RISK ASSESSMENT OF COAL COMBUSTION RESIDUALS:

LEGACY IMPOUNDMENTS AND CCR MANAGEMENT UNITS

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Draft

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Acronyms and Abbreviations

ACAA	American Coal Ash Association
ACH	Air Changes Per Hour
ACI	Active Carbon Injection
ANL	Argonne National Laboratory
ASTM	American Association of Techniques and Methods
BEIR	Biological Effects of Ionizing Radiation
CCR	Coal Combustion Residuals
CCRMUS	Coal Combustion Residual Management Units
COPC	Constituents of Potential Concern
EIA	Energy Information Association
EPACMTP	EPA Composite Model for Leachate Migration with Transformation Products
ESP	Electrostatic Precipitator
FGD	Flue Gas Desulfurization
GWPS	Groundwater Protection Standards
HELP	Hydrologic Evaluation of Landfill Performance
HQs	Hazard Quotients
HSWA	Hazardous and Solid Waste Amendments of 1984
LANL	Los Alamos National Laboratory
LEAF	Leaching Evaluation Assessment Framework
MODFLOW	Modular Three-Dimension Finite-Difference Ground-Water Flow Model
NAICS	North American Industry Classification System
NOAA	National Oceanic and Atmospheric Administration
NRC	National Research Council
OLEM	Office of Land and Emergency Management
OPP	Office of Pesticide Programs
ORNL	Oak Ridge National Laboratory
PNL	Pacific Northwest Laboratory
RCRA	Resource Conservation and Recovery Act
RME	Reasonable Maximally Exposed
RSL	Regional Screening Level
TENORM	Technologically Enhanced Naturally Occurring Radioactive Material
USCHEM	United States geoCHEMical Database
USEPA	United States Environmental Protection Agency
USGS	United States Geologic Survey
USWAG	Utility Solid Waste Activities Group
WIIN	Water Infrastructure Improvements for the Nation

1 Introduction

The United States (U.S.) Environmental Protection Agency (EPA or the Agency) has taken many steps toward characterizing the risks that may result from disposal of coal combustion residuals (CCR) and developing regulations necessary to protect human health and the environment. This characterization of risk is conducted is support of the Resource Conservation and Recovery Act of 1976 (RCRA), as amended by the Hazardous and Solid Waste Amendments (HSWA) of 1984 and Water Infrastructure Improvements for the Nation (WIIN) Act of 2016. A full timeline and summary of regulatory actions related to CCR disposal can be found on the Agency website.[1]

1.1 Regulatory Background

In 2015, EPA finalized national regulations for management of CCR generated at coal-fired electric utilities.[2] This rule established minimum national standards under Subtitle D of RCRA for the design, operation, and closure of landfills and surface impoundments that accept CCR after the effective date of the rule on October 19, 2015. These requirements were designed to address the potential risks EPA identified through environmental modeling documented in "Human and Ecological Risk Assessment of Coal Combustion Residuals" ("2014 Risk Assessment") (U.S. EPA, 2014a) and through a review of relevant damage cases.

The 2015 Rule was challenged by multiple parties, including a coalition of environmental advocacy groups. Among the issues raised by these petitioners was their contention that the scope of the rule violated the RCRA statute. Specifically, they argued that exclusion of inactive surface impoundments at inactive facilities from the regulation (''legacy impoundments'') could result in unmonitored leaks to groundwater and catastrophic structural failures, which violated a baseline requirement of RCRA that promulgated criteria for solid waste disposal pose ''no reasonable probability of adverse effects on health or the environment.'' 42 U.S.C. 6944(a). On August 21, 2018, the U.S. Court of Appeals for the D.C. Circuit issued its opinion in the case of *Utility Solid Waste Activities Group v. EPA*, 901 F.3d 414 (per curiam) (hereafter "*USWAG* decision"). This decision upheld the 2015 CCR Rule on most counts but agreed with the environmental petitioners on the issue of legacy impoundments, holding that EPA acted "arbitrarily and capriciously and contrary to RCRA." As a result, the Court vacated the exemption for legacy impoundments and remanded the issue back to EPA.

In 2023, EPA proposed to revise the CCR Rule in response to the *USWAG* decision and to address additional issues that have arisen since that decision as a result of mandated facility reporting.[3] The Agency first proposed a set of requirements for management of CCR in legacy impoundments that would apply to the inactive facilities where these impoundments are located ("legacy facilities"), which include the same requirements as active units with the exception of certain design requirements and location restrictions. These requirements build off the existing risk record and respond to the *USWAG* decision. The Agency also proposed a separate set of requirements for management practices that result

1) Available online at: https://www.epa.gov/coalash/coal-ash-rule

2) 80 FR 21302, April 17, 2015.

3) 88 FR 31982, May 18, 2023.

in placement of CCR on the land outside regulated disposal units, referred to as CCR management units ("CCRMUs"), which would apply to both active facilities and legacy facilities. This would extend a subset of the requirements for CCR units to CCRMU, including groundwater monitoring, corrective action, closure, post-closure care, and reporting and recordkeeping. These requirements respond to 42 alternate source demonstrations or assessment of corrective measure documents that attribute identified groundwater contamination to these units.

1.2 Purpose and Scope of the Risk Assessment

The 2014 Risk Assessment previously considered the potential risks from disposal of CCR in landfills and surface impoundments that operate onsite at electric utilities (U.S. EPA, 2014a). This assessment utilized site-specific data, where available, supplemented by regional and national data sets, to best reflect the variability of disposal practices, environmental conditions, and receptor behavior across the country. This assessment considered a range of exposure pathways that were modeled in a stepwise fashion, culminating in national-scale, probabilistic modeling. Based on the results of this probabilistic analysis, EPA identified potential for long-term risks from leakage to groundwater and that warranted regulatory action.

The purpose of the current risk assessment is to evaluate the potential for risk from placement of CCR in legacy impoundments and CCRMUs, which fell outside the scope of the 2014 Assessment. Because the 2014 Risk Assessment previously identified a subset of contaminants most likely to drive risk from leakage to groundwater, the current assessment of groundwater focuses on that list of contaminants. EPA started from the same methodology and data sources detailed in the 2014 Risk Assessment for selecting appropriate data and characterizing facility environmental setting, CCR waste properties, contaminant transport, and receptor exposure. EPA found the same methodology and data sources were sufficient to support conclusions about the risks from the landfills and surface impoundments covered in this rulemaking. EPA adjusted the methodology as necessary to better reflect an updated conceptual model for smaller CCRMUs placed for purposes other than disposal and to incorporate more recent data. Finally, EPA considered the potential for additional, non-groundwater exposures specific to these smaller CCRMUs.

The regulatory scope of the current rulemaking is limited to management of CCRs generated by coal-fired electric utilities and independent power producers covered by the North American Industry Classification System (NAICS) Code 221112.[4] The scope of this risk assessment is limited to the disposal or other placement of CCR on the land at electric utilities.

1.3 Overview of Assessment Methodology

This risk assessment is divided into eight main sections and three appendices. The main text summarizes the different data sources relied upon, analyses performed, model results and final Agency conclusions. The appendices provide a more detailed discussion of the data and model results underlying the analyses summarized in the main text. The remainder of this subsection provides further information about the contents of each section and appendix.

4) See: https://www.bls.gov/oes/current/naics5_221112.htm

- **Section 2, Problem Formulation:** describes the conceptual models used to identify relevant exposure pathways, summarizes new data sources used to characterize these pathways, and presents the list of constituents to be evaluated.
- **Section 3, Disposal Unit Groundwater Risk:** describes the review of available data conducted to characterize how risks from historical and inactive disposal in landfills and surface impoundments compare with those previously reported in 2014.
- **Section 4, CCRMU Fill Groundwater Risk:** describes modeling approach used to 1) estimate the magnitude of leakage from smaller CCRMUs to groundwater, 2) model contaminant fate and transport through the underlying soil and aquifer, and 3) calculate the magnitude of resulting exposure and corresponding risk.
- **Section 5, CCRMU Fill Soil Risk:** describes the modeling approach used to 1) estimate the rate at which gamma radiation and radon gas are released from smaller CCRMUs placed within the soil, 2) model contaminant fate and transport through the overlying soil and indoor air, and 3) calculate the magnitude of resulting exposure and corresponding risk.
- **Section 6, Uncertainty and Sensitivity Analyses:** describes the results of various analyses conducted to identify new sources of uncertainty and sensitive parameters that exert the greatest influence on modeled risks. To the extent possible, these sources are quantitatively and qualitatively characterized to identify the potential for higher or lower risks than those previously modeled.
- **Section 7, Summary and Conclusions:** synthesizes available information from all sections of the risk assessment to reach final conclusions about the risks that may result from CCR management practices.
- **Section 8, References:** provides citations for all documents referenced throughout the text.

2 Problem Formulation

This primary purpose of this section is to describe the conceptual models developed for legacy impoundments and different types of CCRMUs that form the basis for this risk assessment. This section also provides a summary of major data sources that have been updated since the 2014 Risk Assessment was finalized. These data on facility conditions and environmental setting are applied to the conceptual models to characterize the potential risks associated with placement of CCR on the land.

2.1 Overview of Coal Combustion and Residuals

CCR is a broad term that refers to a range of byproducts generated directly by coal combustion or as a result of applying certain pollution control devices to emissions from coal-fired combustion units. CCRs may be generated wet or dry, but this can change after generation. Some CCRs are dewatered after generation, while others are later mixed with water to facilitate transport. When multiple types of CCRs are generated at the same facility, mixing and codisposal may occur.

- **Fly ash** is the fraction of combusted coal that becomes suspended in plant flue gases. It is a very fine, powdery material composed primarily of silica. Fly ash is removed from the plant exhaust gases primarily by electrostatic precipitators (ESPs) or baghouses that contain fabric filters. In facilities that use activated carbon injection (ACI) before fly ash collection, the fly ash waste stream will also contain the carbon, along with other mercury control wastes. However, where ACI occurs after fly ash collection, a separate waste stream may result.
- **Bottom ash** consists of ash particles that are too large to become entrained in the flue gas during combustion. It is coarse, with grain sizes that range from fine sand to fine gravel, and quite angular, with a porous surface structure. Bottom ash is collected from the furnace after it collides with and agglomerates to furnace walls or falls through open grates to an ash hopper beneath the furnace.
- **Boiler slag** is molten bottom ash that has been quenched with water. When the molten ash comes in contact with the water, it crystallizes, fractures and forms pellets that are hard with a smooth, glassy appearance. Boiler slag is collected from the base of either slag tap or cyclone type furnaces.
- **Flue Gas Desulfurization (FGD) materials** are produced through a process used to reduce sulfur dioxide (SO_2) emissions from the exhaust gas system of a coal-fired boiler. The physical nature of these materials varies from a wet sludge to a dry powdered material, depending on the pollution control technology, and the composition consists of sulfites, sulfates, or a mixture thereof.

Figure 2-1 provides the layout of a generic coal-fired plant. This simplified layout is intended to demonstrate some major pollution control technologies, waste streams, and points of generation associated with coal combustion. It is intended to be illustrative and so does not capture all possible technologies or plant layouts.

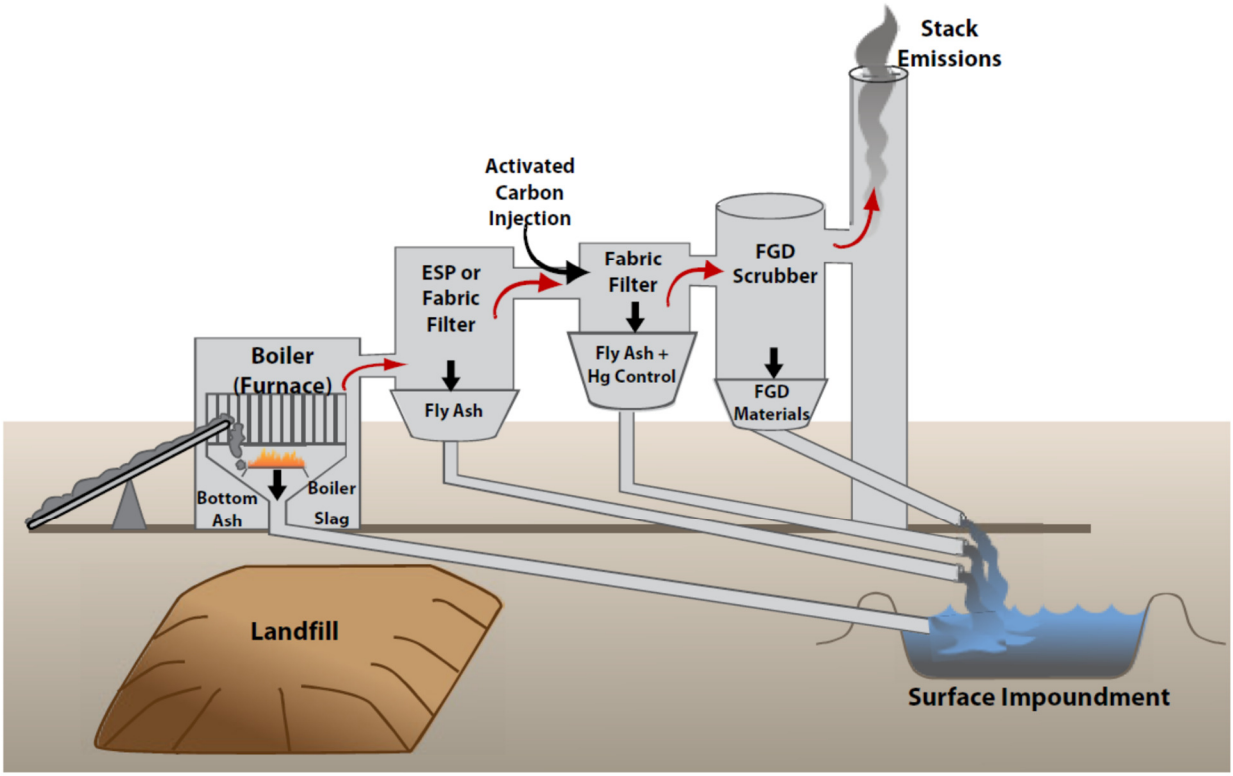

Figure 2-1. Generalized coal-fired power plant layout.

Since promulgation of the 2015 CCR Rule, the rates at which different CCR types are generated and the prevalence of various management practices may have shifted. However, the general descriptions provided here are still valid. For example, while there has been an increasing trend toward dry handling of CCR and landfill disposal, impoundments still operate across the country.

2.2 Conceptual Models

Once placed on the land, the chemical constituents present in CCRs may leach or otherwise be released into the surrounding environment. To evaluate the potential for risk associated with such releases in the absence of regulatory action, EPA developed conceptual models with the intent to broadly depict the relevant characteristics of different management practices considered in the proposed rule. Thus, these definitions and conceptual models do not reflect any specific facility or unit. Nor are they intended to mirror distinctions made in the regulatory text. Instead, they form the basis for identification of complete exposure pathways and subsequent data collection efforts for use in this risk assessment.

2.2.1 Historical and Inactive Landfills

Historical landfills are defined for purposes of this document as landfills that ceased receipt of CCR prior to the effective date of the 2015 Rule, have no intention of accepting further CCR, and have installed some form of cover. Closure of these landfills may or may not be consistent with the requirements of 40 CFR § 257.102(d). As a result, the current configuration of these units can range anywhere between an active or closed landfill.

Inactive landfills are defined for purposes of this document as landfills that ceased receipt of CCR prior to the effective date of the 2015 Rule, have the potential to accept CCR in the future, and have taken no formal steps toward closure. These units may be entirely open to the air or have some limited soil cover for purposes such as dust control. As a result, the current configuration of these units is expected to most closely resemble an active landfill.

EPA believes the national-scale risks from historical and inactive landfills are best characterized using the same conceptual model as the 2014 Risk Assessment (U.S. EPA, 2014a). EPA modeled only one stage of the landfill lifecycle at the time because the groundwater model required a static unit configuration. The primary difference among landfill lifecycle stages is the presence of a cover system over the CCR following closure, which can reduce infiltration to some degree. Yet a cover constructed exclusively with natural soil is still expected to be relatively permeable and allow for infiltration. Given the prevalence of unlined units, EPA previously modeled all active landfills as closed under the assumption this stage of the landfill lifecycle contributes the most to long-term risk as a result of the longer time period that releases can occur. This conceptual model accurately identified potential risks from active landfills and is expected be equally applicable to historical and inactive landfills, which have configurations ranging between active and closed landfills.

EPA previously established the following conceptual model for closed landfills. During closure, waste is left in place and the entire landfill is capped with a minimum of two feet of native soil. Landfills may contain one or more of the different CCR types, as well as other wastes such as coal refuse. For purposes of modeling, landfills are assumed to be constructed with a square footprint and located anywhere from entirely above grade to entirely below the ground surface. **Figure 2-2** depicts a conceptual model for one potential configuration of a closed landfill.

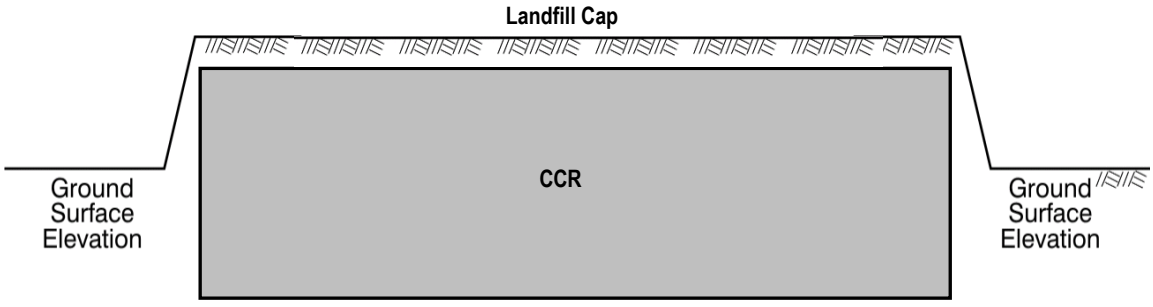

Figure 2-2. Cross-section view of closed landfill constructed above grade.

The 2014 Risk Assessment evaluated the potential risks to offsite receptors up to a mile away that result from disposal of CCR in landfills located at active facilities (U.S. EPA, 2014a). It considered multiple exposure pathways as part of a national-scale, probabilistic analysis, which included human ingestion of impacted groundwater and fish caught from impacted streams, as well as ecological exposure to impacted surface water and sediment. On a national scale, the evaluation found potential for risk to human health from impacted groundwater to occur within the range the EPA Office of Land and Emergency Management (OLEM) typically considers to warrant regulation.[5] In particular, unlined landfills that account for a majority of regulated units were found to result in cancer risks up to 2×10^{-5} for arsenic. Based on these results, groundwater exposure is considered the principal risk driver for regulated landfills. Given the similar design and siting of historical and inactive units, the same exposure pathway will be the focus of further analysis for these units.

2.2.2 Historical and Legacy Surface Impoundments

Historical surface impoundments are defined for purposes of this document as impoundments that ceased receipt of CCR prior to the effective date of the 2015 Rule, have no intention of accepting CCR in the future, and have moved to dewater the unit and install some form of cover system. Closure of these impoundments may or may not be consistent with the requirements of 40 CFR § 257.102(d). As a result, the current configuration of these impoundments is expected to range anywhere between an active surface impoundment and a closed landfill.

Legacy impoundments are defined for purposes of this document as impoundments located at inactive electric utilities, that ceased receipt of CCR prior to the effective date of the 2015 Rule, have the potential to accept CCR in the future, and still contain free water. These units might be open to the air or have some form of soil accumulation on top of the ash. As a result, the current configuration of these units is expected to most closely resemble an active surface impoundment.

EPA believes the national-scale risks from historical and legacy surface impoundments are best characterized using the same conceptual model as the 2014 Risk Assessment (U.S. EPA, 2014a). EPA modeled only one stage of the impoundment lifecycle at the time because the groundwater model required a static unit configuration The primary difference among impoundment lifecycle stages is the presence of free water in the unit before closure. The hydraulic head from this water forces leachate into subsurface soils at a higher rate than can occur from gravity alone. EPA chose to model all surface impoundments during the active stage of their lifecycle under the assumption the constant presence of a hydraulic head would result in the highest releases (U.S. EPA, 2014a). Although the current configuration of the legacy and historical impoundments may vary, both types of units held free water during the active stage of their lifecycle. In the case of legacy impoundments, that free water is still present. Thus, EPA believes the risks from both historical

5) See: 80 FR 21,449

and legacy impoundments are best characterized using the same conceptual model developed for active impoundments.

EPA previously established the following conceptual model for active surface impoundments. During operation, surface impoundment wastewater may be lost to a combination of infiltration into subsurface soils, evaporation into the atmosphere, and discharge to adjacent impoundments or nearby water bodies. CCRs may accumulate until the surface impoundment's capacity is reached or the ash may be periodically dredged for disposition elsewhere. Impoundments may contain one or more of the different CCR types, as well as other wastes like coal refuse. To reflect that a majority of impoundments are dredged, the conceptual model assumes that dredging losses are balanced out by continued loading from the facility, resulting in a constant ash thickness and water depth over the active life of the unit. For the purposes of modeling, surface impoundments are assumed to be constructed with a square footprint and located anywhere from entirely above grade to entirely below ground surface. **Figure 2-3** depicts a conceptual model for one potential configuration of an active impoundment.

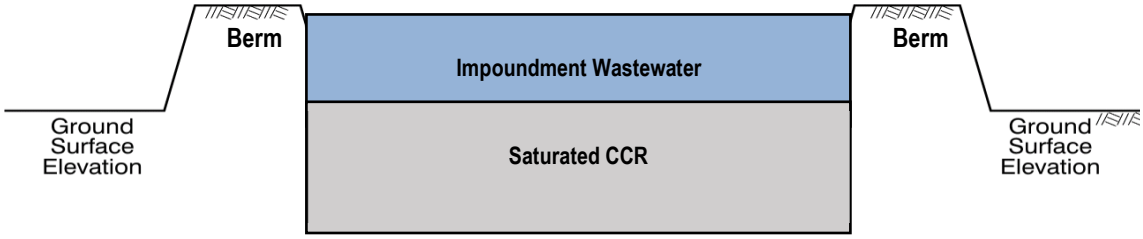

Figure 2-3. Cross-section view of active surface impoundment constructed above grade.

The 2014 Risk Assessment evaluated the potential risks to offsite receptors up to a mile away that result from disposal of CCR in surface impoundments located at active facilities (U.S. EPA, 2014a). It considered multiple exposure pathways as part of a national-scale, probabilistic analysis, which include human ingestion of impacted groundwater and fish caught from impacted streams, as well as ecological exposure to impacted surface water and sediment. On a national scale, the evaluation found potential for risk to human health from impacted groundwater to occur within the range that OLEM typically considers to warrant regulation.[6] In particular, unlined impoundments that account for a majority of regulated units were found to result in cancer risks up to 3×10^{-4} for arsenic and noncancer risks up to hazard quotients (HQs) up to 3, 4, and 2 for lithium, molybdenum, and thallium, respectively. Based on these results, groundwater exposure is considered the principal risk driver for regulated impoundments. Given the similar design and siting of legacy and historical units, the same exposure pathway will be the focus of further analysis for these units.

6) See: 80 FR 21,449

2.2.3 CCRMU Fills

EPA proposed to define CCRMUs as any area of land on which non-containerized accumulations of CCR are received, placed, or otherwise managed, and is not otherwise regulated as a landfill or surface impoundment under the 2015 Rule. These units would include historical impoundments and landfills, inactive landfills, and other areas where CCR has been managed directly on the land. The intent of this broad definition is to capture management practices that fall outside the scope of the 2015 CCR Rule, but still have demonstrated potential to contaminate groundwater. As a result, CCRMUs consist of a broad array of units unified by a single set of regulatory requirements.

As previously discussed, historical and inactive units have direct counterparts in the landfills and impoundments subject to the 2015 CCR Rule and so are best understood through the conceptual models for those regulated units. Therefore, a separate conceptual model was developed for the subset of CCRMUs placed on the land outside of a landfill or impoundment and intended for a purpose other than disposal. It is anticipated the vast majority of such placement is associated with subsurface fills or similar uses. For clarity, this subset is hereafter referred to as "CCRMU fill(s)."

CCRMU fill involves placement of dry CCR on or within the soil. In this way, the conceptual model for fills mirrors that of a landfill. During construction, a specified amount of CCR is placed in the fill. CCRMU fills are generally anticipated to be constructed with one or more of fly ash, bottom ash, or boiler slag. FGD solids tend to be far more soluble than other CCR types and so are generally not anticipated to be suitable as fill. The timeframe for construction of a CCRMU fill is anticipated to be shorter than for landfills, on the order of weeks to months instead of years to decades. At the end of construction, waste is left in place and some form of cover is placed over the ash. This cover may be native soil, concrete, or another material based on project specifications. For the purposes of modeling, CCRMU fills are assumed to be constructed with a square footprint and located anywhere from entirely above grade to entirely below the ground surface. **Figure 2-4** depicts the conceptual model for one potential configuration of a CCRMU fill.

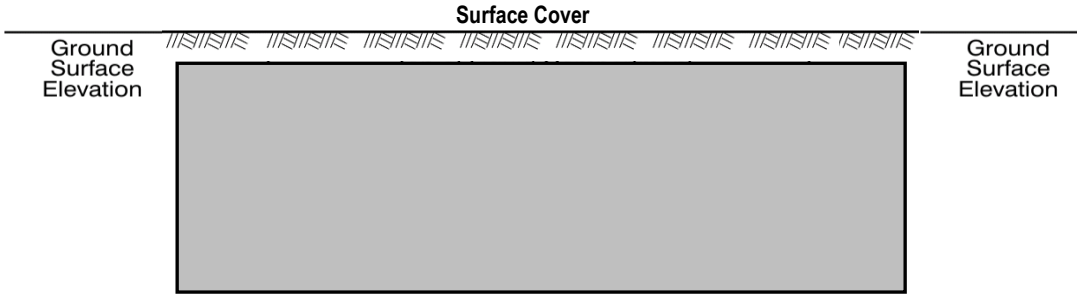

Figure 2-4. Cross-sectional view of generic CCRMU fill below grade.

CCRMU fills are located at the same facilities as both historical and inactive units. However, the exposure pathways for CCRMU fills can diverge from landfills and surface impoundments because

facilities have not historically regarded such placements as a form of disposal. There is no indication facilities have reliably tracked or maintained these placements over time, as is required for disposal units. As a result, it is anticipated that, in the absence of further regulation, these fills will remain in place when ownership of the property changes.

EPA is concerned the potential risks from CCRMU fills have not been adequately characterized for all the stages of the fill lifecycle. The presence of engineering controls, such as an impervious structure constructed on top of the fill, might limit exposures while the facility is active. Yet, in the absence of land use restrictions, there is no guarantee these engineering controls will remain in place when the property is redeveloped. Disturbance of fill can bring ash to the ground surface and create new pathways through which future receptors may be exposed. For these reasons, EPA identified future land use as the stage of the CCRMU fill lifecycle most likely to pose risk.

The Risk Assessment Guidance for Superfund and subsequent Agency policy directives instruct EPA to "assume future residential land use if it seems possible based on the evaluation of available information" (U.S. EPA, 1989; 2010). None of these facilities are likely to operate as electric utilities forever. Indeed, review of the U.S. Energy Information Administration (EIA) Form EIA-860 identified at least 85 utilities that burned coal and have closed since 2015.[7] Many of these utilities are located along water bodies, which increases the value of the property and the potential options for future redevelopment. EPA is aware of at least one instance where a closed property is already slated for use as a mixed-use development.[8] Further, EPA estimates the facilities considered in this risk assessment already have 38,000 individuals living in a five-mile radius on average, and over 100,000 in extreme cases.[9] Therefore, consideration of future residential land use is considered to be appropriate when identifying relevant exposure pathways.

EPA identified three complete exposure routes for future residential receptors that warrant further investigation. These pathways are not intended to represent a comprehensive list of all possible pathways. Instead, the focus is on those anticipated to result in the greatest risks and so to be the focus of any regulatory action. First, chemical constituents present in CCRMU fill can be released by dissolution into precipitation and other water that comes in contact with the CCR. Dissolved constituents can infiltrate down to the underlying water table and migrate through the aquifer to downgradient wells where residents are exposed through ingestion of drinking water. Second, radioisotopes present in CCRMU fill may decay and release radiation in the form of either gamma rays or radon gas. Gamma rays can pass through soil and other materials to reach the ground surface where residents are exposed directly to ionizing radiation. Radon gas may migrate through the soil

7) See: https://www.eia.gov/electricity/data/eia860/

8) See: https://www.alexandriava.gov/neighborhood-development/potomac-river-generating-station-prgs-power-plant-redevelopment-old-town

9) Data collected according to the same procedures outlined in **Section 4.2.7**.

and accumulate in nearby buildings, where residents are exposed through inhalation of indoor air. Finally, the CCR may become intermingled with surface soils, where residents are exposed through incidental ingestion of soil and dust present outdoors and tracked into the home.

2.3 Data Sources

The current assessment builds on the 2014 Risk Assessment and incorporates many of the same data sources previously applied to characterize facility environmental setting, waste composition and release potential, fate and transport through groundwater, and contaminant toxicity (U.S. EPA, 2014a). Many of these data sources and the approach to selecting these data for individual groundwater model runs are unchanged from the 2014 Risk Assessment and were previously discussed in detail as part of that previous assessment. These data sources were made available for public comment and external peer review and were found to represent the best available data. Therefore, EPA focuses subsequent discussion in this document on those data sources that have since been updated. The following text details the major data sources relied upon in the current assessment that differ from or expand upon previous modeling.

2.3.1 Facility Data

The 2014 Risk Assessment relied on two EPA surveys conducted in 2009 and 2010 to identify the location of facilities with onsite disposal units ("2009/2010 EPA Surveys"). These surveys include data on 952 surface impoundments and 431 landfills located across 383 facilities. EPA ultimately determined that 218 of the surface impoundments and 122 of the landfills fell outside the scope of the final CCR Rule for one of the following reasons and so were not incorporated in the final risk model:

- The facility was no longer a coal-fired electric utility according to the 2012 EIA database;
- The landfill or surface impoundment was found to be inactive or retired; or
- The impoundment was not designed to accumulate CCRs (e.g., cooling water ponds).

Since 2014, EPA has identified 42 additional facilities subject to the 2015 Rule that were not modeled in the 2014 Risk Assessment. These facilities were identified as a result of rule reporting requirements. Therefore, these regulated facilities were considered as part of current assessment. A list of these regulated facilities is provided in **Appendix A**.

In 2021, EPA published an Advance Notice of Proposed Rulemaking requesting information on legacy impoundments. A review of comments received resulted in identification of 127 legacy impoundments at 59 new facilities. EPA subsequently received additional data suggesting the existence of an additional 89 legacy impoundments, 73 of which are located across an additional 33 legacy facilities. The research and analysis needed to validate all these newly identified legacy impoundments is extensive. EPA generally lacks unit-specific data to confirm whether a unit meets the definition of a legacy impoundment. Yet, to accommodate the possibility these units are legacy

impoundments, all of the 92 identified facilities were considered as part of current assessment. A list of these facilities is provided in **Appendix A**.

2.3.2 Meteorological Data

The 2014 Risk Assessment relied on the Hydrologic Evaluation of Landfill Performance (HELP) Model Version 3.0 to characterize meteorological conditions in the vicinity of modeled landfills and impoundments (U.S. EPA, 1994). HELP v3.0 provides data on precipitation, temperature, and solar radiation from a 30-year period between 1961 and 1990 at up to 183 meteorological stations at cities across the United States. Each disposal unit was assigned to the closest meteorological station to identify relevant data for environmental modeling.

Since the 2015 Rule was finalized, EPA released HELP Version 4.0 (U.S. EPA, 2020). This model update incorporates a meteorological dataset developed by the EPA Office of Pesticide Programs (OPP) with National Oceanic and Atmospheric Administration (NOAA) data. The OPP dataset is a grid of over 13,000 points distributed evenly across the country on a 0.25 × 0.25 degree grid (latitude/longitude) across the conterminous United States. Meteorological data are available for a 30-year period between 1985 and 2014 at each point on this grid. The current version of the HELP model is available for download on the EPA website.[10] Further information on how the OPP dataset was developed is described in Frye et al. (2016).

HELP v4.0 provides more recent data at a finer spatial resolution. Thus, the current risk assessment relied on the updated model to assign meteorological data to individual facilities. Once assigned, meteorological data drawn from HELP was applied to modeling of groundwater fate and transport in the same way as in the 2014 Risk Assessment. A list of the grid location assigned to each facility is provided in **Appendix A**.

2.3.3 Bulk Concentration Data

The 2014 Risk Assessment relied on a constituent dataset consisting of all the bulk concentration data for CCR that EPA had identified since 1998. These data were variously drawn from state submissions, public comments, Agency studies, and peer-reviewed journal articles. Because these sources all had different goals in collecting the data, the amount of data available for each CCR type and chemical constituent varied widely. For example, the dataset did not contain data on the bulk concentration of lithium in any type of CCR. As a result, the 2014 Risk Assessment was unable to directly model release of lithium from closed landfills. Therefore, EPA has continued to review other data sources in order to expand upon the existing dataset.

EPA identified the U.S. Geological Survey (USGS) coal quality (COALQUAL) database as a new source of bulk concentration data. COALQUAL contains information on nearly 7,500 samples of

10) Available online at: https://www.epa.gov/land-research/hydrologic-evaluation-landfill-performance-help-model

coal and associated rocks collected between 1973 and 1989. It was initially published in 1994 as part of the U.S. geoCHEMical (USCHEM) database. The most recent Version 3.0 was released in 2015 as a standalone database (USGS, 2015). COALQUAL contains data on up to 136 parameters for each sample, which include coal source, elemental composition, and ash yield. EPA used this information to estimate constituent concentrations in the coal ash remaining after combustion. The full COALQUAL database is provided in **Appendix B**.

COALQUAL improves upon the previous constituent dataset for a number of reasons. First, the database includes a large number of geographically diverse samples collected from across 36 states. This provides confidence these samples adequately capture regional variability in coal quality. Second, the samples represent a composite of entire coal beds weighted by the thickness of each discrete interval of minable coal (or "bench"). This ensures the data are representative of broader coal quality from a mine and so less likely to be skewed by any isolated hotspots. Third, samples were collected by USGS and state agency partners, so there is no potential for bias toward individual facilities that might be more likely to provide samples for research. Finally, USGS has undertaken extensive data verification and validation efforts to ensure consistency and reliability of the database.

It is difficult to merge the data from COALQUAL database and the previous constituent dataset because the latter often does not include information about the coal source. However, given the large size of the COALQUAL dataset, merging the two would not be expected to substantially alter summary statistics. Therefore, EPA relied solely on the COALQUAL dataset to characterize CCR bulk content. Data from the previous constituent dataset were instead used for comparison in **Section 6 (Uncertainty and Sensitivity Analyses)**.

2.3.4 Leachate Concentration Data

The 2014 Risk Assessment relied on a constituent dataset consisting of all the CCR leachate data EPA had identified since 1998. These data were drawn from state submissions, public comments, Agency studies, and peer-reviewed journal articles. These data include porewater samples and leaching test data collected with SW-846 Methods, including extraction procedure toxicity test (Method 1310), toxicity characteristic leaching procedure (Method 1311), synthetic precipitation leaching procedure (Method 1312), and the leaching evaluation assessment framework (LEAF) methods (Methods 1313-1316). Following a review of the available data, EPA determined it was most appropriate to use porewater data to model leakage from surface impoundments and LEAF data to model leakage from landfills in the 2014 Risk Assessment.

EPA has since identified LEACHXS Lite as a source of new leachate data. LEACHXS Lite is a free data management and visualization tool that was developed by Vanderbilt University and others

in partnership with EPA.[11] The LEACHXS Lite database contains all the LEAF data relied upon in the 2014 Risk Assessment, as well as LEAF data from other sources and for other materials. Review of the database identified one additional sample of fly ash leachate (Sample FA39), so EPA incorporated this sample into the larger constituent dataset. The additional leachate data is provided in **Appendix B**.

11) Available online at: https://www.vanderbilt.edu/leaching/leach-xs-lite/

3 Disposal Unit Groundwater Risk

As previously discussed, all landfills and surface impoundments progress through similar lifecycle stages during and after operation. Inactive units can be understood as a pause at some point in that progression. The fact that some historical and inactive units may no longer contain free water, or may have installed a soil cover, places these units in a different stage of their lifecycle. However, as noted during development of conceptual models, that alone does not necessarily differentiate the long-term risks from those of previously modeled active units. The potential for risk, and thus the need for groundwater monitoring, must be considered over the full lifecycle of the unit.

The potential risks associated with historical and inactive units can be understood in relation to those previously modeled in the 2014 Risk Assessment. EPA reviewed these previous model results and the currently available data about historical and inactive units to understand any differences between these units anticipated to affect modeled risks. The purpose of this section is to summarize the data EPA reviewed and document the potential for risks from historical and inactive disposal units to meaningfully diverge from those previously modeled.

3.1 Previously Excluded Units

The 2014 Risk Assessment relied on the 2009/2010 EPA Surveys. These surveys include data on 952 impoundments and 431 landfills located across 383 facilities. Of these units, EPA ultimately determined that 218 surface impoundments and 122 landfills fell outside the scope of the 2015 CCR Rule for one of the following reasons, and so even though EPA modeled the risks associated with these units, the results were not incorporated in the final risk results:

- The facility was no longer a coal-fired electric utility according to the 2012 EIA database;
- The landfill or surface impoundment was found to be inactive or retired; or
- The impoundment was not designed to accumulate CCRs (e.g., cooling water ponds).

After removing impoundments not designed to contain CCR, there remained 122 landfills and 163 surface impoundments that represent either historical or inactive disposal units. As noted, EPA previously modeled the risks associated with these units prior to excluding them from the final risk results. Therefore, these model results provide the most direct comparison of risks between regulated and excluded units. EPA reviewed these results to understand how the risks associated these specific units compare with the active units previously reported in the 2014 Risk Assessment.

Table 3-1 presents the 90th percentile model risks to human receptors resulting from groundwater exposure. Both cancer and noncancer risks are presented for the highest exposed receptor group. For drinking water ingestion, the highest cancer risks were for adults (Ages > 21 years), while the

highest noncancer risks were for infants (Age < 1 year). All values are rounded to the nearest digit. Values that exceed the selected risk criteria (i.e., cancer risk > 1×10^{-5} or HQ > 1) are shown in **bold**.

Table 3-1. 90th Percentile Nationwide Risks for Human Health from Excluded Units

Constituent	Surface Impoundments	Landfills
Carcinogenic Effects		
Arsenic III	**8×10^{-5}**	7×10^{-6}
Arsenic V	4×10^{-6}	3×10^{-7}
Noncarcinogenic Effects		
Arsenic III	**2**	0.2
Arsenic V	0.1	< 0.01
Molybdenum	**1**	< 0.01
Lithium	**2**	--*
Thallium	0.5	0.2

* Method 1313 data were not available in the literature to model this constituent for landfills.

For impoundments, the 90th percentile risks associated with ingestion of ground water are above cancer and noncancer criteria for arsenic III (risk = 8×10^{-5}, HQ = 2) and noncancer criteria for both lithium (HQ = 2) and molybdenum (HQ = 1). For landfills, the 90th percentile risks are below the cancer criteria for arsenic III, but still within OLEM risk range (risk = 7×10^{-6}). The risks associated with this set of excluded units vary somewhat from those previously reported, with slightly higher risks for landfills and slightly lower risks for impoundments. Yet there is strong agreement on the overall magnitude of risk.

Since finalization of 2015 CCR Rule, facility reporting has revealed that a far greater percentage of active units are unlined than previously modeled. EPA has seen no evidence that would indicate the actual prevalence of liners at these excluded landfills is markedly different. As part of the 2014 Risk Assessment, 71 percent of the excluded landfills and 57 percent of excluded impoundments were modeled as unlined. Risks for the unlined subset of landfills fall above the criteria for arsenic III (risk = 1×10^{-5}). Again, these results closely mirror those of the 2014 Risk Assessment (risk = 2×10^{-5}). EPA previously modeled the covers for unlined units as two feet of soil with a permeability equivalent to that of the surrounding soils. EPA believes these model results are equally applicable to historical and inactive landfills that have not closed consistent with the requirements of the 2015 Rule.

3.2 Facility Locations

Facility location is a useful proxy for the environmental conditions to which a unit may be exposed. Nearby facilities are likely to be subject to a similar range of weather and hydrogeologic conditions. Therefore, EPA reviewed the locations of facilities that were modeled in 2014 and that have since been identified to understand any differences in the geographic distribution of these facilities. **Figure 3-1** provides a map of the locations of different facility identified as having onsite disposal.

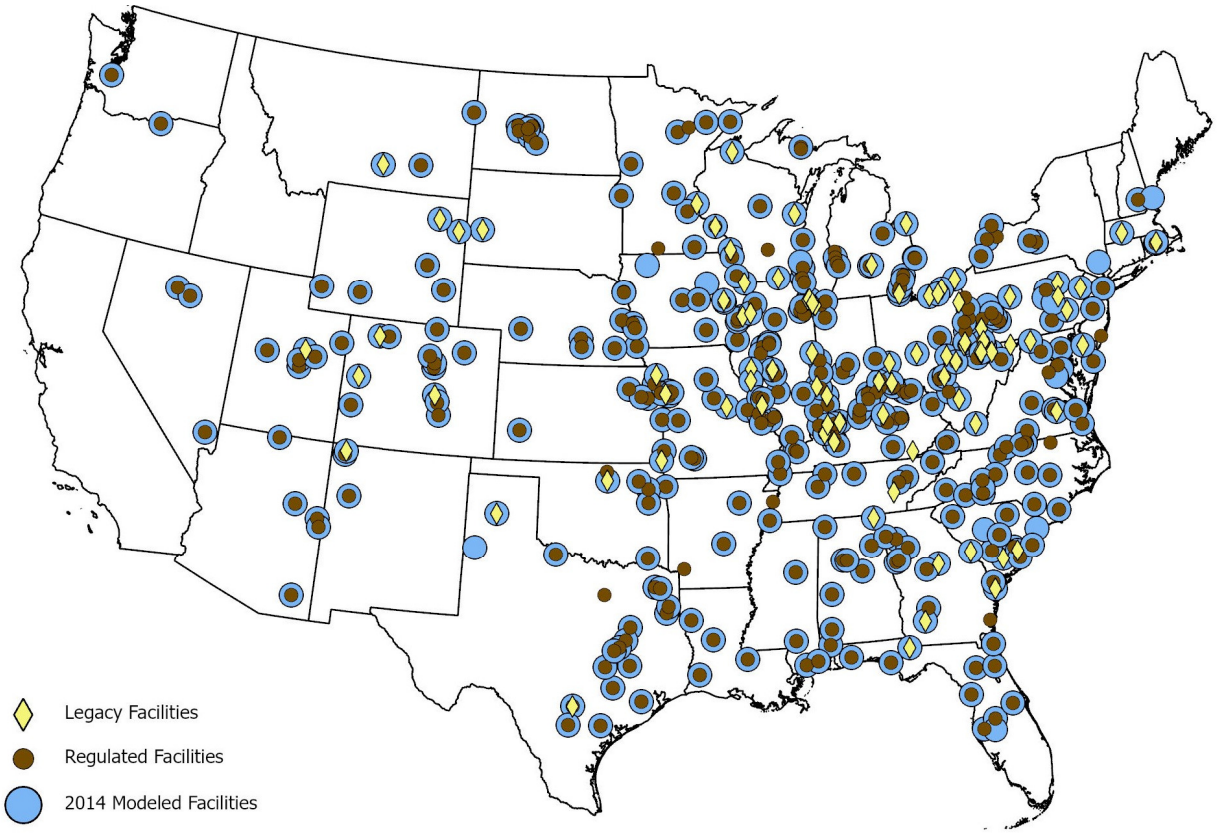

Figure 3-1. Identified facilities with onsite disposal in the conterminous United States.

The 2014 Risk Assessment modeled disposal units at 380 facilities across the conterminous United States, as well as two additional facilities in Alaska and Puerto Rico not depicted here. That count includes 74 facilities that are not currently subject to the 2015 CCR Rule but have been identified as still having an inactive impoundment onsite. After subtracting out a few apparent duplicates from the reported list of legacy facilities, there are 14 legacy facilities that have not been previously modeled.

Some historical and inactive disposal units are located at previously modeled facilities. In many cases, these units are expected to be located in close proximity to previously modeled landfills and impoundments. As a result, these units would be modeled with the exact same weather conditions and hydrogeologic zones used for previously modeled landfills and impoundments. EPA previously drew environmental data from a mile around the units to capture the prevalence of different soil

types and related parameters. This approach is expected to capture broader conditions at a facility. So, while the overall distribution of collected parameters might shift if the sampling area is moved to another point at the same facility, it is highly unlikely it would result in a distribution so dramatically different that it would alter the conclusions drawn from probabilistic modeling. Further, a shift could result in either higher or lower risk estimates. As shown for the previously excluded landfills, different site conditions can result in somewhat higher risk. As a result, there is no reason to believe the environmental conditions at these historical and inactive disposal units would meaningfully differ from those previously modeled.

All legacy facilities that were not previously modeled are clustered among the facilities that were. There is no indication that this subset of facilities is unique. The proximity to other facilities makes it highly unlikely the range of environmental conditions at these additional legacy facilities have not already been captured through modeling of nearby facilities. Furthermore, the location of these remaining legacy facilities are concentrated in the eastern half of the country. This will tend to result in facilities more concentrated in regions of higher precipitation than considered nationally. Thus, the modeled risks for these units may be somewhat higher than those previously modeled for the entire country.

3.3 Unit Size

EPA previously considered the potential for unit size to affect modeled risks (U.S. EPA, 2019a). To do this, EPA first filtered the 2014 Risk Assessment groundwater model runs of unlined landfills where contamination reached private wells before being intercepted by surface water. These limits were placed to focus on model runs that best reflect CCRMU fills and to eliminate confounding factors that could obscure wider trends. EPA focused on cancer risks from arsenic III because it was the primary risk driver identified for unlined landfills. This filtering of model results did not dramatically alter the overall risk profile. The 90th percentile cancer risk of all filtered results is 2.8×10^{-5}, which compares well with the risk of 2.0×10^{-5} for all unlined landfills. So, there was little concern this filtering excessively skewed the overall results.

To better understand the effects of disposal area on modeled risk, EPA organized the filtered model runs from the smallest to largest value of an independent variable (e.g., acreage) and calculated running summary statistics in increments of 2,000 runs (i.e., 1-2000, 2-2001, 3-2002, etc.). This number of runs was selected based on trial and error as the number that provided the clearest view of trends with the fewest number of runs. A running 90th percentile was then calculated for the dependent variable (i.e., risk) because it represents high-end risks considered in the 2014 Risk Assessment. **Figure 3-2** presents the relationship between modeled high-end risks and both total area and mass disposed per area.

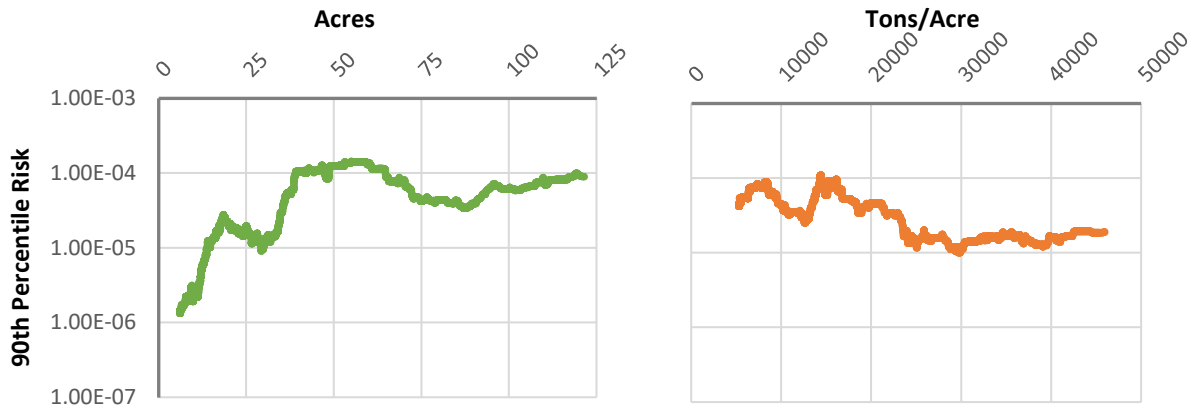

Figure 3-2. Modeled risks as a function of area and mass per area.

The resulting curves are not smooth because the summarized model runs incorporate site-specific data. Thus, individual peaks and troughs may reflect a subset of units that tend to pose more risk (e.g., more permeable soil) or less risk (e.g., more arid environments). The Agency did not attempt to isolate the cause of each fluctuation and instead focused on broader trends, as there are several conclusions that can be readily drawn from these figures.

First, the mass per unit area is not a sensitive parameter for disposal units. High-end risks fell above the 1×10^{-5} point of departure across the full range of values. Risks resulting from around 5,000 tons per acre (thickness of around 2.5 feet) (4×10^{-5}) are not dramatically different from those at 50,000 tons per acre (2×10^{-5}). This indicates that even 2.5 feet of ash can contain enough arsenic mass to sustain leaching and pose risk, if applied over a wide enough area.

Second, disposal area ceases to be a sensitive parameter above a certain size. This is because the area eventually becomes so large that further increases no longer contribute to the groundwater concentrations closer to the centerline of the plume. As the contaminant plume grows larger, the modeled well locations selected at random will fall in regions of higher concentration more often. Thus, beyond a certain plume size, modeled probabilistic risks will eventually plateau.

Finally, modeled landfill risks start to consistently decline only beyond a certain size, falling below the 10^{-5} around 15 acres, but remaining within the risk range. The corresponding acreage for impoundments would be even smaller due to the higher overall risks identified for those units. Based on these results, it is clear both legacy impoundments and CCRMU disposal units, with estimated average sizes of at least 31 and 33 acres, can be expected to pose a similar magnitude of risk as currently regulated units. It is possible based on these findings that smaller placement of CCR would not pose risk within the OLEM risk range for receptors up to a mile away from these units. However, this does not mean such placements are without any risk.

The 2014 Risk Assessment modeled contaminant transport based on a national distribution of the closest residences anticipated to rely on groundwater as a source of drinking water. These locations were defined using a combination of Census reports, satellite imagery, and other geographic data. This modeling approach assumed that, while the proximity of receptors around each unit may shift over time, the overall distribution of receptors across the country would remain the same. The risks identified based on these receptors provided a sufficient basis for national regulations. Yet, this does not mean these are the only receptors relevant for consideration.

Reliance on a distribution of current receptors was necessary because peak risks sometimes occur after hundreds or thousands of years and there is no reliable way to predict population dynamics that far in the future. It was these risks to future receptors that formed the basis for regulation. Whether receptors are presently exposed has no bearing on the need to address the contamination potential identified for these units, as it may not be feasible to fully remove contamination from groundwater once released. A broad goal of RCRA regulations is to protect groundwater. Thus, placements of CCR that does not allow for unrestricted future land use may warrant regulation now to ensure such placements are tracked and properly managed, so the site does not later become a Superfund site after responsible parties have dissolved.

3.4 Conclusions

EPA previously modeled a number of historical and inactive disposal units. These model results were ultimately excluded from the 2014 Risk Assessment because these specific units were judged to not be subject to the 2015 CCR Rule. These model results provide direct, quantitative evidence of the relative risk between the currently regulated and other disposal units. This comparison found no evidence of a meaningful difference in the magnitude of modeled groundwater risks.

Next, EPA compared the location of regulated and legacy facilities to understand the potential for environmental conditions to differ from those previously modeled. The legacy facilities that were not previously modeled are all clustered near regulated facilities in the eastern half of the country. This proximity makes it highly unlikely the range of environmental conditions at these sites will diverge from those already modeled on a national scale. Instead, the clustering of these facilities in the east may result in higher overall rainfall, infiltration, and risk. Thus, there is no indication the locations or environmental settings of these facilities are unique or would alter conclusions about the potential for risk from disposal of CCR in these types of units.

Finally, EPA reviewed the results of previous model runs to understand the effect of unit size on landfill risks. This review found risks effectively plateau for units above a certain size because the resulting plume has become so large. Based on available data, the majority of disposal units are expected to be so large that variation in size among them will have not have a meaningful impact on groundwater risk.

Based on the findings above, EPA has identified no evidence that the potential for risk from legacy impoundments, inactive landfills, or historical units would be expected to meaningfully differ from those of active units. Thus, EPA concludes that the results of the 2014 Risk Assessment provide a reliable basis to draw conclusions about the long-term risks from these additional disposal units. EPA did identify some potential for lower risks to receptors located up to a mile away from smaller placements of CCR. Therefore, EPA conducted further modeling to understand the potential risks associated with such placements. The design and results of this modeling is documented in **Section 4 (CCRMU Fill Groundwater Risk)**.

4 CCRMU Fill Groundwater Risk

EPA conducted further modeling of the exposures that may result if contaminated groundwater is used as a source of drinking water by future residents. The goal of this modeling is to characterize the risks associated with smaller quantities of CCR and understand whether a lower limit exists below which even smaller placement pose no concerns. The section of the document describes the overarching framework for this modeling effort, as well as the specific models and inputs used to predict the fate and transport of constituents through subsurface soils and ground water.

4.1 Modeling Framework

The placement scenario considered for the current evaluation is CCRMU fills located onsite at both active and legacy facilities. In the absence of any requirements to identify and track these smaller placements, it is assumed that the site will be redeveloped in the future for residential use. As part of redevelopment, it is assumed any current engineering controls have been disturbed (e.g., clay cap breached, overlying building demolished). It is assumed there will be some type of soil or other cover placed over the CCR so that it is not exposed to the open air. This is because CCR is not expected to support a robust vegetative cover and because exposed ash may require active measures for dust control. EPA used a combination of two models to characterize the potential impacts to groundwater quality and resulting risks to these residential receptors.

The first model is the EPA Composite Model for Leachate Migration with Transformation Products v2.22 (hereafter "EPACMTP") (U.S. EPA, 1996a, 1997a, 2003a,b,c). EPACMTP consists of two coupled modules: a one-dimensional module that simulates infiltration and dissolved constituent transport through unsaturated soils in the vadose zone and a three-dimensional module that simulates transport through groundwater. As described by the 2014 Risk Assessment, EPACMTP has undergone multiple rounds of internal and external review, including several by the EPA Science Advisory Board (Kool et al., 1994; U.S. EPA, 1996b, 1999, 2004). EPA used this model to calculate groundwater concentrations that result from waste disposal at specified locations and times. The outputs from EPACMTP were used to inform the second model.

The other model is the Modular Three-Dimensional Finite-Difference Ground-Water Flow Model - Unstructured Grid Transport (hereafter "MODFLOW") Version 1.10 (USGS, 2013a; Panday, 2022). MODFLOW is a three-dimensional model simulating flow and contaminant transport through groundwater. This model allows for more direct consideration of subsurface heterogeneity by dividing the aquifer into a grid and assigning different values to each cell in that grid. This version of MODFLOW was chosen because it is a publicly available model; allows for consideration of transport through unsaturated soils, which provides the most direct comparison to EPACMTP; has undergone extensive review and validation; and has been identified by EPA as appropriate for use at corrective action sites (EPA, 1997b, 2009a, 2015; Panday, 2022).

4.2 EPACMTP Setup

Model inputs for EPAMCTP were drawn from a range of site-based, regional, and national datasets based on a combination of government sources and peer-reviewed journal articles. Many of these sources are the same as previously used in the 2014 Risk Assessment (U.S. EPA, 2014a). These data sources were made available for public comment and external peer review and were found to represent the best available data. Therefore, subsequent discussion focuses on where the Agency incorporates new data sources or applies the same data sources in different ways. For example, when site-based data were not available for CCRMU fills, EPA drew data from more regional datasets.

4.2.1 CCRMU Fill Size

EPA has identified little available information on the total size of CCRMU fills present onsite at facilities. EPA believe it is unlikely that a consistent or reliable set of records could be identified for the purposes of characterizing CCRMU fill size. The available record indicates the location of such fills has not been closely tracked by facilities and the short timeframe for construction makes it unlikely these units would all be readily identified through aerial photography or similar means. Instead, EPA identified 74,800 tons as an upper limit of for the current assessment. This amount represents the smallest landfill size identified based on the 2009/2010 EPA Surveys (Kastner, 2015). Despite potential for larger fills, a 74,800 ton upper bound is believed to provide a clear distinction between landfills and CCRMU fills, and thus allow for a better understanding of potential risks associated with smaller placements of CCR not reflected in the 2014 Risk Assessment. Therefore, EPA modeled CCRMU fills based on a flat distribution ranging anywhere from 1 to 74,800 tons.

4.2.2 CCRMU Fill Dimensions

EPA has identified little available information on the relative dimensions of CCRMU fills present onsite at facilities. Therefore, for purposes of modeling, EPA placed upfront limits on the thickness and area of these fills to prevent these relative dimensions from becoming unrealistically extreme. EPA constrained the minimum thickness for all model runs to be one foot, intended to represent limited placement for grading or to promote drainage. EPA constrained the maximum thickness for a model run based on the tonnage selected for that run. To do this, EPA first conceptualized the fill as a conical pile. EPA then calculated the area of the most efficient pile possible based on tonnage, CCR density, and angle of repose (i.e., steepest angle the pile can sustain without collapsing). Finally, EPA spread the CCR equally over the pile area to achieve a uniform thickness. This thickness was set as the maximum depth for the model run to ensure the modeled depth did not exceed the area of the fill. The thickness for a model run was allowed to vary anywhere between these established minimum and maximum values. The area of the fill was then calculated based on the selected tonnage and thickness.

4.2.3 Depth Below Grade

It is assumed CCRMU fills can be constructed anywhere from entirely below the ground surface (e.g., structural fill) to entirely above grade (e.g., embankment). However, even if the entire unit is constructed above grade, it is assumed the fill will have at least 2 ft of soil cover because CCR is unlikely to support robust vegetation on its own. It is also assumed the top of a fill will not begin more than 10 ft below the ground surface because of the compounding cost of thicker covers. The exact thickness of the soil above the CCR is not a sensitive parameter in the model and will not affect long-term infiltration. Instead, the selected maximum is intended to prevent shallow fills from being modeled at unrealistic depths and potentially intersecting with a deeper water table (e.g., a 10 ft thick fill located 100 ft below ground surface).

CCRMU Fills were not allowed to be placed in direct contact with the groundwater table. This is due in part to the constraints inherent in EPACMTP resulting from assumptions made to allow efficient derivation of flow and transport equations. First, the model assumes that flow in the unsaturated zone is one-dimensional and directed down toward the water table. Second, the model assumes that flow in the unsaturated zone is driven only by leakage from a contaminant source and can be represented by a constant rate. Contact between the fill and water table may violate these assumptions as one-dimensional and steady vertical flow cannot be guaranteed, particularly in cases of groundwater mounding. Furthermore, contact between the waste and water may shift redox conditions in ways that alter leaching behavior, which cannot be accounted for with available leaching test methods. Therefore, if the random sampling of model inputs resulted in a scenario where a fill was in contact with the water table, those inputs were discarded and resampled.

4.2.4 Bulk Content

COALQUAL reports constituent concentrations on a whole coal basis. Therefore, additional steps were required to estimate concentrations in the resulting ash. COALQUAL reports two parameters in the datasheet that represent the amount of ash remaining after a coal sample has been burned. The parameter "Standard Ash" in the "CQ_Prox_Ult" datasheet represents the percent ash yield as determined by the American Society for Testing and Materials Method D-3174 following combustion at 750 degrees Celsius (ºC), while the parameter "GSAsh Dry" in the "CQ_Trace" datasheet represents the percent ash yield determined by the USGS laboratories following combustion at 525 ºC. For some constituents in COALQUAL, USGS had used the concentrations measured in the coal ash to back-calculate concentrations in the whole coal. Therefore, a goal of combustion at a lower temperature was to limit loss of volatile constituent mass from the ash and provide a more representative concentration in the whole coal. As a result of the lower combustion temperature, the resulting ash yield for "GSAsh Dry" is slightly higher. EPA selected "GSAsh Dry" to estimate concentrations in the ash because combustion of coal is often incomplete, with some

residual volatile matter (e.g., loss of ignition). Calculated concentration represents a whole ash concentration consisting of fly ash mixed with bottom ash or boil slag.

After calculating ash concentrations, EPA took additional steps to ensure that the data relied upon best reflect real world conditions. EPA first filtered out any samples that were not associated with a specific mine or utility under the column "Mine/PowerPlant" in the "QC_Descript" datasheet. This was done to ensure the samples had been collected from economically viable coal beds. Some samples in COALQUAL may reflect coal deposits that are not thick enough to support a mining operation. Others may reflect deposits that are not suitable for combustion, but might be utilized in other ways, such as for coal bed methane. EPA then filtered out any non-standard coal types listed under "Sample Description" in the "QC_Descript" sheet. Specifically, EPA removed samples listed as: "bone," "shale," "clay," "pyrite zone," "coal," and "carbonaceous to coaly shale." These samples might represent roof, floor, partings, or other non-coal samples.[12] This filtering removed only 17 samples and so is not anticipated to have a substantial impact on estimated concentrations. After filtering, the remaining samples consisted of lignite, bituminous, sub-bituminous, semi-anthracite, and anthracite.

Not all coal in the United States is mined in equal volumes. Therefore, EPA randomly sampled the filtered COALQUAL database a total of 150,000 times in rough proportion to recent production statistics. EPA sampled coal from eastern and western states at a relative proportion of 40:60, respectively (EIA, 2022a). EPA also sampled the different coal types in a relative proportion of 55:45 sub-bituminous to all other types (EIA, 2022b). Because anthracite and lignite represent a very small portion of the database, samples of "all other" types are predominantly bituminous coal.

For thallium, EPA found the calculated 90th and 50th percentile summary statistics were similar. This occurred because elevated detection limits in COALQUAL resulted in much of the thallium data being non-detect. No other constituent was affected by this issue. As a result, EPA was unable to rely on the full probabilistic distribution for thallium. Instead, EPA retained only the maximum and 90th percentile values from the distribution because these reflected measured concentrations. EPA set median and minimum concentrations based on the summary statistics for fly ash from the Agency constituent dataset. These four summary statistics were used to form a rough probability distribution that was sampled for this assessment. **Table 4-1** presents the resulting summary statistics used to characterize bulk concentrations. A comparison of these values with previous data sources is provided in **Section 6 (Uncertainty and Sensitivity Analyses)**.

12) Roof, floor, and partings are non-coal rocks found above, below, and interbedded within minable coal deposits.

Table 4-1. Calculated Bulk Constituent Concentrations (mg/kg)

Constituent	50th Percentile	90th Percentile
Arsenic	40	328
Molybdenum	15	53
Lithium	81	206
Thallium	0.5	10.8

4.2.5 Leachate Concentration

Leachate pH is a primary factor used to define relevant leachate concentrations. The 2014 Risk Assessment relied on a pH distribution generated from 580 samples collected from 42 landfills to approximate overall waste properties within landfills (U.S. EPA, 2014a). However, landfill disposal can include disposal of CCR types not relevant to CCRMU fills (e.g., FGD wastes) and mixing with non-CCR wastes (e.g., coal refuse). Therefore, for the current assessment, EPA instead relied on available LEAF Method 1313 leachate data to identify the range of natural pH for fly ash (i.e., the final pH when the ash is exposed only to water) (U.S. EPA, 2009b). EPA then applied an error bar of 0.5 pH units to either end of the reported natural pH to better capture potential variability and ensure a more continuous distribution. **Figure 4-1** depicts the resulting distribution of leachate pH, expressed both as a cumulative frequency and as a number of samples captured in discrete pH bins.

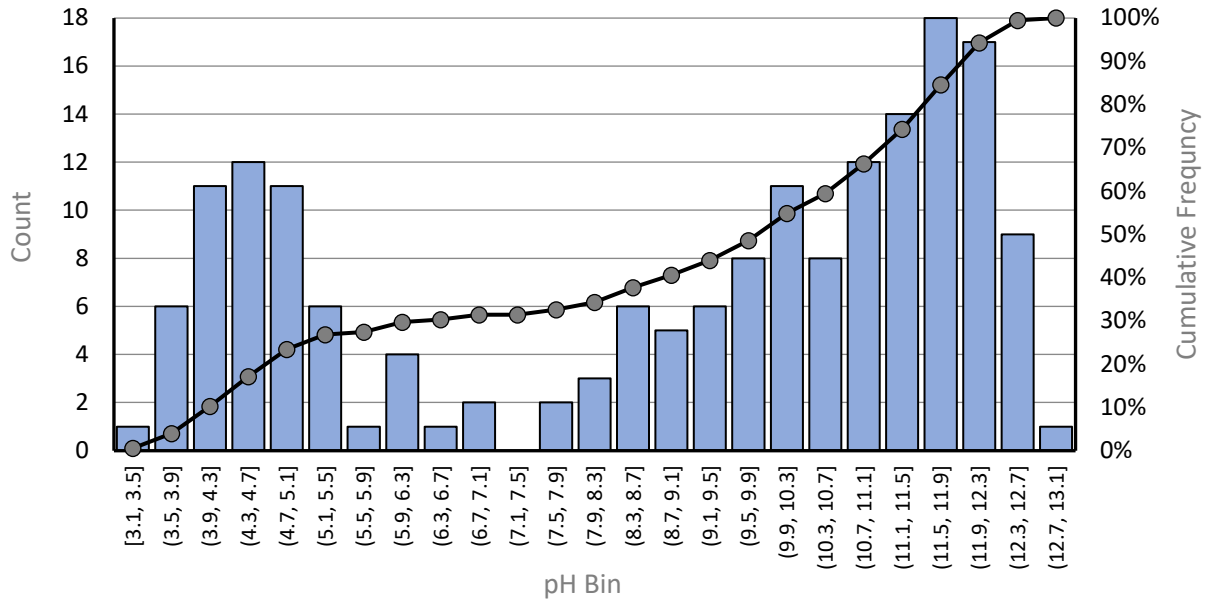

Figure 4-1. Modeled Leachate pH Distribution.

The current assessment relies on available data for fly ash measured with LEAF Method 1313 to characterize leachate concentrations from CCRMU fills. Data on mixed CCR were not used because of the potential inclusion of FGD scrubber sludge that is not considered relevant for this type of

placement because of its high solubility. Method 1313 does not provide data at the exact same pH for every sample. Therefore, EPA first interpolated between measured leachate concentrations to obtain values at consistent 0.25 pH increments. EPA then sorted the interpolated values into bins of 0.5 pH increments shown in the **Figure 4-1**. Thus, each of these bins had two values for each ash sample. To identify the relevant leachate concentration for each model run, the pH distribution was probabilistically sampled and the associated pH bin was selected. Next, a sample ID from the bin was randomly selected. This ensured the model was not biased toward individual samples that had been analyzed under different conditions (i.e., ACI turned on and off). Finally, one of the leachate concentrations associated with the sample ID in that bin was randomly selected for use in the model.

For lithium, only a single Method 1313 sample was identified. Based on review of this sample and the larger constituent dataset, EPA concluded it was appropriate to supplement this sample with available fly ash porewater data for which the associated pH is known. Given that lithium is a monovalent ion, it is unlikely that specific impoundment conditions, such as changes in redox, will substantially affect solubility. Indeed, the two sets of measurements were found to fall within a similar concentration range. Therefore, EPA included the porewater data in the bins associated with the measured pH and sampled using the same approach as for Method 1313 data.

4.2.6 Available Content

Available content (also commonly referred to as "leachable content" or "soluble content") is the constituent mass that can leach from a material over time. The remaining constituent mass may be tightly bound in poorly soluble mineral phases, such as alumina-silicate. Most laboratory leachate tests measure the constituent mass that can be released into a fixed amount of water, but do not provide a direct measurement of the total mass available to be released over time. Instead, the available content was estimated with Method 1313 data as the highest concentration released over the entire pH range (in mg/L) multiplied by the liquid to solid ratio of that sample (in L/kg). This is reasonable because the highly acidic pH will dissolve iron hydroxides and other complexations that may initially limit the release of constituents, and the high liquid to solid ratio will ensure that all of the trace constituent mass can be dissolved (U.S. EPA, 2014b). The resulting available content must, by definition, be less than or equal to the bulk content. However, the calculated available content may be slightly higher that the measured bulk content for some highly soluble constituents as a result of measurement uncertainty for two different media (i.e., solid ash vs. liquid leachate). In these instances, the available content was set equal to the measured bulk content.

To make the best use of all available data, EPA used available and bulk content for each Method 1313 sample to calculate a fraction of the bulk content that is leachable. The calculated values were then compiled into a distribution. Review of these distributions did not identify any clear patterns

between available and bulk content. Therefore, the full distribution was sampled and used to scale the bulk content from COALQUAL data.

For lithium, there was only one Method 1313 sample available and it did not have associated data on bulk content. Yet it was still notable that leachate concentrations were all relatively low compared to bulk content, indicating the potential that a sizable fraction of the lithium present is insoluble. Thus, rather than assume all lithium is leachable, EPA calculated an available content based on the highest measured leachate concentration and compared that against a low-end bulk concentration. This comparison yielded an upper limit on available content of around 40 percent, and so lithium available content was modeled as a flat distribution between 0 and 40 percent. This limit is considered appropriately conservative based on the available data.

4.2.7 Environmental Setting

As previously discussed, there is little information available on the exact locations of CCRMU fills onsite at facilities. These fills are placed for reasons other than disposal and so are not subject to the same siting considerations as landfills and impoundments (e.g., proximity to boilers, surface water). There are fewer limits to where placement for reasons such as grading and contouring may occur. As such, EPA assumed CCRMU fills could be located anywhere in the facility boundary.

EPA drew a 5-mile radius around each regulated and legacy facility centroid. EPA then randomly sampled from within that radius to identify relevant hydrogeologic and meteorologic conditions. This approach is in line with the 2014 Risk Assessment, which relied on a 1.2-mile (2.0-kilometer) radius around the facility centroid when the location of a disposal unit was not known. A larger radius was applied here because the aim is to more broadly capture conditions within and around a facility, rather than to stand-in for the location of a specific unit. The radius was set to match the scale commonly considered for evaluation of impacts to environmental justice communities. Once a sample location was identified, environmental data was extracted for use in EPACMTP, the same as previously described for the 2014 Risk Assessment.

The infiltration rate through disturbed CCRMU fill is assumed be equivalent to the overlying soil, as a substantially lower hydraulic conductivity could result in undesirable drainage issues. This assumption is considered reasonable because the reported hydraulic conductivity of CCRs have considerable overlap with natural soils. For example, the default values for uncompacted ash in the HELP model are 3.0×10^{-5} cm/s for fly ash, 4.1×10^{-3} cm/s for bottom ash, and 4.1×10^{-2} cm/s for boiler slag.

4.2.8 EPACMTP Sampling Location

EPACMTP requires users to specify a fixed point some distance downgradient of the contaminant source where the model will calculate resulting groundwater concentrations. This point may be

conceptualized as a well location where water is drawn from the ground. The current assessment considers two types of wells. The first type is a downgradient compliance well similar to those required at landfills by the 2015 CCR Rule. This well is located at a fixed location as close to the fill boundary as feasible. In this case, a distance of 3 ft (1m) from the centerline of the fill boundary was chosen because a plume will be thinnest directly adjacent to the fill. Therefore, the well was placed a short distance away from the fill boundary to provide a chance for the plume to mix somewhat with groundwater and to ensure the model did not miss evidence of contamination by inadvertently sampling beneath the plume. The second type is a monitoring well similar to those used to delineate contaminant plumes as part of corrective action. These wells were placed at fixed locations 10 ft, 100 ft, and 1,000 ft away from the centerline of the fill boundary. Because these fills may be located anywhere at a facility and because the intent of this modeling is to understand the potential magnitude of groundwater contamination to which future onsite receptors may be exposed, the current assessment did not assume interception by surface water. **Figure 4-2** shows a schematic of the well locations relative to a CCRMU fill.

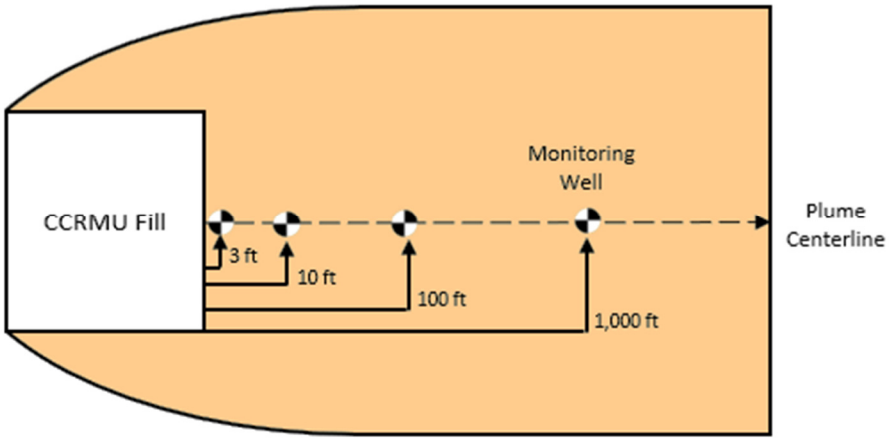

Figure 4-2. Aerial view of conceptual model for well locations.

For each model run and each well, a sample depth was randomly assigned within the top five feet of the surficial aquifer. The purpose of this interval is two-fold. First, it best reflects groundwater concentrations that would be measured by the low-flow sampling used in remedial investigations. Thus, this interval is considered most appropriate to understand potential for exceedance of groundwater protection standard (GWPS). Second, it ensures a consistent frame of reference among the different wells. EPA has found that default of sampling in the top 30 ft of the aquifer can result in oversampling below the plume where it is thinnest, resulting in the appearance of lower concentrations at the wells closest to the unit.

4.2.9 Risk Benchmarks

For every model iteration, the maximum (i.e., peak) concentration at each time step was identified. Then the single year of highest concentration across all modeled time steps was identified. Finally, a time-weighted exposure concentration was calculated by averaging the concentrations for each time step over the specified exposure duration, centered around the year of highest concentration. In cases where the ground water concentration was found to still be increasing after 10,000 years, EPACMTP stopped modeling and used the ground water concentration at that final year as the peak concentration. The groundwater concentrations calculated for each well were aggregated into probability distribution for that location. Summary statistics were drawn from this distribution and compared against relevant risk benchmarks.

For the compliance well, the averaging time used was a single year. If the modeled concentrations at this location exceed promulgated GWPS, that is evidence CCRMU fills have potential to result in the same concentrations that would trigger corrective action in regulated landfills. This is a relevant consideration not only for the potential for CCRMU fills to directly impact groundwater quality, but also for the potential for unmonitored releases to migrate from the CCRMU fills and interfere with groundwater monitoring at nearby regulated units.

For each of the monitoring wells, risk was calculated for a reasonable maximally exposed (RME) residential receptor, relying on the Agency's approach to assessment of CERCLA sites (U.S. EPA, 2014c). EPA chose this approach to ensure that the full risk potential of individual sites is properly characterized, even if such high-end exposures ultimately occur at only a small percentage of sites nationwide. Because the future locations of these receptors are unknown and may occur relatively infrequently, the risk to this sensitive population may not be adequately captured by a nationwide assessment of receptor behavior. Including these considerations reflects the fact EPA is directed not only to issue nationwide rules, but also to issue site-specific permits. Thus, the exposure duration was set to 26 years for cancer risk and one year for non-cancer risk. If the modeled risks at these locations exceed risk criteria, that is evidence that contamination can spread far enough to pose risk to receptors. When risks were identified at these wells, EPA conducted additional modeling with MODFLOW to characterize the extent and duration of the identified plume.

4.3 MODFLOW Setup

EPACMTP is designed to model groundwater concentrations at a single compliance location. Thus, it can be difficult to get a broader sense of full magnitude and extent of an individual plume from those results alone. MODFLOW was selected for use in this evaluation because it is a fully three-dimensional model that is able to track the full shape and size of a contaminant plume over time. However, MODFLOW is not designed to be run in the same iterative and probabilistic manner as EPACMTP. Instead, EPA used MODFLOW for a focused analysis intended to represent the 90th percentile groundwater concentrations identified from the 100,000 EPACMTP model runs.

EPA first identified the 90th percentile groundwater concentration at the 1,000 ft monitoring well of EPACMTP model runs. EPA then pulled all model runs within 0.5 percent of that concentration for further review, as these represent the model runs with the greatest potential for transport away from the CCRMU fill. From these model runs, EPA then selected 25 at random from different states to capture a diversity of scenarios that resulted in high-end contaminant plumes.

4.3.1 Model Inputs

Inputs for MODFLOW were drawn directly from selected EPACMTP runs without modification. This was done to ensure the conditions captured by MODFLOW mirror those from EPACMTP. **Table 4-2** lists the input parameters drawn from EPACMTP and their equivalent in MODFLOW. Some of the inputs used by EPACMTP do not have direct equivalents in MODFLOW because that model calculates them from other data. A list of the 25 EPACMTP model runs and the associated inputs is provided in **Appendix C**.

Table 4-2. Comparison of EPACMTP and Corresponding MODFLOW Model Inputs

EPACMTP Input Name	MODFLOW Input Name	Parameter Description	Units	MODFLOW Package
AREA	Modeled, not directly specified	Area of fill	m^2	N/A
XW	Modeled, not directly specified	Length of fill in direction of groundwater flow	m	N/A
RECHRG	RECH	Infiltration rate outside fill footprint. Modeled same as inside.	m/yr	RCH
SINFIL	RECH	Infiltration rate inside fill footprint. Modeled same as outside.	m/yr	RCH
TSOURC	PERLEN	Duration of leaching	yr	DISU
DEPTH	Modeled, not directly specified	Depth or thickness of fill	m	N/A
DGBS	Modeled, not directly specified	Depth of fill bottom below ground surface	m	N/A
CZERO	CONC	leachate concentration/constant conc. for piles and for fills initial conc. decay over time	mg/L	RCH
POR	PRSITY	Effective porosity of saturated soils	-	BCT
BULKD	BULKD	Bulk density of saturated soils	g/cm^3	BCT
ZB	Modeled, not directly specified	Thickness of saturated zone	m	N/A
XKX	HK and VKA	Hydraulic conductivity of saturated zone (aquifer)	m/yr	LPF
GRADNT	Modeled, not directly specified	Regional hydraulic gradient in the aquifer	-	N/A
AL	DL	Longitudinal dispersivity in the aquifer	m	LPF
AT	DT	Transverse dispersivity in the aquifer	m	LPF
AV	DTYZ and DTXZ	Transverse and vertical dispersivity in the aquifer	m	LPF
SATK	KSAT	Saturated hydraulic conductivity of unsaturated soils	cm/hr	LPF

Table 4-2. Comparison of EPACMTP and Corresponding MODFLOW Model Inputs

EPACMTP Input Name	MODFLOW Input Name	Parameter Description	Units	MODFLOW Package
ALPHA	ALPHA	Moisture retention parameter (Van Genuchten) Alpha	cm^{-1}	LPF
BETA	BETA	Moisture retention parameter (Van Genuchten) Beta	-	LPF
WCR	SR	Residual water content	-	LPF
WCS	PRSITY	Saturated water content (effective porosity)	-	LPF
DSOIL	Modeled, not directly specified	Depth from ground surface to water table	m	N/A
DISPR	DL	Longitudinal dispersivity in unsaturated zone	m	LPF
RHOB	BULKD	Bulk density of unsaturated soil	g/cm^3	LPF
UFCOF	ADSORB	Kd of unsaturated zone	cm^3/g	LPF
SFCOF	ADSORB	Kd of saturated zone	cm^3/g	LPF

4.3.2 Model Design

MODFLOW is divided into a series of components called "packages." Each package performs a specific task, which can be added to or omitted from the model structure to represent the scenario of interest. For example, one package may define properties of individual soil layers, while another may introduce a point of groundwater withdrawal (e.g., pumping activities). **Table 4-3** describes the specific packages used in the current modeling effort.

Table 4-3. MODFLOW Packages Used

Model Package	Acronym	Reason for Use
BASIC	BAS	The package handles a number of administrative tasks for the model as a whole. It opens files and determines options that will be active. It declares and allocates memory for variables that can then be used by other packages to define parameters.
Block Centered Transport	BCT	This package was used to simulate solute transport in the entire model domain.
Output Control	OC	This package was used to instruct the model when and how to save outputs.
Sparse Matrix Solver	SMS	This package was used to solve groundwater flow and transport equations. It incorporates nonlinear methods for conditions where conductance is a function of hydraulic head and linear solution schemes to solve for matrix equations
Recharge	RCH	This package was used to simulate the rate and location of infiltration into the soil and fill.
Transient Constant Head	CHD	This package was used to specify the water level in boundary cells and hydraulic gradient across cells.
Layer-Property Flow (LPF)	LPF	This package was used to simulate flow in the saturated zone. It specified layer types, grid dimensions, and material properties.
Unstructured Discretization	DISU	This package is used to organize and interrelate the location of different cells within the grid and to define initial time steps for the numerical solution.

Table 4-3. MODFLOW Packages Used

Model Package	Acronym	Reason for Use
Adaptive Time Stepping	ATS	This package allows the model to determine the appropriate length of time (time step) between each set of calculations to ensure efficient computation. If the model fails to converge on a solution to transport equations for a given time step, it will attempt to correct the problem by reducing the time step and solving again. It can also increase a time step length if a time step is quickly solved.
Prescribed Concentration Boundary	PCB	This package is used to specify a constant set of boundary conditions upgradient and downgradient of the unit.

MODFLOW allows for greater consideration of complex hydrogeology in modeled scenarios than EPACMTP. However, there is no reasonable means for EPA to assemble the level of detailed, site-specific data necessary to incorporate additional complexities on a national scale. Indeed, EPA has raised concerns through Part A and B reviews that facilities have not adequately characterized site hydrogeology even in the immediate vicinity of the regulated units. Therefore, some additional assumptions are required to enforce consistency in the modeling approach. Below is a list of the major assumptions to ensure consistency in the overall modeling approach. Additional parameter values are identified below:

- EPACMTP is unable to model waste in contact with the water table. Thus, EPA established the top two grid layers to represent the total unsaturated thickness of CCR and soil above the aquifer using the DISU package. The thickness of the first layer equals that portion of fill that extends below ground surface and the second layer starts at the bottom of the fill and extends to a depth calculated based on the total thickness of unsatured soil minus the depth of the fill below ground surface.
- EPACMTP assumes the unsaturated soil and aquifer are both homogenous and isotropic. To mirror these conditions in MODFLOW, EPA specified identical aquifer properties (e.g., hydraulic conductivity, porosity) in the X- and Y- directions. EPA then set the vertical anisotropy ratio (Kx/Kz) in the Z-direction equal to 1.
- EPACMTP assumes there is a constant aquifer thickness across the modeled area. It derives flow boundary conditions along the edge of the modeled area based on specified hydraulic gradient, hydraulic conductivity, and recharge rate. MODFLOW allows the saturated thickness of an unconfined aquifer to vary. To prevent a scenario where the aquifer is discontinuous, fixed water table depths are applied to the upgradient and downgradient ends of the model based on the specified saturated thickness and flow gradient for that model run.
- EPACMTP simulates steady groundwater flow in which groundwater head elevations do not change over time. MODFLOW simulates transient flow and so requires an additional input parameter, the specific yield (S_Y), to model an unconfined aquifer. S_Y is a unitless parameter

that accounts for gravity drainage from aquifer media. S_Y was set to a small number, 1×10^{-4}, to approximate steady flow conditions in MODFLOW

- MODFLOW requires a set of boundary conditions for concentration. EPA specified a zero concentration boundary upgradient of the fill to ensure no contributions from background. A zero concentration boundary was also set at the base of the aquifer to prevent contaminant mass from flowing through that confining layer. At the bottom of the fill, EPA specified a boundary condition equal to the leachate concentration to serve as the contaminant source. Finally, EPA set a boundary condition at the downgradient edge of the model domain equal to the concentration in the final cell, which allowed contaminant mass to continue to flow beyond the downgradient distance that was explicitly modeled.

4.3.3 MODFLOW Sampling Location

EPACMTP models the unsaturated and saturated zones as distinct components, each defined by a single set of parameters. MODFLOW does not explicitly define separate unsaturated and saturated zones. Instead, it relies on boundary and initial conditions to determine which parts of the model domain are saturated at any given time. MODFLOW allows the domain to be subdivided into cells that can each accept different parameter values. For purposes of this risk assessment, the model grid consists of 286 grid columns in the direction of groundwater flow, 181 grid rows perpendicular to groundwater flow, and 12 layers. This results in a total of 621,192 cells across the entire model domain. The specific number of layers and cells were defined through sensitivity analyses that identified the lowest number of cells at which the calculated plume volume and concentration had stabilized for all model runs (i.e., adding more cells did not refine the model outputs). **Figure 4-3** provides an example of how site hydrogeology is translated into a grid structure.

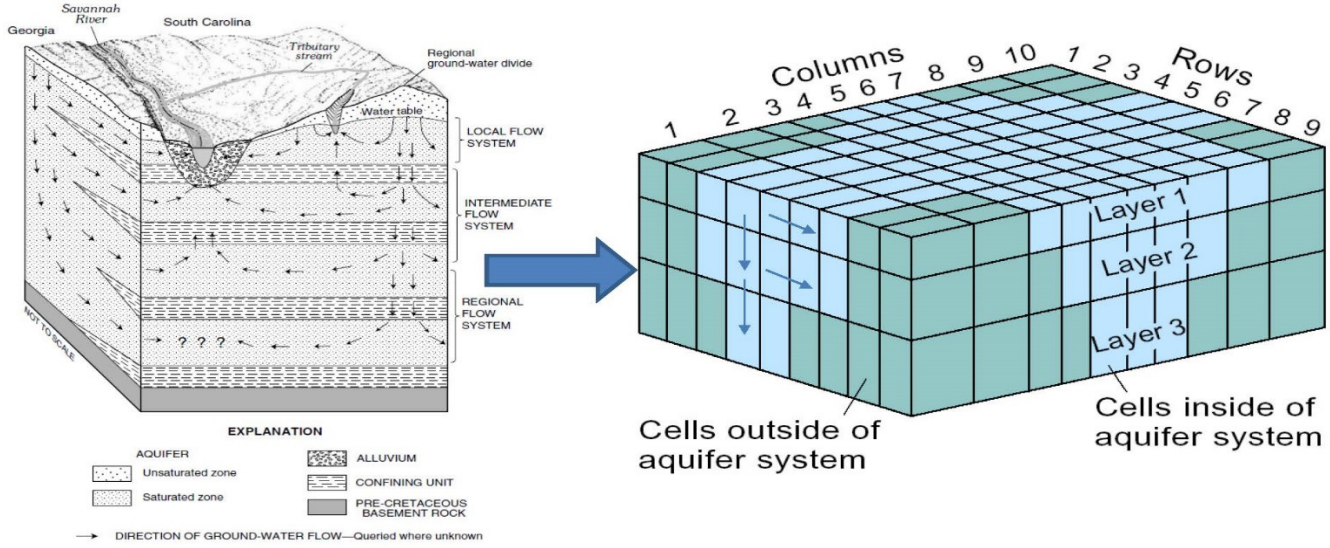

Figure 4-3. Conceptual model of three-dimensional groundwater flow.
(Adapted from USGS, 1996, 1997)

The model domain was set at 1,515 m (4,970 ft) in length, with the upgradient edge of the fill located 476 m (1,562 ft) away from the upgradient boundary. The model domain was set at 411 m (1,348 ft) in width, with the fill positioned along the centerline. The individual cells were shaped as a mixture of squares and rectangles, with a length that varies from 1 m at the smallest to 10 m at the largest. This layout was selected for several reasons: to allow finer resolution of calculations in the vicinity of the fill and along the plume flow path, to accommodate a range of fill sizes within the zone of higher resolution, and to reduce computational burden in areas furthest away from the plume. **Table 4-4** lists the sizes assigned to each cell within a layer. All layers in a run were assigned identical cell layouts. **Figure 4-4** shows an annotated screenshot of MODFLOW that depicts how different cell sizes are distributed within a modeled layer.

Table 4-4. Cell Layout in a MODFLOW Layer

Cell Length	Column Number	Row Number	Cell Aspect Ratio
10 m	1 to 39, 195 to 286	1 to 10, 171 to 181	1.5
6.5 m	40 to 41, 193 to 194,	11 to 12, 170 to 171	1.4
4.5 m	42 to 43, 191 to 192	13 to 14, 168 to 169	1.5
3 m	44 to 45, 189 to 190	15 to 16, 166 to 167	1.5
2 m	46 to 47, 187 to 188	17 to 18, 164 to 165	1.3
1.5 m	48 to 49, 185 to 186	19 to 20, 162 to 163	1.5
1 m	50 to 184	21 to 161	1.0

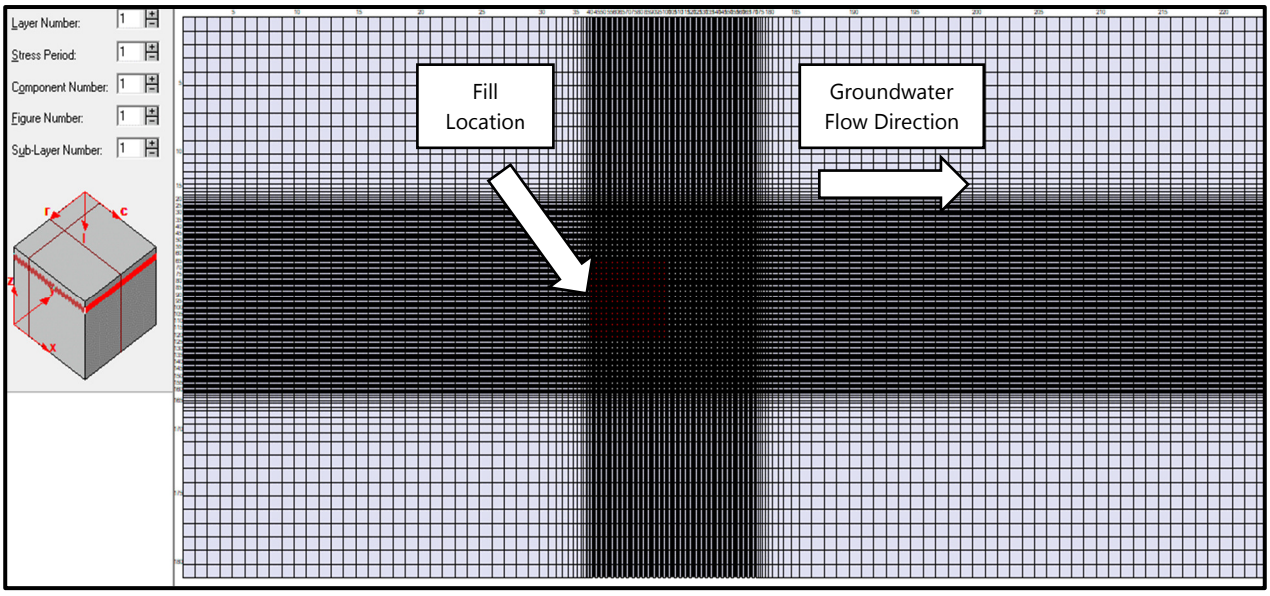

Figure 4-4. Depiction of grid layout for individual layer visualized within MODFLOW.

Generally, the top two layers were assigned as the unsaturated zone representing 1) the depth of fill below ground surface and 2) the unsaturated soil between the fill and aquifer. In rare cases where the thicknesses of the top two layers were too different to achieve model convergence, only

one layer was used to represent the unsaturated zone. The remaining 10 layers were all saturated aquifer. The approach to assigning thicknesses to these 10 layers was to maintain a thicknesses aspect ratio of less than 2.0 across all layers to ensure numerical convergence of the model. The purpose to the subdivision of layers is instead to refine the estimate of modeled plume volume. Details of grid dimensions for all scenarios are presented in **Appendix C**. All cells within the unsaturated and saturated layers are assigned corresponding parameter values from the EPACMTP runs. Contaminant concentrations are calculated as an average at the center node of each cell. Because a contaminant plume is typically curved, use of cells that are too large can result in underestimation of concentrations closer to the boundary.

For each model run, the volume of groundwater above previously specified health benchmarks and the risk associated with the average concentrations over that volume was recorded at each time step, which was used to calculate risk based on the same RME scenario previously described for EPACMTP. The volume of the plume was calculated by summing the saturated volume of each cell across all layers identified as within the plume boundary. Together, these outputs were used to understand the full magnitude and extent of the resulting plume, as well as the potential for sustained exposure for future receptors.

4.4 Model Results

The 90th and 50th percentile exceedances of established GWPS modeled with EPACMTP are presented in **Table 4-5** at the fill boundary. This includes all CCRMU fills at currently regulated and legacy facilities. Values represent the ratio of modeled concentrations and corresponding GWPS. Values that exceed the respective GWPS are shown in **bold**.

Table 4-5. Modeled Exceedance of GWPS at Fill Boundary.

Constituent	GWPS (µg/L)	90th Percentile	50th Percentile
Arsenic III	10	**24**	0.2
Arsenic V	10	**9**	0.1
Lithium	40	**349**	**1**
Molybdenum	100	**184**	**2**
Thallium	2	**19**	0.9

At the 90th percentile concentrations, all modeled constituents exceeded the respective GWPS. The constituents with the greatest exceedances are lithium and molybdemum at 349 and 184 times higher. Notably, arsenic III and V both exceed GWPS at levels 24 and 9 times higher. At the 50th percentile concentrations, lithium and molybdenum still exceed GWPS at levels 1 and 2 times higher. Concentrations of the remaining constituents fell below GWPS.

90th and 50th percentile risk exceedances modeled with EPACMTP are presented in **Table 4-6** for various distances away from the fill boundary. This includes all CCRMU fills at both currently regulated and legacy facilities. Both cancer and noncancer risks are presented for the highest exposed receptor group. For drinking water ingestion, the highest cancer risks were for adults, while the highest noncancer risks were for children. Values that exceed the selected risk criteria (i.e., cancer risk > $1x10^{-5}$ or HQ > 1) are shown in **bold**.

Table 4-6. Modeled Risk at Different Distances from CCRMU Fill

Constituent	Risk for 90th Percentile Groundwater Concentration			Risk for 50th Percentile Groundwater Concentration		
	10 ft	100 ft	1,000 ft	10 ft	100 ft	1,000 ft
Carcinogenic Effects						
Arsenic III	**2×10^{-3}**	**1×10^{-3}**	**2×10^{-4}**	**3×10^{-5}**	**2×10^{-5}**	3×10^{-6}
Arsenic V	**1×10^{-3}**	**6×10^{-4}**	**8×10^{-5}**	**1×10^{-5}**	7×10^{-6}	3×10^{-7}
Noncarcinogenic Effects						
Arsenic III	**3**	**2**	0.3	0.03	0.02	< 0.01
Arsenic V	**1**	0.7	0.1	0.02	0.01	< 0.01
Lithium	**317**	**160**	**27**	0.9	0.5	0.1
Molybdenum	**143**	**77**	**14**	**2**	**1**	0.1
Thallium	**164**	**88**	**16**	**7**	**4**	0.6

For 90th percentile concentrations, all constituents exceed risk benchmarks at every distance. These risks decline further away from the fill as the plume has greater opportunity to mix within the aquifer and disperse. However, all constituents still exceeded benchmarks at 1,000 ft away from the fill. At this distance, EPA identified cancer risks from arsenic (risk = 2×10^{-4} for trivalent and 2×10^{-4} for pentavalent) and noncancer risks from lithium (HQ = 27), molybdenum (HQ = 14), and thallium (HQ = 16). For 50th percentile concentrations, several constituents still exceeded risk benchmarks up to 100 ft away. EPA identified cancer risks from arsenic III (risk = 2×10^{-5}) and noncancer risks from molybdenum (HQ = 1) and thallium (HQ = 4).

As previously noted, the risk estimates reported here represent concentrations within the top five feet of the aquifer to ensure the model did not oversample outside the plume and along the plume centerline to identify regions of highest concentrations. While these results do not provide much information about the broader magnitude and extent of the plume, identification of risks up to 1,000 ft away from the fill indicates substantial potential for spread. Thus, EPA first identified the model runs for arsenic V that fell within 0.5 percent of the 90th percentile concentration modeled with EPACMTP at 1,000 ft. EPA then selected a subset of 25 these runs from different states at random to ensure consideration of geographic diversity. EPA modeled each of these runs in MODFLOW for both arsenic III and V with the same set of waste and environmental conditions

as used in EPACMTP. **Figure 4-5** presents a time series plot based on EPACMTP model run #57,957 that shows the volume of groundwater of that exceeds 1×10^{-5} risk for arsenic III and V and the average risk over that volume. Results for all 25 model runs and the associated model inputs are presented in **Appendix C**.

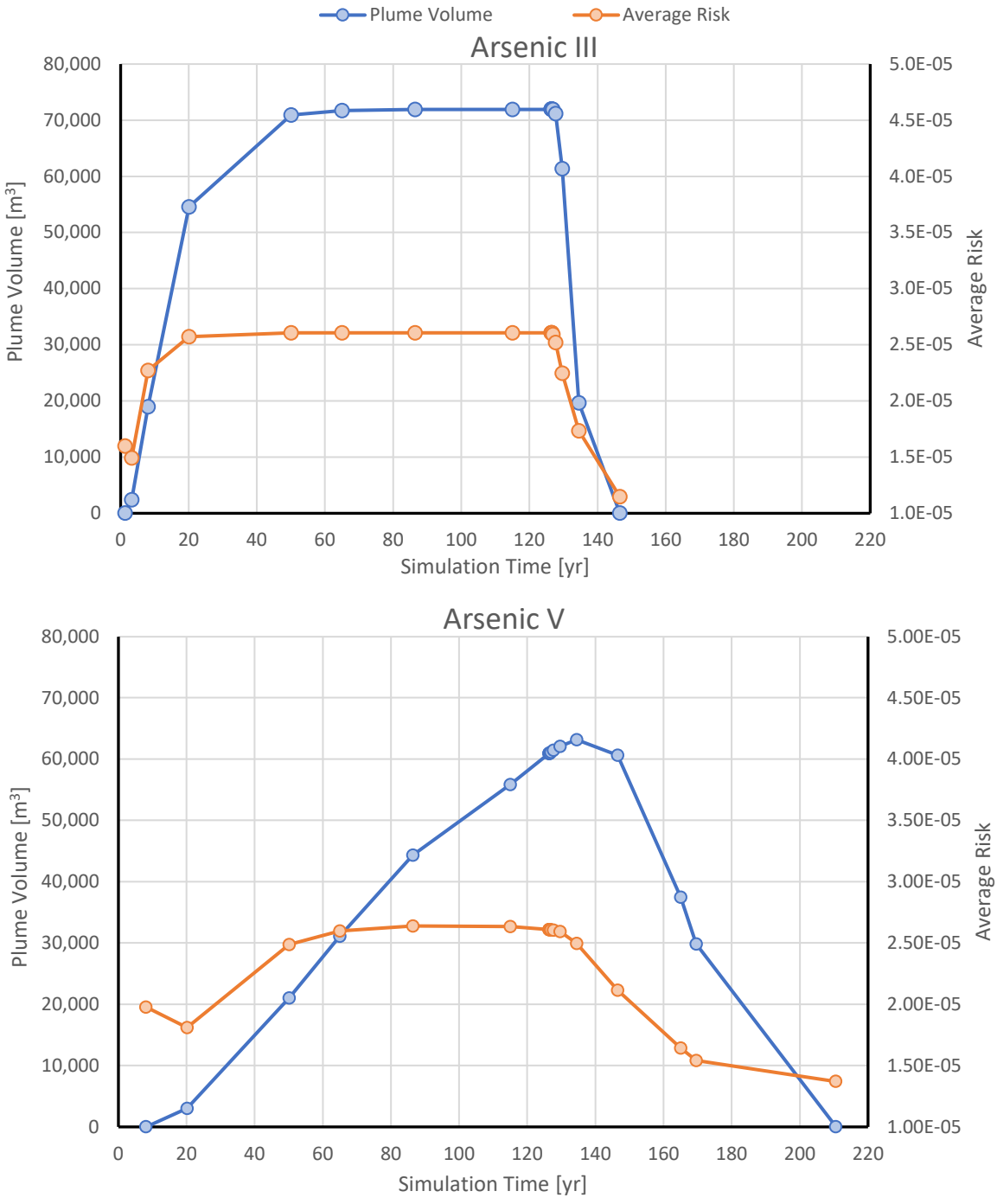

Figure 4-5. MODFLOW Model Result for Arsenic III and V.

The depicted run is a CCRMU fill of 18,500 tons placed over an area of 1.4 acres (5,506 m^2) and at a thickness of 8.1 ft (2.5 m). This run was selected for presentation because it represents the midpoint of the 25 runs in terms of volume and risk. Thus, it is expected to provide a representative understanding of the potential for high-end exposure. The remaining 24 MODFLOW runs reflect a wide range of conditions that also result in some of the highest groundwater concentrations at 1,000 ft from the fill boundary. Placement of CCR across the other 24 runs ranged from around 814 to 74,023 tons placed over areas between 0.2 and 2 acres.

As shown in the **Figure 4-4**, the average risk associated with the plumes change over time as the volume grows due to mixing with the aquifer and lateral dispersion. In both cases, the average risk across the plume eventually achieves a steady state until the leachable mass in the fill is depleted and the plume begins to shrink. This period of steady state lasts around 100 years for both arsenic species. Over this time period, the arsenic III plume maintained a steady volume around 19 million gallons (71,000 m^3) and the arsenic V plume increased from around 5 to 17 million gallons (21,000 to 63,000 m^3). Each plume maintained an average risk around 2.5×10^{-5} over this time period, though actual risks could be higher depending on where within the plume groundwater is drawn. At this volume and duration, these plumes could both sustain exposure for a typical residential receptor estimated to use 82 gallons of water a day at home (USGS, 2018), resulting in use of less than 0.8 million gallons over the duration of exposure.

4.5 Conclusions

EPA modeled the potential magnitude and extent of groundwater contamination resulting from CCRMU fills with EPACMTP and MODFLOW. This modeling effort incorporated many of the same data sources previously used to characterize leakage from CCR in the 2014 Risk Assessment, applied to the conceptual model for smaller fills. EPACMTP model runs identified potential for these smaller fills to result in groundwater contamination under high-end and moderate scenarios. In particular, high-end scenarios demonstrated potential for substantial plume spread. Therefore, MODFLOW was used to further model the full extent the plumes for a subset of high-end scenarios identified with EPACMTP. Based on these results, leakage of arsenic from smaller fills can still result in contamination that can extend over millions of gallons of groundwater and persist for centuries or more. For all these reasons, EPA finds the potential for risk to future residential receptors to be within the range OLEM typically considers for regulation.

5 CCRMU Fill Soil Risk

CCR is recognized as a type of technologically enhanced naturally occuring radioactive material (TENORM).[13] "Technologically enhanced" in this context means naturally occuring radioactive material has been concentrated or altered, such as through combustion, in a way that increases the potential for exposure. Therefore, EPA conducted further modeling of the exposures to radiation that may result from living in a home built on or around a CCRMU fill. The goal of this modeling is to characterize the risks associated with placement of smaller quantities of CCR. The section of the document describes the overarching framework for this modeling effort, as well as the specific models and inputs used to predict risks from radiation.

5.1 Model Framework

The placement scenario considered for the current evaluation is CCRMU fills located onsite at both active and legacy facilities. It is assumed there will be some type of soil or other cover placed over the CCR so that it is not exposed to the open air. This is because CCR is not expected to support a robust vegetative cover and because exposed ash may require active measures for dust control. In the absence of requirements to identify and track these smaller placements, it is assumed that the site will be redeveloped in the future for residential use.

This evaluation considered potential for exposure to gamma radiation and radon gas from placement within the soil. Because the CCR is buried, EPA did not consider potential for direct ingestion or inhalation of the CCR or uptake of contaminants to crops and livestock. Further, EPA did not separately consider leaching to groundwater due to a lack of data on leaching potential of these constituents from CCR. However, the contributions from these additional pathways to overall exposures is expected to be low compared to modeled pathways. These additional pathways are instead discussed an uncertainty in **Section 6 (Uncertainty and Sensitivity Analyses)**.

EPA selected (RESRAD)-ONSITE v7.2 (hereafter "RESRAD") for use in the current evaluation. The version of the model calculates risk with cancer slope factors based on data from "Biological Effects of Ionizing Radiation (BEIR) VII Phase 2: Health Risks from Exposure to Low Levels of Ionizing Radiation" (NRC, 2006). RESRAD was chosen because it is a publicly available model that addresses relevant the relevant exposure scenario, allows user specification of key parameters, and has undergone extensive validation and benchmarking (e.g., ANL, 1994, 1997a,b; ORNL, 1995; BIOMOVS II, 1995; IAEA, 1995).[14]

13) See: https://www.epa.gov/radiation/tenorm-coal-combustion-residuals

14) Use of this model does not represent an endorsement by EPA for use in other applications. EPA offices may evaluate similar scenarios with other models based on the specific needs and requirements of each program.

5.2 Model Inputs

EPA selected model inputs based on a review of RESRAD default values, EPA guidance, and the wider scientific literature. For some parameters, the available data would not support development of continuous probability distributions. This precludes the type of fully probabilistic modeling conducted for groundwater exposures. Instead, EPA conducted a more deterministic analysis by specifying high, moderate, and/or low values to capture the range of potential exposures. All three values were not specified for every parameter. Unspecified values generally reflect scenarios that would result in the lowest exposures and are not considered relevant to risk management decisions (e.g., low activity, short exposure duration). As a result, model runs with these lower risk inputs would not alter the conclusions of this evaluation and so are not considered relevant. **Table 5-1** presents a summary of inputs used for parameters where those values differ from or expand upon model defaults. These inputs are all organized in order from lowest to highest numerical value; however, lower values for each parameter do not always correspond to lower risk.

Table 5-1. Inputs for RESRAD Model

Parameter	Low	Moderate	High
Waste Characteristics			
Ra 226 + 228 Activity (pCi/g)	--	6.7	12.5
Radon Emanation Coefficient (%)	0.2	1	5
Environmental Characteristics			
Soil Cover Diffusion Coefficient (m^2/s)	1×10^{-7}	5×10^{-7}	1×10^{-6}
Contaminated Zone Diffusion Coefficient (m^2/s)	1×10^{-7}	5×10^{-7}	1×10^{-6}
Building Foundation Diffusion Coefficient (m^2/s)	8×10^{-9}	3×10^{-7}	2×10^{-5}
Cover Soil Depth (m)	0.3	0.6	1.2
Erosion Rate (m/yr)	0	--	--
Human Exposure Factors			
Residential Air change per hour (1/hr)	0.18	0.5	1.26
Fraction of Time Spent Indoors (%)	55	68	85
Fraction of Time Spent Outdoors (%)	3	6	13
Exposure Duration (years)	--	9	26

EPA did not specify values for every model input because some are unlikely to have a substantial impact on model results. For example, RESRAD defaults for area and thickness of the contaminated zone are 2 m (6.6 ft) and 10,000 m^2 (2.5 acres), respectively. Yet model results are not particularly sensitive to a thickness greater than 1.5 m (5 ft) or areas greater than 100 m^2 (0.025 acres). The resulting volume of CCR is equivalent to around 250 tons of ash, which is less than anticipated to be associated with typical placement. While larger amounts of CCR could be placed in ways both thicker and wider than modeled, it is not anticipated to substantially alter the results. Therefore, EPA retained default values for both parameters. Further discussion of the remaining parameters is provided in the following subsections.

5.2.1 Bulk Activity

Activity is a measurement of the rate at which radioisotope mass within a sample disintegrates (or decays), expressed in units of picocuries per gram (pCi/g). One pCi is equal to 2.22 disintegrations per minute. Each disintegration releases ionizing radiation in the form of alpha particles, beta particles, or gamma waves that have the potential to damage genetic material and increase an individual's lifetime cancer risk. Activity determines the amount of each radioisotope present in that can release radiation into the surrounding environment. RESRAD does not specify default values for the bulk activity of different radioisotopes.

EPA applied data from COALQUAL to define a distribution of bulk concentration for uranium and thorium using the same approach as described in **Section 4.2.4**. All uranium and thorium mass is unstable and will eventually decay. However, this mass may consist of multiple different isotopes that decay at different rates and with different decay products. The primary isotopes of interest for the current assessment are uranium-238 (U-238) and thorium-232 (Th-232). These isotopes are both the most naturally abundant (99.27 and 99.98 percent of uranium and thorium, respectively) and serve as the starting point for their respective decay chains. EPA calculated the bulk activity of U-238 and Th-232 from the bulk content of uranium and thorium using the following equation, adapted from U.S. EPA (2014d):

(5-1) $$A_B = \frac{(C_B)(NA)}{(2.8 \times 10^{-12})(M_A)(T_{1/2})}$$

Where:

A_B = Isotope Bulk Activity [pCi/g]
C_B = Element Bulk Concentration [mg/kg]
NA = Isotope Natural Abundance [%]
M_A = Isotope Atomic Mass [amu]
$T_{1/2}$ = Isotope Half-Life [years]
2.8×10^{-12} = Unit Conversion Constant

Each measurement of uranium and thorium bulk content were converted to corresponding U-238 and Th-232 bulk activity prior to calculating summary statistics for CCRs. Because these calculations rely on innate properties of the isotopes (e.g., half-life), it is unlikely these calculations introduced much additional uncertainty into the dataset. **Table 5-2** presents summary statistics used to characterize bulk activity of U-238 and Th-232.

Table 5-2. Calculated Bulk Activity (pCi/g)

Constituent	50th Percentile	90th Percentile
Uranium-238	3.9	8.1
Thorium-232	2.4	5.0

U-238 and Th-232 will decay through the natural chain of different isotopes before reaching a stable end product. Each of these decays will release radiation into the surrounding environment. Of the isotopes in these decay chains, radium and its short-lived decay products are expected to contribute most to cancer risk. Thus, it is critical to understand the activity of these isotopes. COALQUAL did not report bulk concentration of radium in coal. The bulk concentration of this element is typically very low, often on the order of picograms per kilogram, and so is typically reported only on the basis of bulk activity.

Previous studies have found that U-238 and Th-232 are in approximate secular equilibrium with the respective radium isotopes, radium-226 (Ra-226) and radium-228 (Ra-228) (Beck and Miller, 1980; LANL, 1982; Lauer et al., 2015). Secular equilibrium is the state in which the mass of a radioisotope remains constant because its production rate (e.g., due to decay of a parent isotope) is equal to its decay rate. Under secular equilibrium, the activity of all isotopes in the decay chain is identical. Therefore, EPA used data on the activity of U-238 and Th-232 to also represent the activity of Ra-226 and Ra-228.

A benefit of the COALQUAL dataset is that it provides consistent reporting of each element across samples. This can allow identification of trends in relative constituent concentrations, which can be an important consideration for cumulative risk because a sample with the highest activity of one isotope may not have the highest of another. Therefore, EPA calculated summary statistics for combined radium activity (Ra-226+228) across all samples. **Table 5-3** presents summary statistics used to characterize combined activity of Ra-226+228.

Table 5-3. Combined Radium Bulk Activity (pCi/g)

Constituent	50th Percentile	90th Percentile
Radium 226 + 228	6.7	12.5

Summary statistics for combined activity do not allow for subsequent breakout of the relative contributions from each isotope. Thus, EPA separately reviewed the relative activity of samples in the database. In a typical sample, Th-232 (Ra-228) activity is around two-thirds of U-238 (Ra-226) activity. Applying this ratio to the combined percentiles yields 50th and 90th percentile activities of 4.0 and 7.5 pCi/g for Ra-226 and 2.7 and 5 pCi/g for Ra-228. These results differ only slightly from summary statistics calculated for the individual isotopes. Therefore, these values were used to establish moderate and high activities for both isotopes. EPA assumed secular equilibrium across the decay chain.

5.2.2 Radon Emanation Coefficient

The radon emanation coefficient (or "emanation power") is the fraction of generated radon able to escape from CCR and migrate into empty pore spaces between the ash particles. This parameter determines the fraction of radon that is available to migrate through the subsurface and enter

overlying buildings. It is generally accepted that recoil is the dominant means by which radon gas is able to escape from solid particles. Recoil occurs because an alpha particle is ejected from the atom when radium decays to radon.[15] The force of ejection causes the newly formed radon atom to recoil in the opposite direction, which can result in release of radon from CCR if it occurs close enough to the surface of the ash particle. The distance radon can push through solid materials by recoil is small, typically on the order of a few micrometers. As a result, the radon emanation coefficient is influenced by waste properties, such as the size and shape of individual particles. Default values in RESRAD are 15 percent for Rn-220 and 25 percent for Rn-222. These values assume the media of interest is contaminated soil.

The Argonne National Laboratory summary of the data reviewed to establish model defaults (ANL, 2015) includes a report by Sakoda et al. (2011) on radon emanation from CCR. This literature review reported emanation coefficients from 46 samples of fly ash across six studies to derive an average emanation coefficient for Rn-222 of 3 percent. Other available data on CCR generated in the United States confirm the magnitude of that average. Beck et al. (1980) summarized data of fly ash and bottom ash samples from three power plants and reported average coefficients for both of less than 1 percent. Beck and Miller (1980) summarized data of 11 samples of fly ash and 10 samples of bottom ash or slag and reported an average emanation coefficient less than 2 percent and a maximum of 5 percent. The American Coal Ash Association, working with the Laurence Berkley Laboratory, reported data on 20 samples of fly ash, with an average and maximum emanation of 1.2 and 3.5 percent, respectively (ACAA, 1981). The Los Alamos National Laboratory reported data on nine samples of both fly ash and bottom ash. The fly ash had average and maximum coefficients of 0.7 and 2.8 percent, while the bottom ash had average and maximum coefficients of 0.2 and 0.4 percent (LANL, 1981). Based on the overall data, EPA set the moderate emanation coefficent to 1 percent for both Rn-220 and Rn-222. EPA applied factors of five to effectively bound the range of reported values and obtain low and high values of 0.2 and 5 percent, respectively.

5.2.3 Diffusion Coefficient for Soil Cover and Contaminated Zone

The radon diffusion coefficient (or diffusivity) is the unitized rate at which radon diffusion will increase over an area for a given change in concentration gradient. Radon diffusion is a passive process driven by the random movement of individual atoms that results in a net migration in the direction of decreasing air concentration. It is a function of the size and connectivity of pore spaces in the subsurface. This parameter determines how quickly individual radon atoms are able to travel through the pore spaces between individual particles and toward the ground surface. The default value in RESRAD for contaminated media and overlying cover soil is 2×10^{-6} m^2/s, which assumes both media are dry soil.

15) An alpha particle is a positively charged subatomic particle that consists of two protons and two neutrons tightly bound together.

ANL (2015) does not directly discuss radon diffusivity through CCR. EPA identified two studies that measured the rate of radon diffusion through fly ash. First, Chauhan and Chakarvarti (2002) placed a source consisting of uranyl acetate at the bottom of vessels filled with samples of fly ash and soil at various levels of compaction. This study reported radon diffusivity through fly ash to be 2.07×10^{-6} m^2/s at the lowest density of 1,260 kg/m^3 and 0.91×10^{-6} m^2/s at the highest density of 1,460 kg/m^3. There was limited overlap in densities tested for the soil sample, but the radon diffusivity through soil was reported to be 2.07×10^{-6} m^2/s at a density of 1,420 kg/m^3. Second, Narula et al. (2010) placed a source consisting of uranium ore at the bottom of vessels filled with various construction materials without compaction. This study reported the radon diffusivity through fly ash to be 2.06×10^{-6} m^2/s at a density of 1,220 kg/m^3, through soil to be 1.65×10^{-6} m^2/s at a density of 1,360 kg/m^3, and through sand to be 4.29×10^{-6} m^2/s at a density of 1,530 kg/m^3. Together, these studies indicate that radon diffusivity through fly ash can be similar to soil. This is considered reasonable because, as previously noted in **Section 4.2.7**, there is substantial overlap in the range of hydraulic conductivities reported for various CCR and soil. Both gas diffusivity and water conductivity are strongly linked to the size and connectivity of interstitial pore spaces. While the size of fly ash particles is similar to silt, bottom ash and boiler slag are more similar to sand. Therefore, EPA further reviewed the available literature for radon diffusivity through soil to better understand the potential diffusivity of radon through both CCR and cover soil.

ANL (2015) summarized results of multiple studies that evaluated the radon diffusivity of soil. These studies generally found that gas diffusion decreased with increasing moisture content. This is because radon diffusivity through water is far lower than through air, so water will tend to further inhibit radon transport as it occupies a greater fraction of the soil pore space. The Pacific Northwest Laboratory evaluated 34 samples of silty sand and clayey sand soils considered for use as cover for uranium mill tailings (PNL, 1983). All samples were compacted to greater than 80 percent of maximum dry density. The compaction process resulted in variable gravimetric water content among these samples ranging from 1 to 10 percent, and so a best fit curve was developed to relate the data. Based on this analysis, the authors concluded that radon diffusivity for the evaluated soils could range between 4.0×10^{-7} to 6.0×10^{-6} m^2/s at a gravimetric water content between 4 and 9 percent and between 1.0×10^{-7} to 2.0×10^{-6} m^2/s at a water content between 10 and 13 percent. Strong et al. (1981) evaluated one sample of silty sandy clay soil at different gravimetric water contents. The authors reported a radon diffusivity of 2.7×10^{-6} m^2/s at 1.5 percent water content, 2.5×10^{-7} m^2/s at 10.5 percent water content, and 6.0×10^{-8} m^2/s at 17.3 percent water content. The results of these studies generally align with each other and values reported in more recent studies (Papachistodoulou et al., 2007; Hosoda et al., 2012).

The available literature indicates a relatively consistent range of values for radon diffusivity in soil on the order of 10^{-6} and 10^{-7} m^2/s, depending on the specific soil type and moisture content. Although far lower values have been reported in the literature, these often represent high water content approaching saturation. However, soils cannot sustain these conditions and the excess

water will infiltrate further into the subsurface. Therefore, saturated conditions will be transient and unrepresentative of long-term radon transport. Based on these data, EPA set the high-end radon diffusion coefficient for both CCR and cover soil at 1×10^{-6} m^2/s, similar to default RESRAD values for dry soil. EPA distributed the moderate and low-end values evenly over the remaining range of reported values at 5.0×10^{-7} and 1.0×10^{-7} m^2/s.

5.2.4 Diffusion Coefficient for Concrete

The radon diffusion coefficient (or diffusivity) is the unitized rate at which radon diffusion increases over an area for a given change in concentration gradient. Radon diffusion is a passive process driven by the random movement of individual atoms that results in a net migration in the direction of decreasing air concentration. This parameter determines how quickly individual radon atoms are able to travel through the pore spaces in the foundation and into the home. The default value in RESRAD for a concrete slab is 3.0×10^{-7} m^2/s. This value represents a typical value for intact concrete drawn from the available literature.

The range of values considered in establishing the default for RESRAD all reflect intact concrete. However, there is potential that concrete will weather and crack as it ages. Cracks in concrete can serve as a preferential transport pathway that allows radon to bypass the remainder of the less permeable concrete matrix, creating the potential for much higher diffusivity. The user's manual for another of the RESRAD family of codes, RESRAD-Build, acknowledges the effects of cracks on radon transport and recommends a default value of 2.0×10^{-5} m^2/s for cracked brick and concrete (ANL, 2022). Therefore, EPA set the high-end value for radon diffusivity at 2.0×10^{-5} m^2/s to represent a cracked concrete slab. EPA set a moderate value at 3.0×10^{-7} m^2/s based on the recommended default for intact concrete and a low-end at 8.0×10^{-9} m^2/s based on the minimum value identified in the literature by ANL (2015).

5.2.5 Cover Soil Depth

Cover soil depth is the thickness of uncontaminated soil separating CCRMU fill from the ground surface and the building foundation. A thicker cover will result in lower exposure because the soil serves as a shield that will absorb some of the gamma radiation and slow radon migration to allow for greater decay before either reaches the ground surface. The default value in RESRAD for a cover soil depth is 0 m. This assumes that the contaminated zone extends all the way to the ground surface.

EPA did not identify any data sources that could be used to define a representative distribution of depths that a cover may be placed over fill and similar uses. It is generally assumed that placement of CCR will not extend up to the ground surface because the ash is unlikely to support significant vegetative cover and may result in issues from windblow dust. EPA set a moderate cover depth at 0.6 m (2.0 ft) to mirror the soil cover requirements for landfill closure and accommodate ground

cover. EPA set low and high depths at 0.3 and 1.2 m (1.0 and 4.0 ft), respectively. It is considered unlikely that substantially thicker soil cover would be common due to compounding costs.

The exact distance between a CCRMU fill and the receptor is further informed by the building foundation depth, which is the distance that the home foundation extends below ground surface. The default value in RESRAD is 1 m below ground surface, which represents a crawl space beneath a home. EPA set this value equal to 0 ft for all model runs. Given the uncertainty associated with the specific thickness of the cover soil, EPA chose to use soil cover thickness as a stand in for the overall distance between the CCRMU fill and receptor. The goal of this approach is to better define risk as a function of overall distance, rather than the relative interplay of multiple parameters.

5.2.6 Erosion Rate

The rate of erosion is the thickness of cover soil lost to wind, runoff, or other forces on an annual basis. A higher rate will result in greater long-term exposures because it will gradually reduce the cover separating the fill from the ground surface. The default value in RESRAD for erosion rate for both the cover soil and contaminated zone is 0.001 m/yr.

Given the extremely long half-lives of the modeled isotopes, specifying any rate of erosion will eventually result in elimination of all the cover soil. At the default rate of 0.001 m/yr, this would occur sometime between 300 and 1,200 years. As a result, consideration of erosion would cause long-term risks to be more similar regardless of the initial cover depth. However, because one focus of this analysis is understanding the effects of a given cover thickness on risks from the underlying fill, EPA set the erosion rate equal to 0 m/yr for all model runs. The potential for the CCR to be disturbed and brought to the surface will instead be addressed as a separate sensitivity analysis.

5.2.7 Air Exchange Rate

The air exchange rate is the number of times the total volume of air in a building is exchanged with outside air over a specified time period. This parameter determines the extent to which radon is able to accumulate in the home before it is cycled out. The default value in RESRAD is 0.5 air changes per hour (ACH). Thus, it would take two hours for all the air in the building to be replaced by.

EPA drew values for air exchange rate are based on national data reported in the 2011 Exposure Factors Handbook (US EPA, 2011). EPA selected low, moderate, and high values as the 10th, 50th, and 90th percentile values from Table 19-24. The corresponding values are 0.18, 0.5, and 1.26 ACH.

5.2.8 Time Spent Indoors/Outdoors

The time spent indoors and outdoors is the fraction of a day a resident spends inside and outside around the home. This parameter determines the level of exposure to gamma radiation and radon. When inside, there is less exposure to gamma radiation because concrete and other building materials serve as shields that absorb some of the radiation before it can reach the resident. When

outside, there is less exposure to radon because the gas is diluted far more quickly in open air compared to in a closed building. This parameter is expressed as a percent of a given 24-hour day. The RESRAD default values for time spent indoors is 50 percent of the day and for time spent outdoors is 25 percent of the day.

EPA drew values for air exchange rate based on national data reported in the 2011 Exposure Factors Handbook (US EPA, 2011). EPA selected the low, moderate, and high values as the 25th, 50th, and 75th percentile values from Table 16-16 "indoors in a residence (all rooms)" and Table 16-20 "at home in the yard or other areas outside the house." For time spent indoors, these associated values are 55, 68, and 85 percent of the day. For time spent outdoors, the associated values are 3, 6, and 13 percent of the day. The values for time spent indoors and outdoors were allowed to vary independently among model runs. Therefore, EPA selected this range of values in part to ensure the total time spent at home did not exceed 24 hours. The remaining time not accounted for between these two fractions is assumed to be spent away from home.

5.2.9 Exposure Duration

The exposure duration is the number of years a receptor lives at a single residence. It determines the total amount of time a receptor is near the waste and potentially exposed. The RESRAD default for exposure duration is 30 years.

EPA drew values for exposure duration based on national data reported in the 2011 Exposure Factors Handbook (U.S. EPA, 2011). These data represent the time an individual lives at a given residence before moving. EPA selected moderate and high values as the 50th and 90th percentiles from Table 16-108. The associated values are 9 and 26 years.

5.3 Model Results

The available data for some parameters do not support development of continuous probability distributions. Thus, even if EPA were to run every possible combination of the previously discussed model inputs, it would be difficult to assign an likelihood of occurrence to individual model results. Instead, EPA first modeled risk with all parameters set to moderate values. This combination of inputs resulted in a modeled risk of 1.6×10^{-5} for future residents. From this baseline, EPA adjusted each individual parameter to lower or higher values to better understand which exert the greatest influence on modeled risks. **Figure 5-1** presents the results of this comparison. EPA was unable to model the lower-end radon emanation coefficient because RESRAD does not allow this parameter to be set lower than 1 percent. Specific model outputs are provided in **Appendix C**.

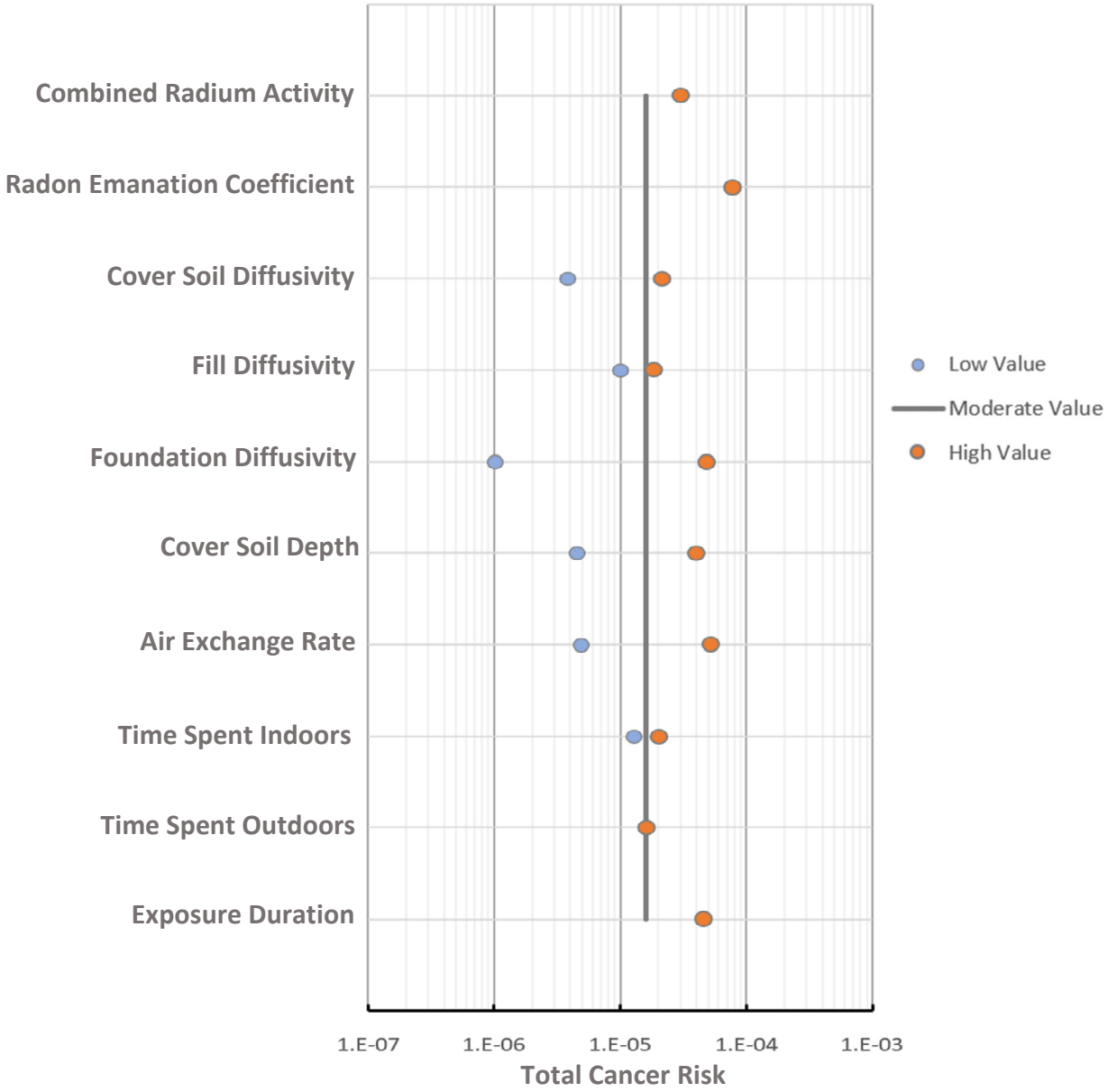

Figure 5-1. Sensitivity analysis of RESRAD model inputs.

Two initial conclusions can be drawn from these model runs. First, radon is the primary driver of risk in this scenario. Contributions from gamma radiation are often an order of magnitude or more below those from radon. This is because the cover soil acts as a shield, adsorbing the energy of the gamma radiation before it reaches the receptors above. Thus, the thicker the cover soil present, the lower the gamma radiation risk. Second, Rn-222 is the primary source of risk for all model runs. Rn-220 has a half-life less than a minute, so there is far less opportunity for it to diffuse through the soil cover and accumulate in indoor air under most conditions. In all model runs, risks for Rn-220 were reported as zero. Therefore, the Ra-226 activity of CCR and resulting decay to Rn-222 is the primary driver of risk to receptors from subsurface CCR.

The average and 90th percentile activities of Ra-226 in CCR of 4.7 and 8.1 pCi/g are higher than the corresponding values in background soil of 1.1 and 1.6 pCi/g, based on nation-wide data from Oak Ridge National Laboratory (ORNL, 1979). However, the range of radon emanation coefficients identified for CCR is considerably lower than for soils. The moderate values for CCR of 1 percent is over an order of magnitude lower than the RESRAD default for soil of 25 percent. As a result, despite higher radium activities, radon emanation from CCR is generally expected to be lower than from background soil. Even with both the CCR radium activity and emanation coefficent set to higher values, radon emanation from CCR would fall within the range expected for soils based on the RESRAD default coefficient. Although other model parameters, such as the building air exchange rate or foundation integrity, can result in higher radon accumulation and associated risk, these factors would all apply equally to CCR and background soils. Thus, based on the available data, the radon risks from CCR are not distinguishable from background soil and highly unlikely to result in the radon accumulation within the range EPA recommends for remediation. Therefore, EPA did not retain radon for further consideration in this risk assessment.

Risks associated with only gamma radiation were 5.4×10^{-7} under the moderate scenario. Many of the parameters adjusted in the model did not have any direct effect on risks from gamma radiation. This is reasonable because the migration of gamma radiation is not as dependent on the specific configuration of the soil or residence. The greatest individual increase in risk occurred when the thickness of the cover soil was reduced to 0.3 m (1 ft), resulting in a risk of 1.1×10^{-5}. This indicates the potential for higher risks if CCR is located closer to the ground surface. While it is considered unlikely that a future resident would live on top of an uncovered CCRMU fill, there is potential for the CCR to become mixed in with the surface soil if the fill is disturbed. However, modeling such exposures would require additional assumptions about the degree of disturbance and mixing, which would inject additional uncertainty into the calculated risks. Therefore, this scenario is discussed further in **Section 6 (Uncertainty and Sensitivity Analyses)**.

5.4 Conclusions

CCR is a type of TENORM that contains radioisotopes at levels greater than typically observed in background soil. EPA modeled potential radiation exposure and resulting cancer risk for future residential receptors with RESRAD under the assumption that some level of cover separates the CCR and the receptor. Because the available data does not support development of continuous probability distributions for some key environmental parameters, EPA separately considered the effects of low, moderate, and/or high values for each model input to understand the relative effects on modeled risk. The modeled risk for a future residential receptor who lives on top of a fill is 1.6×10^{-5} with all parameters set to moderate values. The identified risks resulted primarily from inhalation of radon that accumulates in a home. However, available data indicate the potential for radon emanation and associated risk from CCR is not distinguishable from that of background soils. Therefore, this exposure route was not retained for further consideration. The remaining modeled

risks from gamma radiation is 5.4×10^{-7} with all parameters set to moderate values. The parameter with the greatest influence on risk is the amount of cover soil separating the CCR and the receptor, with modeled risks increasing of 1.1×10^{-5} when the cover is reduced to 0.3 m (1 ft). This is because the soil serves as a shield and limits exposure to gamma radiation. As a result, there is greater potential for risk if the CCR comes to be located closer to the ground surface through future disturbance. This scenario is further explored in **Section 6 (Uncertainty and Sensitivity Analyses)**.

6 Uncertainty and Sensitivity Analyses

EPA reviewed the models used, as well as the data and assumptions input into the models, to better understand the potential sources of uncertainty inherent in the quantitative analyses. The Agency qualitatively and, to the extent possible, quantitatively analyzed these sources to understand the potential effects each may have on modeled risks. EPA also conducted further sensitivity analyses to understand how the modeled national risks vary in response to changes in sensitive parameters and to evaluate the potential for risks through exposure pathways that could not be fully modeled on a national scale. The purpose of this section is to document the results of these additional analyses.

6.1 Uncertainty Analyses

Uncertainty exists to some degree in any quantitative evaluation, and can bias the calculated results higher or lower than actual values. It is important to understand both the direction and magnitude of uncertainties present in a risk assessment. The direction of uncertainty is the tendency for that uncertainty to push a predicted value higher or lower than the actual value, while the magnitude of uncertainty is the extent to which that uncertainty may push a predicted value away from the true value. Characterizing these uncertainties helps to ensure that the overall conclusions of the evaluation would not change with the consideration of additional information. There are three primary causes of uncertainty:

- Variability is the extent to which the characteristics of an environmental system are heterogeneous, and is reflected in the parameter distributions used as inputs for the models. Although variability can be better captured by collecting additional data, it cannot be eliminated and must be treated explicitly in the assessment.
- Data uncertainty is a description of the imperfection in knowledge of the true value of a particular parameter. Uncertainty is generally reducible through additional research and information-gathering.
- Model error occurs because models and their mathematical expressions are simplifications of reality that are used to approximate real-world conditions, processes, and their relationships. These assumptions are sometimes necessary to solve complex mathematical equations or to fill gaps in available knowledge. However, the simplification of complex systems may misrepresent real-world conditions to an unknown degree.

Uncertainties identified in the evaluation were managed to the extent practicable to minimize the potential effects on model results. Variability was addressed by compiling available data into probabilistic distributions for each parameter. Uncertainty about the exact range or distribution of

a parameter was addressed through use of estimated point values or distributions intended to appropriately bound the true range, while ensuring protection of human health and the environment.

As previously documented in **Section 4.1** and **Section 5.1**, the publicly available models used in the current evaluation have undergone extensive review and validation. Together, these reviews verified that the mathematical formulation of the models is scientifically sound, the code executes properly, and the results can provide a reasonable representation of real-world conditions. Due to the extent of past review, EPA has a high degree of confidence in the design and functionality of these models. While some sources of uncertainty based on the model design are known to remain, such as the inability to fully quantify the effects of disposal below the water table, EPA aimed to constrain the scope of the evaluation to minimize the effects of such uncertainties on quantitative model results. Thus, EPA limited the discussion here to uncertainties associated with key inputs selected for use in the models. Many of the inputs used to characterize groundwater fate and transport are drawn from the same data sources as the 2014 Risk Assessment (U.S. EPA, 2014a). Uncertainties associated with these sources were previously discussed in the 2014 Assessment and these sources were found to represent the best available data available on a national scale. As a result, the focus of this uncertainty analysis is new sources of data incorporated in this evaluation.

6.1.1 Bulk Concentration Data

COALQUAL includes data on as-mined coal samples. These samples will undergo processing and combustion prior to disposal as coal ash, both of which will alter the overall composition of the sample. Therefore, EPA considered the potential for these processes to result in a disproportionate loss of constituent mass that might skew estimates of concentrations in the resulting ash.

Coal naturally contains impurities, such as pyrite and quartz, which can contribute to undesirable residuals (i.e., ash) and air pollutants (e.g., sulfur dioxide) during combustion. Coal cleaning is the process by which impurities are removed to the extent practicable from coal prior to combustion. Coal cleaning is a longstanding practice because it can increase the heating value and improve fuel consistency. Today it is employed just as often to reduce emissions of sulfur dioxide and other air pollutants (U.S. EPA, 1977). Although a wide array of cleaning methods have been proposed, the most common approach is still washing (NRC, 2007). Washing is accomplished by first crushing the coal to expose impurities that are not chemically bound within the coal. Afterward, the coal is placed in water, where the impurities separate from the coal based on differences in specific gravity. A secondary benefit of washing is it can greatly reduce concentrations of certain trace elements closely associated with the impurities, particularly sulfur minerals. However, it is not anticipated it will reliably reduce concentrations in the resulting ash. This is because washing also reduces the overall amount of CCR generated by combustion. Thus, the reduced constituent mass

will be further concentrated into a reduced volume of ash. One study of eastern coals reported an average of 56 percent reduction in arsenic mass, but a 70 percent reduction in ash (EPRI, 1998). Based on this, EPA assumed the results of washing on ash concentrations will be negligible.

Coal combustion occurs at extremely high temperatures that can exceed 1,000 °C (1,832 °F). These temperatures are higher than the boiling points of many trace constituents. As a result, certain constituents can vaporize from the coal during combustion and escape from the boiler along with the flue gas. However, flue gas will not remain at such a high temperature. For example, at a plant equipped with an FGD unit, the flue gas will generally exit at temperature between 55 to 70°C (130 to 160°F) (NETL, 2016). That is below the boiling point of most elements and so it is expected the majority of constituent mass will condense out onto ash particulates and be captured in pollution control devices, such as baghouses. This is supported by EPA's previous comparison of FGD and mined gypsum, which found no substantial enrichment of arsenic, molybdenum, or thallium in FGD gypsum (U.S. EPA, 2023). Therefore, EPA assumes the effects of volatilization on whole ash concentrations are negligible for the constituents considered in this evaluation. However, this may not be the case for the most highly volatile constituents, such as boron, mercury, and selenium.

Based on these findings, EPA concludes that the COALQUAL database can be relied upon to draw conclusions about the constituent concentrations present in CCRs. To corroborate this conclusion, EPA compared the bulk content calculated with COALQUAL data to measurements of fly ash from other available data sources. **Table 6-1** summarizes this comparison of calculated and measured bulk content. Upper bound values represent a maximum reported value unless otherwise indicated. No data sources were identified for comparison with lithium.

Table 6-1. Comparison of Bulk Concentration Data.

Constituent	Data Source	Detection Frequency	50th Percentile (mg/kg)	90th Percentile (mg/kg)	Upper Bound (mg/kg)
Arsenic	COALQUAL[1]	3,516 / 3,575	40	328	1,421
	U.S. EPA (2014a)	36 / 36	54	211	980
	EPRI (2008a)	NR	50	NR	NR
Lithium	COALQUAL[1]	3,577 / 3,588	81	206	371
Molybdenum	COALQUAL[1]	3,287 / 3,414	15	53	149
	U.S. EPA (2014a)	16 / 16	14	62	260
	EPRI (2011)	81 / 81	16	NR	236
Thallium	COALQUAL[1]	208 / 3,143	10	11	31
	EPRI (2008b)[2]	NR	0.5	NR	32
Uranium-238 / Radium-226	COALQUAL[1]	3,335 / 3,573	3.9	8.1	19.5
	Appendix B Uranium-238	234 / 234	3.0	6.8	24.3
	Appendix B Radium-226	146 / 146	4.2	7.6	28.4

Table 6-1. Comparison of Bulk Concentration Data.

Constituent	Data Source	Detection Frequency	50th Percentile (mg/kg)	90th Percentile (mg/kg)	Upper Bound (mg/kg)
Thorium-232 / Radium-228	COALQUAL[1]	3,289 / 3,318	2.4	5.0	9.7
	Appendix B Thorium-232	103 / 103	2.0	2.7	3.8
	Appendix B Radium-228	61 / 61	2.1	3.1	3.8

NR – Not Reported
1) Upper bound is 99th percentile to exclude outlier values for purposes of comparison.
2) Upper bound value reported in document excludes two outliers.

The first source of data considered is the 2014 Risk Assessment (U.S. EPA, 2014a). The reported sample number represents the number of sites with data, rather than the number of individual samples. EPA calculated quartiles for each site to more evenly weigh the variable amounts of data from different site and avoid biasing summary statistics those sites with the most data. The second source is industry reports on CCR composition and behavior (EPRI, 2008a,b; 2011). These reports aim to summarize the results of sampling conducted by industry partners, but provides limited information that can be used to understand where samples were collected or the full distribution of measured bulk content. The third source is government reports and peer-reviewed journal articles identified as part of EPA's regular review of the available literature (**Appendix B**). Specifically, EPA identified a number of studies that characterize the bulk activity of various CCRs. Summary statistics were calculated after averaging samples from each study that were collected from a single source.

It was not expected the summary statistics from each data source would be identical. The fly ash samples represent a compilation of data from individual sampling events conducted at different times and for different purposes. As a result, there are no measures in place to ensure the available data proportionally reflect different regions of the country and other factors that might affect composition. Additionally, COALQUAL represents concentrations in the whole ash, which is all the ash that remains after combustion. As a result, there is potential the calculated concentrations may differ somewhat from samples of fly ash. Yet, despite these differences, there was generally good agreement among the different data sources.

There were two notable instances where COALQUAL diverged from the other data sources. The first instance is the median value for thallium. It is clear this difference is caused by high detection limits for thallium in COALQUAL. COALQUAL reports concentrations less than the detection limit at 75% of the detection limit, which biased median values high. Therefore, EPA relied on the other data sources to characterize lower percentiles for this constituent. Given how few samples were non-detect for other constituents, EPA accepted non-detect values as reported. The second instance is high-end values for thorium. Both 90th percentile and upper bound are notably higher

in COALQUAL. However, EPA did not identify any similar data quality issues for this constituent. Instead, the extremely narrow range of values reported by other data sources make it more likely the other data sources do not fully capture the potential variability of this constituent. Therefore, EPA retained the COALQUAL data for use in this evaluation.

Altogether, this review provides a great deal of confidence the COALQUAL dataset can be used to represent the bulk content and activity of CCRs. While the dataset may not provide precise data for any individual ash sample, the overall dataset can provide an accurate representation of overall ash composition suitable for use in a probabilistic evaluation. Therefore, the uncertainty associated with use of this dataset for the constituents considered in this evaluation is considered to be low.

6.1.2 Coal Combustion Residual Type

The constituent bulk content calculated with the COALQUAL database represents the whole ash generated by combustion, which is a mixture of fly ash and bottom ash or boiler slag. It is not possible to further break out the contributions from each type of CCR. In a typical boiler, the ratio of generated ash types falls somewhere around 80 percent fly ash to 20 percent bottom ash (U.S. EPA, 1981). This ratio has remained relatively consistent. Recent statistics on national generation rates show that fly ash accounts for 74 percent of the annual mass of these three ash types (ACAA, 2022). Thus, the whole ash can be understood as predominantly fly ash.

There are potential differences in the composition of fly ash and other CCR types that may result from differences in the volatility of individual constituents. As previously noted, more volatile constituents have a greater tendency to escape from the boiler and settle out onto fly ash. This may result in higher concentrations in fly ash compared to bottom ash and boiler slag. Generally, there are far less data available on constituent concentrations present in and released from bottom ash and boiler slag. This may be due in part to the smaller quantities of ash generated. The most recent American Coal Ash Association (ACAA) report indicates that coal combustion across the United States results in 74 percent fly ash, 23 percent bottom ash, and 3 percent boiler slag (ACAA, 2022). Thus, from a waste management perspective, fly ash has historically been a dominant concern.

In 2014, EPA did not have sufficient data on bottom ash or boiler slag to separately model these CCR types. This was not considered a major source of uncertainty because of the prevalence of co-management of different CCR types in landfills and impoundments. Since then, the Agency has not identified any substantial new sources of data to further inform groundwater modeling for these CCR types. As a result, EPA was again unable to separately model these CCR types as part of the current evaluation. However, it is assumed that there is similar potential for co-management in CCRMU fills.

Despite the lack of waste characterization data for bottom ash and boiler slag, the monitoring data that the 2015 CCR Rule required facilities to report provides ample evidence that these two CCR types have similar potential to contaminate groundwater based on facility monitoring reports as of July 2022. A total of 23 of the 80 disposal units identified as dedicated bottom ash have initiated corrective action (29 percent). A total of 4 of the 12 disposal units identified as dedicated to slag have initiated corrective action (30 percent). This ratio is similar for units that manage other ash types (40 percent). Thus, it appears that any differences in the composition of bottom ash and boiler slag are not substantial enough to prevent groundwater releases. As a result, the magnitude of the uncertainty as it related to groundwater exposure is considered low.

As part of the Agency's regular review of the available literature, EPA did identify a number of sources that characterized the bulk activity of bottom ash. Altogether, these sources are considered sufficient to characterize the anticipated bulk activity of this CCR type. **Table 6-2** provides a comparison of summary statistics for Ra-226 activity in fly and bottom ash. Summary statistics were calculated after averaging samples from each study that were collected from a single source. The underlying raw data are made available in **Appendix B.** Little data was identified for Ra-228 and so a similar comparison could not be conducted.

Table 6-2. Comparison of Fly and Bottom Ash Bulk Content.

Constituent	Ash Type	Sample Count	50th Percentile (mg/kg)	90th Percentile (mg/kg)
Radium-226	Fly Ash	146	4.2	7.6
	Bottom Ash	42	4.5	8.8

Based on these data, there is no indication the Ra-226 activity of bottom ash will differ substantially from that of fly ash. Therefore, EPA concludes the use of COALQUAL data to represent the bulk activity of bottom ash is appropriate. EPA is not aware of any reason the overall composition of boiler slag would differ dramatically from that of bottom ash. As a result, the magnitude of the uncertainty as it relates to radiation exposure is considered low.

6.1.3 Alternate Sources

As part of the current risk assessment, EPA initially considered whether there might be a quantity of CCR small enough to pose no reasonable risk of adverse impacts to groundwater quality. Such an analysis might be feasible for individual placements of CCR. However, management of CCR onsite at electric utilities is considered unique from management offsite in part because there is far greater potential for placement of CCR at multiple discrete locations, both across the facility and in close proximity. The presence of unidentified accumulations of CCR are a particular concern for groundwater monitoring around currently regulated disposal units. There is presently limited

data available on the size and extent of placement across these facilities and the available record indicates that documentation of past placement has not always been maintained. As a result, EPA does not believe it is possible to compile a reliable record of such placements in the absence of further facility inspection and reporting.

The regulatory framework of the 2015 CCR Rule does not capture contamination arising from CCRMUs. Therefore, at present, previous, and ongoing leakage from such placements can affect groundwater quality at wells installed around the monitored CCR units without running afoul of the rule. The statistical methods used to identify statistically significant increases and statistically significant levels are formulated based the assumption there is a common background that would be found both upgradient and downgradient of a CCR unit, provided that unit has not leaked. However, this assumption would not be valid if upgradient wells have been affected by leakage from nearby CCRMUs. Upgradient placements can leak all the same constituents as currently regulated units. If concentrations in upgradient wells increase as a result of leakage from upgradient CCRMUs, then the resulting characterization of background groundwater would not provide an accurate baseline for comparison. Any leakage from the regulated CCR unit would then need to progress even further and faster than from the upgradient CCRMU to be distinguishable from the skewed background. At a minimum, this could delay identification of a potential or actual release.

Leakage from CCR placement does not have to constitute a release by itself in order to confound the groundwater monitoring at nearby CCR units. Elevated levels of the common ions and other constituents listed on Appendix III could still delay or prevent a monitored CCR unit from entering into assessment monitoring. Further, leakage from smaller sources can still contribute to overall risk by supplementing leakage from a regulated CCR unit, resulting in a larger downgradient plume than would have otherwise occurred.

EPA previously identified potential for risk to human health and the environment from operating landfills and surface impoundments. If identification of a release from these currently regulated CCR units is delayed or prevented by leakage from CCRMUs, then previously identified risks from these CCR units to nearby receptors would persist. The longer that contamination is allowed to spread, the greater potential that full remediation will not be feasible as a result of complex site geology or other factors. As a result, EPA concludes that, just because a particular CCRMU might be unlikely to trigger corrective action in isolation, this does not mean there is no potential for concern. EPA is unable to reliably identify a minimum quantity of CCR at which interference with groundwater monitoring is unlikely. This would depend not only on the specific quantity of ash in the CCRMU, but also the number of nearby CCRMUs, the proximity of those CCRMUs to regulated CCR units and to each other, and the relative timeframe over which each unit has leaked.

This represents a large source of uncertainty. However, it does not directly affect the risks from individual CCRMU discussed in this risk assessment.

6.2 Sensitivity Analyses

Sensitivity analyses identify the parameters that exert the greatest influence on modeled risks. These analyses provide further insight as to whether specific waste management scenarios can result in risks substantially different than those modeled nationally. EPA relied on the findings of these analyses to draw additional conclusions about the potential risks associated with CCR management and to refine the scope of its proposed regulatory action.

6.2.1 Updated Weather Data

EPA used HELP v4 to model infiltration of precipitation into CCRMU Fills. Changes in the recent update focused on the underlying weather data. No changes were made to the method by which the model calculates infiltration. This update incorporates NOAA weather data from 1961 to 2014, which includes daily data precipitation, air temperature, and solar radiation; long-term quarterly data on relative humidity; and long-term data on average windspeed. There is a high degree of confidence in these data. The procedure used to interpolate these data and generate a uniform grid of values has been published in the peer-reviewed journal (Frye et al., 2016) and previously applied in Agency modeling (U.S. EPA, 2019b). Incorporating this updated weather data serves to address a key uncertainty identified in the 2014 Risk Assessment.

EPA used HELP v3 to model infiltration into landfills and the surrounding soil for the 2014 Risk Assessment. Disposal units were matched to the closest available weather stations to identify relevant precipitation data. To understand how updates to the HELP model would affect previously modeled risks, EPA compared infiltration rates calculated with both model versions for a specified soil type. **Figure 6-1** depicts the differences in infiltration rates modeled with both the current and previous versions of HELP.

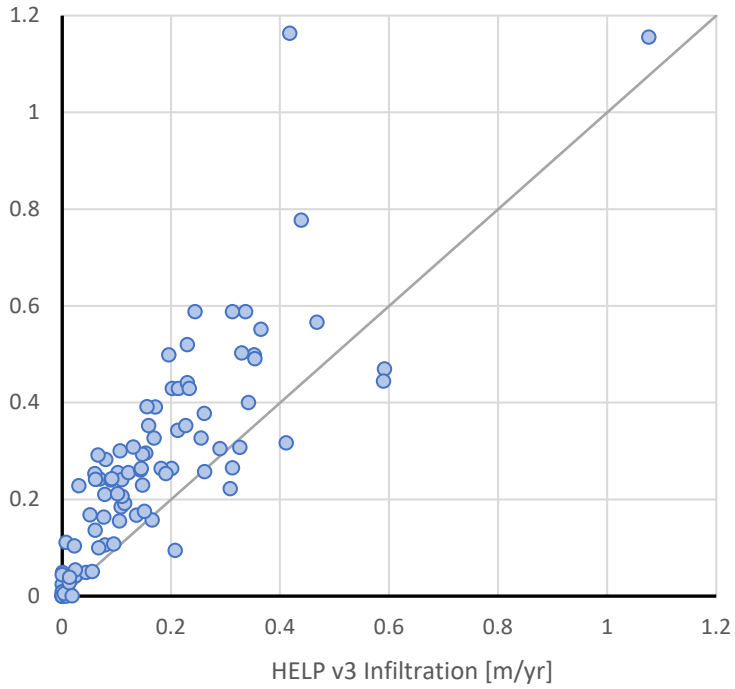

Figure 6-1. Comparison of modeled infiltration into silty loam soils.

Each dot on the figure represents annual infiltration a different weather station. Based on these data, it is expected the more recent weather data in HELP v4 would, on average, result in higher infiltration than previously modeled. Higher infiltration into a landfill will result in greater risks because of the greater volume of resulting leakage into groundwater over time. Therefore, EPA believes it is likely the 2014 Risk Assessment underestimated the magnitude of potential leakage and associated risk from CCR landfills after closure. The updated data is unlikely to similarly affect the modeled risks for surface impoundments. Leakage from impoundments was modeled only during the active life of the unit when it was filled with water. The presence of a constant hydraulic head from this water means that the rate of leakage from the unit would be far less sensitive to the specific contributions from precipitation.

6.2.2 Additional Exposure Pathways

In **Section 5 (CCRMU Fill Soil Risk)**, EPA evaluated the risks associated with CCRMU fills assuming a scenario where the fills remained covered by around 0.6 m (2 ft) of soil. However, there is no guarantee that any cover currently in place will be maintained in the future in the absence of land use restrictions. Mixing of CCR with surface soil will result in increased exposures not only to radiation, but also to any chemical constituents present. There is substantial uncertainty about the degree of mixing that could occur in the future. Thus, EPA instead conducted a sensitivity analysis to understand how risks may change as the quantity of ash mixed in the soil increases. For purposes of this analysis, EPA considered cumulative cancer risk from exposure to arsenic and radionuclides.

This approach is in line with Agency policy to address risks from radionuclide exposures in the same manner as chemical contaminants (U.S. EPA, 2014d).

To accomplish this, EPA drew health-based benchmarks for ingestion and direct gamma exposure from both Regional Screening Level (RSL)[16] calculator for arsenic and Preliminary Remediation Goal (PRG)[17] calculator for Th-232 and U-238 decay chains. Values were obtained assuming secular equilibrium and an area of 100 m^2 (0.025 acres). The benchmarks corresponding to 1×10^{-5} risk are 6.77 mg/kg arsenic, 0.127 pCi/g Th-232, and 0.162 pCi/g U-238. EPA applied these benchmarks to first calculate a cumulative risk associated with undiluted exposure to each individual sample from the COALQUAL database. EPA then calculated the risk from exposure to undiluted CCR at the 50th and 90th percentile bulk concentration and activity. Finally, EPA scaled these summary statistics based on different degrees of mixing with the surface soil. **Figure 6-2** depicts the risk from CCR as it becomes an increasing fraction of the overall surface soil, up to 50 percent. For 90th and 50th percentile concentrations, cancer risks to residential receptors above 1×10^{-5} are possible at mixing of more than 1 and 2 percent, respectively. Cancer risks above 1×10^{-4} are possible at mixing of more than 8 and 17 percent, respectively.

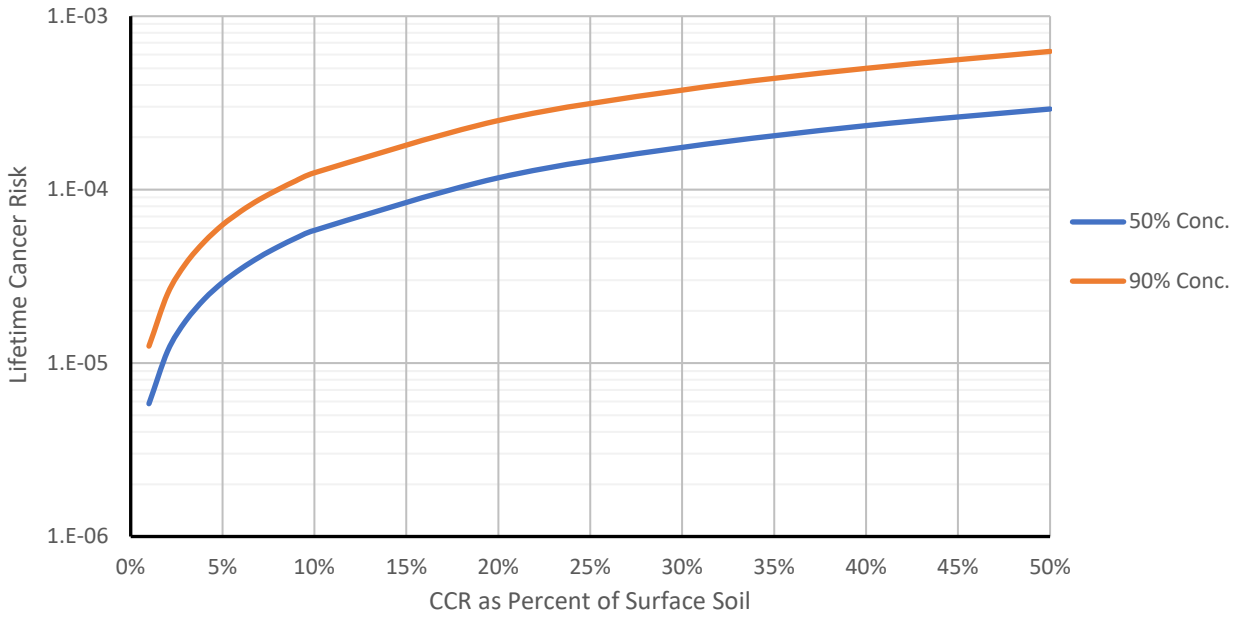

Figure 6-2. Progressive risk from mixing of CCR with surface soils.

Although background soil can also contain arsenic and radium, it is expected that levels present in CCR will generally be higher and so result in increased exposures to these two contaminants. The average and 90th percentile levels of arsenic in background surface soil are estimated to be 6.4 and 10.8 mg/kg (USGS, 2013b), compared to 136 and 328 mg/kg for CCR. The average and 90th

16) See: https://www.epa.gov/risk/regional-screening-levels-rsls

17) See: https://epa-prgs.ornl.gov/radionuclides/

percentile levels of combined Ra-226+228 in background surface soil are estimated to be 2.1 and 3.0 pCi/g (ORNL, 1979),[18] compared to 7.7 and 12.5 pCi/g in CCR. In particular, it is notable the combined bulk activity of Ra-226+228 in CCR is, on average, more than 5 pCi/g above background. This is considered as a potential relevant and appropriate requirement under CERCLA, regardless of whether the contamination is found in surface or subsurface soils (U.S. EPA, 1998). Based on these results, there is substantial potential for risk to future residents if a CCRMU fill is disturbed.

This analysis focused on direct exposure to soil because it is the most direct exposure pathway. There is potential for exposure through other pathways, though each is expected to have greater associated uncertainty. For example, it is possible to identify more stringent benchmarks from the Preliminary Remediation Goal calculator for a site used as a home garden. However, calculated risk will depend on additional factors, such as the specific crops grown and how much of the diet is supplemented with food from the garden, which would require a number of further assumptions on a national scale. Given the magnitude of risk identified from exposure directly to soil, further examination of these additional pathways was not necessary to drawn firm conclusions about risk from CCRMU fills.

6.2.3 Post-Closure Exposures

The main model and sensitivity analyses identified potential risks resulting from gamma radiation and radon gas if CCRMU fills are disturbed. To ensure that current disposal standards are sufficient to mitigate the identified risks, EPA conducted a further analysis of closed disposal units. A major consideration is the fact that land use controls imposed on these units will prevent construction of habitable structures on top of the cover system. This will greatly limit the types of exposures and amount of time any individual will spend on top of the unit in a given day. In the absence of residential receptors, a reasonably maximally exposed receptors under a future land use scenario might be an individual who uses the open area for recreation.

EPA has not established recommended exposure factors for this type of receptor, as actual behavior can vary widely at different sites. Instead, EPA started with an absolute worst-case exposure scenario. This would be someone who spends 8 hours every day for 26 years in the open air on top of a soil cover with a maintained thickness of 0.6 m (2 ft). The radium activity was set to the high values listed in **Table 5-1**. Under this worst-case scenario, RESRAD identified a cancer risk attributed to gamma radiation of around 1.8×10^{-6}.

The scenario is expected to overestimate risk for multiple reasons. For example, it is highly unlikely any receptor would be present on top of a closed unit all day, every day, for over two decades.

18) This comparison assumes that the Th-232 activity reported in ORNL (1979) is in equilibrium with Ra-228, which may not always be the case. Actual levels of Ra-228 may be somewhat higher or lower on a case-by-case basis.

Additionally, many units will contain CCR with lower activity than modeled. Based on these various considerations, it is expected the risks associated with release of gamma radiation from disposal units closed in a manner consistent with the requirements of the 2015 CCR Rule will fall entirely outside the OLEM risk range.

6.3 Conclusions

EPA identified and reviewed major sources of uncertainty identified since 2014 to understand the potential effects on modeled risks. Uncertainties associated with newer data sources are expected to have a minimal effect on the conclusions of this assessment. Uncertainties associated with scenarios that could not be quantitatively modeled have the potential to result in underestimation of risk in some circumstances. To the extent practicable, EPA aimed to minimize the influence of such uncertainties by focusing on the most direct exposure pathways and applying best available data.

EPA also conducted several sensitivity analyses to understand the potential for substantially higher risk than was modeled on a national scale. One analysis identified potential for risk to future residents above the levels at which OLEM typically considers to warrant regulation from exposure to soil if CCRMU fills are disturbed and mixed with surface soil. Another identified potential for higher risks to groundwater from CCR landfills than were previously modeled in the 2014 Risk Assessment. The final analysis affirmed that current regulatory requirements for closure of disposal units are adequate to protect human health and the environment from anticipated exposures to radiation.

The results of all these analyses reinforce the conclusions from previous modeling that disposal in historical and inactive landfills and surface impoundments, as well as placement in CCRMU fills, have the potential to result in risk to future receptors that warrant regulatory action.

7 Summary and Conclusions

The purpose of this document is to characterize the risks associated with management practices across the United States outside of currently regulated landfills and surface impoundments. To accomplish this task, EPA used mathematical models to estimate the rate at which constituents are released from legacy impoundments and CCRMUs, the fate and transport of these constituents through the environment, and the potential risk of adverse effects to individual receptors. EPA then conducted additional sensitivity and uncertainty analyses to identify any potential for higher risks than those identified in the broader, national analysis. The purpose of this section is to summarize the various analyses conducted and results obtained for different exposure pathways, provide further context for these results, and present the final Agency conclusions.

7.1 Groundwater Exposures

When CCRs are placed on the ground, they may leach metals and other inorganic contaminants to groundwater. Once mixed with groundwater, the contamination may migrate downgradient to private wells where it is ingested by receptors who rely on groundwater as their primary source of drinking water. Because groundwater contamination from inorganic constituents can persist for years, a receptor does not need to be presently exposed for there to be concern. The further the contamination is allowed to spread, the more difficult it may become to clean it up due to factors, such as complex site geology.

EPA previously identified risks to groundwater from active landfills and surface impoundments in the 2014 Risk Assessment, which are now regulated under the CCR Rule. The Agency maintains that previous findings on risk from active units are equally applicable to units that ceased receipt of waste prior to 2015 and either closed or became inactive. The operational lifecycle of all disposal units progresses from construction to closure. The fact that some historical and inactive units may no longer contain free water or have since installed a soil cap only places them in a different stage of the same lifecycle. EPA previously modeled active landfills and impoundments at the lifecycle stage anticipated to contribute most to long-term risks. Thus, these estimates of future risk are relevant to all disposal units, regardless of the current lifecycle stage. Indeed, a number of landfills and impoundments that are now either historical or inactive were previously modeled in 2014. These model results were not reported at the time because the units were found to fall outside the scope of the 2015 Rule, but confirm the associated risks are of the same magnitude as for active units. Other available information reviewed as part of this risk assessment further supports the conclusion there is nothing unique about the location or size of these historical and inactive units that could result substantially different risks from those previously modeled. In contrast, there is evidence the 2014 Risk Assessment underestimated the actual magnitude of risk from landfills and impoundments as a result of data available at the time. Data that has since become available indicates a greater proportion of all disposal units are unlined and the rate of infiltration into

landfills is higher than previously modeled. As a result, historical and inactive units would also be expected to have risks even higher than previously modeled.

EPA also conducted further modeling to understand the full magnitude and extent of groundwater impacts that could be associated with smaller placements in CCRMU fills. Through this modeling, EPA identified potential for these fills to contaminate onsite groundwater. Model results indicate potential for exceedance of GWPS at the fill boundary under both high-end and moderate conditions. These results also show potential for substantial spread of the resulting groundwater plume. Under high-end conditions, these plumes are large and persistent enough to sustain exposures for over a century or more at average risks of 2×10^{-5} or higher.

Even when leakage from legacy impoundments and CCRMUs does not pose direct risk, there remains potential for this leakage to affect monitoring wells located further downgradient. If characterization of background groundwater quality at another unit becomes skewed by leakage from legacy impoundments or CCRMUs, then it may delay or entirely prevent identification of a release and implementation of corrective action to address risks previously identified by the 2015 CCR Rule.

7.2 Soil Exposures

When CCRMU fills are placed on the ground and unmonitored, there is potential for the ash to be disturbed in the future when the property is redeveloped. As a result, any engineering controls currently present that would serve to limit exposure cannot be guaranteed to remain in place. For this reason, EPA considered the additional exposure pathways that could result under a future residential land use. Future residents who live on or around the fill may have increased exposed to gamma radiation and radon gas emitted from the CCR and to contaminants present in the CCR as a result of incidental ingestion of CCR mixed in with surface soils. EPA conducted initial modeling to understand the potential for exposure to radiation from CCR remaining in the subsurface. EPA found the amount of radon emitted by CCR is not distinguishable from background soil and so did not retain this pathway for further consideration. EPA also found greater potential for risk from gamma radiation as CCR comes to be located closer to the ground surface due to a reduction in shielding. An additional sensitivity analysis identified potential for further risk if CCR becomes mixed with surface soil. Accumulation of CCR can result in elevated cancer risk from incidental ingestion of arsenic and radium, in addition to direct exposure to gamma radiation from radium. For high-end waste concentrations, an eight percent mixture of CCR in surface soil was found to result in risk on the order of 1×10^{-4}.

7.3 Final Conclusions

Based on the analyses summarized in this document, EPA concludes that leaching from both legacy surface impoundments and CCRMUs has the potential adversely affect groundwater quality and cause risks to future receptors in the range OLEM typically considers for regulation. These risks

are driven by ingestion of arsenic, lithium, molybdenum, and thallium released to groundwater. Health effects associated with arsenic ingestion are an increase in the risk of cancer in the skin, liver, bladder, and lungs, as well as nausea, vomiting, abnormal heart rhythm, and damage to blood vessels. Health effects associated with lithium ingestion are neurological and psychiatric effects, decreased thyroid function, renal effects, cardiovascular effects, gastrointestinal effects, and skin eruptions. Health effects associated with molybdenum ingestion are higher levels of uric acid in the blood, gout-like symptoms, and anemia. Health effects associated with thallium ingestion are hair loss, ocular effects, and behavioral changes.

EPA also concludes that the unmonitored accumulation of CCR in surface and subsurface soils has the potential to result in risk to future receptors in the range OLEM typically considers for regulation. These risks are driven by exposure to gamma radiation from radium and its decay products, and incidental ingestion of arsenic and radium in soil and dust. Health effects associated with exposure to radium include increased risk of several types of cancer, particularly lung and bone cancer.

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Appendix A: Facility List

Since finalization of the 2014 Risk Assessment, EPA has identified additional facilities that were not previously modeled. These include both additional active facilities subject to the 2015 CCR Rule and inactive facilities believed to have inactive surface impoundments. EPA incorporated all these additional facilities along with those previously modeled in current groundwater modeling for CCRMU fills. **Attachment A-1** provides a list of all facilities modeled, the current regulatory status of that facility and a summary of the environmental parameters assigned to that facility for purposes of fate and transport modeling.

Appendix B: New Characterization Data

Since finalization of the 2014 Risk Assessment, EPA has identified additional sources of data that were used in this risk assessment to supplement and corroborate the Agency's characterization of CCR composition and behavior. The COALQUAL database includes data on the composition of coal samples from across the country and was used to estimate CCR bulk composition and activity. **Attachment B-1** provides the 2015 COALQUAL database. The LEACHXS Lite database includes a repository of LEAF leachate data on a range of materials. Recent review of this database identified additional CCR data that was used together with previously collected leachate data. **Attachment B-2** provides new leachate data drawn from LEACHXS Lite. The bulk activity dataset represents data compiled by the Agency from the broader literature. This dataset was used to corroborate the bulk activity calculated from COALQUAL. **Attachment B-3** provides the bulk activity data identified through a review of the literature.

Appendix C: Model Outputs

This risk assessment modeled the fate and transport of metallic and other inorganic constituents identified as constituents of concern for CCRs. As part of this effort, EPA applied multiple models to characterize the magnitude and extent of adverse impacts to different environmental media. EPACMTP is groundwater model designed to calculate concentrations at a specified distance away from the source. **Attachment C-1** provides Access databases that contain the EPACMTP inputs and associated outputs for landfills and impoundments previously modeled, but not incorporated in the results reported in the 2014 Risk Assessment. **Attachment C-2** provides Access databases that contain the EPACMTP inputs and corresponding outputs for CCRMU fills at four distances away from the unit boundary. MODFLOW is a groundwater model that can be used to calculate concentrations in three dimensions. **Attachment C-3** summarizes the MODFLOW model inputs and outputs. RESRAD is a radiation exposure model designed to calculate exposure to gamma radiation and radon gas from radioisotopes present in the soil. **Attachment C-4** provides RESRAD model inputs and outputs.