to estimate protective soil concentrations.

Algae and Aquatic Plant Benchmarks

For algae and aquatic plants, adverse effects concentrations are 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.

Terrestrial Plant Benchmarks

For the terrestrial plant community, ecotoxicological data were identified from a summary document prepared at the Oak Ridge National Laboratory: *Toxicological Benchmarks for Screening Contaminants of Potential Concern for Effects on Terrestrial Plants: 1997 Revision* (Efroymson et al., 1997b). The measurement endpoints are generally limited to growth and yield parameters because (1) they are the most common class of response reported in phytotoxicity studies and, therefore, will allow for criterion calculations for a large number of constituents, and (2) they are ecologically significant responses both in terms of plant populations and, by extension, the ability of producers to support higher trophic levels. As presented in Efroymson et al. (1997a), benchmarks for phytotoxicity were selected by rank ordering the LOEC values and then approximating the 10th percentile. If there were 10 or fewer values for a chemical, the lowest LOEC was used. If there were more than 10 values, the 10th percentile LOEC was used.

H.1.2 Ingestion Exposure

The ingestion route of exposure addresses the exposure of terrestrial mammals and birds through ingestion of plants and prey and incidental soil ingestion. Thus, the CCW ecological benchmarks for ingestion exposure express 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.

The derivation of ingestion benchmarks begins with the selection of appropriate ecotoxicological data based on a hierarchy of data sources. The assessment endpoint for the CCW ecological risk assessment is 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 are preferred over benchmarks for individual organisms; however, very few population-level benchmarks have been developed. Therefore, the CCW risk assessment uses benchmarks derived from individual organism studies, and protection is 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 soil contaminated by CCW waste constituents. The methods reflect exposure through the ingestion of contaminated plants, prey, and various 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-2) (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).

$$Benchmark_w = LOAEL_t \times \left(\frac{bw_t}{bw_w}\right)^{1/4} \qquad \text{(H-2)}$$

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 and terrestrial organisms that may be used as food sources (e.g., fish, plants, earthworms). The Oak Ridge National Laboratory has proposed methods and data that are useful in predicting bioaccumulation in earthworms and small mammals (Sample et al. 1998a,b). These values were typically identified in the open literature and EPA references.

Step 3: Calculate Benchmarks

The following equation provides the basis for calculating surface water benchmarks using a population-inference benchmark (e.g., endpoint on fecundity).

$$Benchmark = \frac{\left[I_{fish} \times \left(BAF \;\; C_w\right)\right] + \left(I_w \times C_w\right)}{bw} \tag{H-3}$$

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 will tend 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-3, resulting in Equation H-4:

$$Benchmark \;=\; \frac{I_{fish} \;\times\; \left(BAF \;\times\; C_w\right)}{bw} \tag{H-4}$$

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-4 can be rewritten to solve for the surface water ($CSCL_{sw}$) as follows:

$$CSCL_{sw} \;=\; \frac{benchmark \;\times\; bw}{I_w \;+\; \left(I_{fish} \;\times\; BAF\right)} \tag{H-5}$$

For wildlife populations of mammals and birds in terrestrial systems, the soil benchmark ($CSCL_{soil}$) for a given receptor was calculated using Equation H-6:

$$CSCL_{soil} = \frac{benchmark \times bw}{I_{food} \sum \left(BCF_j \times F_j \times AB_j\right) + I_{soil}} \tag{H-6}$$

where

bw	=	body weight (kg)
I_{food}	=	total daily food intake of species (kg/d)
I_{soil}	=	total daily soil intake of species (kg/d)
BCF_j	=	bioaccumulation factor in food item *j* (assumed unitless)
F_j	=	fraction of diet consisting of food item *j* (unitless)
AB_j	=	absorption of chemical in the gut from food item *j*.

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 soil, sediment, and aquatic receptors; and the source for each benchmark.

Table H-3. Ecological Benchmarks Used in the CCW Risk Assessment

Constituent	Soil Criterion (mg/kg)	Terrestrial Receptor	Sediment Criterion (mg/kg)	Sediment Receptor	Aquatic Criterion (mg/L)	Aquatic Receptor	Source
Aluminum	ID	--	ID	--	0.09	Aquatic Biota	U.S. EPA (1998)
Antimony	14	Raccoon	2	Sediment biota	0.03	Aquatic Biota	U.S. EPA (1998)
Arsenic total	10	Plants	0.51	Spotted sandpiper	ID	--	U.S. EPA (1998)
Arsenic III	ID	--	ID	--	0.15	Aquatic Biota	U.S. EPA (1998)
Arsenic IV	ID	--	ID	--	8.10E-03	Aquatic Biota	U.S. EPA (1998)
Barium	500	Plants	190	Spotted sandpiper	4.00E-03	Aquatic Biota	U.S. EPA (1998)
Beryllium	ID	--	ID	--	6.60E-04	Aquatic Biota	U.S. EPA (1998)
Boron	0.5	Plants	ID	--	1.60E-03	Aquatic Biota	U.S. EPA (1998)
Cadmium	1	Soil invertebrates	0.68	Sediment biota	2.50E-03	Aquatic Biota	U.S. EPA (1998)
Chromium total	64	Soil invertebrates	16.63	Spotted sandpiper	ID	--	U.S. EPA (1998)
Chromium IV	ID	--	ID	--	0.09	Aquatic Biota	U.S. EPA (1998)
Chromium VI	ID	--	ID	--	0.01	Aquatic Biota	U.S. EPA (1998)
Cobalt	1000	Soil invertebrates	ID	--	0.02	Aquatic Biota	U.S. EPA (1998)
Copper	21	Soil invertebrates	18.7	Sediment biota	9.30E-03	Aquatic Biota	U.S. EPA (1998)
Lead	28	Soil invertebrates	0.22	Spotted sandpiper	3.00E-04	River Otter	U.S. EPA (1998)
Mercury	0.1	Soil invertebrates	0.11	Spotted sandpiper	1.90E-07	Kingfisher	U.S. EPA (1998)
Molybdenum	42.08	Amer. woodcock	34	Spotted sandpiper	0.37	Aquatic Biota	U.S. EPA (1998)
Nickel	30	Plants	15.9	Sediment biota	0.05	Aquatic Biota	U.S. EPA (1998)
Selenium total	1	Plants	ID	--	5.00E-03	Aquatic Biota	U.S. EPA (1998)
Selenium IV	ID	--	ID	--	0.03	Aquatic Biota	U.S. EPA (1998)

(continued)

Table H-3. (continued)

Constituent	Soil Criterion (mg/kg)	Terrestrial Receptor	Sediment Criterion (mg/kg)	Sediment Receptor	Aquatic Criterion (mg/L)	Aquatic Receptor	Source
Selenium VI	ID	--	ID	--	9.50E-03	Aquatic Biota	U.S. EPA (1998)
Silver	ID	--	0.73	Sediment biota	3.60E-04	Aquatic Biota	U.S. EPA (1998)
Thallium	ID	--	ID	--	0.01	Aquatic Biota	U.S. EPA (1998)
Vanadium	130.00	Soil invertebrates	18	Spotted sandpiper	0.02	Aquatic Biota	U.S. EPA (1998)
Zinc	50	Plants	120	Sediment biota	0.12	Aquatic Biota	U.S. EPA (1998)

ID = insufficient data.

H.3 References

CCME (Canadian Council of Ministers of the Environment). 1997. *Recommended Canadian Soil Quality Guidelines*. Science Policy and Environmental Quality Branch, Ecosystem Science Directorate, Environment Canada, Ottawa, Ontario. ISBN 1-895-925-92-4.

Efroymson, R.A., M.E. Will, and G.W. Suter. 1997a. *Toxicological Benchmarks for Contaminants of Potential Concern for Effects on Soil and Litter Invertebrates and Heterotrophic Process: 1997 Revision*. ES/ER/TM-126/R2. Oak Ridge National Laboratory, Oak Ridge, TN.

Efroymson, R.A., M.E. Will, G.W. Suter, and A.C. Wooten. 1997b. *Toxicological Benchmarks for Screening Contaminants of Potential Concern for Effects on Terrestrial Plants: 1997 Revision.* ES/ER/TM-85/R3. Oak Ridge National Laboratory, Oak Ridge, TN.

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, G.W. Suter, II, and T.L. Ashwood. 1998a. *Development and Validation of Bioaccumulation Models for Earthworms*. ES/ER/TM-220. 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.